



# Mitigating the Legacy Effects of Ditching in a New England Salt Marsh

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## Abstract

The legacy effects of mosquito ditching have made salt marshes more vulnerable to flooding impacts from climate change, presenting management challenges in New England where the majority of salt marshes have been ditched and greater rates of sea level rise and storm events are being observed. One legacy effect of mosquito ditching appears to be subsidence of the marsh, with greater effects near the ditches and extending into the marsh plain. We found an average of 9 cm subsidence midway between ditches that averaged 14 m apart. Ditch Remediation is a new approach to filling ditches that uses existing hydrology and vegetation to mend ditches from the bottom up to restore marsh plain elevations. Smaller, auxiliary ditches are selected for treatment. Hay is mown, allowed to dry, and rolled into the treatment ditch where it is held using twine. Nine ditches in four areas were treated in fall 2014 and 2015. Depth of treated ditches decreased an average of 18 cm by fall 2017, and *Spartina alterniflora* colonized the ditch centers (plant cover and stem density increased). It is unknown whether the trajectory of filling and revegetation of the ditches will continue on their own or if the reduction in drainage depth will stimulate marsh plain building. Ditch remediation could remove excessive drainage effects of half the ditches, thereby approaching an optimal drainage density that may allow tidal marshes to rebuild elevation. This simple and inexpensive technique to reduce ditch impacts should be considered by partnerships that include mosquito control agencies.

**Keywords** Ditch remediation · Subsidence · Cordgrass colonization · Resilience

## Introduction

Salt marshes in New England were prized for their productive perennial grasses by colonists arriving from Western Europe. In the past two centuries, declining agricultural need and

development of cities, ports, and transportation led to filling and dredging of marshes resulting in losses ranging from 10 to over 50% of pre-colonial acreage (Bromberg and Bertness 2005). Current regulations now regulate human activities in salt marshes protecting them from direct human impacts, but not from continuing impacts associated with tidal restrictions (Roman and Burdick 2012) or other hydrologic alterations, including ditching, which can reduce resilience of the marsh and surrounding communities to coastal storms and sea level rise by leading to marsh drowning (Costanza et al. 2008; Raposa et al. 2016).

Most salt marshes in New England have been ditched, although the density of ditches, their origins, and maintenance vary from marsh to marsh (Bromberg and Bertness 2005; Bromberg Geden and Silliman 2009). In a survey of 20 New England salt marshes, Adamowicz and Roman (2005) found ditch density on the order of  $217 \pm 100$  m/ha. In Delaware, Corkran (1938) estimated 124 m/ha for 1916 ha of marsh in 1938, and in a small 200 ha site on Long Island NY, Corman and colleagues reported a density of 370 m/ha in 2012. The intentional isolation of large areas (> 100 ha) of salt marsh

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from all tides using berms or dikes fitted with one-way drainage valves (Bromberg Geden and Silliman 2009) was less common in New England than elsewhere in North America. Along the mid-Atlantic coast, some have recently been restored to tidal flow (Teal and Weishar 2005). Our focus was on densely ditched areas not associated with large agricultural berms.

Farmer ditching dates back to the 1600s and helped provide salt hay (*Spartina patens*, *Juncus gerardii*, and *Distichlis spicata*) as a ready source of fodder and bedding for use with livestock (Clift 1862; Hawes 1986; Sebold 1992). Salt farms were agricultural *systems*, and their management was promulgated in national journals (e.g., American Farmer 1820). A simple ditching formula used by salt hay farmers generally increased ditching density ~ 50–75% over the pre-colonial tidal channel density based on comparisons of unditched and farming-ditched areas (G. Wilson, pers. obs.). This reflects the natural creek versus ditch density documented in Adamowicz and Roman (2005) still present today.

Ditching used to control mosquito production started in Connecticut in 1912, with laborers paid by the linear foot (Britton and Walden 1916). This management effort was amplified during the Great Depression (Wolfe 1996). Bourn and Cottam (1950) estimated that 90% of tidal marshes from Virginia through Maine had been ditched. A more recent estimate based on 32 marshes in New England by Crain et al. (2009) found 94% of tidal marshes were ditched.

