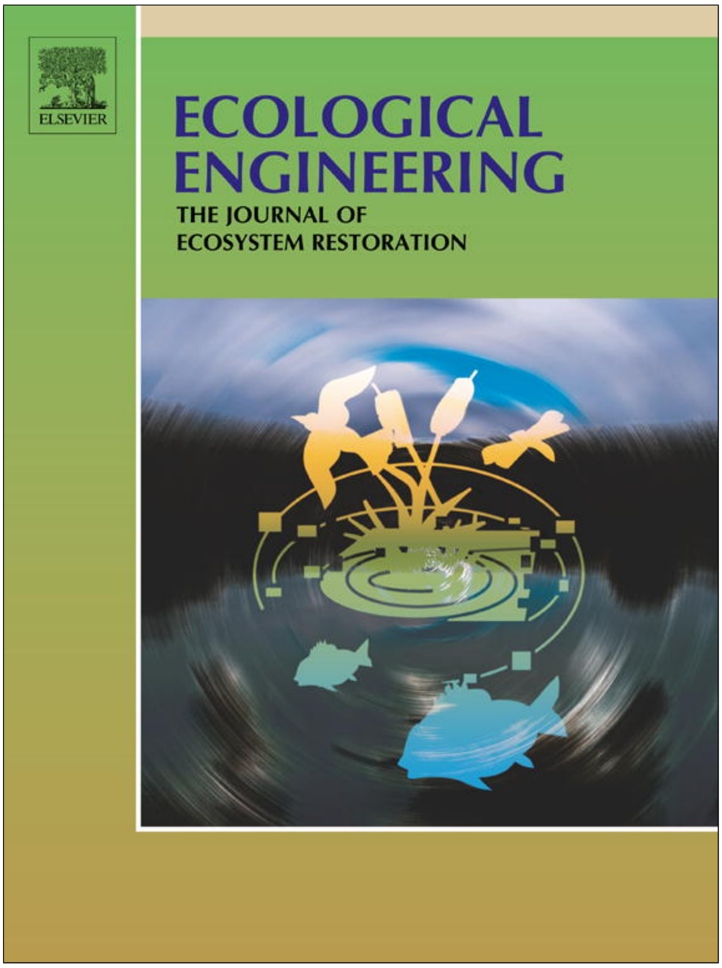
ELSEVIER
ECOLOGICAL ENGINEERING
THE JOURNAL OF
ECOSYSTEM RESTORATION

Ecological Engineering 51 (2013) 264–274

Contents lists available at SciVerse ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Functional assessment of differential sediment slurry applications in a deteriorating brackish marsh

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ARTICLE INFO

Article history:
Received 23 May 2012
Received in revised form 5 October 2012
Accepted 3 December 2012

Keywords:
Brackish marsh
Ecosystem function
Elevation change
Louisiana
Primary production
Restoration
Sediment slurry

ABSTRACT

We applied sediment slurries of varying thicknesses to deteriorating vegetated brackish marsh areas with organic soils. Our objective was to determine if overall elevation change and its component processes of soil compression and sediment consolidation are differentially affected by different amounts of sediment and to assess sediment effects on key ecosystem functions and physico-chemical drivers. We found that sediment nourishment (2.3–20.3 cm) increased soil surface elevation initially, but by the end of the ~2.5 year study period, sediment-nourished areas, averaged over all thicknesses, subsided to pre-sediment surface elevations, and were no different from reference area surface elevations. Rates of elevation change, soil compression, and sediment consolidation were all strongly related to applied sediment thickness; plots that received more sediment lost elevation at faster rates. Elevation change following sediment nourishment was driven by soil compression that occurred within the underlying native soil. In contrast, applied sediment consolidation over time had little negative influence on elevation change, though applied sediment bulk density increased linearly with increasing sediment thickness. Sediment nourishment <10 cm thick resulted in plots having lower elevations (up to ~2 cm lower) compared to pre-sediment conditions without enhanced function. However, plots that received >15 cm of sediment nourishment had both small elevation gains (<3 cm) compared to pre-sediment surface elevations and greater plant production compared to unamended reference plots. Factors influencing enhanced ecosystem function included a combination of greater soil bulk density, mineral matter content, available nutrient and trace metal concentrations, higher soil redox potential, and reduced salinity and sulfide stress that occurred with increasing sediment nourishment.

1. Introduction

Deltaic wetlands maintain intertidal elevations by accreting organic matter and trapping mineral sediment. Under natural conditions, periodic river flooding provides the mineral sediment that helps these wetlands maintain high rates of plant production and organic matter accumulation, which counterbalance the combined effects of subsidence and sea level rise (Nyman et al., 1990). However, if the primary source of mineral sediment is cut off, as is the case in the Mississippi River Delta, wetland plants can become vulnerable to flooding stresses. Excessive flooding can cause soil oxygen deficiency and accumulation of the phytotoxin hydrogen sulfide in soil porewater, which negatively affect wetland plant growth (Mendelssohn et al., 1981; Mendelssohn and McKee, 1988; Koch and Mendelssohn, 1989). Wetland plants experiencing flooding stress produce less belowground biomass (Wilsey et al., 1992; Howard and Mendelssohn, 1995) and have decreased rooting-depth (Megonigal and Day, 1992; Xie et al., 2008). Under these conditions, wetland soils can be more susceptible to erosion during extreme meteorological events, such as hurricanes, because of a reduced ability of the root system to stabilize the substrate (Howes et al., 2010). If the flooding stress is lethal, the denuded substrate can rapidly subside or erode creating shallow ponds that can coalesce into larger ponds over time, leading to large-scale land loss (DeLaune et al., 1994; Barras et al., 2003).

Due to these and other related factors (see Day et al., 2007), land loss in coastal Louisiana from 1978 to 2000 exceeded 75 km^2 yr^{-1} (Barras et al., 2003). Further degradation and loss of this resource is of utmost concern because these wetlands are some of the most productive ecosystems in the world and provide critical ecological functions and services, which include carbon storage, pollution

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http://dx.doi.org/10.1016/j.ecoleng.2012.12.031

abatement, nutrient sorption, soil formation, fisheries support, flood and storm protection, as well as many others (Costanza et al., 1997; Dawes, 1998; Keddy, 2000; Mitsch and Gosselink, 2000). Extensive efforts to rehabilitate, restore, and manage coastal wetlands are currently underway in Louisiana, with approximately $50 billion in new coastal protection and restoration projects planned for the future (CPRA, 2012). Although many restoration strategies have been employed, including hydrologic manipulation or marsh management, marsh terracing, river diversions, and beneficial-use of dredged sediment, the direct application of dredged sediments to deteriorating marshes has become increasingly common to offset elevation deficits and reestablish marshes because this method, unlike other restoration techniques, allows the placement of sediment in specific areas of critical importance (CPRA, 2012).

Considerable research has concluded that the hydraulic dispersal of sediment is an efficient and effective means of nourishing and rebuilding salt marshes, having an almost immediate impact on marsh function when appropriately applied. When applied to salt marshes, hydraulically conveyed sediment increases soil surface elevation and soil bulk density, decreases the frequency and duration of inundation, supplies minerals such as iron and manganese that precipitate hydrogen sulfide, and fertilizes plants with nutrients, which increases primary production (Ford et al., 1999; Mendelssohn and Kuhn, 2003; Slocum et al., 2005; Schrift et al., 2008; Stagg and Mendelssohn, 2010). These combined effects, in turn, produce a plant community that is more resilient to disturbance (Slocum and Mendelssohn, 2008; Stagg and Mendelssohn, 2011), and thus, increases sustainability compared to unamended salt marshes.

