2001 Annual Report Of The Hatches Harbor Salt Marsh Restoration Project

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January 4, 2002

Introduction

Hatches Harbor is a 200-acre salt marsh that was bisected in 1930 by a dike built to reduce population levels of nuisance mosquitoes. To allow drainage of fresh water, a small 2-foot wide circular culvert was constructed. In the decades after the dike construction, reduced tidal exchange has caused hydrological and geochemical changes severely impacting the natural salt marsh upstream of the dike.

From 1930 to 1987, tidal exchange of saline waters was essentially eliminated in the restricted marsh. This kept the restricted marsh peats continuously dewatered for months during the growing season. In 1987, the flap valve in the old culvert was removed during repair work following storm damage to the dike. Although tidal exchange was appreciably increased, the reduced tidal range and altered hydroperiod in the restricted marsh impounded water on the marsh surface during spring tides. As as result, pore water salinity dropped over time. These conditions led to a shift in the restricted marsh plant communities from natural salt marsh, characterized by *Spartina alterniflora* and *S. patens*, to degraded salt marsh habitat, characterized a variety of wetland plant species. The most important of these is *Phragmites australis*, which formed dense stands in the restricted marsh. Test peat cores indicate that *Phragmites* may have overtopped the existing *Spartina patens* habitat. Eventually the degraded salt marsh habitat extended to 100 acres as reduced pore water salinity favored *P. australis*.

The National Park Service, in cooperation with the Provincetown Airport Commission, Provincetown Municipal Airport and the Federal Aviation Administration, initiated a restoration of the degraded habitat. This restoration was initiated by replacing the single 2-foot diameter circular culvert with four 7-foot wide rectangular box culverts with adjustable heights (see Figure 1). Since April 1999, these culverts have been progressively opened. It is expected that the resultant increase in tidal exchange in the restricted marsh will increase tidal ranges, increase the inundation (during high tide) and dewatering (during low tide) of the marsh surface. Eventually increased tidal range should increase porewater salinity and favor the replacement of the salt-intolerant vegetation now present with salt marsh grasses, while improving surface water quality.

Figure 1. New culverts at Hatches Harbor dike

Environmental Monitoring during the Restoration Project

A comprehensive environmental monitoring program was begun in 1997, prior to construction of the new culverts, and continues to the present. There are three main goals of environmental monitoring: 1) to document pre-restoration environmental conditions in the unrestricted and restricted marsh basins. 2) to document the pace of restoration by comparing tide heights, water quality, estuarine fauna and vegetation above and below the Hatches Harbor Dike 3) to monitor water levels in the vicinity of the airport to ensure that tidal increases do not impede airport operations.

Our approach was to devise monitoring parameters that characterize environmental and ecological responses to increasing tidal exchange. Our general hypothesis was that increasing tidal exchange would bring hydrological and physical changes to the restricted marsh making these parameters more closely resemble those of the adjacent unrestricted marsh basin. These changes would be followed by changes in the geochemical parameters and finally by ecological changes in plant and animal communities bringing the habitats in the restricted and unrestricted basins closer in character to each other. Interpretation of monitoring data will involve and pre- and post-restoration comparisons as well as between the two basins for selected parameters. The monitoring would thus be of the form BACI (before-after-control-impact) for the bases of analysis and interpretation.

In this annual report, we summarize major findings from monitoring programs in 2000 and 2001. Table 1 summarizes the monitoring parameters to be characterized.

Table 1. Monitoring parameters discussed in this report

Factor	Parameter	Method
Hydrological	Water level and tidal range changes	Data logger units
	Porewater levels	Monitoring well
Geochemical	Salinity and sulfide	Monitoring wells
Ecological	Vegetative cover	Permanent vegetation plots
	Vegetative biomass	Biomass sampling plots
	Adult mosquito production	Light/CO_2 traps
	Fish population surveys	Throw traps

These reported findings demonstrate that the culvert openings in 2000 and 2001 have led to significant environmental changes in the degraded salt marsh habitat in the restricted marsh. Significant shifts in mosquito population structure and vegetation productivity were observed from 1997 to 2001. Concurrent changes in species composition of the vegetation and fish population in the restricted marsh have not as yet fully occurred.

Tidal Range And Tidal Height Response

C. N. Farris, November 2001

As has been discussed in previous annual reports, increasing the tidal exchange in tidally restricted salt marshes leads to a range of environmental responses. These include increases in marsh surface inundation and a greater penetration of saline water into the marsh peats (Harvey et al 1997) with a resultant alteration of porewater chemistry (Portnoy and Valiela 1997; Caetano et al 1997). Increasing tidal range also facilitates greater peat dewatering during ebb tides (Montague et al 1987; Harvey et al 1997). These factors over time favor the growth of *S. alterniflora* and *patens* over *Phragmites australis*, especially changes in porewater chemistry and salinity (Steever et al 1976; Odum et al 1995). Increasing tidal exchange in restricted salt marsh should also improve water quality in the restricted marsh and alter the species composition and abundance of mosquitoes.

Tidal exchange in tidally restricted marshes is driven by differences in hydraulic head between the basins, the volume of the restricted marsh and the size, shape and elevation of the culverts (e.g. Roman et al 1995). Of equal importance is the elevation of the bottom of the culvert, as this determines how much of the tidal prism remains at this level during an ebbing tide. It is important to maximize the amount of the culvert cross-sectional area that is nearest this critical elevation. This would maximize the extent of dewatering during ebb tides. Such dewatering helps encourage re-establishment of native cord grass stands.

The interaction between increased tidal inundation, due to a larger culvert, and the culvert shape will affect tidal changes in observed water levels in the restricted marsh. Discrete events (storms, breaches in the spit at the outer mouth of the harbor, etc.) can also cause transient changes in water level. These episodic events can be important if they occur during ecologically critical periods, for example, the peak of the growing season.

Methods

Four multi-parameter data loggers (YSI UPG6000, Yellow Springs, OH) were deployed in both the restricted and unrestricted marsh on either side of the culvert approximately 0.7 – 0.9 m (2 – 2.5 ft) above the bottom (see Figure 2). Two YSIs were located in the restricted marsh, one adjacent to the culvert and one near the edge of the marsh bank in the restricted marsh approximately 300 m from the culvert. The third YSI was located adjacent to the culvert in the unrestricted marsh. Data loggers measured water levels with on-board pressure transducers that compensated for salinity and temperature. Tide stage was continuously recorded ten times an hour for two-week periods. Sampling commenced in October 1998 and continues to the present day. On May 6 2001, the YSI at the culvert station in the restricted marsh had to be removed because migrating sand was burying its mount. Tidal range was calculated by subtracting average low tide levels from high tide levels. Tidal range comparisons were made among spring tides.

Results

Over the last three years, the annual openings of the culverts have progressed according to the Operation Plan (see Appendix 1). Last year, the culverts were opened two notches to mitigate silting in of the culverts. This silting was promptly reversed. Table 2 shows the culvert configurations for these years.

Table 1. Culvert configurations

Year	**Number of open culverts**	**Height of opening**	**Total cross-sectional area of outlet**	**Resultant mean spring high tide measured in restricted marsh**
1998	1	Old culvert	0.29 m^2 (3.12 ft^2)	1.46 m-MSL
1999	2	0.1 m (0.32 ft)	0.43 m^2 (4.58 ft^2)	1.48 m-MSL
2000	4	0.1 m (0.32 ft)	0.85 m^2 (9.1 ft^2)	1.55 m-MSL
2001	4	0.4 m (1.32 ft)	3.41 m^2 (36.7 ft^2)	1.70 m-MSL

Note that the measured high tides did not change from 1998 to 1999, but showed changes for the three subsequent years (significant $p<0.001$). As reported in previous annual reports, the change in tidal range was probably due to lower low tides. The new culvert configuration greatly enhanced drainage of the restricted basin as most of the culvert opening was at or near the invert elevation of the original culvert.

These culvert openings resulted in the tidal range more than doubling since the inception of the restoration project, increasing by approximately 30% between 2000 and 2001 at the two locations of the restricted marsh where monitoring was performed (see Figure 3). Examining the magnitudes of the two daily high tides in consecutive years reveals an example of the convergence in hydroperiod characteristics between the two basins (see Figure 4). In the unrestricted basin there is an asymmetry between the two daily high tides because of the interaction of tidal components that were filtered out in restricted marsh by the old culvert as of 1998. Over the next three years, the records show a gradual increase in the magnitude of one of the daily high tides as tidal exchange is increased. This increase in consecutive high tides may presage significant changes in the hydroperiod (timing, extent and duration of marsh surface inundation and dewatering) with increasing tidal exchange. It is expected that geochemical and ecological factors susceptible to changes in hydroperiod will follow these trends.

