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Ecological responses to tidal restorations of two northern New England salt marshes

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Abstract

Efforts are underway to restore tidal flow in New England salt marshes that were negatively impacted by tidal restrictions. We evaluated a planned tidal restoration at Mill Brook Marsh (New Hampshire) and at Drakes Island Marsh (Maine) where partial tidal restoration inadvertently occurred. Salt marsh functions were evaluated in both marshes to determine the impacts from tidal restriction and the responses following restoration. Physical and biological indicators of salt marsh functions (tidal range, surface elevations, soil water levels and salinities, plant cover, and fish use) were measured and compared to those from non-impounded reference sites. Common impacts from tidal restrictions at both sites were: loss of tidal flooding, declines in surface elevation, reduced soil salinity, replacement of salt marsh vegetation by fresh and brackish plants, and loss of fish use of the marsh.

Water levels, soil salinities and fish use increased immediately following tidal restoration. Salt-intolerant vegetation was killed within months. After two years, mildly salt-tolerant vegetation had been largely replaced in Mill Brook Marsh by several species characteristic of both high and low salt marshes. Eight years after the unplanned, partial tidal restoration at Drakes Island Marsh, the vegetation was dominated by *Spartina alterniflora*, a characteristic species of low marsh habitat.

Hydrologic restoration that allowed for unrestricted saltwater exchange at Mill Brook restored salt marsh functions relatively quickly in comparison to the partial tidal restoration at Drakes Island, where full tidal exchange was not achieved. The irregular tidal regime at Drakes Island resulted in vegetation cover and patterns dissimilar to those of the high marsh used as a reference. The proper hydrologic regime (flooding height, duration and frequency) is essential to promote the rapid recovery of salt marsh functions. We predict that functional recovery will be relatively quick at Mill Brook, but believe that the habitat at Drakes Island will not become equivalent to that of the reference marsh unless the hydrology is further modified.

Introduction

Impacts from development have destroyed much of the salt marsh area in New England since colonial times (Niering and Bowers 1966, Mitsch and Gosselink 1986). Although they have been protected in the United States since 1972 from direct anthropogenic impacts such as filling (Section 404 of the Clean Water Act), salt marshes continue to be impacted indirectly by the long-term effects of coastal structures (*e.g.*, roads, dikes) that result in tidal restrictions (Niering and Warren 1980, Roman *et al.* 1984, Boumans and Day 1994, Roman *et al.* 1995). Studies in Connecticut and New Hampshire indicate that 10% and 20%, respectively, of the remaining marsh area is currently being degraded by the indirect impacts associated with inadequate tidal exchange (Roman *et al.* 1984, USDA SCS 1994). The most notable impact is the loss of salt marsh vegetation.

Two primary determinants of salt marsh vegetation within a climatic region are the flooding and salinity regimes (Niering and Warren 1980, Zedler *et al.* 1982). Tides flood low-lying salt marshes with saline waters. Therefore, one would expect the loss of salt marsh vegetation following the restriction of tidal exchange and subsequent salinity decrease and where plants that are competitively superior under less saline conditions exclude salt marsh species. The reduction of tidal exchange in New England marshes has been linked to vegetation die-back and replacement of typical salt marsh plants (*Spartina alterniflora* and *Spartina patens*) by invasive species such as *Phragmites australis* and *Lythrum salicaria* (Roman *et al.* 1984, Shisler 1990, Sinicrope *et al.* 1990, USDA SCS 1994, Burdick and Dionne 1994). Tidal restrictions reduce not only saltwater flooding, but also sediment inputs (Boumans and Day 1994), prevent landward migration of coastal marshes with rising sea level (Pethick 1993, Bird 1993), and reduce biological exchange with adjacent estuarine or coastal waters (Herke *et al.* 1992). Thus salt marsh habitat and associated functions will continue to be lost in New England due to existing tidal restrictions.

Resource managers have determined that the impacts from tidal restrictions are significant and are promoting the hydrologic restoration of impacted salt marshes. However, there is little information available to guide these restoration efforts. Most quantitative assessments of impacts and responses to restoration are found in conference proceedings and agency reports (Josselyn *et al.* 1990, Shisler 1990). Systematic studies of the impacts caused by tidal restrictions and responses following restoration are needed. Re-establishing tidal exchange is more easily achieved than the broader habitat restoration goals for such projects (Race 1985, Rozsa 1988, Frenkel and Morlan 1991), thus highlighting the need for a better understanding of ecological responses to determine appropriate solutions. Project goals have typically been to restore habitats to the pre-restriction conditions (Zedler *et al.* 1982). In the New England region, common mechanisms to improve tidal exchange include the installation or modification of culverts under roads and railways and the creation or re-establishment of tidal creeks blocked or filled by soil berms or dikes.

We examined the ecology of two salt marshes following changes to road culverts that restored their tidal exchange using an approach that measures indicators of marsh functions (Zedler 1992). Tidal restrictions reduce the ability of a marsh to perform several important functions, including support of salt marsh vegetation, the associated habitat, and secondary consumers. We examined functional indicators to determine impacts from restrictions and responses following restoration of tides. Our set of hypotheses tested whether tidal restriction led to changes in the marsh (pre-restored *vs.* reference marshes under the assumption that the reference marshes found downstream represent the original pre-restriction conditions of the impacted marshes), and whether re-establishment of tidal exchange restores the ability of a marsh to perform these functions (post-restored *vs.* reference marsh). Using our indicators, we tested these hypotheses separately for each of the two systems (Mill Brook in New Hampshire and Drakes Island in Maine) and then compared the results to contrast approaches to tidal restoration.

Although we did not have the opportunity to design and carry out a comprehensive pre- and post-restoration study, we used our existing data (Short 1987, Burdick and Dionne 1994), along with a sampling program to assess the two restorations and develop an ecological basis for tidal restoration of salt marshes. Improvement of salt marsh functions due to major hydrologic changes is indicated by rapid changes in tidal flooding, soil water depths and salinities, and fish use (days to months), whereas several years may be required to document the development of plant and fish communities (Fig. 1). Even more time is required for the surface elevations of impacted marshes to become re-established. We measured ecological changes in marshes within several years of hydrologic restoration to determine the current functional benefits, to predict the long term functional benefits, and to assess whether further hydrologic modifications are needed to support the functions of salt marsh habitats.

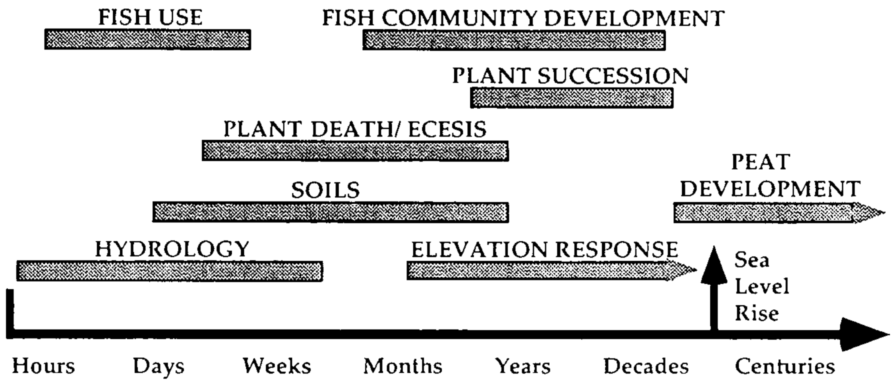

Fig. 1. Hypothesized time scales of processes related to indicators of marsh functions in restored and created salt marshes.

