Quantifying Vegetation and Nekton Response to Tidal Restoration of a New England Salt Marsh

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Abstract

Tidal flow to salt marshes throughout the northeastern United States is often restricted by roads, dikes, impoundments, and inadequately sized culverts or bridge openings, resulting in altered ecological structure and function. In this study we evaluated the response of vegetation and nekton (fishes and decapod crustaceans) to restoration of full tidal flow to a portion of the Sachuest Point salt marsh, Middletown, Rhode Island. A before, after, control, impact study design was used, including evaluations of the tide-restricted marsh, the same marsh after reintroduction of tidal flow (i.e., tide-restored marsh), and an unrestricted control marsh. Before tidal restoration vegetation of the 3.7-ha tide-restricted marsh was dominated by *Phragmites australis* and was significantly different from the adjacent 6.3-ha *Spartina*-dominated unrestricted control marsh (analysis of similarities randomization test, $p < 0.001$). After one growing season vegetation of the tide-restored marsh had changed from its pre-restoration condition (analysis of similarities randomization test, $p < 0.005$). Although not similar to the unrestricted control marsh, *Spartina patens* and *S. alterniflora* abundance increased and abundance and height of *Phragmites* significantly declined, suggesting a convergence toward typical New England salt marsh vegetation. Before restoration shallow water habitat (creeks and pools) of the unrestricted control marsh supported a greater density of nekton compared with the tide-restricted marsh (analysis of variance, $p < 0.001$), but after one season of restored tidal flow nekton density was equivalent. A similar trend was documented for nekton species richness. Nekton density and species richness from marsh surface samples were similar between the tide-restored marsh and unrestricted control marsh. *Fundulus heteroclitus* and *Palaemonetes pugio* were the numerically dominant fish and decapod species in all sampled habitats. This study provides an example of a quantitative approach for assessing the response of vegetation and nekton to tidal restoration.

Key words: *Fundulus heteroclitus*, **nekton,** *Phragmites australis*, **restoration, Rhode Island, salt marsh,** *Spartina.*

Introduction

Restricting tidal flow by roads, bridges, dikes, and other structures can significantly alter the ecological structure and function of salt marshes. In New England altered hydrology of tide-restricted marshes has dramatic effects on sedimentation processes (Anisfeld et al. 1999), sediment chemistry and biogeochemical processes (Anisfeld & Benoit 1997; Portnoy 1999), water chemistry (Portnoy 1991), vegetation (Roman et al. 1984; Rozsa 1995), and nekton communities (Allen et al. 1994; Raposa & Roman 2001). Many efforts are now underway to reestablish natural tidal regimes to impacted salt marshes by removing dikes or replacing small culverts and bridges with larger openings. Increasing tidal exchange to tide-restricted salt marshes often results in restored ecological functions (Sinicrope et al. 1990; Barrett & Niering 1993; Peck et al. 1994; Roman et al. 1995; Burdick et al. 1997; Dionne et al. 1999; Fell et al. 2000; Warren et al. 2002, this issue; Raposa 2002 in press).

Restoration of salt marshes has rapidly accelerated over the past few decades with government agencies and conservation organizations leading initiatives and providing support. However, many restoration sites have limited quantitative data aimed at assessing ecological responses to restoration practices. Researchers and resource management professionals must acquire long-term data under pre-restoration, post-restoration, and reference marsh conditions to objectively evaluate if restoration is proceeding as expected and to assist with verification of models aimed at predicting marsh responses to tidal restoration (e.g., Boumans et al. 2002,

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this issue). Moreover, quantitative assessments are essential for understanding ecological processes of restoration and for assisting in the design and evaluation of future restoration efforts. Here we examine vegetation and nekton (fishes and decapod crustaceans) of a southern New England salt marsh under pre-restoration (tide-restricted) conditions and the initial 2 years under restored tidal conditions. The research design and associated statistical evaluations used in this study serve as a model for natural resource management efforts aimed at evaluating the response of vegetation and nekton to tidal restoration. Using a before, after, control, impact study design (Stewart-Oaten et al. 1986) we quantified vegetation composition and nekton use of a tide-restricted marsh before restoration and then evaluated the same marsh under tide-restored conditions (i.e., after). A tide-unrestricted control marsh was assessed to ensure that any changes detected from tide-restricted to restored conditions were due to the tidal restoration (i.e., impact) and not other factors.

Materials and Methods

Study Site

The Sachuest Point salt marsh (Middletown, RI; lat 41°29′N, long 71°15′W), associated with the U.S. Fish and Wildlife Service's Sachuest Point National Wildlife Refuge, is located behind an estuarine beach at the mouth of the Sakonnet River, a portion of the Narragansett Bay estuary (Fig. 1). An inlet through the beach provides tidal exchange to a 6.3-ha portion of the marsh, hereafter referred to as the unrestricted control marsh. A causeway bisecting the marsh, with only a 51-cm diameter culvert, allowed for limited tidal exchange to a 3.7-ha portion of the Sachuest Point marsh. Water elevation was monitored over several tidal cycles in 1996, prior to tidal restoration, and tidal range within the unrestricted control marsh varied from 0.25 to 0.50 m (Fig. 2a). Mean tidal creek salinity, recorded throughout the unrestricted marsh at low tide during August through October, was 12 ppt (range, 4–22 ppt). Before tidal restoration the causeway and culvert, which had become mostly filled with sediment, effectively impounded water and severely reduced tidal range within the restricted marsh to 1 to 4 cm (Fig. 2a).