Although sediment-slurry application has been extensively investigated for salt marsh restoration (as cited above), a systematic and experimental evaluation of this restoration approach is lacking in lower salinity, brackish and freshwater marshes (only La Peyre et al., 2009). For instance, sediment loading on the marsh surface may cause the organic soils of brackish marshes to compress, potentially having a negative impact on elevation. However, the degree to which elevation change will occur with different amounts of applied sediment is currently unknown. Therefore, it cannot be assumed that sediment-slurry amendments in low salinity wetlands, which generally have more permeable organic soils, will produce the same results as in salt marshes, where soils are generally more mineral. Because net elevation gain is a function of (1) the amount of sediment applied, (2) the amount of compression that occurs within the underlying substrate due to the additional mass overlying it, and (3) consolidation of the applied sediment layer, an understanding of how different amounts of added sediment affect net elevation gain, especially in low salinity marshes with organic soils, is complex but critical for successful restoration of wetland function. Coastal management agencies must accurately predict subsidence and consolidation rates prior to project initiation in order to instruct dredging companies during construction. However, accurately achieving a target elevation that both minimizes costs (i.e., the amount of material applied) and maximizes benefits (i.e., ecosystem function and sustainability) is a challenge for coastal managers. Hence, it is presently difficult to predict the outcome of sediment-slurry nourishment in these wetland types, which dominate in coastal Louisiana. Therefore, the objectives of this research were to (1) determine if elevation change, soil compression, and sediment consolidation were differentially affected by different amounts of sediment applied to a brackish marsh, (2) identify the primary driver of elevation change following sediment nourishment, (3) assess sediment effects on ecosystem function, and (4) identify the physico-chemical drivers of observed sediment effects.

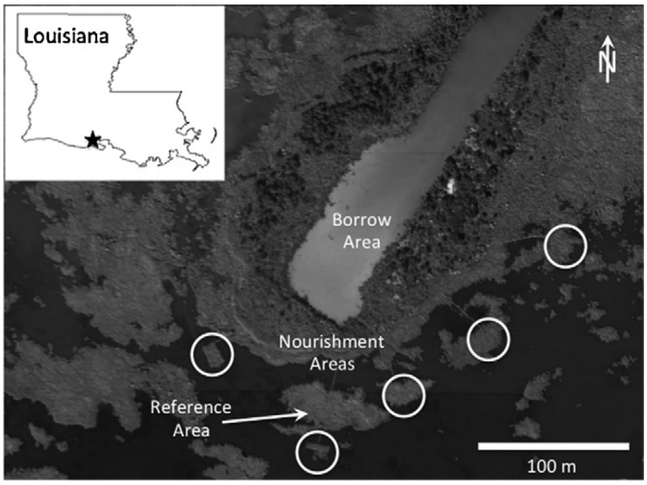

Fig. 1. Site location map showing the borrow area, sediment nourishment areas (circled), and the reference area.
Source: Google Maps 2010.

2. Materials and methods

2.1. Study site

To carry out our objectives, we initially established 20, 3 m × 4 m, vegetated plots in 5 areas within an 800 ha, semi-impounded and deteriorating *Spartina patens* (Aiton) Muhl. dominated brackish marsh located within the Paul J. Rainey Wildlife Sanctuary in Vermillion Parish, Louisiana (N29.692824°, W92.228194°; Fig. 1). Prior to adding sediment, a water level recorder (Remote Data Systems, Inc.) was installed at the site to monitor hydrology on an hourly basis. Elevated wooden boardwalks were also constructed around each plot so that sediment applications and sampling could take place without any disturbance to the marsh surface within the plots. Wire-backed silt fencing was then attached to the boardwalk around the perimeter of each plot and trenched 30 cm in to the marsh soil to contain the sediment. At 3 locations within each plot, 20.3 cm diameter perforated-aluminum settling-disks were set flush with the soil surface and tethered to PVC stakes inserted into the marsh surface to serve as survey points for surface elevation measurements. The soil surface elevation at each settling disk in each plot, as well as the elevation of the calibration point on the water level recorder, was then determined using a class 1 laser level (Sokkia LP30A). All elevation measurements were referenced to a 10.7 m deep permanent benchmark that we installed and surveyed relative to NAVD88 using a Trimble R7 RTK-GPS receiver and Zephyr Geodetic antenna. The pre-sediment soil surface elevations provided a baseline to which all subsequent measurements (i.e., post-sediment addition) could be compared.

Sediment was pumped from the adjacent canal as a slurry of water (∼70–80%) and sediment (∼20–30%) into each of the twenty vegetated plots using a small, hand-operated dredge (Piranha PS-135-E). While dredging, we collected 10 composite samples of the discharged sediment prior to deposition on the marsh for physico-chemical analysis (see Section 2.2.4). Dredging occurred over a 5-week period from July 7, 2008 to August 13, 2008. However, on September 13, 2008, prior to the completion of dredging operations and reference plot establishment, Hurricane Ike made landfall approximately 250 km west of our site. During the storm, the study site experienced a 3 m storm surge, which resulted in the physical removal of approximately 50 ha of marsh from the semi-impounded wetland unit (Yvonne Allen, U.S. Army Corps of

Engineers – Engineer Research and Development Center, unpublished GIS data). Our post-storm site assessment in October 2008 revealed that a substantial portion of our boardwalk system was also destroyed, and the canal from which we were dredging and also used to access the site was filled with detached marsh and debris. However, only one plot was deemed unusable due to excessive debris deposition, leaving 19 sediment-nourished plots. At that time, we established 4 reference plots that did not receive sediment and continued with our assessment. We re-built the necessary site infrastructure in November and December 2008 for continued sampling, although additional dredging was not possible.

2.2. Sample collection and analysis

2.2.1. Elevation change

After adding sediment, we re-surveyed the soil surface elevation at each settling disk in each sediment-nourished plot at approximately 3-month intervals from August 2008 to January 2011 (i.e., 10 measurements over time). We also simultaneously measured the thickness of sediment over each disk by inserting a ruler into the sediment until it made contact with the disk. The elevation of each disk was determined by subtracting the sediment thickness from the measured surface elevation. All post-sediment surface elevation measurements were then compared to the pre-sediment surface elevation measurements to determine elevation change. Likewise, all post-sediment disk elevation measurements were compared to the pre-sediment disk (i.e., surface) elevation measurements to determine compression of the underlying native substrate resulting from the sediment additions. Sediment consolidation was determined by changes in sediment thickness over time.

Surface elevation in the reference plots was measured nine times from October 2008 to January 2011. Since sediment was not added to the reference plots, it was not necessary to install settling disks. Rather, elevation change was determined based on changes in the soil surface over time at three locations within each reference plot using the same survey method described above.

In all plots, the 3 replicate measurements were averaged, and then regression analysis was used to estimate elevation change (all plots), soil compression, and sediment consolidation (sediment nourished plots) relationships with time. To estimate rates of change in sediment-nourished plots, we assumed that the maximum possible elevation was achieved (i.e., elevation increase = sediment thickness) on the final day of dredging, and no compression or consolidation occurred until the day after dredging ended. However, we acknowledge that dewatering of the applied sediment had occurred prior to our first post-sediment measurements.

Time-series measurements of elevation change, soil compression, and sediment consolidation in sediment-nourished plots were then fit to the log-linear equation:

$$y = a \times \ln(t) \qquad (1)$$

where (y) is elevation change, soil compression, or sediment consolidation (cm), (a) is the regression coefficient, and $\ln(t)$ is the natural log of time in days. Elevation change in reference plots was not well represented by curvilinear or linear regression, and rates were not calculated. Instead, we determined cumulative change on the final measurement for reference plots.

2.2.2. Plant production

Net aboveground plant production was estimated by clipping randomly chosen 0.1 m^2 sub-plots within each plot every 3 months from February 2009 to October 2010 for a total of 7 biomass harvests. Clipped plant biomass was separated into live and dead categories, dried to a constant weight at 60 °C, and weighed. Estimates of aboveground production in 2009 and 2010 were then calculated based on changes in both live and dead biomass over time using the Smalley Method (Smalley, 1959).

