Salt marsh restoration with sediment-slurry application: Effects on benthic macroinvertebrates and associated soil–plant variables

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A B S T R A C T

We analyzed the effects of various levels of sediment-slurry addition on the restoration of the macroinvertebrate community and its related habitat (i.e., sediment and vegetation) 7 years after application to a subsided Louisiana salt marsh affected by sudden marsh dieback. Moderate sediment additions restored macroinvertebrate species richness, diversity, density, and total biomass to levels equivalent to those in reference marshes, although individual species and taxa had variable recovery depending on treatment-level. Total aboveground plant biomass and live Spartina alterniflora biomass, stem density and height were equivalent to those in reference marshes. In contrast, total belowground biomass had not yet reached equivalency with reference marshes. Although moderate sediment application created conditions that were ecologically equivalent with reference marshes for most macroinvertebrate and plant variables, degraded areas that received high sediment addition had impaired recovery across all metrics, even 7 years after sediment application. Thus, when sediment-slurries are applied to proper elevations, the macroinvertebrate community, as well as aboveground marsh vegetation, can recover to reference conditions. However, too much sediment impairs recovery. Consequently, greater consideration must be given to establishing suitable post-construction marsh elevations to insure successful ecosystem restoration.

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1. Introduction

Salt marshes, distributed worldwide in estuaries and coastal areas in middle and high latitudes, are one of the most productive ecosystems in the world and have garnered much attention in the last 50 years (Mitsch and Gosselink, 2000; Lefevre et al., 2003; Silliman et al., 2009). Unfortunately, these coastal marshes are being lost at high rates across the globe (Mitsch, 2005). For example, Louisiana, one of the most wetland-rich regions in the world, is suffering coastal wetland loss rates of up to 69.7 km\textsuperscript{2} yr\textsuperscript{-1} (Barras et al., 2008). Multiple factors have contributed to such high land loss, including large-scale coastal development projects, land reclamation, sea level rise and geological subsidence (McFalls et al., 2010; Day et al., 2011a,b; Lee et al., 2011). In addition, extreme weather events, such as the sever droughts that have occurred during the last decade in the southeastern United States (Alber et al., 2008), are believed responsible for salt marsh dieback (McKee et al., 2004) and subsequent subsidence and marsh loss. Excessive inundation in many dieback marshes in coastal Louisiana prevents plant reestablishment (McKee et al., 2004; Slocum et al., 2005; Stagg and Mendelssohn, 2010).

Increasing marsh elevation with sediments is often key in reversing the processes of coastal wetland loss caused by sea level rise, geological subsidence and also drought-induced subsidence (Day et al., 2011a,b; Stralberg et al., 2011). However, man-made constructions such as dams and levees have disrupted the sediment supply. As a result, coastal and deltaic wetlands, historically sustained by river sediment input, are now facing decline, as exemplified by the Mississippi (Blum and Roberts, 2009; Tweel and Turner, 2012) and Yangtze Deltas (Yang et al., 2005a,b, 2006a). Consequently, the artificial enrichment of sediment supply appears essential as a means of maintaining marsh elevation. One effective method is sediment-slurry addition (Mendelssohn and Kuhn, 2003). Previous studies have shown this technique’s effectiveness in the restoration of vegetation and soil conditions (Mendelssohn and Kuhn, 2003; Slocum et al., 2005; Schriff et al., 2008) and related ecosystem functions (Stagg and Mendelssohn, 2010, 2011). Based on the known relationship among vegetation, abiotic environmental condition, and invertebrate species composition and abundance (Levin and Talley, 2000; Bolam et al., 2004; Yuan et al., 2005), it

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is reasonable to deduce that such restoration actions must have effects on the invertebrate populations. Previous studies examining invertebrate community structure in created marshes have found restoration of faunal density can take between 3 and 25 years depending on species (Craft and Sacco, 2003). Factors influencing recovery time include soil bulk density and organic content, salinity, vegetation type, and reproductive strategy of individual macroinvertebrate species among others (Levin et al., 2001; Craft and Sacco, 2003; Moseman et al., 2004). However, the effects of sediment-slurry addition on the restoration of the macroinvertebrate community of subsiding and degraded marshes has received little attention.

Macroinvertebrates play a key role in the functioning of salt marsh ecosystems by bridging the gap between primary producers and higher trophic levels (Levin and Talley, 2000; Levin et al., 2001; Moseman et al., 2004). Specifically, macroinvertebrates are often a food source for birds, fish, and other taxa (Sacco et al., 1994; Levin et al., 1996; Cardoso et al., 2008), and can contribute to the decomposition of organic matter in the soil (Levin et al., 2001). Macroinvertebrates are also important indicators for wetland condition (Weihoef er, 2011). Understanding how the macroinvertebrate community responds to various levels of sediment-slurry addition is essential in determining how effective sediment-slurry additions are at restoring ecosystem structure and function.