Methods

Study Area

Drakes Island Marsh is in the Wells National Estuarine Research Reserve at Wells, Maine (Fig. 2). The marsh formed landward of a barrier beach system in a lagoon estuary approximately 4,000 years BP (Kelley *et al.* 1995). Use of the 40 hectare marsh as a pasture led to hydrologic manipulations since *ca.* 1848 when a dike was built. A road providing access to the Drakes Island beach and running parallel to the dike had a box culvert with a water control structure that operated from the 1920s to the 1950s. Another beach access road was built at the north end of the marsh, preventing spring tides entering the marsh from the lagoon estuary to the east (Little River Estuary). The sedimentary record indicates that the impacted area was originally dominated by high marsh, and was similar to the adjacent salt marsh found downstream of the road today (D. Belnap pers. comm.). We used this downstream marsh (described previously by Kelley *et al.* 1995) as our reference marsh. The present culvert under Drakes Island Road was installed in the 1950s as a 1.2 meter diameter pipe with a flap gate that prevented salt water from entering the marsh. Subsequent repairs have resulted in a narrower cross section (0.9 m diameter) for a portion of the pipe. Storm tides flooded the marsh infrequently (*ca.* 1/decade). The flap gate fell off in spring, 1988. Salt intolerant vegetation was soon killed across 40% of the 40 ha site. March 1988 is considered the date that this inadvertent and unplanned tidal restoration occurred. Subsequent efforts to replace the tide gate were discouraged by permit reviews (Section 404 of the Clean Water Act).

Mill Brook Marsh formed in a minor fluvial valley near the mouth of the Squamscott River that flows into the southwest corner of Great Bay in Stratham, New Hampshire (Fig. 2). When an access road cutting across the valley to Stuart Farm was upgraded to accommodate larger milk trucks in 1970, its wooden bridge was removed and the tidal creek was replaced by a pipe culvert with a flap gate. Over the years, the 4.5 ha area became a wet meadow that flooded annually in the spring following snow melt and occasionally by salt water from storm tides over the road. Like the Drakes Island Marsh, we believe that the impacted marsh was originally dominated by a high marsh plant community with low marsh along creek banks, and that it was similar to the reference marsh found downstream across the road. Remnant patches of *S. patens* were found at both impacted sites before restoration. A 2.1 meter diameter arched culvert was installed and the flap gate on the existing culvert was removed in October 1993 to restore salt marsh habitat upstream of the road at Mill Brook.

Water levels and surface elevations

Pressure transducers were positioned at Drakes Island Marsh on both sides of the culvert to measure water levels between April 30 1996, and May

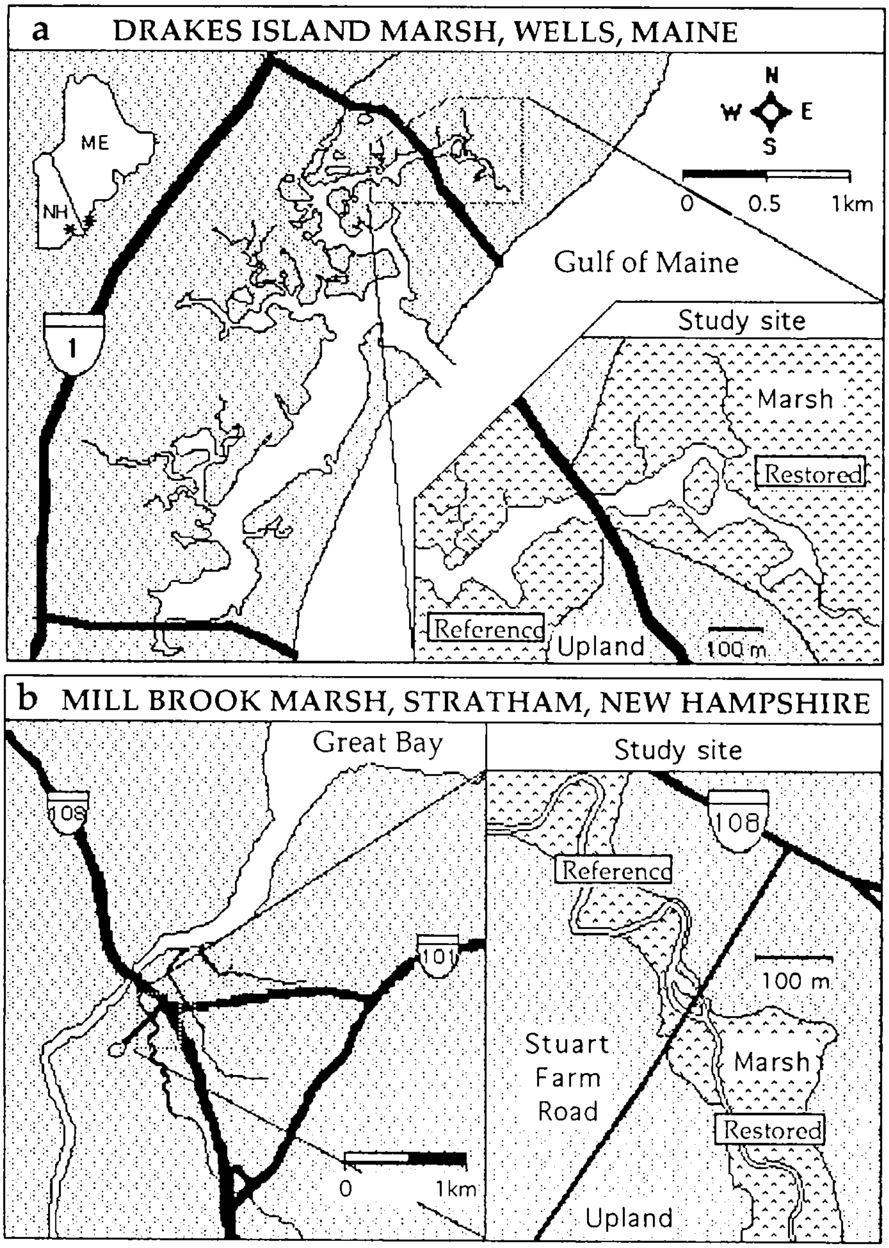

Fig. 2. Study sites: a. Drakes Island Marsh, Wells National Estuarine Research Reserve, Wells, Maine; b. Mill Brook Marsh at Stuart Farm, Great Bay National Estuarine Research Reserve, Stratham, New Hampshire.

7, 1996. The pressure output was recorded and converted to water depth every 15 minutes using a YSI 6000 UPG. Marsh elevations were surveyed in May 1996, along one transect at the restored and reference sites in Drakes Island Marsh, and along two transects at the restored and reference sites in the Mill Brook Marsh. Elevations are reported relative to the top of the culverts. The transects ran perpendicular to the main tidal channel and culvert from one marsh/upland interface to the other. The first point was chosen haphazardly, with elevations and plants then surveyed every 10 m and at important transition points (upper and lower edges of the high marsh, lower edge of the low marsh, and the channel center). Downstream of the restored area, the reference marsh was divided into the two major plant communities: those dominated by *S. patens* were called "high marsh" while areas dominated by the tall form of *S. alterniflora* were defined "low marsh". No clear zonation was found upstream of the culverts in the restored marsh areas and the community was simply called "new" marsh.