Net belowground plant production was estimated using in-growth bags (5 cm diameter by 30 cm long, 1.5 mm × 1.5 mm woven mesh bags, packed with finely ground peat). Four in-growth bags were installed in each plot in January 2009 by removing soil cores of the same dimensions and replacing them with the in-growth bags. Two randomly selected in-growth bags were then retrieved from each plot after 1 and 2 years by removing slightly larger soil cores that encapsulated the bags. Upon removal, the bags were rinsed of all mud, trimmed of external root/rhizomes, and washed over a 2-mm mesh screen to remove the peat packing material. The roots and rhizomes that remained were dried to a constant weight at 60 °C, and weighed. An average production rate was calculated for each plot during each time period prior to statistical analysis.

To calculate net total plant production, aboveground production was calculated to match the belowground production (i.e., aboveground production after 2 years = (2009 NAPP + 2010 NAPP)/2). Aboveground production and belowground production were then summed to get total production after 1 and 2 years.

2.2.3. Litter decomposition

Above- and belowground plant litter decomposition was measured using 6 cm wide × 30 cm long litterbags constructed from 1-mm^2 nylon mesh. Aboveground decomposition was estimated by placing 6 litterbags filled with 7 g (oven-dried) of *S. patens* stems/leaves on the marsh surface. Belowground decomposition was estimated by inserting 6 litterbags filled with 3 g (oven-dried) of *S. patens* roots/rhizomes (2.23:1 ratio) into the marsh soil at a depth of 15 cm. One randomly selected above- and belowground litterbag was then retrieved after approximately 2 weeks, 1 month, 3 months, 6 months, 1 year, and 2 years. Upon retrieval, the bags were rinsed of all mud, and any identifiable in-grown roots/rhizomes were removed. The remaining material was dried to a constant weight at 60 °C, and weighed. The percent (%) mass remaining over time was then fit to the exponential decay function:

$$y = e^{-kt} \qquad (2)$$

where (y) is the percent (%) mass remaining, (k) is the exponential decay constant, and (t) is time in years.

2.2.4. Soil and sediment physico-chemistry

In addition to the 10 composite sediment slurry samples collected while dredging, 2 soil cores were extracted from each plot following sediment application using a 5 cm diameter Russian Peat Borer (Aquatic Research Instruments, Hope, ID) during 8 samplings from February 2009 through January 2011. Both physical and chemical analyses were performed on each homogenized composite sediment slurry sample, while one core from each sampling date was designated for physical characterization and the other for chemical characterization. Physical characterization included measurements of soil bulk density (Blake and Hartge, 1986), percent moisture (Gardner, 1986), percent organic matter (LOI at 550 °C; Christenson and Malmros, 1982), and particle size (pipette method; Gee and Bauder, 1986). However, bulk density of the composite samples could not be determined because the sediment slurry was in transit (i.e., undefined volume). Chemical characterization included measurements of pH (1:1 water) and exchangeable nutrient concentrations. Ammonium-N was determined using the automated phenate method (APHA, 2005) following 2 M KCl extraction and 0.45 μm filtration (Mulvaney, 1996). Additional mineral nutrients and trace metals were measured by inductively coupled plasma (ICP) spectrometry (Spectro Ciros) following extraction with either Mehlich 3 test solution (calcium, magnesium, phosphorus, potassium, sodium, and sulfur; Mehlich, 1984), DTPA (copper, iron, manganese, and zinc; Lindsay and Norvell, 1978), or 0.1 N HCl (nickel, soil cores only; Wears and Sommer, 1948). All samples for physico-chemical analysis were stored on ice while in the field and at 4 °C upon returning to the lab. Soil extractions were performed within 1 week of sample collection, and sample analyses occurred within 1 month of collection. All nutrient and trace metal samples were analyzed at field moistures. Soil moisture and bulk density correction factors were subsequently applied to present values on a dry volume (g cm^{-3}) basis, except for composite samples collected while dredging, which were presented on a dry mass (mg kg^{-1}) basis.

A third soil core was also collected from the sediment-amended plots in January 2011 only, and sectioned into two increments: (1) the applied-sediment layer and (2) the top 15 cm of underlying native soil. Bulk density (Blake and Hartge, 1986) was then determined for both increments.

Soil accretion, defined as the vertical accumulation of organic and/or mineral matter, was determined in the field from cores collected from all plots in December 2009, July 2010, and January 2011. In the sediment-amended plots, accretion was measured as any accumulation above the applied sediment layer. In reference plots, soil accretion was measured as any soil accumulation above a thin (~1–2 cm) hurricane deposition layer composed primarily of sand and clay (S.A. Graham, personal observation). The rate of accretion over time was then estimated for all plots using linear regression.

Soil oxidation–reduction potential (Redox) was measured in situ during all samplings by inserting 3 bright platinum electrodes and a calomel reference electrode into the sediment of each plot to a depth of 15 cm. Platinum electrodes were allowed to equilibrate for at least 30 min before reading the potential using a digital mV meter. The 3 readings within each plot were averaged and corrected for the reference electrode value prior to statistical analysis.

2.2.5. Porewater chemistry

Porewater water samples were extracted from each plot in May 2009, December 2009, and July 2010 using a porewater "sipper" similar to that described by McKee et al. (1988). The sipper was inserted into the soil and 4 separate samples were extracted under suction pressure over a depth range of 10–20 cm below the soil surface. The first sample collected was used to determine pH and salinity (APHA, 2005). The second sample was immediately preserved at a 1:1 ratio with antioxidant buffer for sulfide analysis (APHA, 2005). The third sample was filtered (0.45 μm) and analyzed for plant available NH_4-N by the automated phenate method (APHA, 2005). The fourth sample was filtered (0.45 μm), preserved with HCl, and analyzed for a suite of plant available mineral nutrients and trace metals (i.e., Al, B, Ca, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, P, S, and Zn) by inductively coupled plasma (ICP) spectrometry (Spectro Ciros) (APHA, 2005). All porewater samples were stored on ice while in the field. Upon returning to the lab, pH/salinity and sulfide samples were stored at 4 °C and 25 °C, respectively, and analyzed within 24 h of collection. Ammonium-N samples were immediately frozen at −20 °C upon returning to the lab, and analyzed within 28 days of collection, while ICP samples were stored at 4 °C and analyzed within 2 months.

2.3. Statistical analysis

All statistical analyses were conducted using SAS (Statistical Analysis Systems, version 9.2, SAS Institute, Inc., Cary, NC). Linear regression analyses (PROC REG) were used to identify predictive relationships and trends with increasing applied sediment thickness, while one-way ANOVA (PROC MIXED) was used to identify causal relationships associated with sediment application or time. For ANOVA of sediment nourishment effects, plots were grouped into 4 treatment categories based on applied sediment thickness: Control (no sediment; $n = 4$), Low (<10 cm; $n = 4$), Medium (10–15 cm; $n = 10$), and High (>15 cm; $n = 5$). When no significant treatment effect existed, all plots that received sediment ($n = 19$) were grouped and compared to the control ($n = 4$).

Time-series measures such as plant production, soil physico-chemistry (0–15 cm), and porewater chemistry, not previously converted to rates by regression were initially analyzed using two-way ANOVA (PROC MIXED) to determine if the time of sampling was a significant effect alone or in interaction with sediment thickness. However, there was no effect of year on plant production estimates, and although a significant effect of time was present for several soil and porewater variables, in all cases statistical differences were isolated to single sampling periods, with no discernable trends. Thus, prior to ANOVA, all plant, 0–15 cm soil physico-chemical variables, and porewater chemical variables were time-averaged to obtain a mean value for each plot.