Mosquito ditching effects on wildlife populations were argued in the 1930s with mosquito control officials suggesting minor impacts and wildlife scientists noting sharp declines in waterfowl, especially where ditches drained permanent pools (Urner 1935; Cottam 1938; Headlee 1939; Tonjes 2013). Ditches have been shown to lower the water table in the marsh (Stearns et al. 1940; Turner and Lewis 1997), and oxygenation of the peat, mediated by respiring microbes, leads to subsidence and loss of elevation (Bourn and Cottam 1950; Vincent et al. 2013). A study of three New England marshes with moderate ditch density reports over 20 cm of elevation was lost to subsidence (grading from 24 cm at ditch edge to 6 cm at 7 m away and to 2 cm at 15 m away; Vincent et al. 2013). There also could be lower below-ground production and accretion as well as increased erosion at ditch edges, but subsidence due to oxidation, as found for tidally restricted marshes (Roman and Burdick 2012), may be the most important factor reducing elevation. Since water levels are reported relative to the marsh surface, subsidence can effectively conceal the extent to which ditches lower water tables. Subsidence in densely to moderately ditched marshes (less than 30 m apart) makes them even more vulnerable as sea level rise increases worldwide (Nicholls and Cazenave 2010) and especially in New England where it may be rising faster than the global average (IPCC 2014; Goddard et al. 2015; Raposa et al. 2016). In Boston, the closest long-

term tide gauge, sea level rise averaged 3.4 mm/year from 2008 to 2018 (25 years) based on a linear regression of monthly means for mean higher high water (NOS, CO-Ops; <https://tidesandcurrents.noaa.gov>).

In addition, the loss in elevation creates poor habitat for saltmarsh sparrows (*Ammodramus caudacutus*) that nest only in marsh vegetation associated with high marsh habitat (DiQuinzio et al. 2002). They are highly susceptible to flooding from storms or flooding tides that occur within the bird's 28-day nesting cycle (Benvenuti et al. 2018). Any loss of salt marsh elevation makes the birds more susceptible to marsh flooding. A current model (Field et al. 2017) anticipates possible extirpation of this species by 2035 if no steps are taken to protect their nesting habitat. Given the prevalence of marsh ditching, restoring marsh elevation and vegetation (function and condition) is imperative if this species is to be conserved.

Mosquito ditching maintenance continues to this day, but in the past 60 years, new mosquito control drainage systems were developed with greater consideration of the marsh ecosystem. Open marsh water management (Wolfe 1996), closed systems (James-Pirri et al. 2011), and integrated marsh management (Rochlin et al. 2012) all seek to balance mosquito control with improved ecological function, though some have failed according to Riepe (2010). While ditching may have had positive results in terms of mosquito control, we are concerned that the practice in New England has negative ecological impacts and has left marshes particularly vulnerable to accelerated rates of sea level rise and landward communities more vulnerable to storms.

Another approach to mitigating impacts was ditch plugging, where the seaward end of the ditch was filled with peat (often excavated from on-site) and fortified with a low berm (if necessary). Water trapped behind the plug led to severe waterlogging, plant death, and peat collapse (Adamowicz et al. 2004; Riepe 2010; Vincent et al. 2013, 2014). Other efforts to fill ditches as a pilot were investigated at Fire Island National Seashore (Corman et al. 2012) but were not implemented (Riepe 2010). Ditches were filled as part of an integrated marsh management project on Long Island (Rochlin et al. 2012) but excavated peat from the high marsh—exactly the habitat favored by saltmarsh sparrows.

Wilson and Adamowicz developed ditch remediation as an approach to set unwanted ditches on a self-healing trajectory that is simple and very low-tech and has shown positive results in an earlier, unpublished trial. To formally test the effectiveness of ditch remediation, we applied it to four groups of ditches in a marsh surrounding an undeveloped 9.7 ha island in the Great Marsh, Massachusetts, USA. Here we report changes over 3 years in ditch depth and natural colonization by *Spartina alterniflora*. We further anticipate that reduction in marsh drainage may lead to increased elevation of the marsh platform in the future.