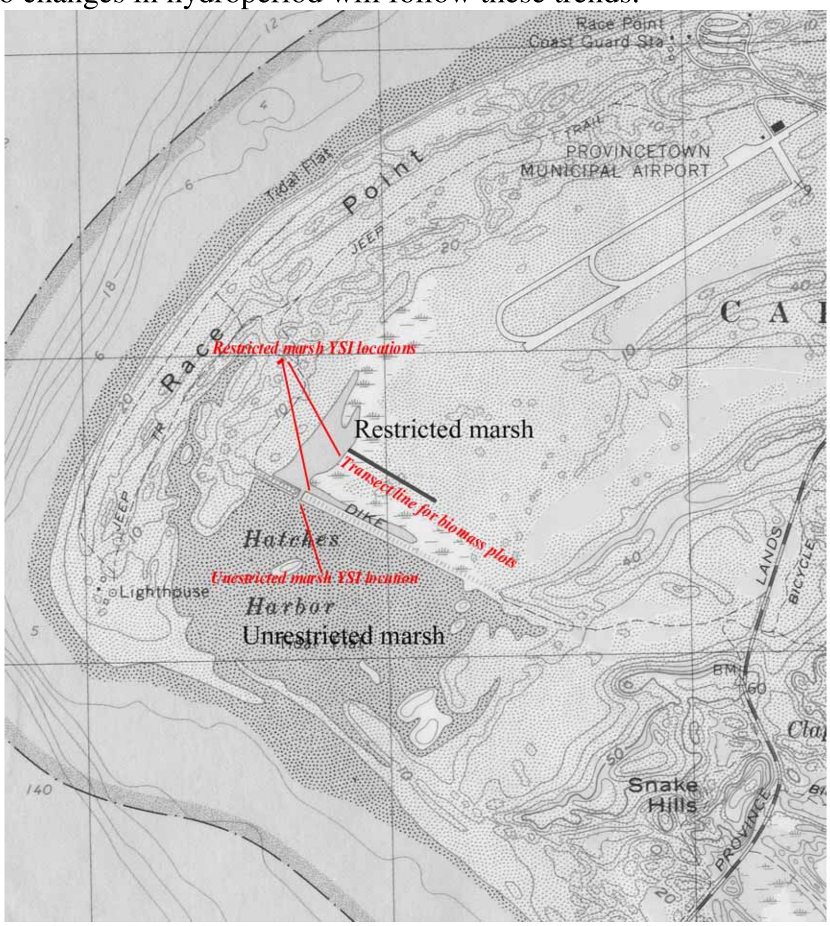
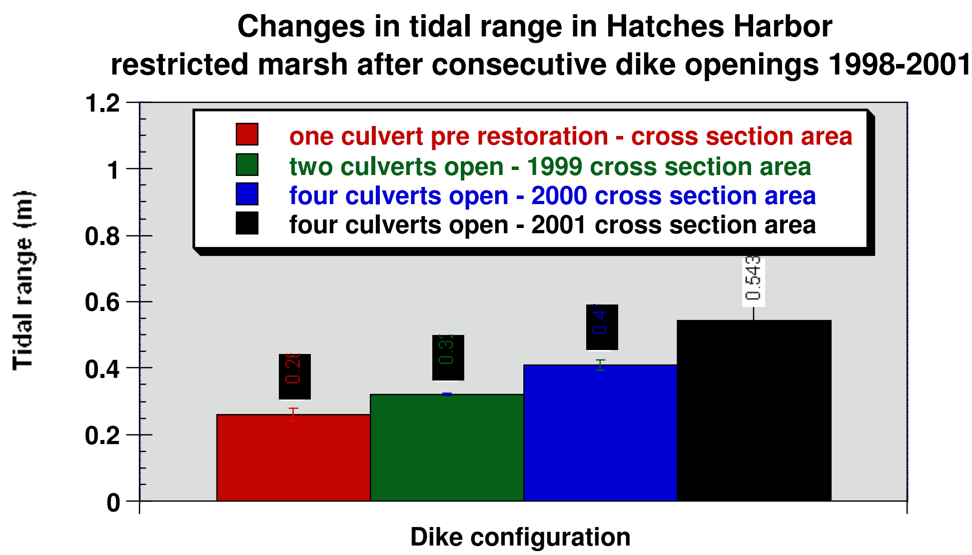


Figure 2. Locator map for YSI and vegetation transect sampling

Figure 3. Tidal range at two sites in the restricted marsh



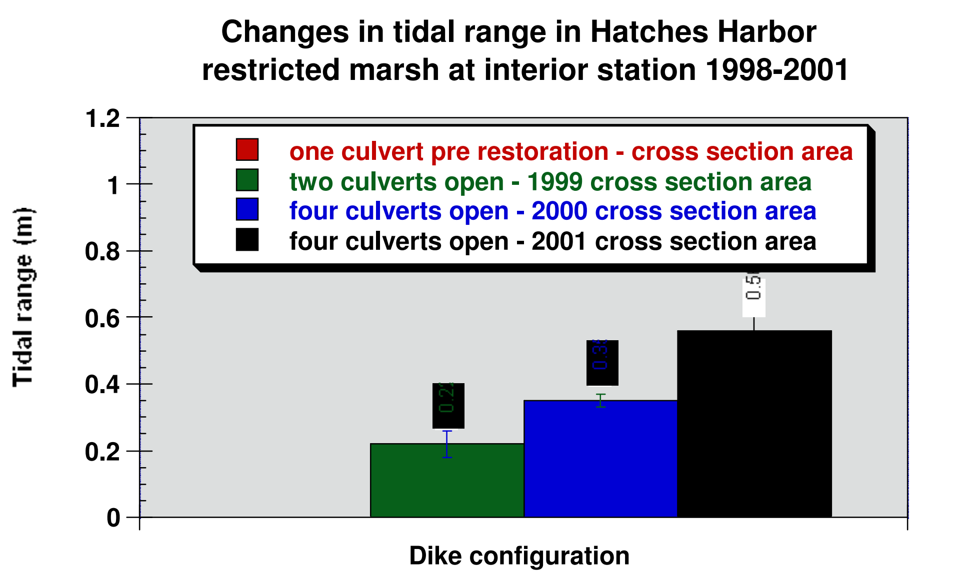
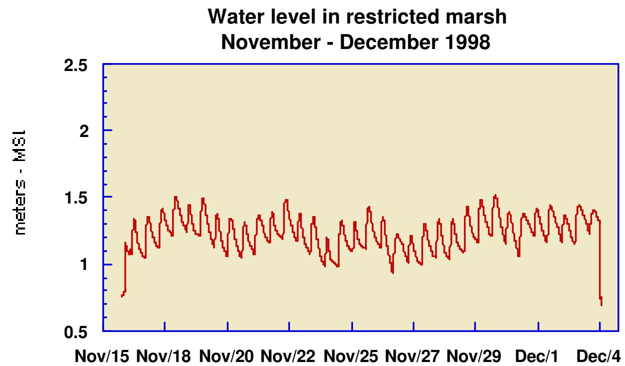
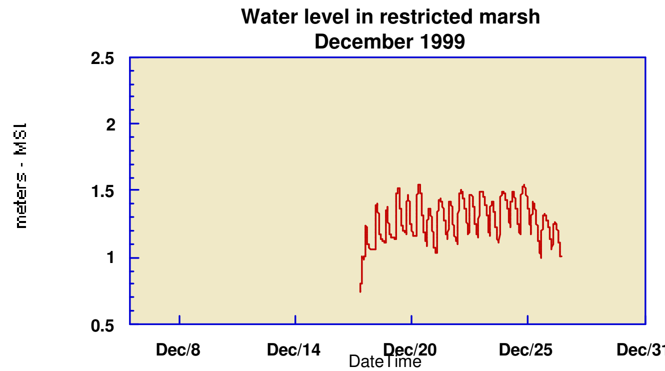
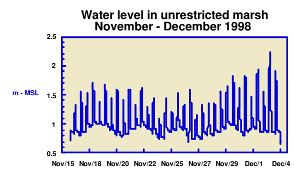
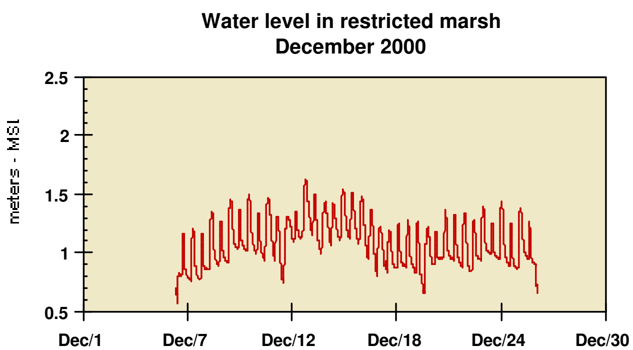


