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Fish Utilization of Restored, Created, and Reference Salt-Marsh Habitat in the Gulf of Maine

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Abstract.—In the Gulf of Maine region, projects to restore or create salt-marsh habitat to replace salt-marsh functions and values reduced or lost to tidal restriction are increasing. We assess fish utilization of marsh restoration and creation projects along the central Gulf of Maine coastline by addressing three questions: (1) how do fish assemblages in manipulated and reference marshes compare, (2) how do differences between manipulated and reference marshes change over time, and (3) how do fishes respond to different types of restoration? Fish utilization of restored and created marshes in New Hampshire and Maine (two created and four tidally restored marshes) is compared to adjacent reference marshes. The comparison of manipulated marshes with local reference marshes provides an internal standard for the monitoring of each restoration project, making it possible (1) to follow changes over time while accounting for natural variation and (2) to make valid comparisons about the magnitude and direction of changes between independent restoration projects. Our study provides the first density estimates for fish utilization of vegetated salt-marsh habitat in the Gulf of Maine. The highest fish densities from this study just overlap with the lowest fish densities reported from more southerly marshes. Overall, fish were distributed similarly among manipulated and reference marshes, and fish distribution did not change with time. Trends in the data suggest that fish utilize elevated marshes restored by dug channels to a lesser degree than impounded marshes restored by culverts. It appears that fish will readily visit restored and created marshes in assemblages similar to those found in reference marshes over the short term (one to five years post-restoration) but are subject to the influence of differences in tidal regime, access to marsh habitat, and vegetation density. In the large majority of cases, hydrologic restoration of tidally restricted marshes will improve a much larger area of fish habitat per unit cost than creation of new marsh and will not be subject to many of the constraints that limit the function of created marshes. The primary consideration in tidal restoration projects is not necessarily the cost of construction but the social, economic, and political issues that must be addressed. Often, tidally restricted marshes are in highly developed coastal areas where many individual property owners may perceive the increased tidal flow as a threat, even when flood hazard studies show that no such threat exists. In spite of this caution, thousands of hectares of coastal fish habitat can be improved through a concerted program to restore the hydrology of tidally restricted marshes in the Gulf of Maine.

The coastline of the Gulf of Maine is characterized in large part by existing or former salt marsh. High marsh, dominated by *Spartina patens*, typically fills large areas behind barrier beaches or in protected drowned valleys both within the Bay of Fundy region and from the Kennebec River in Maine to Provincetown, Massachusetts, while low marsh, dominated by *Spartina alterniflora*, forms a vegetated fringe along the shores of larger rivers and bays throughout the Gulf (Nixon 1982). Recent estimates of the areal coverage of salt marsh throughout the Gulf are 171 km^2 (Jacobson et al. 1987; Gordon and Cranford 1994), in spite of vast losses of this vulnerable ecosystem since colonial times that range from 84% in the Bay of Fundy (Gordon and Cranford 1994) to 25–50% in Massachusetts, New Hampshire, and Maine (Cook et al. 1993). Given the important place of salt marshes the coastal landscape, it is surprising how little we know of the role of salt-marsh ecosystems in supporting the fish community of the Gulf of Maine. The occurrence of fishes has been studied in a small number of salt-marsh in estuaries (Lamborghini 1982; Roman 1987; Murphy 1991; Ayvazian et al. 1992; Doering et al. 1995; Lazzari et al. 1996; Cartwright 1997), and fish diets and food webs have been investigated in fewer marshes still (Lamborghini 1982; Cartwright 1997; Deegan and Garritt 1997). The use of salt marshes as nurseries by postlarval and juvenile marine fishes and the value of salt marshes as feeding grounds by adult marine fishes are poorly understood for the Gulf of Maine (Acadian

coastal province, using classification of Cowardin et al. 1979) compared to the Viriginian (Smith et al. 1984; Rountree and Able 1992a, 1992b; Rountree and Able 1993; Szedlmayer and Able 1996); Carolinian (Shenker and Dean 1979; Weinstein 1979; Bozeman and Dean 1980; Weinstein and Walters 1981; Rogers et al. 1984; Hettler 1989; Kneib 1993; Kneib and Wagner 1994; Miltner et al. 1995; Irlandi and Crawford 1997); and Louisianan coasts (Boesch and Turner 1984; Felley 1987; Deegan et al. 1990; Deegan 1993; Peterson and Turner 1994; Minello and Webb 1997). Current understanding of the value of salt-marsh estuaries as fish habitat (derived from studies of marshes to the south of the Gulf of Maine) is comprehensively reviewed by Day et al. (1989), Rozas (1995), and Kneib (1997a).

Tidal restrictions such as those created by dikes, roads, railroads, and other marsh crossings present a large potential impact to the fish production value of salt-marsh habitat in the Gulf of Maine. A recent survey of New Hampshire coastal marshes determined that 25% of the total marsh area was being impacted by reduced tidal flow due to human-made structures (USDASCS 1994). Most tidal restrictions have been in place for decades and have been maintained with little thought to their impact on marsh ecosystems. These restrictions can reduce or eliminate the access of estuarine and marine fish to estuarine salt-marsh habitat and over time can lead to major changes in marsh geomorphology and vegetation (Roman et al. 1984; Rozsa 1988; Sinicrope et al. 1990; Frenkel and Morlan 1991; Rozsa 1995; Buchsbaum et al. 1997; Burdick et al. 1997; Chadwick 1997; Portnoy and Valiela 1997; Weinstein et al. 1997; Orson et al. 1998). The abnormal patterns of draining and flooding created by tidal restriction can lead to subsidence of salt-marsh soils, loss of *Spartina patens* high marsh, expansion or invasion of the salt-marsh cattail *Typha angustifolia*, and invasion by *Phragmites australis* or purple loosestrife *Lythrum salicaria*.

There is growing interest in restoring Gulf of Maine salt marshes that have been degraded by tidal restrictions (Reiner 1989; Cook et al. 1993; Dionne 1994; USDASCS 1994; Bryan et al. 1997; Dionne et al. 1998). Salt-marsh creation is already used as an approach to mitigate functions and values lost when areas of marsh are disturbed or destroyed by development or other human activities (Reiner 1989; Bosworth and Short 1993). Restoration refers to activities that aim to enhance degraded sites within naturally occurring systems or habitats; creation refers to activities designed to establish habitats de novo at a site where no such habitat exists naturally (Reiner 1989; Kusler and Kentula 1990; Sinicrope et al. 1990; Matthews and Minello 1994; Burdick et al. 1997; Weinstein et al. 1997). In this study, we assess the utilization by fishes of restored and newly created salt marshes along the central Gulf of Maine coastline by asking (1) how does the fish assemblage of the manipulated marsh compare with that of an appropriate reference marsh, (2) how does the difference between the fish assemblage of the manipulated marsh and the reference marsh change over time, and (3) how do fishes respond to different types of marsh restoration? We are interested in these questions specifically as they apply to vegetated marsh as opposed to intertidal or subtidal creeks or open estuarine waters.

Methods

Study Sites

The restored, created, and reference marshes in this study were along the New Hampshire and southern Maine coast (Figure 1, Table 1). For each restored or created marsh we selected an adjacent undisturbed area of natural marsh to serve as a reference (Figure 1) so that each manipulated marsh had a paired reference marsh. At each study site, manipulated and reference marshes were sampled simultaneously so that they were flooded by the same tide and, insofar as possible, at similar points during the cycle of tidal inundation. The one exception to this procedure was at the Submarine (SUB) site (Figure 1E) in 1997, where the reference marsh at Inner Cutts Cove (ICC) was separated from the manipulated site by a roadway and was sampled 24 h after sampling of the SUB created marsh. In each year, fish sampling occurred in the early and late growing season (June–July and August–October, respectively). Percentage of plant cover, water table, soil salinities, soil organic carbon, and fish abundances were measured at treatment and reference marshes at each site.