Well water depths and salinities

Two different approaches were used to measure soil water salinities. In 1986 and 1992 at Drakes Island Marsh, salinity of interstitial water was sampled using sippers (Short *et al.* 1985) at a depth interval from -7.5 to -12.5 cm. All subsequent salinity samples and water depth measurements were obtained from a randomly selected subset of permanent wells. The wells were made of 2.2 cm ID PVC pipe that sampled water from a depth interval of -5 to -20 cm. The wells were capped to prevent rain water inflow, yet they allowed for free gas exchange. Salinity of each sample was measured in the field using a temperature-corrected optical refractometer. Sampling was performed at low tides on 2 to 6 occasions throughout each of the 1993 to 1995 growing seasons and included both spring and neap tides. A minimum of 4 samples were collected to characterize a marsh area on any given date and the data were analyzed as means of these samples. For each marsh, means of water depths and salinities for the pre- and post-restored sites were compared to those from reference sites using One-Way ANOVA. Means were compared using Scheffe's test because the sample sizes had a wide range (from n=29 and 34 from the reference low marsh to n=245 and 259 from the post-restoration marsh for water table depth and salinity, respectively).

Vegetation

A total of five vegetation samples were collected in 1986 at Drakes Island Marsh from two sites within the impounded marsh using 1/16 m^2 clip plots. The percent species cover was estimated from stem counts and dry weights. All subsequent samples (1993 through 1995) were obtained by direct estimates of percentage cover by species for single or paired 1 m^2 areas at 4 to 18 locations to characterize each marsh area (impacted/restored, reference low, and reference high marshes). Cover estimates were made in August or September, the time of peak standing crop, except for the 1994 Mill Brook data which were collected in late June. Plant nomenclature is described as in Tiner (1987).

Nekton

Fish and shrimp abundance, biovolume, and species composition were sampled using fyke nets. The fyke nets were equipped with two 15-m wings attached to the largest (1.2 m) of a series of four successively smaller square fykes (1.27 cm mesh) leading to a cod end (0.63 cm mesh). To sample, a fyke net was set up on the lower edge of the marsh with the cod end anchored in a tidal channel. The wings were staked away from the first fyke at 45 degree angles so that fish restrained by the net would swim towards the nets. At high tide, the area of marsh covered with water between the wings of the net was staked and measured to determine the area of marsh fished. Once the tide fell below the nets connecting the fykes, the catch was collected from the cod end, placed in water-filled buckets, sorted to species, and counted. The lengths of up to 30 of each species were sampled haphazardly from each bucket with a dip net, and biovolumes of each estimated by displacement in a graduated cylinder.

Two evening fish collections were made at reference and restored areas in the fall of 1993 following restoration at Mill Brook Marsh. A more complete program sampled fish at both Mill Brook and Drakes Island in June and September of 1995. Nets were set during rising spring tides, and fish were caught simultaneously in restored and reference areas. Each sampling period included an evening and a daytime tide; at Mill Brook an extra set of day and evening samples was taken in June 1995. The area of marsh fished ranged from 23 to 800 m^2; one sample (reference area of Mill Brook) was removed from the data set due to the small area of marsh fished (6 m^2).

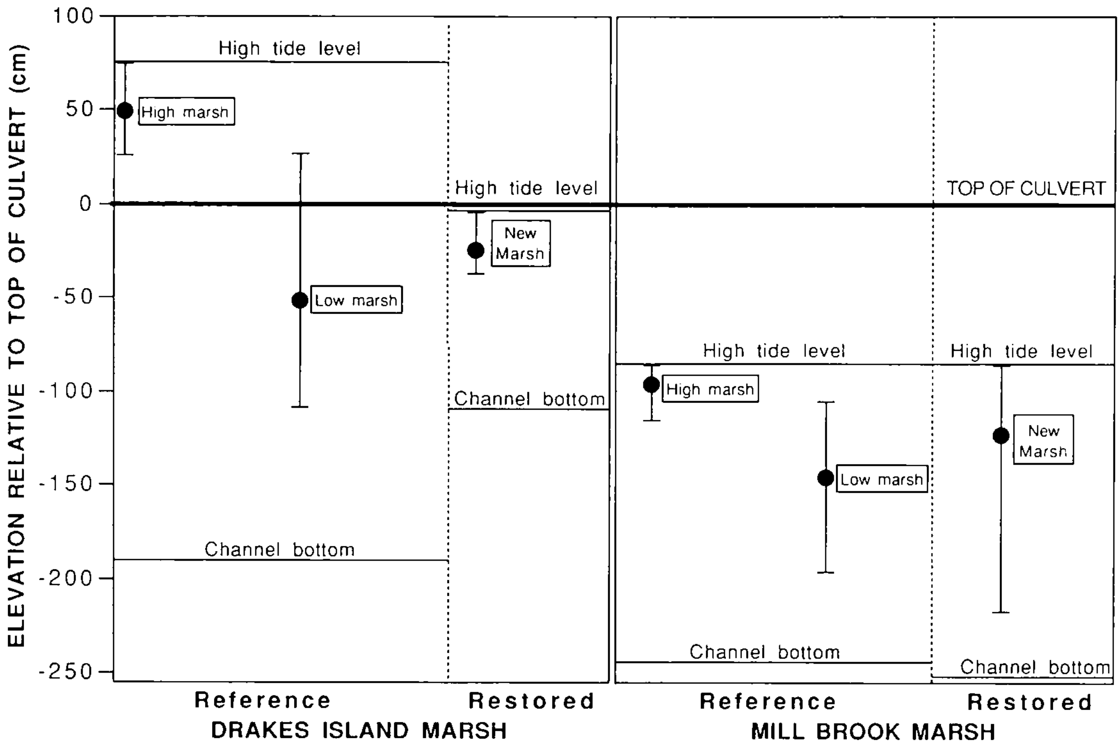

Fig. 3. Mean elevations (•) relative to the top of the upstream side of the culverts at Drakes Island Marsh and Mill Brook Marsh. Vertical bars show the range of elevations where habitats were observed. High and low marsh communities were not distinguishable in the restored "new" marshes. High tide elevations were defined by wrack lines and plant zonation (upper edge of *Juncus gerardii*, where present). The lower end of the tidal range was defined by the channel bottom.

Results

Surface elevations relative to tidal levels

The reference areas in both marshes had clearly defined high and low marsh communities, reflecting distinct elevation ranges (Fig. 3). The restored areas, however, had poorly developed zonation, and we were unable to differentiate between high and low marsh communities. Although the elevational range of the reference marsh vegetation was smaller at Mill Brook than Drakes Island (120 *vs.* 180 cm), the elevational range of the restored marsh at Mill Brook was 120 cm. In contrast, the elevational range at the restored marsh at Drakes Island was less than 30 cm (Fig. 3). In addition, the mean elevations for the broad flat areas of the marsh (excluding data from the creeks and transition zones used in Fig. 3) were 11 cm lower on the impacted side at Mill Brook Marsh and 73 cm lower on the impacted side at Drakes Island Marsh when compared to reference areas, suggesting subsidence had occurred at both sites.

