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Source: Journal of Coastal Research, 2008(10055) : 180-192

Published By: Coastal Education and Research Foundation

URL: <https://doi.org/10.2112/SI55-015>

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Early Ecological Responses to Hydrologic Restoration of a Tidal Pond and Salt Marsh Complex in Narragansett Bay, Rhode Island

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ABSTRACT

RAPOSA, K. B., 2008. Early ecological responses to hydrologic restoration of a tidal pond and salt marsh complex in Narragansett Bay, Rhode Island. *Journal of Coastal Research*, SI(55), 180–192. West Palm Beach (Florida), ISSN 0749-0208.



Tidal exchange was restored to the flow-restricted, 2.3-ha Potter Pond salt marsh on Prudence Island in the Narragansett Bay National Estuarine Research Reserve in April 2003. Ecological monitoring was conducted for 1 year before and 2 years after restoration to quantify ecological changes. Simultaneous monitoring was conducted in a nearby marsh that served as an experimental control. Tidal restoration increased the tide range in Potter Pond from approximately 4 cm in 2000 to 120 cm in 2003. After 2 years of restoration, the height of *Spartina alterniflora* remained unchanged, and the same was true of the composition of the emergent marsh vegetation community. However, by 2004, the percent cover of live *Phragmites australis* decreased by 69%, and the average height of *Phragmites* decreased by 76 cm. Seven additional bird species were observed at Potter Pond after 1 year of restoration, and the number of birds observed increased from 6 to 85 per viewing effort, mostly due to large numbers of shorebirds using the newly exposed mud flats. Nekton density decreased from 100 m⁻² to 38 m⁻² after 1 year, probably because of the change from subtidal to mostly intertidal conditions and increased predation by birds. Initial results from monitoring demonstrate that restoration of the Potter Pond marsh complex improved tidal exchange, negatively impacted *Phragmites*, and increased bird use, resulting in an overall shift to a more natural functioning salt marsh system. This study also demonstrates that restoring even very small tide-restricted marshes can result in impressive ecological improvements.

ADDITIONAL INDEX WORDS: Nekton, estuarine birds, emergent vegetation, water quality, national estuarine research reserve system, New England.

INTRODUCTION

Roads, dikes, levees, and other structures that block or restrict tidal flow to salt marshes are common throughout coastal New England and elsewhere. These tidal restrictions result in altered hydrology and biogeochemistry, and degraded vegetation, nekton, and bird communities compared to unrestricted salt marshes (BURDICK *et al.*, 1997; EERTMAN *et al.*, 2002; GRAY *et al.*, 2002; LLANSO, BELL, and VOSE, 1998; MYSHRALL *et al.*, 2000; PORTNOY, 1991, 1999; PORTNOY and GIBLIN, 1997; RAPOSA, 2002; ROMAN, GARVINE, and PORTNOY, 1995; RAPOSA and ROMAN, 2003; ROMAN, NIERING, and WARREN, 1984; ROMAN *et al.*, 2002; SINICROPE *et al.*, 1990; VOSE and BELL, 1994; WARREN *et al.*, 2002). Salt marshes are integral to the proper functioning of estuarine systems, and in recognition of the negative effects of tidal restrictions, efforts are now underway to restore tidal flow and ecological function to many of these degraded systems. Unfortunately, salt marsh restoration projects are not always accompanied by comprehensive ecological monitoring, and monitoring programs that compare pre- and postrestoration conditions are even less frequent. For example, in a review of Gulf of Maine

marsh restoration projects, it was found that basic environmental parameters (*e.g.*, salinity and vegetation) were monitored at more than 78% of the sites, but more biological parameters such as nekton and birds were monitored at only about half of all sites (KONISKY *et al.*, 2006). Ecological monitoring is critical to understand the effects of each individual restoration project and improve our understanding of how these systems respond to restoration in general in order to better guide future restoration efforts. Indeed, a growing number of case studies demonstrate that restoring tidal flow to restricted marshes can return ecological functionality by improving water quality conditions, vegetation, and nekton and bird communities (BARRETT and NIERING, 1993; BURDICK *et al.*, 1997; JIVOFF and ABLE, 2003; RAPOSA and ROMAN, 2003; ROMAN *et al.*, 2002; TEO and ABLE, 2003; THOM, ZEIGLER, and BORDE, 2002; WARREN *et al.*, 2002).

Potter Pond is a tide-restricted estuarine pond and salt marsh complex located on Prudence Island, Rhode Island, in the Narragansett Bay National Estuarine Research Reserve. Under tide-restricting conditions, tidal exchange with Narragansett Bay was negligible, low oxygen conditions were frequent, and the pond was virtually covered by excessive macroalgal growth. Tidal exchange was returned to this site by replacing a pair of crushed culverts with a new, larger culvert

DOI: 10.2112/SI55-015.

in April 2003. A second crushed culvert at the back end of the pond was also replaced to return estuarine tidal flow to what had become an essentially freshwater impoundment dominated by a monostand of common reed, *Phragmites australis*. Therefore, this restoration effort was essentially composed of two individual, yet connected, restorations.

The purpose of this study was to monitor ecologically relevant parameters including water quality, vegetation, nekton, and birds before and after restoration to document the effects of tidal restoration at Potter Pond. This study also uses a nearby natural marsh, Coggeshall Marsh, as an experimental control for Potter Pond. Results from this study add to the knowledge base of the effects of salt marsh restoration. Comparisons are made between results from this study and those from previous studies in New England. This study adds a unique perspective because the tide-restricted Potter Pond was consistently impounded with excessively high water levels, whereas other restricted marshes described in the literature are instead typically deprived of adequate estuarine water (e.g., BURDICK *et al.*, 1997; JIVOFF and ABLE, 2003; MYSHRALL *et al.*, 2000; ROMAN *et al.*, 2002).

STUDY SITES

The Potter Pond restoration site is located in the Narragansett Bay National Estuarine Research Reserve on Prudence Island, Rhode Island (Figure 1). Potter Pond is a small (2.3 ha) estuarine pond fringed by a narrow band of salt marsh vegetation. It has been almost completely tide-restricted by a farm access road since at least the 1930s (the date of the earliest aerial photographs of the area), but the exact duration of the restriction is unknown. Before restoration, Potter Pond was connected to Narragansett Bay under the road by two crushed concrete 38-cm-diameter culverts. A second crushed plastic culvert further restricted the back portion of the pond under another road, resulting in an impounded fresh/brackish area dominated by *P. australis* with minimal open water. This 0.24-ha *Phragmites*-dominated area is hereafter referred to as the Upper Impoundment, whereas the estuarine pond and fringing marsh area (2.04 ha) is hereafter referred to as the Lower Impoundment. Potter Pond (or Pond) refers to the entire complex, consisting of both impoundments. All crushed culverts were replaced in April 2003 to restore proper tidal exchange. A 1.5-m \times 1.2-m, 15.25-m-long aluminum arch culvert was installed to connect the Lower Impoundment with Narragansett Bay, and a 61-cm-diameter round pipe with flared ends was installed to reconnect the back end of the Lower Impoundment to the Upper Impoundment.

The control site, Coggeshall Marsh, is a 25.5-ha salt marsh located 1.5 km to the north of Potter Pond on Prudence Island. Coggeshall is completely unrestricted and is an excellent example of a mature southern New England meadow marsh, interspersed with tidal creeks, some remnant marsh pools, and unmaintained mosquito ditches. It is dominated by low marsh *Spartina alterniflora* and salt meadow species (e.g., *Spartina patens*, *Distichlis spicata*, and *Juncus gerardii*), with the shrubs *Iva frutescens* and *Baccharis halimifolia* bordering the upland.

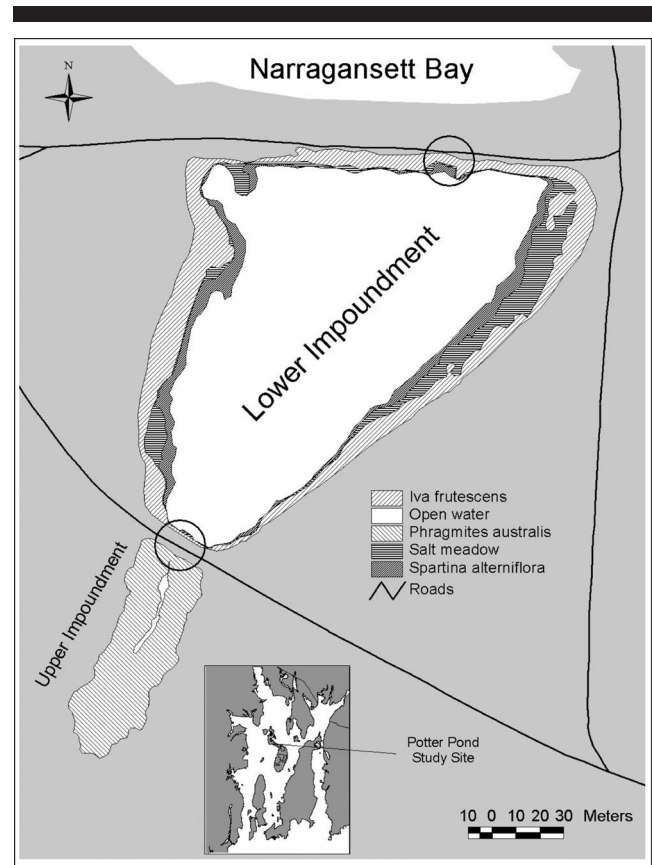


Figure 1. Location of the Potter Pond restoration site on Prudence Island, Rhode Island. Vegetation is based on prerestoration mapping efforts in 2000. Open circles indicate the locations of the two sets of culverts.

MATERIALS AND METHODS

Monitoring was conducted before and after culvert replacement to quantify the ecological responses of Potter Pond while using Coggeshall Marsh as an experimental control. In this way, a before-after-control-impact ([BACI] STEWART-OATEN, MURDOCH, and PARKER, 1986; UNDERWOOD, 1992) experimental design was utilized. The distribution and area of habitats, along with water quality, emergent and submersed vegetation, nekton, and birds were all monitored before and after restoration.

