

Ditching and Ditch-Plugging in New England Salt Marshes: Effects on Hydrology, Elevation, and Soil Characteristics

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Abstract Anthropogenic activities in New England salt marshes have altered hydrologic flows in various ways, but unintended consequences from some types of habitat modifications have received little attention. Specifically, ditches have existed on salt marshes for decades, but the effects of these hydrologic alterations are only poorly understood. Ditch-plugging is a more recent methodology used for salt marsh habitat enhancement and mosquito control, but the long-term effects from this management practice are also unclear. The interactions involving marsh surface elevation, soil characteristics, and hydrologic regimes result in feedbacks that regulate the salt marsh self-maintenance process, and these interactions vary with hydrologic modification. Using natural tidal creeks and pools as controls, we examined the effects of ditching and plugging, respectively, on hydrology, surface elevations, and soils. Results showed the most apparent effects of altered hydrology from ditching are prolonged pore-water retention in the rooting zone and significantly lower soil bulk density and mineral content when compared with natural creek habitat. From a management perspective, the important question is whether the combined alterations to physical and biological processes will hinder the marsh's

ability to keep pace with increasing rates of sea level rise, especially in more heavily ditched marshes. In contrast, ditch-plugging results in the decoupling of feedback processes that promote salt marsh self-maintenance and in doing so, threatens marsh stability and resilience to climate change. High surface water levels, permanently saturated soils, marsh subsidence, and significantly lower bulk density, carbon storage, soil strength, and redox levels associated with hydrologic alterations from ditch-plugging all support this conclusion.

Keywords Hydrologic alteration · Self-maintenance · Subsidence · Carbon storage · Sea level

Introduction

Salt marshes develop over thousands of years through a dynamic process involving hydrology, vegetation, soil accretion, and marsh surface elevation (Redfield 1972). In response to dynamic environmental conditions (e.g., sea level rise, suspended sediments), salt marshes will continue to evolve but can also maintain their form and function within limits of environmental change. Self-maintenance is a complex process regulated by tidal sediment transport and deposition along with plant growth, soil trapping, and organic matter accumulation which, combined with hydrologic regime, provide a negative feedback mechanism that facilitates marsh stability within certain thresholds (Morris et al. 2002; Burdick and Roman 2012). Hydrologic regimes mediate the development and persistence of biological communities, which in turn influence plant community patterns and salt marsh accretion rates. For example, Goodman et al. (2007) determined the mean rate of accretion in several Maine salt marshes was 2.8 mm/year over a 17-year period, similar to the relative sea level rise of approximately

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2.4 mm/year for the same time period. The salt marsh self-maintenance process breaks down when marsh accretion rates fail to keep pace with the rate of sea level rise (Morris et al. 2002).

Salt marshes have a long history of human use and anthropogenic impacts, resulting in a mosaic of natural and created water features throughout the marshes of New England (Daiber 1986; Silliman et al. 2009). The naturally occurring water features found within our study sites include meandering and often multi-ordered creek channels that conduct tidal waters, along with a variety of pools, both permanent and ephemeral. Created water features characteristic of our study sites include ditches constructed by farmers that were expanded and ramified by the Civilian Conservation Corps (CCC) more than 70 years ago and ditch-plugs constructed more recently over the past decade. Ditch-plugging is a habitat alteration method that increases surface water habitat for larvivoracious fish in an attempt to control mosquitoes, as well as provide wading bird and waterfowl habitat (Meredith et al. 1985; Taylor 1998). In addition, some managers believe that ditching led to a loss of natural pools, so they construct ditch-plugs in an attempt to replace the open water habitat. More detailed characteristics of the natural and created channel and pool habitats found within our study sites are provided in Vincent et al. (2012).

Hydrologic alteration is a common management tool used in salt marsh restoration and enhancement projects, yet the long-term effects of such modifications on marsh processes with regard to sea level rise and habitat persistence receive little attention. Our objective was to characterize the hydrology of natural and created water features and determine the localized effects of altered hydrologic regimes on physical soil parameters in adjacent habitat (i.e., within 20 m). Thus, our study focuses on localized responses of the habitats to altered hydrologic regimes, rather than whole-marsh habitat comparisons made in other studies (Adamowicz and Roman 2002; James-Pirri et al. 2012).

We tested the hypothesis that created hydrologic features alter physical parameters, soil processes, and surface elevations of adjacent salt marsh habitat, disrupting the salt marsh self-maintenance process, and resulting in distinct differences between natural and created habitats. Specifically, we examined how ditching and ditch-plugging in salt marshes affect hydrologic regimes and how the altered hydrologic regimes affect soils in adjacent habitat. Our approach to answering these questions was to use natural creeks and pools as controls in comparison with created ditches and ditch-plugs. Our results provide insights into the effects of ditching and ditch-plugging on hydrologic and soil components of the salt marsh self-maintenance process and in doing so, allow us to infer habitat stability with regard to sea level rise.

Materials and Methods

Study Areas

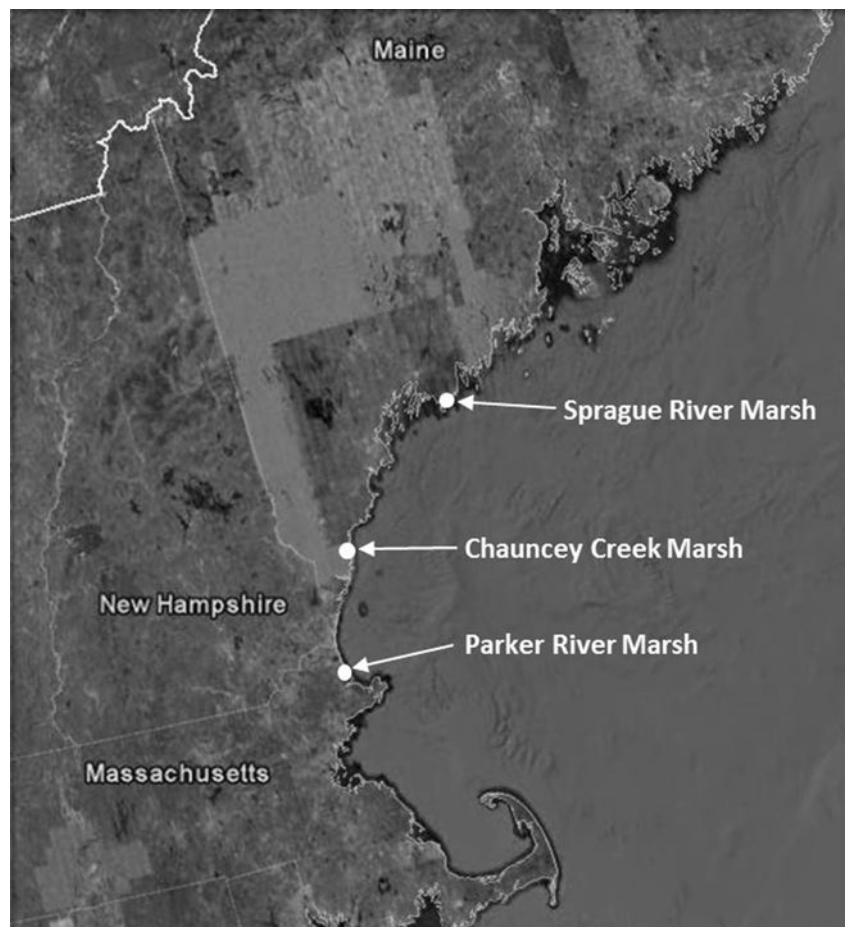
Sampling took place during the summer of 2005 at three back-barrier salt marshes located along the Gulf of Maine coastline from Massachusetts to central Maine (Fig. 1). Habitat in each marsh was similar, consisting of typical New England high marsh dominated by *Spartina patens* mixed with *Distichlis spicata*, *Juncus gerardii*, short-form *Spartina alterniflora*, and a variety of halophytic forbs. Creek banks were dominated by tall-form *S. alterniflora*, and ditch banks were mixed with *Atriplex patula*, *Suaeda* spp., *S. patens*, and *Salicornia europaea*, among others. Natural pool edges were dominated by short-form *S. alterniflora*, *S. patens*, and *D. spicata*, while ditch-plug edges were dominated by clumps of tall-form *S. alterniflora* and *S. europaea*.