The goal of this study was to determine the effects caused by various levels of sediment-slurry addition on the salt marsh macroinvertebrate community and related habitat. Our previous investigations documented that the moderate levels of sediment addition generated the most beneficial and long lasting effects to vegetation and soils at various restoration sites in the Mississippi Delta (Slocum et al., 2005; Stagg and Mendelsson, 2010, 2011). Macroinvertebrate structure also naturally varies along elevational gradients in salt marshes (Netto and Lana, 1997; Salgado et al., 2007). We hypothesized that intermediate levels of sediment-slurry addition to degraded salt marshes would yield the most beneficial effects to the macroinvertebrates. We assessed the macroinvertebrate community in a restored salt marsh with a distinct elevational gradient 7 years after sediment addition. We also measured elevation, vegetation and soil responses to provide a more comprehensive assessment of ecosystem restoration and to determine how these factors contribute to macroinvertebrate recovery. We compared ecological responses in the restored marshes to nearby marshes unaffected by the dieback as well as to dieback marshes that received no sediment addition.

2. Methods

2.1. Study area

The study area is located approximately 8 km south-southwest of Leeville, Louisiana, USA (Fig. 1), at the same location used in previous studies by Schrift et al. (2008) and Stagg and Mendelsson (2010). This area is characterized by Spartina alterniflora Loisel dominated salt marshes, high rates of relative sea level rise (1.11 cm yr⁻¹, 1947–1986; Penland and Ramsey, 1990), and consequently, high rates of wetland loss (50.9 km² yr⁻¹: Barras et al., 2008). Average tidal range is approximately 0.3 m and sea levels tend to be lower in winter and higher in late summer (Turner, 1991). Mean monthly temperature for Leeville, LA, in April is 20 °C. Mean high and low temperatures are 26 °C and 14 °C, respectively. Mean monthly precipitation is 134 mm.

An intense drought affected the study area from 1999 until the fall of 2000. In conjunction with the drought, large expanses of marsh within the study area were severely affected by sudden marsh dieback (McKee et al., 2004). Much of the denuded substrate became mudflat, subsided, and remained devoid of vegetation, while other adjacent areas were unaffected.

In an effort to reestablish vegetation within the areas affected by sudden marsh dieback, the Louisiana Department of Natural Resources used hydraulically dredged sediment amendments to raise marsh surface elevation to a level conducive for vegetation regrowth. The sediment slurry was pumped into five separate 1.5 ha cells, and within each cell a range of elevations resulted. Following sediment application, the cells were hydraulically connected with culverts, and the levees broken to provide tidal exchange. The ring levees that surround the sediment cells were constructed by digging material from within the levee perimeter and placing it on top of the marsh. The hydrologic condition of the reference plots was not affected by the levees since they were outside the “ring” where normal tidal exchange occurs. Fig. 1 shows the borrow pit running along the interior perimeter of the levee system. Locations of reference plots were selected based on two criteria: (1) prior to the dieback event, reference plot locations relative to adjacent sediment cells were within a contiguous marsh (identified using pre-dieback aerial photography) and (2) reference plot locations were undisturbed areas at least 10 m from the outer edge of the levee (identified using aerial photography taken during construction). Criteria (1) allowed us to use a randomized block design. For additional information on the study site and dredging activities see Schrift et al. (2008).

2.2. Experimental design

For this research, we utilized 4 of the 5 sediment cells (A, B, D, E) as replicate blocks (Fig. 1). Adjacent to each cell, we established 2 reference conditions: (1) healthy reference areas unaffected by the sudden marsh dieback event and dominated by S. alterniflora (Reference) and (2) degraded reference areas that experienced dieback in 2000, received no sediment addition, and remained devoid of vegetation (Degraded). Within each cell, the irregular distribution of sediment created 4 distinct elevation zones corresponding to 4 different sediment addition levels: High, Medium, Medium-Vegetated and Low. In total, we had 6 treatments all of which were represented within each block of the randomized block design (four blocks, n = 4). The relative elevation at each of these treatment locations was measured with a class 1 laser level (Sokkia LP30) and tied in to a previously surveyed permanent benchmark (Schrift et al., 2008). The surface elevation (mean ± SE) of the ambient healthy reference sites (Reference, abbreviated to Ref) was 22.4 ± 0.7 cm-NAVD88. The surface elevation of the sediment amended sites were +39.4 ± 1.0 cm (High), +36.8 ± 0.5 cm (Medium, abbreviated to Med), +34.1 ± 1.2 cm (Medium-Vegetated, abbreviated to Veg), and +30.7 ± 0.7 cm – NAVD88 (Low). The Medium-Vegetated sites were initially distinguished from the Medium sites because they contained live vegetation immediately after sediment addition (see Schrift et al., 2008). The surface elevation of the degraded reference sites (Degraded, abbreviated to Deg) was 6.0 ± 3.7 cm – NAVD88.