Two of the four tidal restoration sites were in Stratham, along the southwest shore of New Hampshire's Great Bay. The Sandy Point (SDPT) marsh fringes the open waters of Great Bay and lies within the Great Bay National Estuarine Research Reserve (Figure 1F, Table 1). The marshes fringing Great Bay tend to be dominated by *Spartina alterniflora* low marsh with bands of *Spartina pat-*

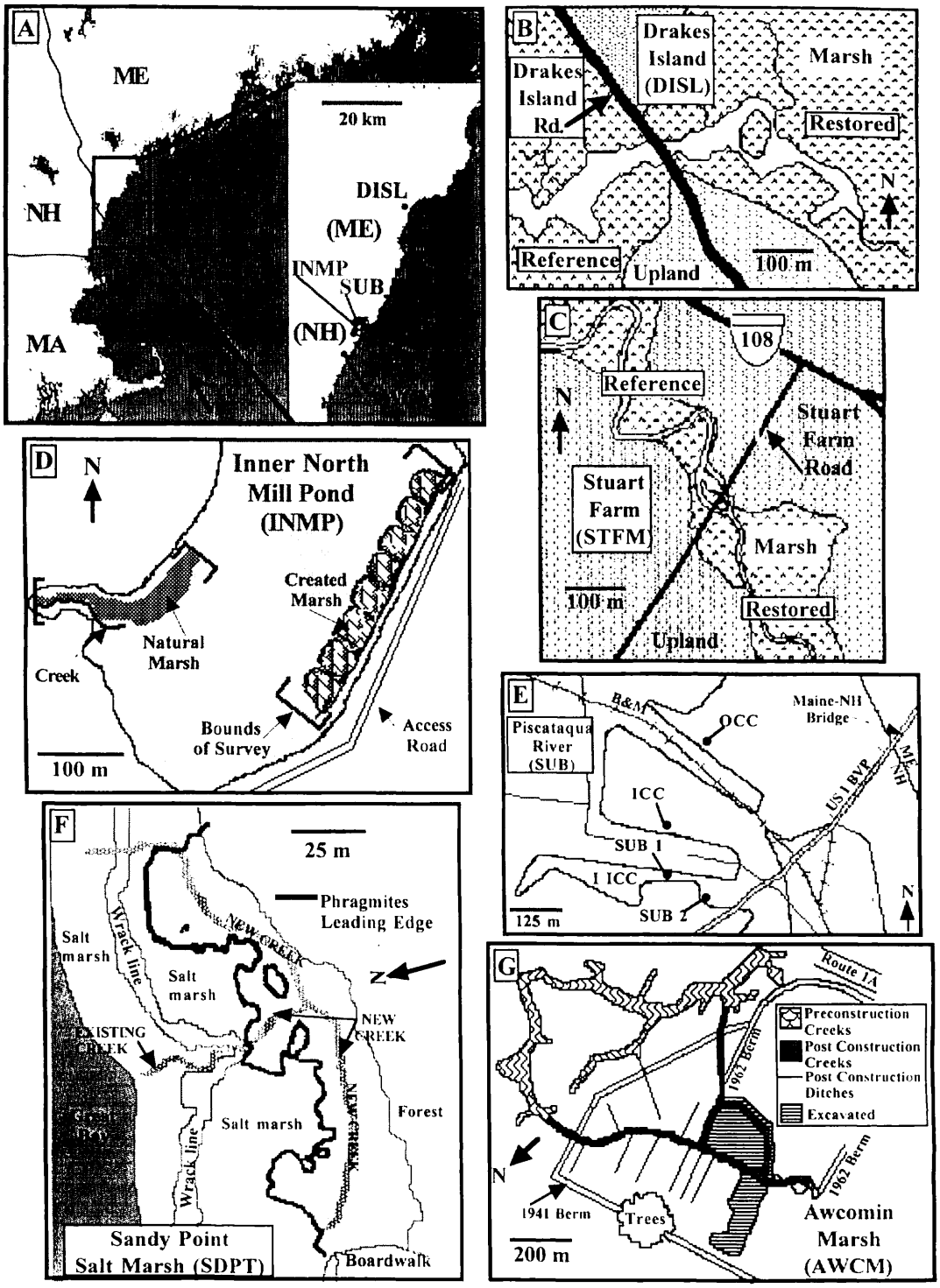

FIGURE 1.—Location of study sites and size, shape, and juxtaposition of restored, created, and reference marshes.

TABLE 1.—Overview of marsh restoration and creation study sites. Sample size (n) refers to the total number of fish samples collected at each reference and each treatment marsh during the entire study (e.g., n = 4 indicates four samples at the reference marsh and four samples at the treatment marsh).

Site	Location	History	Treatment	Monitoring
INMP	Inner North Mill Pond, Portsmouth, New Hampshire (NH); 0.4 ha	Railroad and industrial waste dump; mitigation for port expansion	Planted *Spartina alterniflora* in 1992 and 1993	1993, 1997; n = 4
SUB	Inner Inner Cutts Cove, Submarine Site, Portsmouth, NH; 0.1 ha	Ledge shoreline; mitigation for port expansion	Planted *Spartina alterniflora* in 1994 and 1996	1997; n = 4
AWCM	Awcomin Marsh, Rye, NH; 6.0 ha	Dredge spoil berm (20 cm layer) with *Phragmites* invasion	Dug channels to restore tidal flow and drainage in 1992	1993, 1995, 1996; n = 6
SDPT	Sandy Point, Great Bay, Stratham, NH; 1.5 ha	*Phragmites* berm (10 cm to 20 cm)	Dug channels to restore tidal flow and drainage in 1994	1995, 1996; n = 4
STFM	Stuart Farm Marsh, Squamscott River, Great Bay, Stratham, NH; 4.5 ha	Marsh impounded by flap gate since 1970; marsh surface subsided (15 cm to 30 cm)	Replacement of flap gate with large culvert in 1993	1993, 1995, 1996; n = 7
DISL	Drakes Island Marsh, Webhannet River, Wells, Maine; 16.0–20.0 ha	Marsh impounded for 100 years, most recently by a flap gate; marsh surface subsided (0.6 m to 0.9 m)	Loss of flap gate in 1988	1995, 1996; n = 4

ens and *Scirpus robustus* high marsh. The tidal connection to the landward half of this marsh was impaired by a berm of unknown origin 10–20 cm high. The restriction was associated with a stand of *Phragmites* that had invaded the site. In 1994, channels were dug in the berm to improve tidal exchange. The second Stratham restoration site, Stuart Farm Marsh (STFM), lies within a *Spartina patens*-dominated riverine marsh (technically a fluvial minor salt marsh as described in Kelley et al. 1988) near the mouth of the Squamscott River as it flows into Great Bay (Figure 1C, Table 1). This marsh is upstream of the access road to Stuart Farm and had been deprived of tidal exchange since the 1970s when a bridge was replaced by a narrow culvert and flap gate. Tidal flow was restored by the replacement of the limiting structures by a much larger (2.1 m diameter) arched culvert in October 1993. Measurements of marsh surface elevations in the area deprived of tidal exchange averaged 33 cm lower on average than elevations in the unimpounded reference marsh downstream of the access road, indicating marsh soil subsidence.

