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ORIGINAL PAPER

Utilization of a citizen monitoring protocol to assess the structure and function of natural and stabilized fringing salt marshes in North Carolina

Carolyn A. Currin · Priscilla C. Delano · Lexia M. Valdes-Weaver

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Abstract Narrow fringing salt marshes dominated by *Spartina alterniflora* occur naturally along estuarine shorelines and provide many of the same ecological functions as more extensive marshes. These fringing salt marshes are sometimes incorporated into shoreline stabilization efforts. We obtained data on elevation, salinity, sediment characteristics, vegetation and fish utilization at three study sites containing both natural fringing marshes and nearby restored marshes located landward of a stone sill constructed for shoreline stabilization. During the study, sediment accretion rates in the restored marshes were approximately 1.5- to 2-fold greater than those recorded in the natural marshes. Natural fringing marsh sediments were predominantly sandy with a mean organic matter content ranging between 1.5 and 6.0%. Average *S. alterniflora* stem density in natural marshes ranged between 130 and 222 stems m^{-2}, while mean maximum stem height exceeded 64 cm. After 3 years, one of the three restored marshes (NCMM) achieved *S. alterniflora* stem densities equivalent to that of the natural fringing marshes, while percentage cover and maximum stem heights were significantly greater in the natural than in the restored marshes at all sites. There was no significant difference in the mean number of fish, crabs or shrimp captured with fyke nets between the natural and restored marshes, and only the abundance of *Palaemonetes vulgaris* (grass shrimp) was significantly greater in the natural marshes than in the restored ones. Mean numbers of fish caught per 5 m of marsh front were similar to those reported in the literature from marshes adjacent to tidal creeks and channels, and ranged between 509 and 634 fish net^{-1}. Most of the field data and some of the sample analyses were obtained by volunteers as they contributed 223 h of the total 300 h spent collecting data from three sites in one season. The use of fyke nets required twice as many man-hours as any other single task. Vegetation and sediment parameters were sensitive indicators of marsh restoration success, and volunteers were capable of contributing a significant portion of the labor needed to collect these parameters.

Keywords Citizen monitoring · Fish utilization · Fringing marsh · Restoration · Salt marsh · Shoreline stabilization · Sill · *Spartina alterniflora*

C. A. Currin (✉) · P. C. Delano · L. M. Valdes-Weaver
NOAA Center for Coastal Fisheries and Habitat Research, 101 Pivers Island Road, Beaufort, NC 28516, USA
e-mail: Carolyn.Currin@noaa.gov

P. C. Delano
e-mail: Priscilla.Delano@noaa.gov

L. M. Valdes-Weaver
e-mail: LexiaW@nccoast.org

Introduction

Fringing salt marshes are a common feature of the estuarine shoreline in the lagoons and sounds behind the barrier islands of the southeast coast of the United States. These fringing marshes often consist of a 10- to 20-m-wide swath of *Spartina alterniflora* Loisel, with a narrow band of upper marsh vegetation [*Spartina patens* (Aiton) Muhlenberg, *Salicornia virginica* L., *Distichlis spicata* (L.) E. Greene and/or *Juncus roemerianus* Scheele] between the lower marsh and uplands. Salt marshes provide a variety of ecosystem functions, including provision of fishery habitat, sediment stabilization, primary and secondary productivity, nutrient cycling, filtering of sediments and contaminants from the water column and interception of sediments and contaminants delivered by land. Many of these functions have been demonstrated to occur within 10 m of the marsh edge (Knutson 1988; Minello et al. 1994; Peterson and Turner 1994; Kastler and Wiberg 1996; Leonard et al. 2002) and thus are provided by fringing marshes. However, the location of these marshes at the interface between land and sea place them in a tenuous position. Sea level rise, subsidence and wave energy combine to erode fringing marshes from the estuarine front, while coastal development poses both direct and indirect threats to marshes from the landward edge.

Hardened structures, such as seawalls or bulkheads, are often used to stabilize eroding shorelines, but these structures may result in a loss of habitat structure and function (Mock 1966; Bozek and Burdick 2005; Seitz et al. 2006). An alternative to seawall installation is the planting of salt marsh grass, a natural buffer against wave energy (Knutson 1988; Rogers et al. 1992). More recently, marsh grass planting has been combined with offshore stone breakwaters, a design which is often referred to as a 'living shoreline' (Barnard 2004). This approach has gained popularity among environmental groups and coastal management agencies as a compromise between providing sufficient shoreline stabilization and avoiding the destruction of coastal habitat (NRC 2007). The design typically consists of a stone sill, parallel to the shore, set at a base elevation near mean low water, with sill heights extending from mean sea level to mean high water. Sill placement or construction frequently requires the transplanting of marsh landward of the sill to create a salt marsh or to supplement existing marsh plants. Although similar designs have been evaluated along ocean beaches (Martin et al. 2005), a detailed assessment of the ecological function of these structures and associated created marsh in estuarine settings has not been conducted (NRC 2007).

Salt marsh acreage in the continental United States has been in decline for decades, and habitat restoration along with conservation are necessary in order to achieve the National Wetlands Policy Forum's goal of "no net loss" of wetlands for the United States set in 1988 (Weinstein and Kreeger 2000). The recovery of ecological function in created or restored marshes may take years to decades, and the rate of the recovery process varies according to the parameter measured (Simenstad and Thom 1996; Craft et al. 1999; Zedler and Callaway 1999; Morgan and Short 2002). Although large (>10 ha) marsh restoration projects often include a comprehensive monitoring program (Simenstad and Thom 1996; Weinstein et al. 2001), smaller, community-based restoration projects generally lack sufficient means to gauge restoration success or collect information for adaptive management (Thom 1997; Palmer et al. 2005) A significant obstacle to monitoring habitat restoration projects is labor cost, and community volunteers may provide some relief (Sharpe and Conrad 2006; Oscarson and Calhoun 2007). Several guidelines for monitoring restored salt marshes have been developed, and these identify the core variables necessary to monitor restoration effectiveness and the need for selecting appropriate reference sites (Zedler 2001; Neckles et al. 2002; Thayer et al. 2005). The study reported here monitored many of those core variables for a period of 2–3 years after marsh restoration.

We utilized citizen volunteers to help map marshes and obtain data on sediment, elevation, vegetation and nekton use for restored marshes associated with 'living shorelines' and for natural reference marshes. Natural reference marshes used in this study were selected based on their proximity to the restored marsh and their similar tidal elevation, overall size, landscape position and sediment characteristics (Neckles et al. 2002). The overall objectives of the study were to assess the habitat value of living shoreline marsh restorations compared to their natural fringing marsh counterparts and the effectiveness of a volunteer monitoring program. It addresses the following questions: (1) What are the structure and function of narrow fringing marshes that are typical of shoreline stabilization restoration efforts? (2) Do living shoreline marsh restorations achieve natural levels of sediment and vegetation characteristics and fishery utilization? (3) Which of the core variables recommended for monitoring marsh restoration projects are best suited for a citizen monitoring program?

Methods

Description of study sites

Study sites were located in the southern Outer Banks section of North Carolina, an area characterized by barrier island and lagoon geomorphology (Fig. 1). The North Carolina Coastal Federation (NCCF), a non-profit environmental organization, designed and constructed the three 'living shoreline' sites used in the study. At each site, a low stone sill, consisting of granite boulders amassed in an unconsolidated low-relief structure running parallel to the shoreline, was constructed at a base elevation near the mean low water mark, extending to a height just above mean sea level. Marsh grass, *Spartina alterniflora* and *S. patens*, was planted behind the stone sill at elevations consistent with nearby natural marshes. The sites were bordered by seawalls and/or fringing salt marshes consisting primarily of *S. alterniflora*.

In September 2001, the NCCF removed a steel breakwater adjacent to the North Carolina Maritime Museum (NCMM) on the Newport River Estuary near Beaufort (34.7291°N, −76.6678°W) (Fig. 1) and replaced it with three 50-m low stone sills placed parallel to the shore. Behind these, volunteers planted *S. alterniflora* and *S. patens* to supplement existing marsh vegetation. This site stretches along 500 m of restored and natural marsh and is bordered by a navigable waterway seaward and developed uplands landward. Data were obtained from survey transects behind each of the sills and from a natural reference marsh adjacent to the sills.

The second site was adjacent to Duke University Marine Lab (DUML) (34.7177°N, −76.6736°W), approximately 1 km south of the NCMM site (Fig. 1). Seawall and fish pens were removed in February 2002 and replaced with a stone sill 80 m in length. The sill was curved to adjoin the existing seawall on each side. Two sill gaps, approximately 2 m wide, were created to increase fish utilization of the site. *S. alterniflora and S. patens* were planted landward of the sill in June 2002. The planted marsh dimensions were 10 m wide at both ends and 27 m at the widest point in the middle. The sill was bordered by a navigable waterway seaward and developed uplands landward.