2.3. Vegetation measurements

All sampling took place in April 2010. At each sampling location, two replicate 0.25 m² plots were randomly located. Within each plot, all aboveground plant biomass was clipped to the soil surface, bagged, and transported back to the laboratory for processing. The total (live + dead) biomass of each plot was determined after the samples were washed, dried at 60 °C, and weighed. All the samples were dried for more than 48 h until a constant weight was attained. Accurate measurements of stem density (i.e., the number
of individuals per plot) and height were difficult to make when plots contained Salicornia virginica and pneumatophores of Avicennia germinals, so these data were collected for only the S. alterniflora plots. Mean stem height was estimated by measuring ten individual shoots per plot. If the shoot number was less than ten, then all shoots were measured.

Importance values (IV) were used to determine the dominance of each plant species (Mueller-Dombois and Ellenberg, 1974). Importance values were calculated as the summation of the relative frequency (%) and relative biomass (%) of each species identified. We used relative biomass instead of relative cover, and relative density was not included in the calculation.

Within each clipped plot, total (live + dead) belowground biomass to a depth of 30 cm was estimated by taking a 15 cm diameter core and rinsing it over a 500 µm mesh sieve. The remaining organic material was then dried at 60 °C and weighed to the nearest 0.01 g.

2.4. Benthic macroinvertebrate sampling

We collected all of the benthic macro-epifauna visible on the clipped vegetation and the soil surface within each clipped plot immediately following the aboveground vegetation harvest. Macro-infauna (>500 µm) were picked from the vegetation belowground biomass cores after sieving through a 500 µm mesh sieve (Vinagre et al., 2008). All macroinvertebrates were preserved in 10% formalin, sorted, enumerated and identified to species (some rare species were sorted to order, family, or genus and assigned a nominal species designation). After identification and counting, all samples were dried at 60 °C for more than 48 h to a constant weight and weighed to the nearest 0.01 g.

2.5. Soil measurements

Within each clipped plot, a 30 cm deep (5 cm diameter) core was extracted with a Russian Peat Borer (Aquatic Research Instruments). Each core was cut into six 5-cm segments, which were placed into individual plastic bags, and kept on ice until returning to the laboratory. In the laboratory, the segments were weighed to the nearest 0.01 g, then dried at 60 °C and re-weighed to the nearest 0.01 g. All the samples were dried more than 48 h until a constant weight was achieved. Based on these weights, water content (%) and bulk density were calculated.

The reduction–oxidation potential (Eh) of the soil was measured in situ at 3 locations adjacent to the clip-plots (Mendelssohn and Kuhn, 2003). At each location, Eh was recorded at six depths below the surface: 2.5 cm, 7.5 cm, 12.5 cm, 17.5 cm, 22.5 cm, 27.5 cm, which corresponded to the midpoint of the 5-cm soil core segments.
2.6. Data analysis

We used the PROC MIXED procedure in SAS (version 9.1.3, SAS Institute) to analyze the vegetation, soil, and benthic macroinvertebrate data. Analysis of variance (ANOVA) was used to determine treatment-effects for plant and macroinvertebrate variables and repeated measures ANOVA to determine treatment-effects for soil variables with depth. We used Fisher’s protected LSDs to identify significant differences between treatment-means. When necessary data were natural log or power transformed prior to analysis to meet ANOVA assumptions of normality and homogeneity of variance (Mendelssohn and Kuhn, 2003).

Macroinvertebrate community diversity was estimated with the Shannon–Weiner diversity index ($H'$; log$_e$) and with species richness (number of species). The relationships between benthic macroinvertebrate community composition and environmental variables, including soil and vegetation, were tested with the procedure BVSTEP and RELATE of PRIMER (version 5.2.8; Clarke and Warwick, 2001). BVSTEP was used to select the best combination of environmental variables correlated to the community composition, and RELATE was employed to test the significance of the correlation between the matrices of the community composition and these variables (Lercari et al., 2002). All these procedures were based on the rank correlation analysis technique. To meet the requirement of these procedures, all the data except those for individual item analysis (e.g., each species, each class, etc.) were fourth root ($\times 0.14$ c is a constant used for zero and negative value adjustment) and standardization transformed. These transformations can reduce the effects of extreme values and distribution pattern of some variables on the distribution characteristics of the entire data collection (e.g., the distribution pattern of the dominant species relative to the whole community of invertebrates). Standardization transformation can also reduce the effects of different units (e.g., the units of the different environmental variables). We only did the fourth root transformation on individual item analysis because it only has one variable (e.g., one species, one class, etc.). Next, Bray–Curtis similarity indices and Euclidean distances were used to build the similarity matrices of the community composition and the selected environmental variables, respectively. The commonly used spearman rank correlation coefficient was computed based on these matrices, and the stepwise method was employed to select the environmental variables in BVSTEP. The combination that had the highest correlation with the corresponding community composition was thought to be the best selection. Then the environmental similarity matrices were rebuilt with these selections; the significance of the correlations between them and their corresponding community composition were tested ($P<0.05$ was considered statistically significant).