The two remaining tidal restoration projects in this study were behind Gulf of Maine barrier beaches. One site was in the Awcomin Marsh (AWCM) in Rye, New Hampshire (Figure 1G, Table 1). This marsh had been used as a disposal site for dredge material from Rye Harbor, resulting in a 20 cm increase of surface elevation of the marsh within a higher berm. In 1992, intertidal creeks were dug to improve tidal flow as a strategy to halt the invasion of *Phragmites*. The channels followed the path of original creeks that had been filled with dredge material. The new creeks were shallower (0.5–1.0 m) than the natural tidal channels into which they drained (2.0 m). The second back-barrier tidal-marsh restoration site was in the Drakes Island Marsh (DISL) of the Webhannet River estuary in Wells, Maine, within the boundaries of the Wells National Estuarine Research Reserve (Figure 1B, Table 1). This site has a long history of impoundment for use as pasture (since circa 1848), first by a dike and then by a road with a box culvert and water control structure from the 1920s to the 1950s. The existing culvert under the road was installed in the 1950s and

fitted with a flap gate to prevent tidal flow into Drakes Island Marsh. In the winter or spring of 1988, the flap gate fell off, and tidal flow was partially restored through the existing narrow pipe (0.9 m minimum diameter). The mean elevation of the impounded marsh surface was 73 cm lower than the marsh downstream of the culvert. The tidal restoration sites were of two types: (1) those restricted by berms (SDPT and AWCM) and (2) those restricted by culverts (DISL and STFM).

The two created marshes were established as mitigation for the expansion of the New Hampshire Port Authority ship docking facilities on the Piscataqua River in Portsmouth. Degraded shoreline and debris were excavated, and fine-grained sediment was imported to the sites and contoured to appropriate elevations. One marsh, Inner North Mill Pond (INMP), was in an urban tidal basin (Figure 1D, Table 1). The other created marsh was planted in two discrete areas (SUB 1 and SUB 2 in Figure 1E) near a decommissioned submarine used as a museum in a tidal backwater of the Piscataqua River known as Inner Inner Cutts Cove, downstream of Inner North Mill Pond (see also Table 1). All created marshes were planted with the low-marsh dominant *Spartina alterniflora*. The INMP marsh was planted in 1992 and 1993, and the SUB marsh was planted in 1994 and 1995.

Field Sampling

In this study, we adapted fyke nets (Figure 2) to collect fishes, using a known area of marsh, as the fishes left the marsh as water ebbed during evening spring tides. Nekton assemblages were assessed with fyke nets set to sample evening ebb spring tides. In general, a given manipulated marsh and its corresponding reference marsh were within the same water body and sampled on the same tide. We would have been unable to appropriately compare restored, created, and reference marshes without adjusting for the area of vegetated marsh sampled. Flume weirs (Kneib 1991), drop samplers (Zimmerman et al. 1984), and other enclosure sampling methods are used specifically to estimate nekton densities in vegetated marsh habitats (Kneib 1997a; Rozas and Minello 1997). In this study, shallow flooding, low nekton densities, and a need for nondestructive, removable gear led us to a low-cost alternative to enclosure sampling. Most methods of estuary and salt-marsh nekton sampling (trawls nets, throw nets, seines, flume nets, weirs, pit traps, minnow traps) are not dimension-adjusted in practice (Varnell and Havens 1995), that is, the contents of the sample is not related to the dimensions of the habitat from which it is sampled. Therefore, these data do not provide fish abundance estimates per unit of area sampled (i.e., density), although it is possible to estimate densities with some of these methods. Trawling, seining, and throw-net techniques are not effective in emergent vegetation.

The nylon mesh nets used in this study were constructed of a series of four compartments held open by square frames or fykes, with 15-m wings attached to the first and largest (1.2-by-1.2-m frame) fyke opening. After the first compartment, each compartment contained an internal net funnel connected to the previous fyke frame, with the smaller end of the funnel

FIGURE 2.—Photograph of fyke net in place before sampling an evening high tide. (See "Methods" section for explanation of the sampling process.)

emptying into the center of the compartment. This design compelled fish to swim into the cod end of the net with little chance of escape. The wings and the first three compartments were made of 1.27-cm bar mesh, with the final cod end made of 0.63-cm bar mesh. At low tide, nets were set at the lower edge of the marsh with wings at 45°, fykes upright, and cod end and wings anchored. The wing top line was buoyed and set so that nekton could enter the marsh area by the side or tied down until high tide. At high tide, we staked the flooded area of marsh that would be fished by the net to calculate the area fished. The catch was placed into buckets of water once the tide had receded below the level of the first compartment. All fish and crustaceans were counted and identified to species, and total biomass of each species was measured. Up to 30 individuals of each fish species were measured for total length, sampled haphazardly from the bucket with an aquarium net. For crustaceans, we measured the maximum carapace width for green crab *Carcinus maenus* only.

For most years at most sites, fish were sampled once in the early salt-marsh growing season (June or early July) and once in the late growing season (late August through October), providing the minimum replication for the purposes of statistical analyses. The small sample size was dictated by the very limited funding available to monitor marsh restoration success. In 1993, nekton were sampled on two dates in the fall only at the AWCM, STFM, and INMP sites after restoration and creation efforts at these sites were completed. A total of 3,275 fish were collected from 62 samples during the course of the study, excluding 2,017 of 2,018 fish from a school of Atlantic herring *Clupea harengus* captured at Outer Cutts Cove (an additional reference site associated with the SUB site [Figure 1E]). These fish were removed from the data for analysis because their extremely high density represented an outlier. The area of marsh sampled ranged from 8.7 to 870.0 m^{-2}, with a mean of 317.0 m^{-2}. A total of 1.89 ha of vegetated marsh was sampled during the course of the study.

Analyses

Comparisons between treatment (restoration and creation) sites and reference sites for fish density, total length, and species number were made using two-way analysis of variance (ANOVA) ($p < 0.05$) to test the effects of treatment (i.e., restoration or creation) and year (change over time). The same approach was used to test for spatial and temporal variation among three reference sites within the same tidal reach of the Piscataqua River (Figures 1D and 1E). Fish density data were transformed (ln, sine or square root) as needed to reduce heterogeneity of variance. Data transformations did not change the incidence of statistical outcomes; therefore, all results are presented in original (i.e., not transformed) values. Crustacean data were not analyzed for this report.

Results

We collected 15 fish and 4 crustacean species from the 13 marsh areas sampled (Table 2). Seven species of fishes were marsh residents, four were diadromous, three were transients, and one was an "accidental" freshwater visitor. Comparison of pooled presence–absence data within treatment (restored and created) marshes with such data in reference marshes found nine species in common, four unique species in the treatment marshes (*Apeltes quadracus, Lepomis* sp., *Morone americana, Alosa sapidissima*), and two unique species in the Outer Cutts Cove (OCC) (Figure 1E) reference site (*Clupea harengus, Morone saxatilis*). The number of species ranged from two to eight in the marsh areas sampled. *Fundulus heteroclitus* occurred in all marshes whereas *Menidia menidia* occurred in 71% of the reference and 67% of the treatment marshes. *Anguilla rostrata* were taken in all reference marshes and in 67% of manipulated marshes. *Pseudopleuronectes americanus,*[1] *Gasterosteus aculeatus, Pungitius pungitius, Fundulus majalis*, and *Microgadus tomcod* were seen less frequently in both treatment and reference marshes (each species was present in 15–43% of reference marshes and 17% of treatment marshes). *Apeltes quadracus, Alosa sapidissima*, and *Morone americana* were only in treatment marshes, with *A. quadracus* at two of the six sites (33% occurrence) and the latter species at only one of the six sites (17% occurrence). The largest differences in the number of species between treatment and reference marshes were found at the STFM (culvert) site, where more species were found in the restored marsh, and at the SDPT (berm) site, where more species were found in the reference marsh (Table 2).

The marsh surface species assemblage identified in this study represents 29% of the species known to occur in marsh-dominated estuaries in the Gulf of Maine (Table 3). Fish life history habits were derived from the classification by McHugh as de-

[1] Cooper and Chapleau (1998) have revised the scientific name of winter flounder to *Pseudopleuronectes americanus*.

TABLE 2.—Fish species occurrences at all study sites. The column headings Created, Culvert, and Berm refer to types of restoration or creation projects. Data are pooled across all samples (n = 35 for reference sites; n = 27 for restoration or creation sites). O = reference marsh, X = restored or created marsh. See Table 1 for full spellings of site names.