Parker River Marsh, located on Plum Island in Newburyport, Massachusetts, is part of the US Fish and Wildlife Service Parker River National Wildlife Refuge (NWR) and is the southern-most marsh in this study (Fig. 2). Parker River Marsh is a 500-ha back-barrier salt marsh (42° 44' 36.06" N; 70° 48' 10.05" W) and receives tidal flow from the Gulf of Maine primarily via Plum Island Sound. Several creek channels meander through the marsh, with smaller tributaries extending into the high marsh. Natural and created pools occur throughout the marsh. Ditches connected to the main channels were dug perpendicular to the upland edge throughout the marsh during the 1930s by the CCC. The three northern-most ditch-plugs that were sampled during our study were constructed in 2002, and the southern-most ditch-plug was constructed in 1995.

Chauncey Creek Marsh in Kittery, Maine, is a 22-ha back-barrier salt marsh that is part of the US Fish and Wildlife Service Rachel Carson NWR (Fig. 3). It is located approximately 40 km north of Parker River Marsh (43° 05' 13.58" N; 70° 39' 52.46" W). Chauncey Creek Marsh receives tidal flow through Chauncey Creek, which is a 2 km-long tributary to the Piscataqua River at Portsmouth Harbor. A meandering creek channel with small tributaries facilitates tidal exchange, and natural and created pools are distributed throughout the marsh. Ditches constructed by the CCC in the 1930s extend from the main channel to the upland edge. All four ditch-plugs sampled during our study were constructed in 2002.

Sprague River Marsh in Phippsburg, Maine, is an 86-ha back barrier salt marsh that formed over time in a narrow glacial valley (Fig. 4). Part of the Bates Morse Mountain Conservation Area (43° 44' 00.63" N; 69° 44' 42.33" W), Sprague River Marsh is jointly managed by Bates College and The Nature Conservancy. Sprague River Marsh is located approximately 100 km north of Chauncey Creek

Fig. 1 Study marsh locations showing a central Gulf of Maine coast distribution (Google Earth 2010)



Marsh and is the northern-most marsh in this study. The marsh receives tidal flow from the Gulf of Maine via the Sprague River Inlet. Although ditching had occurred at Sprague River Marsh for agricultural purposes since the 1700s, the most extensive human impact occurred in 1958 when a local resident dredged a large central channel the length of the marsh and smaller ditches extending perpendicularly from upland borders to the dredged central channel (Sebold 1998; Bohlen 2007). A narrow gravel and dirt causeway built in the 1940s cuts across the northern portion of the marsh, and a bridge over the main channel allows tidal flow upstream of the causeway. In 2002, riprap was removed from below the bridge, improving flow to the northern portion of the marsh; however, the causeway itself is slightly higher in elevation than the surrounding marsh surface and impedes sheet flow over the marsh surface to some degree even during the highest tides. Remnants of the primary creek meander through the marsh with several smaller tributaries extending into the high marsh. Natural pools occur throughout the marsh. The two northern-most ditch-plugs that were sampled during our study were constructed north of the causeway in 2002, and the two ditch-plugs at the southern end of the marsh were constructed in 2000.

Study Design

Data collection at all three marshes targeted habitat adjacent to four types of water features (ditch, ditch-plug, creek, and natural pool). To ensure a broad distribution across the marsh, aerial photos of each marsh were divided into three equal sections (north, middle, and south). A grid was placed over each marsh aerial photo, and each grid intersection was numbered. A random-numbers table was used to choose grid intersections in each of the marsh sections, and water features for each habitat type closest to the chosen grid intersections were used as sample stations for the study. There were no more than two replicates in each section of the marsh, and the number of natural habitats equaled the number of created habitats of similar feature (i.e., pool or channel) in each section.

Each study marsh contained four replicates of each of the four habitat types. Eight transects of 20 m in length were established for each habitat type (two as subsamples per replicate). Transect locations were sometimes limited by the availability of specific habitat types (i.e., ditch-plugs) and at other times by proximity to other water features. Transect locations were not randomly selected but were selected without bias by extending transects perpendicularly

Fig. 2 Parker River Marsh-Newburyport, Massachusetts (MassGIS 2010), *white line* represents the study area boundary



from the edge of each water feature at a location closest to the center of the water feature that allowed the 20 m transect to end at least 10 m away from any other water feature. Although preliminary data suggested that the hydrologic influence of water features extended outward up to 18 m from the edge of the water body, 10 m was the maximum distance from adjacent water features that a 20-m transect could consistently be located throughout each study marsh. To minimize the effect of isolated marsh characteristics, representative habitats were subsampled on both sides of each water feature. The Global Program of Action Coalition for the Gulf of Maine protocol (Neckles et al. 2002) was used as a basis for field data collection. Attributes sampled included ground and surface water levels, marsh surface elevations, and soil characteristics (salinity, redox potential (Eh), bulk density, percent organic content, carbon storage, and soil strength). Salinity and Eh were measured four times during the period when water level recorders were deployed at each replicate. Soil strength and soil cores were collected once from each replicate. All data were recorded in field books or on data sheets and transferred

to electronic databases. Sampling methods were consistent among all study sites.