The macroinvertebrate community was not only analyzed at the species and class levels, but also at the level of functional group. The
macroinvertebrates were categorized into three functional groups according to their living habit: epifauna, infauna and semi-infauna. The semi-infauna has the characteristics of epifauna and partial infauna, e.g., Uca longisignalis, that live on the sediment surface but can also burrow into the sediment.

3. Results

3.1. Benthic macroinvertebrates

Within the benthic macroinvertebrate community we distinguished 161 specimens from 3 phyla, 5 classes, 10 families, and 10 species. The dominant species, by number, were the polychaete Notomastus sp. (24.06%) and the gastropod Littoraria irritata (20.78%), while the bivalve Geukensia demissa (46.74%) and the gastropod L. irritata (45.19%) accounted for the highest portion of the total biomass. However, the dominant species, class and functional group varied with treatment (Table 1). In the Reference sites, the epifauna took the leading place. Gastropoda dominated in both abundance and biomass, the dominant species was L. irritata. In the Low and Med-Vegetated sites, the infauna, especially Polychaeta, had the highest abundance. The dominant species was Notomastus sp., but biomass was dominated by epifauna – Gastropoda in the Low sites with L. irritata dominating, and the semi-infaunal Bivalvia in the Med-Vegetated sites with G. demissa dominating. In the Medium sites, the infaunal Insecta and Polychaeta had the highest abundance and the insect Pyralidae sp. appeared to be dominant, while the epifaunal Gastropoda had the highest biomass and L. irritata took the highest portion. In the High sites only one species of semi-infaunal Crustacea was found, and in the Degraded sites only one species of infaunal Insecta was recorded.

Macroinvertebrate density, biomass, and species richness were significantly affected by sediment-slurry treatment. Macroinvertebrate densities were highest in the Reference sites and with moderate sediment addition (Low, Med-Vegetated and Medium sites) and were significantly greater than in the Degraded and High sites (Fig. 2a). Macroinvertebrate biomass also tended to be lowest in the High and Degraded sites, although there was high variation within treatment-levels (Fig. 2b). For example, in the Med-Vegetated sites, one plot had a large cluster of G. demissa with a biomass of nearly 800 g/m², which greatly influenced variability. Macroinvertebrate species richness also differed significantly with sediment-slurry application (Fig. 2c). Species richness was highest in the Reference and moderate sediment-addition sites, which were significantly greater than in the Degraded and High sites (Fig. 2c). Similar to macroinvertebrate density, species richness was equally low and did not differ between the degraded sites and the High treatment. There was no treatment effect on the Shannon–Weiner diversity index (0.18 ± 0.07).

However, the effects of the treatments on the macroinvertebrate also varied with species, class and functional group (Table 1). Only the density of Neritina usnea, L. irritata and Pyralidae sp. and the biomass of the last two were significantly affected by the treatment. The density of N. usnea appeared to be significantly higher in Low sites than in all the other. L. irritata in the Low and Reference sites had significantly higher density and biomass than in the Degraded, High and Med-Vegetated sites, while the density and biomass of Pyralidae sp. in the Medium sites happened to be significantly higher than in the others. Relative to the composition of the class and functional groups, only the density and biomass of the Gastropoda and epifauna were significantly affected by the treatment, and in the Low and Reference sites were significantly higher than in the Degraded and High sites.

Fig. 2. Macroinvertebrate density (a), biomass (b), and species richness (c) in the sediment-slurry amended areas and reference sites (mean ± SE).
3.2. Vegetation

A total of four plant species were distinguished in the clipped plots, with *S. alterniflora* being the dominant species at each site (Table 2). The dominance of *S. alterniflora* in Low, Medium, and Med-Vegetated sites was similar to that in Reference sites, and higher than the High sites.

Average total aboveground biomass in the Low, Med-Vegetated, and Medium sites was not significantly different from the Reference sites (Fig. 3). Total aboveground biomass in the High sites was significantly lower than the reference and moderate sediment-addition marshes. We found no aboveground biomass in the Degraded sites.

Average total belowground biomass was significantly higher in the Reference sites compared to all other treatments (Fig. 3). Among the sediment-amended sites, belowground biomass in the Low, Medium and Medium Vegetated treatments was significantly greater than the High treatment. The belowground biomass in the Degraded sites was composed of decomposed dead biomass only—no live belowground biomass was present in any of the plots within this treatment.

*S. alterniflora* stem densities, and live and dead aboveground biomass, in the Low, Med-Vegetated, and Medium sites were equal to or greater than the Reference sites, while the Degraded sites had no vegetation (Table 3). The height of live *S. alterniflora* in the Reference treatment was equal to that in the Low, Med-Vegetated, and Medium sites, but higher than in the High sites (Table 3). Again, the Degraded sites were devoid of vegetation and therefore stem densities, heights and aboveground biomass were all zero.