Scientific name (common name)	Life history [a]	Created			Culvert		Berm	
		INMP	SUB	OCC [b]	DISL	STFM	AWCM	SDPT
		Fish						
Fundulus heteroclitus (mummichog)	r	O X	O X	O	O X	O X	O X	O X
Menidia menidia (Atlantic silverside)	r	O X			O X	O X	O X	O
Pungitius pungitius (ninespine stickleback)	r					X	O	
Anguilla rostrata (American eel)	r or m (c)	O X	O X	O	O	O X	O	O X
Fundulus majalis (striped killifish)	r					O		O X
Gasterosteus aculeatus (threespine stickleback)	r				O X			O
Gasterosteus wheatlandi (blackspotted stickleback)	r				O X			O
Pseudopleuronectes americanus (winter flounder)	t			O		X		
Microgadus tomcod (Atlantic tomcod)	t	O	O X	O				
Morone saxatilis (striped bass)	m (a)			O				
Clupea harengus (Atlantic herring)	t			O				
Apeltes quadracus (fourspine stickleback)	r				X	X		
Lepomis sp. (sunfish sp.)	f					X		
Morone americana (white perch)	m (a)					X		
Alosa sapidissima (American shad)	m (a)	X						
Total (15 species)		4 4	3 3	6	5 5	4 8	4 2	6 3
		Crustaceans						
Palaemonetes sp. (shore shrimp)		X	O X	O		O X	O X	O X
Crangon septemspinosa (sevenspine bay shrimp)					O X	O X	O X	O X
Carcinus maenas (green crab)		O X	O X	O	O X	O X	O X	O X
Homarus americanus (American lobster)			O					
Total (4 species)		1 2	3 2	2	2 2	3 3	3 3	3 3

[a] r = marsh resident, f = freshwater, m = migratory, c = catadromous, a = anadromous, and t = marine transient.
[b] OCC is an additional reference site (see Figure 1E).

scribed in Ayvazian et al. (1992). Here we use a simplified version, collapsing the seven categories to three: resident, migratory, and transient. Estuarine residents are species that spawn and spend a significant part of their life in the estuary. Migratory fish are those with an anadromous or catadromous life

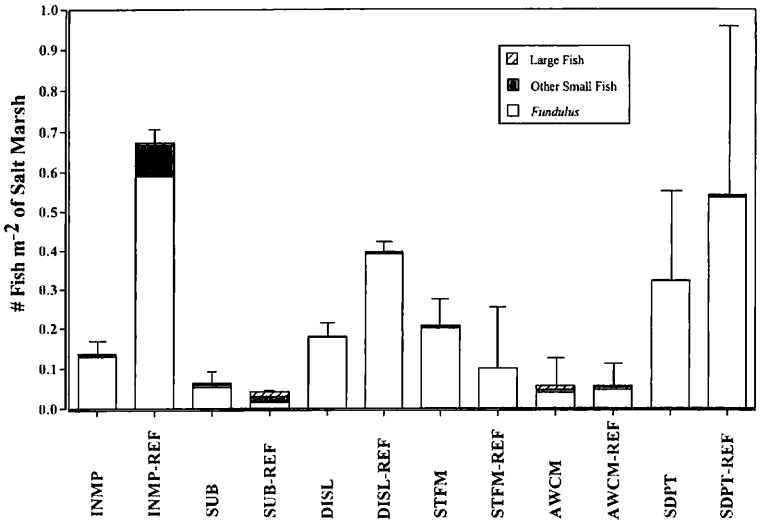

FIGURE 3.—Mean total fish density at all paired treatment (restored and created) and reference (REF) study sites, showing contribution of *Fundulus*, other small fish, and large fish to total. Vertical bars = 1 standard error of the mean for total density. See Table 1 for full spellings of site names.

history. Transient life history classifications for this study include the "spawner" (marine species that spawn in estuaries), "nursery" (marine species that spawn in ocean waters but use the estuary as a nursery), and "marine" (marine fish that visit the estuary as adults) classifications.

Mean total fish density ranged from $0.04/m^{-2}$ for the SUB reference marsh to $0.67/m^{-2}$ for the INMP reference marsh (Figure 3). *Fundulus* comprised from 39 to 99% of mean fish density at each marsh, followed by other small fish (0–31%) and large fish (0–29%; see below for large and small

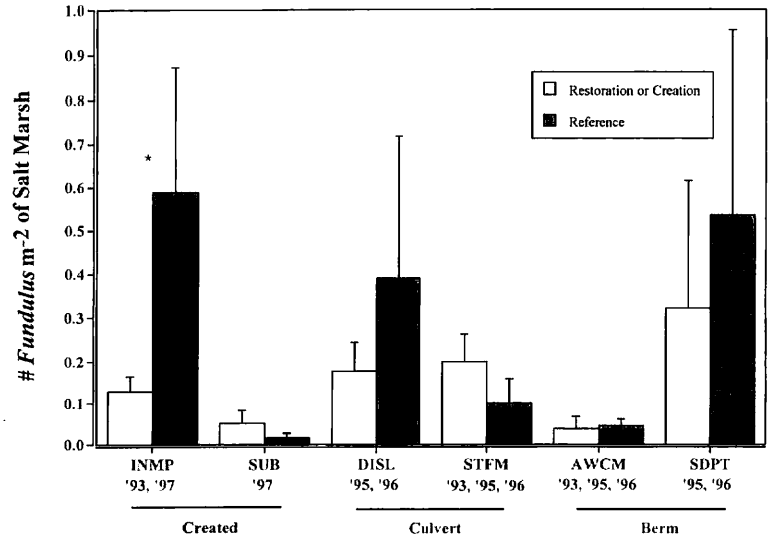

FIGURE 4.—Comparison of mean density of *Fundulus* between treatment and reference marshes at each study site, by two-way ANOVA. Treatment, year, and treatment by year were all significant ($p < 0.05$) for Inner North Mill Pond (INMP). There were no other significant main effects or interactions. Vertical bars = 1 standard error of the mean. The asterisk indicates the significant comparison. See Table 1 for full spellings of site names.

TABLE 3.—Fish species known to occur in Gulf of Maine salt-marsh estuaries. With the exception of the present study, all data are from unvegetated intertidal and subtidal creeks and channels. The Herring River study was located in Massachusetts, and the other studies were located in Maine, with the exception of the present study, which includes sites in New Hampshire and Maine. (For the second Little River and Webhannet River studies, data from vegetated marsh were not included due to paucity of species; these species also occurred in the present study's creek data. The Little and Webhannet rivers occur within the Wells National Estuarine Research Reserve, with Wells Harbor located at the mouth of the Webhannet River.)

Scientific name (common name)	Life history [a]	Present study	Herring River [b]	Little River [c]	Little River [d]	Wells Harbor [e]	Webhannet River [f]	Georgetown Little River [g]	Kennebec Point [h]	Bass Harbor [i]
Alosa aestivalis (blueback herring)	m(a)		X	X		X		X	X	
Alosa mediocris (hickory shad)	m(a)		X							
Alosa pseudoharengus (alewife)	m(a)		X	X		X		X	X	
Alosa sapidissima (American shad)	m(a)	X		X						
Brevoortia tyrannus (Atlantic menhaden)	t		X	X						X
Clupea harengus (Atlantic herring)	t	X		X		X		X	X	X
Ammodytes americanus (American sand lance)	t			X				X	X	
Anguilla rostrata (American eel)	m(c)	X	X	X	X	X		X		X
Apeltes quadracus (fourspine stickleback)	r	X	X	X	X	X	X	X	X	X
Gasterosteus aculeatus (threespine stickleback)	r	X		X	X	X	X	X	X	X
Gasterosteus wheatlandi (blackspotted stickleback)	r	X		X	X	X	X	X	X	X
Pungitius pungitius (ninespine stickleback)	r	X		X	X	X	X	X	X	X
Cyclopterus lumpus (lumpfish)	t			X				X	X	
Liparis atlanticus (seasnail)	t			X						
Decapterus macarellus (mackerel scad)	t								X	
Fundulus heteroclitus (mummichog)	r	X	X	X	X	X	X	X	X	X
Fundulus majalis (striped killifish)	r	X	X						X	
Gadus morhua (Atlantic cod)	t								X	
Microgadus tomcod (Atlantic tomcod)	t	X		X	X	X		X	X	
Pollachius virens (pollock)	t							X	X	X
Urophycis chuss (red hake)	t								X	
Urophycis tenuis (white hake)	t			X		X		X	X	

TABLE 3.—(continued.)