In March 1998 tidal flow was restored to the restricted marsh by installing two 76-cm diameter culverts adjacent to the 51-cm culvert. With this installation tidal range in the unrestricted control marsh and tide-restored marsh was equivalent (Fig. 2b). The 3.7-ha marsh before the installation of new culverts is hereafter called the tide-restricted marsh. The same area is referred to as the tide-restored marsh after the 1998 cul-

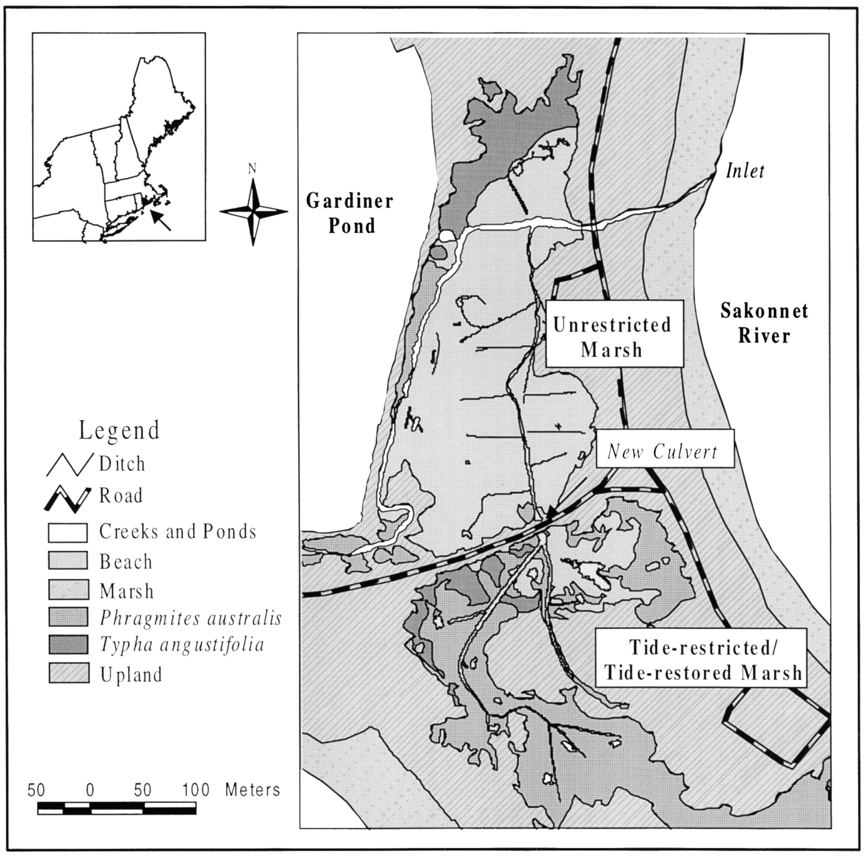

Figure 1. Sachuest Point Salt Marsh (Middletown, RI, U.S.A.) showing the unrestricted control marsh, tide-restricted/tide-restored marsh, and location of new culverts that allowed increased tidal flow to the tide-restored marsh. Gardiner Pond, a freshwater reservoir, is hydrologically isolated from the salt marsh by an earthen dike.

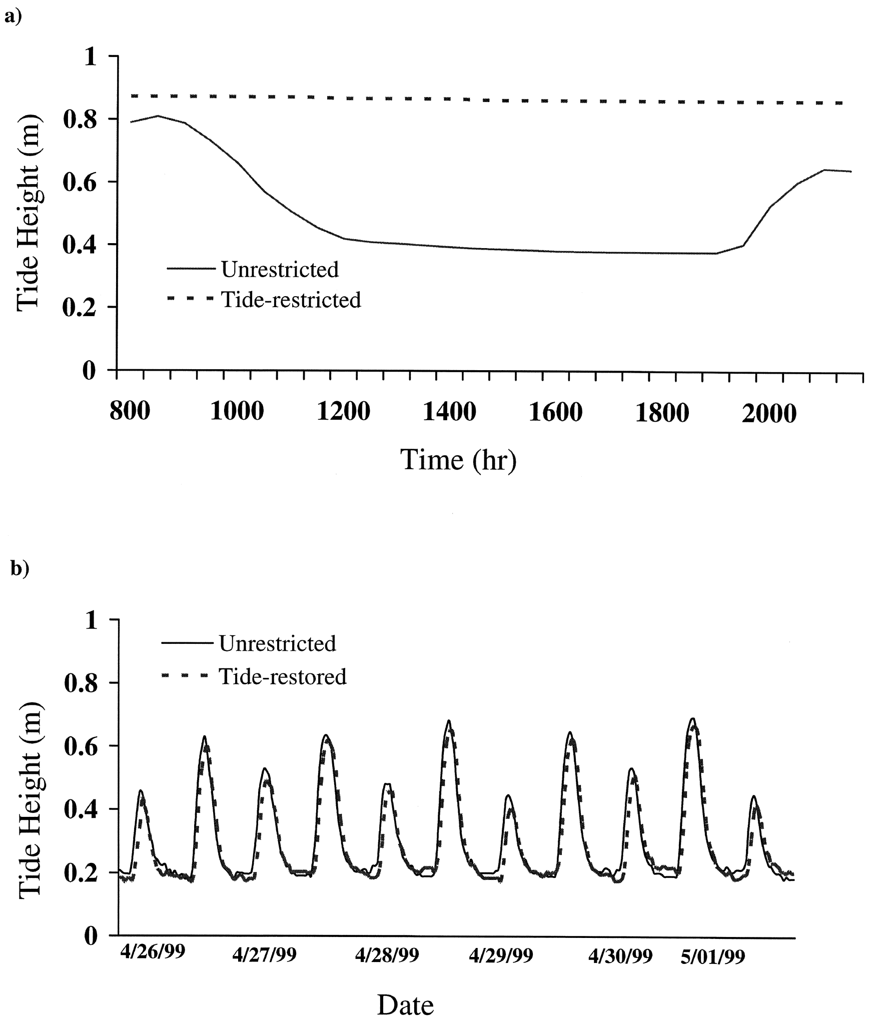

Figure 2. (a) Water elevation of the unrestricted and tide-restricted marsh creeks measured over one tidal cycle before restoration activities. Data from November 1996. (b) Water elevation of the unrestricted and tide-restored marsh creeks for several tidal cycles in April 1999 after installation of new culverts.

vert installation. Tidal creek salinity in the tide-restored marsh averaged 18 ppt (range, 13–26 ppt).

Associated with the restoration activities, new tidal creeks and marsh pools were created in the tide-restored marsh and the unrestricted control marsh. Before restoration there were 3,220 m^2 of shallow water habitat within the unrestricted control marsh. This increased in 1998 (to 3,815 m^2) with creation of pools and creeks in conjunction with open marsh water management activities, a practice of mosquito control described by Ferrigno and Jobbins (1968). Alterations to the unrestricted control marsh were minimal in spatial extent (the new pools and creeks represented less than 1% of the 6.3-ha marsh), and thus use of this marsh as a control was probably not compromised. In the tide-restricted marsh there were only 490 m^2 of shallow water habitat, but after tidal restoration and the creation of pools and creeks open water habitat increased by almost five times (2,360 m^2).