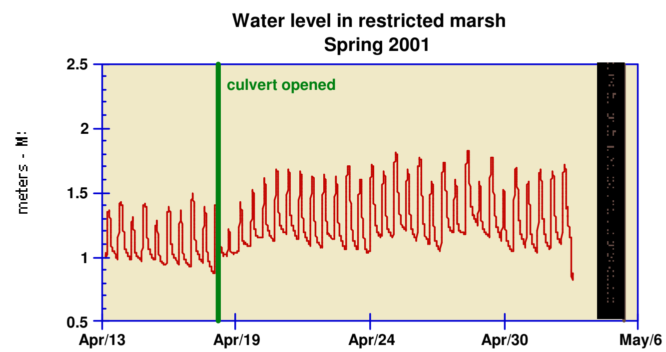
Figure 4. Water level records for four years in restricted marsh (comparison to 1998 record in unrestricted marsh











Summary

1. Cross-sectional area has been increased over ten-fold from 1997 to 2001.
2. Tidal ranges have increased two to three-fold in the restricted marsh.
3. Hydroperiod characteristics in the unrestricted marsh and the tidally restoring marsh seem to be converging presaging tidal inundation patterns in the restricted marsh that more closely resemble the unrestricted marsh.

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Monitoring Flooding Depth, Porewater Salinity and Sulfides Over A Spring Neap Tidal Cycle

J.W. Portnoy

The species composition and growth form of coastal wetland vegetation is determined by site-specific salinity and flooding regimes. Elevation relative to the local tidal range is a basic determinant of both the salinity and the flooding depth of salt marsh soils. Proximate causes of plant stress with prolonged seawater flooding include osmotic imbalance and sulfide toxicity, inhibiting nitrogen uptake. High salinity and/or sulfide is stressful for all wetland plants, causing mortality in fresh-brackish species and stunted growth in typical salt marsh grasses (e.g. *Spartina alterniflora*).

The wetland surface immediately above the Hatches Harbor dike is about 15 cm below the elevation occupied by intertidal *S. alterniflora* in the unrestricted marsh seaward of the structure. This is an expected result of 1) reduced tidal range, allowing peat dewatering, pore-space collapse, and aeration and increased decomposition (Portnoy & Giblin 1997) and 2) decreased tidal transport of inorganic particles which in an unrestricted marsh accumulate on the wetland surface (Thom 1992). The relatively low elevation of the marsh surface relative to modern sea level indicated that flooding depths and durations might exceed those of the unrestricted marsh once tidal range is restored. Thus there was concern that excessive flooding and consequently high salinity and/or sulfide could hinder the re-establishment of salt marsh vegetative cover in the Hatches Harbor restoration site. This would occur if flooding heights increased faster during tidal restoration than the rate of sedimentation. During spring tides flooding depth and duration behind the dike already exceeded those of the unrestricted natural marsh before any tidal restoration. This was because of the small (2-ft) diameter of the dike's original culvert that impeded discharge during low tides.

To establish a basis for future assessments of the effects of increased tidal volume on wetland soil conditions, we have monitored water depth, porewater salinity and sulfides along transects both seaward (1997, 1999 and 2001) and landward (1997, 1999, 2000 and 2001) of the dike. Pre-restoration monitoring was conducted in September 1997 with the original 2-ft culvert still in place; a clapper valve in place since 1930 to prevent seawater flow into the diked marsh had been removed in 1987. With the installation of new enlarged culverts in winter 1998-1999, annual porewater monitoring has resumed along with the incremental increases in culvert openings (see Table 2)

Methods

Marsh water levels and porewater salinity and sulfide concentrations were measured during low tide (tide height seaward of the dike < 0.93 m-NGVD) along vegetation Transect 2 located landward of the dike (Fig. 2). Sampling was conducted every 2-3 days with seven observation dates in 1997, nine in 1999, and eight in 2000 and 2001. All sampling has been conducted between 23 August and 1 October to coincide with reported fall maximum sulfide concentrations in New England salt marshes (Howarth & Teal 1979, Howes et al. 1983).

Water levels were monitored in 60 cm long, 3-cm ID PVC well screens driven 50 cm below the marsh land surface, leaving 10 cm exposed. Elevations of casings were determined by total station.

Porewater for salinity and sulfide determinations was withdrawn from the sediment with a 2-mm ID stainless steel probe with slotted point. The probe was inserted 10 cm into the sediment and water was drawn into a 3-ml syringe fitted onto silicone tubing attached to the probe's upper end. Any air aspirated into the syringe was discharged prior to collecting an anoxic sample. If the peat water level was deeper than 10 cm from the surface, no sample was collected. All but 0.5 ml of sample contained in the syringe was discharged onto a refractometer to read salinity (± 1 ppt). The remaining 0.5 ml was discharged directly into a 20-ml scintillation vial containing 12 ml of 2% ZnAc to precipitate sulfides. Total sulfides were subsequently determined colorimetrically in the laboratory (Cline 1969, detection limit 10 µM).

Results and Discussion

Flooding regime

Prior to the 1999 installation and opening of the new culverts, the restricted marsh surface was always covered with 10-20 cm of water for seven days during spring tide periods (Fig. 5). The small cross-sectional area (0.29 m^2) of the old two-foot culvert impeded drainage resulting in impoundment and waterlogging in the diked marsh.

In April 1999, two of the four new culverts were opened 10 cm, increasing the cross-sectional area available for discharge; importantly, all of this cross-sectional area (0.43 m^2) is low, between 0.53 and 0.63 m-NGVD, improving discharge at low tide. As a result, at nearly all sampling stations along Transect 2, 1999 low-tide water levels were below the peat surface even during spring tides (Fig. 6). This represented a major qualitative change: peat that was constantly waterlogged for about half the spring-neap cycle before restoration was now dewatered and aerated on each low tide (Fig. 6).

With the partial opening of the second two culverts in April 2000, the open cross-section was doubled to 0.85 m^2; however, resulting low tide elevations were similar to those observed in 1999 (Fig. 6).

The large increase in culvert opening in 2001 to 3.40 m^2 caused a major increase in tidal volume and a major qualitative change in the wetland flooding regime. For the first time since at least 1987, the wetland surface is being exposed at low tide throughout the spring-neap cycle. Increased low-tide drainage and peat aeration should benefit survival and production of salt marsh grasses as they re-colonize the diked flood plain.

Salinity

Porewater salinity significantly increased (ANOVA, P<0.05) along Transect 2 in 1999, 2000 and 2001, as increasing tide heights carried higher-salinity water further into the emergent wetland (Fig. 7). Salinities ranged from about 30 ppt at the creek bank to 5 ppt 240 m from the creek in the interior marsh. Brackish water now permeates the peat over a broad area of previously freshwater wetlands extending to the airport approach lights.

Sulfides

In general, since monitoring began in 1997 total sulfides in the diked marsh have remained very low (<0.05 mM), or about two orders of magnitude below concentrations known to cause plant stress in less well-drained salt marshes (Pezeshki et al. 1988, Koch & Mendelssohn 1989). Sulfide concentrations did not change significantly along the monitored Transect 2 until this year's (2001) increase in culvert opening and tidal range, when sulfide concentrations decreased significantly below those observed in 1997, 1999 and 2000 (ANOVA, P<0.05) (Fig. 8). Although salinity has substantially increased, especially this past year (Fig. 7), so has drainage and aeration (Fig. 6), favoring aerobic decomposition and abiotic sulfide oxidation. An increase in sulfides would be expected if salinity, providing abundant sulfate, and waterlogging, promoting anaerobic carbon catabolism by sulfate reduction, both increased.

Summary

1) Recent physical changes to the Hatches Harbor Dike have significantly affected porewater levels, root-zone salinity and sulfides in the restricted wetland upstream.
2) With increased high tide heights and decreased low tide heights, the wetland root zone is now being exposed to both high salinity (30 ppt) at high tide and air (dewatering) at low tide over a broad area extending about 160 m from the creek bank. These conditions are similar to the natural (unrestricted) salt marsh seaward of the dike.
3) Improved drainage at low tide has caused sulfides to decrease significantly since 2000. The combination of high salinity, which stresses salt-sensitive vegetation, and low concentrations of toxic sulfides should favor the re-establishment of native salt marsh grasses.

Figure 5. Land surface and spring low tide porewater levels along Hatches Harbor Transect 2 in Aug-Sep 1997, 1999, 2000 and 2001.