Scientific name (common name)	Life history [a]	Present study	Herring River [b]	Little River [c]	Little River [d]	Wells Harbor [e]	Webhannet River [f]	Georgetown Little River [g]	Kennebec Point [h]	Bass Harbor [i]
Pholis gunnellus (rock gunnel)	t			X				X		
Menidia beryllina (inland silverside)	r					X				
Menidia menidia (Atlantic silverside)	r	X	X	X	X	X	X	X	X	X
Menidia peninsulae (tidewater silverside)	r		X							
Morone americana (white perch)	t	X	X						X	
Morone saxatilis (striped bass)	t	X		X					X	
Mugil cephalus (striped mullet)	t			X						
Hemitriperus americanus (sea raven)	t								X	
Myoxocephalus aeneus (grubby)	t			X						
Myoxocephalus octodecimspinosus (longhorn sculpin)	t			X					X	
Myoxocephalus scorpius (shorthorn sculpin)	t							X	X	
Cryptacanthodes maculatus (wrymouth)	r							X		
Osmerus mordax (rainbow smelt)	m(a)			X		X		X	X	
Peprilus tricanthus (butterfish)	t			X						
Petromyzon marinus (sea lamprey)	m(a)			X						
Pleuronectes ferrugineus (yellowtail flounder)	t									X
Pseudopleuronectes americanus (winter flounder)	t		X	X	X			X	X	
Scopthalmus aquosus (windowpane flounder)	t			X						
Pomatomus saltatrix (bluefish)	t		X	X					X	
Salmo salar (Atlantic salmon)	m(a)			X						
Salmo trutta (brown trout)	m(a)			X						
Salvelinus fontinalis (brook trout)	m(a)			X						X
Scomber scombrus (Atlantic mackerel)	t		X	X						
Sphyraena borealis (northern sennet)	t			X						

TABLE 3.—(continued.)

Scientific name (common name)	Life history [a]	Present study	Herring River [b]	Little River [c]	Little River [d]	Wells Harbor [e]	Webhannet River [f]	Georgetown Little River [g]	Kennebec Point [h]	Bass Harbor [i]
Syngnathus fuscus (northern pipefish)	t		X	X			X	X	X	X
Tautogolabrus adspersus (cunner)	t			X	X					
Total (48 species)		14	15	35	10	15	6	21	27	13

[a] r = marsh resident, m = migratory, c = catadromous, a = anadromous, and t = marine transient.
[b] See Roman 1987.
[c] M. Dionne, Wells National Estuarine Research Reserve, unpublished data.
[d] See Murphy 1991.
[e] See Ayvazian et al. 1992.
[f] See Murphy 1991.
[g] See Lamborghini 1982.
[h] See Lazzari et al. 1996.
[i] See Doering et al. 1995.

fish designation). Estimates of standing crop were obtained from fish density estimates (Figure 3) and area of the site (Table 1). The two created marshes supported mean standing crops of 544 fish (INMP) and 80 fish (SUB). The two berm marshes supported populations of 2,760 (AWCM) and 4,830 (SDPT) fish, and the two culvert marshes provided habitat for 9,900 (STFM) and 28,800–36,000 (DISL) fish.

Comparisons of fish occurrence between treatment and reference marshes at the six study sites revealed few significant differences in density, total length, or species number. Mean *Fundulus* density was significantly higher at the INMP reference site than at the INMP creation site (Figure 4) due to the great difference between creation sites and their reference sites in the first year after planting. Four years

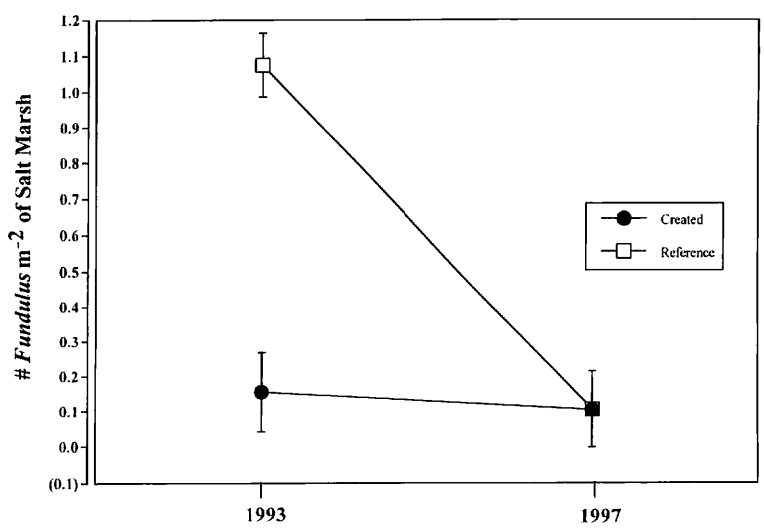

FIGURE 5.—Interaction of density and year effects for the created and reference marshes at Inner North Mill Pond. Data were analyzed as in Figure 4. Treatment, year, and treatment by year were all significant ($p < 0.05$). Vertical bars = 1 standard error of the mean.

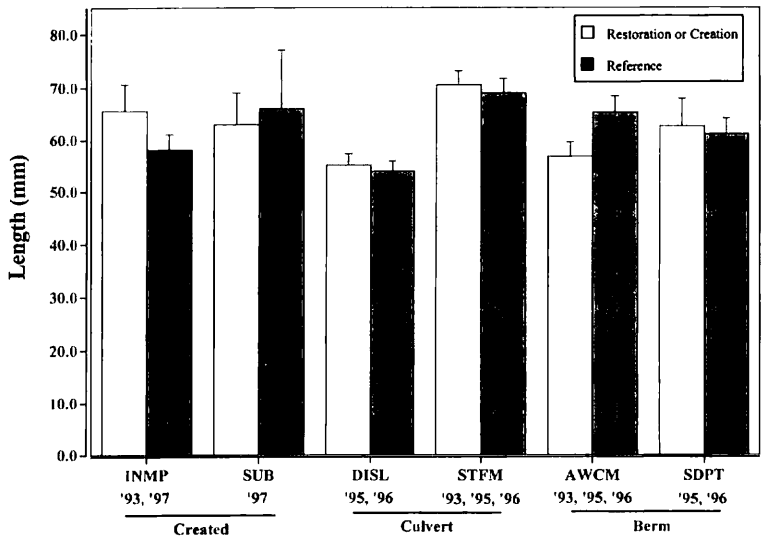

FIGURE 6.—Comparison of mean *Fundulus* total length between treatment and reference marshes at each study site. Vertical bars = 1 standard error of the mean. See Table 1 for full spellings of site names.

later, there was no difference between these marshes (Figure 5) as fish density in the reference marsh was lower in 1997 than in 1993. The total length (TL) of *Fundulus* in the fyke nets ranged from 30 to 130 mm TL, and the means were remarkably similar between treatment and reference marshes at every site (Figure 6), ranging from 53.9 to 70.5 mm. The occurrence and direction of significant statistical outcomes ($p < 0.05$) of analyses for density and total length of all small fishes (*Fundulus* plus other small fish) were the same as the analyses of density and total length for *Fundulus* alone.