Based on aerial photographic analysis and associated vegetation mapping the tide-restricted marsh was dominated by *Phragmites australis* (common reed) before tidal restoration, covering almost 75% of the 3.7-ha system. *Phragmites* is an invasive grass that often proliferates in tide-restricted New England salt marshes (Roman et al. 1984). Remnants of salt marsh (<15%), dominated by *Spartina patens* (salt hay) and *S. alterniflora* (saltwater cordgrass), also were present. In contrast, almost 80% of the unrestricted control marsh was dominated by species typical of southern New England salt marshes (e.g., *S. alterniflora*, *S. patens*, *Juncus gerardii* [blackgrass], *Iva frutescens* [marsh elder]) (Niering & Warren 1980).

Restoration of the Sachuest Point salt marsh was intended to enhance finfish nursery habitat damaged by the 1989 M/V *World Prodigy* oil spill in Narragansett Bay. The spill did not affect the Sachuest Point Marsh, but the site was selected to compensate for finfish injuries related to the spill.

Vegetation

To provide quantitative data on changes in plant species composition and relative abundance, permanent vegetation plots were sampled within the unrestricted control marsh and the tide-restricted/tide-restored marsh. Data were collected toward the end of the growing season (August to early October) in 1996 (pre-restored tidal conditions), 1998 (one growing season of restored tidal conditions) and 1999 (two growing seasons). Transects, extending from creekbanks to the upland border, were randomly located within the unrestricted control and tide-restricted/tide-restored marsh. Vegetation plots (1 m^2) were located systematically along each transect, with the first plot randomly located and others spaced at equal intervals from the first ($n = 22$ plots in unrestricted control marsh, $n = 28$ in tide-restricted/tide-restored marsh). Within each 1-m^2 plot the percent cover of each species present was ranked by visual inspection according to the Braun-Blanquet scale (Kent & Coker 1992): less than 1 to 5%, 6 to 25%, 26 to 50%, 51 to 75%, and more than 75%. Other cover-type categories such as bare and litter were used as appropriate.

A nonparametric multivariate permutation procedure was used to analyze vegetation data (PRIMER statistical package, Carr 1997; Clarke & Warwick 1994). Using a Euclidean distance similarity matrix this method allows the objective identification of sample plots that have similar (or dissimilar) vegetation in terms of species composition and abundance. An analysis of similarities randomization test (ANOSIM) is applied to the matrix to test for significant differences between groups of sample plots. ANOSIM is a nonparametric analog to multivariate analysis of variance (Clarke & Green 1988). Nine pair-wise comparisons between groups of sample plots that were defined a priori were evaluated to detect differences in vegetation between the unrestricted control, tide-restricted, and tide-restored marshes over time. For example, all 28 tide-restricted marsh plots in 1996 were compared with all 28 tide-restored marsh plots in 1998. Bonferroni adjustment was applied to the alpha level. For pair-wise comparisons that were significant or had dissimilar vegetation communities we calculated the proportion of the overall dissimilarity that was contributed by each individual cover type:

$$1 - \frac{D}{D_{\max}} = 1 - \frac{(C_{1i} - C_{2i})^2}{\Sigma(C_{1i} - C_{2i})^2}$$

where D is the distance, C_{1i} is the cover of species i in marsh 1, and C_{2i} is the cover of species i in marsh 2. The outcome is a list of cover types or species ranked in order of their percent contribution to the dissimilarity between significant pair-wise comparisons. $D_{\max}$ provides an overall measure of dissimilarity for each pair-wise comparison.

With tidal restoration it was hypothesized that the height of *Phragmites* would be stunted, and thus in each 1-m^2 plot containing *Phragmites* the maximum height of randomly selected stems was measured (12 stems in 1996; 16 stems in 1998, 1999, and 2000). If a plot contained fewer than the target number of stems, then all were measured. Comparisons of *Phragmites* stem heights among years was analyzed by one-way analysis of variance (ANOVA) and significantly different means evaluated by the least-squares means test.

Polyvinyl chloride wells (3.8 cm diameter, 50 cm length) were installed at a subset of the vegetation plots to measure groundwater elevation and porewater salinity. Before sampling the wells were pumped, allowed to refill, and then salinity was measured with a hand-held salinity refractometer (model A366ATC, Vista). Groundwater level was measured in centimeters below the marsh surface. Measurements were taken near low tide on several dates in 1996, 1998, and 1999. If wells were dry during sampling, depth to water table was recorded as greater than 45 cm below the marsh surface.

Nekton Sampling

Nekton was collected in tidal creeks and marsh pools with a throw trap (1 $m^2 \times 0.5$ m high), after Kushlan (1981). The trap frame was constructed of narrow aluminum bars covered on four sides with 3-mm mesh hardware cloth and open at the top and bottom. Permanent sampling stations were randomly established in both the unrestricted control marsh and tide-restricted/tide-restored marsh. Under pre-restoration conditions in 1997, 10 permanent sampling stations were randomly selected in the unrestricted marsh and 10 stations in the tide-restricted marsh. In 1998 and 1999 restoration activities included the creation of new marsh pools and tidal creeks, and thus the number of sampling stations increased ($n = 20$ unrestricted marsh; $n = 20$ tide-restored marsh).

Throw trap sampling was conducted monthly during the period of peak abundance (August, September, October) in 1997, 1998, and 1999. All sampling occurred near low tide after the marsh surface had drained. Samples were collected by approaching each station from the marsh surface and then throwing the trap into the creek or marsh pool. To minimize disturbance of the nekton the investigator would quietly wait on the marsh surface for 2 minutes, adjacent to the sampling station, before throwing the trap 4 to 5 m into the water. Captured nekton was removed from the trap with a large dip net (1 m wide $\times$ 0.5 m; 1-mm mesh) that fit snugly in the trap. All nekton was considered collected when three consecutive scoops with the large dip net were empty. All individuals were enumerated by species in the field.