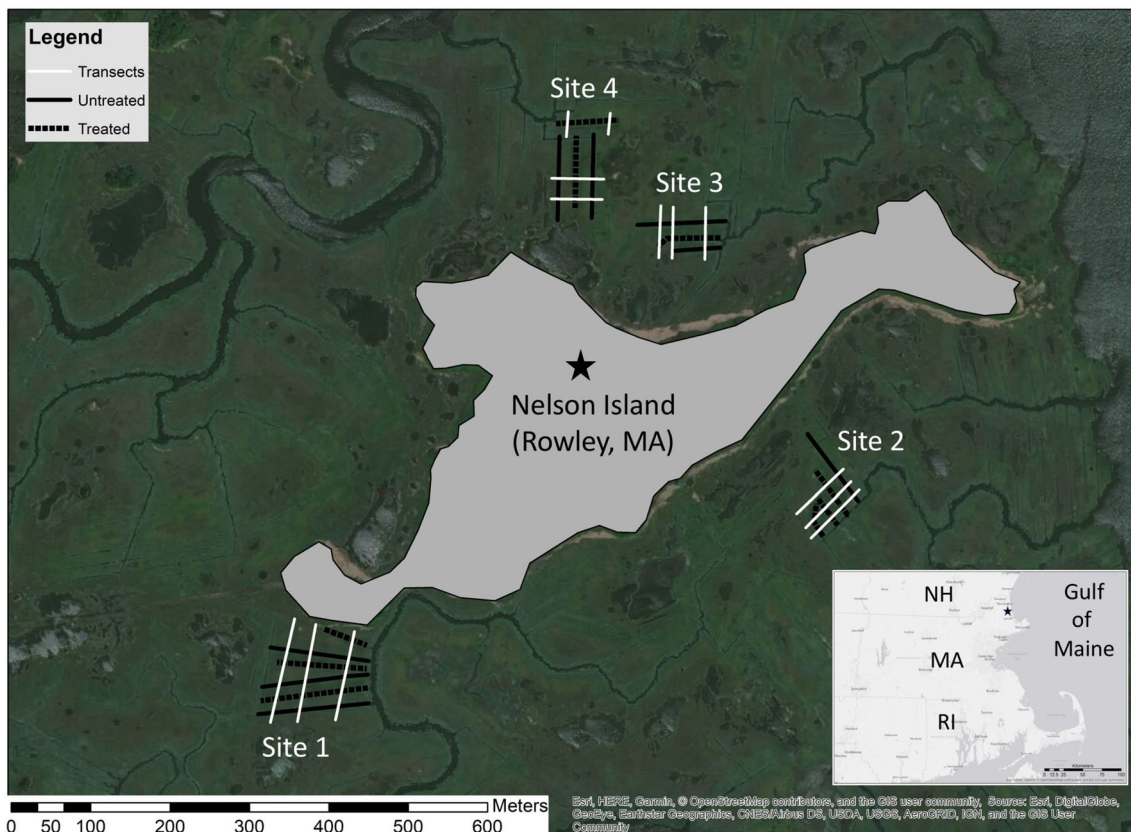
## Methods

Four ditched areas surrounding Nelson’s Island were chosen to be treated following examination of aerial photography and a site visit (Fig. 1). Ditch selection was based on ditch hierarchy; only first order ditches (*sensu* Horton) were filled. Auxiliary (smaller, shallower) ditches were selected for treatment, while care was taken to avoid ditches reverting to primary tidal channels. The four areas were predominantly high marsh with parallel ditches cut between panels dominated by *Spartina patens*. Some of the ditches supported tall form *Spartina alterniflora* and drained directly to, or in close proximity to, natural creeks. Most ditches were open and unvegetated their entire lengths. Working in nearby marshes, LeMay (2007) found suspended sediments averaged 10 mg/L at slack high tides. Application of the treatments were conducted outside of the annual fall migratory waterfowl seasons and during the neap tide cycles of September 30–October 4, 2014, and November 3–November 7, 2015.