The Spring tidal range in the Drakes Island restored site was 30 cm, which was much smaller than in the reference site (220 cm, Fig. 4). Visual observations indicated that such a difference did not occur in Mill Brook where the culvert cross section is very large. Using the upper edge of the marsh defined by vegetation changes and wrack-lines, we found that the high tide line in the Drakes Island restored site was 85 cm lower than the reference site, while no large difference was observed between the Mill Brook restored and reference marsh high tide elevations (Fig. 3). Similarly, mean channel depth was shallower in the Drakes Island restored marsh than at the reference site, while such a difference was not found in the Mill

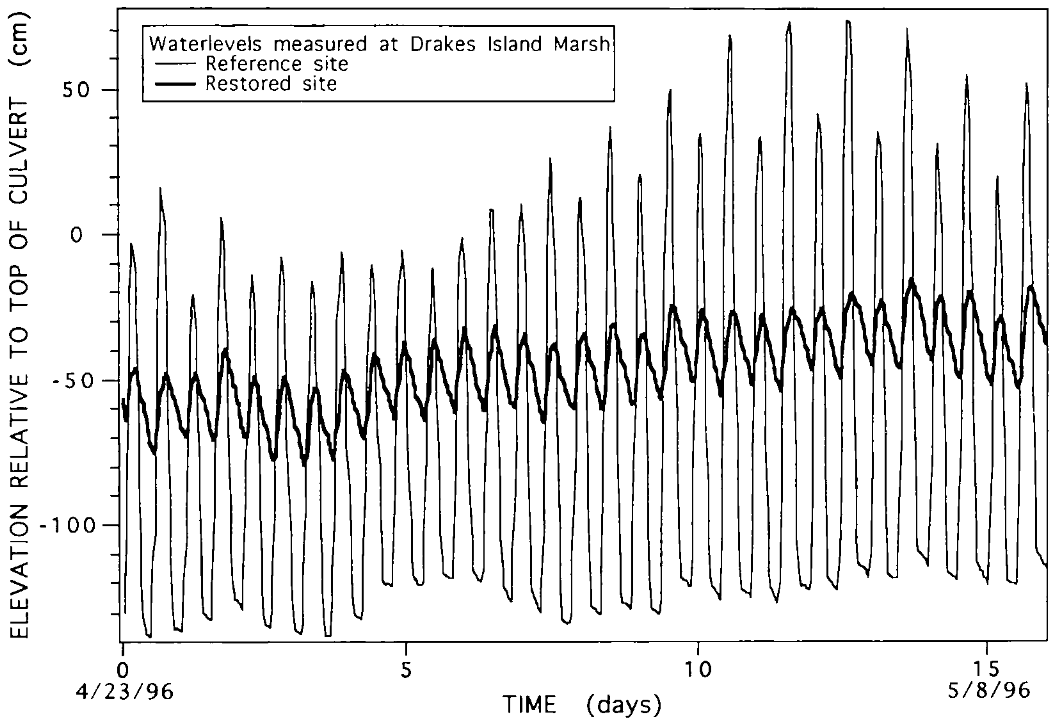

Fig. 4. Water levels in the main tidal channel of restored and reference areas over a two week period at Drakes Island Marsh. Elevations are relative to the top of the upstream side of the culvert.

Brook Marsh. Clearly the range of marsh surface and tidal levels upstream of the culvert is compressed relative to downstream at Drakes Island but not at Mill Brook. We believe this is a reflection of the tidal hydrology following partial tidal restoration. The causeway across the marsh still results in a large head of water at most high tides downstream and at all low tides upstream of the culvert.

Furthermore, the tidal levels within the restored area shifted from -50 to -75 cm during neap tides up to -50 to -25 cm during spring tides relative to the top of the culvert at Drakes Island (Fig. 4). Thus the tidal regime could be interpreted as occurring fortnightly rather than semidiurnally, which is normal for marsh environments in the Gulf of Maine. The elevation of the vegetated marsh surface (-40 to -10 cm on Fig. 3) occurs where the sediments are continually exposed during neap tides and experience semi-diurnal tides during spring tide periods.

Well water depth and salinity

Restoration of tidal exchange at Mill Brook increased water levels in sampling wells to levels that were similar to those found at the downstream reference marsh (Fig. 5). Although no pre-restoration water depths were measured in Drakes Island Marsh, water depths there were similar to those of reference areas during the post-restoration period. In general, the water table depths were about 3 cm greater at Mill Brook than at the Drakes Island Marsh.

The salinity of the well water showed dramatic increases after restoration for both marshes (Fig. 6). The mean salinities for both restored marshes suggest these areas may be slightly more saline than reference marshes (2 to 5 ppt). This difference was statistically significant for Mill Brook ($P<0.05$), but not for Drakes Island. Overall, the Mill Brook Marsh was less saline than the Drakes Island Marsh, probably because of the influence of the nearby Squamscott River, and because the

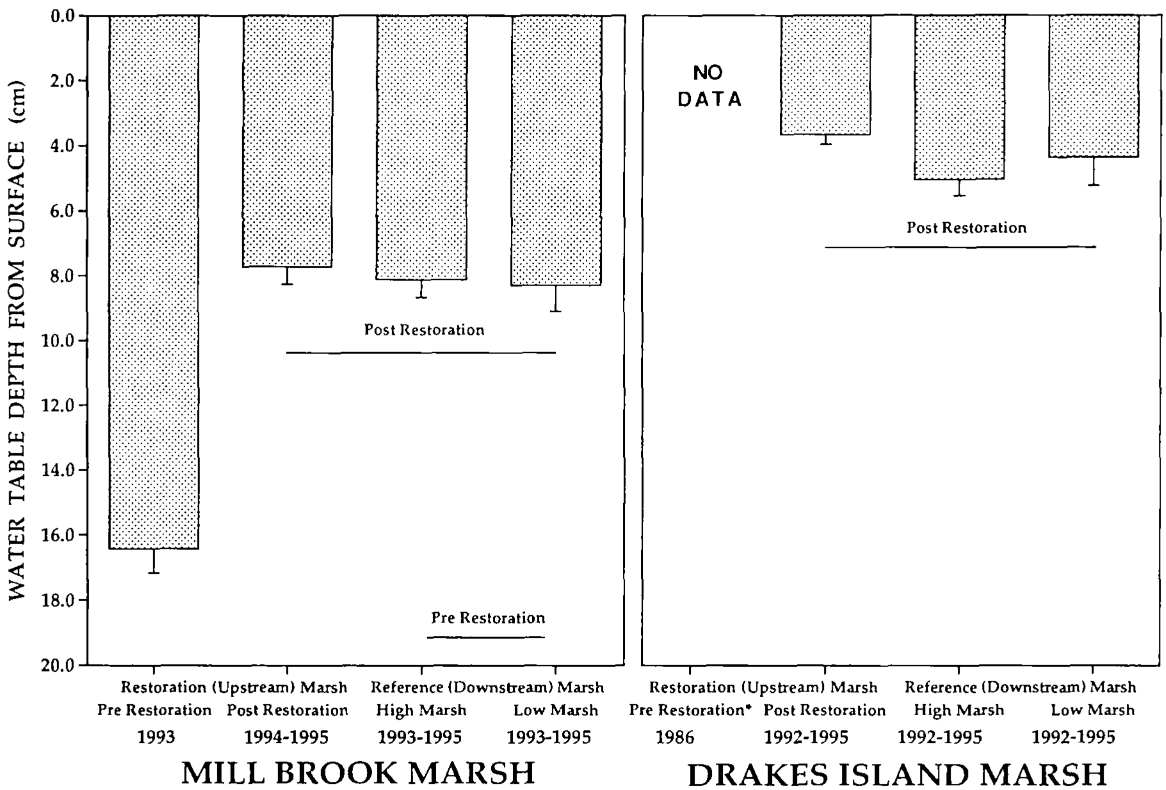

Fig. 5. Depths to the water table from the surface of the marsh using 20 cm deep wells are shown as means with standard errors. For each marsh, the mean of water depths for the upstream restoration site at each period (pre-restoration and post-restoration) was compared to means from the downstream reference sites using ANOVA. Horizontal bars indicate means that are not significantly different (P>0.05 by Scheffes test) for each set of comparisons: pre-restoration and post restoration.

downstream source of salt water, Great Bay, ranges from 5 to 30 ppt.