For each parameter evaluated (species density, species richness, community composition), there were six treatments encompassing the 3 sample years and the unrestricted (U) control marsh and tide-restricted/tide-restored (R) marsh (97U, 98U, 99U, 97R, 98R, 99R). Comparisons were made to contrast the unrestricted control marsh and tide-restricted or tide-restored marsh in each year, to assess changes in the tide-restricted/tide-restored marsh over time, and to evaluate changes in the unrestricted marsh over time as a control. Nekton density data were log (× + 1) transformed and then analyzed by two-way ANOVA, using the six treatments and month as factors. Nine a priori comparisons were evaluated by the Student-Newman-Keuls multiple range test (97U vs. 98U, 97U vs. 99U, 98U vs. 99U, 97R vs. 98R, 97R vs. 99R, 98R vs. 99R, 97U vs. 97R, 98U vs. 98R, 99U vs. 99R). Species richness was calculated according to Heltshe and Forrester (1983), and a priori pair-wise comparisons were evaluated by Student's *t*-test using a Bonferroni adjusted alpha level. Nine pair-wise comparisons of nekton community composition were evaluated by two-way ANOSIM using a Bray-Curtis similarity matrix, with Bonferroni adjusted alpha levels. To determine the individual species that contributed to nekton community differences similarity percentages were calculated. Similarity percentage provides an overall measure of dissimilarity for each comparison and generates a list of species ranked in order of their percent contribution to that dissimilarity.

Nekton use of the marsh surface habitat was evaluated using bottomless lift nets (Rozas 1992). Sampling was conducted monthly from June through October in the unrestricted control marsh and the tide-restored marsh during 1998 and 1999. No pre-restoration data were collected because site conditions were not conducive to lift net use. To be used effectively the bottomless lift net requires that the marsh surface is flooded and then drained by tidal action, but under pre-restoration conditions tidal range in the tide-restricted marsh was minimal (Fig. 2a). On each sampling date five randomly selected permanent stations were sampled in the unrestricted marsh and five in the tide-restored marsh. All sampling stations were within 1 m of a creekbank to ensure flooding of the marsh surface. Each sampling station included a 6-m^2 area surrounded by netting (3-mm mesh) buried in the sediment flush with the marsh surface. When the marsh surface was flooded at high tide the net was quickly pulled up from the sediment, thereby trapping all nekton within the enclosed area. As the marsh surface drained on the ebb tide nekton would migrate to a sump (23 × 23 cm) located within the 6-m^2 area. All fish and crustaceans within the sump and those occasionally collected outside the sump but still within the 6-m^2 area were enumerated by species.

Results and Discussion

Vegetation Responses to Restoration

After just one growing season of restored tidal conditions vegetation of the tide-restored marsh (1998) was significantly different from the same marsh under tide-restricted conditions in 1996 (Table 1). There were no significant vegetation changes in the tide-restored marsh between 1998 and 1999, indicating that vegetation changes slowed after the rapid response that occurred during the first growing season of restored tidal conditions. However, vegetation in the unrestricted marsh (i.e., control of the before, after, control, impact model) remained similar among sampling years, suggesting that the plant community changes in the tide-restored marsh were related to tidal restoration activities.

Dissimilarity in vegetation from 1996 pre-restoration conditions in the tide-restricted marsh to the 1998 tide-restored state was mostly attributed to an increase in relative abundance of bare areas, whereas the salt marsh species, *S. patens*, *S. alterniflora*, and *Salicornia europaea* (saltwort), increased (Table 2). As expected, *Typha angustifolia* (narrow-leaved cattail) declined in abundance, a species less tolerant of saline conditions. The increase in bare areas resulted mostly from removal of a small berm and activities of mechanical equipment used to create creeks and pools in the tide-restored marsh. As vegetation colonized these regions of the marsh the relative cover of bare areas decreased during the second year of restoration, dead *Phragmites* increased, and restoration of the *Spartina* community continued (Table 2).

Vegetation of the tide-restored Sachuest Point salt marsh after 2 years of increased tidal flow remained significantly different from the unrestricted control marsh (Table 1). However, the vegetation was responding in a trajectory toward the typical New England salt marsh as represented by the unrestricted marsh. D_{max}, a measure of dissimilarity based on the Euclidean distance metric, was 16.5 in 1996 when comparing vegetation of the unrestricted control marsh and tide-restricted marsh but decreased to 14.5 in 1999, indicating a convergence of the tide-restored marsh vegetation toward that of the unrestricted control marsh. As D_{max} diminishes toward zero the marshes are becoming more similar.

Table 1. Results of one-way ANOSIM tests on pair-wise comparisons of vegetation data.

Comparisons	*p Value*
Tide-restricted/tide-restored marsh	
Restricted 1996 vs. restored 1998	0.004
Restricted 1996 vs. restored 1999	0.001
Restored 1998 vs. restored 1999	NS
Unrestricted control marsh	
1996 vs. 1998	NS
1996 vs. 1999	NS
1998 vs. 1999	NS
Unrestricted vs. tide-restricted or tide-restored	
Unrestricted vs. restricted 1996	0.001
Unrestricted vs. restored 1998	0.001
Unrestricted vs. restored 1999	0.001

Bonferroni adjusted alpha = 0.05/9 = 0.0055.

Table 2. Individual cover types contributing most to dissimilarities noted between years for the tide-restricted (1996) and then tide-restored marsh (1998, 1999).

	Average Rank		
Cover Type	*Tide-restricted 1996*	*Tide-restored 1998*	*Percent Contribution*
Bare	1.2	3.7	60.1
Phragmites australis (dead)	0.1	0.9	6.6
Iva frutescens (seedlings)	0.9	0.2	4.6
Spartina patens	2.3	3.0	4.6
Salicornia europaea	0.3	1.0	4.5
Typha angustifolia	1.2	0.6	3.9
Spartina alterniflora	0.5	1.0	2.9
	Tide-restricted 1996	*Tide-restored 1999*	
Bare	1.2	3.4	45.4
Phragmites australis (dead)	0.1	1.4	15.2
Typha angustifolia	1.2	0.2	9.2
Spartina patens	2.3	3.2	7.4
Spartina alterniflora	0.5	1.3	5.9
Iva frutescens	0.0	0.6	3.8
Phragmites australis	3.2	2.6	3.6

Percent cover data are presented as the average of the five ranked cover classes (e.g., 1 = <1–5% cover, 2 = 6–25%, 3 = 26–50%, 4 = 51–75%, 5 = >75%). Cover types are ranked by the percent that each cover type contributes to the dissimilarity indicated. Cover types cumulatively contributing up to 90% of the dissimilarity are presented.

A goal of salt marsh restoration in the northeastern United States often targets a reduction in *Phragmites* cover and height. At Sachuest Point, in addition to the noted decline in *Phragmites* cover with tide-restored conditions, average height also declined significantly after just one growing season (Fig. 3). The stunting of *Phragmites* continued after two growing seasons of tidal restoration. In 2000, three seasons after restoration, the height decline did not continue.