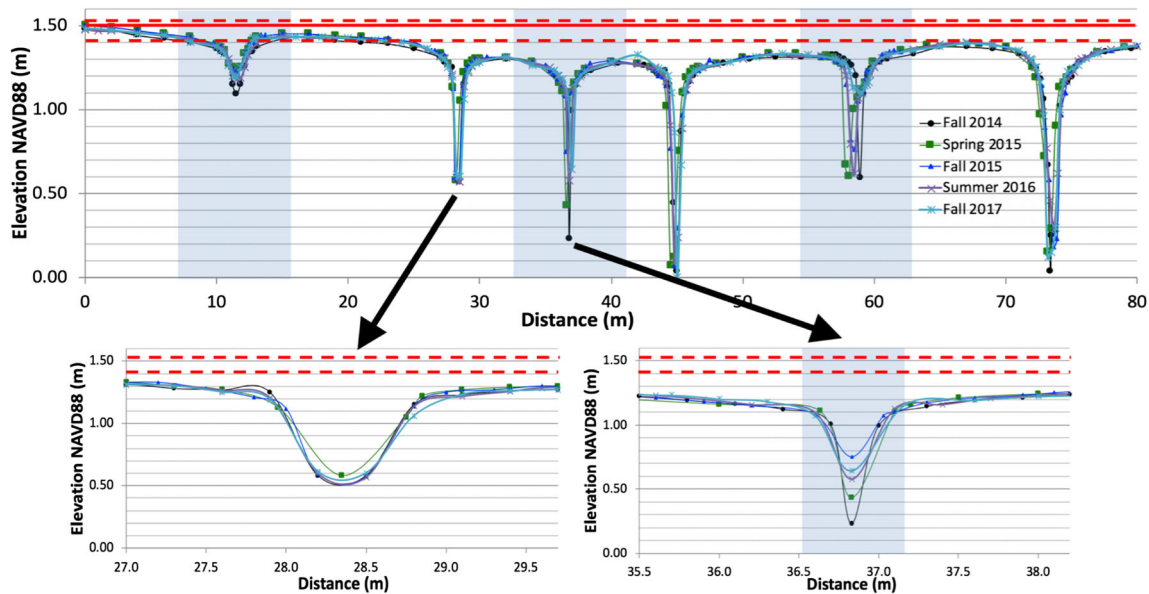
Treatment consisted of mowing a sufficient area of salt marsh perennial grasses on one or both sides of the treatment ditch to supply the entire length with a uniform 15–20 cm hay layer prior to compaction. Mowing was completed with a self-propelled brushcutter with a 66 cm wide deck. After mowing, hay was allowed to air dry for 24 h prior to loosely braiding

and placement into the treatment ditch. The loosely braided hay was lightly compacted by foot and secured to the ditch bottom with 59 kg tensile strength sisal baler twine pretreated with vegetable oil and softwood grade stakes.

Generally, a series of 3 parallel transects were established in each area perpendicular to the parallel ditches with the ends marked using 20 mm diameter PVC pipe and locations recorded using an RTK-GPS. Temporary benchmarks were established in upland using oak stakes and rock outcrops where available. Elevations and distances along transects were made with field tape and laser level in fall 2014 (before treatment) and again in spring 2015, fall 2015 (following second treatment), summer 2016, and summer 2017. Elevations were always taken at the sediment surface; treated ditches quickly acquired a thin layer of fine-grained sediments over the salt hay, and this was the surface measured. Data were collected every 2–3 m on the marsh plain and every 30 cm throughout the ditch feature. An example transect showing data from all five sampling dates is presented in Fig. 2. Plants were assessed at key locations along each transect (ditch center, transition zone to marsh plain, center of marsh plain) on either side of each transect using a 25 by 100 cm quadrat (0.25 m<sup>2</sup> area). Here only vegetation in the ditch center is reported, based on 0.5 m<sup>2</sup> survey area. Plant cover as a percentage of total plot area was assessed using ocular estimate, shoots > 5 cm in



**Fig 1** Study area surrounding Nelson Island in the Great Marsh, Massachusetts, showing arrangement of treated and untreated (control) ditches and the transects used to measure elevation and vegetation



**Fig. 2** Cross section of transect elevations showing treated (shaded) and untreated ditches for five sampling dates: fall 2014 pre-treatment, spring 2015 post-treatment, fall 2015 post-treatment, summer 2016, and fall 2017. Insets show details from an example of an untreated ditch (lower

left) and a treated ditch (lower right). Solid horizontal line in upper graph shows presumed marsh surface without ditching; mean high water (lower dashed line) and mean higher water (upper dashed line) were calculated from a nearby water level recorder operating for 1 month in 2016

height were counted, and height was measured to the cm for the three tallest shoots.