3.3. Soil

In general, soil bulk density in the Reference and Degraded sites was lower than in the sediment addition sites (Fig. 4a). However, bulk density in the top 5 cm of the Low and Med-Vegetated sites was not significantly different from the Reference sites, and the Medium sites were intermediate to the High sites. Trends in soil water content between treatments were similar to those for bulk density (Fig. 4b). Soil Redox potential tended to decrease with depth in all treatments (Fig. 4c). The High sites had the least reduced soils, the Degraded sites had the most reduced soils, and the Low, Medium, Medium-Vegetated, and Reference sites had intermediately reduced soils.

3.4. Relationships between macroinvertebrate community composition and the environmental variables

The best combination of environmental variables that had significant correlations with macroinvertebrate community composition appeared to vary with species, class and functional group; elevation, however, appeared in each combination (Table 4). At the species level, the density of the macroinvertebrate community was significantly correlated with the combination of elevation, bulk density of specific sediment layers, *S. alterniflora* dead standing biomass and density. The biomass was significantly correlated with the combination of elevation, bulk density of specific sediment layers, *S. alterniflora* dead standing biomass. However, different species had different environmental variable combinations corresponding to it. For example, the combination of bulk density at 5–30 cm soil depth and elevation had a significant correlation with density of *L. irrorata*. This combination together with *S. alterniflora* dead standing biomass was significantly correlated with biomass of *L. irrorata*, while the combination of Eh at 20–25 cm soil depth, bulk density at 15–30 cm soil depth and elevation appeared to have significant correlations with density and biomass of the *G. demissa*. At the class level, there was a combination which had a significant correlation to biomass of the classes, and none to density, while at the functional group level, no combination of environmental variables had significant correlations to the density and biomass of the functional groups. However, there were still some combinations of environmental variables that had significant correlations to some classes and functional groups.

### Table 2

<table>
<thead>
<tr>
<th>Scientific name</th>
<th>Region</th>
<th>Ref</th>
<th>Low</th>
<th>Med</th>
<th>Low</th>
<th>Veg</th>
<th>High</th>
<th>Deg</th>
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<td>195.08</td>
<td>157.36</td>
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<td>67.26</td>
<td>92.52</td>
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<td>55.78</td>
<td>83.04</td>
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<td>0.00</td>
<td>12.66</td>
<td>26.02</td>
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### Table 3

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<tr>
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<th>Stem density (stems/m²)</th>
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<td>Deg</td>
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<td>0 ± 0²</td>
<td>0 ± 0²</td>
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<tr>
<td>Ref</td>
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<td>838 ± 87²</td>
<td>214 ± 17²</td>
</tr>
<tr>
<td>Low</td>
<td>317 ± 29²</td>
<td>724 ± 105²</td>
<td>228 ± 18²</td>
</tr>
<tr>
<td>Veg</td>
<td>382 ± 60²</td>
<td>831 ± 140²</td>
<td>265 ± 28²</td>
</tr>
<tr>
<td>Med</td>
<td>476 ± 96²</td>
<td>835 ± 108²</td>
<td>322 ± 35²</td>
</tr>
<tr>
<td>High</td>
<td>157 ± 45²</td>
<td>148 ± 46²</td>
<td>206 ± 53²</td>
</tr>
</tbody>
</table>

Fig. 3. Average total aboveground and belowground biomass in the sediment-lurry amended areas and reference sites. Treatments sharing letters within the shaded area are not significantly different belowground (P > 0.05), and treatments sharing letters in the open bars are not significantly different aboveground (P > 0.05).
but the combination to each of them was different. Among those combinations of the environmental variables, which had significant correlations to the community compositions, the most frequently selected variable was elevation, dead standing biomass of *S. alterniflora*, bulk density and Eh of the specific layers of the sediment. Although correlations do not suggest cause and effect, they suggest the importance of selected environmental variables to macroinvertebrate community composition.

4. Discussion

4.1. Effects of sediment-slurry addition

The primary goal of sediment-slurry addition is to increase marsh surface elevation to the point where marsh structure and function is restored and sustained. In general, marsh surface elevation, and resulting hydrology, greatly influences salt marsh ecosystem structure and function, including vegetation composition and primary productivity (Anastasiou and Brooks, 2003; Edwards and Mills, 2005; Hughes et al., 2009), environmental condition (Pennings and Callaway, 1992; Fariña et al., 2009; Hickey and Bruce, 2010), and benthic fauna (Levin and Talley, 2000; Rodrigues et al., 2006; Quan et al., 2008). Our results support the proposition that elevation differences caused by sediment-slurry addition have significant effects on the macroinvertebrate community and the associated vegetation and soils. This conclusion was supported by the frequent inclusion of elevation in the combinations of the environmental variables related to macroinvertebrate composition. However, the specific effects of sediment-slurry addition vary depending on the extent to which surface elevation is modified as well as the particular response variable of interest.