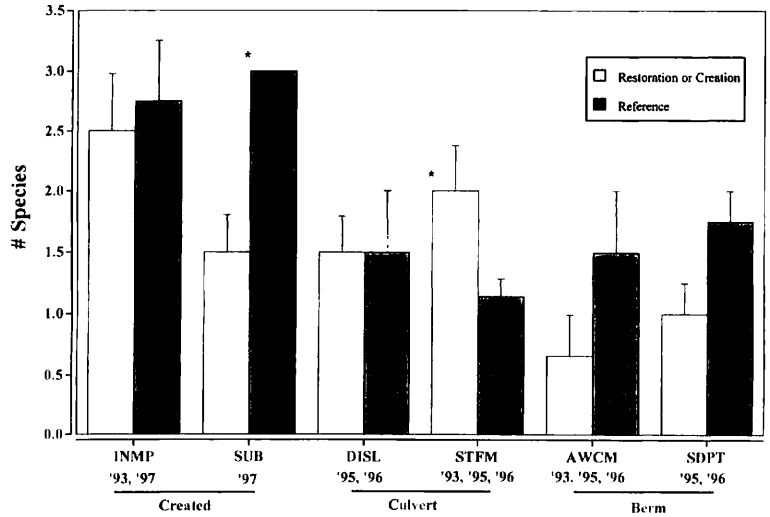

FIGURE 7.—Comparison of mean number of species of fishes between treatment and reference marshes at each study site. Data were analyzed as in Figure 4, except for SUB, where one-way ANOVA was used because the site was sampled in a single year (1997). There were significant differences ($p < 0.05$) in species number between the restored and created marshes at the STFM and SUB sites and their respective reference sites. There were no other significant main effects or interactions. Vertical bars = 1 standard error of the mean. Asterisks indicate significant comparisons. See Table 1 for full spellings of site names.

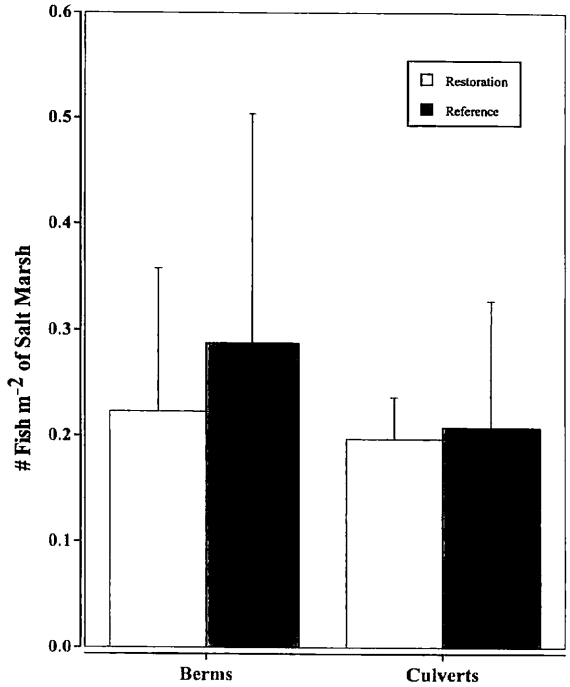

FIGURE 8.—Mean density of small fish (*Fundulus* plus other species) at culvert and berm sites (n = 10). Data were analyzed as in Figure 4. There was no significant effect of treatment or year for either culvert or berm restoration sites. Vertical bars = 1 standard error of the mean.

Mean number of fish species per sample was significantly greater at the STFM restored marsh than at the reference marsh and significantly greater at the SUB reference site at Inner Cutts Cove (ICC, Figure 1E) than for the SUB created marsh (SUB 1 and SUB 2 in IICC, Figure 1E) at Inner Inner Cutts Cove (Figure 7). There were no other significant differences for number of species between treatment marshes and their respective reference marshes. There was a trend toward fewer species in restored berm marshes (compared to reference marshes), whereas species numbers in the restored culvert marshes were the same as or greater than the numbers in the reference marshes.

When data were pooled within berm sites and culvert sites, there were no significant differences in small-fish density between restoration and reference marshes or between marsh types (Figure 8). The mean density was essentially the same for restored culvert marshes and reference culvert marshes ($0.20/m^{-2}$ and $0.21/m^{-2}$, respectively). The mean fish density was lower for the restored berm marshes than reference berm marshes ($0.223/m^{-2}$ and $0.288/m^{-2}$, respectively).

Of the 14 marine and estuarine fish species identified from all study sites, 3 species, *Anguilla rostrata*, *Microgadus tomcod*, and *Morone saxatilis*, were designated as "large fish" due to their three- to eightfold greater length compared to small fish. *Anguilla* occurred in 32% of all samples, and the other two species were rare (5% and 2% of all samples for *M. tomcod* and *M. saxatilis*, respectively). The mean density of large fish across all study sites did not differ between treatment and reference marshes (Figure 9). The same result was found for mean total length (Figure 9), but in this case, there was a trend toward larger mean size for fish captured in the reference marshes.

A comparison of the three reference marshes within a series of linked tidal basins draining into the Piscataqua River revealed significant effects of marsh basin, year, and the interaction of marsh basin and year for the mean density of *Fundulus* and small fish (Figure 10). There was a significant effect of year alone for large fish due to an increase in mean density from $0.002/m^{-2}$ in 1993–$0.010/m^{-2}$ in 1997. Six species were identified from OCC, compared to 4 and 3 species respectively from INMP

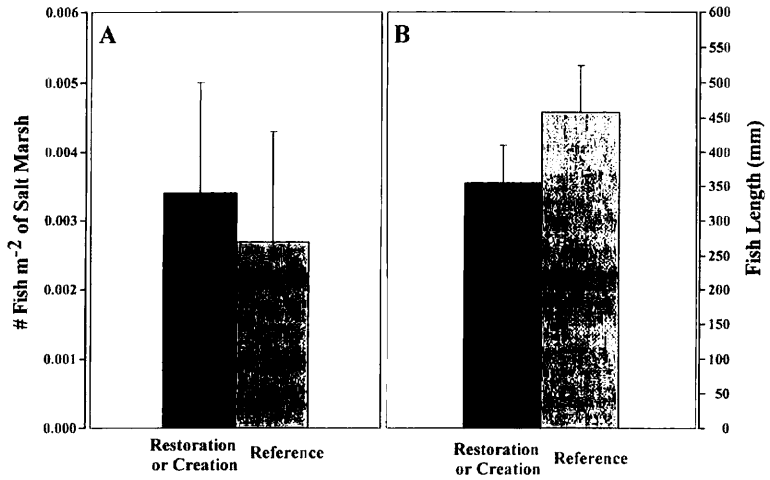

FIGURE 9.—Comparison between reference (n = 29) and treatment (data pooled across all study sites; n = 28) marshes for large fish density (A) and total length (B). There were no significant differences for either variable ($p > 0.05$ by Kruskal Wallis for fish density). A nonparametric test was used for the density analysis due to the large number of zero values. Vertical bars = 1 standard error of the mean.

and ICC (Table 2). Two of these fishes, *Morone saxatilis* and *Clupea harengus*, were unique to Outer Cutts Cove. There were no significant differences by marsh or year for mean species number, which ranged from 2.5 (ICC) to 3.5 (OCC).