Photographs were taken in October 2014 and in late August thereafter to illustrate treatment responses compared to untreated ditches. Photo stations were established at both ends of each ditch, marked in the field with a 5 by 5 cm wooden stake and documented with GPS. Photos were taken at 1.3 m elevation by setting the camera (Canon EOS 60D with an EF-S 18–200 mm f/3.5–5.6 IS lens) on a fixed tripod. Without point of reference features in the marsh, photo numbers were recorded with the experimental treatment number on data sheets in the field for ease of comparison over time.

Ditch is considered the experimental unit. When several transects crossed the same ditches, values were averaged for each ditch prior to least squares analysis (ANOVA using JMP™). Variation between ditch depths was removed by subtracting the 2014 ditch depth from all data and analyzing the differences in elevation. For smooth cordgrass cover and stem density within ditches, the reduced dataset was analyzed using ANOVA with year of sampling and treatment as main effects (interactions were not significant). Data transformations to achieve normality and homoscedasticity were LN (stem density + 1) and LN (arcsinCover\*100) + 1).

### Results

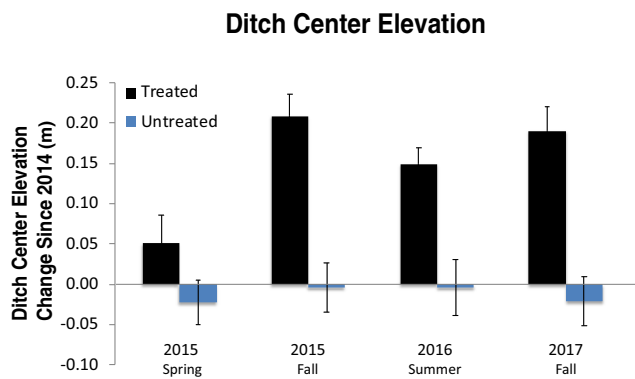
Transects ranged from 14 to 110 m in length and crossed from 1 to 6 ditches. Ditches were roughly parallel and averaged 13.9 m apart but ranged from 10 to 18 meters apart. Nine

ditches were treated with grass fill, and eight were left as controls. First measured in fall 2014 before any treatments, elevations of the high marsh plain between ditches averaged  $9.1 \pm 0.8$  cm lower than transect ends where ditches were less prevalent (Fig. 2 and Table 1).

The deepest part of the ditch crossed by the transect was measured on each of the 5 sampling dates. Prior to treatments, the depth of ditch centers receiving the treatment averaged  $0.67 \pm 0.21$  m NAVD88 and ranged from 0.35 to 1.09 m. Untreated ditch centers averaged  $0.35 \pm 0.27$  m NAVD88 and ranged from -0.04 to 0.88 m. Treated ditches were significantly shallower than untreated, despite considerable overlap. Our management approach was not to eliminate hydrologic channels, just reduce their number by converting the smaller ditches into vegetated swales. As planned, treatment in 2014 and 2015 led to shallower ditches, but the effect was only significant following the second year of treatment and thereafter (Fig. 3). Although average ditch depth increased slightly in 2016, when measured in 2017 after 2 years without