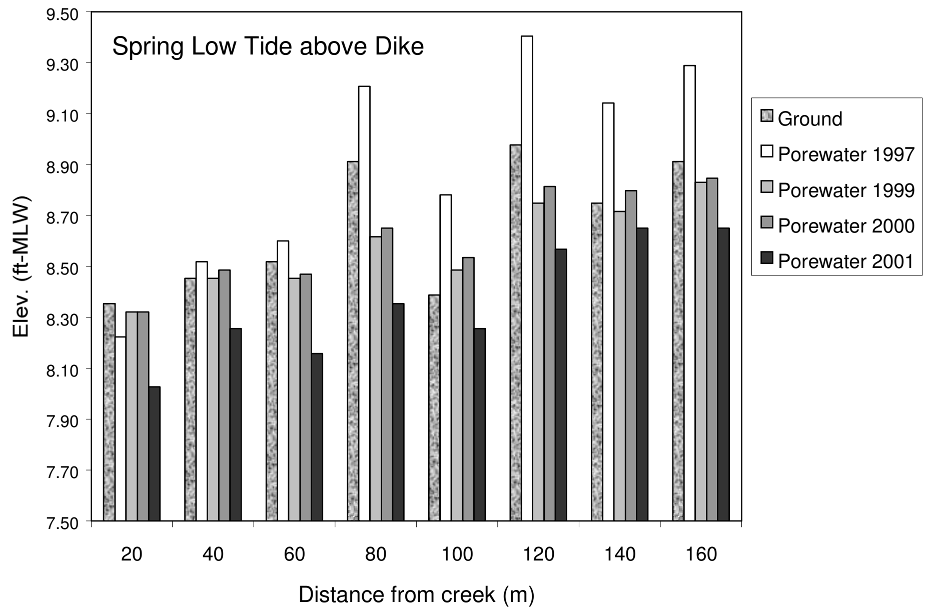


Figure 6. Low-tide porewater levels along Transect 2 through a spring-neap tidal cycle in 1997, 1999, 2000 and 2001.

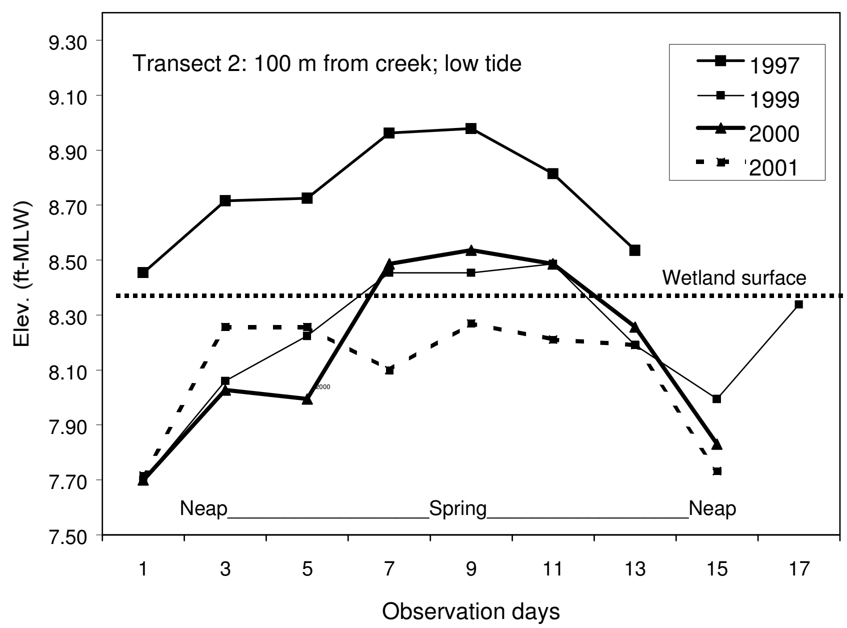


Figure 7. Mean (±SD) porewater salinity along Transect 2 over a spring-neap tidal cycle in 1997, 1999, 2000 and 2001.

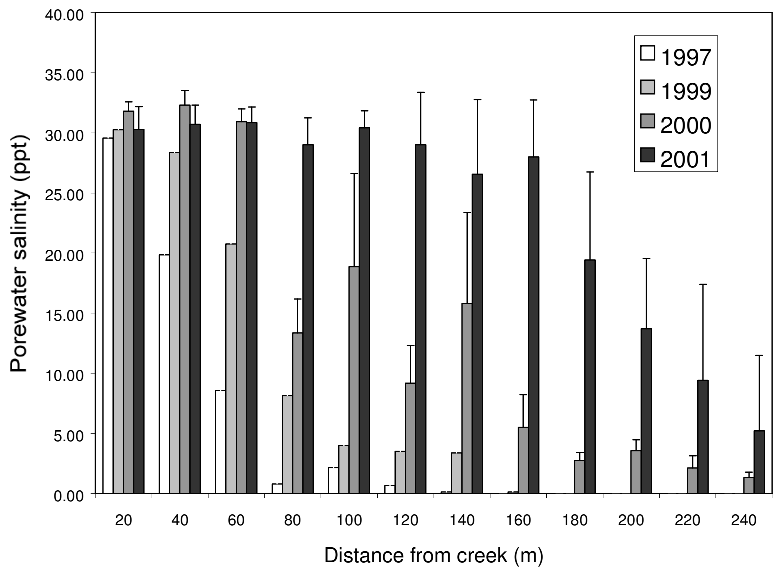
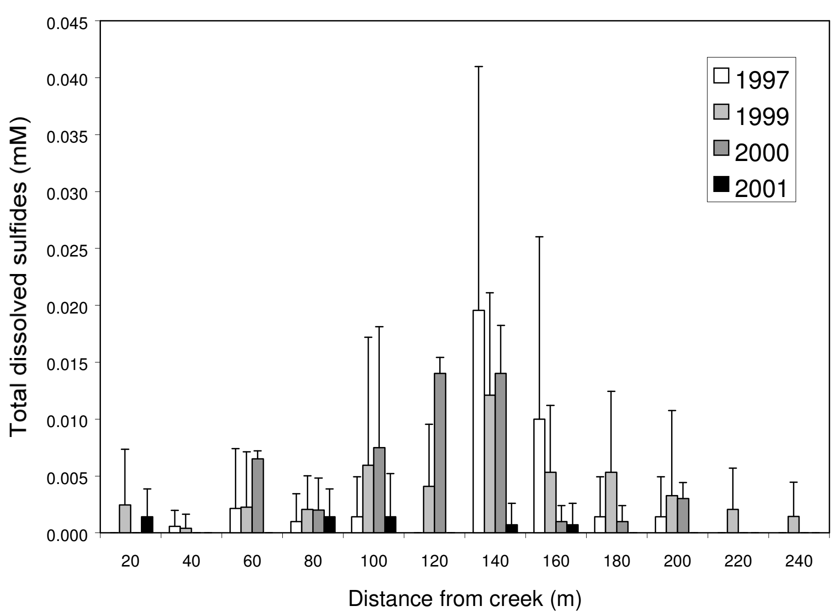


Figure 8. Mean (±SD) porewater sulfide along Transect 2 over a spring-neap tidal cycle in 1997, 1999, 2000 and 2001.



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Hatches Harbor Adult Mosquito Monitoring, Summer 2000 & 2001
J.W. Portnoy, October 2001

With increased tide heights within the flood plain above the Hatches Harbor Dike, there is concern that floodwater mosquito breeding habitat may increase, thereby increasing mosquito production and the mosquito nuisance at the Provincetown Airport. As part of an interdisciplinary program of pre- and post-restoration monitoring, nuisance mosquito monitoring was undertaken to assess any changes in abundance and/or species composition that could in turn reflect changes in breeding habitat.

The abundance and species composition of nuisance mosquitoes depend in large part on wetland flooding regimes and salinity. On outer Cape Cod, summertime floodwater *Aedes* species are comprised chiefly of *A. canadensis* and *A. cinereus* emerging from freshwater and *A. cantator* and *A. sollicitans* from brackish or saltwater habitats. *A. cantator*, the principal nuisance mosquito on the outer Cape during June, July and August, emerges primarily from tidally restricted salt marshes like Hatches Harbor (Portnoy 1984). *Coquilletidia perturbans* larvae develop only during winter in emergent freshwater wetlands; thus, episodic spring and summer flooding of coastal marshes should not increase the abundance of this species.

Adult mosquito trapping has been conducted for two summers (1997 and 1998) prior to tidal restoration and has continued for three more years (1999, 2000 and 2001) as NPS has incrementally increased tidal flow through the new, enlarged dike culverts. The objective of the monitoring is to represent seasonal abundance and species composition of nuisance mosquitoes over the entire flood plain using repeatable methods as tidal restoration proceeds. Species composition should indicate primary breeding habitats, especially with regard those habitat variables that are most sensitive to changes in tidal flow through the Hatches Harbor Dike, i.e. salinity and wetland hydroperiod. It is hypothesized that tidal restoration will increase the extent and depth of flooding of the diked wetland surface during high tides; however, improved drainage of freshwater runoff and tidal water through the enlarged culverts during the ebb will limit floodwater mosquito breeding habitat. It is further hypothesized that improved fish (especially *Fundulus* spp.) access to the wetland surface will reduce mosquito reproductive success. Increased salinity may alter the species composition of nuisance mosquitoes landward of the dike, perhaps favoring salt marsh species (*A. sollicitans* and *A. cantator*).