Hydrology

Hydrology was sampled at each marsh separately. Water levels were sampled along one transect randomly selected at one replicate of each of the four habitat types during a spring/neap tidal cycle. Sampling each habitat type at the same time allowed for comparison of hydrologic conditions during the same tidal cycle and reduced sampling bias due to variable tidal conditions. Three automatic capacitance water-level data loggers (Odyssey Data Recording 2005) were placed along one transect for each habitat type at 2, 7, and 15 m from the edge of the water feature (12 water level loggers). Water level data loggers were enclosed in 5.08-cm diameter PVC pipes with 6.4-mm holes drilled on four sides every 10 cm for the length of the pipe. Capacitance loggers use a Teflon-coated wire that extends the length of the PVC pipe, with a brass counter weight located at the base of the tube below the water

Fig. 3 Chauncey Creek Marsh-Kittery, Maine (MaineGIS 2010), *white line* represents the study area boundary



table. The PVC pipes extended 200 cm above and 100 cm below the marsh surface to record both surface and ground water levels. One additional data logger was placed in the center of the main tidal channel as a downstream tidal reference. Elevations of the water-level loggers were determined using a self-leveling laser level (CST/Berger 2005) tied into

state highway benchmarks, and water levels were recorded relative to NAVD 88. Data loggers were programmed to record water levels every 10 min, and data were uploaded to Excel files. This procedure was repeated by rotating sampling among all four habitat replicates for a total of four transects sampled per habitat type at each study marsh.

Fig. 4 Sprague River Marsh-Phippsburg, Maine (MaineGIS 2010), *white line* represents the study area boundary



Soil Characteristics

Soil cores were collected at 2, 7, and 15 m along eight transects for each habitat type (ditch, ditch-plug, creek, and natural pool) in each marsh. A 3-cm diameter Eijkkelkamp gouge auger was used to remove the first 20 cm of soil at each core location. The gouge auger has an open-face design that allows it to be inserted without compacting soils. Each core was visually examined at the time of collection, and no signs of compaction were observed in any sample. Cores were cut into four sections, each five centimeters in length. Core segments were placed in individual Ziploc bags, stored on ice in coolers, and transported to the lab for processing. Each core was placed in individual tins, weighed, and dried at 60 °C for 3 days to achieve constant weight. Dried cores were re-weighed to determine bulk density as milligrams per cubic centimeter. Cores were then combusted in a muffle oven at 450 °C for 3 h, cooled, and reweighed to determine organic matter calculated as the proportion of dry core weight. Carbon storage was calculated as milligrams carbon per cubic centimeter of soil based on the equation presented in Craft et al. (1991):

$$\text{Organic carbon} = (0.40)\text{LOI} + (0.0025)\text{LOI}^2$$

Soil strength was measured using a Field Scout SC-900 soil compaction meter (Spectrum Technologies, Inc. 2005). Soil compaction readings were collected at 0, 2, 7, 15, and 20 m along all eight transects per habitat at each marsh. The soil compaction meter measured soil strength in kilograms per square centimeter at 1-cm intervals down to a depth of 50 cm below the soil surface.

Soil salinity was measured in parts per thousand (ppt) in each habitat type using a handheld temperature-corrected refractometer. Salinity wells were made of 1.27 cm diameter PVC pipe with 4 mm holes drilled on four sides and spaced every 2.5 cm along the length of the below ground portion of the well. The bottom of each well was sealed, and the top of the wells had removable elbows for ventilation that prevented entry of surface water. Salinity wells were inserted 15 cm into the marsh soils at 2, 7, and 15 m along four transects for each habitat type (one transect per replicate). The wells were placed 25 cm away from each water-level data logger. A syringe with plastic tubing was used to extract all water from the well. When water had re-filled the well, a water sample was collected and the salinity was measured.

Soil redox potential was measured using a handheld redox meter, double junction reference, and platinum-tip electrodes (Faulkner et al. 1989) inserted 10 cm into the soil at 2, 7, and 15 m along four transects for each habitat type (one per replicate). Soil redox readings were collected at the

same locations used for hydrology and soil salinity data collection. Eh was calculated by adding +244 mV to each value to account for the potential of the reference electrode (Burdick et al. 1989).

Marsh surface elevations were measured using a self-leveling laser level (CST/Berger 2005) tied into state highway benchmarks. Marsh elevations were surveyed at 0, 2, 7, 15, and 20 m along each transect. A nylon meter tape was stretched out on the ground between wooden stakes marking the ends of each transect. A sighting rod was placed on the tape at each survey point. Placing the sighting rod on the tape prevented the stick from sinking into the soil and helped to ensure the survey reading was representative of the marsh surface.

Statistical Analysis

Data were compiled into Microsoft Excel® spreadsheets and statistically analyzed using JMP statistical software (SAS Institute, Inc. 2010). The alpha level was set at 0.05 to control for type I error during statistical analyses. Pearson *r* correlation was used to assess relationships among soil characteristics and water levels. Each dependent metric (i.e., water level, soil characteristics, or marsh surface elevation) was analyzed separately using a two-way fixed-effects ANOVA model blocked by marsh, with water feature (pool versus channel) and habitat origin (natural versus created), as independent variables, along with the interaction of water feature by origin. Data are presented as means±Standard Error (SE). All parametric analyses included Tukey-Kramer HSD post hoc means comparisons.

Data were examined to ensure they satisfied the assumptions of the general linear model (normal distributions, no extreme outliers, and evenness of variance). Where necessary, data transformations were performed to meet these assumptions. Water level and soil compaction data were log(*x*+1)-transformed, and bulk density data were square-root-transformed. Transformations of marsh surface elevation data did not satisfy the assumptions of normality, so the Kruskal–Wallis non-parametric test was used to compare means for this metric.

Discriminant function analysis (DFA) was used to see if data could predict habitat type (creek, ditch, natural pool, and ditch-plug) based on marsh physical characteristics (bulk density, carbon storage, percent organic matter, soil strength, salinity, redox potential, and water level). The number of discriminant functions used in a model is based on either (a) *g*−1, where *g* is the number of categories in the grouping variable (habitat, dependent variable), or (b) the number of discriminating (independent) variables, *p*, whichever is less. This study used *g*−1, which resulted in three discriminant functions included in analysis of the means of each of the 48 transect means.

Results

A comparison of the three study marshes using the two-way ANOVAs revealed differences in water level and soil strength only. Lower water levels were found for channel habitats at Chauncey Creek marsh relative to the levels recorded at Parker River and Sprague River marshes, and an inverse relationship was found between soil strength and marsh latitude. However, hydrologic, soil characteristics, and pore water patterns were consistent for each habitat type among all three marshes.