The dimensionalities of correlated soil and porewater chemical variables were further reduced by factor analysis (PROC FACTOR), using the principal-axis method of extraction and squared multiple correlations of each variable with all other variables as prior communality estimates. Factors with eigenvalues greater than 1 were retained and orthogonally rotated with a varimax rotation (Kaiser, 1958). Variables with correlation coefficients ≥0.6 were used to define the retained factors. Factors scores generated for each plot were then analyzed using one-way ANOVA (PROC MIXED) and linear regression (PROC REG), with time-averaged (mean) sediment thickness as the fixed effect and independent variable, respectively.

Treatment means in PROC MIXED were tested using the least-squares (LS) means procedure with a Tukey–Kramer adjustment to maintain an experiment-wise error rate. Measures of significance were identified at $p < 0.01$, 0.05, and 0.1. Assumptions of homogeneity of variance and goodness of fit to a normal distribution were verified by examining residual plots and normal probability plots, respectively (PROC UNIVARIATE). When necessary, these data were logarithmically, square root, or square transformed prior to analysis to validate model assumptions.

3. Results

3.1. Elevation change, soil compression, and sediment consolidation

Prior to adding sediment, marsh surface elevation within the vegetated plots averaged 36.6 ± 0.5 cm-NAVD88 (Fig. 2a). Sediment slurry applications ranged from 2.3 to 20.3 cm, and average surface elevation of all plots receiving sediment ($n = 19$) increased significantly ($p < 0.05$) to 41.7 ± 0.7 cm-NAVD88 immediately (16 days) after applications ended. Due to variation in marsh surface microtopography, plots that received the most sediment did not necessarily end up at the highest elevation, although a highly significant ($p < 0.002$) positive linear relationship with increasing sediment thickness accounted for 43% of the variability in marsh surface elevation (cm-NAVD88 = 36.8 + 0.4 × applied sediment thickness). An additional increase in overall soil surface elevation occurred immediately after Hurricane Ike, due to the deposition of material scoured from the surrounding marsh (Fig. 2a). However, within 365 days of sediment application, the mean surface elevation of sediment-nourished plots had decreased to where the initial increase in elevation resulting from sediment application was no longer significantly different from the surface elevation in

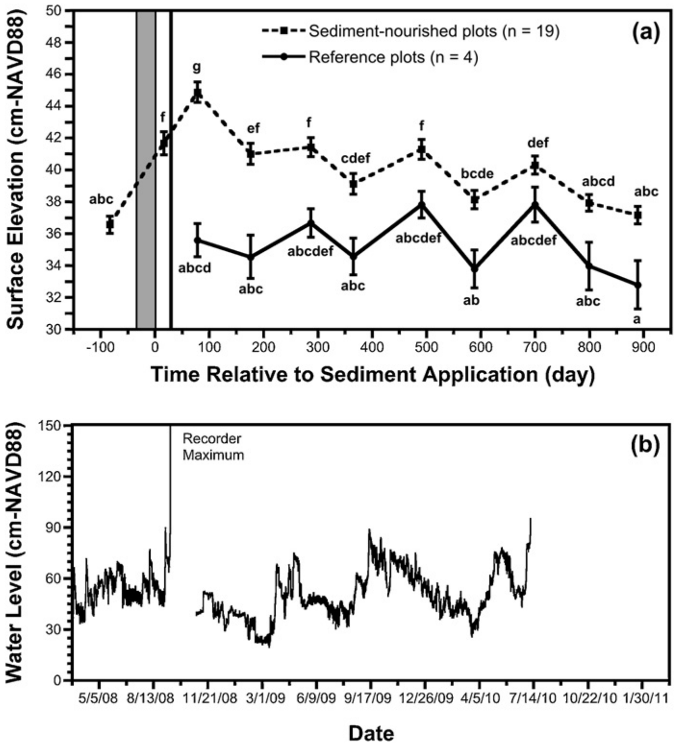

Fig. 2. (a) Mean (plot-averaged) soil surface elevation (±1 SE) of sediment-nourished plots (before and after sediment applications) and reference plots. The vertical gray bar shows when dredging took place. The vertical black line shows when Hurricane Ike made landfall. Means separated by different letters are significantly different ($p < 0.05$, Tukey–Kramer multiple comparison test). Lines connecting measurements over time do not imply that regular changes occurred during time periods between measurements. (b) Hourly water level record: 3/16/08 through 7/7/10 (end of record). Water level recorder maximum = 150 cm-NAVD88. *X*-Axis time-period and labels in graph (b) correspond to those in graph (a).

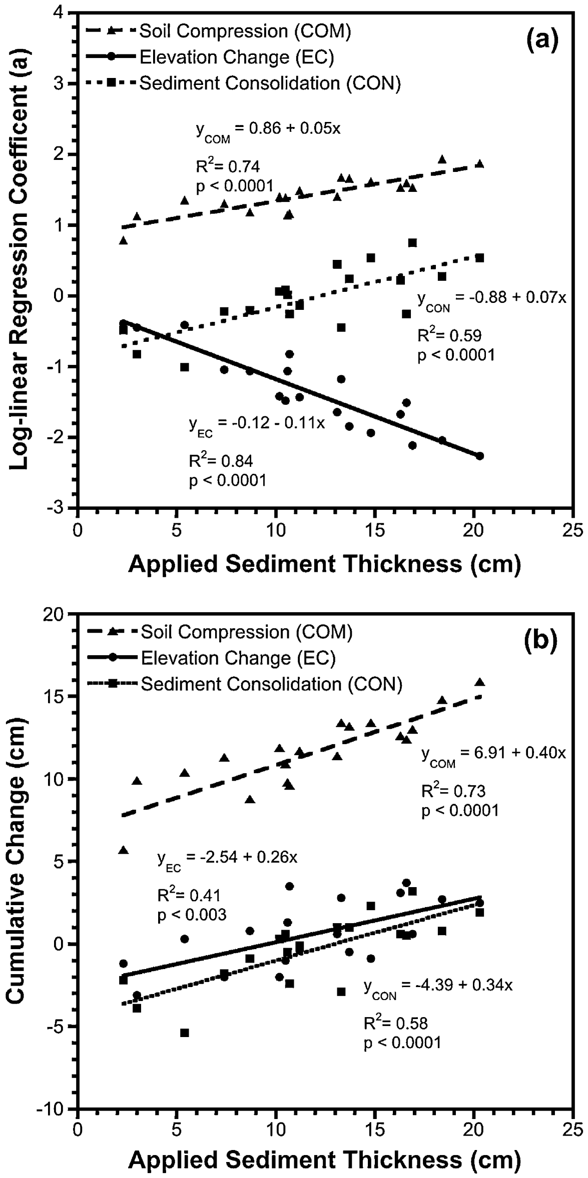

Fig. 3. The relationship between applied sediment thickness, elevation change, soil compression, and sediment consolidation presented as (a) the rate of change, as determined by the log-linear regression coefficient, *a*, from the equation $y = a \times \ln(t)$, and (b) the cumulative change 889 days after sediment application.

reference plots or the pre-sediment mean elevation of sediment-nourished plots. In contrast, there was no significant change in reference plot surface elevations over time. Although, the effect of water level fluctuations was apparent. High water during samplings that occurred 287, 491, and 700 days after sediment applications likely contributed to increases in surface elevation on average compared to the preceding measurement (Fig. 2a and b). But, by the end of the study, 889 days after sediment addition, the average surface elevation of sediment-nourished plots ($n = 19$) had decreased to 37.2 ± 0.5 cm-NAVD88 (mean ± 1 SE), similar to both pre-application and reference elevations.

The statistical model (Eq. (1)) used to estimate elevation change over time in sediment-nourished plots (i.e., log-linear regression estimates on a per plot basis) explained 66 ± 6% of the variability, on average (data not shown). However, model estimates in plots that received <5 cm of sediment ($n = 3$) accounted for substantially less variability (15 ± 2%) compared to estimates in plots that received greater amounts (75 ± 4%; $n = 16$). The amount of variability explained by the log-linear model was mixed for measured components of elevation change (i.e., soil compression and sediment consolidation). On average, 75 ± 2% of the variability in the rate of compression within the underlying 15 cm of native soil was explained by the model, compared to 31 ± 6% for the rate of sediment consolidation over time. Low coefficients of determination associated with estimated rates of sediment consolidation can likely be attributed to deposition/scour resulting from Hurricane Ike, as well as root/rhizome in-growth into the newly deposited sediment.