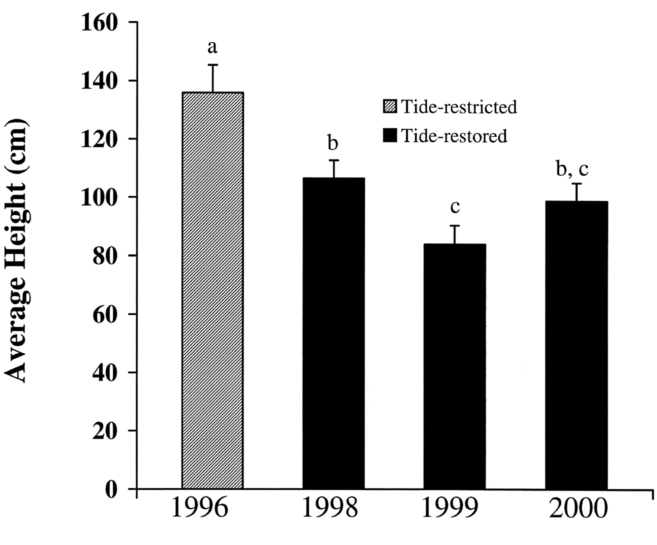

Figure 3. Average height (± SE) of *Phragmites australis* from the 1996 tide-restricted marsh and the same marsh but tide-restored in 1998, 1999, and 2000. Analysis by one-way ANOVA ($p < 0.0001$) with different letters indicates significantly different means by least-squares means test ($p < 0.05$).

Increased porewater salinity is a factor contributing to the reduction in *Phragmites* height (Hellings & Gallagher 1992). Before reintroduction of tidal flow, porewater salinity of the restricted marsh averaged 18 ± 9 ppt (± SD) compared with 26 ± 4 ppt in the 1999 tide-restored marsh (ANOVA, $p < 0.10$). Porewater salinity data for 2000 are not available, but it is noted that the year-to-year decline in average *Phragmites* height did not occur in 2000, perhaps due to a growing season with more rainfall than the previous year (average rainfall from May to August: 1999, 22 cm; 2000, 37 cm). In addition to increased porewater salinity sulfide toxicity from anoxic waterlogged sediments can stress *Phragmites* (Chambers et al. 1998). Mean water table level of the tide-restricted marsh was just 6 ± 6 cm below the marsh surface, suggesting waterlogged conditions. However, drainage within the sandy salt marsh peat improved after reintroduction of tidal flow and resulted in a lowered water table (43 ± 8 cm below the marsh surface; ANOVA, $p < 0.0001$). Given reduced soil waterlogging and assumed reduction in sulfide toxicity stress it will be interesting to evaluate how *Phragmites* responds over the long term. In nearby Connecticut tide-restored marshes with water table levels nearer the surface seem to have a more rapid conversion from *Phragmites*-dominated to *Spartina* marsh when compared with marshes with deeper mean water table levels (Warren et al. 2002, this issue).

In summary after just 2 years of restored tidal exchange vegetation of the tide-restored Sachuest Point salt marsh is developing toward the typical pattern of a southern New England marsh. *Phragmites* cover and height were reduced, whereas *S. patens* and *S. alterniflora* increased in abundance (Fig. 3, Table 2). This represents just the beginning of a restoration process that may proceed for decades. For example, at a Connecticut site vegetation data collected 4 and 10 years after reintroduced tidal flow revealed a progressive conversion of an impounded *Typha angustifolia* marsh to short-form *S. alterniflora* and other common salt marsh plants, with vegetation recovery after 10 years still ongoing (Sinicrope et al. 1990). In southern Maine Burdick et al. (1997) monitored vegetation plots for 8 years after tidal restoration and found conversion of a *Typha latifolia* (common cattail) and *Spartina pectinata* (prairie cordgrass) brackish marsh to a *S. alterniflora*-dominated marsh. Rozsa (1995) reported a 40-year process of vegetation change from intertidal flats to low *S. alterniflora* marsh after removal of tide-restricting gates in a Long Island Sound marsh. Subtle changes in vegetation of a coastal New Hampshire freshwater impounded meadow toward *S. alterniflora* and high marsh species were observed after only 2 years of tidal restoration (Burdick et al. 1997), but similar to Sachuest Point it is expected that vegetation change in response to tidal reintroduction will continue.

Nekton Responses to Restoration

Fourteen fish species and 4 decapod species were collected from creeks and pools of the Sachuest Point salt marsh (Table 3). In 1997, before restoration, average density of nekton in shallow water habitats (creeks and pools) of the unrestricted control marsh was dramatically higher than in the tide-restricted marsh (26.8 vs. 4.9 individuals/m^2; ANOVA, $p < 0.001$; Fig. 4a). Species richness was also greater in the unrestricted marsh (t-test, $p < 0.005$; Fig. 5). Regarding the nekton community nine pairwise comparisons evaluated by ANOSIM revealed a significant difference only between the unrestricted control and tide-restricted marshes. Similarity percentage analysis indicated that this difference is because three of the most common species (*Fundulus heteroclitus*, *Palaemonetes pugio*, *Menidia menidia*) had higher densities in the unrestricted marsh (Table 3), accounting for 64% of the dissimilarity that was detected by ANOSIM.

The nekton community quickly responded to reintroduction of tidal flow to the restricted marsh and creation of additional marsh pools and tidal creeks. Average nekton density and species richness increased significantly on the tide-restored marsh (Fig. 4a & 5). Comparisons of data between the unrestricted and tide-restored marshes in 1998 and 1999 provides further evidence of the rapid response by nekton as total density, species richness, and community composition were all equivalent.

The nekton density data in Table 3 are the number of individuals occupying 1 m^2 of shallow water habitat (creeks and marsh pools), but it is interesting to evaluate the nekton community response to tidal restoration

Table 3. Mean nekton density (number of individuals/m^2 ± SE in parentheses) in creeks and pools of the unrestricted marsh and tide-restricted/tide-restored Sachuest Point salt marsh from 1997 to 1999.