Hydrology

Water levels relative to the soil surface were significantly different between created and natural (drier) and between pool and channel (drier) habitats based on the two-way ANOVA (Table 1). Although the interaction between the two main effects was not significant, a Tukey–Kramer post hoc test showed significantly higher water levels in ditch-plug habitat than all other habitat types. In addition, water level in habitat adjacent to natural pools was significantly higher than in habitat adjacent to creeks and ditches (Table 2).

In support of these water level differences, tidal stage duration curves are presented for each habitat type in Fig. 5. Water levels were observed to be higher for a greater percentage of time in habitat adjacent to pools than channels and higher for longer periods of time in created than natural habitats. Water-level recorders also showed tidal ranges were greater and drainage was more extensive in natural habitats compared with created habitats (Table 3). Water level was positively correlated with soil salinity and negatively correlated with redox (Table 4).

Marsh Surface Elevations

Comparisons of marsh surface elevations by habitat feature and type followed what might be expected from the water

level results but with some exceptions. Marsh surface elevations were significantly lower in ditch-plug habitat than in all other habitats (Table 2), whereas surface elevations were highest for natural pool habitat even though water levels were greater than creeks and ditches. Notably, habitat directly adjacent to ditches was significantly greater in elevation than adjacent to ditch plugs.

Since marsh surface elevations were measured at five distances along each transect, we can examine the trends with distance for the four habitat types. Elevations in natural pool habitat remained fairly consistent for the 20 m sampled and were consistently higher than the other three habitats (Fig. 6). Elevations in creek habitat trended lower than ditch habitat from 0 to 2 m, but elevations were similar for the two habitats from 7 to 20 m distance. Elevations in the two channel habitats trended higher with distance and converged towards natural pool elevations at around 15–20 m. Marsh surface elevations in ditch-plug-habitat trended slightly higher with distance but were considerably lower than elevations in natural pool habitat from 0 to 20 m and were lower than channel habitats from 7 to 20 m, indicating subsidence over time in ditch-plug habitat.

Soil Characteristics

Soil salinity was different among habitats, and the interaction of marsh feature and habitat origin was significant (Table 1). The Tukey–Kramer post hoc test showed significantly higher salinity levels in ditch-plug habitat relative to all other habitat types (Table 2). There was no significant difference in soil salinity for habitat adjacent to natural pools, creeks, or ditches. Soil salinity was positively correlated with water level and negatively correlated with bulk density, redox potential, and carbon storage (Table 4).

Soil redox potential was different among habitats, and the interaction of marsh feature and habitat origin was significant (Table 1). The Tukey–Kramer post hoc test showed significantly lower redox potential in ditch-plug habitat than

Table 1 Two-way ANOVA results for hydrology and soil characteristics (DF = 5, 6; $n=12$)

Metric	Feature (channel vs. pool)		Origin (created vs. natural)		Interaction (feature by origin)	
	<i>F</i>	<i>p</i> value	<i>F</i>	<i>p</i> value	<i>F</i>	<i>p</i> value
Water level	52.56	<0.0001	12.99	0.0020	1.56	0.2101
Salinity	41.70	<0.0001	21.71	<0.0001	22.73	<0.0001
Redox	19.74	<0.0001	30.48	<0.0001	12.18	0.0011
Soil strength	1.12	0.2967	19.51	<0.0001	26.92	<0.0001
Bulk density	11.70	0.0014	6.58	0.0141	3.34	0.0751
Carbon storage	2.53	0.1192	20.54	<0.0001	13.76	0.0006
Percent organic matter	6.44	0.0151	0.16	0.6938	11.82	0.0014

Bold values indicate significant effects; feature=channel or pool; type =natural or created

Table 2 Marsh physical characteristics by habitat (values are means from paired 20 m transects of four habitat replicates from three marshes, pooled together; $n=12$)

Habitat	Water level (cm)	Marsh elevation NAVD88 (cm)	Soil salinity (ppt)	Soil redox potential Eh (mV)	Soil strength (kg/cm ²)	Bulk density (mg/cm ³)	Carbon storage (mg/cm ³)	Organic content (%)
Created ditch	-2.52 c	110.98 b	26.17 b	7.85 a	4.51 b	605.01 b	34.17 a	18.97 a
Natural creek	-4.01 c	105.24 bc	26.21 b	52.18 a	4.21 bc	761.76 a	34.08 a	14.67 b
Ditch plug	5.51 a	102.57 c	30.96 a	-165.69 b	3.57 c	550.79 b	28.66 b	17.98 ab
Natural pool	-0.49 b	121.04 a	26.94 b	31.29 a	5.87 a	575.11 b	37.43 a	21.58 a

Comparisons with the same letters are not significantly different according to Tukey–Kramer post hoc tests for each variable ($p<0.05$)

all other habitat types (Table 2). Differences in redox potential for habitat adjacent to natural pools, creeks, or ditches were not significant. Redox potential was positively correlated with soil strength, bulk density, and carbon storage and negatively correlated with water level and salinity (Table 4).

Soil strength was different among the habitats, and the interaction of marsh feature and habitat origin was significant (Table 1). The Tukey–Kramer post hoc test showed significantly higher soil strength in natural pool habitat than all other habitat types (Table 2). Soil strength was lowest in ditch-plug habitat but not significantly different from creek habitat. In each habitat, an increase in soil strength was observed at 3 cm below the marsh surface, reflecting the dense root mass that typically occurs (Fig. 7). After 10 cm depth, soil strength increased consistently with depth and exhibited strikingly similar patterns, regardless of habitat type. Soil strength in natural pool habitat was consistently higher than all other habitats, followed by ditches. Soil strength was positively correlated with redox potential and carbon storage (Table 4).

Bulk density was significantly different between created and natural (higher bulk density) and between pool and channel (higher bulk density) habitats based on the two-way ANOVA (Table 1). The interaction of the main effects

was not significant; however, the Tukey–Kramer post hoc test showed significantly higher bulk density in creek habitat than all other habitat types (Table 2). Bulk density was positively correlated with redox potential and negatively correlated with percent organic matter and salinity (Table 4).

Carbon storage was different among habitats, and the interaction of marsh feature and habitat origin was significant (Table 1). The Tukey–Kramer post hoc test showed significantly lower carbon storage in soils adjacent to ditch-plugs than all other habitat types (Table 2). Carbon storage was higher in soils adjacent to natural pools but was not significantly different from soils adjacent to ditches or creeks. Carbon storage was positively correlated with soil strength and redox potential and negatively correlated with salinity (Table 4).

Percent organic matter was different among habitats, and the interaction of marsh feature and habitat origin was significant (Table 1). The Tukey–Kramer post hoc test showed percent organic matter was significantly lower in soils adjacent to creeks but was not different from that of ditch-plugs (Table 2). Percent organic matter was negatively correlated with bulk density (Table 4).