Rates of elevation change, soil compression, and sediment consolidation, as determined by the log-linear regression coefficient (*a*), were all strongly related to applied sediment thickness (Fig. 3a). Soil compression and sediment consolidation both increased with increasing sediment thickness, although soil compression occurred on average at a much greater rate than sediment consolidation, regardless of sediment thickness. Linear regression showed that plots receiving less than ∼12.5 cm of sediment had negative rates of sediment consolidation, indicating that sediment expansion occurred. Regardless, all plots that received sediment decreased in elevation after application, and plots that received more sediment lost elevation at a faster rate.

After 889 days, cumulative elevation change in sediment-nourished plots ranged from −1.9 to 2.7 cm, on average. Over approximately that same time period, elevation in reference plots decreased by 2.8 ± 1.6 cm. Cumulative elevation change in

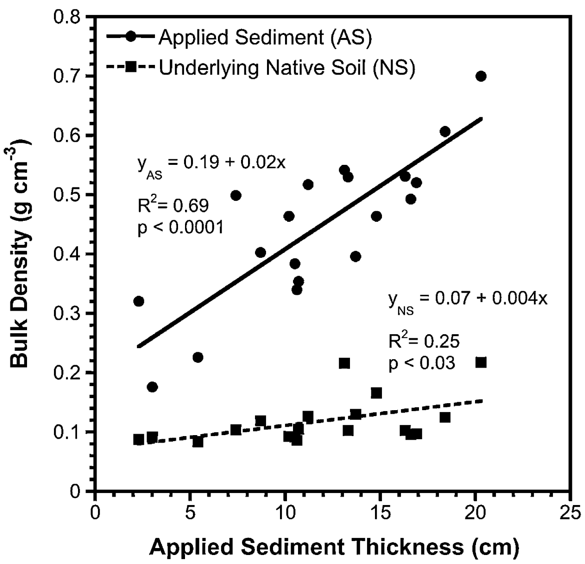

Fig. 4. The relationship between applied sediment thickness and bulk density of the applied sediment layer and the top 15 cm of underlying native soil 889 days after sediment application.

sediment-nourished plots had a strong positive relationship with applied sediment thickness (Fig. 3b). However, compression of the underlying soil had almost completely counterbalanced any elevation gain associated with the sediment applications. In fact, linear regression showed that plots nourished with low amounts of sediment (i.e., <10 cm) actually lost elevation relative to pre-application elevation, despite sediment expansion likely caused by root growth into sediment layer over the approximate 2.5 year period. In contrast, sediment consolidation accounted for very little elevation change overall, though it displayed a highly significant positive linear trend with increasing applied sediment thickness (Fig. 3b).

3.2. Applied sediment layer and underlying soil bulk density

Bulk densities of the applied sediment layer and the top 15 cm of underlying native soil increased with increasing sediment thickness (Fig. 4). The linear relationship with applied sediment thickness accounted for 69% of the variability in applied sediment bulk density, and 25% of the variability in underlying native soil bulk density. On average, applied sediment bulk density increased from approximately 0.24 to 0.63 g cm^{-3}. A much more modest increase from 0.08 to 0.11 g cm^{-3} occurred within the underlying native soil, despite high rates of soil compression.

3.3. Marsh function

Both above- and belowground plant production increased with increasing sediment thickness (Table 1). Nourishment with more than 15 cm of sediment (i.e., high treatment) resulted in greater rates of both compared to control plots ($p < 0.06$ for both). Results were similar, albeit more significant ($p < 0.05$), for total plant production (aboveground + belowground production). However, enhanced plant production did not translate in to enhanced soil accretion. Plots that received moderate levels of sediment had lower ($p < 0.1$) accretion compared to control plots, while plots receiving either low or high levels of sediment had intermediate accretion rates (Table 1).

Sediment addition also increased ($p < 0.1$) belowground decomposition, but the treatment groupings (i.e., high, medium, and low) could not be distinguished from the overall sediment effect (Table 1). In contrast, sediment nourishment had no effect on aboveground decomposition. Overall, aboveground decomposition progressed with an exponential decay constant (k) approximately twice that belowground.

3.4. Soil physico-chemistry

The sediment slurry pumped on to vegetated marsh plots was composed primarily of silt and clay (combined 82%), and sediment nourishment resulted in significantly greater soil bulk density ($p < 0.01$) and significantly lower soil moisture ($p < 0.05$) and organic matter ($p < 0.01$) for all treatments when compared to control plots and each other (Table 2, Appendix A). The effect of sediment nourishment on soil texture (i.e., sand, silt, and clay content), however, was not as straightforward. In general, sediment nourishment significantly ($p < 0.05$) decreased sand content compared to the control plots, whereas both silt and clay increased significantly ($p < 0.05$).

The chemical composition of the sediment slurry was reflective of its high mineral content, and, in general, essential plant nutrient concentrations were present in relatively high concentrations (Appendix A). Factor analysis identified 3 factors that explained a total of 89% of the variation in the soil (0–15 cm) chemical data following applications (Table 3, Appendix B). Soil factor 1 explained 66% of the variation, with high positive correlations for K, Ca, pH, Mg, Zn, Cu, P, Mn, and Ni, and was interpreted as being related to soil mineral/nutrient content. Soil factor 2 explained 14% of the variation in the soil data and had high positive correlations with Fe, S, and Na, which were interpreted as also being related to soil

Table 1
Effects of sediment slurry amendments on aboveground, belowground, and total plant production, above- and belowground decomposition (k = exponential decay constant), and soil accretion. Treatment means (±1 SE) are presented and identified as significantly different by different letters (*$p < 0.1$, **$p < 0.05$; Tukey–Kramer multiple comparison test). When no significant treatment effect existed, all plots that received sediment (=sediment) were grouped and compared to the control (=no sediment). If there was no significant sediment effect, the overall mean (=all plots) is displayed.

Sediment treatment	Aboveground production (g m^{-2} yr^{-1})*	Belowground production (g m^{-2} yr^{-1})*	Total production (g m^{-2} yr^{-1})**	Aboveground decomposition (k)	Belowground decomposition (k)*	Soil accretion (cm yr^{-1})*
Control (0 cm)	1301 ± 260^a	200 ± 28^a	1501 ± 285^a	0.61 ± 0.02 (all plots)	0.26 ± 0.01^a (no sediment)	2.1 ± 0.5^b
Low (<10 cm)	1931 ± 316^{ab}	412 ± 62^{ab}	2343 ± 363^{ab}		0.33 ± 0.02^b (sediment)	1.8 ± 0.1^{ab}
Med (10–15 cm)	2011 ± 160^{ab}	462 ± 77^{ab}	2473 ± 170^{ab}			1.3 ± 0.2^a
High (>15 cm)	2461 ± 347^b	472 ± 77^b	2932 ± 327^b			1.4 ± 0.1^{ab}

Table 2
Effects of sediment slurry amendments on time-averaged (mean) soil bulk density, moisture, organic matter, sand, silt, and clay (0–15 cm). Treatment means (±1 SE) are presented and identified as significantly different by different letters (**$p < 0.05$, ***$p < 0.01$; Tukey–Kramer multiple comparison test).