Importantly, adult mosquito abundance as measured by light/CO_2 trapping varies greatly both spatially (tens of meters) and temporally (days) depending upon local temperature, humidity and wind speed. Therefore, species composition, as a reflection of potential changes in breeding habitats, is emphasized over abundance in assessing trends in the trapping data.

Methods

Duplicate Bioquip #2803 EVS mosquito traps baited with light and 500 g of dry ice were set 1.5 m above the ground at three locations within the Hatches Harbor flood plain during July and August of each year (Fig. 9). Trapping stations represented natural salt marsh ("seaward"), diked marsh ("taxiway"), and airport terminal, where mosquitoes were most likely to encounter and bite people. With increases in tidal flow since 1999, these three stations now span a salinity gradient from full-strength seawater (32 ppt) at the Dike, to brackish water (5-15 ppt) near the Taxiway, to freshwater surrounding the Terminal. Traps were hung at about 1800 (dusk) and retrieved at about 0600 the following morning. Adults were taken back to the laboratory and immediately killed by freezing. With the exception of only a few badly damaged specimens, adult mosquitoes were identified to species (Darsie & Ward 1981) and enumerated.

Results and Discussion

Adult mosquito abundance is favored by high precipitation, especially that occurring during the spring and summer when warm temperatures promote egg hatching and larval development. This relationship was evident in high trap counts during 1998, 2000, and 2001 when heavy precipitation occurred in spring and early summer, and low trap counts in the dry summers of 1997 and 1999 (Fig. 10). Heavy summer rains create floodwater breeding sites on wetland surfaces, but apparently favor some species more than others. Brackish water mosquito *(Aedes cantator)* production was especially high in wet summers (1998, 2000 and 2001), and almost nil in the drought years of 1997 and 1999. In contrast, production of the salt marsh mosquito *Aedes sollicitans* appeared much less sensitive to precipitation; this is expected given its tendency to breed in high-salinity tidal waters especially after spring tides.

In 2001, *A. cantator* trap counts generally increased about four-fold over preceding years. This increase is probably attributable to increased wetland flooding which resulted from the large increase in sluice gate openings this April. As noted elsewhere in this report, mean high tides increased 0.2 m over 2000 conditions. Although larvivorous mummichogs were often abundant over the flooded wetland surface, larvae apparently survived in protected micro-habitats, e.g. stagnant pools that were not accessed by fish.

Adults of freshwater-breeding species have not increased since tidal restoration began in 1999, nor has the salt marsh breeding *Aedes sollicitans* (Fig. 10) despite large increases in salinity and seawater incursion within the diked flood plain.

Inter-site differences in species composition appear related to the proximity of appropriate breeding habitat. Salt marsh species *(A. sollicitans*) were generally more abundant in traps at the Dike, just landward of extensive natural salt marshes. With tidal restoration behind the Dike since 1999, brackish-water mosquitoes have become more dominant farther inland including around the Terminal building. Meanwhile, freshwater-breeding mosquitoes have become scarce, i.e. relative to brackish species, at all three trapping stations from the Dike to the Airport Terminal (Fig. 11).

Although mosquito densities and the mosquito nuisance appear to be largely controlled by the amount and timing of precipitation, with wet years (1998, 2000 and 2001) yielding many more biting mosquitoes than dry years, brackish water mosquito production was much higher in 2001 than in 2000 despite comparable precipitation (Fig. 10). Therefore, the recent major changes in hydrography and salinity consequent with the tidal restoration project appear to have favored brackish water mosquito production in 2001.

Persistence of this trend will depend on the persistence of pools on the wetland surface which are presently poorly flushed and/or inaccessible to predatory fish. As mentioned above, increased brackish mosquito breeding is probably centered in stagnant micro-habitats in the interior marsh well back from creek banks. These stagnant pools have developed in depressions probably caused by peat subsidence since the 1930 diking. The persistence of these depressions depends on sedimentation. With the return of tidal flow over the wetland surface, sediment transport and deposition should increase to compensate for elevational loss since diking.

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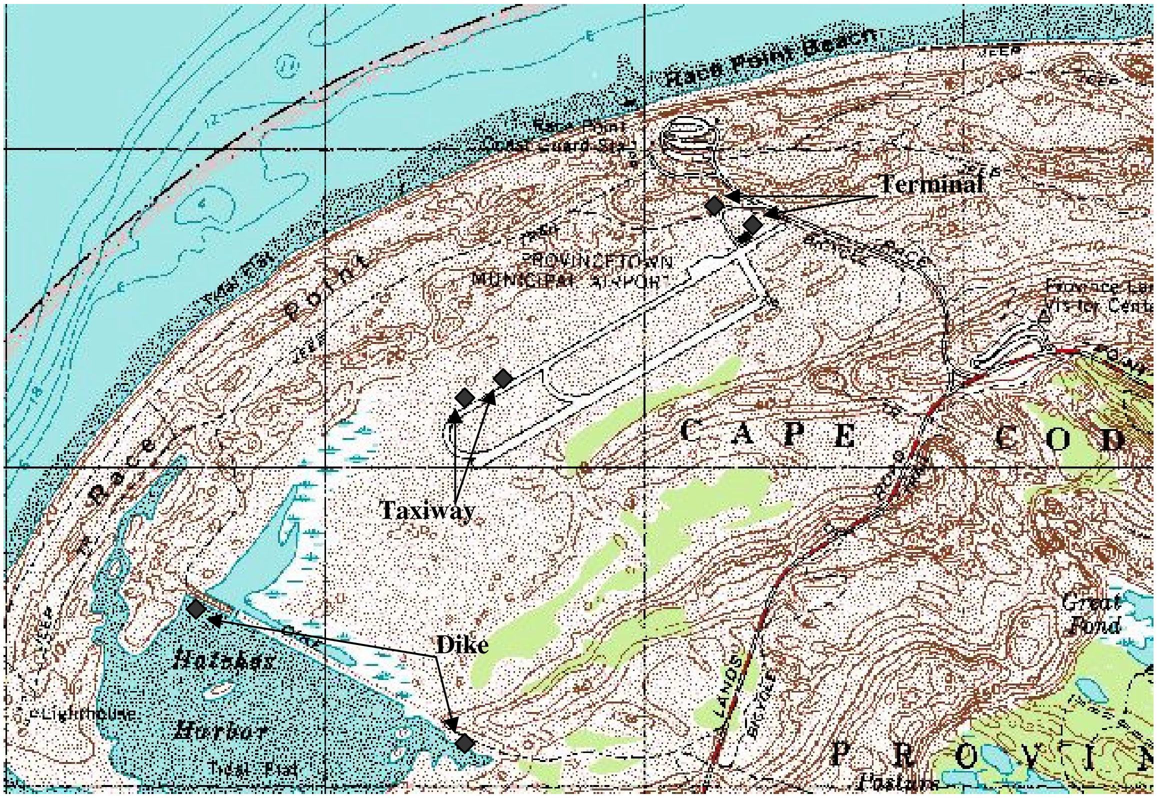

Figure 9. Hatches Harbor study area showing mosquito trapping sites at black diamonds (◆).