Vegetation types and habitats in Potter Pond were mapped to submeter accuracy using a Trimble Pro-XR GPS unit in 2000 and again in 2004. The outlines of all major habitat types were delineated by walking with the GPS and then importing into ArcView 3.2a. Mapped habitats included estuarine open water (subtidal), mud flats, *Salicornia* spp., *S. alterniflora*, salt meadow (composed primarily of *S. patens* and *D. spicata*), *I. frutescens*, and *P. australis*. In addition, individual tufts of *S. alterniflora* were counted and marked on the 2004 map to record the colonization of this species onto the new mud flats.

Basic water quality parameters were monitored before and after tidal restoration using YSI model 6000 sondes pro-

grammed to collect data every 30 minutes. Two sondes were deployed in Potter Pond: one was deployed in the Lower Impoundment, near the connection with Narragansett Bay, and the second was deployed in the Upper Impoundment. Control data were obtained with a third YSI sonde that is permanently deployed nearby in Potter Cove (part of Narragansett Bay) as part of the National Estuarine Research Reserve's System-Wide Monitoring Program. Prerestoration data were collected between July 12 and July 25, 2000, and for approximately 3 months immediately before the culvert replacement in April 2003. Monitoring continued for approximately 3 weeks immediately after restoration and then again from July through early September 2003. However, because of problems with the sondes, the only period when high quality data were simultaneously collected from all three sites in summer after restoration was from late August to early September. Every 30 minutes, each sonde collected data for temperature, salinity, dissolved oxygen, water depth, and pH. Daily changes in water levels (a proxy for tide range) were calculated for each area by subtracting the lowest depth reading from the highest reading each day.

Emergent vegetation at Potter Pond and Coggeshall was monitored using 1-m² plots located at intervals along randomly selected transects in the marshes. All transects ran from the upland edge to a primary open water body (either a main tidal creek or open water pond, but not marsh pools). Twenty-two plots along seven transects were sampled in the Lower Impoundment, 12 plots along two transects were sampled in the Upper Impoundment, and 21 plots along three transects were sampled in Coggeshall Marsh. In addition, three transects were sampled in Coggeshall Marsh. Vegetation was monitored in each plot using the point-intercept method (ELZINGA, SALZER, and WILLOUGHBY, 1998) at the end of the growing seasons (late August to September) in 2000, 2003, and 2004. In each quadrat, the percent cover of all species was quantified, and, when present, the heights of up to 12 *S. alterniflora* and *P. australis* stems were measured. Percent cover of macroalgae was sampled in the Lower Impoundment in September 2000 and 2003 using 30 1-m² plots that were randomly established throughout the open water. Macroalgae had disappeared by 2004, and sampling did not occur during that year.

Nekton was sampled in Potter Pond and Coggeshall Marsh using a throw trap according to protocols described in RAPOSA, ROMAN, and HELTSHE (2003). The trap measured 1 m² × 0.5 m high and was constructed of an aluminum frame surrounded by 3-mm mesh hardware cloth on the four sides. Captured nekton was removed from the trap using a 1-m × 0.5-m dip net with 3-mm mesh. Each site was sampled in July and September of 2000, 2003, and 2004. On each date, 25 stations were sampled from each site. All samples were collected from aquatic habitats adjacent to the vegetated marsh surface (e.g., creeks, pools, ponds) on ebbing tides, beginning after the marsh surface had drained of tidal water. A station was sampled by slowly approaching it from the marsh surface and quickly throwing the trap into the water. After the trap was secured into the sediment, dip netting was conducted from all four sides of the trap until nekton was

absent from three consecutive swipes. Each individual animal was identified, counted, and returned live to the field.

Birds were monitored in Potter Pond and Coggeshall Marsh in 2000, 2003, and 2004 using visual point counts (e.g., HOWE *et al.*, 1997). One monitoring point was located in the Lower Impoundment, one was in the Upper Impoundment, and four were located at Coggeshall Marsh. All birds observed using the marsh within 10 minutes at each point were identified and counted. Fly-overs were not counted unless they were actively feeding above the marsh. Each point was sampled approximately biweekly from late June through September (seven dates each year). To facilitate comparisons, all data were converted to the mean number of birds observed per viewing effort.

Changes in ecological parameters were analyzed using an array of statistical tests. Water quality comparisons were made using analysis of variance (ANOVA) and Student-Newman Keuls (SNK) pairwise comparisons. Changes in emergent vegetation, nekton, and bird community composition were analyzed among the 3 years of the study using analysis of similarity (ANOSIM). If any statistical differences in composition were detected, the species most responsible for the differences, based on percent contribution, were determined using similarity percentages (SIMPER). Both ANOSIM and SIMPER are part of the PRIMER 6 statistical package (PRIMER-E Ltd.). Changes in the total abundance of all nekton and of all birds were analyzed using ANOVA followed by SNK pairwise comparisons after transforming the data to address the assumptions of normality and equal variance. Richness of nekton and of birds was estimated using the jackknife technique described by HELTSHE and FORRESTER (1983) and compared among years using Student's *t* test.

For all water quality parameters, statistical comparisons were only made among sites during the same year; interannual comparisons were not made within each site because monitoring was not conducted during the exact same time each year. For all biological parameters, all analyses were performed within the same marsh among years, and no attempt was made to directly compare parameters between marshes (e.g., between Potter Pond and Coggeshall Marsh) within the same year. Such analyses can be useful to track the degree to which restoration results in increased similarity between the restoring marsh and a reference marsh. However, in this study, it was not felt that Potter Pond would necessarily become more similar (in terms of ecological structure) to Coggeshall Marsh with time. Although they are both salt marshes, the two sites remain very different in terms of size and habitat characteristics. Instead, Coggeshall Marsh was simply used as an experimental control to account for interannual variability in the parameters that were monitored.

For all biotic parameters except nekton, the Upper and Lower Impoundments were treated as separate experimental units. For nekton, however, data from the two impoundments were pooled before any statistical analyses because nekton freely moved between the two sites with the tides on a daily basis after restoration, and the two areas essentially functioned as one contiguous dynamic tidal pond for nekton.

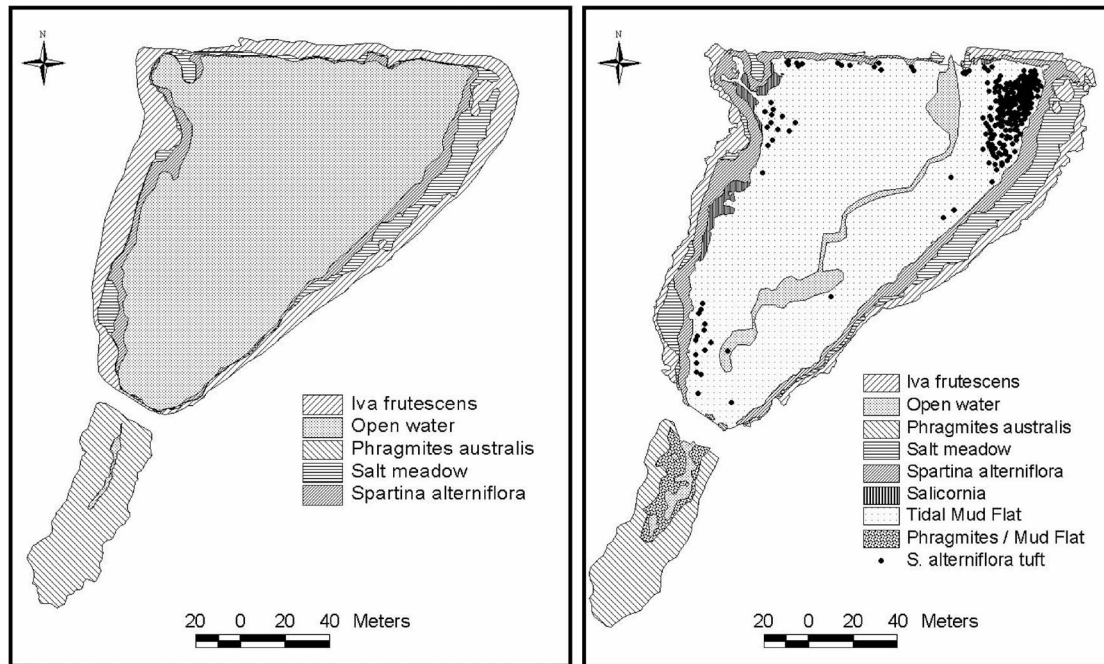


Figure 2. Composition and extent of habitat and vegetation types at low tide in Potter Pond before and after restoration. The left panel shows conditions in 2000; the right in 2004. Note the extension of the *Spartina alterniflora* zone, emergence of a *Salicornia* zone, and the numerous tufts of *S. alterniflora* that encroached on the new mud flats after restoration. Note that the outside edge of the *Iva frutescens* zone in 2000 was not delineated in the field and should not be directly compared to 2004.

RESULTS

Mapping

Figure 2 shows the changes in vegetation and habitats in Potter Pond before and after culvert replacement in 2003. Some notable differences between years are apparent. After restoration, substantial areas of newly exposed mudflats were present at low tide, with a small tidal creek running through the middle of the former pond. This is in contrast to conditions in 2000 when open water persisted throughout the Lower Impoundment at all tide stages. After restoration, the *S. alterniflora* zone began expanding onto newly exposed areas of denuded marsh peat via rhizome growth. New individual tufts of *S. alterniflora* and *Salicornia* spp. also began growing on the new mudflats in various areas along the lower edge of the *S. alterniflora* zone. Overall, the area of *S. alterniflora* grew by 0.08 ha (67%) between 2000 and 2004, whereas *Salicornia* spp. increased from zero to 0.30 ha. Low-tide levels of open water decreased by 1.34 ha (93%), while mud flats increased from zero to 1.24 ha. By all accounts, the stagnant pond that existed before replacement of the culverts was replaced by a dynamic system with new mudflats that were quickly colonized by encroaching salt marsh vegetation.

Water Quality

In 2000, daily changes in water levels were severely reduced behind the restricting culverts (4 cm in the Lower Impoundment; 5 cm in the Upper Impoundment) compared to

the adjacent Narragansett Bay (139 cm) (Table 1). Mean daily temperatures were also higher in the Lower Impoundment than in Narragansett Bay or the Upper Impoundment. Salinity in the Upper Impoundment was significantly lower than the other sites, with a mean level of 1.5 ppt in the summer of 2000. Mean pH was significantly different between all three areas, with highest levels occurring in the Lower Impoundment and lowest levels found in the Upper Impoundment. Mean oxygen levels were significantly higher in the Upper Impoundment compared to the Lower Impoundment and the adjacent area of Narragansett Bay.