Discriminant Function Analysis

Our rationale for using discriminant function analysis was to determine if marsh physical characteristics could predict habitat type. DFA identified significant differences in group membership for soil characteristics and hydrology among the four habitat types (Wilks' Lambda=0.14; $p<0.0001$), with 69 % of the transect memberships correctly classified into each of the four habitats (Fig. 8). Most misclassifications occurred between creeks and ditches.

The first discriminant function axis (Canonical 1) captured 77 % of the explained variation in soil and hydrology characteristics among the four habitats and represented edaphic stress associated with the degree of flooding. The highest positive correlation for Canonical 1 was redox potential, and the most negative correlation was water level (Table 5). The lower redox potential and higher water level identified in Canonical 1 for ditch-plugs identified ditch-plug habitat as distinctly more waterlogged than all others.

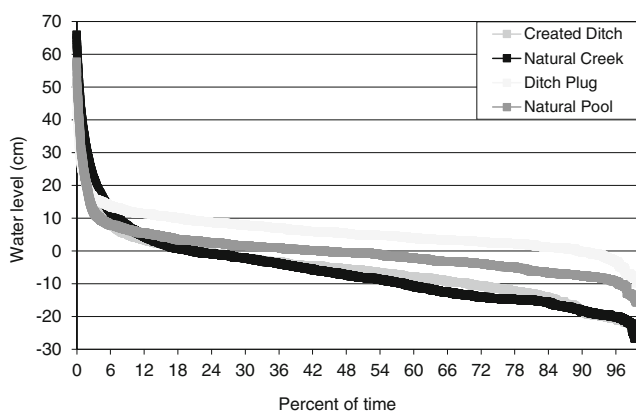


Fig. 5 Tidal stage duration curves showing the extent and duration of surface and ground water inundation by habitat. The marsh surface is at zero centimeters (values are means from 20 m transects of four habitat replicates from three marshes, pooled together; $n=12$)

Table 3 Water levels by habitat (values are means from 20 m transects of four habitat replicates from three marshes, pooled together; $n=12$)

Habitat	Range (cm)	Mean (cm)	% of time \geq marsh surface	% of time ≤ -5 cm ^a	% of time ≤ -10 cm ^a	% of time ≤ -15 cm
Ditch	83.00	-2.52	23.69	53.05	29.49	14.43
Creek	93.00	-4.01	22.19	58.32	40.53	16.59
Ditch-plug	63.00	5.51	91.03	2.17	0.07	0.00
Natural pool	74.00	-0.49	46.74	20.21	2.36	0.10

^a Root zone 0–10 cm below marsh surface

The second discriminant function (Canonical 2) captured 22 % of the explained variation in dependent variables and represented soil characteristics associated with soil bulk properties. Canonical 2 was positively correlated with soil strength and carbon storage, and most negatively correlated with bulk density. Canonical 2 identified the relationship between soil strength and carbon storage and distinguished natural pool habitat as different from all others in terms of these variables (Fig. 8). Also, Canonical 2 identified higher bulk density as a distinguishing factor for creek habitat. Canonical 3 contributed very little to the analysis (Table 5); calling out Sprague River marsh which exhibited lower salinity.

Discussion

Our study showed that hydrologic regimes and self-maintenance in created habitats differed from natural habitats, but differences were more distinct for ditch-plugs and less so for ditches. Higher tidal range has been associated with salt marsh building processes and the alleviation of stressful edaphic conditions (Chmura et al. 2001) and is directly related to marsh surfaced elevation (this study). In South Carolina marshes, Morris et al. (2002) determined that flooding had positive effects on primary production and marsh accretion, yet flooding beyond optimal levels crossed a threshold and negatively impacted plants and marsh accretion processes. Marsh accretion ceases when

peat building processes (i.e., mineral deposition and below-ground organic production) are interrupted (Morris et al. 2002). Although hydrologic alterations of high marsh habitat were the focus of ditches and ditch-plugs, our study shows that only in ditch-plug habitat were altered tidal ranges and flooding thresholds severe enough to have dramatic impacts on the surrounding habitat. Taken together with work by Chmura et al. (2001) and Morris et al. (2002), we expect that habitat surrounding ditch-plugs will further degrade over time.

Our results do not support previous studies that suggested ditches decreased soil pore-water levels (Bourn and Cottam 1950; Lesser 1982; Adamowicz and Roman 2005), but rather found that ditching did not result in lower water levels relative to the surface. Since ditches were originally created in high marsh habitat, presumably at the same elevation as high marsh distant from water features, the lower marsh surface elevations of ditch habitat from 0 to 7 m (Fig. 6) indicate some subsidence (relative to natural high marsh habitat) since the time of ditch construction. Marsh soils likely became better drained when ditches were originally dug (Anisfeld 2012); but over time with decomposition and subsidence, elevations have decreased, and waterlogging has increased. We observed that ditched marshes relaxed to new equilibriums following the initial disturbance, yet maintained accretion rates that kept pace with sea level for the past several decades. However, it remains to be seen if increasing rates of sea level rise will out-pace accretion and have negative effects on the self-maintenance process in

Table 4 Pearson's r correlation results for hydrology and soil characteristics

	Bulk density	Percent organic	Soil strength	Salinity	Redox potential	Water level
Carbon storage	0.28 (0.0568)	0.24 (0.0934)	0.47 (0.0007)	-0.32 (0.0288)	0.37 (0.0094)	-0.18 (0.2266)
Bulk density		-0.82 (<0.0001)	-0.01 (0.9479)	-0.30 (0.0413)	0.29 (0.0432)	-0.23 (0.1220)
Percent organic			0.28 (0.0586)	0.08 (0.6053)	-0.14 (0.3312)	0.08 (0.5928)
Soil strength				-0.22 (0.1359)	0.32 (0.0264)	-0.04 (0.7816)
Salinity					-0.30 (0.0404)	0.33 (0.0226)
Redox potential						-0.35 (0.0155)

Values are based on means from paired 20 m transects of four habitat replicates from three marshes, pooled together and presented as Pearson r (p value); $n=12$; bold values are significant

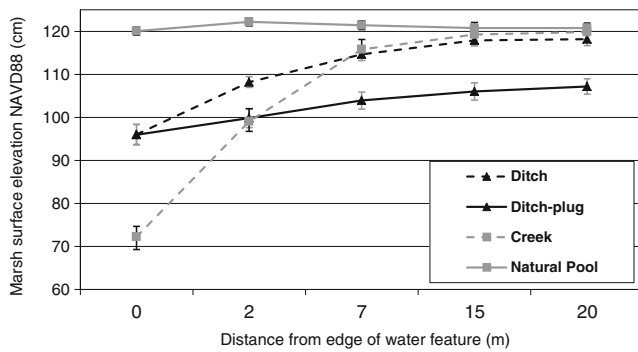


Fig. 6 Marsh surface elevations by habitat (values are means from paired 20 m transects of four habitat replicates from three marshes, pooled together \pm SE; $n=12$)

ditched marshes. In addition, where marshes are intensively ditched (every 10 m) subsidence may be more noticeable and severe, as indicated by LeMay (2007) and illustrated by the darker, wetter areas surrounding ditches in Seabrook, NH (Fig. 9). These marshes may be more vulnerable to the effects of increasing sea-level rise rates on self-maintenance.