Sediment treatment	Bulk density ($g\,cm^{-3}$)***	Moisture (%)**	Organic matter[a] (%)***	Sand[a] (%)**	Silt[a] (%)***	Clay[a] (%)**
Control (0 cm)	0.09 ± 0.004[a]	84.2 ± 0.4[d]	43.6 ± 2.0[d]	24.0 ± 0.9[c]	17.0 ± 1.7[a]	16.0 ± 1.7[a]
Low (<10 cm)	0.23 ± 0.05[b]	70.8 ± 4.0[c]	23.8 ± 3.7[c]	18.7 ± 1.9[ab]	29.1 ± 2.4[b]	28.6 ± 2.8[b]
Med (10–15 cm)	0.32 ± 0.02[c]	62.8 ± 1.6[b]	15.6 ± 1.4[b]	15.1 ± 0.5[a]	36.0 ± 1.4[c]	33.4 ± 0.6[c]
High (>15 cm)	0.44 ± 0.03[d]	56.5 ± 1.3[a]	10.6 ± 0.8[a]	21.8 ± 2.8[bc]	36.7 ± 0.7[c]	30.9 ± 1.6b[c]

[a] Presented as percent of total soil (organic matter + sand + silt + clay = 100%).

Table 3
Factor analysis of time-averaged soil chemical variables. Indicator variables corresponding to bolded correlation coefficients (≥±0.6) define the factor, unless noted otherwise.

Indicator variable	Correlation coefficients		
	Factor 1	Factor 2	Factor 3
Potassium	**0.92673**	0.29468	−0.00561
Calcium	**0.91451**	−0.09267	−0.00896
pH	**0.90446**	−0.37326	−0.13780
Magnesium	**0.90259**	0.33928	−0.08266
Zinc	**0.88405**	0.22782	−0.07127
Copper	**0.87980**	0.34384	0.13164
Phosphorus	**0.80129**	0.34143	0.20728
Manganese	**0.75511**	0.37494	0.01630
Nickel	**0.71081**	0.25534	0.41093
Iron	0.02865	**0.90489**	0.28276
Sulfur	**0.60465**[a]	**0.66700**	−0.11004
Sodium	0.49495	**0.59047**	−0.50358
Red-ox	0.32283	0.24710	**0.74723**
Ammonium-N	0.02921	0.00777	−0.17267
Eigenvalue	8.19	1.73	1.11
Variance explained	66%	14%	9%

[a] Soil sulfur is more strongly correlated with soil factor 2.

Table 5
Factor analysis of time-averaged porewater chemical variables. Indicator variables corresponding to bolded correlation coefficients (≥±0.6) define the factor.

Indicator variable	Correlation coefficients			
	Factor 1	Factor 2	Factor 3	Factor 4
Salinity	**0.94224**	−0.09385	0.16238	0.05074
Sodium	**0.93192**	−0.15601	0.00853	−0.00198
Magnesium	**0.92053**	0.09529	0.23779	0.05219
Potassium	**0.89675**	−0.1837	0.07352	0.12931
Boron	**0.84797**	−0.08741	−0.24446	−0.00953
Sulfur	**0.79025**	−0.41753	−0.12332	0.0197
Sulfide	**0.73081**	−0.6065	−0.03834	−0.00362
Ammonium-N	**0.62058**	−0.15332	−0.231	0.10384
Copper	**−0.927**	0.1231	−0.07235	−0.10057
Manganese	−0.22253	**0.8611**	0.35442	−0.20788
Phosphorus	−0.01064	**0.84057**	0.01063	−0.04739
Nickel	−0.2527	**0.80057**	−0.41916	−0.09407
Iron	−0.22823	**0.70307**	−0.4202	−0.00537
Zinc	−0.36885	**0.59861**	−0.24426	0.49079
Molybdenum	0.06339	0.51388	0.22469	0.49675
pH	−0.07391	−0.08035	**0.93117**	0.19895
Aluminum	0.05864	−0.07143	0.19948	**0.63194**
Calcium	0.35587	−0.32631	−0.17976	0.51742
Eigenvalue	8.20	2.84	1.79	1.17
Variance explained	50%	17%	11%	7%

mineral content as well as soil salinity. Soil factor 3 explained 9% of the variation in the soil data with a high positive correlation for soil redox potential only, and thus appeared to be a soil waterlogging factor.

The soil chemical environment, as represent by soil factor 1, was similar between the control plots and plots that received low amounts of sediment, but plots that received medium and high sediment were significantly different ($p < 0.001$) from them, as well as each other (Table 4). In fact, a highly significant positive linear relationship ($R^2 = 0.76$, $p < 0.0001$) existed between mean sediment thickness and soil factor 1. Sediment additions also had a significant positive effect on soil factors 2 ($p < 0.05$) and 3 ($p < 0.10$), but only when sediment-nourished plots were compared as a group ($n = 19$) to control plots ($n = 4$).

3.5. Porewater chemistry

Factor analysis of the interstitial water variables resulted in 4 factors that explained 85% of the variance in the data (Table 5, Appendix B). The first factor explained 50% of the variation and had high positive correlations with salinity, Na, Mg, K, B, S, S^{2-}, and NH_4-N, and a high negative correlation with Cu. The second porewater factor had high positive correlations with Mn, P, Ni, Fe, and Zn and explained an additional 17% of the porewater data variability. The first two factors were interpreted as being related to salinity/flooding and nutrients, respectively. Porewater factors 3 and 4 explained 11% and 7% of chemical variability, respectively,

Table 4
Effects of sediment slurry amendments on soil chemical factors. Variables that define each factor are identified in parentheses. Factor score means (±1 SE) for each treatment are identified as significantly different by different letters (*$p < 0.1$, **$p < 0.05$, ***$p < 0.01$; Tukey–Kramer multiple comparison test). When no significant treatment effect existed, all plots that received sediment (=sediment) were grouped and compared to the control (=no sediment).

Sediment treatment	Factor scores		
	Factor 1*** (K, Ca, pH, Mg, Zn, Cu, P, Mn, Ni)	Factor 2** (Fe, S, Na)	Factor 3* (Redox)
Control (0 cm)	−1.36 ± 0.07[a]	−0.96 ± 0.25[a] (no sediment)	−0.75 ± 0.55[a] (no sediment)
Low (<10 cm)	−0.50 ± 0.30[ab]		
Med (10–15 cm)	0.18 ± 0.21[b]	0.20 ± 0.22[b] (sediment)	0.16 ± 0.20[b] (sediment)
High (>15 cm)	1.13 ± 0.27[c]		

Table 6
Effect of sediment slurry amendments on porewater chemical factors. Variables that define each factor are identified in parentheses. Factor score means (±1 SE) for each treatment are identified as significantly different by different letters (**$p < 0.05$, ***$p < 0.01$; Tukey–Kramer multiple comparison test). When no significant treatment effect existed, all plots that received sediment (=sediment) were grouped and compared to the control (=no sediment). If there was no significant sediment effect, the overall mean (=all plots) is displayed.

Sediment treatment	Factor scores			
	Factor 1*** (salinity, Na, Mg, K, B, S, S^{2-}, NH_4-N, –Cu)	Factor 2**(Mn, P, Ni, Fe, Zn)	Factor 3 (pH)	Factor 4 (Al)
Control (0 cm)	1.63 ± 0.47[b]	−0.99 ± 0.16[a]		
Low (<10 cm)	−0.46 ± 0.37[a]	−0.53 ± 0.42[ab]	0.00 ± 0.21 (all plots)	0.00 ± 0.20 (all plots)
Med (10–15 cm)	−0.48 ± 0.19[a]	0.01 ± 0.22[bc]		
High (>15 cm)	0.03 ± 0.15[ab]	1.20 ± 0.37[c]		

and each were related to a single indicator variable (factor 3 = pH, factor 4 = Al).

Sediment nourishment had significant effects on porewater factors 1 and 2 but not factors 3 and 4 (Table 6). Compared to control plots, scores for porewater factor 1 were significantly ($p < 0.01$) lower for plots that received either low or medium amounts of sediment, and plots that received high amounts of sediment were intermediate. Conversely, porewater factor 2 scores increased significantly ($p < 0.05$) with increasing sediment nourishment.