Culvert replacement immediately returned regular tidal patterns to the formerly impounded areas (Figure 3) and increased daily changes in water levels in both impounded areas (Table 1). Restoration also resulted in other water quality parameters reverting to levels similar to those found in the adjacent part of Narragansett Bay. For example, during the first year of restoration, mean temperature did not differ among the Bay, the Lower Impoundment, and the Upper Impoundment. Mean salinity remained significantly lower in the Upper Impoundment compared to the Lower Impoundment and Bay, but increased from the near-fresh levels observed before restoration to 26.8 ppt after restoration. Mean pH levels were more consistent among the three areas in 2003, but remained statistically different among the sites, with highest levels observed in the Bay and lowest levels in the Upper Impoundment. After restoration, mean dissolved oxygen was lower in the Upper Impoundment compared to

Table 1. Water quality conditions (mean and one standard error) in Narragansett Bay, the Lower Impoundment, and the Upper Impoundment in 2000 and 2003. Data were collected from July 12, 2000, to July 24, 2000, and from August 26, 2003, to September 7, 2003. Different superscripts indicate significant differences in mean daily values among sites based on one-way ANOVA ($p < 0.05$). Tests were not run to compare daily water level ranges.

| | | Narragansett Bay | Lower Impoundment | Upper Impoundment |
|------|------------------------------|---------------------------|---------------------------|---------------------------|
| 2000 | Daily water depth range (cm) | 120.25 (1.70) | 3.67 (0.47) | 5.09 (1.40) |
| | Temperature (C) | 21.95 (0.03) ^b | 24.49 (0.04) ^a | 22.15 (0.06) ^b |
| | Salinity (ppt) | 29.74 (0.01) ^a | 30.43 (0.01) ^a | 1.54 (0.04) ^b |
| | Dissolved oxygen (mg/l) | 6.35 (0.15) ^b | 6.25 (0.13) ^b | 7.84 (0.26) ^a |
| | pH | 7.24 (0.01) ^b | 8.69 (0.01) ^a | 6.53 (0.00) ^c |
| 2003 | Daily water depth range (cm) | 142.60 (0.05) | 119.57 (1.60) | 44.70 (1.30) |
| | Temperature (C) | 20.82 (0.04) | 21.59 (0.11) | 21.28 (0.14) |
| | Salinity (ppt) | 28.78 (0.02) ^a | 28.44 (0.05) ^a | 26.81 (0.09) ^b |
| | Dissolved oxygen (mg/l) | 5.27 (0.10) ^a | 5.30 (0.15) ^a | 3.09 (0.12) ^b |
| | pH | 7.70 (0.01) ^a | 7.52 (0.01) ^b | 7.12 (0.02) ^c |

the Lower Impoundment and Bay, which did not differ statistically from one another.

Vegetation

Before restoration, the Lower Impoundment of Potter Pond was composed of typical New England salt marsh vegetation, including low marsh *S. alterniflora* flanked at higher elevations by *S. patens* and *D. spicata* (Table 2). Landward, these marsh species were flanked by bands of *I. frutescens* and *B. halimifolia*. In contrast, the Upper Impoundment consisted of a monoculture of *P. australis* surrounded by a diverse community of brackish, freshwater, and upland plants. Coggeshall Marsh was similar to Potter Pond's Lower Impoundment in that it consisted of plants common to salt marshes throughout the region (Table 2).

Restoration did not result in a compositional change in the emergent vegetation community in the Lower Impoundment because typical salt marsh species were already present before restoration (ANOSIM, Global $R = 0.002$, $p = 0.39$). It also did not result in a change in the vegetation community in the Upper Impoundment (ANOSIM, Global $R = -0.01$, $p = 0.64$). Similarly, the vegetation community at Coggeshall Marsh did not change among the 3 survey years (ANOSIM,

Global $R = -0.02$, $p = 0.84$). However, the height of live *P. australis* in the Upper Impoundment decreased from 2000 to 2004 (ANOVA, $F = 4.14$, $p = 0.03$; SNK, $p < 0.05$) (Figure 4), as the stress of reintroduced estuarine water negatively impacted this species. Interestingly, there was no significant decrease in *P. australis* height during the first year of restoration (SNK, $p > 0.05$). *Phragmites australis* was not present in any of the sample plots in the Lower Impoundment or at Coggeshall. The height of *S. alterniflora* tended to increase in the Lower Impoundment during the course of this study (Figure 4), but this change was not statistically significant (ANOVA, $F = 0.62$, $p = 0.55$) nor was there a significant change at Coggeshall (ANOVA, $F = 1.84$, $p = 0.17$).

Before restoration, dense green algal mats covered approximately 72% of the Lower Impoundment, but this was reduced to only 10% cover 5 months after tidal restoration in 2003. Macroalgae was not sampled in 2004 because it had virtually disappeared from the pond.

Nekton

Only 10 species of nekton were found in Potter Pond (again, for nekton this includes pooled data from both impounded areas) in 2000 before restoration, and only *Palaemonetes* spp., *Fundulus heteroclitus*, *Lucania parva*, and *Cyprinodon variegatus* were abundant (Table 3). The remaining six species were not abundant, and some were caught in only one sample. In contrast, 14 nekton species were captured in Coggeshall Marsh, including some common marsh species that were conspicuously absent from Potter Pond, including *Pagurus* spp., *Crangon septemspinosa*, *Panopeus herbstii*, and *Pseudopleuronectes americanus*.

Nekton community composition changed during the first year of restoration in Potter Pond (ANOSIM 2000 vs. 2003, Global $R = 0.68$, $p < 0.001$). Based on SIMPER, these changes were mostly caused by the disappearance of *L. parva* (31% of the total dissimilarity between years was caused by this species) after restoration and the similarly large decrease in *C. variegatus* (16%). A change in nekton community composition was also observed in Coggeshall Marsh between 2000 and 2003 (ANOSIM, Global $R = 0.03$, $p < 0.05$). However, a SIMPER analysis shows that this change was due to increases in *Palaemonetes* spp. (27%) and *Pagurus* spp. (15%) in

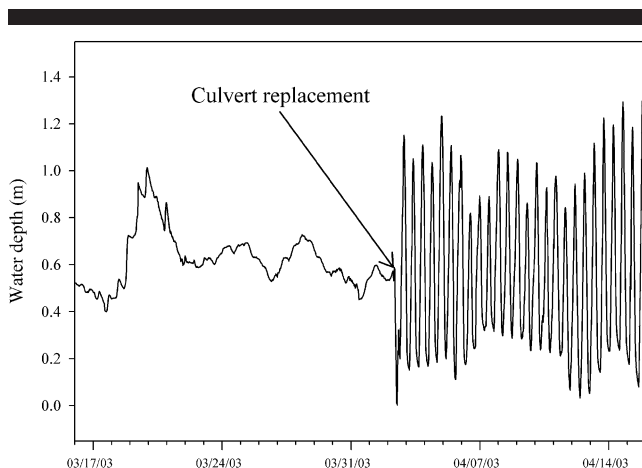


Figure 3. Water depths in the Lower Impoundment of Potter Pond immediately before and after culvert replacement in April 2003.

Table 2. Mean (and one standard error) percent cover of vegetation species in the Lower Impoundment, the Upper Impoundment, and Coggeshall Marsh in 2000, 2003, and 2004. Species are sorted in descending order based on overall cover among the three sites.