Vegetation

Drakes Island Marsh

Pre-restoration conditions were documented in a 1986 vegetation survey that showed *Typha latifolia* and *Spartina pectinata* dominated the marsh which supported other salt-tolerant species as well: *Scirpus robustus*, *Triglochin maritimum*, and *Baccharis halimifolia* (Fig. 7). Because of the saline nature of the site (from storm overwash), the system was then characterized as a fresh to brackish non-tidal marsh (Short 1987). Salt-intolerant vegetation was killed over several years following the loss of the flap gate from the culvert and tidal restoration in 1988 (N. McReel and C. Ferris, personal communication). By the sixth growing season (1993), cover by vascular plants was about 70%. The quantitative cover data collected from 1993 to 1995 shows increased colonization by *Spartina alterniflora* and declines in *Vaucherria* sp., filamentous green algae, *Salicornia europaea*, and *Atriplex patula* (Fig. 7). Some changes were due to small scale local variation (*e.g.*, changes in *S. pectinata*) and may not reflect long-term trends.

Downstream of the culvert, the reference area is primarily high marsh. Low-marsh vegetation (tall-form *S. alterniflora*) is found along channel bars, creek banks, and the drainage paths distributed throughout the marsh. Lower areas of the high marsh that hold water over low tides support the stunted short form of *S. alterniflora*. The high marsh is dominated by *Spartina patens*, but supports a rich mixture of other grasses and forbs: *S. europaea*, *Plantago maritima*, *T. maritimum*, *A.*

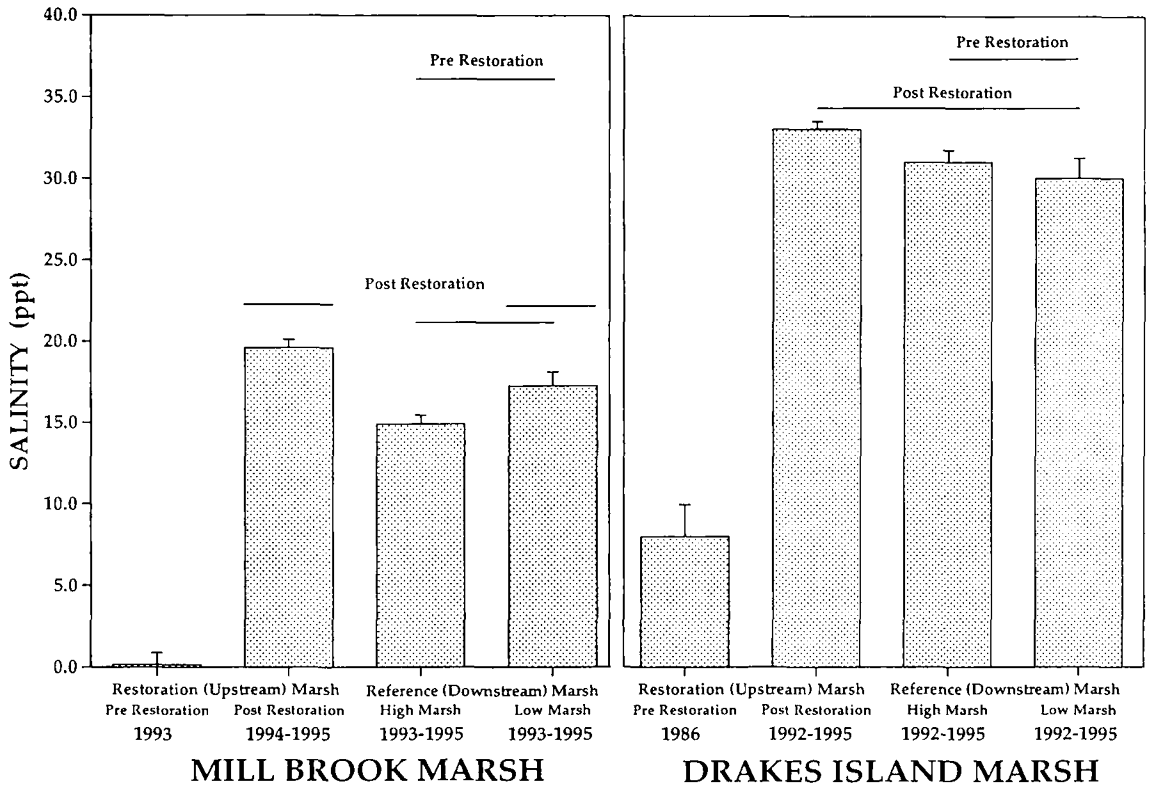

Fig. 6. Soil water salinities in 20 cm deep wells are shown as means with standard errors. For each marsh, the mean of water depths for the upstream restoration site at each period (pre-restoration and post-restoration) was compared to means from the downstream reference sites using ANOVA. Horizontal bars indicate means that are not significantly different (P>0.05 by Scheffes test) for each set of comparisons: pre-restoration and post restoration.

patula (Fig. 7) as well as *Limonium nashii*, *Glaux maritima*, *Distichlis spicata*, and *Agalinis maritima*. The restored marsh, dominated by *S. alterniflora*, more closely resembled low marsh than high marsh in 1995 (Fig. 7). Eight years following the loss of the flap gate, high marsh species comprised only 10% of the restored marsh cover, yet this area was surely high marsh when it was first impounded for grazing cattle over a century ago. *Spartina patens* is more common on hummocks and natural levees, *S. pectinata* and *Typha* only occur far from the main channel, but *S. alterniflora* occurs throughout the system. The elevation range of the restored system is small (Fig. 3) and zonation is not clear.

Mill Brook Marsh

Pre-restoration conditions were assessed in 1993 and species characteristic of a fresh water, irregularly flooded meadow were found: *Lythrum salicaria* was the dominant overstory species, with an understory of aster and other forbs and several grasses including *Agropyron repens*, *Agrostis stolonifera*, and *Festuca* spp. (Fig. 8). Small extant populations of *S. pectinata* and *S. patens* were the only salt tolerant plants found upstream and none fell within samples. In June 1994, nine months following restoration, loss of all asters, *Lythrum* and much of the meadow grasses was evident. Overall, plant cover was reduced by 50% (Fig. 8), although this may be because plants were surveyed in spring rather than summer. New populations of *Juncus gerardii* appeared, but no *Vaucherria* was found. The following year meadow grasses were still declining, and overall cover by vascular plants fell to 40%. However, *S. alterniflora*, *S. europaea*, *A. patula*, *Scirpus* sp., and *Vaucherria* appeared (Fig. 8), along with new populations of *J. gerardii* and *S. patens*.