	Unrestricted			*Tide-Restricted*	*Tide-Restored*	
Species (Common Name)	*1997* $n = 30$	*1998* $n = 60$	*1999* $n = 60$	*1997* $n = 30$	*1998* $n = 60$	*1999* $n = 60$
Fundulus heteroclitus (mummichog)	19.83 (7.36)	10.62 (2.17)	12.05 (1.99)	4.03 (1.02)	13.02 (3.11)	14.67 (2.79)
Cyprinodon variegatus (sheepshead minnow)	0.20 (0.10)	1.27 (0.62)	2.52 (0.97)	0.60 (0.23)	0.77 (0.32)	2.60 (0.73)
Anguilla rostrata (American eel)	0.10 (0.07)	0.03 (0.02)	0.27 (0.07)	0.03 (0.03)	0.02 (0.02)	0.23 (0.07)
Palaemonetes pugio (daggerblade grass shrimp)	2.93 (1.04)	7.68 (3.69)	6.57 (2.79)	0.27 (0.15)	1.62 (0.63)	3.97 (1.51)
Carcinus maenas (green crab)	0.17 (0.08)	0.18 (0.08)	0.07 (0.03)	0	0.02 (0.02)	0
Crangon septemspinosa (sand shrimp)	0.03 (0.03)	0.02 (0.02)	0	0	0.02 (0.02)	0
Fundulus majalis (striped killifish)	0.70 (0.35)	0.17 (0.06)	0.55 (0.22)	0	0.08 (0.04)	0.02 (0.02)
Mugil curema (white mullet)	1.63 (1.53)	0.05 (0.05)	0.12 (0.09)	0	0	0
Menidia menidia (Atlantic silverside)	1.23 (0.41)	0.20 (0.10)	0.07 (0.05)	0	0	0
Menidia beryllina (inland silverside)	0	0.35 (0.13)	0.52 (0.21)	0.03 (0.03)	0.40 (0.13)	0.42 (0.15)
Callinectes sapidus (blue crab)	0	0.03 (0.03)	0.20 (0.06)	0	0.05 (0.03)	0.02 (0.02)
Alosa aestivalis (blueback herring)	0	0	0	0	0.02 (0.02)	0
Pungitius pungitius (ninespine stickleback)	0	0	0	0	0.05 (0.03)	0
Notropis sp. (shiner)	0	0	0	0	0.03 (0.03)	0.02 (0.02)
Brevoortia tyrannus (Atlantic menhaden)	0	0	0.10 (0.10)	0	0	0
Centropristis striata (black sea bass)	0	0	0.03 (0.02)	0	0	0.37 (0.15)
Lucania parva (rainwater killifish)	0	0	0	0	0	0.03 (0.03)
Gobiosoma ginsburgi (seaboard goby)	0	0	0.37 (0.18)	0	0	0
Total fishes	23.70 (7.51)	12.68 (2.28)	16.58 (2.66)	4.69 (1.01)	14.40 (3.14)	18.35 (2.86)
Total decapods	3.13 (1.03)	7.92 (3.69)	6.83 (2.80)	0.27 (0.15)	1.70 (0.65)	3.98 (1.51)
Total nekton	26.83 (7.91)	20.60 (4.12)	23.42 (3.81)	4.93 (1.09)	16.17 (3.19)	22.33 (3.24)

Nekton was collected using a 1-m^2 throw trap from August through October of each year.

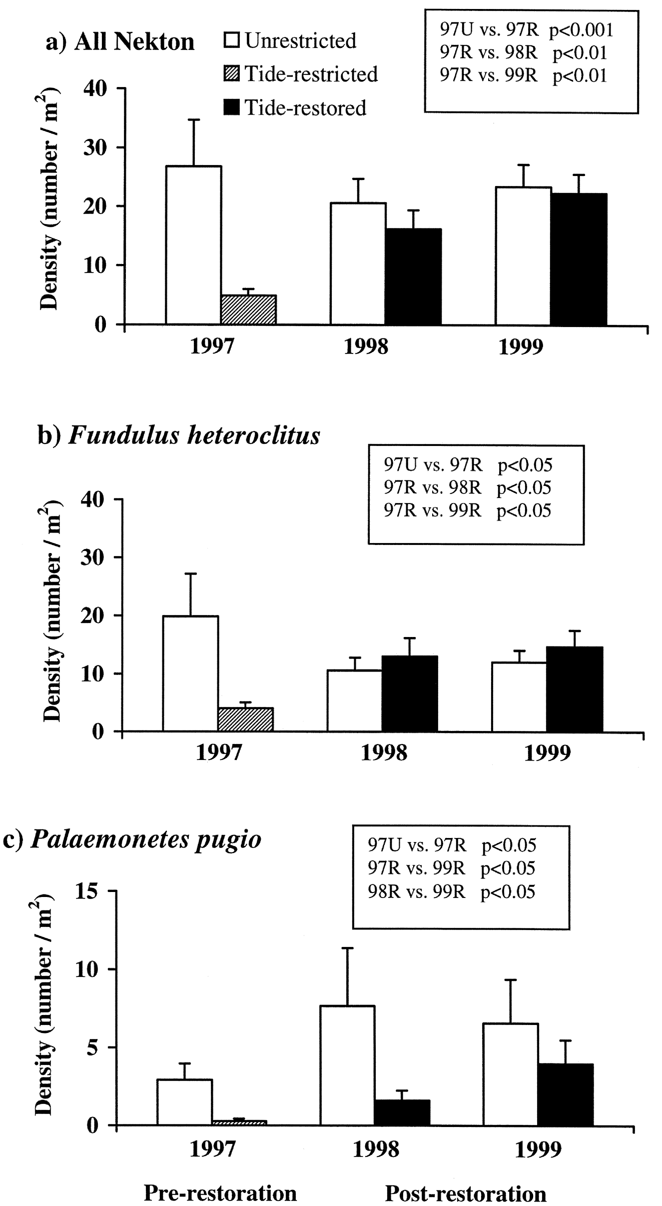

Figure 4. Density of nekton (number of individuals/m^2) from shallow water habitats (marsh creeks and pools) for the unrestricted control marsh, tide-restricted marsh, and tide-restored marsh. Pre-tide restoration conditions are 1997 and post-restoration, 1998 and 1999. Data were collected using a 1-m^2 throw trap from August to October of each year. Analysis by two-way ANOVA with significant comparisons by SNK multiple range test. (a) All nekton combined, (b) *Fundulus heteroclitus*, and (c) *Palaemonetes pugio.*

when accounting for all available shallow water habitat within the unrestricted control and tide-restricted/tide-restored marshes. An estimate of nekton utilization by the total number of individuals was calculated by multiplying total nekton density, on a 1-m^2 basis (from Table 3), by the area of available shallow water habitat.