Combined organic and mineral inputs to soils are necessary for sustaining marsh accretion. Stumpf (1983) studied sedimentation associated with creeks in Delaware marshes and concluded that periodic storm events contribute more to marsh accretion than sediment delivered by creeks during daily tidal cycles. However, creeks in Stumpf's study had levees of several centimeters in elevation that trapped coarse materials. Our study showed that prominent levees are not typical of creek and ditch banks in back-barrier marshes of northern New England. As Stumpf, Chmura and Hung (2004) determined that in marshes of eastern Canada, vertical accretion in the middle and high marsh zones decreased with distance from creeks due to less frequent flooding, and marsh accretion was greater in areas associated with channel flow. In addition, LeMay (2007) found that soil transport and deposition from tidal flow associated with ditches was similar to that of creeks. Our study adds to findings by

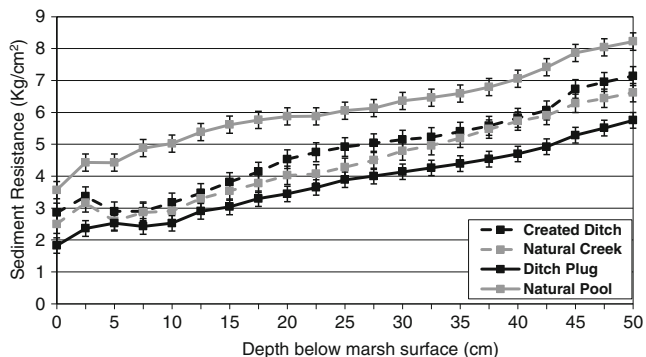


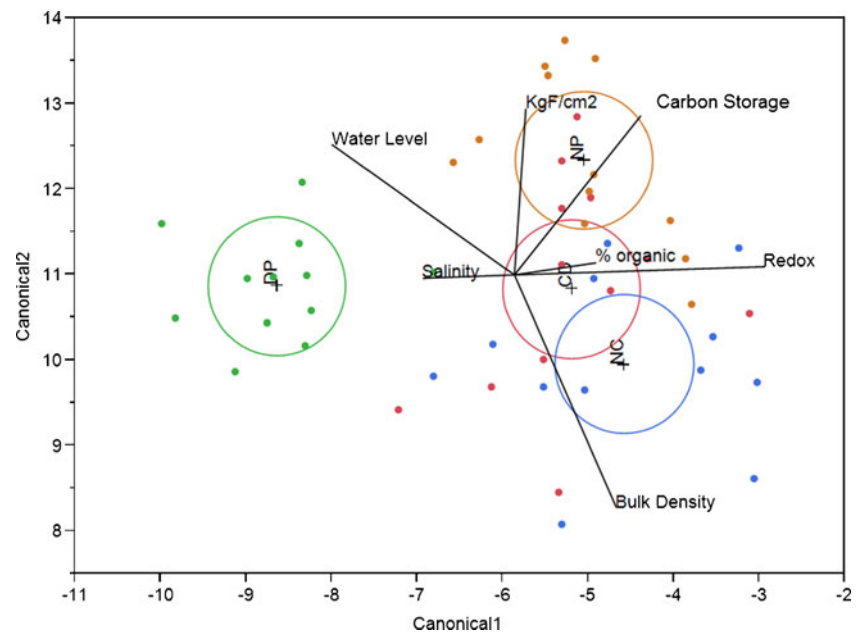
Fig. 7 Soil strength with depth below marsh surface by habitat (values are means from paired 20 m transects of four habitat replicates from three marshes, pooled together \pm SE; $n=12$)

Chmura and Hung (2004) and LeMay (2007) by confirming that hydrologic regimes of ditches that extend into the marsh interior have maintained processes associated with marsh development and self-maintenance. Although subsidence appears to have occurred within 7 m of ditches, elevations are similar to creek habitat 7 to 20 m out, leading us to infer habitat adjacent to ditches has been supported by tidal hydrology following drainage and subsidence, and kept pace with marsh building processes similar to creek habitat for the past several decades.

In contrast, creation of ditch-plug habitat does not support processes that maintain marsh surface elevations and self-maintenance but rather facilitates processes that lead to subsidence, as evidenced by the consistently lower elevations at all replicates. This argument is supported by the difference in mean elevation (10 cm) between ditch-plug and ditch habitats relative to the mean accretion rate (2.8 mm/year) determined by Goodman et al. (2007). At this rate, ditches have the potential to accrete 1.4 cm of elevation over the past 5 years since the ditch-plugs were created, which would leave 8.6 cm accounted for by subsidence. Low elevations are not likely to be an artifact of site selection since ditch-plug locations are unbiased with respect to marsh surface elevation (Meredith et al. 1985; Hruby and Montgomery 1988; Northeastern Massachusetts Mosquito Control and Wetlands Management District 2011). Since the other habitats could not have accreted ≥ 10 cm during the time since the ditch-plugs were created, we attribute the lower ditch-plug elevations to subsidence. Thus, altering hydrologic regimes by plugging ditches inhibits the salt marsh self-maintenance process and leads to increased open water and loss of salt marsh habitat quite a distance beyond the created ditch-plug pool itself (Fig. 10).

Stressors such as salinity and sulfides that accumulate in response to varying hydrologic regimes can influence organic matter contributions to marsh soils and accretion and thus alter the self-maintenance process within different habitats. Working in New England, southern US Atlantic, and Gulf Coast marshes, Bertness and Pennings (2000) observed reduced salt accumulation in association with frequent tidal flushing. The less frequent flushing, greater retention of saline water, and evaporation contributed to salinity concentrations in ditch-plug habitat. Bertness and Ellison (1987) concluded that soil salinity in New England salt marshes reached its highest levels at unvegetated mid-marsh locations due to high surface temperatures and evaporation rates. During hot dry summer periods that coincide with neap tides, we found that large areas of unvegetated soil concentrate salinity in ditch-plug habitat (Fig. 11). Mendelssohn and McKee (1988) conducted plant transplant experiments in Louisiana marshes and observed low redox potential and high sulfide levels in association with waterlogged soils at lower elevations of the marsh platform. Burdick et al.