**Table 1** Subsidence of marsh plain center relative to transect ends

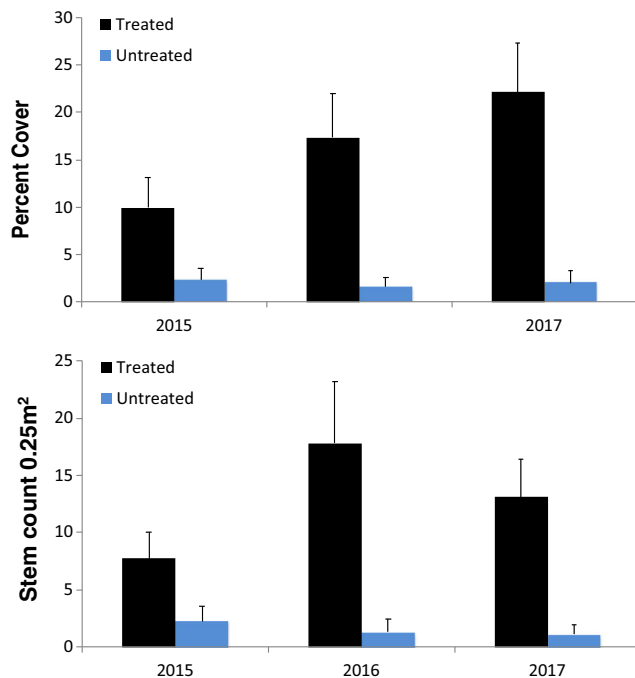
Area	Inter-ditch plains (n)	Subsidence (m)	Standard error (m)
1	13	0.113	0.012
2	7	0.059	0.008
3	4	0.070	0.022
4	4	0.100	0.015
Average	-	0.0914	0.0081



**Fig. 3** Shallowing of ditch depth (in m) from pre-treatment in 2014 and following 2014 and 2015 treatments

treatment, the average depth shallowed and was more similar to 2015 (Fig. 3).

Visual estimates of cover and shoot counts of plots placed in the ditch centers showed colonization by *Spartina alterniflora* in the treated ditches (Fig. 4). Quantitative results are supported by the chronosequences from the photo stations (Fig. 5). No vascular plant other than *S. alterniflora* was found growing in the ditch centers, and cover was about 8% greater in treated plots than controls in 2015 and 20% greater in 2017 (Fig. 4). Average stem density of *S. alterniflora* was greater in treated ditches in 2015 (about 20 shoots/m<sup>2</sup>) and increased dramatically in 2016 to over 15 per quadrat (60/m<sup>2</sup>). No significant change in vegetation occurred in the control ditches (Fig. 4).



**Fig. 4** Colonization of ditches by *S. alterniflora* following treatment in 2014 and 2015

## Discussion

Salt hay was added to the treated ditches in fall 2014 and 2015, and the depths decreased immediately but then increased slightly over time as the hay compacted and decomposed with the final measurement averaging 18 cm. The technique has starkly different results compared with previous efforts to mitigate ditch effects: ditch plugging (Adamowicz and Roman 2005; Vincent et al. 2013, 2014) or filling (Rochlin et al. 2012). Rather than disrupting hydrologic processes that sustain marshes by plugging or completely filling ditches, our technique works with hydrology to capture fine grain sediments and stimulates recolonization by *S. alterniflora* that can further catch sediments aboveground and rebuild peat with organic inputs belowground. This is similar to the model of rebuilding peat in pools drained by new connections to tidal creeks that was described by Wilson et al. in 2014. Ditch remediation also has the advantage of selectively treating auxiliary ditches, reinforcing flow to primary channels, and thus re-establishing “tide-shed” boundaries that were fractured by extensive ditch networks.

The low-impact technique does cut marsh grass to infill the ditches. Grass is cut at the close of the growing season that is similar to (but 1–2 months later than) the colonial practice of haying. Buchsbaum and colleagues (2006) report that haying is still practiced on 400 ha of marsh in this area of Massachusetts. Their field study found no impacts to end-of-year standing biomass or plant species density, but species composition differences favoring *S. patens* over *S. alterniflora* were found in areas subject to 2-year haying rotation compared to controls. Our preferred method is to use salt hay harvested *on-site*. Other sources of fill materials have the potential to result in unintended impacts of trampling (due to transporting off-site material to treatment areas) and invasive species introduction (seeds in offsite material). Commercially harvested salt hay could be used as long as propagules from *Phragmites* or perennial pepperweed (*Lepidium latifolium*) are not present.