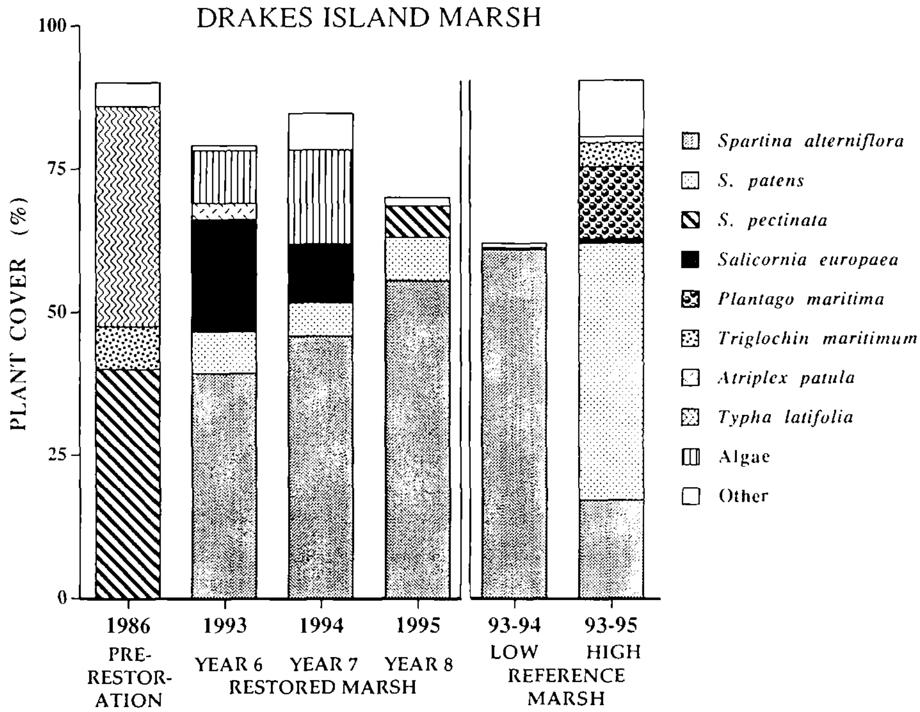

Fig. 7. Percentage cover of vegetation in late summer at Drakes Island Marsh from 1986 to 1995. Data from the pre-restoration site were estimated from five clip plots of 0.0625 m^2; all other data are means of direct estimates from ten to eighteen, 1.0 m^2 samples. Yearly data from the restored area are shown to illustrate trends. Data from the reference area are averaged over the years and divided into low and high marsh zones.

In the reference marsh downstream of the culverts, low marsh was dominated by tall *S. alterniflora* and restricted to the steep creek banks and point bars. Low marsh samples included *S. patens* and *Scirpus* spp. at the upper edges of the banks (Fig. 8). As for Drakes Island, most of the Mill Brook reference area was considered high marsh, being dominated by *S. patens*, and including *J. gerardii* and *Scirpus* spp. (Fig. 8), as well as *T. maritimum*, *Potentilla anserina*, *L. nashii*, and a large stand of *Carex paleacea*.

Nekton

There were no significant differences in fish density between restored and reference sites at either marsh (Fig. 9). Fish biovolume, a correlate of biomass, showed a pattern very similar to fish density, with no significant differences between restoration and reference sites at each marsh. Our data show that the tidal restoration has allowed fish typical of salt marshes to enter and begin to use these marshes. *Fundulus* size (biovolume) was considerably greater at Mill Brook than Drakes Island (5.12 *vs.* 2.25 ml per fish for restored sites and 5.20 *vs.* 1.68 for reference sites).

Fish assemblages at all sites were dominated by the resident salt marsh species *Fundulus heteroclitus*. Other species present in both marsh systems include *Menidia menidia*, *Gasterosteus aculeatus*, *Gasterosteus wheatlandi*, *Apeltes quadracus* and the caridean shrimp *Crangon septemspinosa*. In addition, *Anguilla rostrata* and a single freshwater *Lepomis* sp. (sampled upstream one month after restoration) were present at Mill Brook Marsh. The maximum number of fish species in a single sample was four, which were found in one of the four samples at both the restored and reference areas at Drakes Island, and in two of the eight samples at the Mill Brook restored marsh. There were no significant differences in species number when restoration and reference sites were compared within either marsh (P>0.05 by Wilcoxon

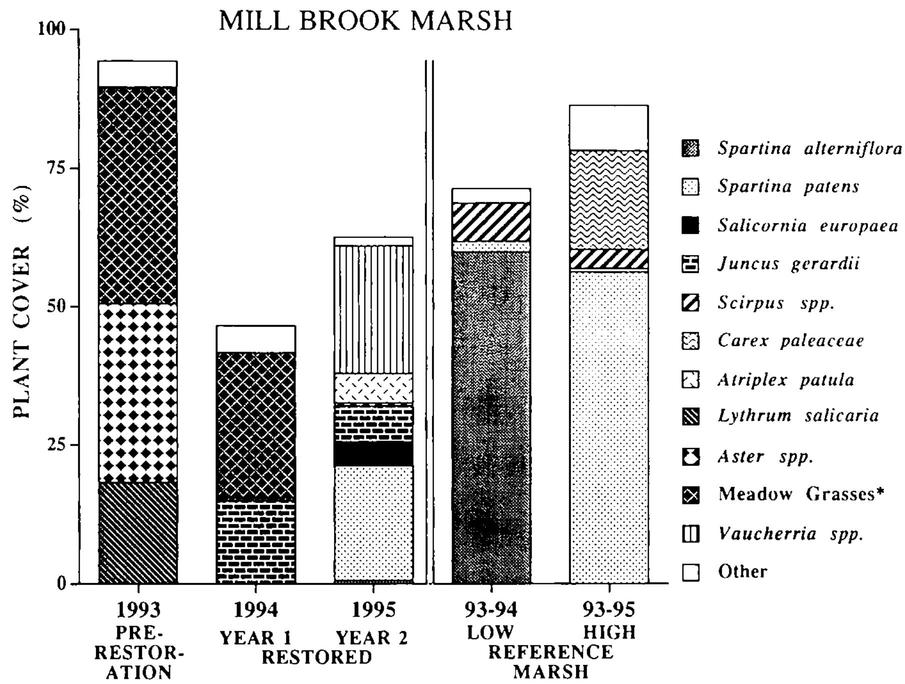

Fig. 8. Percentage cover of vegetation at Mill Brook Marsh from 1993 to 1995. Data were collected in late summer in 1993 and 1995, but in June in 1994. Annual data are means of direct estimates from 4 to 6 pairs of 1.0 m^2 samples. Yearly data from the restored area are shown to illustrate trends. Data from the reference area are averaged over the years and divided into low and high marsh zones. Meadow grasses* include, in order of importance: *Agropyron repens*, *Agrostis stolonifera*, *Festuca* spp., *Typha angustifolia* and *Phalaris arundinacea*.

rank sum), but the mean species number was slightly greater at each of the restored sites (2.75 *vs.* 2.50 at Drakes Island and 2.13 *vs.* 1.50 for Mill Brook).

Discussion

Impacts from tidal restriction

The ecological impacts from tidal restrictions in salt marshes of northern New England are similar in scope and degree (qualitative and quantitative) to impacts found by others along the United States and Europe coasts (Beeftink 1979, Roman *et al.* 1984, Frenkel and Morlan 1991, Roman *et al.* 1995). Assuming the two marshes we studied were similar on both sides of each road before impoundment, the surface elevations of the high marshes not only failed to equal the surface accretion necessary to offset sea level rise, but actually fell (Fig. 3; Table 1). The marsh surface likely subsided because the sediment supply from downstream was cut off while lowered water tables accelerated the oxidation of organic matter (Roman *et al.* 1984, Rozsa 1988, Sinicrope *et al.* 1990, Frenkel and Morlan 1991). As indirect evidence, we found that the water table was lower at Mill Brook before restoration (Fig. 5) and organic matter was 20% lower at restricted *vs.* reference areas (Burdick and Dionne 1994). However, we have no comparable data for the Drakes Island Marsh site.