The third site, which was near the North Carolina Aquarium at Pine Knoll Shores (PKS), is located 15 km west of Beaufort, on the northern side of

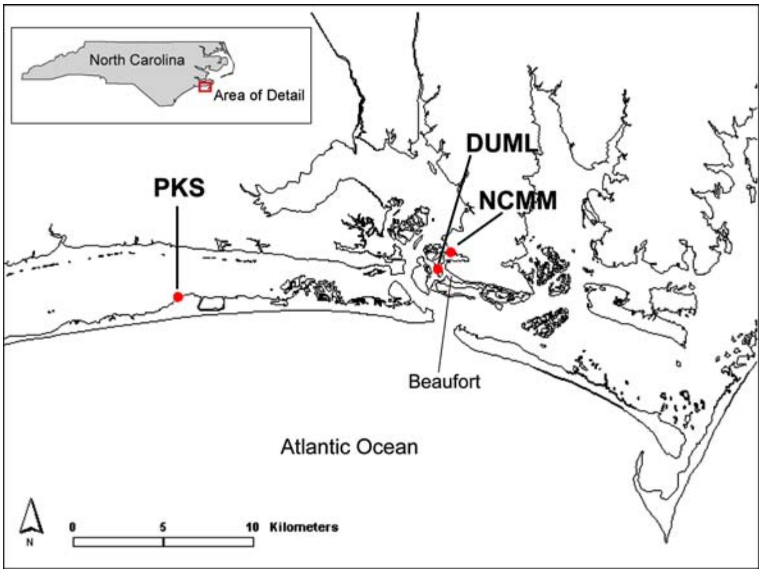

Fig. 1 Location of the three study sites monitored by the volunteers: Pine Knoll Shores (*PKS*), Duke University Marine Laboratory (*DUML*) and North Carolina Maritime Museum (*NCMM*)

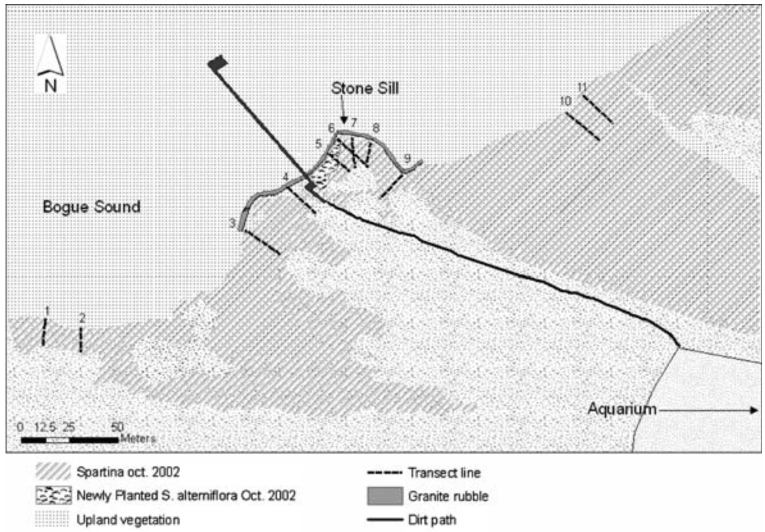

Fig. 2 Example of the base map prepared for the Pine Knoll Shores (PKS) site, illustrating the distribution of the salt marsh, location of the stone sill and sampling transects (*1–11*) and nearby landmarks

Bogue Banks (34.7011°N, −76.8319°W) (Figs. 1, 2). Here the marsh is bordered by Bogue Sound and the Roosevelt State Forest. Although much of the shoreline is composed of fringing marsh, portions of the marsh extend 100 m or more from the shore back to woods along a small tidal creek. The NCCF installed a 100-m-long stone sill around an existing dock and included two fish drop downs. Volunteers planted *S. alterniflora* and *S. patens* behind the sill in June 2002 although much of the area was already vegetated with marsh plants.

Natural reference marshes were selected near each restored site, based on physical similarity and proximity to the restored marshes (Neckles et al. 2002). The NCMM and PKS natural marshes were directly adjacent to the restored marshes, ensuring similar hydrology, geomorphology, tidal range, water quality, orientation and fetch. As there was no adjacent natural marsh at the DUML site, a nearby natural site was selected based on the similarity of these criteria. It is located approximately 300 m north of the restored marsh and sill location.

A Trimble ProXR unit was used to delineate marsh habitat, sill extent, location of transects and fyke net locations at each site. ARC VIEW was used to generate maps for each study site.

Volunteer assessment

In addition to habitat evaluation, this study assessed the effectiveness of using community volunteers in restoration monitoring. Over 60 volunteers of diverse ages and backgrounds were recruited from the local community and trained to collect scientific data. They participated in most aspects of field and laboratory work. During one season (fall 2003), a record was kept of National Oceanic and Atmospheric Administration (NOAA) staff and volunteer time to allow for an analysis of labor costs by parameter – i.e. sediment, vegetation or nekton utilization. Volunteers participated in all aspects of data collection. NOAA staff developed the sampling transect design and instructed volunteers in proper data collection and measurement techniques; they also supervised all field collection and laboratory processing of samples.

Monitoring of marsh parameters

Monitoring of all parameters was conducted during the spring (April) and fall (September or October) of each year, beginning in fall 2001 (at NCMM) and fall 2002 (at DUML and PKS) through to spring 2004. All fall 2003 sampling occurred after Hurricane Isabel (category 2) made landfall on the North Carolina coast on September 18, 2003, approximately 45 km northeast of the study area. Measurements of surface elevation, sediment characteristics and vegetation were obtained from permanent transects that ran perpendicular to the shoreline at each site. Transect locations were selected using restricted random sampling, which has been shown to produce better density estimates than simple random sampling

(Elzinga et al. 1998). Transects began at the seaward edge of the marsh and continued to the start of upland vegetation. This paper will present only data from the *S. alterniflora*-dominated portion of the marshes, which reached no more than 20 m from the start of the transect at most sites.

The number of transects within a site depended on the extent of the natural marsh and the length of the stone sill. Sampling plots began at the lower marsh edge, and samples were collected every 3 or 5 m along the transect, depending on total transect length. The NCMM site consisted of five natural reference transects and 14 restored transects. At DUML, samples were taken from six natural references and five restored transects, while at PKS, there were four natural references and five restored transects. For the analysis aimed at determining whether changes in marsh sediment and vegetation characteristics occurred with increased distance from the marsh edge, sampling plots were divided into four groups: plot group 1 included plots sampled at 0 and 3 m from the lower marsh edge, plot group 2 consisted of plots at 5 and 6 m from the lower marsh edge, plot group 3 included plots at 9 and 10 m from the lower marsh edge and plot group 4 consisted of plots sampled at 15 and 20 m from the lower marsh edge.

Surface elevation

Surface elevation measurements (±5 mm 100 m^{-1}) were performed by NOAA staff and volunteers using a leveling rod and rotary laser level. Elevations were obtained at the same locations used for vegetation and sediment data collection. Tidal datums or elevation benchmarks were only in close proximity to the DUML site, so intersite comparison of elevations was not performed. All elevations within a site were corrected to the same reference point during a collection and among collection periods.

Elevation data collected from DUML in fall 2002 and fall 2003 and those collected from PKS in spring 2003 and spring 2004 were used to calculate elevation change (cm) and slope (ratio of height over distance). Data from NCMM were not used due to variability in baseline elevations: the low number of replicate plots greater than 15 m landward of the marsh edge precluded their use in these calculations.

Sediment parameters (porewater salinity, grain size and organic matter)

Porewater salinity was measured by the volunteers at each plot and sampled on all transects. A small amount of surface sediment was placed in a 10-cc syringe, and the porewater was extracted through two Whatman glass fiber (grade 1) filters. Salinity (±1 ppt) was measured in the field using a refractometer.

Grainsize and organic matter samples (top 2 cm of sediment layer) were collected by the volunteers at every plot sampled on select transects and taken back to the lab to be frozen for later analysis. Grainsize analysis was performed by NOAA staff and volunteers. For the organic matter analysis, a subsample of about 20 g (wet weight) was dried overnight in a 100°C oven, and then placed in a 450°C oven for 6–8 h to obtain ash weight. To determine particle size content, we washed a subsample of about 20–30 g through 2-mm and 63-μm sieves to determine the gravel (>2 mm), sand (>63 μm, <2 mm) and silt-clay (<63 μm) size fractions, in a method adapted from Plumb (1981).

Vegetation parameters

At each sampling plot, volunteers recorded plant species composition and the percentage cover, stem density and stem height of each species. Percentage cover was estimated over the whole plot (m^{-2}) according to the Braun–Blanquet visual percentage estimation scale (Braun-Blanquet et al. 1932). Stem density was determined by counting the total number of stems of each species found in the landward right 0.25-m^{-2} area of the quadrat. Stem height was recorded for the three tallest stem heights for each plant species found within the whole plot (m^{-2}).

Nekton

Fyke nets were used by volunteers and NOAA staff to determine the utilization of marsh habitat by fish and decapod crustaceans at the DUML and NCMM locations. The reduced tidal amplitude at PKS made this site unfavorable for the use of fyke nets as they require complete marsh inundation and drainage within a tidal cycle. The nets were made of 3-mm black mesh and consisted of a compartmentalized net bag with two wings that stretched out from the bag at 45° angles, opening onto 5 m of marsh edge (Hettler 1989). Nets were located near vegetation transects. During collections, three nets were set at both the DUML restored and natural site and four nets were set at the NCMM restored marsh locations, with two at its paired natural marsh site. Fyke nets were set at high tide and fished at low tide to capture fish and decapods exiting the marsh as the tide dropped. Species were identified, counted, measured for total length (fish and shrimp) or carapace length (crabs) and then released in the field to minimize mortality. Unknown species were placed on ice and returned to the lab for identification.

Volunteer time/cost of data collection

Total volunteer and administrative hours were recorded for the fall 2003 monitoring season. Volunteer hours were logged and characterized according to type of task – i.e. vegetation, fyke nets, sediment, etc. Time spent by NOAA personnel in support of data collection was also recorded. Other duties, such as the recruiting and scheduling of volunteers, were recorded as administrative tasks. Hours required for data entry and analysis were not included.

Statistics

Statistical tests were performed using the analyst application of the statistical package SAS, ver. 8.02 (SAS Institute 1999). Because of the violations of the assumptions for parametric statistical testing, Kruskal–Wallis or Mann–Whitney nonparametric tests were performed to test for significant differences in sediment characteristics, *S. alterniflora* percentage cover, stem density and maximum stem height as well as for fish and decapod abundance between restored and natural marshes, sampling seasons (spring vs. fall), spring and fall samplings (i.e. spring 2002 vs. spring 2003, etc.) and plot groups. Sample sizes for the various components analyzed are included in Tables 1 through 7 or in the text, where applicable. An alpha level of 0.05 was used for all hypothesis testing.