| Species | Lower Impoundment | | | Upper Impoundment | | | Coggeshall Marsh | | |
|------------------------------------|-------------------|--------------|--------------|-------------------|--------------|--------------|------------------|--------------|--------------|
| | 2000 | 2003 | 2004 | 2000 | 2003 | 2004 | 2000 | 2003 | 2004 |
| <i>Spartina alterniflora</i> | 23.32 (7.28) | 15.55 (5.81) | 21.68 (7.14) | 0.00 | 0.00 | 0.00 | 48.62 (8.12) | 56.33 (7.34) | 40.10 (8.58) |
| <i>Spartina patens</i> | 31.27 (7.46) | 17.86 (6.80) | 12.50 (4.89) | 0.00 | 0.00 | 0.00 | 49.48 (8.81) | 45.00 (7.95) | 49.00 (9.01) |
| <i>Phragmites australis</i> (dead) | 0.00 | 0.00 | 0.00 | 35.33 (7.22) | 41.50 (9.20) | 34.58 (8.91) | 0.00 | 0.00 | 0.00 |
| <i>Distichlis spicata</i> | 21.82 (8.78) | 22.77 (8.57) | 22.45 (8.29) | 0.00 | 0.00 | 0.00 | 18.00 (6.09) | 14.57 (5.44) | 10.00 (3.66) |
| <i>Iva frutescens</i> | 29.00 (8.92) | 24.82 (8.43) | 20.14 (7.13) | 0.00 | 0.00 | 0.00 | 5.67 (4.08) | 7.33 (4.03) | 9.67 (5.65) |
| <i>Phragmites australis</i> | 0.00 | 0.00 | 0.00 | 37.08 (5.51) | 11.08 (2.63) | 11.33 (3.25) | 0.00 | 0.00 | 0.00 |
| Bare | 13.05 (5.51) | 15.36 (6.69) | 4.41 (2.00) | 0.00 | 0.00 | 0.00 | 0.62 (0.42) | 0.62 (0.44) | 0.62 (0.46) |
| <i>Baccharis halimifolia</i> | 10.41 (5.79) | 13.55 (6.64) | 14.00 (6.91) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.43 (0.43) |
| <i>Salicornia</i> spp. | 2.32 (1.57) | 5.82 (2.63) | 17.77 (4.90) | 0.00 | 0.00 | 0.08 (0.08) | 1.86 (1.06) | 3.05 (1.19) | 3.81 (1.00) |
| <i>Panicum virgatum</i> | 0.41 (0.30) | 0.14 (0.10) | 0.18 (0.18) | 10.83 (8.33) | 9.42 (7.92) | 7.92 (7.92) | 0.43 (0.43) | 0.38 (0.38) | 0.33 (0.33) |
| Water | 0.00 | 0.00 | 0.00 | 6.92 (6.92) | 7.92 (7.92) | 7.92 (7.92) | 0.00 | 0.00 | 0.00 |
| <i>Acer rubrum</i> | 0.00 | 0.00 | 0.00 | 6.33 (6.33) | 7.92 (7.92) | 7.92 (7.92) | 0.00 | 0.00 | 0.00 |
| <i>Celastrus orbiculatus</i> | 0.00 | 0.00 | 0.00 | 6.00 (6.00) | 8.67 (7.19) | 5.67 (3.85) | 0.00 | 0.00 | 0.00 |
| <i>Juncus gerardii</i> | 1.18 (0.80) | 4.91 (4.33) | 0.59 (0.59) | 0.00 | 0.00 | 0.00 | 4.57 (4.52) | 3.33 (3.33) | 5.52 (4.27) |
| <i>Spartina patens</i> (dead) | 0.00 | 4.91 (3.39) | 4.55 (3.15) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Viburnum recognitum</i> | 0.00 | 0.00 | 0.00 | 1.25 (1.25) | 3.25 (3.25) | 5.17 (5.17) | 0.00 | 0.00 | 0.00 |
| <i>Iva frutescens</i> (dead) | 0.32 (0.32) | 1.82 (1.26) | 5.00 (2.46) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Juncus effuses</i> | 0.00 | 0.00 | 0.00 | 6.50 (6.50) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Lycopus virginicus</i> | 0.00 | 0.00 | 0.00 | 5.58 (5.15) | 0.25 (0.25) | 0.67 (0.67) | 0.00 | 0.00 | 0.00 |
| <i>Ilex verticillata</i> | 0.00 | 0.00 | 0.00 | 5.08 (5.08) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Rubus alleghaniensis</i> | 0.00 | 0.00 | 0.00 | 1.08 (1.08) | 1.83 (1.50) | 0.17 (0.11) | 0.00 | 0.00 | 0.00 |
| <i>Limonium carolinianum</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.38 (0.21) | 1.10 (0.53) | 1.24 (0.61) |
| <i>Solidago sempervirens</i> | 0.64 (0.44) | 0.23 (0.16) | 0.14 (0.10) | 0.00 | 0.00 | 0.00 | 0.33 (0.33) | 0.38 (0.38) | 0.10 (0.10) |
| <i>Rhus radicans</i> | 0.00 | 0.00 | 0.00 | 1.00 (1.00) | 0.50 (0.50) | 0.25 (0.25) | 0.00 | 0.00 | 0.00 |
| <i>Parthenocissus quinquefolia</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.75 (1.50) | 0.00 | 0.00 | 0.00 |
| <i>Atriplex hastate</i> | 1.32 (0.60) | 0.23 (0.19) | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Myrica pennsylvanica</i> | 0.55 (0.55) | 0.00 | 0.00 | 0.58 (0.58) | 0.00 | 0.00 | 0.00 | 0.29 (0.29) | 0.00 |
| <i>Smilax rotundifolia</i> | 0.00 | 0.05 (0.05) | 0.09 (0.09) | 0.00 | 0.25 (0.25) | 0.67 (0.67) | 0.00 | 0.00 | 0.00 |
| Wreck | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.24 (0.24) | 0.86 (0.86) | 0.00 |
| <i>Rubus flagellaris</i> | 0.00 | 0.00 | 0.00 | 0.00 | 1.08 (1.08) | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Mikania scandens</i> | 0.00 | 0.00 | 0.00 | 0.58 (0.40) | 0.42 (0.42) | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Solidago rugosa</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.75 (0.75) | 0.25 (0.25) | 0.00 | 0.00 | 0.00 |
| <i>Juncus gerardii</i> (dead) | 0.00 | 0.59 (0.44) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Myrica pennsylvanica</i> (dead) | 0.00 | 0.59 (0.59) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Vaccinium corymbosum</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.67 (0.67) | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Typha angustifolia</i> | 0.00 | 0.00 | 0.00 | 0.33 (0.33) | 0.17 (0.17) | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Thelypteris</i> sp. | 0.00 | 0.00 | 0.00 | 0.08 (0.08) | 0.25 (0.25) | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Setaria italica</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 (0.33) | 0.00 | 0.00 |
| <i>Rosa rugosa</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.25 (0.25) | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Trifolium</i> sp. | 0.00 | 0.00 | 0.00 | 0.17 (0.17) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Galium palustre</i> | 0.00 | 0.00 | 0.00 | 0.08 (0.08) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Carex</i> sp. | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.05 (0.05) | 0.00 |

Table 3. Mean nekton density (number m^{-2} and one standard error) in Potter Pond and Coggeshall Marsh before and after tidal restoration. Prerestoration data are from 2000; postrestoration data are from 2003 and 2004. Species are sorted in decreasing order based on overall density averaged between sites and among years.

| Species | Potter Pond | | | Coggeshall | | |
|--|--------------|--------------|--------------|---------------|----------------|----------------|
| | 2000 | 2003 | 2004 | 2000 | 2003 | 2004 |
| <i>Palaemonetes</i> spp. (Grass shrimp) | 13.34 (5.86) | 15.90 (5.35) | 3.36 (1.30) | 90.22 (22.21) | 175.84 (34.48) | 143.74 (39.60) |
| <i>Fundulus heteroclitus</i> (Mummichog) | 20.94 (3.09) | 16.42 (2.57) | 13.70 (2.77) | 19.24 (3.83) | 19.88 (3.13) | 15.02 (3.73) |
| <i>Lucania parva</i> (Rainwater killifish) | 51.24 (5.15) | 0.00 | 0.00 | 0.62 (0.51) | 0.08 (0.08) | 0.00 |
| <i>Pagurus</i> spp. (Hermit crab) | 0.00 | 0.00 | 0.10 (0.10) | 4.96 (1.37) | 7.50 (1.92) | 12.62 (5.28) |
| <i>Cyprinodon variegatus</i> (Sheepshead minnow) | 15.16 (3.57) | 0.22 (0.08) | 0.76 (0.38) | 1.42 (0.55) | 0.52 (0.27) | 0.74 (0.27) |
| <i>Fundulus majalis</i> (Striped killifish) | 0.02 (0.02) | 2.36 (0.53) | 3.44 (0.69) | 1.06 (0.30) | 1.46 (0.35) | 1.34 (0.40) |
| <i>Crangon septemspinosa</i> (Sand shrimp) | 0.00 | 0.00 | 0.00 | 2.32 (1.17) | 0.40 (0.15) | 3.82 (1.41) |
| <i>Menidia menidia</i> (Atlantic silverside) | 1.24 (0.57) | 2.10 (0.89) | 0.80 (0.25) | 0.14 (0.06) | 0.24 (0.13) | 0.28 (0.15) |
| <i>Carcinus maenas</i> (Green crab) | 0.28 (0.15) | 0.40 (0.22) | 0.74 (0.33) | 0.24 (0.09) | 0.38 (0.13) | 0.64 (0.18) |
| <i>Menidia beryllina</i> (Inland silverside) | 0.48 (0.19) | 0.24 (0.12) | 0.24 (0.09) | 0.84 (0.44) | 0.50 (0.27) | 0.00 |
| <i>Limulus polyphemus</i> (Horseshoe crab) | 0.00 | 0.00 | 0.94 (0.38) | 0.00 | 0.00 | 0.00 |
| <i>Anguilla rostrata</i> (American eel) | 0.08 (0.05) | 0.00 | 0.04 (0.03) | 0.40 (0.19) | 0.16 (0.07) | 0.02 (0.02) |
| <i>Panopeus herbstii</i> (Mud crab) | 0.00 | 0.02 (0.02) | 0.00 | 0.38 (0.18) | 0.12 (0.05) | 0.08 (0.04) |
| <i>Apeltes quadracus</i> (Fourspine stickleback) | 0.08 (0.04) | 0.00 | 0.00 | 0.00 | 0.00 | 0.30 (0.13) |
| <i>Pseudopleuronectes americanus</i> (Winter flounder) | 0.00 | 0.00 | 0.00 | 0.12 (0.08) | 0.04 (0.03) | 0.04 (0.03) |
| <i>Ovalipes ocellatus</i> (Lady crab) | 0.00 | 0.00 | 0.02 (0.02) | 0.00 | 0.00 | 0.16 (0.16) |
| <i>Brevoortia tyrannus</i> (Atlantic menhaden) | 0.00 | 0.02 (0.02) | 0.00 | 0.00 | 0.00 | 0.10 (0.10) |
| <i>Syngnathus fuscus</i> (Northern pipefish) | 0.00 | 0.00 | 0.00 | 0.00 | 0.02 (0.02) | 0.04 (0.03) |
| <i>Callinectes sapidus</i> (Blue crab) | 0.00 | 0.00 | 0.00 | 0.04 (0.04) | 0.00 | 0.00 |
| <i>Pungitius pungitius</i> (Ninespine stickleback) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.04 (0.03) |
| <i>Mugil curema</i> (White mullet) | 0.00 | 0.00 | 0.00 | 0.00 | 0.02 (0.02) | 0.00 |
| <i>Tautoga onitis</i> (Tautog) | 0.00 | 0.00 | 0.00 | 0.00 | 0.02 (0.02) | 0.00 |
| <i>Gasterosteus aculeatus</i> (Threespine stickleback) | 0.00 | 0.02 (0.02) | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Hemigrapsus sanguineus</i> (Asian shore crab) | 0.00 | 0.00 | 0.02 (0.02) | 0.00 | 0.00 | 0.00 |

2003; thus, these changes were not related to the changes observed in Potter Pond during this time.

Nekton community composition in Potter Pond in 2004 also differed from 2000 (ANOSIM, Global $R = 0.66$, $p < 0.001$) as well as from 2003 (ANOSIM, Global $R = 0.13$, $p < 0.001$), indicating that the nekton community continued to undergo changes during the second year of restoration. Based on SIMPER, the changes in nekton between years one and two of postrestoration were due to a decrease in *F. heteroclitus* (responsible for 21% of the overall community dissimilarity), a large drop in *Palaemonetes* spp. (20%), and the continued increase in *Fundulus majalis* (15%). However, a change was also observed at Coggeshall Marsh between 2003 and 2004 (ANOSIM, Global $R = 0.07$, $p < 0.01$) which was primarily due to a decrease in *Palaemonetes* spp. (SIMPER, 29%), an increase in *Pagurus* spp. (15%), and a decrease in *F. heteroclitus* (13%). These results, therefore, suggest that some of the second year changes in Potter Pond (e.g., the decreases in *F. heteroclitus* and *Palaemonetes* spp.) may not be entirely because of restoration and instead may simply be due to natural variability. Thus, the only change clearly caused by restoration of Potter Pond during the second year of restoration is the continued increase in *F. majalis* density.