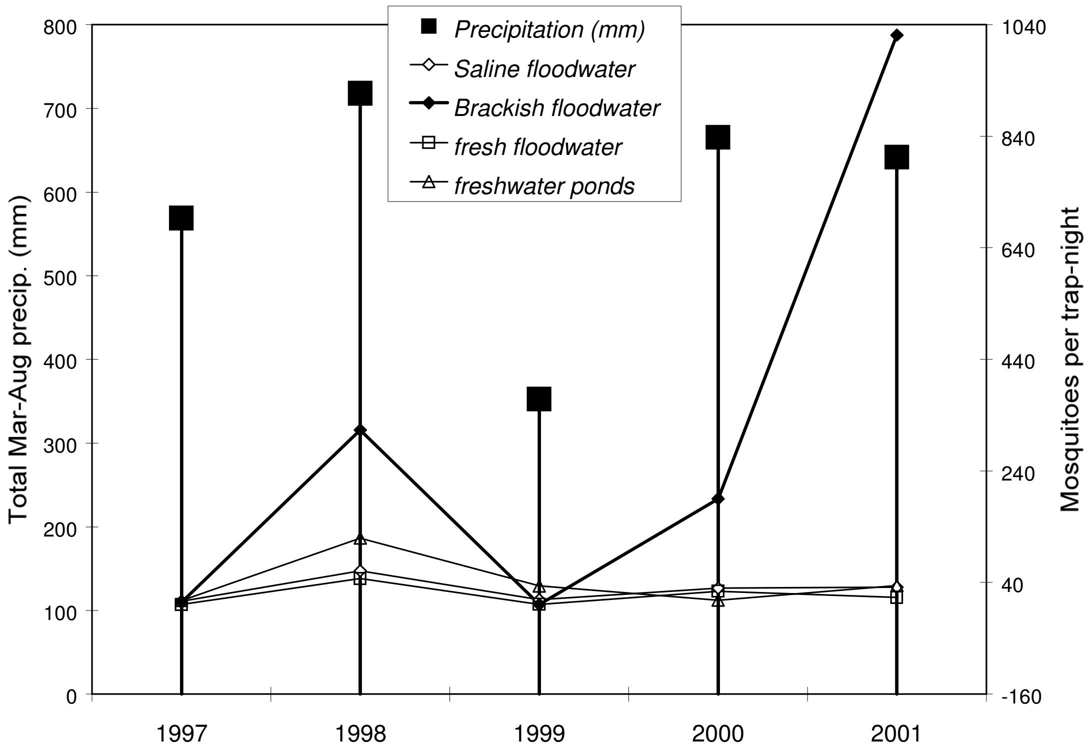

Figure 10. March through August precipitation and adult mosquito abundance by Hatches Harbor breeding habitat for 1997 through 2001. Mosquito data are means for 24 trap-nights in 1997, 1998, 2000 and 2001, and 46 trap-nights in 1999.

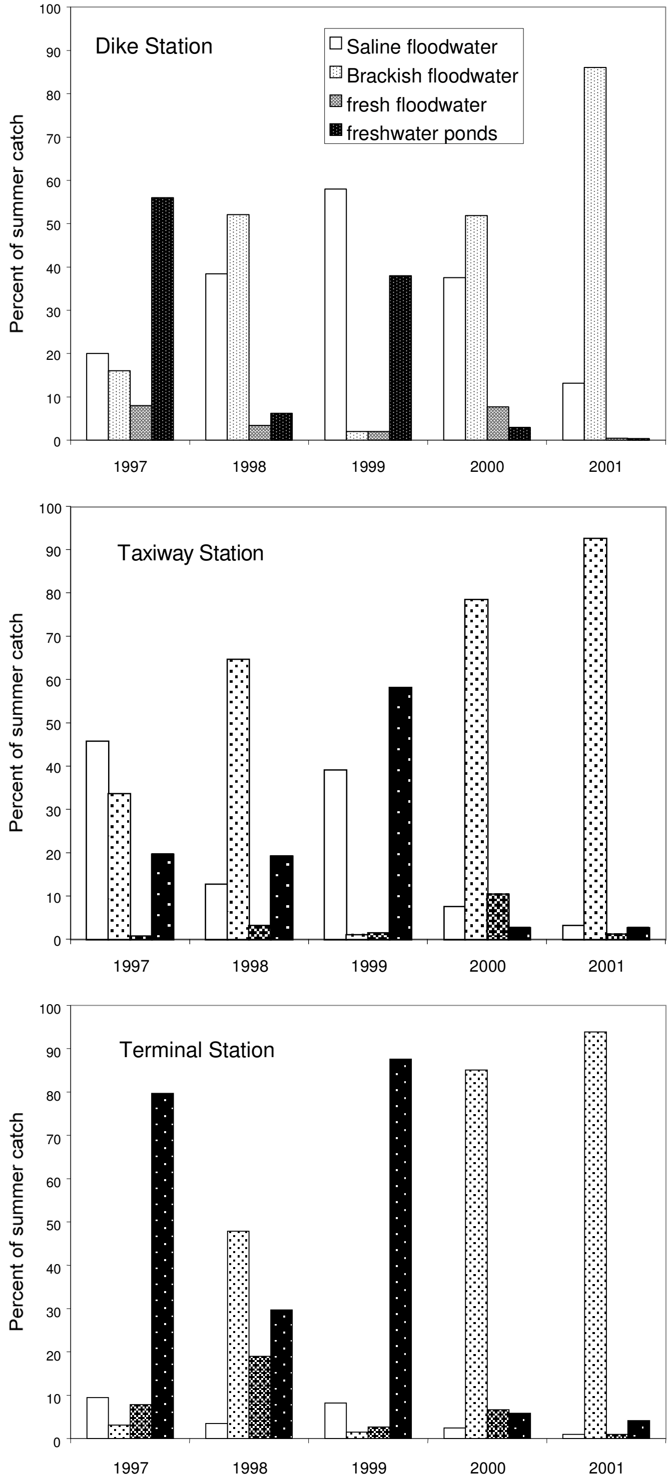

Figure 11. Species composition of adult mosquitoes captured at three sampling stations at Hatches Harbor. See map for trapping locations.

Hatches Harbor salt marsh restoration:
Preliminary summary of vegetation and nekton responses to initial tidal restoration

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April 6, 2001

Tidal restoration of the Hatches Harbor salt marsh has been ongoing since Spring 1999. Salt marsh vegetation and nekton (fish and decapod crustaceans) are just two of many variables being monitored to quantitatively evaluate the ecological response of the marsh to tidal restoration. Vegetation and nekton data were collected before tidal restoration and after the initial increase in tidal flushing through the new culverts. Monitoring of these variables will continue for the long-term.

The fundamental study design follows a BACI (before, after, control, impact) approach, with sampling <u>before</u> tidal restoration and <u>after</u> tidal restoration. The portion of marsh located downstream of the Hatches Harbor dike, referred to as the unrestricted marsh, serves as a <u>control</u>. In the case of a BACI design, re-introduction of tidal flow is the <u>impact</u>. With this study design it is possible to make several comparisons aimed at determining, with statistical certainty, the response of vegetation and nekton to tidal restoration. The following comparisons, for vegetation and nekton, are presented in this summary report:

- Unrestricted marsh v. restricted marsh before tidal restoration. This will document the effect that tidal restriction has had on the marsh system. Also, with continued monitoring (e.g., unrestricted vs. restoring marsh year 1, unrestricted vs. restoring year 2, etc.), it is possible to evaluate the trajectory or direction of vegetation or nekton change. Overtime, it is assumed that the unrestricted and restoring marshes will become more similar.
- Restricted marsh v. the restoring marsh. This documents the response to tidal restoration. Overtime, if restoration is successful the restoring marsh would be expected to differ when compared to the restricted marsh (i.e., habitat quality would improve).
- Unrestricted marsh before restoration v. unrestricted marsh after restoration. It is important to document vegetation or nekton changes of the control marsh overtime. If after tidal restoration, vegetation changed within the restoring marsh, but the control marsh vegetation did not change, then it could be suggested, with some

certainty, that the changes in the restoring marsh were due to increased tidal flow and not some other factors.

The purpose of this report is to summarize the findings, with regard to the above-mentioned comparisons. This is not intended as a comprehensive report, and thus, details on methods, data analysis techniques, and comparisons of the Hatches Harbor data with other marsh restoration sites are not provided. Comprehensive reports are presently under peer technical review or are in preparation.

Vegetation

Permanent $1m^2$ vegetation plots were established throughout the unrestricted marsh (37 plots) and restricted marsh (92 plots) and sampled during the summer of 1997 (pre-restoration conditions) and again in 2000 (during the 2^{nd} growing season after tidal restoration). Within each plot, the species composition and cover or abundance of each species were determined by the point-intercept method. To evaluate vegetation responses to tidal restriction and tidal restoration, the vegetation plot data were analyzed by Analysis of Similarity (ANOSIM), a non-parametric procedure aimed at comparing the similarity (or dissimilarity) between groups of community (i.e., species composition and abundance) data. In this case, the groups are restricted vs unrestricted, restricted vs. restoring, etc.

The unrestricted salt marsh was dominated by *Spartina alterniflora* and was typical of New England salt marshes. Just five (5) marsh plant species (*Spartina alterniflora*, *Salicornia* sp., *Limonium nashii*, *Spartina patens*, *Sueada maritima*) and 2 species of macroalgae (*Ascophyllum nodosum*, *Fucus vesiculosus*) were encountered on the unrestricted marsh. Over 70 plant species were encountered throughout the tide-restricted marsh.