Fish and shrimp species diversity was compared between marsh types at each site by calculating Shannon diversity indices using PRIMER-E software (Clarke and Gorley 2001). One-Way ANOVA was used to decipher significant differences in fish and

Table 1 Marsh elevation and slope change over time at Duke Marine Lab (DUML) and Pine Knoll Shores (PKS) (*ND* not determined)

	Elevation change (cm)			
	DUML		PKS	
Plot location (m)	Natural	Restored	Natural	Restored
0	−5.96 ± 4.19 (6)	7.27 ± 7.19 (5)	13.08 ± 4.49 (4)	25.18 ± 2.52 (7)
5	−0.55 ± 12.84 (5)	12.67 ± 8.31 (5)	14.59 ± 3.00 (4)	19.05 ± 0.96 (7)
10	16.41 ± 9.86 (5)	10.06 ± 13.41 (5)	7.68 ± 5.68 (4)	29.47 ± 9.53 (6)
15	22.70 ± 6.72 (5)	6.78 ± 9.76 (4)	ND	21.52 ± 1.22 (3)
Marsh average	7.48 ± 4.79 (21)	9.32 ± 4.61 (19)	11.78 ± 2.53 (12)	23.96 ± 2.60 (23)
	Slope			
	DUML		PKS	
Time	Natural	Restored	Natural	Restored
T1	0.07 ± 0.01 (6)	0.09 ± 0.01 (5)	0.04 ± 0.01 (4)	0.03 ± 0.01 (7)
T2	0.08 ± 0.001 (6)	1.00 ± 0.01 (5)	0.03 ± 0.01 (4)	0.03 ± 0.01 (7)

Data were collected from DUML in fall 2002 (T1) and fall 2003 (T2), and from PKS in spring 2003 (T1) and spring 2004 (T2). Plot locations represent distance landward from the lower marsh edge. Hurricane Isabel passed near the study area in September 2003, prior to the fall sampling at DUML. Standard error and number of samples (in parentheses) are also shown shrimp diversity between natural and restored marshes and between seasons (spring vs. fall). The PRIMER software was also used to perform non-metric multidimensional scaling (MDS) and analyses of similarities (ANOSIM) to test for differences in fish and shrimp community structure between marsh types at each site. Data were fourth root transformed prior to running these analyses.

Results

Base maps

Maps of each site were prepared that illustrate the location of stabilization structures, distribution of marsh plant cover and the location of sampling transects. An example of the base map for PKS is shown in Fig. 2. These maps were used by volunteer groups to facilitate the locating of sampling locations and will provide a valuable reference point for detecting long-term changes in marsh habitat and shoreline features.

Surface elevation

Both natural and restored marshes at DUML showed an overall increase in elevation between fall 2002, shortly after marsh establishment, and fall 2003. The mean elevation change for the DUML natural marsh was 7.5 cm, while the mean elevation change for the DUML restored marsh was 9.3 cm (Table 1). The DUML natural marsh exhibited a trend of greater elevation increase 15 m into the marsh than at the marsh edge as elevations decreased at the 0- and 5-m plot locations. The DUML restored marsh showed a greater elevation increase 5 m into the marsh. All plot locations at both PKS marshes exhibited net sediment accretion between spring 2003 and spring 2004. The trend in the PKS natural marsh was for a greater elevation increase near the marsh edge and less at 10 m inside the marsh, while the restored marsh showed similar and larger increases in sediment elevation at all plot locations. Overall, sediment accretion rates in the restored marshes behind the stone sills were 1.2- (DUML) to 2-fold (PKS) greater than those recorded in the natural marsh (Table 1).

There was little change in the marsh slope during the study and, within a site, the slope among marsh types was similar (Table 1). The slope at the DUML marshes increased only slightly from 0.07 to 0.08 (natural marsh) and 0.09–1.0 (restored marsh) between sampling times. The PKS marshes exhibited a constant and more gradual slope than did marshes at DUML.

Sediment porewater, organic matter, and grain size

Mean porewater salinity ranged between 29 and 34 ppt, which is reflective of the polyhaline setting and regular tidal flooding at all sites (Table 2). Sediment salinity did not vary significantly between natural and restored marshes at any site, regardless of whether all the data were pooled ($P > 0.1324$) and regardless of season ($P > 0.0583$) or distance into the marsh ($P > 0.1344$).

Table 2 Mean porewater salinity, percentage organic matter (OM) and percentage sediment composition (sand, silt and gravel) for natural and restored marshes at each site

	DUML		NCMM		PKS	
	Natural (27)	Restored (35)	Natural (31)	Restored (70)	Natural (31)	Restored (49)
Salinity	31.27 ± 1.21 (60)	33.87 ± 1.90 (32)	28.88 ± 1.06 (67)	32.10 ± 1.34 (135)	32.29 ± 1.15 (63)	34.29 ± 1.38 (86)
OM	**1.48 ± 0.23**	0.83 ± 0.28	1.56 ± 0.52	**3.59 ± 0.73**	**6.02 ± 1.04**	1.61 ± 0.15
Sand	87.84 ± 1.75	**94.86 ± 0.40**	**90.37 ± 1.28**	74.31 ± 2.03	77.89 ± 3.35	**87.95 ± 1.05**
Silt	**9.21 ± 1.12**	4.55 ± 0.35	8.67 ± 1.23	**12.99 ± 1.13**	**20.64 ± 3.25**	9.01 ± 0.56
Gravel	**2.95 ± 1.17**	0.58 ± 0.15	0.96 ± 0.37	**12.70 ± 1.67**	1.47 ± 0.29	3.05 ± 0.85

Standard error and number of samples (in parentheses) are also shown. Because porewater salinity was measured at all plots and transects, the number of salinity samples was greater than the number of samples used for determining organic matter and particle size and is shown immediately following its value. Within a site, values in bold indicate that the parameter was significantly greater in that marsh type than the other

Mean sediment organic matter content was relatively low at all three sites regardless of marsh type, ranging from 0.83% at the DUML restored marsh to 6.02% at the PKS natural marsh (Table 2). Sediments at all locations were primarily comprised of sand, with average sand content ranging from 74.31% at the NCMM restored marsh to 94.86% at the DUML restored marsh.

Significant differences in sediment characteristics were observed between natural and restored marshes at all sites ($P < 0.0302$), with the exception of gravel content at PKS ($P = 0.6892$) (Table 2). Overall, organic matter, silt and gravel content were significantly greater in the natural marshes than in the restored marshes at DUML and PKS, while the reverse was observed at NCMM. Significant differences in organic matter, sand and silt content were observed between plot groups at all three natural marshes ($P < 0.0460$), while only sand and silt at the DUML restored marsh and gravel at the PKS restored marsh were significantly different between plot groups (Table 3). The general trend was for increasing organic matter content with increased distance from the marsh edge.

Vegetation

The percentage cover and maximum stem height of *S. alterniflora* was significantly lower in restored marshes than in natural marshes within each site, regardless of whether the data was pooled by site ($P < 0.0176$) or by site and season ($P < 0.0464$) (Table 4). In addition, stem density was also significantly lower in restored marshes than in natural marshes at all sites ($P < 0.0015$) except NCMM ($P = 0.6311$).

Seasonal differences in vegetation characteristics were only observed for maximum stem heights of *S. alterniflora*, which were significantly greater in the fall than in the spring at all locations ($P < 0.0001$) (Fig. 3). Also, percentage cover at the PKS restored marsh was significantly greater during the spring than during the fall ($P = 0.0136$).

During the fall 2001 through to the spring 2004 study period, changes in the vegetation characteristics of *S. alterniflora* occurred at all locations, with the exception of the NCMM natural marsh (Fig. 3). At the NCMM restored marsh, percentage cover of *S. alterniflora* was significantly greater in spring 2003 ($P = 0.017$) and spring 2004 ($P < 0.0001$) than in spring 2002 and was significantly greater in fall 2003 than in fall 2001 ($P = 0.0004$) and fall 2002 ($P = 0.0099$). Stem densities at this site also increased significantly with time as they were significantly greater in fall 2003 than in fall 2001 ($P = 0.0005$) and fall 2002 ($P < 0.0001$). Maximum stem heights at the NCMM restored marsh were also significantly greater in fall 2003 than in fall 2001 ($P = 0.0051$). With the exception of maximum stem height at the DUML natural marsh ($P = 0.6806$), all vegetation parameters at the DUML natural and restored marshes significantly increased from fall 2002 to fall 2003 ($P < 0.0204$). The PKS natural and restored marshes showed the same patterns with respect to vegetation changes through time as stem densities significantly increased from fall 2002 to fall 2003 at both marsh types ($P < 0.0043$). Stem density at the PKS natural marsh also increased from spring 2003 to spring 2004 ($P < 0.0001$).

With the exception of the PKS natural marsh ($P > 0.1281$), the percentage cover and stem density of *S. alterniflora* varied significantly with distance into the marsh at all locations ($P < 0.0355$) (Table 5). Percentage cover and stem density showed very similar patterns along the marsh slope within the restored marshes of all three sites. For example, at the DUML restored marsh, values for both of these parameters were significantly lower at interior plots than at plots closer to the shoreline ($P < 0.0008$). At both the NCMM and PKS restored marshes, plots 5 m from the edge showed significantly greater percentage cover and stem density than did other locations within the marsh ($P < 0.0328$ NCMM; $P < 0.0152$ PKS). Maximum stem heights, however, were not significantly different between marsh locations at any of the sites ($P > 0.1127$).

Fish and invertebrate utilization of marsh habitat

Overall, there was no significant difference in the mean number of crabs, fish or shrimp between the natural and restored marshes at the DUML and NCMM sites regardless of whether all the data was pooled per site [$P > 0.0904, n = 33$ (DUML), $n = 36$ (NCMM)] (data not shown) or by site and season ($P > 0.0668$) (Table 6). Season was a major factor influencing the

Table 3 Mean percentage organic matter (OM) and percentage sediment composition (sand, silt and gravel) for natural and restored marshes at each site by distance into the marsh

	DUML							
	Natural				Restored			
Distance into marsh (m)	OM[a]	Sand[a]	Silt[a]	Gravel	OM	Sand[a]	Silt[a]	Gravel
0–3	1.41 ± 0.34 (8)	84.05 ± 4.69 (8)	10.32 ± 2.16 (8)	5.63 ± 3.74 (8)	0.85 ± 0.12 (8)	92.44 ± 0.84 (8)	6.67 ± 0.78 (8)	0.89 ± 0.36 (8)
5–6	2.12 ± 0.39 (8)	84.15 ± 1.86 (8)	13.54 ± 1.96 (8)	2.31 ± 0.77 (8)	0.46 ± 0.08 (8)	95.36 ± 0.62 (8)	4.38 ± 0.58 (8)	0.25 ± 0.06 (8)
9–10	0.49 ± 0.19 (7)	93.75 ± 1.35 (7)	5.43 ± 1.20 (7)	0.82 ± 0.54 (7)	0.37 ± 0.11 (8)	96.16 ± 0.60 (8)	3.05 ± 0.26 (8)	0.80 ± 0.50 (8)
15–20	2.07 ± 0.91 (4)	92.44 ± 2.59 (4)	4.96 ± 0.97 (4)	2.60 ± 1.85 (4)	1.42 ± 0.89 (11)	95.32 ± 0.61 (11)	4.24 ± 0.52 (11)	0.44 ± 0.12 (11)