It is clear that salinity was lowered by restrictions in both systems (Fig. 6). These lowered water tables likely led to the major vegetation changes that produced fresh to brackish wet meadow communities (Figs. 7 and 8). Exclusion of salt water from these marshes was not so complete that upland glycophyte communities became established as found in other systems (Beeftink 1979, Frenkel and Morlan 1991). Storm surges produced

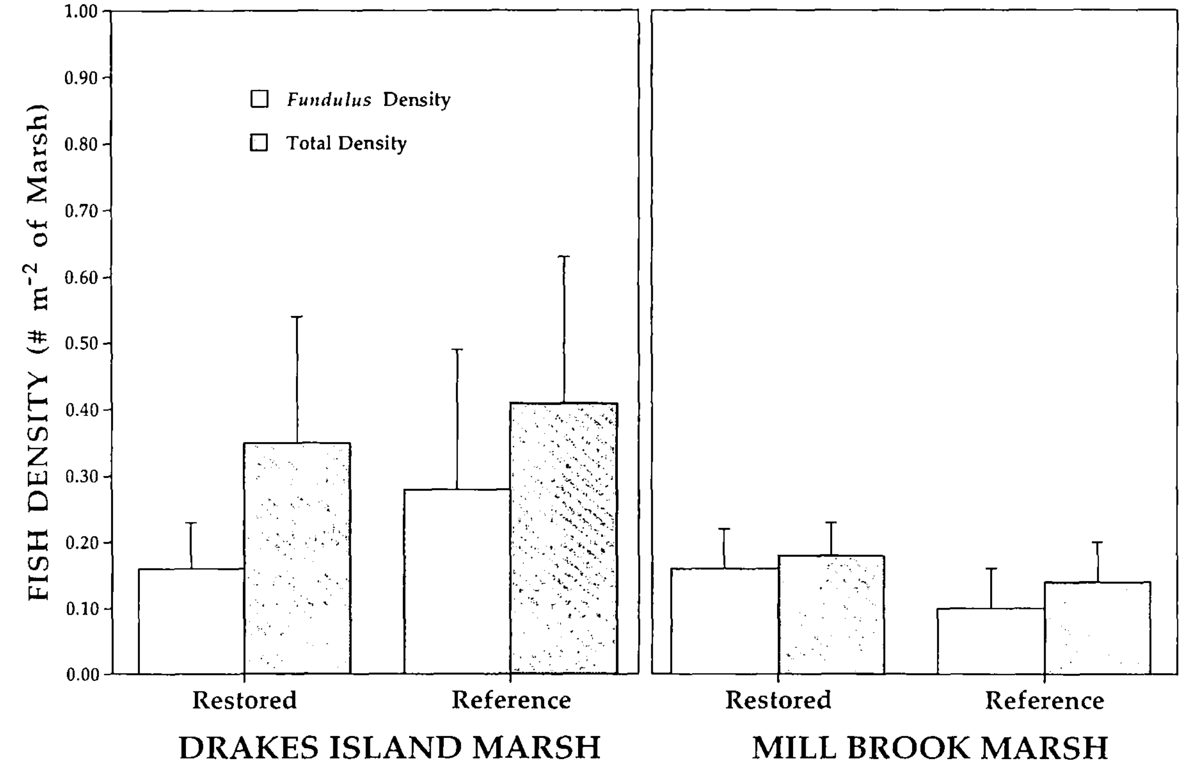

Fig. 9. Number of fish and shrimp captured in fyke nets were standardized to the area of marsh fished and presented as means with standard error bars. No significant differences were found between restored and reference sites within each marsh when means were compared using a nonparametric test (Wilcoxon rank sum). Sample size was four at both sites at Drakes Island Marsh and seven at both sites at Mill Brook Marsh.

tides that overtopped causeways in both systems about once per decade. Thus both marshes had vegetation dominated by mildly salt tolerant grasses and forbs (*Spartina pectinata* and *Typha latifolia* in Drakes Island Marsh; *Agropyron repens* and *Lythrum salicaria* in Mill Brook Marsh). Beeftink (1979) reviewed the impacts to vegetation from tidal modifications and other disturbances in the Netherlands, and found that loss of *Spartina townsendii* (a low marsh variety) was a good indicator of decreasing tidal influence while loss of *Triglochin maritimum* was not. Similarly, we found *Triglochin* performing well even though no *Spartina alterniflora* could be found before restoration at Drakes Island Marsh (Fig. 7). Tidal restrictions at both sites prevented estuarine fish from utilizing upstream areas before restoration.

Response to tidal restoration

Surface elevation relative to water levels

Tidal flooding of the marsh increased dramatically following restoration at Mill Brook and appeared to exhibit the full tidal range found here before restriction. At Drakes Island, the restoration was unplanned, partial, and without an initial hydrologic study. Here the culvert was too small to conduct the potential tidal prism (volume of water needed to realize the potential tidal range) upstream of the culvert. This resulted in inadequate flooding and, especially, draining of the restored marsh. Other tidal restoration projects lacking hydrologic or ecologic analyses and that have required further hydrologic modification are not uncommon (Rozsa 1988, Sinicrope *et al.* 1990).

The surface elevation of both marshes had fallen,

Table 1. Response of indicators of selected salt marsh functions to tidal restriction and restoration.

Function	Indicator	Evidence of Restriction	Evidence of Restoration
1) Support of salt marsh vegetation	Surface elevation relative to tides	No tides Depressed surface elevation (Fig. 3)	Tides present (Fig. 4) ** no data yet **
	Water table and salinity	Water levels and salinity low (Figs. 5 and 6)	Water levels and salinity increased (Figs. 5 and 6)
2) Provide salt marsh habitat	Plant cover by species	Fresh to brackish species invade (Figs. 7 and 8)	Fresh plants die, salt marsh species increase (Figs. 7 and 8)
3) Support of consumers	Fish use	Estuarine fish excluded from marsh	Estuarine fish using marsh (Fig. 9)

but we have not determined whether the restored areas experience rapid accretion rates (Table 1). Marsh sedimentation and accretion results from the interaction of an inorganic sediment supply brought by flood waters with the vegetation that retains it while producing organic matter above and below ground (Stumpf 1983, Stevenson *et al.* 1986, Reed and Cahoon 1992). Most sediment is captured in the low marsh (Stumpf 1983). Frenkel and Morlan (1991) found that lower elevations accrete more rapidly following tidal restoration, though they predict that 50 years is required to approach reference elevations.