Results from the ANOSIM are presented in Table 1 and summarized as follows;

- In 1997, prior to tidal restoration, vegetation of the unrestricted and restricted marshes was very different.
- In 2000, after tidal restoration, vegetation of the unrestricted marsh and the now restoring marsh remained different. It is expected that in subsequent years under a regime of restored tidal flow that vegetation of the restoring marsh will become more and more similar to the unrestricted "control" marsh.
- Vegetation of the restricted marsh in 1997 and the now restoring marsh in 2000 was similar. Again, with continued restoration, it is expected that the restoring marsh vegetation will no longer be similar to the 1997 restricted marsh.
- Vegetation of the unrestricted marsh did not change from 1997 to 2000, as would be expected. Salt marsh vegetation is constantly changing in response to natural processes and events, however, these changes are often very subtle and not detected over the short-term. Because no significant changes in the unrestricted marsh or control marsh were noted, it can be stated that any detected changes within the restoring marsh were in fact due to tidal restoration, and not some other factors.

Table 3 also presents a dissimilarity percentage between the unrestricted and tide-restricted or restoring marsh. Dissimilarity near 100% suggests that the vegetation is very dissimilar. It is clear that after just 2 growing seasons of restored tidal flow, vegetation changes within the restoring marsh are not detectable. However, as restoration proceeds and as the culvert opening increases, it is expected that dissimilarity will decrease, suggesting that restoration is converging toward or becoming more similar to the unrestricted control marsh.

As noted from the ANOSIM results, vegetation of the unrestricted and restricted marshes is very different. By calculating similarity percentages (SIMPER), individual species that contribute to the community differences are ranked, in order of importance (Table 4). For 2000 data, *Spartina alterniflora* had an average percent cover rank in the unrestricted marsh of 3.14 (equates to an actual percent cover of 26-50%) and a much reduced average cover rank of 0.54 in the restricted marsh, as would be expected. This difference accounted for 9.4% of the overall dissimilarity between the unrestricted and restricted marsh. As the Hatches Harbor vegetation monitoring proceeds, shifts in the SIMPER percentages are expected, perhaps with *S. alterniflora* cover increasing in the restoring marsh and *Phargmites* or *Myrica* decreasing.

Nekton (Fish and Decapod Crustaceans)

Nekton was sampled from subtidal creeks and marsh pools using a $1m^2$ x 0.5m high throw trap, with a 3mm-mesh frame. Fifteen creek stations and 5 marsh pool stations were randomly established within the unrestricted marsh and 10 creek and 5 pool stations in the restricted marsh. To evaluate pre-restoration conditions, sampling was conducted at two-week intervals from June 1997 through September 1998 (additional sampling, fall, winter and spring, was conducted in 1997, but those data are not presented here). In 1999, following tidal restoration, the same stations were sampled bi-weekly from June through September.

Table 5 presents nekton density data (number m^{-2}), with creek and pool data combined and averaged over the June to September period. As noted, 9 fish species and 3 decapod crustaceans were collected, with the common mummichog (*Fundulus heteroclitus*) as the dominant species. Based on ANOSIM (Table 4), the following trends were found;

- In 1997, prior to restoration, the nekton community of the unrestricted marsh was much different from the tide-restricted marsh.
- Following tidal restoration the nekton community of the unrestricted marsh in 1999 was similar to the restoring marsh. This suggests that tidal restoration was successful at restoring the nekton community, at least within marsh creeks and pools.
- Following tidal restoration, there was no difference in nekton communities between the 1997 restricted marsh and the 1999 restoring marsh. It is difficult to interpret this finding. Given that the unrestricted and restoring marsh were similar in 1999, it was expected that the nekton community of the restoring marsh in 1999 would have changed from the 1997 restricted marsh.

- Nekton of the unrestricted marsh in 1997 and 1999 was similar, as expected.

Regarding the nekton findings, it should be stressed that these data relate only to tidal creeks and marsh pools and should not be interpreted to suggest that the entire tide-restricted Hatches Harbor marsh has been restored for nekton. The marsh surface is an important habitat for estuarine fishes, but has not been included in this preliminary analysis. Data are available on marsh surface utilization by nekton with Hatches Harbor before tidal restoration, but no post restoration data have been collected to date.

Overall Summary

- Prior to tidal restoration, vegetation of the tide-restricted Hatches Harbor salt marsh was very different from the unrestricted marsh. The unrestricted marsh was dominated by *Spartina alterniflora*, typical of New England salt marshes, while the restricted marsh was dominated by a diversity of non-salt marsh species, like *Rubus* and *Myrica pensylvanica*, and an aggressive grass common to tide-restricted marshes, *Phragmites*.
- Following restoration of tidal flow for 2 growing seasons, the vegetation remained different. As the culvert openings increase, it is expected that significant changes in the vegetation of the restoring marsh will occur. A trend toward a vegetation community like the unrestricted marsh is likely.
- Prior to restoration, the nekton community associated with marsh creeks and marsh pools of the tide-restricted marsh was different from the unrestricted marsh.
- During the first year of restored tidal flow, the nekton community of the restoring marsh became similar to the unrestricted marsh, documenting a positive influence of restored tidal flow on nekton communities. However, monitoring nekton use of the marsh surface is required to derive a more complete interpretation of the response of nekton to tidal restoration at Hatches Harbor.

Table 3. Results of one-way ANOSIM tests on pairwise comparisons of vegetation community data from Hatches Harbor salt marsh. Unrestricted marsh serves as the control. In 1997 the treatment marsh was under a regime of restricted tidal flow, while in 2000 the same treatment marsh was under restored tidal flow conditions. Bonferroni adjusted alpha = 0.05/4 = 0.0125.

Pairwise ANOSIM Comparisons	*p* value	% Dissimilarity
1997 Unrestricted vs. 1997 Restricted	<0.001	94%
2000 Unrestricted vs. 2000 Restoring	<0.001	95%
1997 Restricted vs. 2000 Restoring	0.441	
1997 Unrestricted vs. 2000 Unrestricted	0.286	

Table 4. SIMPER results showing the contribution of each species to the vegetation community dissimilarity noted by ANOSIM. Data presented are for the 2000 Unrestricted vs. 2000 Restoring marsh. Average cover ranks are according to the Braun-Blanquet scale (0=absent, 1=<1-5%, 2=6-25%, 3=26-50%, 4=51-75%, 5=76-100%). SIMPER provides data for all species encountered; only the top ranked species are shown here.

Species	Avg Cover Rank 2000 Unrestricted	Avg Cover Rank 2000 Restoring	% Dissimilarity
Spartina alterniflora	3.14	0.54	9.4
Salicornia sp.	1.78	0.04	5.8
Rubus sp.	0.00	1.91	5.1
S. alterniflora (dead)	1.30	0.37	4.7
Phragmites australis	0.00	1.24	3.8
Ascophyllum nodosum	1.27	0.00	3.7
Myrica pensylvanica	0.00	1.17	3.4

Table 5. Nekton density (number m^{-2}) in the unrestricted and restricted Hatches Harbor salt marsh in 1997, before tidal restoration, and 1999 with restored tides. Two-way ANOVA was performed to evaluate differences in species density between restricted 1997 and restoring 1999, and then between unrestricted 1997 and 1999. Nekton density data were log (x+1) transformed. Only one significant difference was noted (***, *Carcinus maenas*, p<0.01).

Species	Restricted 1997 n=90	Restoring 1999 n=90	Unrestricted 1997 n=120	Unrestricted 1999 n=120
Fundulus heteroclitus (mummichog)	22.84	28.38	18.99	14.77
Carcinus maenas (green crab)	0.46	1.47 ***	1.38	1.69
Crangon septemspinosa (sand shrimp)	0.18	0.14	2.16	0.69
Fundulus majalis (striped killifish)	0.10	0.36	0.28	0.58
Menidia menidia (Atlantic silverside)	0.12	0.20	0.65	0.18
Anguilla rostrata (American eel)	0.36	0.12	0.00	0.00
Apeltes quadracus (4-spine stickleback)	0.20	0.09	0.00	0.00
Gasterosteus aculeatus (3-spine stickleback)	0.10	0.10	0.00	0.00
Mugil curema (white mullet)	0.02	0.00	0.04	0.00
Syngnathus fuscus (pipefish)	0.03	0.01	0.00	0.00
Neopanopeus sayii (mud crab)	0.00	0.00	0.00	0.01
Pseudopleuronectes americanus (winter flounder)	0.00	0.00	0.01	0.00
Total Nekton	24.41	30.86	23.50	17.93

Table 6. Results of one-way ANOSIM tests on pairwise comparisons of nekton community data from Hatches Harbor salt marsh. 1997 represent pre-restoration data, and 1999 is one-year post tidal restoration. Bonferroni adjusted alpha = 0.05/4 = 0.0125. NS = not significant.