Species richness in Potter Pond tended to increase each year after restoration (Figure 5), but these changes were not significantly different among years (Student's t test, $p > 0.05$ for each pairwise comparison). Richness also did not change at Coggeshall Marsh during the 3 years of the study (Student's t test, $p > 0.05$ for each pairwise comparison). In contrast, overall nekton density in Potter Pond was significantly lower in both 2003 and 2004 than in 2000 (two-way ANOVA,

$F = 14.55$, $p < 0.001$) after the return of tidal flow with restoration (Figure 5). Nekton density in Coggeshall Marsh did not differ among the years of this study (two-way ANOVA; $F = 1.93$; $p = 0.15$), indicating that the decrease in nekton density in Potter Pond was the result of restoration.

Birds

In 2000, the Lower Impoundment of Potter Pond functioned as a permanent estuarine pond that attracted birds that forage in open water such as *Sterna albifrons*, *Megasceryle alcyon*, and *Phalacrocorax auritus*. *Iridoprocne bicolor* and *Hirundo rustica* also regularly foraged over the open water of the impounded pond. Wading birds were not common, but the fringing marsh vegetation provided habitat for edge-associated songbirds (e.g., *Tyrannus tyrannus*, *Carduelis tristis*, and *Dendroica petechia*) (Table 4). The Upper Impoundment did not provide habitat for many bird species, and *Agelaius phoeniceus* was the only species regularly found there. Coggeshall Marsh supported a rich assemblage of birds that are common to New England salt marshes, including the songbirds *I. bicolor*, *A. phoeniceus*, *H. rustica*, and *Ammospiza* spp., and the wading birds *Casmerodius albus*, *Egretta thula*, and *Ardea herodias* (Table 4).

Tidal restoration induced a significant change in the bird community in the Lower Impoundment during the first year of restoration (ANOSIM, Global $R = 0.69$, $p = 0.001$). The second year after restoration was also different from prerestoration conditions (ANOSIM, Global $R = 0.63$, $p = 0.001$), but not from the first year after restoration (ANOSIM, Global $R = 0.02$, $p > 0.05$), indicating that, as with nekton, the most

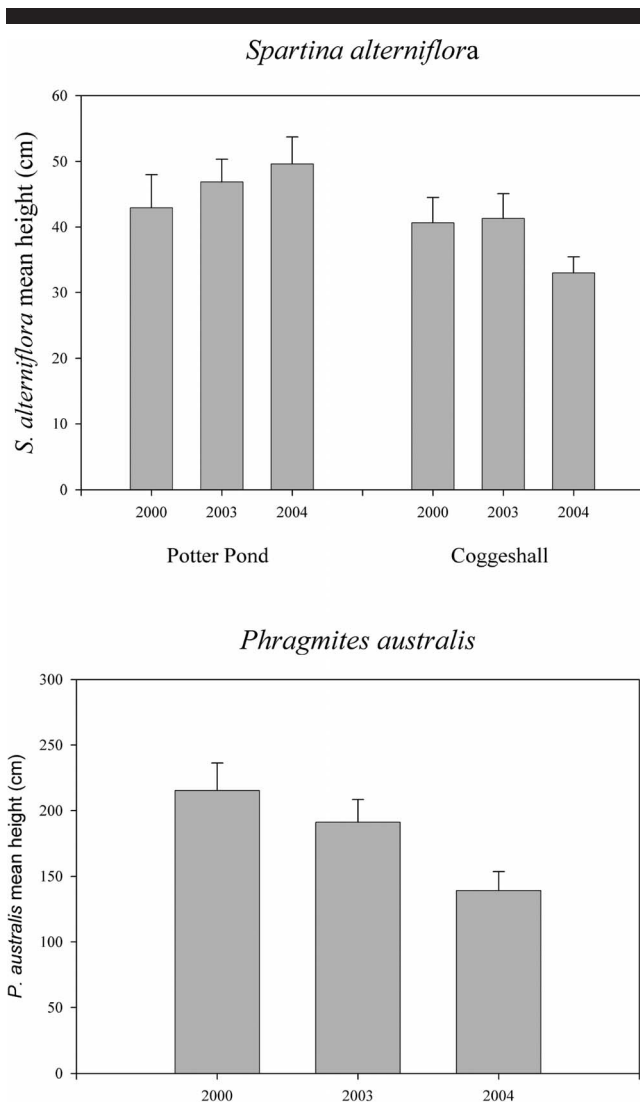


Figure 4. Average heights of *Spartina alterniflora* and *Phragmites australis*. *Spartina alterniflora* heights were measured in the Lower Impoundment of Potter Pond and Coggeshall Marsh, whereas *P. australis* heights were measured in the Upper Impoundment of the Pond. Error bars are one standard error.

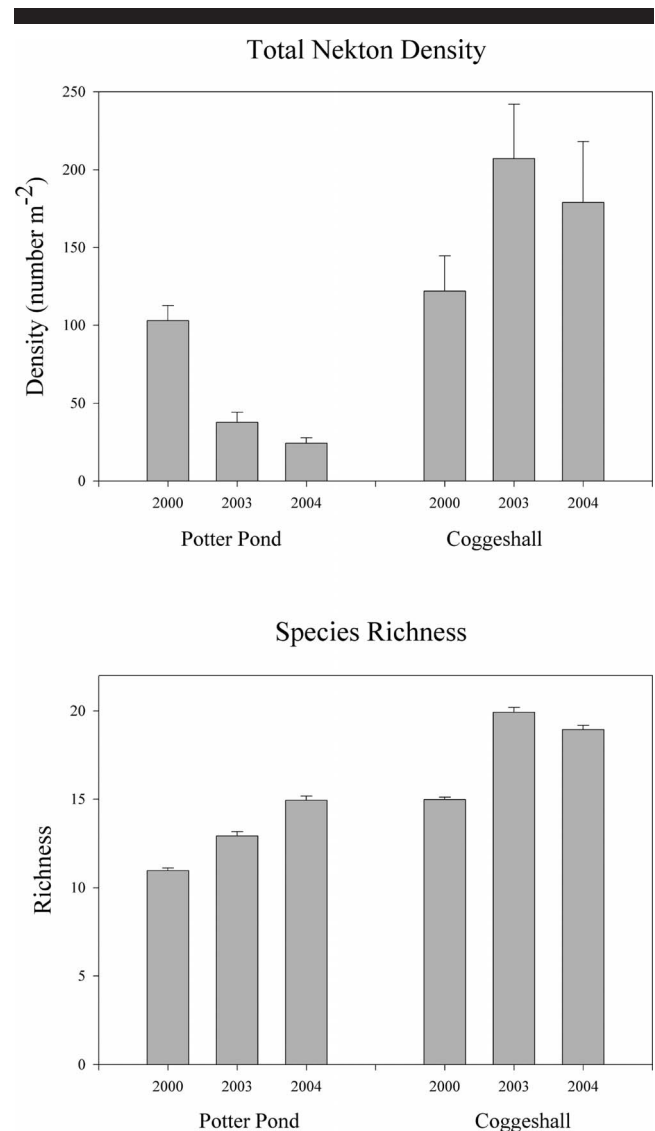


Figure 5. Mean density of all nekton species combined and species richness of nekton in Potter Pond and Coggeshall Marsh. Prerestoration conditions were in 2000; postrestoration was in 2003 and 2004. Error bars are one standard error.

dramatic shift occurred during the first year of restoration. The species most responsible for the shift in composition during the first year of restoration, according to SIMPER, include *Charadrius semipalmatus* (15%), *Calidris pusilla* (15%), and to a lesser extent *E. thula* (6%), *Limnodromus griseus* (5%), and *Charadrius vociferous* (5%). Bird communities did not change among the 3 years of the study in the Upper Impoundment (ANOSIM, Global $R = -0.01$, $p > 0.05$) nor did they change at Coggeshall Marsh (ANOSIM, Global $R = 0.08$, $p > 0.05$), indicating that the changes observed in the Lower Impoundment were due to restoration.

Bird abundance in the Lower Impoundment during both years of postrestoration was significantly higher than before restoration (ANOVA, $F = 5.58$, $p = 0.01$; SNK comparison

between year 2000 and 2003, $p = 0.01$; SNK comparison between year 2000 and 2004, $p = 0.04$) (Figure 6). Bird abundance did not change in the Upper Impoundment among the 3 years of this study (ANOVA, $F = 1.31$, $p = 0.30$) nor did it change at Coggeshall Marsh (ANOVA, $F = 1.30$, $p = 0.30$), indicating again that the changes observed in the Lower Impoundment were due to restoration. Bird species richness in the Lower Impoundment increased from 2000 to 2003 and again from 2003 to 2004 (Student's t test, $p < 0.005$ for each test) (Figure 6). Patterns in species richness in the Upper Impoundment and at Coggeshall Marsh were similar to each other, but did not follow trends observed at the Lower Impoundment. At Coggeshall and the Upper Impoundment, richness was lower in 2003 than in 2000, but then higher in

Table 4. Bird abundance (mean number per viewing session and one standard error) in the Lower Impoundment, the Upper Impoundment, and Coggeshall Marsh before and after tidal restoration. Pre-restoration data are from 2000, post-restoration data are from 2003 and 2004. Species are sorted in decreasing order based on overall density averaged between sites and among years.