Discussion

In this study we monitored fish utilization of manipulated and reference marshes at six locations within a small geographic area (less than 50 km along the coast from Drakes Island Marsh in Wells, Maine to

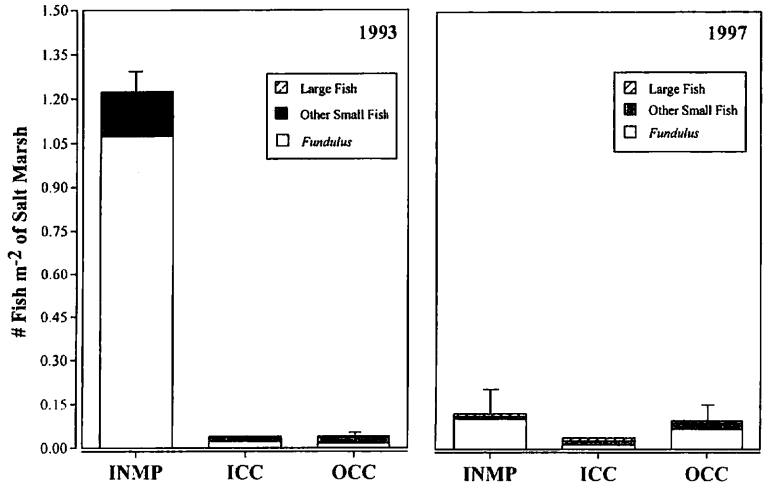

FIGURE 10.—Mean fish densities at three reference marshes in Portsmouth, New Hampshire during 1993 and 1997 (number of samples collected = 4 for each marsh). Both main effects (marsh, year) and their interaction were significant ($p < 0.05$ by two-way ANOVA) for mean *Fundulus* density and mean small-fish density due to the high densities of fish in the INMP reference marsh in 1993. Vertical bars = 1 standard error of the mean. See Table 1 for full spellings of site names.

Awcomin Marsh in Rye, New Hampshire). We sampled fish populations for one, two, or three years within the initial five-year postrestoration period (eight years in the case of Drakes Island) during the early response phase of marsh creation or restoration. The marsh resident *Fundulus heteroclitus* dominated the fish assemblage and comprised 93% of the mean total fish density. Of the 14 marine and estuarine species identified, more than half were marsh residents (8 species), followed by transients (3 species) that spawn in marine waters and use the marshes as larvae or juveniles, and migratory species (3 species) that use the marshes both as juveniles and adults. This assemblage represents 29% of the 48 fishes known to occur in salt-marsh-dominated estuaries in the Gulf of Maine (Lamborghini 1982; Roman 1987; Murphy 1991; Ayvazian et al. 1992; Doering et al. 1995; Lazzari et al. 1996; Cartwright 1997), including most of the marsh resident species in the Gulf of Maine (Table 3). Elsewhere along the Atlantic coast, marsh residents are known to use the vegetated marsh surface to a much greater degree than other groups of fish that occur in salt-marsh estuaries (Kneib 1984; Talbot and Able 1984; Kneib 1987a, 1987b; McIvor and Odum 1988; Murphy 1991; Rozas 1995; Kneib 1997a).

Overall, we found that fish densities, total length, and species numbers were similar and temporally stable among manipulated and reference marshes. The exception to this pattern was for the INMP created marsh, where fish densities were significantly greater in the reference marsh than the created marsh in 1993 but not in 1997. Sampling error is the parsimonious explanation for this result. Alternatively, the high fish density in the reference marsh could have been an effect of natural annual variation rather than sampling error. The development of the created marsh over five growing seasons could have caused the convergence of fish densities between planted and reference marshes from the initial difference during the first growing season. This interpretation assumes that fish density in the created marsh would have been significantly lower in 1997 if it had been planted in 1996 and 1997 rather than in 1992 and 1993. Fish utilization of the other planted marsh site (SUB) was similar to the reference marsh upon its initial sampling in 1997, but this marsh was sampled during the fourth growing season since its creation, allowing the vegetation to increase in density. Our picture of fish utilization of these created marshes over time would be improved if mitigation plans required that sampling occur at least biannually over the first five years of marsh development (year one, year three, and year five). Biannual sampling provides the minimum replication necessary for statistical analysis of annual change.

A nonsignificant trend in the data suggested that fish utilize elevated marshes restored by dug channels to a lesser degree than impounded marshes restored by culverts (Figure 8). Even though these data are merely suggestive, it is useful to discuss some potential differences in the restoration of elevated marshes and culverted marshes. Reduced fish densities may persist after hydrologic restoration of marshes elevated by berms because of the inverse relationship between the period of inundation (over the diurnal tidal cycle) and elevation. Channels dug through a raised area of marsh may improve drainage of freshwater from the marsh, and therefore help control *Phragmites*, but the channels will not always restore the pattern of tidal inundation critical for fish access. Depending on their construction, dug channels could provide significant low-marsh fish habitat along their banks or in the lower-elevation area within berms. The channels at Awcomin Marsh and Sandy Point had shallow, vertical banks that did not support plant growth, and adjacent high marsh (especially at Stuart Farm) experienced considerably shorter periods of tidal inundation than reference marsh sites even though flooding was increased during spring tides. Conversely, the restored marshes at Drakes Island and Stuart Farm had lower elevations than the reference sites due to soil subsidence typical of impounded and drained salt-marsh peats. The period of tidal inundation at both of these restored culvert marshes was considerably longer than at the reference marshes due to the lower elevations of the restored marshes. In addition, the period of tidal inundation at the Drakes Island salt marsh was augmented by slow drainage on the ebb tide through the undersized culvert (Burdick et al. 1997). The relationship between marsh elevation and pattern of inundation should be accounted for in designing fish monitoring programs for marsh restoration projects.

At Stuart Farm, the restored marsh area underwent a process of colonization and succession by salt-marsh plants after saltwater from the tidal restoration killed the freshwater marsh vegetation. At Drakes Island, the entire 16–20-ha restored marsh area was dominated by *Spartina alterniflora*, creating an expanse of contiguous low marsh unusual in New England. For marsh resident fishes, prolonged inundation of emergent vegetation can create optimal conditions for feeding, growth, and survival. The

dimensions and vertical placement of culverts can also influence the movement of fish to and from restored marshes. At Drakes Island, the small diameter and long traverse of the culvert pipe likely had a negative influence on fish movement compared to the shorter and much wider culvert at Stuart Farm. The significantly higher number of species at the restored marsh at Stuart Farm compared to the reference marsh coupled with a trend toward higher fish densities suggested that there was little interference in the movement of fish by the culvert. The restored and reference marshes at Drakes Island had the same mean number of fish species, although fish density trended lower in the restored marsh in spite of the expansive low-marsh habitat. Tidal movement of the fishes may have been reduced by the long, narrow culvert. Physical and behavioral barriers to fish passage should be considered in the design of marsh restoration and creation projects.

The comparison of manipulated marshes with local reference marshes not only provides an internal standard for the monitoring of each restoration project but also makes it possible (1) to follow changes over time while accounting for natural annual variation and (2) to make valid comparisons of the magnitude and direction of these changes between unrelated restoration projects. If the "appropriate" reference is selected (see below for discussion of reference site selection), this approach will provide a more valid test of the functioning of a manipulated site than the use of fixed benchmarks (Brinson and Rheinhardt 1996; Simenstad and Thom 1996) based on the average performance of undisturbed systems within the region, especially over the short term.