Fig. 8 Discriminant function analysis comparing habitat type with soil characteristics (values are means from paired 20 m transects of four habitat replicates from three marshes, pooled together; $n=12$; plus sign mean; circles 95 % mean confidence; CD Ditch; DP Ditch-Plug, NC Creek, and NP Natural Pool)



(1989), Reed and Cahoon (1992), and Howarth and Teal (1979) came to similar conclusions with their work in marshes of Louisiana and Massachusetts. Mendelsohn and McKee (1988) noted that such conditions lead to positive feedbacks that expand the radius of waterlogged degraded soils and gradually increase the footprint of open water habitat. This “dieback” process was evident in ditch-plug habitat for each replicate in our study, resulting in a loss of carbon input that can inhibit marsh accretion over time.

Soil structure characteristics were the primary variables that distinguished natural pool habitat from the others, suggesting a dynamic equilibrium in the feedback between elevation and hydrology exists, creating habitat stability and maintenance of marsh building processes. In ditch-plug habitat, the influence of water level over soil characteristics was most apparent, which clearly distinguished ditch-plugs from all others and indicated that the level of edaphic stress and habitat degradation observed in ditch-plug habitat was unique (Fig. 8). The higher carbon storage we found in natural pool habitat contributed to the significantly higher soil strength and marsh stability compared with all other

habitats and lower carbon storage in soils adjacent to ditch-plugs created the opposite effect. Turner (2010) working in Louisiana determined that organic content (root and rhizome biomass) contributes greatly to soil strength. Furthermore, the lower soil strength associated with ditch-plug habitat indicates greater susceptibility to erosion and a higher degree of habitat instability compared with the other habitats. Differences in sediment strength among habitat types did not vary with depth. Soils at 50 cm are approximately 400 years old (Redfield and Rubin 1962), and the difference in soil strength with depth between creeks and natural pools represents the effects of channel and pool hydrologic regimes on natural habitat over long periods time. Ditches and ditch-plugs were presumably created in high marsh habitat with elevations similar to natural pools. This would account for the similarity in patterns for soil strength with depth compared to natural habitats, since the effects of hydrologic regimes on soil processes would have been similar prior to ditch and ditch-plug creation. Therefore, the lower overall soil strength with depth in created habitats compared with natural pools can be attributed to the short-term effects of altered hydrology on soil processes since the

Table 5 Discriminant function standardized scoring coefficients used to indicate the relative importance of each independent variable for each discriminant function

Canonical function	Eigenvalue	Canonical correlation	Percent organic	Carbon storage	Bulk density	Soil strength	Salinity	Redox	Water level
Canon1	2.85	0.86	0.24	0.37	0.3	0.03	-0.27	0.73	-0.53
Canon2	0.8	0.67	0.04	0.47	-0.68	0.49	-0.01	0.02	0.38
Canon3	0.03	0.18	-0.34	0.15	0.24	0.31	0.73	0.17	0.18

Bold values identify independent variables that contribute the most to a given canonical. Eigenvalues are based on the three canonical functions. Canonical correlations are a measure of the association between groups formed by independent variables and a particular discriminant function



Fig. 9 Extensive grid and parallel ditches that were dug in the 1930s by the Civilian Conservation Corps in Seabrook, NH; darker and wetter areas are seen surrounding ditches in the *center* and *right* portions of the photo (Google Earth, photo taken in 2010)

creation of ditches (>70 years) and ditch-plugs (5–10 years). Furthermore, ditch-plugs are excavated out of previously existing ditches; consequently, significantly lower soil strength in ditch-plug habitat compared with ditch habitat indicates habitat degradation over time in response to the altered hydrologic regime. In addition, the spike in soil strength at 3 cm in each habitat is due to the peak in root density present at this depth (Vincent et al. 2012), indicating soil aeration in the top several centimeters is important for maintaining belowground biomass production.

Soil strength counteracts erosive forces and is thus a determining factor for channel formation and marsh stability. The relative amount of organic to mineral content (i.e., percent organic content) in soils of ditch habitat was higher than creek habitat, which contributed to higher soil strength in ditch habitat. Computer models developed by Fagherazzi and Sun (2004) showed how channels with greater soil strength were more resistant to erosion and, therefore, maintained more linear and less branching channel development. Hughes et al. (2009) analyzed creek development in North Carolina marshes and determined that tidal range, soil strength, and plant structure on slumping banks contributed to branching channel morphology. Therefore, the higher soil strength, lack of slumping banks, and smaller tidal range relative to creeks are contributing factors in maintaining the vertical channel walls and linear character of ditches over the past several decades.

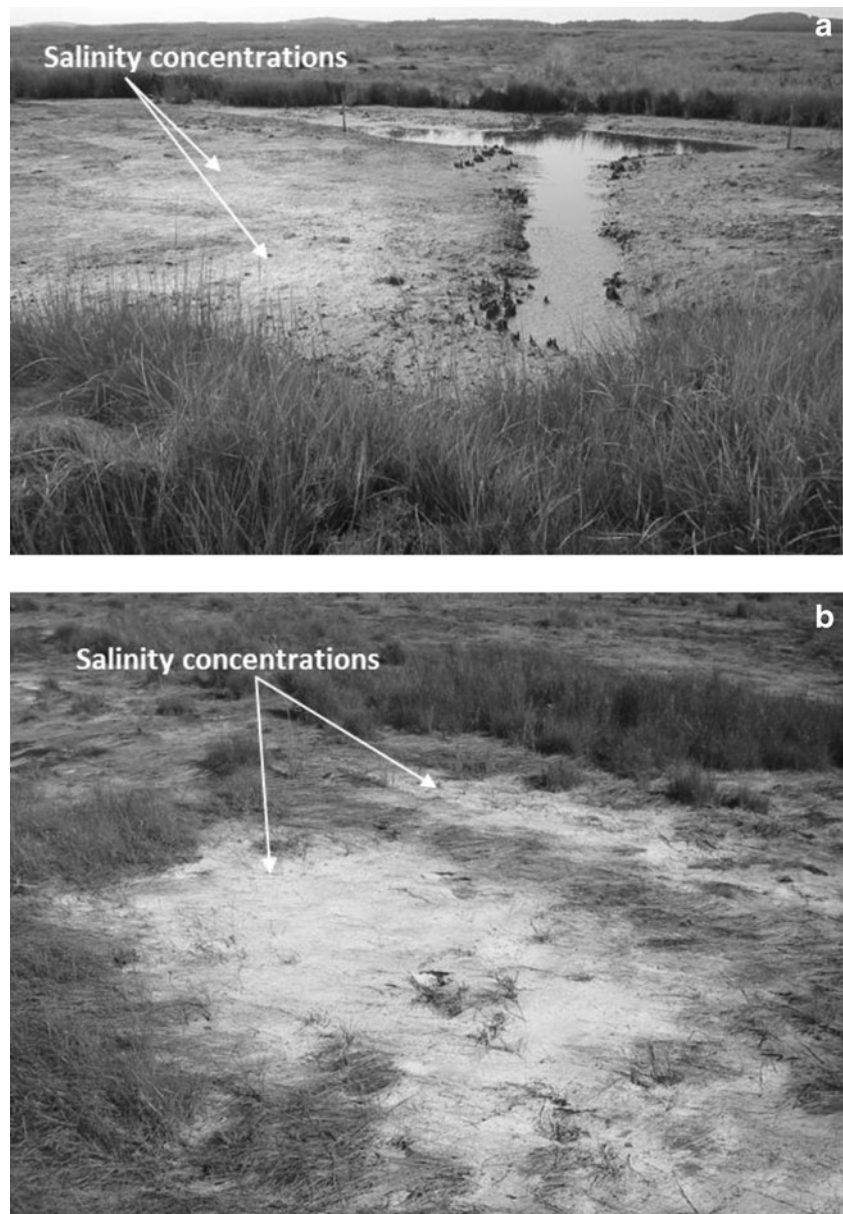
Salt marshes provide large carbon pools that trap atmospheric CO₂, providing an important buffer against the effects of climate change (Kirwan and Mudd 2012). Thus, the influence of habitat type on carbon storage (milligrams per cubic centimeter) and self-maintenance plays an important role in greenhouse gas sequestration and offsetting the