4. Discussion

This research confirms that the application of sediment to deteriorating brackish marshes provides an immediate increase in soil surface elevation, as documented in salt marshes (Mendelssohn and Kuhn, 2003; Slocum et al., 2005; Schrift et al., 2008; Stagg and Mendelssohn, 2010). However, contrary to sustained elevation gains documented over the long-term in salt marsh research (Edwards and Proffitt, 2003; Slocum et al., 2005; Stagg and Mendelssohn, 2011), the initial elevation increase resulting from sediment nourishment in this brackish marsh was relatively short lived. During this experiment between 12 and 129 kg m^{-2} of additional mass were deposited on top of the native marsh soil depending on applied sediment thickness (2.3–20.3 cm), and log-linear regression estimates on a per plot basis indicate a rapid rate of soil compression and elevation loss initially that increased with increasing sediment thickness. By study's end (~2.5 year), realized elevation gain was on average 74–339% less than potential elevation gain due to soil compression following sediment application (i.e., soil compression = 6.91 + 0.40 × sediment thickness; Fig. 3b). In contrast, applied sediment consolidation over time had little negative influence on elevation change, though its influence increased with increasing sediment thickness, up to 12% of applied sediment thickness (i.e., sediment consolidation = −4.39 + 0.34 × sediment thickness; Fig. 3b). In plots that received less than 15 cm of sediment, the sediment actually expanded up to 157% beyond applied thicknesses, most likely due to root and rhizome in-growth, although the effects of Hurricane Ike cannot be ruled out because the amount of storm deposition increased as elevation decreased ($R^2 = 0.46$, $p < 0.001$). However, due to the type of material deposited (i.e., highly organic scoured marsh soil), rates of soil compression were likely not affected.

These results show that the native marsh soil at our research site, having high organic matter content, low bulk density, and high moisture content, is highly compressible. Similarities among our references plots and published values from other brackish marshes in Louisiana (Nyman et al., 1990; DeLaune et al., 2003; La Peyre et al., 2009) suggest that brackish marsh soils in general may be highly compressible as well. Within 1 year, the surface elevation of plots that received sediment was approaching equivalency with the reference plots, and by the end of the study 889 day after sediment additions, sediment-nourished areas on average had subsided to pre-sediment surface elevations. Depending on the amount of sediment applied, approximately 87–183% of the potential elevation gain, as applied sediment thickness, was lost to subsidence (i.e., elevation change = −2.54 + 0.26 × sediment thickness; Fig. 3b). Likewise, elevation gain was <14% of the amount of sediment applied, which resulted in a net elevation increase of <3 cm with sediment thicknesses ranging from 10 to 20 cm. Although deposition from Hurricane Ike likely enhanced, to some extent, measured elevation gains due to the net elevation increase immediately following the storm, the effect was minimal at higher elevations plots that received the greatest amount of sediment.

Our results are different from those of La Peyre et al. (2009), who found that elevation losses following sediment nourishment in vegetated marsh areas were minimal 1 year after application. Our results, however, are more in agreement with those reported by Curole and Dearmond (2010), who measured rapid post-construction elevation and sediment volume decreases for the restoration project studied by La Peyre et al. (2009). Over a 3-year period, and using methods similar to that in the present study to measure elevation change, Curole and Dearmond (2010) found that elevation and sediment volume decreased by 26 cm and 44%, respectively. However, they did not mention sediment thickness (only sediment volume), the amount of soil compression that occurred beneath the applied sediment layer, or specify which elevation measurements were in previously vegetated marsh areas, so a direct comparison cannot be made. Though based on information gleaned from both sources, it is likely that sediment thickness was much greater than the amount we applied. Furthermore, the project studied by Curole and Dearmond (2010) and La Peyre et al. (2009) was much larger in scale than the sediment-nourished plots in the present study, and sediment thicknesses likely varied substantially across an area consisting of both interior ponds and vegetated marsh, which could explain differences in results.

Despite having only a small effect on surface elevation, nourishment with more than 15 cm of sediment stimulated plant growth in the present study. Compared to control plots, plant production in sediment-nourished plots increased by as much as 95%, with similar percentage increases both above- and belowground. La Peyre et al. (2009) found a similar response for aboveground biomass, but not belowground biomass, which neither increased nor decreased after sediment nourishment, and was lower when compared to reference sites. However, measurements of belowground biomass and belowground production differ substantially in interpretation, especially when new sediment is added. It is, therefore, likely that while belowground production rates can increase following sediment nourishment (present study; Edwards and Mills, 2005), increased production does not necessarily translate into increased belowground biomass accumulation compared to unamended areas, as found by La Peyre et al. (2009), because belowground biomass must accumulate in sediment that contains no (or negligible) root mass initially.

Corresponding with increased plant production, sediment nourishment also stimulated belowground decomposition and reduced the rate of soil accretion; although accretion in sediment-nourished plots was strongly influenced by surface elevation differences during Hurricane Ike (see Fig. 2). Nonetheless, the belowground exponential decay constant (k) increased by 27%, regardless of sediment thickness, which may slow the rate at which organic matter accumulates within the soil matrix over time in plots that received <15 cm of sediment. However, in plots that received >15 cm of sediment, each year of enhanced belowground production will provide a net increase in belowground organic matter accumulation that will last for approximately 10 years, despite enhanced belowground decomposition (Table 1). Nyman et al. (2006) concluded that soil organic matter accumulation controls vertical accretion in coastal marshes of Louisiana. Therefore, when supplied in sufficient quantity (i.e., >15 cm), sediment nourishment will help maintain realized elevation gains and prolong the effects of sediment nourishment on marsh elevation by enhancing organic matter accumulation, thus contributing to longer-term marsh stability.

The effects of sediment nourishment on marsh ecosystem function during the present study were driven by changes in a combination of interrelated soil and porewater characteristics that resulted from sediment nourishment. Our results show that increasing the quantity of applied sediment increased the mineral matter content of the soil, soil bulk density, soil exchangeable nutrient and trace metal concentrations, and soil redox potential (Tables 3 and 4, Appendices A and B). Corresponding changes in porewater chemistry resulting from greater amounts of applied sediment, in turn, resulted in greater nutrient and trace metal concentrations available for plants, and lower porewater salinity and sulfide concentrations known to inhibit plant growth (Tables 5 and 6, Appendix B). Higher elevation and greater concentrations of available nutrients have also been shown to accelerate belowground decomposition (Neckles and Neill, 1994; Morris and Bradley, 1999).

Similar effects from sediment nourishment have been documented in salt marshes (Edwards and Proffitt, 2003; Mendelssohn and Kuhn, 2003; Slocum et al., 2005; Slocum and Mendelssohn, 2008; Schrift et al., 2008; Stagg and Mendelssohn, 2010) and to a lesser extent in brackish marshes (La Peyre et al., 2009). However, the later study found that although aboveground biomass increased after sediment addition, decadal-scale trajectories suggested that growth stimulation decreases over time. Therefore, it is currently unclear how long enhanced function will prolong marsh sustainability, although it is reasonable to assume that benefits from sediment addition will have a finite duration.

5. Conclusions

Current, and proposed, restoration projects in coastal Louisiana using hydraulically conveyed sediment target both brackish and salt marsh areas that are vegetated, but in a state of deterioration, as well as those that have previously deteriorated to open water (CPRA, 2012). Furthermore, most sediment addition projects involve large landscape areas that contain substantial variation in surface elevation, with sediment additions occurring over both shallow ponds and vegetated areas, which can result in sizable ranges in sediment thicknesses; as much as 10–50 cm difference, or more. Overall, results from the present study suggest that a proactive approach – wetland rehabilitation by adding sediment to deteriorating but still vegetated wetlands – should be considered a priority by State and Federal coastal protection and restoration officials. Although only small elevation gains were realized with greater sediment additions, increased plant productivity resulting from nourishment with as little as 15–20 cm of sediment will help to maintain realized elevation gains and prolong the effects of sediment nourishment on marsh surface elevation. Obviously, project construction costs are lower when less sediment is required to restore or enhance marsh function. However, nourishment with <10 cm of sediment has the potential to decrease absolute soil surface elevation, and is not considered effective for increasing soil surface elevation or enhancing the function of vegetated brackish marshes with organic soils such as this one. Therefore, this research suggests that a minimum sediment-application threshold of 10–15 cm exists for this brackish marsh and possibly others, below which elevation is lost, and above which elevation is gained and ecosystem function is enhanced.