	NCMM							
	Natural				Restored			
Distance into marsh (m)	OM[a]	Sand[a]	Silt[a]	Gravel	OM	Sand[a]	Silt[a]	Gravel
0–3	0.57 ± 0.06 (9)	93.67 ± 0.65 (9)	5.62 ± 0.59 (9)	0.71 ± 0.31 (9)	2.13 ± 0.27 (26)	73.31 ± 3.02 (26)	12.10 ± 1.72 (26)	14.59 ± 3.04 (26)
5–6	0.81 ± 0.08 (11)	90.75 ± 0.91 (11)	8.11 ± 0.83 (11)	1.14 ± 0.94 (11)	3.31 ± 1.45 (24)	78.86 ± 2.72 (24)	12.35 ± 1.53 (24)	8.79 ± 1.98 (24)
9–10	2.26 ± 1.42 (9)	91.70 ± 1.13 (9)	7.73 ± 1.23 (9)	0.57 ± 0.29 (9)	5.86 ± 1.88 (19)	71.14 ± 5.05 (19)	15.39 ± 2.84 (19)	13.47 ± 3.36 (19)
15–20	6.98 ± 3.32 (2)	67.46 ± 8.13 (2)	29.64 ± 10.38 (2)	2.91 ± 2.25 (2)	5.44 ± – (1)	51.42 ± – (1)	5.85 ± – (1)	42.73 ± – (1)

	PKS							
	Natural				Restored			
Distance into marsh (m)	OM[a]	Sand[a]	Silt[a]	Gravel	OM	Sand[a]	Silt[a]	Gravel
0–3	4.99 ± 2.89 (7)	87.73 ± 4.48 (7)	10.94 ± 3.62 (7)	1.32 ± 0.87 (7)	1.42 ± 0.28 (13)	89.39 ± 1.54 (13)	9.27 ± 1.21 (13)	1.33 ± 0.60 (13)
5–6	3.41 ± 1.37 (7)	85.47 ± 3.75 (7)	13.12 ± 3.11 (7)	1.41 ± 0.68 (7)	1.26 ± 0.24 (12)	90.94 ± 1.14 (12)	8.23 ± 1.05 (12)	0.83 ± 0.30 (12)
9–10	3.67 ± 0.82 (7)	85.85 ± 2.69 (7)	12.79 ± 2.31 (7)	1.35 ± 0.55 (7)	1.73 ± 0.29 (12)	86.19 ± 1.88 (12)	8.99 ± 1.08 (12)	4.82 ± 1.64 (12)
15–20	10.20 ± 1.77 (10)	60.11 ± 6.66 (10)	38.18 ± 6.74 (10)	1.70 ± 0.38 (10)	2.04 ± 0.37 (12)	85.14 ± 3.17 (12)	9.51 ± 1.21 (12)	5.36 ± 2.86 (12)

Standard error and number of samples (in parentheses) are also shown

[a] Parameters that were significantly different with increasing distance from the marsh edge

Table 4 Mean percentage cover, stem density and maximum stem height of *Spartina alterniflora* for natural and restored marshes at each site

	DUML		NCMM		PKS	
	Natural	Restored	Natural	Restored	Natural	Restored
Percentage cover	**33.04 ± 3.30 (86)**	10.03 ± 1.46 (73)	**32.60 ± 2.96 (93)**	21.54 ± 1.82 (241)	**46.37 ± 2.60 (68)**	26.31 ± 2.26 (104)
Stem density	**149.51 ± 15.16 (82)**	69.83 ± 10.22 (65)	130.21 ± 12.40 (85)	150.93 ± 12.07 (190)	**222.18 ± 16.79 (68)**	161.64 ± 14.93 (100)
Maximum stem height (cm)	**76.62 ± 4.19 (70)**	50.21 ± 4.93 (43)	**82.09 ± 4.60 (84)**	62.32 ± 2.90 (165)	**64.21 ± 3.54 (67)**	53.34 ± 2.59 (94)

Standard error and number of samples (in parentheses) are also shown. Within a site, bold values indicate the parameter was significantly greater in that marsh type than the other

abundance of fish at all locations as the numbers of fish were significantly greater during the spring than during the fall for all years combined ($P < 0.0184$) (Table 6). The numbers of shrimp were also significantly greater during the spring than in fall; however, only at the DUML natural marsh ($P = 0.0237$). On the contrary, crabs were significantly more numerous in the fall than in the spring at the DUML restored marsh ($P = 0.0382$). The overall abundance of crabs, fish and shrimp at all locations did not exhibit significant inter-annual change between the spring and fall samplings ($P > 0.0775$) (data not shown).

During the study, a total of 47 fish species, six crab species and six shrimp species were caught at the DUML and NCMM natural and restored marshes. The ten most abundant species and their mean seasonal abundance at each site and marsh type are listed in Table 7. During the spring, *Palaemonetes vulgaris* Say, 1818 ($P = 0.0035$) and *Lagodon rhomboides* Linnaeus, 1766 ($P = 0.0304$) were the only two species to show significantly different abundances between natural and restored marshes at DUML (significantly greater abundance at the natural marsh than the restored marsh) (Table 7). In the fall, only the mean abundance of *Fundulus heteroclitus* Linnaeus, 1766 was significantly greater in the natural marsh at NCMM than in the restored marsh ($P = 0.0393$).

The two most abundant species, *Leiostomus xanthurus* Lacepède, 1802 and *L. rhomboides* were significantly more abundant during the spring than in the fall at all locations ($P < 0.0037$ and $P < 0.0449$, respectively) (Table 7). *Mugil cephalus* Linnaeus, 1758, (DUML natural marsh), *Paralichthys dentatus* Linnaeus, 1766 (DUML natural marsh), *P. vulgaris* (DUML and NCMM natural marshes) and *Palaemonetes intermedius* Holthuis, 1949 (NCMM restored marsh) were also significantly greater during the spring than during the fall ($P = 0.0028$, $P = 0.0161$, $P < 0.0222$, $P = 0.0328$, respectively). Other species, however, were significantly more abundant in the fall than in the spring; *Eucinostomus argenteus* Baird and Girard in Baird, 1855 (DUML natural and restored marshes and NCMM restored marsh), *Synodus foetens* Linnaeus, 1766 (DUML natural marsh), *Callinectes sapidus* M. J. Rathbun, 1896 (DUML restored marsh), *Eucinostomus* sp. Baird and Girard in Baird, 1855 (DUML and NCMM restored marshes), *Fundulus majalis* Walbaum, 1792

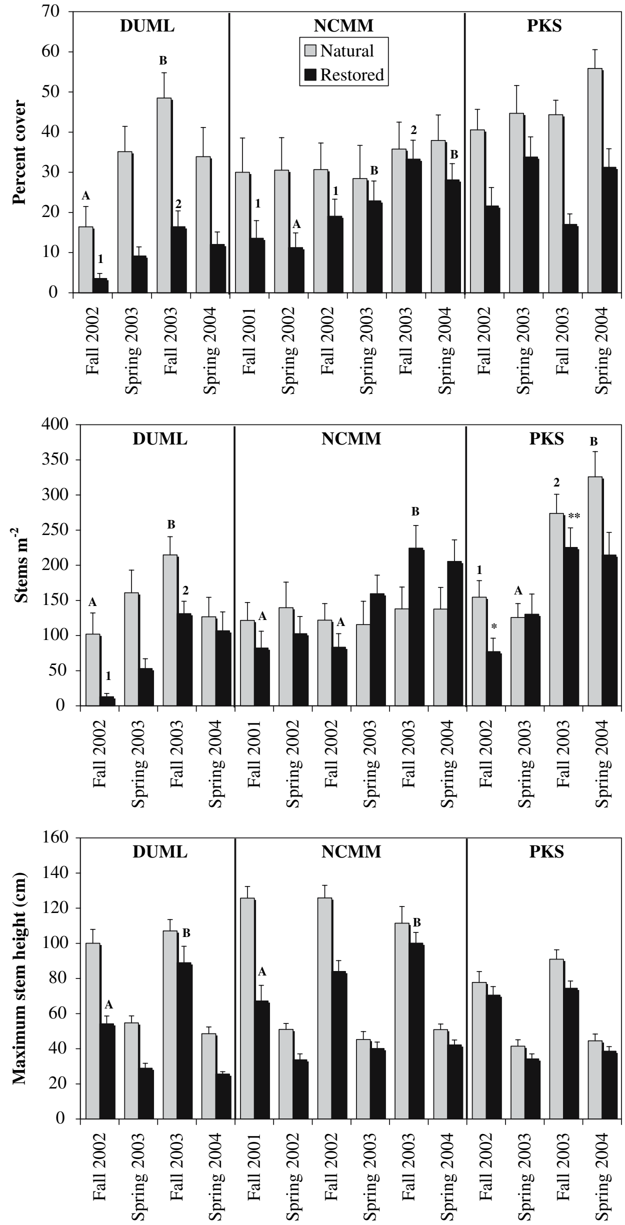

Fig. 3 Mean percentage cover, mean stem density and mean maximum stem height of *Spartina alterniflora* during fall 2001 through spring 2004 for natural and restored marshes at each study site. *Error bars* represent standard error. Significant differences within a study site between spring samplings (i.e., spring 2002 vs. spring 2003 vs. spring 2004) and/or between fall samplings are indicated with combinations of letters (*A*, *B*), numbers (*1*, *2*) or symbols (*, **)

(DUML restored marsh), *Gobionellus boleosoma* Jordan and Gilbert, 1882 (NCMM restored marsh) and *Menidia menidia* Linnaeus, 1766 (NCMM restored marsh) ($P < 0.0229$, $P = 0.0227$, $P = 0.0050$, $P < 0.0328$, $P = 0.0469$, $P = 0.0061$ and $P = 0.0427$, respectively).