| Species | Coggeshall | | | | Upper Impoundment | | | | Lower Impoundment | | | |
|---|---------------|--------------|--------------|------|-------------------|-------------|---------------|------|-------------------|---------------|------|---------------|
| | 2000 | 2003 | 2004 | 2004 | 2000 | 2003 | 2004 | 2004 | 2000 | 2003 | 2004 | 2004 |
| <i>Iridoprocne bicolor</i> (Tree Swallow) | 43.14 (23.68) | 16.14 (6.34) | 14.71 (5.41) | 0.00 | 0.14 (0.14) | 0.00 | 14.86 (14.86) | 0.00 | 2.43 (1.95) | 0.00 | 0.00 | 0.14 (0.14) |
| <i>Calidris pusilla</i> (Semipalmated Sandpiper) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 47.14 (20.80) | 0.00 | 20.14 (8.30) |
| <i>Charadrius semipalmatus</i> (Semipalmated Plover) | 0.00 | 0.00 | 3.57 (1.90) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 26.29 (6.10) | 0.00 | 32.71 (13.77) |
| <i>Agelaius phoeniceus</i> (Red-winged Blackbird) | 1.00 (0.53) | 3.14 (0.94) | 2.86 (1.32) | 0.00 | 1.43 (0.75) | 2.57 (0.92) | 1.00 (0.44) | 0.00 | 0.29 (0.29) | 0.00 | 0.00 | 0.43 (0.43) |
| <i>Hirundo rustica</i> (Barn Swallow) | 3.43 (0.48) | 3.14 (0.51) | 0.86 (0.46) | 0.00 | 0.00 | 0.00 | 0.29 (0.18) | 0.00 | 0.71 (0.36) | 0.00 | 0.00 | 0.57 (0.43) |
| <i>Casmerodius albus</i> (Great Egret) | 1.00 (0.44) | 2.14 (1.01) | 1.29 (0.64) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.14 (0.14) | 0.00 | 0.14 (0.14) |
| <i>Annoszipiza caudacuta</i> spp. (Marsh Sparrow) | 1.43 (0.72) | 0.43 (0.20) | 1.71 (0.68) | 0.00 | 0.00 | 0.00 | 0.43 (0.20) | 0.00 | 0.00 | 0.00 | 0.00 | 0.43 (0.30) |
| <i>Dumetella carolinensis</i> (Gray Catbird) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) |
| <i>Linnodromus griseus</i> (Short-billed Dowitcher) | 2.43 (1.65) | 0.43 (0.20) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 3.57 (3.07) | 0.00 | 0.00 |
| <i>Calidris minutilla</i> (Least Sandpiper) | 1.86 (0.77) | 0.86 (0.46) | 0.43 (0.43) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.71 (0.57) | 0.00 | 0.00 |
| <i>Corvus brachyrhynchos</i> (American Crow) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.29 (0.18) | 0.00 | 0.00 |
| <i>Charadrius vociferans</i> (Killdeer) | 1.00 (0.31) | 0.43 (0.20) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 3.00 (1.83) | 0.00 | 0.43 (0.30) |
| <i>Tyrannus tyrannus</i> (Eastern Kingbird) | 0.43 (0.20) | 0.71 (0.47) | 1.29 (0.57) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.43 (0.20) | 0.00 | 0.00 | 0.00 |
| <i>Egretta thula</i> (Snowy Egret) | 0.43 (0.20) | 0.29 (0.18) | 0.57 (0.37) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.86 (0.14) | 0.00 | 0.29 (0.18) |
| <i>Ardea herodias</i> (Great Blue Heron) | 0.00 | 0.29 (0.29) | 1.43 (0.48) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.29 (0.18) | 0.00 | 0.00 |
| <i>Melospiza melodia</i> (Song Sparrow) | 1.43 (0.72) | 0.57 (0.37) | 1.29 (0.29) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.43 (0.30) | 0.00 | 0.29 (0.18) |
| <i>Larus argentatus</i> (Herring Gull) | 0.00 | 0.14 (0.14) | 0.14 (0.14) | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.29 (0.29) | 0.00 | 0.00 | 0.14 (0.14) |
| <i>Carduelis tristis</i> (American Goldfinch) | 0.00 | 0.00 | 0.86 (0.70) | 0.00 | 0.00 | 0.00 | 0.43 (0.30) | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) |
| <i>Tringa melanoleuca</i> (Greater Yellowlegs) | 0.00 | 0.00 | 0.29 (0.18) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.86 (0.34) | 0.00 | 0.86 (0.26) |
| <i>Quiscalus quiscula</i> (Common Grackle) | 0.29 (0.18) | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 1.29 (1.29) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Dendroica petechia</i> (Yellow Warbler) | 0.14 (0.14) | 0.00 | 0.29 (0.29) | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.29 (0.18) | 0.00 | 0.00 | 0.14 (0.14) |
| <i>Anas platyrhynchos</i> (Mallard) | 0.14 (0.14) | 0.00 | 0.29 (0.29) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.29 (0.29) | 0.00 | 0.43 (0.30) |
| <i>Sayornis phoebe</i> (Eastern Phoebe) | 0.00 | 0.00 | 0.29 (0.18) | 0.00 | 0.14 (0.14) | 0.14 (0.14) | 0.29 (0.29) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Pluvialis squatarola</i> (Black-bellied Plover) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.00 (0.85) |
| <i>Bombicilla cedrorum</i> (Cedar Waxwing) | 0.86 (0.70) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Phalacrocorax auritus</i> (Double-crested Cormorant) | 0.29 (0.18) | 0.29 (0.18) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.29 (0.18) | 0.00 | 0.00 | 0.00 |
| <i>Capodacus mexicanus</i> (House Finch) | 0.57 (0.57) | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) |
| <i>Sterna albifrons</i> (Least Tern) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.57 (0.30) | 0.00 | 0.00 | 0.00 |
| <i>Megasceryle alcyon</i> (Belted Kingfisher) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.43 (0.20) | 0.00 | 0.00 | 0.00 |
| <i>Dendroica coronata</i> (Yellow-rumped Warbler) | 0.00 | 0.00 | 0.43 (0.30) | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Mimus polyglottos</i> (Northern Mockingbird) | 0.14 (0.14) | 0.14 (0.14) | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Geothlypis trichas</i> (Common Yellowthroat) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.29 (0.18) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Larus marinus</i> (Greater Black-backed Gull) | 0.29 (0.18) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Plegadis falcinellus</i> (Glossy Ibis) | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) |
| <i>Empidonax</i> spp. (Flycatcher) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.29 (0.29) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Turdus migratorius</i> (American 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 |
| <i>Florida caerulea</i> (Little Blue Heron) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.14 (0.14) |
| <i>Zenaidura macroura</i> (Mourning Dove) | 0.00 | 0.00 | 0.29 (0.18) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 |
| <i>Pandion haliaetus</i> (Osprey) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Empidonax traillii</i> (Willow Flycatcher) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.29 (0.18) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Buteo striatus</i> (Green Heron) | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Circus cyaneus</i> (Northern Harrier) | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Archilochus colubris</i> (Ruby-throated Hummingbird) | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Icterus galbula</i> (Baltimore Oriole) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Cyanocitta cristata</i> (Blue Jay) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 |
| <i>Toxostoma rufum</i> (Brown Thrasher) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) |
| <i>Chaetura pelagica</i> (Chimney Swift) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Calidris alpina</i> (Dunlin) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 |
| <i>Sturnella magna</i> (Eastern Meadowlark) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Spizella pusilla</i> (Field Sparrow) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Troglodytes aedon</i> (House Wren) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Passerculus sandwichensis</i> (Savannah Sparrow) | 0.00 | 0.00 | 0.14 (0.14) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| <i>Actitis macularia</i> (Spotted Sandpiper) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 (0.14) |

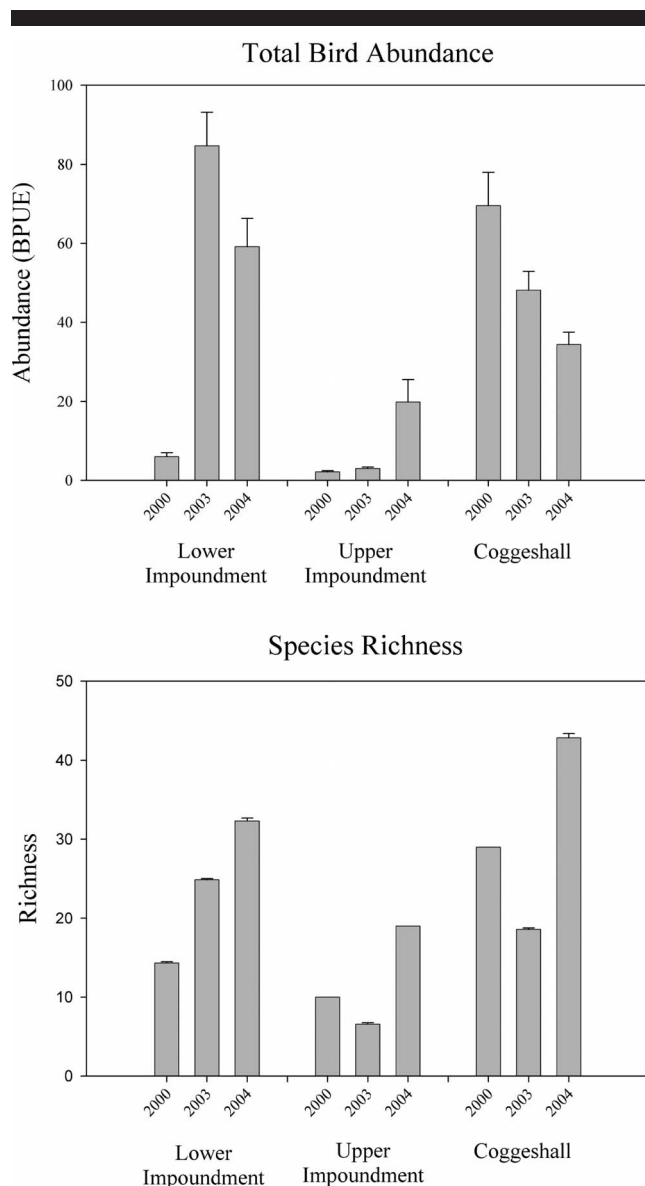


Figure 6. Mean bird abundance and species richness in the three study areas before and after tidal restoration of Potter Pond. Prerestoration conditions were in 2000; postrestoration was in 2003 and 2004. Error bars are one standard error.

2004 than in 2000 (Student's *t* test, $p < 0.005$ for all cases). These patterns indicate the only trend that can be attributed solely to restoration is the increase in richness in the Lower Impoundment during the first year of postrestoration.