Moderate sediment-slurry addition restored the macroinvertebrate community (species richness and density) to that of undisturbed reference marshes, while high sediment addition

Table 4

Correlations between selected environmental variables and benthic macroinvertebrate density and biomass. Significance tests (in parentheses) were performed using the RELATE procedure. When significant (*P* < 0.05; highlighted), selections correspond with most influential environmental variables.

<table>
<thead>
<tr>
<th>Category</th>
<th>Density Correlation</th>
<th>Selections</th>
<th>Biomass Correlation</th>
<th>Selections</th>
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<td>Species</td>
<td></td>
<td></td>
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<td>Neritina usnea</td>
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<td>2, 5, 22, 25–28</td>
<td>0.062 (0.004)</td>
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<td>Littorina irrorata</td>
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<td>14, 25–28</td>
<td>0.255 (0.024)</td>
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<td>13, 23–28</td>
<td>0.158 (0.16)</td>
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<td>Culicidae sp.</td>
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<td>16–18, 25–28</td>
<td>0.339 (0.34)</td>
<td>16–18, 25–28</td>
</tr>
<tr>
<td>Namalycastis abiona</td>
<td>0.133 (0.072)</td>
<td>14, 25–28</td>
<td>0.135 (0.06)</td>
<td>14, 25–28</td>
</tr>
<tr>
<td>Notomastus sp.</td>
<td>−0.057 (0.69)</td>
<td>2, 5, 21, 25–28</td>
<td>−0.038 (0.61)</td>
<td>2, 5, 21, 25–28</td>
</tr>
<tr>
<td>Class</td>
<td>0.006 (0.38)</td>
<td>5, 17, 22, 25–28</td>
<td>0.046 (0.05)</td>
<td>5, 23, 25–28</td>
</tr>
<tr>
<td>Gastropoda</td>
<td>0.142 (0.077)</td>
<td>5, 22, 25–28</td>
<td>0.141 (0.005)</td>
<td>5, 22, 23, 25–28</td>
</tr>
<tr>
<td>Bivalvia</td>
<td>0.275 (0.012)</td>
<td>14, 25–28</td>
<td>0.255 (0.024)</td>
<td>14, 25–28</td>
</tr>
<tr>
<td>Crustacea</td>
<td>0.136 (0.14)</td>
<td>13, 23–28</td>
<td>0.158 (0.14)</td>
<td>13, 23–28</td>
</tr>
<tr>
<td>Insecta</td>
<td>0.105 (0.51)</td>
<td>2, 16–19, 25–28</td>
<td>0.105 (0.55)</td>
<td>2, 16–19, 25–28</td>
</tr>
<tr>
<td>Polyxena</td>
<td>0.069 (0.055)</td>
<td>5, 14, 22, 25–28</td>
<td>0.105 (0.017)</td>
<td>5, 14, 22, 25–28</td>
</tr>
<tr>
<td>Functional group</td>
<td>−0.034 (0.25)</td>
<td>22, 23, 25–28</td>
<td>−0.006 (0.071)</td>
<td>5, 22, 23, 25–28</td>
</tr>
<tr>
<td>Epiplana</td>
<td>0.169 (0.003)</td>
<td>5, 22, 23, 25–28</td>
<td>0.141 (0.007)</td>
<td>5, 22, 23, 25–28</td>
</tr>
<tr>
<td>Semi-infauna</td>
<td>0.158 (0.21)</td>
<td>14, 23, 25–28</td>
<td>0.136 (0.14)</td>
<td>14, 24–28</td>
</tr>
<tr>
<td>Infauna</td>
<td>0.062 (0.08)</td>
<td>5, 14, 22, 25–28</td>
<td>0.097 (0.023)</td>
<td>5, 14, 22, 25–28</td>
</tr>
</tbody>
</table>