Only the DUML natural marsh and the NCMM restored marsh demonstrated significant interannual differences in the abundance of several species between the spring and fall samplings (Table 8). For example, at the DUML natural marsh, the abundance of *Brevoortia tyrannus* Latrobe, 1802 ($P = 0.0339$), *M. cephalus* ($P = 0.0372$) and *P. intermedius* ($P = 0.0466$) was significantly greater in spring 2004 than in spring 2003, while the abundance of *S. foetens* was significantly greater in fall 2003 than in fall 2002 ($P = 0.0339$). At the NCMM restored marsh, *P. dentatus* was significantly more abundant in spring 2003 than in spring 2002 ($P = 0.0472$) and spring 2004 ($P = 0.0472$), while

Table 5 Mean percentage cover, stem density and maximum stem height of *S. alterniflora* for natural and restored marshes at each site per distance into marsh

	DUML					
	Natural			Restored		
Distance into marsh (m)	Percentage cover[a]	Stem density[a]	Maximum stem height (cm)	Percentage cover[a]	Stem density[a]	Maximum stem height (cm)
0–3	26.48 ± 4.98 (27)	126.07 ± 23.80 (27)	80.53 ± 8.41 (23)	12.24 ± 2.66 (25)	86.09 ± 16.95 (23)	50.42 ± 7.31 (21)
5–6	52.81 ± 6.08 (24)	220.50 ± 26.83 (24)	83.13 ± 6.84 (21)	13.53 ± 2.90 (19)	104.24 ± 23.67 (17)	46.08 ± 9.75 (12)
9–10	35.05 ± 6.72 (22)	147.16 ± 30.69 (19)	70.19 ± 8.87 (18)	10.56 ± 3.16 (16)	56.21 ± 17.19 (14)	54.73 ± 9.71 (10)
15–20	6.81 ± 3.08 (13)	64.00 ± 37.46 (12)	62.79 ± 5.68 (8)	0.00 ± 0.00 (13)	0.00 ± 0.00 (11)	–

	NCMM					
	Natural			Restored		
Distance into marsh (m)	Percentage cover[a]	Stem density[a]	Maximum stem height (cm)	Percentage cover[a]	Stem density[a]	Maximum stem height (cm)
0–3	10.18 ± 2.39 (33)	56.00 ± 10.59 (29)	76.10 ± 8.84 (28)	15.65 ± 2.08 (92)	120.70 ± 16.46 (80)	57.45 ± 4.20 (71)
5–6	60.33 ± 3.71 (30)	215.60 ± 22.52 (30)	92.73 ± 7.92 (30)	31.49 ± 3.82 (79)	202.83 ± 23.62 (65)	66.85 ± 5.50 (59)
9–10	26.58 ± 4.80 (26)	108.36 ± 17.21 (22)	78.15 ± 7.76 (22)	18.12 ± 3.43 (67)	132.30 ± 22.60 (43)	65.09 ± 5.56 (33)
15–20	48.75 ± 3.15 (4)	148.00 ± 56.45 (4)	65.83 ± 4.89 (4)	25.00 ± 10.00 (2)	74.00 ± 14.00 (2)	56.17 ± 5.17 (2)

	PKS					
	Natural			Restored		
Distance into marsh (m)	Percentage cover[a]	Stem density[a]	Maximum stem height (cm)	Percentage cover[a]	Stem density[a]	Maximum stem height (cm)
0–3	49.63 ± 4.10 (19)	275.16 ± 37.72 (19)	68.36 ± 8.35 (19)	24.74 ± 4.14 (29)	152.97 ± 27.28 (29)	55.62 ± 6.00 (27)
5–6	50.94 ± 6.01 (16)	229.50 ± 34.84 (16)	62.09 ± 7.10 (16)	38.73 ± 4.35 (28)	252.86 ± 30.55 (28)	46.69 ± 4.23 (27)
9–10	36.25 ± 6.08 (16)	157.75 ± 31.59 (16)	64.78 ± 6.70 (15)	18.88 ± 2.98 (26)	115.69 ± 19.31 (26)	61.35 ± 4.67 (26)
15–20	47.94 ± 4.42 (17)	216.71 ± 23.10 (17)	61.08 ± 5.93 (17)	21.12 ± 5.85 (21)	96.47 ± 32.86 (17)	46.89 ± 4.13 (14)

Standard error and number of samples (in parentheses) are also shown

[a] Parameters that were significantly different with increasing distance into the marsh

Prionotus evolans Linnaeus, 1766 was significantly more numerous in spring 2004 than in spring 2002 ($P = 0.0456$) and spring 2003 ($P = 0.0455$), and the abundance of *P. vulgaris* was significantly reduced in spring 2002 when compared to spring 2003 ($P = 0.0139$) and spring 2004 ($P = 0.0472$). Differences in species abundance between fall samplings were also observed at the NCMM restored marsh for *Callinectes similis* A. B. Williams, 1966 and *Lutjanus griseus* Linnaeus, 1758 (significantly more abundant in fall 2002 than in fall 2001 and fall 2003, $P < 0.0456$), *L. xanthurus* and *Palaemonetes*

Table 6 Mean seasonal abundance of crabs, fish and shrimp for natural and restored marshes at DUML and NCMM for all years combined

	DUML				NCMM			
	Spring		Fall		Spring		Fall	
	Natural (9)	Restored (9)	Natural (6)	Restored (6)	Natural (6)	Restored (6)	Natural (12)	Restored (12)
Crabs	1.00 ± 0.78	0.56 ± 0.34	2.00 ± 0.86	**3.33 ± 1.52**	5.17 ± 1.56	2.50 ± 1.23	4.33 ± 0.99	3.75 ± 0.90
Fish	**773.11 ± 258.25**	**912 ± 185.53**	173.33 ± 79.90	166.17 ± 60.27	**1220.67 ± 588.87**	**986.33 ± 515.07**	47.83 ± 15.34	32.08 ± 9.25
Shrimp	**51.33 ± 28.53**	13.44 ± 9.35	2.00 ± 1.44	20.83 ± 14.11	18.00 ± 8.06	42.83 ± 15.95	15.00 ± 5.42	8.42 ± 4.73

Standard error and number of samples (in parentheses) are also shown. Within each site, seasonal values were not significantly different between restored and natural marshes. Bold values indicate that seasonal abundance per site and marsh type was significantly greater in that season than in the other

pugio Holthuis, 1949 (significantly more abundant in fall 2003 than in fall 2001 and fall 2002, $P < 0.0202$), and *Farfantapenaeus* sp. Fabricius, 1798 (significantly more abundant in fall 2001 than in fall 2002 and fall 2003, $P < 0.0472$).

There was no significant difference in fish and shrimp species diversity between natural and restored marshes at DUML and NCMM regardless of whether all the data was pooled ($P > 0.3232$) (data not shown) or analyzed by season ($P > 0.1555$) (Table 9). However, fish and shrimp species diversity was greater at the restored marshes than at the natural marshes at both sites in the fall. On the contrary, in spring, greater diversity occurred at the natural marshes than at the restored marshes at both sites. Although seasonal differences were observed for fish and shrimp species diversity at all locations (greater in fall than in spring), this difference was not significant ($P > 0.0726$) (Table 9). Although MDS plots showed some weak separation by site as well as by marsh type, shrimp and fish communities did not differ significantly by either of these factors (ANOSIM for site: Global $R = 0$, significance = 48.6%; ANOSIM for marsh type: Global $R = -0.229$, significance = 94.3%).

Monitoring time by parameter

A total of 309 man-hours were spent in the collection of the data in fall 2003. Volunteers contributed 223 total hours. Nekton monitoring with fyke nets was by far the most labor-intensive monitoring task (Fig. 4). Volunteers contributed 68.5 h in the field and another 67 h in the lab sorting, identifying and measuring of fish, shrimp and crabs. NOAA personnel spent 16 h (administrative) in preparation and organization, and training and supervision by NOAA personnel was required for both fieldwork and laboratory sample processing.

Discussion

Habitat characteristics of fringing salt marshes

Marsh restoration projects undertaken with the objective of shoreline stabilization generally tend to occur in narrow (<25 m) bands of marsh with a navigable waterway on one side and upland

Table 7 The ten most abundant fish and decapod species and their mean seasonal abundance (all years combined) for natural and restored marshes at DUML and NCMM

	DUML				NCMM			
	Spring		Fall		Spring		Fall	
Species	Natural (9)	Restored (9)	Natural (6)	Restored (6)	Natural (6)	Restored (12)	Natural (6)	Restored (12)
Leiostomus xanthurus (spot)	**414.22 ± 159.38**	**740.22 ± 166.79**	4.17 ± 3.20	23.67 ± 17.98	**870.67 ± 453.44**	**818.92 ± 496.60**	2.67 ± 1.26	8.42 ± 4.52
Lagodon rhomboides (pinfish)	**322.78 ± 91.96**[a]	**107.56 ± 21.56**	54.50 ± 21.59	55.33 ± 24.57	**257.67 ± 105.88**	**118.67 ± 27.11**	32.17 ± 15.05	13.08 ± 3.46
Mugil cephalus (striped mullet)	**29.33 ± 14.80**	47.33 ± 27.30	0.00 ± 0.00	1.00 ± 0.68	20.50 ± 15.59	19.75 ± 12.06	0.17 ± 0.17	0.25 ± 0.13
Palaemonetes vulgaris (marsh grass shrimp)	**47.00 ± 27.87**[a]	0.44 ± 0.24	0.17 ± 0.17	0.67 ± 0.67	**10.33 ± 6.07**	25.92 ± 12.55	0.00 ± 0.00	4.58 ± 4.23
Menidia menidia (Atlantic silverside)	0.22 ± 0.22	0.11 ± 0.11	94.67 ± 84.15	0.50 ± 0.34	1.17 ± 0.98	0.25 ± 0.25	4.67 ± 3.68	**1.08 ± 0.65**
Palaemonetes pugio (daggerblade grass shrimp)	0.56 ± 0.24	12.89 ± 9.19	1.83 ± 1.47	20.17 ± 14.26	4.17 ± 3.58	9.50 ± 4.56	9.00 ± 5.55	1.83 ± 0.94
Brevoortia tyrannus (Atlantic menhaden)	0.56 ± 0.44	0.78 ± 0.78	0.00 ± 0.00	0.00 ± 0.00	25.00 ± 24.01	18.00 ± 17.73	0.00 ± 0.00	0.00 ± 0.00
Eucinostomus argenteus (spotfin mojarra)	0.00 ± 0.00	0.00 ± 0.00	**11.17 ± 7.49**	**32.67 ± 18.73**	0.17 ± 0.17	0.00 ± 0.00	2.83 ± 1.83	**2.92 ± 1.47**
Eucinostomus sp. (small spotfin mojarra <30 mm)	0.00 ± 0.00	0.00 ± 0.00	3.67 ± 3.67	**39.50 ± 28.35**	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	**1.33 ± 0.91**
Anchoa mitchelli (bay anchovy)	1.67 ± 1.43	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	28.00 ± 27.80	3.00 ± 2.65	0.17 ± 0.17	1.00 ± 0.72
Others	0.18 ± 0.06 (450)	0.33 ± 0.19 (450)	0.14 ± 0.04 (300)	0.34 ± 0.12 (300)	0.52 ± 0.22 (300)	0.35 ± 0.09 (600)	0.31 ± 0.09 (300)	0.20 ± 0.04 (600)