DISCUSSION

Initial results from monitoring clearly demonstrate that restoration of the Potter Pond marsh complex improved ecological conditions in this system. The increased tidal exchange with Narragansett Bay improved tidal range in Potter Pond, which flushed out stagnant and decaying mats of macroalgae. *Phragmites* height and cover were significantly re-

duced, and pioneer species such as *Salicornia* spp. and *S. alterniflora* began encroaching on newly exposed mud flats in the Lower Impoundment. Bird abundance (especially shorebird abundance) increased significantly after restoration in the Lower Impoundment. However, the responses of nekton and the emergent marsh vegetation community did not proceed as expected. In contrast to what was observed at other salt marsh restoration sites in Narragansett Bay (RAPOSA, 2002; ROMAN *et al.*, 2002), and despite improved marsh access and water quality, nekton density actually decreased after restoration at Potter Pond. In addition, the composition of the emergent marsh vegetation community in the Lower Impoundment did not change even after 2 years of restoration; this was because most vegetation species typically found in New England salt marshes were already present in this area before restoration. At the very least, these somewhat unexpected findings further illustrate that ecological responses to salt marsh restoration are to some degree site-specific and depend heavily on prerestoration or tide-restricted conditions, making it more difficult to identify generalized ecological responses to restoration and demonstrating that ecological monitoring should be conducted before and after any salt marsh restoration project. In addition, generalized patterns will be all the more difficult to identify unless monitoring at restricted, restoring, reference, and control sites uses standardized or consistent methodologies and is conducted for multiyear times scales (KONISKY *et al.*, 2006).

Some of the more conspicuous ecological changes in Potter Pond were surely in response to the newly formed mudflats that were exposed after tidal restoration. An abundant, mixed-species assemblage of shorebirds arrived during the first year after restoration to feed on the new tidal flats. Although an increase in these birds was expected, the magnitude of the increase was notable. Overall, bird abundance increased by 1400% during the first year of restoration. Abundance decreased somewhat in year two, but was still 1000% higher than it was under tide-restricted conditions. At a marsh restoration site in New Jersey, mudflat area increased after restoration, and this increased the richness and abundance of mudflat foragers, mostly due to large flocks of sandpipers (SEIGEL, HATFIELD, and HARTMAN, 2005). This illustrates that vegetated salt marshes are not the only habitats that are adversely affected by tide-restricting structures; other important estuarine habitats, such as exposed mud flats, should also be considered as targets of tidal restoration.

The new mudflats also facilitated the expansion and colonization of *S. alterniflora* and *Salicornia* spp. By the second year of postrestoration, these species had already firmly established themselves on the flats. In the short term, the Lower Impoundment of Potter Pond should continue to provide foraging habitat for shorebirds, but over the long term it is expected that emergent vegetation will continue to colonize the open mud flats. This should eventually result in a small, well-developed salt marsh with a single major tidal creek bisecting and draining the marsh. The loss of mudflat habitat to vegetation colonization has been linked to a decrease in avian use (EERTMAN *et al.*, 2002), and the increased value of Potter Pond for shorebirds, although impressive initially,

should diminish if marsh development progresses as expected.

The restored tidal patterns in Potter Pond elicited additional ecological changes. Conspicuous among them was the disappearance of large decaying mats of green macroalgae and the decrease or loss of nekton species that are intimately associated with macroalgae. *Cyprinodon variegatus* and *L. parva* are common in submerged aquatic vegetation (SAV) such as eelgrass (*Zostera marina*), widgeongrass (*Ruppia maritima*), and macroalgae (ABLE and FAHAY, 1998; RAPOSA, 2002; RAPOSA, personal observation; RAPOSA and OVIATT, 2000; SOGARD and ABLE, 1991); *Cyprinodon variegatus* density decreased dramatically after restoration of Potter Pond, and *L. parva* disappeared entirely from the pond after only 1 year. Although it was not quantified in this study, some of the decrease in these species may be attributable to increased predation by wading birds. Deeper water and SAV (including macroalgae) both provide a refuge from wading bird predation for small nekton (GAWLIK, 2002; POWELL, 1987; ROZAS and ODUM, 1988). After tidal flow was restored to Potter Pond, macroalgal habitat was lost and shallower water was much more prevalent during ebbing and low tide stages. Both of these conditions are conducive to increased predation by wading birds and may explain the increased abundance of these species (e.g., *E. thula* and *Tringa melanoleuca*) and the decrease in the density of certain nekton species after restoration. Indeed, direct predation by these birds on nekton in shallow water was frequently observed in Potter Pond once macroalgae was lost after restoration (RAPOSA, personal observation).

The overall decrease in nekton density after restoration was not expected and could be misinterpreted as a detrimental ecological response to restoration. However, the decrease in nekton density in this case may have been due to abnormally high numbers of small juveniles of hardy marsh species (e.g., *F. heteroclitus*, *C. variegatus*, and *L. parva*) that took refuge in macroalgae throughout the Lower Impoundment before restoration; when algae was lost because of restored tidal flushing, densities of these species and of nekton overall were significantly reduced. The issue here is not that nekton density decreased, but rather that density alone can sometimes prove to be a poor indicator of ecological function. In the case of Potter Pond under tide-restricted conditions, piscivorous birds and fish were not able to forage extensively on small nekton, resulting in artificially high nekton densities. Larger fish could not enter the pond through the crushed culverts, and wading birds could not effectively forage in the deep, algae-ridden water. The restoration of tidal flow resulted in a decrease in nekton density, but this was surely in part due to increased predation pressure, at least from wading birds. This indicates that density should not be the only indicator of nekton condition in salt marshes, which supports the contention of VAN HORN (1983) that density can be a misleading indicator of habitat quality in general (e.g., high densities are not always better or natural). Other indicators of nekton function, such as richness, biomass, growth rates, and predation rates by larger fish and birds, should also be measured to fully understand the response of nekton to restoration.

A common goal of salt marsh restoration in New England is the reduction in cover and height of *P. australis*, and this goal was achieved in each of the 2 years immediately after restoration of Potter Pond. Both of these parameters decrease sharply as salinities exceed 26 ppt (WARREN *et al.*, 2002), and this level was achieved on average in the Upper Impoundment of Potter Pond after tidal restoration. As live *P. australis* was lost in the Upper Impoundment, it was replaced by an increasing area of open water habitat that was rapidly used by highly mobile nekton such as *F. heteroclitus* and *Pa-laemonetes* spp. These and other nekton species rely heavily on habitats in the upper reaches of salt marshes that retain water at low tide, such as marsh pools and distal ends of small creeks and ditches (ADAMOWICZ, 2002; RAPOSA and ROMAN, 2001; ROZAS, McIVOR, and ODUM, 1988), and the creation of this habitat within the former *P. australis* monoculture was an important result of restoration in this study. Nekton was documented utilizing this new habitat during the summer in this study, and it is expected that species such as *F. heteroclitus* and *C. variegatus* will also use it for overwintering in the marsh (ABLE and FAHAY, 1998; RAPOSA, 2003; SMITH and ABLE, 1994).

In contrast, bird richness, abundance, and community composition remained unchanged in the Upper Impoundment in both years after restoration. This is partly because the large increase in overall bird abundance that was seen in this area in 2004 was mostly due to one flock of *I. bicolor* that was resting on the reeds on a single date. Although there were no overall statistical changes, some subtle effects of restoration on bird use of the Upper Impoundment were apparent. For example, the new edge that formed when some *P. australis* was converted into open water attracted new edge-associated songbirds (e.g., *Dumetella carolinensis*, *Geothlypis trichas*, *D. petechia*, and *Quiscalus quiscula*) to the area after restoration. A nesting pair of *Turdus migratorius* with chicks was even observed in the edge of the *P. australis*. This supports earlier studies showing that low diversity *P. australis* monocultures with little open water are typically used by a bird community dominated by a single species (*A. phoeniceus*) and that increasing habitat heterogeneity after restoration increases marsh use by multiple avian guilds (CRAIG and BEAL, 1992; SEIGEL, HATFIELD, and HARTMAN, 2005).

Phragmites australis has been shown to be important habitat for the songbird *A. phoeniceus* and nesting habitat for wading bird species such as *Egretta caerulea*, *E. thula*, and *Plegadis falcinellus* (PARSONS, 2003; SEIGEL, HATFIELD, and HARTMAN, 2005). The ecological responses to the loss of *P. australis* at Potter Pond further suggest that complete eradication of this species may not be necessary to improve habitat for birds and nekton. Instead, merely increasing habitat heterogeneity by fragmenting a *P. australis* monoculture with open water or tidal creeks can benefit edge-associated bird species and nekton species that are adapted to extreme upstream marsh habitats. Thus, in the case of Potter Pond, if *P. australis* eventually begins to stabilize or even rebound from the initial shock of tidal restoration (e.g., ROMAN *et al.* [2002] observed 2 years of decreasing *P. australis* height after restoration, followed by an increase in year three), the in-

creased habitat benefit can still be maintained as long as the *P. australis*/water mosaic is also maintained.

This study provides an excellent example of the difference between using a nearby or contiguous marsh as a reference site and an experimental control. Coggeshall Marsh and Potter Pond are very different in terms of overall marsh size, habitat composition and configuration, and hydrology. Therefore, in this study, Coggeshall was never intended to function as a reference site to which Potter Pond would become more ecologically similar with time. Indeed, there are no marshes in the immediate vicinity of the study area that are structurally similar enough to Potter Pond to be able to serve as a proper reference site. This problem is not unique to this study; in New Jersey, SEIGEL, HATFIELD, and HARTMAN (2005) were unable to identify a suitable nearby reference site because of issues related to working in an urban environment and because most other marshes in the study area were themselves heavily invaded by *P. australis*. For the Potter Pond study, Coggeshall Marsh was used as the experimental control in a BACI design. Using this approach, this study was able to control for and extract the effects that were due to natural interannual variability, making it possible to clearly illustrate which ecological responses were due to tidal restoration (e.g., the increase in *F. majalis* density in Potter Pond in 2004), and which changes were not (e.g., the decrease in *F. heteroclitus* and *Palaemonetes* spp. densities in Potter Pond in 2004). This demonstrates that even if an appropriate reference site cannot be identified, a nearby marsh can easily serve a useful function by helping to control for natural variability. Without this type of control, it can never be unequivocally determined whether an observed change in a restoration site was actually due to the restoration (e.g., because a control marsh was not used, SEIGEL, HATFIELD, and HARTMAN [2005] were not able to determine whether decreasing sandpiper numbers over time were due to changes in the restoration marsh or to larger regional trends).