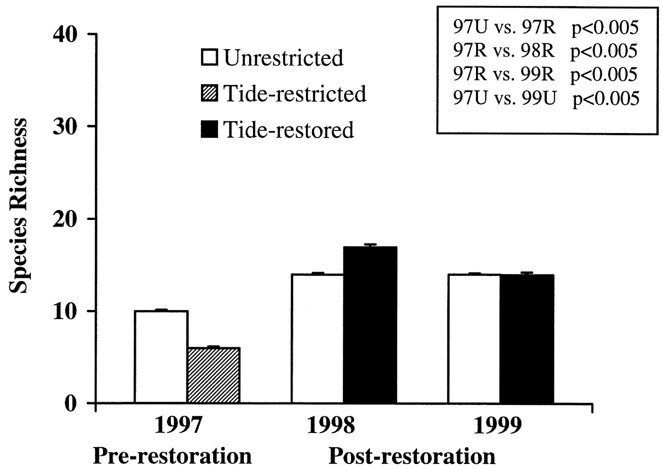

Figure 5. Species richness (± SE) of nekton from shallow water habitats (marsh creeks and pools) for the unrestricted control marsh, tide-restricted marsh, and tide-restored marsh. Data were collected using a 1-m^2 throw trap from August to October of each year. Significant a priori pair-wise comparisons evaluated by *t*-test are indicated (Bonferroni adjusted alpha = 0.0055).

Assuming a homogeneous distribution of nekton throughout open water habitat, the total number of nekton using the unrestricted control marsh was similar throughout the 3-year study period, despite a modest increase in available habitat in 1998 and 1999 (Fig. 6). In contrast, there was a dramatic increase in nekton using shallow water of the tide-restored marsh after reintroduction of tidal flow and creation of pools and creeks. As noted previously shallow water habitat increased by almost five times in the tide-restored marsh.

Regarding individual species, *F. heteroclitus* clearly dominated trends in nekton density (Table 3, Fig. 4b). Density did not differ significantly in the unrestricted marsh for the 3 sampling years. In 1997 density of *F. heteroclitus* in the tide-restricted marsh was less compared with the unrestricted marsh; however, with tidal restoration the density of *F. heteroclitus* became similar to the unrestricted control marsh. The dominant decapod, *P. pugio,* was slower to respond to tidal restoration (Fig. 4c). Density in the tide-restored marsh did not increase until the second year of restoration (1999).

Nekton species composition in shallow water habitats changed notably after restoration activities (Table 3). The significant increase in species richness in the tide-restored marsh (Fig. 5) was due to an influx of species that were not present in 1997. Installation of the new culverts and restored tidal flow enhanced access to the once tide-restricted marsh by several species, such as *Fundulus majalis*, *Carcinus maenas*, and *Crangon septemspinosa* and some commercially/recreationally recognized species like *Callinectes sapidus*, *Alosa aestivalis*, and *Centropristis striata.*

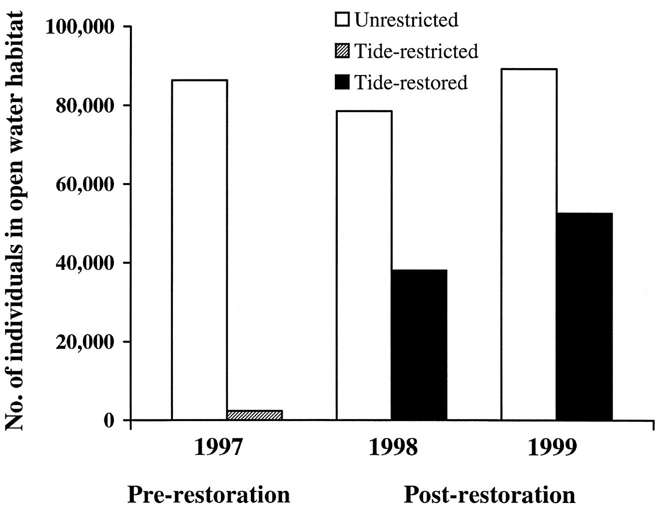

Figure 6. Estimated total number of individual nekton utilizing all shallow water habitats (creeks and pools) on the unrestricted control marsh, tide-restricted marsh, and tide-restored marsh. Nekton density (individuals/m^2) multiplied by total area of creeks and pools was used to derive these estimates.

In addition to shallow water habitats the marsh surface is an important habitat for some nekton. Six fish and four decapod species were collected with the bottomless lift net apparatus from the unrestricted and tide-restored marsh surfaces (Table 4). *Fundulus heteroclitus* and *P. pugio* were numerically dominant, comprising 98% of total nekton density. Average nekton density was statistically similar between the unrestricted marsh and tide-restored marsh for both years after tidal restoration (Fig. 7a). In 1998, during the first year of restoration, species richness between the unrestricted marsh and tide-restored marsh was equivalent, but during the second year it is noted that species richness of nekton utilizing the marsh surface was greater for the tide-restored marsh (Fig. 7b). These results on density and richness suggest a strong similarity between nekton utilizing the unrestricted and tide-restored marsh surfaces; however, it is noted that sampling with the bottomless lift net occurred within 1 m of creekbanks and not from interior portions of recovering *Phragmites* marsh.

Quantitative data sets on nekton use of both marsh surface and shallow water habitats of newly restored salt marshes in New England are quite limited. At a New Hampshire restoration site fish density on the surface of a tide-restored marsh was equivalent to a reference marsh after just 1 month of reintroduced tidal flow (Burdick et al. 1997). Studying a Rhode Island salt marsh, Raposa (2002 in press) reported a rapid colonization of common nekton species after tidal restoration. Not surprisingly, this quick colonization by fish in tidally restoring marshes has been noted in other geographic regions, such as the Pacific Northwest coast (Simenstad & Thom 1996) and the Delaware Bay (Able et al. 2000).

Based on comparisons of nekton density and species richness between the tide-restored and unrestricted portions of the Sachuest Point marsh, it is apparent that nekton communities are becoming progressively similar. Despite this short-term response of marsh nekton we cannot conclude that complete restoration of the marsh fauna has been achieved. For example, after 12 years of tidal restoration at a Connecticut marsh, densities of *Melampus bidentatus* (salt marsh snail) remained significantly lower than at a reference marsh (Fell et al. 1991; Peck et al. 1994). Documenting equivalent densi-

Table 4. Nekton density (number/m^2 ± SE in parentheses) from the marsh surface of the unrestricted and tide-restored Sachuest Point salt marsh in 1998 and 1999.