Pairwise ANOSIM Comparisons	*p* value
1997 Unrestricted vs. 1997 Restricted	<0.001
1999 Unrestricted vs. 1999 Restoring	NS
1997 Restricted vs. 1999 Restoring	NS
1997 Unrestricted vs. 1999 Unrestricted	NS

Biomass changes in the degraded salt marsh habitat
C. N. Farris, November 2001

Another measurement of the impact of increasing tidal exchange on the degraded salt marsh habitats within the restricted marsh was performed by quantifying interannual variation in *Phragmites* biomass. The hypothesis was that increasing tidal exchange would increase intrusion of saline surface waters into the interior of the restricted marsh. Subsequently, the more saline surface waters would penetrate marsh peats (Harvey et al 1997) resulting in sub-optimum conditions in the root zones of the invasive plant species in these habitats. Thsee increasing salinities along with an altered geochemical environment, would lead to physiological stress in the plants. This should be reflected in a decrease in plant productivity over time. To elucidate this trend, *Phragmites* biomass measurements were taken along a transect in the restricted marsh for three years 1997, 2000 and 2001.

Methods

Phragmites biomass was measured in the restricted marsh on regularly spaced plots along a transect perpendicular to the longitudinal axis of the mainstem tidal creek (see Figure 2). Sampling was performed in late August in each year at the peak of *Phragmites* above-ground production. In 1997, 0.25 square-meter plots were sampled every 20 m. In 2000 and 2000, 1.0 square-meter plots were sampled every 20 m. Live stems were counted in 1997 and 2001 and stem heights were measured in each plot for all three years. In 1997 and 2001, all live *Phragmites* stems were harvested by clipping and biomass was expressed as the mass in grams of stems from each plot after drying at 105^{o}C to constant weight.

Results and discussion.

A recent study in a Connecticut River tidal marsh found *Phragmites* biomass measured as 1) stem height to be 208 cm; 2) stem density to be 80 stems m^{-2} and 3) peak live standing crop to be 1300 g DW m^{-2} (Warren et al 2001). In the present study *Phragmites* biomass in the pre-restoration samples were comparable to the Connecticut values. However, stem heights differed significantly among years (ANOVA, $p<0.01$) and declined with increasing cross section area of the culverts and tidal range (significant at $p<0.05$, r^2 = 0.995; see Figure 12). This reduction of plant vigor is likely due to increased root-zone salinity (see Porewater Monitoring, above). The *Phragmites* stem height response was pronounced within 100 m of the creek where pore water sampling demonstrated largest increase in porewater salinity with increasing tidal exchange for all three post-restoration years.

Phragmites biomass and stem density did not change significantly between 1997 and 2001. Between 1997 and 2001, the location of peak biomass levels and peak stem densities seem to have shifted 40 to 60 m in from the creek. As there is an elevational gradient along this transect, the critical soil salinities may have shifted to a higher elevation after each increase in tidal exchange. This shift in root zone salinities may result in a corresponding shift in the location of optimal *Phragmites* vigor. Recent studies show that Phragmites responds rapidly to changes in hydrological and geochemical conditions (Windham and Lathrop 1999; Warren et al 2001).

In conclusion, *Phragmites* stands have been severely impacted by the increasing tidal exchange and the resultant higher salbo. This is especially true in the last two years, as stem density dropped 32% between the 2000 and 2001 growing seasons, compared to a 16% drop from 1997 to 2000.

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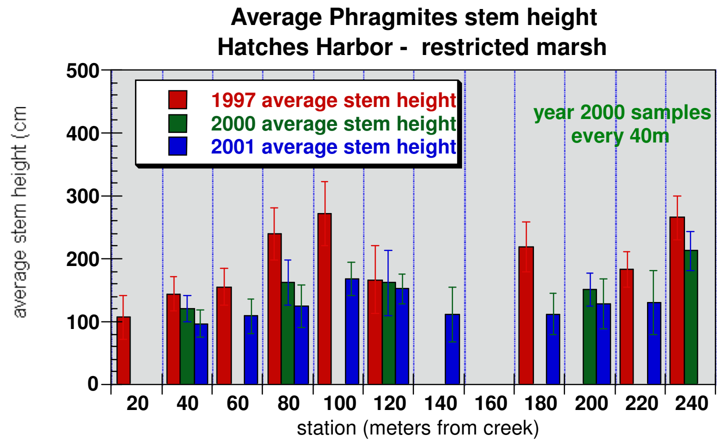

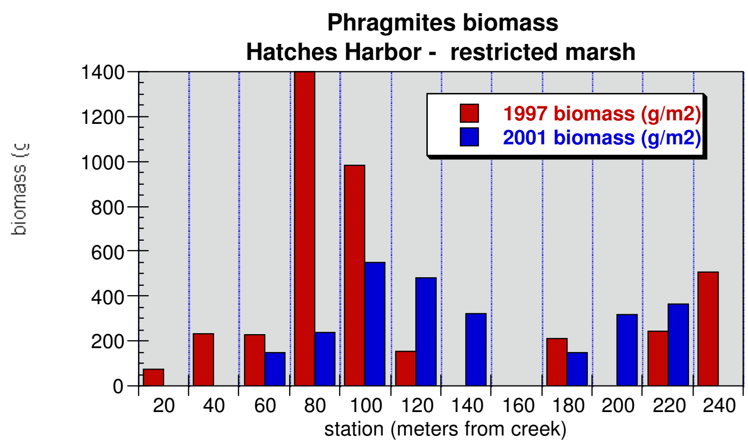

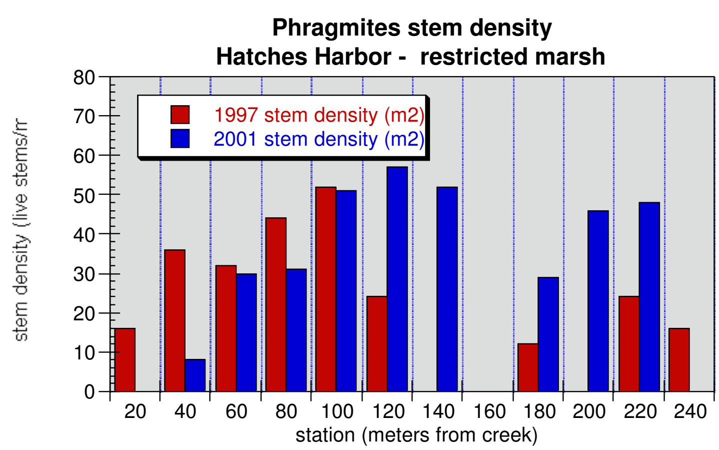

Figure 12. Three measures of biomass in restricted marsh *Phragmites* stands

Annual Report Conclusions

The restricted marsh in the Hatches Harbor is showing a significant response to three years of increasing tidal exchange. As hypothesized, these responses were seen first in hydrological parameters (water level, tidal range and water table). Tidal ranges have increased from 0.26 to 0.4 meters (MSL). The cross-sectional areas of the culverts have not only been increased 10 fold since 1997, but most of this area is at or near critical invert elevation. This enhances dewatering during low tides and allows for aerated peat surface at each low tide.

These were quickly followed by geochemical responses (porewater salinities and sulfides). Porewater salinities have increased significantly and increasing tide heights transported higher salinity porewater further into the marsh. Brackish water is now distributed over broad areas of marsh surface previously existing as freshwater wetlands. Porewater sulfides have remained at low levels since 1997. In 2001, sulfide levels actually decreased as drainage and aeration increased in the restricted marsh basin.

Some responses in ecological parameters were observed, after a temporal lag, as these parameters necessarily are an integrated response to changing environmental conditions. The species composition of the fish reveals interannual changes that may be a function of differences in microhabitat within the restoring marsh. The nekton community of the restoring marsh became more similar to that of the unrestricted marsh. Additional monitoring is needed to discern a better understanding of the nekton population response.

The changes in adult mosquito abundance and species composition may be influenced by interannual changes in seasonal precipitation. This relationship was evident in high trap counts were noted during high spring precipitation years (1998, 2000, 2001) and low trap counts in low precipitation years (1997, 1999). Brackish water mosquito populations proved especially sensitive to summer precipitation levels. These mosquitoes showed the highest population increase (four-fold) in 2001. This increase is probably attributable to increased marsh surface flooding after the last culvert opening.

Phragmites biomass and vigor significantly decreased with progressive increases in tidal exchange. The distribution of *Phragmites* in the part of the transect more distant from the creek may be evidence of peak biomass shifting to adjacent areas as the porewater of the restricted marsh surface at lower elevations become more saline.

Species composition of plant communities within the restricted marsh demonstrated no significant changes in response to the increasing tidal exchange as of 2000. A comparison of the restoring marsh vegetation to the unrestricted marsh vegetation reveals a clear difference between them, but with no significant trend towards greater similarity. It is expected that the two vegetative communities will converge over time, but that this parameter will have a longer response time.