Soil water table and salinity

We found water table levels increased following tidal restoration at Mill Brook Marsh. This apparently occurs at some restorations, but not at others (Rozsa 1988, Sinicrope *et al.* 1990) and depends upon hydrologic changes following restoration. More universal and better documented is the increase in salinity that is found in restored marshes (Beeftink 1979, Sinicrope *et al.* 1990). Sediment salinities at Mill Brook and Drakes Island both increased about 20 ppt, and increases were associated with dramatic vegetation changes. We found slightly greater salinities in restored versus reference marshes. At restoration sites, hypersalinity may be occurring on the surface of sediments due to increased evaporation from exposed peat and bare mudflats (Bertness 1991). Cover by vascular plants fell to less than 40% by year two at Mill Brook Marsh. The early response to restoration at Drakes Island Marsh is not known, but cover by vascular plants was only 70% seven years later. While severe hypersalinity is a problem in marshes of the southern Pacific coast (Zedler *et al.* 1982), elevated surface salinities likely hastened the retreat of salt intolerant and mildly salt tolerant species in our marshes. Thus, restoration at both sites established water level and salinity regimes in intertidal areas that support only salt marsh plants (Table 1).

Vegetation

Most existing vegetation in intertidal areas was killed following the reintroduction of tidal cycles at Mill Brook, but the process required longer than the several months we anticipated (Fig. 1). Major declines of mildly salt-tolerant species coincided with the colonization by important salt marsh species that required two years. This is similar to vegetation responses in other marshes where tides have increased (Beeftink 1979, Frenkel and Morlan 1991).

Further declines in species that had invaded during impoundment appeared to continue at the upper edges of tidal influence in both systems for several years. Whereas rapid change continues at Mill Brook Marsh through 1996 (year 3), dramatic changes at Drakes Island Marsh are unlikely (Fig. 7). The Drakes Island data from 1995 suggest that the cover of some of the opportunistic plants (*Salicornia*, *Vaucherria*) may be declining as *Spartina alterniflora* increases slowly. After 7 to 10 years of monitoring following restoration, some investigators have found that substantial changes in plant communities are continuing (Sinicrope *et al.* 1990, Simenstad and Thom 1996), whereas others have not (Beeftink 1979, Frenkel and Morlan 1991). It must be recognized that some change is likely to occur over time, even in reference marshes (Zedler *et al.* 1982, Clark 1986, Warren and Niering 1993). Although salt marsh vegetation has been re-established at both sites (Table 1), the dominance of *S. alterniflora* at Drakes Island Marsh (Fig. 7) suggests that the high marsh habitat found before restriction will unlikely be attained under the present hydrologic regime. We believe flooding stress associated with the fortnightly tides restricts the establishment of high marsh plants, but allows *S. alterniflora* to dominate this marsh (Bertness and Ellison 1987, Burdick *et al.* 1989). In a review of the elevational distribution of *S. alterniflora*, McKee and Patrick (1988) found that where the tidal range was small (< 0.5 m), smooth cordgrass did not grow below the half tide level, as seen at Drakes Island Marsh (Fig. 3).

Two years after restoration of Mill Brook, no zonation of the emerging plant communities was discerned. However by the third year (1996), we observed most stands of *S. alterniflora* grew bordering or near tidal creeks. Early indications of vegetation change (comparing 1994 and 1995 to high and low reference marshes) suggest the Mill Brook Marsh is developing into a high marsh with low marsh along the creek channels. In 1996 we observed that the cover of vascular plants increased dramatically and the marsh is beginning to resemble the reference area.

Nekton

Some functional values of fish use were recovered in both marshes soon after tidal restoration, independent of culvert size (Table 1). Support of secondary producers (*i.e.*, consumers) by the restored marsh has often been assumed or ignored by those focusing on a narrow project goal (re-establishment of salt marsh vegetation), but recent monitoring has included animals. For example, Simenstad and Thom (1996) reported bird and fish use of a restored brackish marsh in Puget Sound were the highlights of a project that had low plant growth after seven years. In their long-term monitoring, Simenstad and Thom (1996) found that fish occupied the restored area immediately upon hydrologic reconnection. This is also what we found at the Mill Brook Marsh. They attributed a gradual increase in fish density over five years to the development of prey resources and refuges from predation. Using fyke nets, we found similar assemblages of fish in restored and reference areas one month after restoration and again after two years at Mill Brook, and after eight years at Drakes Island.

Specialized fish communities may not become well-developed in salt marshes of northern New England. We found very few species overall, and samples were dominated by the opportunistic mummichog, *Fundulus heteroclitus*. Zedler (1992) reports that the depauperate fish assemblages associated with salt marshes in southern California may be due to the harsh physical environment. In northern New England marshes, stresses associated with 2 to 3 m semi-diurnal tides may also constrain fish community development, yet allow the rapid return of the relatively simple fish assemblage to restored marshes. The similarity between restored and reference sites may simply reflect their close proximity, and the ease of fish movement between restored and reference areas.

At Mill Brook Marsh, the watershed could potentially support runs of migratory fish (*Alosa sapidissima*, *Osmerus mordax*, *Anguilla rostrata*). We observed a run of *A. sapidissima* through the Mill Brook Marsh during the spring of 1994. The time scale of establishment of migratory species as part of the estuarine fish community (Fig. 1) is difficult to predict because establishment depends upon the immigration of individuals from other sites into the restored system. To further determine fish community development and use of these sites, sampling should be timed to coincide with runs at nearby sites and fish foraging success and growth should be measured (Shreffler *et al.* 1992, Simenstad and Thom 1996).

Conclusions

The patterns of marsh degradation we and others have observed in marshes indicates that tidal restrictions negatively impact salt marsh ecosystems. Flap gates on culverts lead to reduced flooding frequency on the marsh, reduced salt and sediment exchange, and retention of fresh water from rain and spring melt events. These conditions eliminate estuarine fish and favor brackish and freshwater plants over salt marsh species. Furthermore, reduced sediment input and the low water table associated with reduced tidal exchange result in decreased marsh elevation.

A small suite of indicators of critical marsh functions has been reported to examine the results of two restoration projects. Currently, benefits from the restorations have been documented for both marshes (Table 1), even though neither site has vegetation that is equivalent to the habitat values of the reference areas. Our analysis of these indicators suggests that the restoration at Mill Brook will support full habitat function soon (perhaps by its fourth year), while the tidal regime at Drakes Island will not. Our analysis of the patterns in tidal flooding and resultant vegetation indicates that the unplanned partial restoration at Drakes Island Marsh still does not provide the tidal regime needed to establish an equivalent cover of salt marsh vegetation, even though salinity is high and fish use is similar to that of the reference area. Further increases in plant cover and productivity will likely require hydrologic modification. A structure that allows a 50 cm semi-diurnal tide (through better drainage to reduce the fortnightly signal) would lead to significantly improved vegetation.

Once restored, salt marshes should be self-perpetuating and require minimal management. To understand such long-term responses to restoration, the changes in surface elevation and organic matter should be included as part of a suite of indicators. Furthermore, indicators of other critical functions such as filtration and export of reduced carbon should be included to determine whether these functions are restored. Additional measures of habitat for invertebrates, use by birds, and production of fish would be useful to estimate the benefits of tidal restoration.

Note for Managers

The partial restoration at Drakes Island Marsh was clearly a rare case of low cost re-establishment of salt marsh vegetation, although it was not a complete success. The Mill Brook restoration was inexpensive in terms of the direct funding for hydrologic analysis and construction ($20,000). However, as a cooperative effort among federal, state and local agencies (U.S. Fish and Wildlife Service, the Natural Resource Conservation Service, the Coastal Program of the Office of State Planning, and the Rockingham County Conservation District), additional costs included many hours spent by public servants assessing the site and planning for this restoration project. Approximately $20,000 is needed per year (1996 dollars), for example, to assess the impacts and responses to tidal restoration at Mill Brook.

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