	Density (number/m^2)			
	Unrestricted		*Tide-restored*	
Species	*1998 n = 25*	*1999 n = 25*	*1998 n = 25*	*1999 n = 25*
Fundulus heteroclitus	0.65 (0.11)	0.74 (0.22)	4.23 (1.44)	1.57 (0.54)
Palaemonetes pugio	0.75 (0.47)	0.61 (0.38)	0.68 (0.37)	0.44 (0.21)
Carcinus maenas	0.17 (0.05)	0.11 (0.04)	0.02 (0.02)	0.03 (0.02)
Fundulus majalis	0.01 (0.01)	0.01 (0.01)	0.01 (0.01)	0.01 (0.01)
Cyprinodon variegatus	0	0.01 (0.01)	0.01 (0.01)	0.06 (0.04)
Anguilla rostrata	0.04 (0.01)	0	0	0
Uca pugnax	0.30 (0.07)	0.13 (0.03)	0	0.06 (0.02)
Mugil curema	0	0	0	0.02 (0.02)
Callinectes sapidus	0	0	0	0.01 (0.01)
Centropristis striata	0	0	0	0.02 (0.01)
Total fish	0.70 (0.11)	0.77 (0.22)	4.25 (1.44)	1.68 (0.56)
Total decapods	1.22 (0.48)	0.85 (0.40)	0.69 (0.36)	0.53 (0.22)
Total nekton	1.93 (0.53)	1.62 (0.42)	4.94 (1.48)	2.22 (0.68)

Nekton was collected using 6-m^2 bottomless lift nets from June through October of each year.

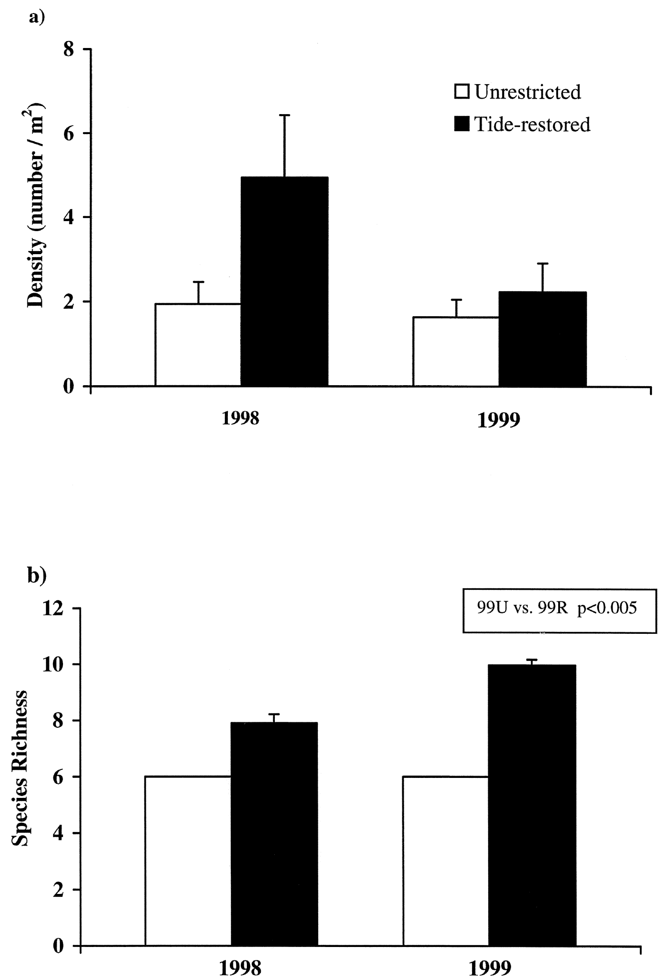

Figure 7. (a) Density (± SE) and (b) species richness (± SE) of nekton collected from the marsh surface with bottomless lift nets. Data for the unrestricted control and tide-restored marshes are presented for 2 years. No significant differences in density were detected by ANOVA. For species richness significant a priori pair-wise comparisons and significance level as evaluated by *t*-test are indicated. Bonferroni adjusted alpha = 0.0125.

ties and species composition of nekton within control and tide-restored marshes represent excellent metrics for evaluating restoration success, but, as concluded by Dionne et al. (1999), it is equally important to assess factors such as growth and survival of nekton species and availability of prey. A recent study of *F. heteroclitus* gut content conducted at the Sachuest Point salt marsh (James-Pirri et al. 2001) found that the tide-restored marsh provided similar food resources when compared with the unrestricted control after just 1 year of tidal restoration. Conversely, gut content studies on *F. heteroclitus* from Connecticut marshes suggest that it may be a decade or more for tide-restored and reference marshes to attain foraging equivalence (Warren et al. 2002, this issue).

Quantifying Restoration Responses

Given the emergence of coastal restoration initiatives over the past few decades from government agencies and conservation organizations, it is surprising that the body of literature documenting restoration responses is not richer. Quantifying restoration is a progressive process that should incorporate pre-restoration assessments, followed by monitoring of initial and longer term responses and inclusion of reference sites. As demonstrated by the quantitative data from the Sachuest Point salt marsh, statements on the trajectory of restoration can be made with a degree of scientific certainty, not merely anecdotally. Vegetation of the tide-restored marsh has responded to enhanced tidal flow with an increase in *S. patens* and *S. alterniflora* abundance and a corresponding decrease in *Phragmites* abundance and height. However, vegetation of the tide-restored marsh remains quite different when compared with the unrestricted control marsh, but they are converging toward similar communities. After just 1 year of restoration the density, species richness, and community composition of fishes and decapods in the tide-restored marsh were similar to the unrestricted control marsh.

Numerous study designs, field methods, and data analysis techniques are available to detect trends in salt marsh vegetation and nekton. Factors to be considered when developing long-term studies to evaluate restoration responses should incorporate careful study design to enable comparisons before and after restoration activities, including reference sites; use of quantitative sampling gear, such as throw traps and bottomless lift nets; and statistical tests that evaluate the composition of entire communities (e.g., nonparametric permutation testing), including a priori establishment of hypotheses to be tested. Quantitative approaches to assess vegetation, nekton, and other salt marsh components will enhance our understanding of the processes that control salt marsh restoration and our ability to predict restoration responses.

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