Substantial gains in the percentage of cover and stem density of *S. alterniflora* show that the treatment has stimulated a process of recolonization of the ditches, though there are still ditch centers that have no vascular plants. It is unclear whether the shallowing of the ditches improved light reaching the center of the ditches or if the grass beds held or caught seeds and aided their germination and survival that stimulated colonization of the ditches. Either way, the establishment and spread of vegetation in the ditches may now act to build elevation through sediment trapping and biomass production and storage (Nyman et al. 2006; Wilson et al. 2014).

Loss in marsh plain elevation appears primarily caused by oxidation and subsidence of peat but could also be associated with slower rates of accretion, lower belowground production and storage, or greater erosion compared to non-ditched areas.

**Fig. 5** Visual comparison of treated versus untreated ditches reveals notable recolonization by marsh grasses, particularly in shallower ditches (left) versus untreated ditch (right)



Measurement of marsh plain subsidence was not the primary intent of the investigation, but conservative estimates show the crest of the inter-ditch plain had subsided at least 9 cm (with greater amounts closer to ditches) since mosquito ditching in the 1930s (approximately 80 years). The average distance between ditches at the sites was  $14.6 \pm 0.6$  m, so it appears subsidence at this location was a bit more intense than the three marshes investigated by Vincent et al. (2013) who found 6 cm as the average subsidence 7.0 m from ditches. In 2007, LeMay working near Nelsons Island reported ditched marsh plains were an average of 7.4 cm lower than marsh areas not ditched. Further, Warren and Niering (1993) found that the un-ditched marsh at Wequetequock Cove was 9.3 cm higher than the ditched Headquarters Marsh, though they ascribed the difference to reduced accretion rates rather than subsidence. Some might argue that subsidence measured in this way is meaningless since farmers and later mosquito control teams were likely to have focused ditching in low elevation areas of the marsh to remove standing water. However, present-day measurements show that the vegetated marsh associated with wetter pool areas on a marsh are actually *higher* in elevation (LeMay 2007; Millette et al. 2010; Vincent et al. 2013).

### Further Research Needs

Three years following the first treatment of old mosquito ditches in a New England salt marsh, there are many unanswered questions. Will revegetation of cordgrass continue and

will ditches continue their trajectory of filling so that further management is unwarranted? Once ditches revegetate, it is assumed the trajectory of infill will continue but slow as the volume of water and sediments passing through diminishes. Without further manipulation of the site, how shallow will the ditches become and will they maintain a hydrologic function over decades or will the ends fill and begin to impound water? In this case, monitoring linked to adaptive management (e.g., Lyons et al. 2008) may allow for simple maintenance solutions. Alternatively, the rate of sea level rise might surpass that of filling and the ditches might deepen and expand.

At the larger scale, will infilling of ditches to a depth of 15 to 25 cm slow deep peat oxidation (subsidence) and stimulate more rapid rise of the surrounding marsh plains through organic matter storage, thus increasing resilience of both the marshes and human communities they protect to climate change? The optimum channel network will result in flooding and ebbing that achieves a certain inundation period both on the surface and in the root zone for plant health, marsh resilience (Howes et al. 1986; Morris et al. 2002; Vincent et al. 2013, 2014), and management (Bourn and Cottam 1950, <https://www2.who.edu/site/marshsustainabilityandhydrology/>). If suspended sediments are a finite resource, will ditch remediation decrease sediment availability for marsh plain building or will the hydraulic forcing through remaining ditches deliver sediments to marsh surfaces more efficiently? Long-term monitoring can determine whether marsh plains build faster if drained less following ditch remediation and measuring accretion at paired controls (unrestored ditched areas) may determine whether ditch infilling “robs” the marsh of sediment.

## Management Implications

Almost all tidal marshes have been ditched in the eastern USA, and the loss in elevation these ditches cause increases the vulnerability of these marshes to sea level rise and subsequent loss (Raposa et al. 2016). Further, marsh loss will increase vulnerability and decreases resilience of coastal communities (Costanza et al. 2008). Marsh managers and restoration practitioners should actively engage mosquito control agencies to mitigate the negative effects of mosquito ditching by reducing the number of deep ditches in marshes. Ditch remediation can be used as one of the several tools in salt marsh restoration, recognizing that each management area needs to consider how many and which ditches should be filled to what depth to optimize marsh building and hydrology.

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