Acknowledgements

Funding for this research was provided by The Cooperative Institute for Coastal and Estuarine Environmental Technology (CICEET), a partnership of the University of New Hampshire and the National Oceanic and Atmospheric Administration (NOAA), and the Louisiana Sea Grant Program, a part of the National Sea Grant Program maintained by NOAA. We thank the National Audubon Society for allowing us access to the site and providing logistical support, especially Timmy Vincent for providing lodging, meals, and airboat transportation. We also thank Yvonne Allen for providing GIS-based estimates of land loss. Joe Bastian, Chester Keating, Martin Maxwell, Pete Lennaker, Andrew Hoefer, Lauren Alleman, Cary Perry, Paul Guillery, and Kate Lingoni provided invaluable assistance in the field. Thoughtful comments by Joe Baustian and 2 anonymous reviewers improved upon an earlier version of this manuscript.

Appendix A. Sediment slurry physico-chemistry. Values are presented as the mean (n = 10) ± 1 SE, except for percent sand, silt, and clay (n = 5).

Constituent	Concentration
Organic matter (%)[a]	6.9 ± 0.7
Sand (%)[a]	11.3 ± 2.2
Silt (%)[a]	42.4 ± 2.1
Clay (%)[a]	39.6 ± 2.8
pH (1:1 water)	7.7 ± 0.1
Ammonium-N (mg kg^{-1})	114.57 ± 20.09
Calcium (mg kg^{-1})	284.32 ± 33.87
Copper (mg kg^{-1})	0.76 ± 0.08
Iron (mg kg^{-1})	178.00 ± 12.48
Magnesium (mg kg^{-1})	200.2 ± 9.66
Manganese (mg kg^{-1})	30.52 ± 4.57
Phosphorus (mg kg^{-1})	7.41 ± 0.70
Potassium (mg kg^{-1})	73.46 ± 4.44
Sodium (mg kg^{-1})	426.33 ± 26.44
Sulfur (mg kg^{-1})	16.94 ± 1.30
Zinc (mg kg^{-1})	0.46 ± 0.05

[a] Presented as percent of total soil (organic matter + sand + silt + clay = 100%).

Appendix B. Soil (0–15 cm) and porewater (10–20 cm) chemistry. Time-averaged treatment means (± 1 SE) are presented as supplemental data for factor analysis results.

Constituent	Treatment			
	Control (0 cm)	Low (<10 cm)	Med (10–15 cm)	High (>15 cm)
Soil				
Ammonium-N ($\mu g\,cm^{-3}$)	0.72 ± 0.16	0.91 ± 0.29	0.60 ± 0.11	0.71 ± 0.13
Calcium ($\mu g\,cm^{-3}$)	27.24 ± 1.53	45.82 ± 8.82	63.44 ± 7.00	99.76 ± 12.95
Copper ($\mu g\,cm^{-3}$)	0.11 ± 0.01	0.32 ± 0.03	0.39 ± 0.02	0.47 ± 0.02
Iron ($\mu g\,cm^{-3}$)	42.62 ± 2.68	75.09 ± 8.69	75.84 ± 7.70	75.50 ± 9.10
Magnesium ($\mu g\,cm^{-3}$)	36.34 ± 2.08	46.51 ± 3.06	56.12 ± 2.07	64.51 ± 0.17
Manganese ($\mu g\,cm^{-3}$)	0.86 ± 0.11	3.96 ± 0.86	5.80 ± 0.55	9.18 ± 2.22
Nickel ($\mu g\,cm^{-3}$)	0.04 ± 0.00	0.17 ± 0.06	0.15 ± 0.01	0.21 ± 0.03
pH (1:1 water)	5.09 ± 0.08	5.26 ± 0.20	5.69 ± 0.21	6.19 ± 0.28
Phosphorus ($\mu g\,cm^{-3}$)	0.25 ± 0.02	0.54 ± 0.10	0.82 ± 0.06	0.95 ± 0.05
Potassium ($\mu g\,cm^{-3}$)	12.76 ± 0.81	18.48 ± 1.65	22.85 ± 0.94	27.75 ± 0.41
Red-ox @ 15 cm (mV)	182.75 ± 23.97	252.82 ± 20.49	232.77 ± 6.29	244.77 ± 2.50
Sodium ($\mu g\,cm^{-3}$)	88.51 ± 6.99	93.32 ± 9.73	104.98 ± 5.36	115.80 ± 3.57
Sulfur ($\mu g\,cm^{-3}$)	18.08 ± 1.66	23.29 ± 1.06	25.60 ± 0.89	28.92 ± 1.89
Zinc ($\mu g\,cm^{-3}$)	0.32 ± 0.03	0.74 ± 0.10	0.88 ± 0.06	1.25 ± 0.14
Porewater				
Aluminum ($mg\,L^{-1}$)	0.90 ± 0.46	0.13 ± 0.01	0.73 ± 0.39	0.12 ± 0.01
Ammonium-N ($mg\,L^{-1}$)	0.86 ± 0.38	0.43 ± 0.18	0.42 ± 0.07	0.36 ± 0.12
Boron ($mg\,L^{-1}$)	0.82 ± 0.03	0.67 ± 0.04	0.65 ± 0.01	0.71 ± 0.02
Calcium ($mg\,L^{-1}$)	471.51 ± 2.59	377.52 ± 25.00	407.07 ± 16.45	384.52 ± 30.61
Copper ($mg\,L^{-1}$)	0.01 ± 0.00	0.02 ± 0.00	0.02 ± 0.00	0.02 ± 0.00
Iron ($mg\,L^{-1}$)	1.35 ± 0.48	16.14 ± 7.04	19.32 ± 5.16	23.21 ± 6.68
Magnesium ($mg\,L^{-1}$)	258.39 ± 14.84	203.60 ± 10.03	208.67 ± 7.66	223.47 ± 7.75
Manganese ($mg\,L^{-1}$)	0.99 ± 0.06	2.66 ± 0.26	3.51 ± 0.36	6.02 ± 0.25
Molybdenum ($mg\,L^{-1}$)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Nickel ($mg\,L^{-1}$)	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00	0.02 ± 0.00
pH	6.10 ± 0.06	6.04 ± 0.12	6.20 ± 0.05	6.25 ± 0.15
Phosphorus ($mg\,L^{-1}$)	0.20 ± 0.04	0.28 ± 0.03	0.48 ± 0.09	1.10 ± 0.27
Potassium ($mg\,L^{-1}$)	69.09 ± 0.70	54.61 ± 2.04	57.11 ± 1.14	57.05 ± 0.84
Salinity ($g\,L^{-1}$)	7.83 ± 0.33	6.13 ± 0.27	6.16 ± 0.15	6.55 ± 0.23
Sodium ($mg\,L^{-1}$)	2099.44 ± 39.92	1697.55 ± 66.90	1685.73 ± 44.78	1737.49 ± 30.71
Sulfur ($mg\,L^{-1}$)	574.03 ± 109.18	288.87 ± 78.86	172.23 ± 21.42	122.92 ± 26.19
Sulfide ($mg\,L^{-1}$)	36.76 ± 2.08	13.98 ± 4.29	5.44 ± 1.18	3.28 ± 0.76
Zinc ($mg\,L^{-1}$)	0.03 ± 0.00	0.06 ± 0.02	0.08 ± 0.01	0.09 ± 0.01

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