The mean seasonal abundance of the remaining species collected is listed as ‘Others’. Standard error and number of samples (in parentheses) are also shown. Note the number of samples for the “Other” category is different than for the individual species. Bold values indicate abundance per site and marsh type was significantly greater in that season than the other

[a] Abundance per site and season was significantly greater in that marsh type than the other

Table 8 The mean abundance of fish and decapod species that demonstrated significant differences in abundance between subsequent spring samplings and/or fall samplings

Species	DUML Natural				NCMM Restored					
	Spring 2003 (6)	Spring 2004 (3)	Fall 2002 (3)	Fall 2003 (3)	Spring 2002 (4)	Spring 2003 (4)	Spring 2004 (4)	Fall 2001 (4)	Fall 2002 (4)	Fall 2003 (4)
Leiostomus xanthurus (spot)	300.50 ± 78.75	641.67 ± 484.79	0.33 ± 0.33	8.00 ± 6.03	56.00 ± 35.01	151.75 ± 37.13	2249.00 ± 1298.45	1.75 ± 0.85	1.00 ± 0.58	**22.50 ± 11.14**
Mugil cephalus (striped mullet)	8.33 ± 7.14	**71.33 ± 32.42**	0.00 ± 0.00	0.00 ± 0.00	9.25 ± 9.25	7.25 ± 3.20	42.75 ± 35.21	0.50 ± 0.29	0.00 ± 0.00	0.25 ± 0.25
Palaemonetes vulgaris (marsh grass shrimp)	47.83 ± 39.99	45.33 ± 36.35	0.33 ± 0.33	0.00 ± 0.00	0.00 ± 0.00[a]	43.75 ± 30.23[a]	34.00 ± 21.61[a]	0.00 ± 0.00	13.00 ± 12.67	0.75 ± 0.75
Palaemonetes pugio (daggerblade grass shrimp)	0.50 ± 0.34	0.67 ± 0.33	0.00 ± 0.00	3.67 ± 2.73	0.00 ± 0.00	10.25 ± 8.09	18.25 ± 10.36	0.00 ± 0.00	0.00 ± 0.00	**5.50 ± 1.71**
Brevoortia tyrannus (Atlantic menhaden)	0.00 ± 0.00	**1.67 ± 1.20**	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	54.00 ± 53.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Palaemonetes intermedius (grass shrimp)	0.50 ± 0.50	**9.00 ± 7.00**	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	2.25 ± 2.25	10.25 ± 5.27	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Paralichthys dentatus (summer flounder)	1.00 ± 0.37	1.67 ± 0.88	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	**6.00 ± 3.24**	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Callinectes similis (lesser blue crab)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	**1.00 ± 0.41**	0.00 ± 0.00
Penaeus sp. (shrimp)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	**2.50 ± 1.32**	0.00 ± 0.00	0.00 ± 0.00
Lutjanus griseus (gray snapper)	0.00 ± 0.00	0.00 ± 0.00	1.33 ± 0.88	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	**0.75 ± 0.25**	0.00 ± 0.00

Table 8 continued

Species	DUML Natural				NCMM Restored					
	Spring 2003 (6)	Spring 2004 (3)	Fall 2002 (3)	Fall 2003 (3)	Spring 2002 (4)	Spring 2003 (4)	Spring 2004 (4)	Fall 2001 (4)	Fall 2002 (4)	Fall 2003 (4)
Synodus foetens (inshore lizardfish)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	**1.33 ± 0.33**	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Prionotus evolans (striped sea robin)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	**1.25 ± 0.48**	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00

Standard error and number of samples (in parentheses) are also shown. Values in bold indicate seasonal abundance was significantly greater in that year than the other year(s)

[a] Abundance in spring 2003 and 2004 was significantly greater than in spring 2002

Table 9 Shannon diversity indices for fish and shrimp species in natural and restored marshes at DUML and NCMM during the spring and fall samplings for all years combined

	DUML		NCMM	
	Natural	Restored	Natural	Restored
Spring	1.08	0.74	0.95	0.83
Fall	1.24	1.80	1.63	2.11

vegetation on the other. In this study, nearby fringing marshes, rather than interior regions of marsh plains or marshes adjacent to small tidal creeks, were selected as natural reference sites in order to match the geomorphology and hydrology of the restoration sites to the greatest extent possible (Neckles et al. 2002). The overall average organic matter content of natural fringing marshes was 3.1%, while the average sand content was 85%. These values indicate that the fringing marshes examined have coarser sediments with lower organic matter content than the majority of mature salt marshes previously examined in the mid-Atlantic (Hettler 1989; Craft et al. 1993; Osgood and Zieman 1993). However, the lower organic matter content and coarser sediments are similar to those reported from geologically young marshes on back-barrier islands (Osgood and Zieman 1993), fringing salt marshes (Craft et al. 1993), and transplanted marshes less than 5 years old (Piehler et al. 1998; Craft et al. 1999). Several studies have reported significant positive relationships between organic matter content and biomass or abundance of infauna (Sacco et al. 1994; Craft 2000), and it is therefore possible that natural fringing marshes might have lower infaunal densities than marshes located on tidal creeks or within marsh plains. However, Craft and Sacco (2003) also note that a sediment organic C content of 0.5% (equivalent to approximately 1% organic matter content, assuming organic matter is 50% C) is sufficient to support infauna communities resembling those of natural marshes with higher organic matter content.

The fringing natural reference marshes we sampled also exhibited a lower overall *S. alterniflora* stem density (overall mean of 164 stems m^{-2}) than has been reported elsewhere. For example, Hettler (1989) reported average stem densities between 516 and 890 stems m^{-2} in channel and rivulet marsh habitats, respectively, within 2 km of the DUML and

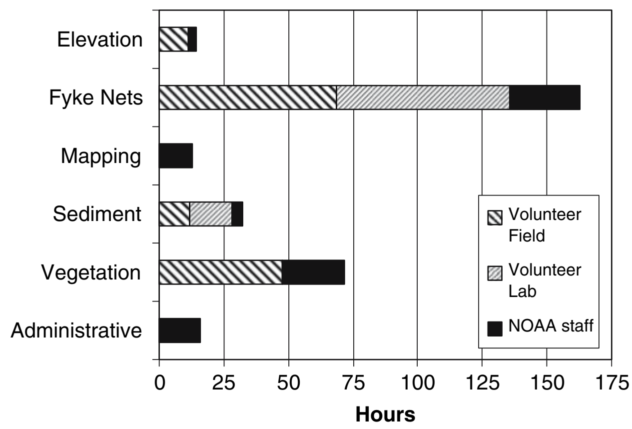

Fig. 4 Monitoring hours by task during the fall 2003 sampling period. For each task, the hours spent by volunteers in the field, by volunteers in the laboratory and by NOAA staff (field and lab combined) are recorded. NOAA staff time did not include data entry or analysis

NCMM study sites, while *S. alterniflora* stem density in South Carolina ranged between 490 and 2200 stems m^{-2} (Morris and Haskin 1990). An average of 300 stems m^{-2} was reported from natural reference marshes in an examination of North Carolina salt marshes adjacent to the intercoastal waterway (M. Fonseca, personal communication; Mark.Fonseca@noaa.gov), which were more similar in their physical setting to the reference marshes we sampled. It is apparent that the relatively high energy setting of these marshes can result in sediment and plant characteristics distinct from those obtained from interior marshes or marsh edges situated on tidal creeks.

The spatial pattern of sediment accretion in the fringing marsh at the DUML natural marsh was different than that reported from marshes bordering tidal creeks. Typically, creekbank marshes demonstrate highest accretion rates at the marsh edge, which ultimately lead to levee formation (Kastler and Wiberg 1996). However, the fringing marsh at the DUML site exhibited erosion at the marsh edge, and the highest accretion rate 15 m landward of the marsh edge. We note that this pattern would help to sustain the relatively steep slope (8%) and is consistent with the absence of levee formation at the marsh. The more gradually sloping marsh (3%) at PKS exhibited the highest accretion rates at the 0- and 5-m plots. The overall extent of sediment accretion recorded during the study period is an order of magnitude above the local relative sea level rise and much greater than that typically recorded in the literature (Childers et al. 1993; Morris et al. 2002), although storm or flooding events can result in higher sediment accretion rates (Stumpf 1983; Byrd and Kelly 2006). Our results are also similar to those reported from a transplanted marsh on a dredge spoil island located approximately 12 km from the study sites (Meyer et al. 1997). That study demonstrated the effectiveness of oyster reefs in stabilizing the marsh edge and promoting marsh sediment accretion. In addition, the sampling periods in Meyer et al. (1997) and in this study include hurricane conditions that affected the study area. Our technique, suitable for a citizen monitoring program, employed survey-grade leveling instruments (Byrd and Kelly 2006), while most studies of marsh accretion rates have used marker horizons, sediment elevation tables or radionuclide distributions to measure long-term accretions rates in salt marshes (Childers et al. 1993). When a benchmark is available as a reference point, this approach can provide accurate elevation data, capable of detecting short-term elevation changes of >5 mm (Byrd and Kelly 2006).