Note: Variables: 1 Spartina living standing density; 2 Spartina dead standing density; 3 Spartina total standing density; 4 Spartina living standing biomass; 5 Spartina dead standing biomass; 6 Total aboveground plant biomass; 7 Total belowground plant biomass; 8 Spartina living standing height; 9 Spartina dead standing height; 10 Eh of 0–5 cm soil; 11 Eh of 5–10 cm soil; 12 Eh of 10–15 cm soil; 13 Eh of 15–20 cm soil; 14 Eh of 20–25 cm soil; 15 Eh of 25–30 cm soil; 16 Water content of 0–5 cm soil; 17 Water content of 5–10 cm soil; 18 Water content of 10–15 cm soil; 19 Water content of 15–20 cm soil; 20 Water content of 20–25 cm soil; 21 Water content of 25–30 cm soil; 22 Bulk density of 0–5 cm soil; 23 Bulk density of 5–10 cm soil; 24 Bulk density of 10–15 cm soil; 25 Bulk density of 15–20 cm soil; 26 Bulk density of 20–25 cm soil; 27 Bulk density of 25–30 cm soil; 28 Elevation.
impaired recovery. In fact, adding too much sediment yielded similar results relative to macroinvertebrate restoration as adding no sediment for most response variables. Species richness, biomass and density, although not species composition, were just as depressed in the High sediment addition treatment as in the Degraded marshes, which received no sediment. However, the mechanisms responsible for this were quite different. For the high sediment addition treatment, the infrequent inundation of these plots created relatively dry soil conditions that likely impaired macroinvertebrate recovery. In contrast, the degraded marshes were almost constantly inundated due to their much lower elevation after dieback and subsidence (Stagg and Mendelsohn, 2010, 2011). Hence, the ability to successfully restore the macroinvertebrate community is dependent on proper sediment application. Furthermore, given the variability in effects of sediment slurry addition on the species, class and functional groups, the optimum amount of sediment addition will be different with different restoration targets. In this study, if we take the entire macroinvertebrate community as the target for restoration, then the sediment slurry addition (Low, Med-Vegetated and Medium sites) meet the requirement, but if we take the epifauna, Gastropoda and L. irrata as the targets, the sediment addition amount of the Low sites would be the best choice.

In some respects, the effects of sediment-slurry addition on the macroinvertebrate community in the present study were similar to their response to natural elevation gains, although effects can differ depending on the particular response variable. Levin et al. (1997) reported a general trend of decreasing polychaete importance and increasing oligochaete and insect importance from the low to high intertidal zone in the Tijuana River Estuary. Our results show a similar trend for polychaete abundance, but we found no oligochaetes larger than 500 μm in the soil. Quan et al. (2008) also determined that elevation influenced the benthic macrofauna in the marsh tidal flats of the Yangtze Estuary and Hangzhou Bay. Macraunal biomass consistently varied with elevation: high > middle > low. We also found a similar response to elevation, but the trend was not as consistent. Faunal biomass was low at the lowest and at highest sites, but variable at intermediate elevations. In addition to marsh surface elevation, vegetation and soils can also influence the extent of macroinvertebrate recovery from sediment-slurry restoration. However, a key factor controlling the difference in environmental and biotic responses to a natural elevational gradient and one created from sediment slurry addition is their respective rates of formation. In natural salt marshes, an elevational gradient often takes time to develop, and once developed, other environmental and biotic conditions associated with the gradient follow (Davy, 2000; Seybold et al., 2007). In contrast, the elevational gradient in the sediment slurry restored salt marshes formed immediately after sediment application. As a result, the development of physical and chemical characteristics of the sediment and associated biota lagged behind the formation of the elevational gradient, which still needed time to develop (Slocum et al., 2005; Schrift et al., 2008). We observed this lag time between the reference and sediment restored marshes for soil bulk density and some of the macroinvertebrate parameters.

The effects of sediment-slurry amendments on vegetation recovery have been previously documented. Specifically, sediment addition immediately increases soil surface elevation, mineral matter content and fertility of the soil, which reduces plant nutrient deficiencies, inundation stress, and phytotoxic sulfide stress over the long-term, thereby promoting plant growth (Mendelsohn and Kuhn, 2003; Slocum et al., 2005; LaPeyre et al., 2009; Stagg and Mendelsohn, 2010). Mendelsohn and Kuhn (2003) determined that aboveground biomass and cover can respond positively within 2 years following sediment addition, and Slocum et al. (2005) concluded that increased elevation was the dominant factor promoting long-term plant vigor. At the same restoration site as ours, Schrift et al. (2008) found that plant recruitment following sediment–slurry application increased when the elevation and resulting inundation regime were within the optimal range of the target vegetation. Schrift et al. (2008) also noted aboveground vegetative cover in the Low and Med-Vegetated sites was similar to the Reference sites 2 years after sediment-slurry addition. A subsequent study by Stagg and Mendelsohn (2010) also concluded that above- and belowground production was restored in the Low and Med-Vegetated sites 5 years after sediment-slurry addition. The results from the present study further support and enhance their research findings. They show that aboveground and belowground biomass do not recover at the same rate following sediment-slurry amendments. After 7 years, total aboveground biomass in Low, Med-Vegetated and Medium sites was similar to the Reference, but total belowground biomass remained significantly lower than the Reference in all the sediment treatments. Regardless, their results support our conclusion that moderate levels of sediment addition generate the most beneficial effects to the vegetation (Slocum et al., 2005; Stagg and Mendelsohn, 2010, 2011).