In summary, restoration of the degraded Potter Pond system quickly resulted in improved ecological conditions after only 1 year of restored tidal flow, and these improvements persisted into the second year. Some responses occurred immediately after restoration (e.g., nekton use of the new water in the Upper Impoundment), whereas others were more gradual (e.g., *S. alterniflora* and *Salicornia* spp. colonization of the newly exposed mud flats). Continued monitoring over the long term will quantify the long-term rates and magnitudes of these changes and determine if these restoration-induced improvements persist. The unique ecological responses that were observed at Potter Pond further emphasize that high-quality pre- and postrestoration monitoring at experimental as well as control sites should be undertaken with every salt marsh restoration. Despite its small size (2.3 ha), Potter Pond provides an excellent example of the benefits that can be derived from restoring tide-restricted habitats in New England and will hopefully encourage coastal resource managers to further seek to restore degraded sites regardless of their size. This is especially important in Rhode Island where the mean size of all marshes in Narragansett Bay is 2 ha (TROCKI, 2003). Larger restorations may result in more publicity, and larger restoring marshes may more effectively provide some

functions (e.g., storm buffering) than smaller marshes, but restoration of multiple smaller sites can also lead to improved estuarine conditions in the aggregate.

ACKNOWLEDGMENTS

Funding for the restoration was provided by the USDA NRCS, Fish America, the U.S. Fish and Wildlife Service, and the NOAA Restoration Center. Funding for ecological monitoring was provided through a grant from NOAA Estuarine Reserves Division to the NBNERR. Robin Weber and Matt Rehor assisted with collecting postrestoration water quality data. Tom Kutcher assisted with field work and reviewed an early draft of the manuscript.

LITERATURE CITED

- ABLE, K.W. and FAHAY, M.P., 1998. *The First Year in the Life of Estuarine Fishes in the Middle Atlantic Bight*. New Brunswick, New Jersey: Rutgers University Press, 342p.
- ADAMOWICZ, S.C., 2002. New England Salt Marsh Pools: Analysis of Geomorphic and Geographic Parameters, Macrophyte Distribution, and Nekton Use. Kingston, Rhode Island: University of Rhode Island, Ph.D. thesis, 207p.
- BARRETT, N.E. and NIERING, W.A., 1993. Tidal marsh restoration: trends in vegetation change using a geographical information system (GIS). *Restoration Ecology*, 1(1), 18–28.
- BURDICK, D.M.; DIONNE, M.; BOUMANS, R.M., and SHORT, F.T., 1997. Ecological responses to tidal restorations of two northern New England salt marshes. *Wetlands Ecology and Management*, 4(2), 129–144.
- CRAIG, R.J. and BEAL, K.G., 1992. The influence of habitat variables on marsh bird communities of the Connecticut River Estuary. *Wilson Bulletin*, 104(2), 295–311.
- EERTMAN, R.H.M.; KORNMAN, B.A.; STIKVOORT, E., and VERBEEK, H., 2002. Restoration of the Sieperda Tidal Marsh in the Scheldt Estuary, the Netherlands. *Restoration Ecology*, 10(3), 438–449.
- ELZINGA, C.L.; SALZER, D.W., and WILLOUGHBY, J.W., 1998. Measuring and Monitoring Plant Populations. BLM National Applied Resources Center, Report BLM/RS/ST-98/005+1730. Denver, Colorado: USDI, 492p.
- GAWLIK, D.E., 2002. The effects of prey availability on the numerical response of wading birds. *Ecological Monographs*, 72(3), 329–346.
- GRAY, A.; SIMENSTAD, C.A.; BOTTOM, D.L., and CORNWELL, T.J., 2002. Contrasting functional performance of juvenile salmon habitat in recovering wetlands of the Salmon River Estuary, Oregon, U.S.A. *Restoration Ecology*, 10(3), 514–526.
- HELTSHE, J.F. and FORRESTER, N.E., 1983. Estimating species richness using the jackknife procedure. *Biometrics*, 39(1), 1–12.
- HOWE, R.W.; NEIMI, J.G.; LEWIS, S.J., and WELSH, D.A., 1997. A standard method for monitoring songbird populations in the Great Lakes region. *Passenger Pigeon*, 59(3), 183–194.
- JIVOFF, P.R. and ABLE, K.W., 2003. Blue crab, *Callinectes sapidus*, response to the invasive common reed, *Phragmites australis*: abundance, size, sex ratio, and molting frequency. *Estuaries*, 26(2), 587–595.
- KONISKY, R.A.; BURDICK, D.M.; DIONNE, M., and NECKLES, H.A., 2006. A regional assessment of salt marsh restoration and monitoring in the Gulf of Maine. *Restoration Ecology*, 14(4), 516–525.
- LLANSO, R.J.; BELL, S.S., and VOSE, F.E., 1998. Food habits of red drum and spotted seatrout in a restored mangrove impoundment. *Estuaries*, 21(2), 294–306.
- MYSHRALL, D.H.A.; GOLET, F.C.; PATON, P.W.C., and TEFPT, B.C., 2000. Salt Marsh Restoration Monitoring at the Galilee Bird Sanctuary, Narragansett, R.I. Narragansett, Rhode Island: RI Department of Environmental Management Division of Fish and Wildlife Report, 94p.
- PARSONS, K.C., 2003. Reproductive success of wading birds using

- Phragmites* marsh and upland nesting habitats. *Estuaries*, 26(2), 596–601.
- PORTNOY, J.W., 1991. Summer oxygen depletion in a diked New England estuary. *Estuaries*, 14(2), 122–129.
- PORTNOY, J.W., 1999. Salt marsh diking and restoration: biogeochemical implications of altered wetland hydrology. *Environmental Management*, 24(1), 111–120.
- PORTNOY, J.W. and GIBLIN, A.E., 1997. Effects of historic tidal restrictions on salt marsh sediment chemistry. *Biogeochemistry*, 36(3), 275–307.
- POWELL, G.V.N., 1987. Habitat use by wading birds in a subtropical estuary: implications for hydrography. *The Auk*, 104(4), 740–749.
- RAPOSA, K. B., 2002. Early responses of fishes and crustaceans to restoration of a tidally restricted New England salt marsh. *Restoration Ecology*, 10(4), 665–676.
- RAPOSA, K.B., 2003. Overwintering habitat selection by the mummichog, *Fundulus heteroclitus*, in a Cape Cod (USA) salt marsh. *Wetlands Ecology and Management*, 11(3), 175–182.
- RAPOSA, K.B. and OVIATT, C.A., 2000. The influence of contiguous shoreline type, distance from shore, and vegetation biomass on nekton community structure in eelgrass beds. *Estuaries*, 23(1), 46–55.
- RAPOSA, K.B. and ROMAN, C.T., 2001. Seasonal habitat-use patterns of nekton in a tide-restricted and unrestricted New England salt marsh. *Wetlands*, 21(4), 451–461.
- RAPOSA, K.B. and ROMAN, C.T., 2003. Using gradients in tidal restriction to evaluate nekton community responses to salt marsh restoration. *Estuaries*, 26(1), 98–105.
- RAPOSA, K.B.; ROMAN, C.T., and HELTSHE, J.F., 2003. Monitoring nekton as a bioindicator in shallow estuarine habitats. *Environmental Monitoring and Assessment*, 81(1–3), 239–255.
- ROMAN, C.T.; GARVINE, R.W., and PORTNOY, J.W., 1995. Hydrologic modeling as a predictive basis for ecological restoration of salt marshes. *Environmental Management*, 19(4), 559–566.
- ROMAN, C.T.; NIERING, W.A., and WARREN, R.S., 1984. Salt marsh vegetation change in response to tidal restriction. *Environmental Management*, 8(2), 141–150.
- ROMAN, C. T.; RAPOSA, K. B.; ADAMOWICZ, S. C.; JAMES-PIRRI, M. J., and CATENA, J.G., 2002. Quantifying vegetation and nekton response to tidal restoration of a New England salt marsh. *Restoration Ecology*, 10(3), 450–460.
- ROZAS, L.P.; McÍVOR, C.C., and ODUM, W.E., 1988. Intertidal rivulets and creekbanks: corridors between tidal creeks and marshes. *Marine Ecology Progress Series*, 47(3), 303–307.
- ROZAS, L.P. and ODUM, W.E., 1988. Occupation of submerged aquatic vegetation by fishes: testing the roles of food and refuge. *Oecologia*, 77(1), 101–106.
- SEIGEL, A.; HATFIELD, C., and HARTMAN, J.M., 2005. Avian response to restoration of urban tidal marshes in the Hackensack Meadowlands, New Jersey. *Urban Habitats*, 3(1), 87–116.
- SINICROPE, T.L.; HINE, P.G.; WARREN, R.S., and NIERING, W.A., 1990. Restoration of an impounded salt marsh in New England. *Estuaries*, 13(1), 25–30.
- SMITH, K.J. and ABLE, K.W., 1994. Salt-marsh tide pools as winter refuges for the mummichog, *Fundulus heteroclitus*, in New Jersey. *Estuaries*, 17(1), 226–234.
- SOGARD, S.M. and ABLE, K.W., 1991. A comparison of eelgrass, sea lettuce macroalgae, and marsh creeks as habitats for epibenthic fishes and decapods. *Estuarine, Coastal and Shelf Science*, 33(5), 501–519.
- STEWART-OATEN, A.; MURDOCH, W.M., and PARKER, K.R., 1986. Environmental impact assessment: “pseudoreplication” in time? *Ecology*, 67(4), 929–940.
- TEO, S.L.H. and ABLE, K.W., 2003. Growth and production of the mummichog (*Fundulus heteroclitus*) in a restored salt marsh. *Estuaries*, 26(1), 51–63.
- THOM, R.M.; ZEIGLER, R., and BORDE, A.B., 2002. Floristic development patterns in a restored Elk River estuarine marsh, Grays Harbor, Washington. *Restoration Ecology*, 10(3), 487–496.
- TROCKI, C. L., 2003. Patterns of Salt Marsh and Farmland Use by Wading Birds in Southern Rhode Island. Kingston, Rhode Island: University of Rhode Island, M.S. thesis, 97p.
- UNDERWOOD, A.J., 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. *Journal of Experimental Marine Biology and Ecology*, 161(2), 145–178.
- VAN HORN, B., 1983. Density as a misleading indicator of habitat quality. *Journal of Wildlife Management*, 47(4), 893–901.
- VOSE, F.E. and BELL, S.S., 1994. Resident fishes and macrobenthos in mangrove-rimmed habitats: evaluation of habitat restoration by hydrologic modification. *Estuaries*, 17(3), 585–596.
- WARREN, R.S.; FELL, P.E.; ROZSA, R.; BRAWLEY, A.H.; ORSTED, A.C.; OLSON, E.T.; SWAMY, V., and NIERING, W.A., 2002. Salt marsh restoration in Connecticut: 20 years of science and management. *Restoration Ecology*, 10(3), 497–513.