How are the results of monitoring affected by the selection of the reference marsh? Although our study was not specifically designed to investigate this question, we attempted to address it by comparing the variation in fish distributions across three natural fringing marshes in a series of tidally connected basins in Portsmouth Harbor. Inner North Mill Pond (INMP, Figure 1D) is the tidal basin at the head of tide, upstream from Inner Inner Cutts Cove (IICC), Inner Cutts Cove (ICC), and Outer Cutts Cove (OCC) (Figure 1E). Because of the general size, micro-topography, and locality of these latter three marshes, each could have been considered as a reference for the INMP or SUB created marshes. There were significant differences in fish density among the reference marshes due to the much higher density in 1993 of fish in INMP (an urban tidal pond) than in ICC or OCC. These two downstream sites were in contiguous basins, one in a tidal backwater flanking a major roadway (ICC), the other adjacent to the main stem of the Piscataqua River (OCC). In addition, the species assemblage from OCC contained two fishes (Atlantic herring and striped bass) not found in any other marsh area. The herring entered the marsh as a large school. These species' presence may have been due to the location of the marsh in a narrow, deepwater cove connected to the Piscataqua River mainstem. The movement of water in Gulf of Maine salt-marsh ecosystems is highly directional. At low tide, nearshore water enters from an inlet and progresses upstream so that the marsh is inundated in a sequential pattern. Fish entering the marsh system with the flooding tide will encounter some marsh sites before others depending on marsh size, drainage density (pattern of channels and creeks), and site location. Reference and manipulated sites should be located to minimize the influence of sequential inundation of the marsh surface on fish distribution and abundance, so as not to confound the influence of restoration-specific differences between the sites.

Our study provides the first density estimates for fishes in vegetated salt-marsh habitat in the Gulf of Maine. Most prior studies of fish distribution and abundance in salt marshes of the Gulf of Maine have relied on measures of relative abundance and have been restricted to sampling within intertidal and subtidal waters (Lamborghini 1982; Roman 1987; Murphy 1991; Doering et al. 1995; Lazzari et al. 1996; Cartwright 1997). Ayvazian et al. (1992) used dimension-adjusted (i.e., accounting for the area sampled) seine and trawl data to estimate fish densities in intertidal channels within a Gulf of Maine salt marsh and determined a mean annual fish density of $0.64/m^{-2}$ for channel samples taken adjacent to vegetated marsh. Mean total fish density in this study ranged from $0.05/m^{-2}$ to $0.67/m^{-2}$. The highest mean densities found at our study sites just overlap with some of the lower mean densities measured for vegetated marsh in Virginia ($1.8/m^{-2}$, Varnell and Havens 1995); Georgia ($0.7/m^{-2}$, Kneib and Wagner 1994); Louisiana ($26.0\ m^{-2}$, Baltz et al. 1993; $5.3\ m^{-2}$, Rozas and Reed 1994); and Texas ($0.3\ m^{-2}$ to $7.3\ m^{-2}$, Minello et al. 1994). The Virginian, Carolinian, and Louisianan tidal wetlands in the preceding studies are low marsh dominated by *Spartina alterniflora*. The greatest fish densities in our study occurred in the reference sites at Inner North Mill Pond ($0.7\ m^{-2}$)

and Sandy Point (0.5/m^{-2}). These marsh areas were fringing *Spartina alterniflora* low marsh, as opposed to the *Spartina patens*-dominated high marsh of the Drakes Island, Stuart Farm, and Awcomin Marsh reference areas. Further, the fish assemblages at the low-marsh-only study sites at INMP, SUB, and OCC did not include any of the four stickleback species (Table 2). The data suggest a difference in fish utilization between low- and high-marsh zones that merits further investigation, given that both of these marsh types are important habitats in the Gulf of Maine. The creation of salt marsh on the Atlantic and Gulf of Mexico coasts has focused primarily on low marsh, but most opportunities for marsh restoration in the Northeast occur in marsh systems with both high and low marsh, where the area of low marsh tends to comprise 10–20% of the total marsh area (Nixon 1982).

In this study, we have documented the occurrence of fishes in restored, created, and reference salt-marsh habitats, but we have not determined how these habitats influence fish survival and growth. Salt-marsh estuaries are physically and hydrologically complex systems with a number of different habitats that can be separated or connected depending on tidal stage. Fish can move with the tides to and from open bay waters, subtidal channels, intertidal creeks, rivulets and puddles, low marsh edge, low marsh interior, high marsh edge, high marsh interior, and salt pannes. Recent studies have focused on understanding how life history and developmental stage interact with tides and seasons to determine occurrence and feeding of fish species in these habitats (Kneib 1984, 1987a, 1987b; McIvor and Odum 1988; Rozas et al. 1988; Deegan 1990; Murphy 1991; Rountree and Able 1992a, 1992b; Baltz et al. 1993; Kneib 1994; Kneib and Wagner 1994; Minello et al. 1994; Smith and Able 1994; Miltner et al. 1995; Szedlmayer and Able 1996; Deegan and Garritt 1997; Kneib 1997b). The emerging view is that small marsh residents feed within the vegetated marsh on the high tides and then overlap with and serve as prey for juvenile transient fishes in the intertidal and subtidal habitats as the tides recede. As the juvenile transients grow and mature they move to the open bay areas of the estuary and coastal ocean, finishing the "trophic relay" (Kneib 1997a) of productivity from the vegetated salt marsh to the open marine ecosystem.

The use of marshes by nekton is often considered a criterion of success for salt-marsh mitigation projects (Matthews and Minello 1994). In the present study, it appears that fish will readily visit restored and created marshes in assemblages similar to those found in reference marshes but are subject to the influence of differences in tidal regime, access to marsh habitat, and possibly density of vegetation. This result is not surprising, given the mobility of fish and the general tendency of many littoral fishes to investigate physical structures (e.g., plant stems).

Visitation is necessary but not sufficient evidence for the value of restored and created marsh as fish habitat. From the results of this study, we cannot determine how fish growth and survival in manipulated marshes compares to that in natural marshes. Studies comparing the benthic invertebrate fauna of manipulated and reference marshes indicate that invertebrate prey abundance and availability for fishes can be significantly reduced in created marshes during the first 10–20 years postcreation (Posey et al. 1997). Reduced abundance of benthic invertebrates may be related to the generally low organic content of created marsh soils (Langis et al. 1991; Moy and Levin 1991; Sacco et al. 1994; Scatolini and Zedler 1996). In at least one study, however, densities and diversity of benthic infauna in a created marsh came to resemble the reference marsh within six months (Levin et al. 1996). Moy and Levin (1991) reported lower fish densities but increased feeding in a created marsh due to the increased abundance of polychaetes, which were more accessible to fish than the oligochaetes that characterized the reference marsh. Comparable studies of marsh benthic infauna have not been carried out for restored marshes, but in a study of fish feeding within intertidal ditches in a tidally restored marsh, fish had reduced feeding success compared to fish in a downstream reference marsh (Allen et al. 1994). From the differing results of these studies, it appears that variation in prey availability is an important factor in determining the value of created and restored salt marshes for fish. Measures of fish feeding or growth should be added to the criteria for determining this value.

It appears from our study that both created and restored marshes can be visited by fish assemblages comparable to those found in reference marshes over the short term (1–5 years), as found at Inner North Mill Pond, Stuart Farm, and Awcomin Marsh. Nonsignificant trends in the data also suggest that fish access to vegetated marsh habitat may be restricted by inadequate culvert dimensions as at Drakes Island or inadequate marsh flooding as at Sandy Point. Additional study is needed to further support this interpretation. After they

gain access to subtidal marsh channels, the ability of fish to utilize the marsh surface is directly related to the period of marsh inundation. Inundation patterns are primarily a function of tidal restriction and marsh elevation. In the large majority of cases, hydrologic restoration of tidally restricted marshes will improve a much greater area of fish habitat per unit cost than creation of new marsh and will not be subject to many of the constraints that limit the function of created marshes (Zedler 1996).

The primary consideration in tidal restoration projects is not necessarily the cost of construction but the social, economic, and political issues that must be addressed. Often, tidally restricted marshes are in highly developed coastal areas where local residents may perceive the restoration of tidal flow as a threat, even when flood hazard studies show that no such threat exists. This has been the case at Drakes Island Marsh, where property owners in the vicinity of the restored marsh still strongly advocate replacement of a culvert flap gate 10 years after tidal flow was restored. In spite of this caution, hundreds of hectares of coastal fish habitat can be improved through a concerted program to restore the hydrology of tidally restricted marshes in the Gulf of Maine (Dionne et al. 1998).

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