Previous research has also shown that soil structure is much slower to recover than vegetation (Edwards and Proffitt, 2003) because bulk density of newly deposited sediment decreases only when plants recruit to the area, consume soil nutrients, and increase the organic content of the soil via root growth (Delaune et al., 1979). Our results agree with this conclusion. After 7 years, bulk density of the added sediment decreased in the top 10 cm of soil as belowground biomass increased. The Low, Medium and Med-Vegetated sites, where the plants recruited, had bulk densities in top 5 cm of the soil equivalent to the Reference sites.

4.2. Holistic response and implications for salt marsh restoration

As the components of ecosystems have complex linkages to each other, changes in one component, resulting from an environmental change, e.g., sediment-slurry addition, may lead to cascading changes in the whole system. Thus, the effects of sediment-slurry addition must be considered from a holistic perspective. The development of the benthic macroinvertebrate community has been linked to plant cover, shoot height, above- and belowground biomass, which are driven by soil development, and soil surface elevation (Netto and Lana, 1997; Craft, 2000; Yuan et al., 2005; Stagg and Mendelsohn, 2012). We found that in addition to elevation, dead standing biomass of S. alterniflora, bulk density, and Eh at a specific soil depth had the most correlations to benthic macroinvertebrate community composition. However, correlations between the combination of these variables and the corresponding community composition were, although statistically significant, not high. It is, thus, reasonable to deduce that there are likely other factors contributing to benthic macroinvertebrate community structure, such as soil organic matter and sand content (Bolam et al., 2004; Ferraro and Cole, 2007), grain size (Barros et al., 2008), sediment heavy metal and petroleum hydrocarbon content (Inglis and Kross, 2000; Goto and Wallace, 2010), salinity (Hampel et al., 2009), food web control (Valiela et al., 2004; Seitz, 2011) and microniche effects (Stockdale et al., 2009), etc. Based on the fact that sites of different elevations had different macroinvertebrate assemblages, and that different species, classes and functional groups had different selections on the correlated environmental variables, we can infer that different biota were controlled by different factors.
Temporal variation including succession can also be an important determinant of the benthic macroinvertebrate community. Recent studies have shown that salt marshes of different growth and development stages can have different benthic faunal assemblages, regardless if they are natural (Yang et al., 2006b; Neira and Levin, 2007) or created/restored sites (Posey et al., 1997; Talley and Levin, 1999; Craft, 2000; Mosteau et al., 2004). In this study, soil bulk density and some macroinvertebrate parameters were different between reference sites and some sediment added areas, which may also be a result of differential stages in the development of the sites. Also, the benthic community can shift after anthropogenic or natural disturbances (Lardicci et al., 2001; Zajac and Whitlatch, 2001; Reiss and Kröncke, 2005; Garbutt et al., 2006; Dolbeth et al., 2007; Dauer et al., 2007). We found that macroinvertebrate biomass and density was significantly lower in Degraded sites that died back from the drought and sites that received the greatest amount of sediment. Thus, it is apparent that a number of factors, many of which are interlinked, can control the macroinvertebrate community in restored salt marshes. Experimental research that manipulates these factors, either singly or preferably in combination, is needed to more comprehensively and unambiguously determine determinants of macroinvertebrate structure and function in restored salt marshes. Regardless, this research has demonstrated that sediment-slurry application to degraded salt marshes can restore the benthic macroinvertebrate community when proper marsh surface elevation, and dependent soil and vegetation conditions, are achieved. The proper amount of sediment-slurry addition will likely vary with environment conditions, including sediment physico-chemical properties, regional hydrology, local topography, etc. Hence, the use of sediment-slurries to successfully restore the ecological structure and function of degraded salt marshes must incorporate accurate measurement of these variables so that the relationship between restored marsh elevation and local hydrology can match that of the restoration target.

5. Conclusions

Sediment-slurry addition is an effective technique for the restoration of salt marshes threatened by the effects of decreases in surface elevation. Sediment-slurry addition directly increases marsh elevation and changes sediment quality, which in turn affects soil biota. Our results show that the quantity of the sediment added to the marsh is one of the most important factors determining the recovery rate of vegetation, soil, and macroinvertebrates. There is an optimal level of sediment addition for any marsh, relative to local hydrology, that can lead to relatively swift recovery, but too much or too little sediment addition can delay recovery of benthic fauna and associated vegetation and soil. We found maximum recovery of macroinvertebrates, vegetation, and soil in the Low, Medium, and Med-Vegetated sites within an elevation range of 8–15 cm above the ambient healthy marsh. However, the optimum amount of sediment-slurry addition varies with the restoration target. Although we did not see the same level of recovery below-ground as occurred above-ground, surface soil in the Low, Medium, and Med-Vegetated sites had begun to show signs of equivalence with reference sites, although the timeframe for complete recovery is unknown. Different ecosystem components of the restored salt marsh will take different durations for complete recovery. Therefore, the continued investigation of this restoration site and others that have received sediment-slurry application will provide a more complete picture of restoration success. Furthermore, the impact of rising sea levels on restoration success and salt marsh sustainability should be incorporated in future investigations.

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References