Fig. 10 Surface water impoundment and plant dieback in ditch-plug habitat at **a** Parker River, **b** Chauncey Creek, and **c** Sprague River marshes

effects of climate forcing. The effects of hydrologic regimes on carbon storage were equivalent in the two channel habitats, and differences in bulk density were attributed to

Fig. 11 Soils and plant dieback areas in ditch-plug habitat of **a** Parker River Marsh and **b** Sprague River Marsh that were exposed to increased temperatures, evaporation, and concentrated salinity levels during hot dry summer neap tide periods. *Arrows* are pointing to salt deposits



higher mineral accretion rates in creek habitat. These results suggest that channel habitats have positive effects on greenhouse gas sequestration. Our study shows that natural pool habitat acts as a carbon sink in the high marsh, which we attribute to less extensive pore-water drainage and lower organic matter decomposition under anaerobic conditions compared with the channel habitats. Turner et al. (2000) analyzed soil cores from marshes in Louisiana, Texas, and Rhode Island. They stressed the importance of below-ground organic content to marsh building processes and concluded that plant biomass facilitates mineral accretion in salt marsh habitats. The authors noted an inverse relationship between marsh building processes and hydrologic alterations that result in increased flood duration and periodic drying that leads to reduction in soil organic matter.

Turner et al. (2000) also concluded that bulk density was inversely related to organic content, and the loss of soil organic content through decomposition is usually permanent in hydrologically restricted salt marsh habitat. Our results were similar. We showed that ditch-plugs led to extended periods of surface water impoundment and brief seasonal dry periods, conditions that stressed plants and reduced biomass (Figs. 10 and 11) and led to organic decomposition in soils and subsidence (Fig. 6). The two pool habitats showed similar bulk density, but ditch-plug habitat had lower carbon storage and lower percent organic content compared with natural pool habitat. These results indicate that the higher concentration of mineral material in ditch-plug soils is due to a higher rate of organic decomposition in ditch-plug habitat, rather than increased mineral accretion.

Thus, using only percent organic content to compare levels of organic matter and mineral accretion in soils of different habitats can be misleading. Our study suggests that ditch-plugs lead to carbon loss and the reduction of greenhouse gas sequestration.

Our study showed that tidal regimes and their effects on the self-maintenance process were unique for each habitat. In natural creek habitat, marsh elevations and tidal flow promote flushing and sediment delivery. The sloping banks with tall *S. alterniflora* promote sediment trapping and accretion, and high vegetation productivity and sedimentation have allowed marsh accretion to keep pace with sea level. These conditions support the self-maintenance process in natural creek habitat. In ditched marshes, habitat initially drained by ditches has subsided over time, reducing the depth to water table and creating wetter conditions than in natural creek habitat. The elevations and hydrologic regime supports vegetation growth, but water retention in the rooting zone alters community composition (Vincent et al. 2012), sediment trapping, and soil organic content. Self-maintenance appears to have been functioning for the past several decades in ditch habitat, but questions remain: Will accretion keep pace with sea level rise and will further subsidence occur and limit self-maintenance? In natural pool habitat, moderate drainage supports vegetation growth and reduces decomposition. High carbon content promotes sediment strength and habitat stability, providing a carbon sink in the high marsh. These carbon inputs combined with moderate sedimentation promote marsh accretion that maintains consistently high surface elevations. The self-maintenance process seems to be functioning well in natural pool habitat. In ditch-plug habitat, poor drainage results in surface and pore-water retention that promotes edaphic stress and vegetation die back. Low organic and mineral inputs, low sediment strength, decomposition, and collapse of the root zone have led to habitat instability and marsh subsidence with lower elevations. The hydrologic restriction from ditch-plugging leads to a decoupling of the self-maintenance process with a transition from vegetated to open water areas and the loss of salt marsh habitat.

Conclusion

Environmental gradients that occur throughout New England salt marshes result from feedbacks involving hydrology, elevation, and soil characteristics, which together with plants regulate the salt marsh self-maintenance process (Harshberger 1911; Chapman 1960; Redfield 1972; Bertness and Ellison 1987; Ewing et al. 1997; Hacker and Bertness 1995; Turner et al. 2000; Morris et al. 2002). The complex gradients influence localized conditions, resources, and biological communities. We hypothesized that created hydrologic features alter

physical parameters, soil processes, and surface elevations of adjacent salt marsh habitat by altering the salt marsh self-maintenance feedback process and would result in distinct differences between natural and created habitats. The hypothesis was accepted for pool habitat comparisons but rejected for channel habitats.

Interrupting tidal flow and increasing surface water impoundment diminishes the ability of the marsh to keep pace with sea level rise and contributes to salt marsh habitat loss over time (Mendelssohn and McKee 1988; DeLaune et al. 1994; Cahoon and Reed 1995; Turner et al. 2000). Our results show that the long-term effects of moderate ditching (>30 m between ditches) on hydrology, soils, and marsh surface elevations appeared minor, and soil conditions reflecting salt marsh building processes in ditch habitat are comparable to those observed in creek habitat. However, the effects of subsidence and pore-water retention may have greater impacts in marshes with more extensive grid ditching, which warrants further study. In contrast, ditch-plugging showed significant effects on the salt marsh self-maintenance process that exceeded thresholds and led to a slow expansion of habitat loss surrounding the created pools. We believe ditch-plugging will lead to additional habitat loss with increased rates of sea level rise.

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