The fyke net data were obtained twice a year, in the spring and fall and, as a result, comparisons with literature values will be sensitive to the effects of season on fish and invertebrate species abundance and composition. Hettler (1989), using similar gear, sampled marsh use by fish over an annual cycle at sites near the DUML and NCMM sites. In that study, September was the time of peak biomass, while the greatest fish abundance was obtained in April. Hettler (1989) also noted that more species were present during the fall than during the spring and summer. Therefore, it is not unreasonable to compare our results with those reported by Hettler (1989) and Fonseca (personal communication) for fish utilization in North Carolina marshes. The three most abundant species in this study, *L. xanthurus*, *L. rhomboides* and *M. cephalus*, were the second, eighth and ninth most abundant fish species, respectively, in the Hettler (1989) study. However, Hettler (1989) noted differences in fish utilization between rivulet and channel marshes as *L. rhomboides* was more common in channel marshes than in rivulet marshes. Similar to our results, Fonseca (personal communication) found that *Paleomonetes* spp. were the most abundant shrimp. *Fundulus heteroclitus* was the most abundant fish sampled by Hettler (1989) and the fourth most abundant fish sampled by Fonseca (personal communication), although we rarely found any in our nets. Hettler (1989) reported that over 75% of the *F. heteroclitus* collected were from rivulet marshes. Therefore, this common marsh resident fish species may not utilize fringing marshes to the same extent as it does interior marshes. Overall, these fringing marshes provided habitat to similar numbers of fish as have been reported from other *S. alterniflora* marsh edge habitats, although species composition was different than that reported from marshes adjacent to tidal creeks (Hettler 1989). In particular, these fringing marshes provided habitat for the large numbers of larval fish entering the estuary in spring and particularly for *L. xanthurus* and *L. rhomboides*, which were significantly more abundant in the spring than in the fall. The results from this study were consistent with the observations made by Allen et al. (1997) in the North Inlet marshes of South Carolina where fish abundance did not vary inter-annually; however, species composition was found to vary significantly between years.

Salt marshes provide a variety of ecosystem functions, including provision of fishery habitat, sediment stabilization, primary and secondary productivity, nutrient cycling, filtering of sediments and contaminants from the water column and interception of sediments and contaminants delivered by land. Many of these functions have been demonstrated to occur primarily at the marsh edge (Knutson 1988; Minello et al. 1994; Peterson and Turner 1994; Kastler and Wiberg 1996; Leonard et al. 2002). We documented the utilization of fringing salt marshes by similar numbers and biomass of fish as were previously found in nearby tidal creek and channel marshes of greater width (Hettler 1989). We also documented significant sediment accretion rates occurring at all interior locations of natural fringing marshes, although the fringing marsh seaward edge exhibited erosion at one site (DUML) and net accretion at another (PKS). Primary production by *S. alterniflora* in fringing salt marshes may be similar to that of interior marshes, as despite the lower stem density in fringing marshes, overall stem height more closely resembles that of tall *S. alterniflora* characteristic of creek banks than that of the short *S. alterniflora* found in interior marshes. This is consistent with the regularly flooded nature of these fringing marshes.

Functional equivalency of natural and restored marshes

Sediment grain size and organic matter content remained significantly different between natural and restored marshes at PKS and DUML throughout the study. The restored NCMM marsh, which was one growing season older than the other restored marshes and which had considerable marsh vegetation present prior to sill installation, had similar or greater organic matter content or fine-grained sediment than did the natural marsh. In contrast to conclusions from earlier examinations of marsh restoration projects, we do not conclude that it will take 10–20 years for these marsh restoration sites to achieve the sediment features of their natural reference marsh counterparts (Craft et al. 1999), as the natural fringing marshes we sampled also consist of relatively coarse-grained, mineral sediments.

Of the three sites, NCMM was the only site where stem densities of *S. alterniflora* in the natural (x = 130 stems m^{-2}) marsh did not exceed those found in the restored marsh (x = 151 stems m^{-2}). As noted previously, *S. alterniflora* was already present behind the sills, and so a fairly quick recovery of *S. alterniflora* biomass may be expected. *S. alterniflora* stem density in transplanted marshes often equals or exceeds stem density found in natural marshes after one to three growing seasons (Broome et al. 1986; Craft et al. 1999). At PKS and DUML, the restored marshes exhibited significantly lower stem densities, less percentage cover and lower stem height than natural marshes throughout the study period. The planting of *S. alterniflora* and *S. patens* at PKS and DUML did not occur until June 2002, 9 months after the planting at

NCMM and relatively late in the growing season for transplanting (Broome et al. 1986). However, the trajectory of the vegetation response suggests that the vegetation parameters in restored marshes behind stone sills will resemble that of natural fringing marshes after three growing seasons.

Our results suggest that the stone breakwaters were very effective at trapping sediments and that sediment accretion rates in marshes behind offshore breakwaters were 1.5- to 2-fold greater than those in adjacent natural marshes. Previous studies have documented increased sediment accretion rates with higher stem density and stem height in natural marshes (Gleason et al. 1979; Morris et al. 2002). Given the lower stem density and canopy height of the restored marshes (Table 5), we would have expected lower sediment accretion rates in restored marshes. The observed higher accretion rates suggest that the breakwater structure was primarily responsible for the observed difference in accretion rate These results are consistent with an examination of stone breakwaters established parallel to the shore at several European sites, where silt-clay and organic matter C content increased landward of the structures (Martin et al. 2005). The coarse texture and low organic matter content found in both natural and restored fringing marshes in this study suggest that most of the sediment accretion observed was due to the deposition of sand-sized particles.

Fish and invertebrates utilized the restored marsh habitat in numbers similar to those found in natural marshes at all sampling times. This rapid fish utilization of restored or transplanted salt marsh habitat has been previously reported (Minello and Zimmerman 1992; Able et al. 2000). Both natural and restored marsh fish assemblages were dominated by *L. xanthurus*, and we did not observe any consistent differences in the fish, crab and shrimp assemblages utilizing natural and restored marsh sites. Fish and shrimp species diversity was greater in natural marshes than restored marshes in the spring when fish abundance was greater, while the opposite pattern (restored > natural) was true in the fall. These results imply that restored marshes provide a similar refuge function as natural marshes. However, we do not know if prey availability is similar between the two marshes.

We did not examine the fauna or flora associated with the stone sill, but note that intertidal hard substrate, other than oyster reefs, is not naturally present in estuaries in the southeast USA, and artificially armored shorelines may serve as a point of entry for invasive or alien species (Davis et al. 2002; Airoldi et al. 2005). We also note that the stone sill occupies what would otherwise be productive shallow subtidal habitat (MacIntyre et al. 1996) and that fishery utilization of marsh habitat is directly linked to elevation and tidal immersion period (Minello and Webb 1997). Caution should be exercised in applying the 'living shoreline' approach, as it can result in replacing a soft-bottom subtidal habitat with a higher elevation hard substrate and potentially increases the elevation of associated marsh habitat, which can reduce the overall ecosystem services provided.

Monitoring protocol and use of volunteers

A current issue facing natural resource managers is defining the variables that should be monitored in order to evaluate the success or failure of a habitat restoration project. It has been noted that the most readily obtained monitoring data are measures of structural attributes, while it is the functional attributes of the marsh ecosystem that resource agencies are striving to restore (Zedler and Lindig-Cisneros 2000; Neckles et al. 2002). However, when the resources available for a monitoring project are limited – as they almost always are – it is only the structural aspects of the marsh that can be measured, and it behooves the project manager to choose only the most important of those variables. A reasonable objective of most monitoring projects should be to confirm that the structural aspects of a restored ecosystem are intact, while it should be the goal of restoration research to establish the link between structure and function in those restored or created ecosystems (Zedler and Lindig-Cisneros 2000).

In this study, the monitoring parameters that were obtained with the least labor costs were plant stem density, percentage cover and stem height, sediment organic matter and particle size and elevation. The most time-consuming data to collect were on fish utilization, and although the fish data were useful in comparing fringing marshes to more extensive marshes bordering tidal creeks, there was no change in the relationship between natural and restored marshes throughout the study period, thereby reducing the effectiveness of this parameter as a metric for success (Minello and Zimmerman 1992; Able et al. 2000; Neckles et al. 2002). However, we note that catching fish was a popular activity with volunteers and has a high educational value, which may be an important aspect of some monitoring programs (Hudson 2001). The collection of vegetation data was completed in the field and, unlike the sediment and fish parameters, required no subsequent lab work, thus making it more adaptable to a wider variety of groups. The sediment analyses require a laboratory for processing, and so are limited to college students or other groups that have access to drying ovens, ashing ovens, sieves and balances. The elevation data were readily collected by volunteers but normalizing to baseline elevations required close attention by professional staff. We conclude that vegetation and sediment parameters are the most efficient and sensitive metrics of success for a citizen-based restoration monitoring program. The acquisition of GIS base maps and elevation data require more professional input but are equally valuable, particularly for any long-term assessment of restoration projects that include shoreline stabilization as an objective. This combination of parameters would also be the most useful for guiding post-restoration adaptive management. The failure of wetland mitigation and restoration programs to successfully offset wetland losses has been well-documented (Brown and Veneman 2001; Race and Fonseca 1996), and the lack of consistent evaluation and monitoring programs contributes to the problem (Palmer et al. 2005; Konisky et al. 2006).

The incorporation of citizen volunteers into environmental monitoring programs has been widely promoted (Sharpe and Conrad 2006; Oscarson and Calhoun 2007), although there are few published assessments of marsh restoration projects that describe the use of citizen volunteers for data collection (Markovchick-Nicholls et al. 2004). We successfully utilized volunteers from a variety of backgrounds to obtain monitoring data following standardized protocols (Neckles et al. 2002), a crucial aspect of developing a useful regional restoration monitoring program (Konisky et al. 2006). Our volunteers included senior citizens helping an environmental non-profit group, a mother–daughter service club, a community college science club and university students from a minority-serving institution. Participation in monitoring programs can be a valuable environmental education tool suitable for a diverse population (Hudson 2001).

Volunteers contributed over 1200 h of work in the field and lab over the course of the study and were capable of adequately performing most tasks with instruction and supervision. Volunteers were incorporated into most aspects of the project, including field and laboratory preparation, clean ups and basic laboratory work (sediment analysis) supervised by NOAA staff. This significant contribution of volunteers facilitated the collection of data which, in turn, allowed us to meet the objectives of the study. We found that (1) fringing salt marshes have sediment and vegetation properties distinct from more extensive marshes associated with tidal creeks and floodtide deltas, although they support similar numbers of fishery organisms, (2) marshes associated with living shoreline projects were on a trajectory to recover the structural attributes of reference marshes in a 3- to 5-year period, with the exception of sedimentation rates, which exceeded rates in natural reference marshes and (3) vegetation, sediment and elevation were the most valuable variables for assessing the success of living shoreline marsh restoration projects, and citizen volunteers could be utilized efficiently in data collection.

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References

Able KW, Nemerson DM, Light PR et al (2000) Initial response of fishes to marsh restoration at a former salt hay farm bordering Delaware Bay. In: Weinstein MP, Kreeger

DA (eds) Concepts and controversies in tidal marsh ecology. Kluwer, Dordrecht
Airoldi L, Abbiati M, Beck MW et al. (2005) An ecological perspective on the deployment and design of low-crested and other hard coastal defence structures. Coastal Eng 52:1073–1087
Allen DM, Edwards D, Feller RF et al. (1997) Detection and analysis of unusual events in long-term zooplankton and nekton data sets from North Inlet Estuary, SC, USA. Oceanol Acta 20:165–175
Barnard T (2004) Preserving the bay's living shorelines: a growing grass-roots effort. Va Wetlands Rep 19:1–2
Bozek CM, Burdick DM (2005) Impacts of seawalls on salt marsh plant communities in the Great Bay Estuary, New Hampshire, USA. Wetlands Ecol Manage 13:553–568
Braun-Blanquet J, Fuller GD, Conard HS (1932) Plant sociology: the study of plant communities. McGraw-Hill, New York
Broome SW, Seneca ED, Woodhouse WW Jr (1986) Long-term growth and development of transplants of the salt-marsh grass *Spartina alterniflora*. Estuaries 9:63–74
Brown SC, Veneman PLM (2001) Effectiveness of compensatory wetland mitigation in Massachusetts, USA. Wetlands 21:508–518
Byrd KB, Kelly M (2006) Salt marsh vegetation response to edaphic and topographic changes from upland sedimentation in a Pacific estuary. Wetlands 26:813–829
Childers DL, Sklar FH, Drakes B et al (1993) Seasonal measurements of sediment elevation in three mid-Atlantic estuaries. J Coast Res 9:986–1003
Clarke KR, Gorley RN (2001) PRIMER-E version 5. Plymouth Marine Laboratory, Plymouth
Craft CB (2000) Co-development of wetland soils and benthic invertebrate communities following salt marsh creation. Wetlands Ecol Manage 8:197–207
Craft CB, Sacco JN (2003) Long-term succession of benthic infauna communities on constructed *Spartina alterniflora* marshes. Mar Ecol Prog Ser 257:45–58
Craft CB, Seneca ED, Broome SW (1993) Vertical accretion in regularly and irregularly flooded microtidal estuarine marshes. Estuar Coast Shelf Sci 37:371–386
Craft C, Reader J, Sacco JN et al. (1999) Twenty-five years of ecosystem development of constructed *Spartina alterniflora* (Loisel) marshes. Ecol Appl 9:1405–1419
Davis JL, Levin LA, Walther SM (2002) Artificial armored shorelines: sites for open-coast species in a southern. Calif Bay Mar Biol 140:1249–1262
Elzinga CL, Salzer DW, Willoughby JW (1998) Measuring and monitoring plant populations. Technical Reference 1730-1. United States Department of the Interior, Bureau of Land Management, National Business Center, Denver
Gleason ML, Elmer DA, Pien NC, Fisher JW (1979) Effects of stem density upon sediment retention by salt marsh cord grass, *Spartina alterniflora* Loisel. Estuaries 2:271–274
Hettler WF Jr (1989) Nekton use of regularly-flooded salt-marsh cordgrass habitat in North Carolina, USA. Mar Ecol Prog Ser 56:111–118
Hudson SJ (2001) Challenges for environmental education: issues and ideas for the 21st century. Bioscience 51:283–288
Kastler JA, Wiberg PL (1996) Sedimentation and boundary changes of Virginia salt marshes. Estuar Coast Shelf Sci 42:683–700
Konisky RA, Burdick DM, Dionne M et al. (2006) A regional assessment of salt marsh restoration and monitoring in the Gulf of Maine. Restor Ecol 14:516–525
Knutson PL (1988) Role of coastal marshes in energy dissipation and shore protection. In: Hook DD et al (eds) The ecology and management of wetlands. Ecology of wetlands, vol 1. Crook Helm, London, pp 161–175
Leonard LA, Wren PA, Beavers RL (2002) Flow dynamics and sedimentation in *Spartina alterniflora* and *Phragmites australis* marshes of the Chesapeake Bay. Wetlands 22:415–424
MacIntyre HL, Geider RJ, Miller DC (1996) Microphytobenthos: the ecological role of the 'Secret Garden' of unvegetated, shallow-water marine habitats. I. Distribution, abundance and primary production. Estuaries 19:186–201
Markovchick-Nicholls L, Dean T, Bloch P et al. (2004) Sound stewardship-using citizen-based restoration monitoring as a tool for adaptive management. Proc 2003 Ga Basin/Puget Sound Res Conf.
Martin D, Bertasi F, Colangelo MA et al (2005) Ecological impact of coastal defense structures on sediment and mobile fauna: evaluating and forecasting consequences of unavoidable modifications of native habitats. Coast Eng 52:1027–1051
Meyer DL, Townsend EC, Thayer GW (1997) Stabilization and erosion control value of oyster clutch for intertidal marsh. Restor Ecol 5:3–99
Minello TJ, Zimmerman RJ (1992) Utilization of natural and transplanted Texas salt marshes by fish and decapod crustaceans. Mar Ecol Prog Ser 90:273–285
Minello TJ, Zimmerman RJ, Medina R (1994) The importance of edge for natant macrofauna in a created salt marsh. Wetlands 14:184–198
Minello TJ, Webb JW (1997) Use of natural and created *Spartina alterniflora* salt marshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. Mar Ecol Prog Ser 151:165–179
Mock CR (1966) Natural and altered estuarine habitats of penaeid shrimp. Proc Gulf Caribb Fish Inst 19:86–98
Morgan PA, Short FT (2002) Using functional trajectories to track constructed salt marsh development in the Great Bay Estuary, Maine/New Hampshire, U.S.A. Restor Ecol 10:461–473
Morris JT, Haskin B (1990) A 5-yr record of aerial primary production and stand characteristics of *Spartina alterniflora*. Ecology 71:2209–2217
Morris JT, Sundareshwar PV, Nietch CT et al. (2002) Responses of coastal wetlands to rising sea level. Ecology 83:2869–2877
National Research Council (2007) Mitigating shore erosion along sheltered coasts. The National Academies Press, Washington D.C.
Neckles HA, Dionne M, Burdick DM et al (2002) A monitoring protocol to assess tidal restoration of salt marshes on local and regional scales. Restor Ecol 10:556–563

Oscarson DB, Calhoun AJ (2007) Developing vernal pool conservation plans at the local level using citizen-scientists. Wetlands 27:80–95

Osgood DT, Zieman JC (1993) Spatial and temporal patterns of substrate physiochemical parameters in different-aged barrier island marshes. Estuar Coast Shelf Sci 37:421–436

Palmer MA, Bernhardt ES, Allan JD et al (2005) Standards for ecologically successful river restoration. J Appl Ecol 42:208–217

Peterson GW, Turner RE (1994) The value of salt marsh edge vs. interior as a habitat for fish and decapod crustaceans in a Louisiana tidal marsh. Estuaries 17:235–262

Piehler MF, Currin CA, Casanova R et al (1998) Development and N_2-fixing activity of the benthic microalgal community in transplanted *Spartina alterniflora* marshes in North Carolina. Restor Ecol 6:290–296

Plumb RH (1981) Procedures for handling and chemical analyses of sediment and water samples. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Race MS, Fonseca MS (1996) Fixing compensatory mitigation: what will it take? Ecol Appl 6:94–101

Rogers SM, Broome SW, Seneca ED (1992) Shoreline erosion control using marsh vegetation and low-cost structures. N C Sea Grant Publ UNC-SG-92-12, Raleigh

Sacco JN, Seneca ED, Wentworth TR (1994) Infaunal community development of artificially established salt marshes in North Carolina. Estuaries 17:489–500

SAS Institute (1999) SAS version 8.02. Cary, N.C.

Seitz RD, Lipcius RN, Olmstead NH et al. (2006) Influence of shallow-water habitats and shoreline development upon abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. Mar Ecol Prog Ser 3426:11–27

Sharpe A, Conrad CS (2006) Community based ecological monitoring in Nova Scotia: challenges opportunities. Environ Monit Assess 113:395–409

Simenstad CA, Thom RM (1996) Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. Ecol Appl 6:38–56

Stumpf RP (1983) The process of sedimentation on the surface of a salt marsh. Estuar Coast Shelf Sci 17:495–508

Thayer GW, McTigue TA, Salz RJ et al. (eds) (2005) Science-based restoration monitoring of coastal habitats, vol 2: Tools for monitoring coastal habitats. NOAA Coastal Ocean Program Decision Analysis Series No.23. NOAA National Centers for Coastal Ocean Science, Silver Spring, Md.

Thom RM (1997) System-development matrix for adaptive management of coastal ecosystem restoration projects. Ecol Eng 8:219–232

Weinstein MP, Kreeger DA (eds) (2000) Concepts and controversies in tidal marsh ecology. Kluwer, Dordrecht

Weinstein MP, Teal JM, Balletto JH et al (2001) Restoration principles emerging from one of the world's largest tidal marsh restoration projects. Wetlands Ecol Manage 9:387–407

Zedler JB, Calloway JC (1999) Tracking wetland restoration: do mitigation sites follow desired trajectories?. Restor Ecol 7:69–73

Zedler JB, Lindig-Cisneros R (2000) Functional equivalency of restored and natural salt marshes. In: Weinstein MP, Kreeger DA (eds) Concepts and controversies in tidal marsh ecology. Kluwer, Dordrecht, pp 565–582

Zedler JB (ed) (2001) Handbook for restoring tidal wetlands. CRC Press, Boca Raton