Drainage enhancement effects on a waterlogged Rhode Island (USA) salt marsh

Kenneth B. Raposa1, Robin L. Weber1, Wenley Ferguson2, Jeffrey Hollister3, Ron Rosza4, Nicole Maher5, and Alan Gettman6

1 Narragansett Bay National Estuarine Research Reserve, Prudence Island, RI, USA

2 Save The Bay

3 EPA

4 CT DEP (retired)

5 The Nature Conservancy Long Island Chapter

6 Rhode Island Department of Environmental Management, Kingston, RI, USA

**Abstract**

Drainage enhancement (e.g., ditch digging, open-marsh water management, runnelling) has long been used to reduce tidal marsh soil waterlogging and surface ponding to promote farming and mosquito control. Now it is also being used as a tool to enhance marsh resilience to sea-level rise despite a lack of studies that evaluate its effectiveness as an intervention approach. We therefore conducted a controlled field experiment to evaluate short-term responses to drainage enhancement of a waterlogged Rhode Island (USA) salt marsh. Drainage enhancement elicited rapid physical changes including declines in water levels and marsh elevation, but biological communities were largely unaffected. In some locations of the marsh, surface inundation declined from > 75% to 3-10% and low water levels dropped by 20 cm. Elevations across the marsh increased 5 mm one year after drainage enhancement but dropped to 11 mm below initial conditions after three years. The decline in elevation varied among habitats, with the greatest decline (18 mm) found in areas dominated by *Spartina alterniflora* and/or bare ground. Vegetation communities were unchanged, but areas that were initially bare had fully revegetated after three years. Drainage enhancement also had no effects on bird communities or marsh sparrow (*Ammodramus* spp.) density. Our study provides evidence that drainage enhancement can relieve marsh waterlogging and help bare areas quickly revegetate, without any apparent adverse effects to existing biological communities. At the same time, it can induce a loss of marsh platform elevation that has the potential to offset declining water levels and inhibit high marsh enhancement. Finally, unanticipated findings from our study provide evidence that drainage enhancement can increase marsh crab abundance and impacts, and that the effects of larger-scale drivers, such as sea-level rise, may predominate over localized responses to drainage enhancement itself.

**Introduction**

Tidal salt marshes worldwide are threatened by sea-level rise (SLR), with the degree of vulnerability and extent of impacts varying widely at different scales (Kirwan et al. 2016, Raposa et al. 2016, Cole Ekberg et al. 2017). The problem is particularly acute in southern New England (USA), where SLR is accelerating rapidly and many marshes are highly vulnerable due to limited sediment supplies, slow rates of elevation gain, minimal elevation capital, and concomitant impacts from crab burrowing and herbivory (Weston 2014, Raposa et al. 2017a, Crotty et al. 2017, Watson et al. 2017, Raposa et al. 2018a). In Rhode Island alone, 17% of the total statewide marsh area has been lost since the 1970s due to impacts associated with SLR, and there are signs this rate of loss may already be accelerating in concordance with future SLR projections (Watson et. al. 2017, Raposa unpublished data).

One consequence for marshes that are not able to maintain their position in the tidal frame with accelerating SLR is an increase in the duration and extent of surface flooding, coupled with excessive soil waterlogging (Hill and Anisfeld 2015, Anisfeld et al. 2016). This leads to conditions that inhibit plant growth, or kill plants outright, which in turn results in conspicuous and increasingly common signs of SLR impacts such as vegetation composition shifts, interior marsh ponding, headward and lateral creek expansion and, eventually, marsh loss (Warren and Niering 1993, Raposa et al. 2017b, Watson et al. 2017). Thus, one strategy for building marsh resilience to SLR is to conduct projects aimed at reducing surface flooding, increasing drainage and alleviating soil waterlogging (Wigand et al. 2017).

There are different approaches that can be used to try to accomplish these goals. Sediment addition projects utilize externally-sourced soils to increase the elevation of low-lying marshes so that surface inundation is reduced, and drainage improved, thereby promoting healthy plant growth (Croft et al. 2006, Stagg and Mendelssohn 2010). This approach appears promising (Schrift et al 2008, Cahoon et al. 2019) but can be expensive and is mostly limited to marshes close to viable soil sources (e.g., marshes adjacent to channel dredging). An alternate approach, and one that is generally less expensive, is to dig new drainage channels into waterlogged marshes to improve tidal exchange between marsh and estuary and increase drainage out of the marsh (hereafter referred to as ‘drainage enhancement’). Options here include digging naturalized sinuous channels that mimic creeks (e.g., Rochlin et al. 2012), creating or cleaning out existing linear ditches, and digging runnels, which are small, shallow channels across the marsh surface whose purpose is to drain specific ponded areas (Dale et al. 1993). In southern New England, many drainage enhancement projects have been undertaken or are ongoing, most in isolation, but others conducted in combination with sediment addition in a more holistic, adaptive management approach. For example, 23 drainage enhancement projects totaling x ha affected have been conducted in RI just in the past decade, largely with a focus on digging runnels to reduce surface ponding; similar projects have occurred in other areas within the southern New England/Long Island Sound region, but not as extensively as in RI (Table 1).

Many drainage enhancement projects do not have the capacity to include a comprehensive monitoring component, and are thus unable to assess effects (Table 1); substantial gaps therefore remain in understanding how these kinds of projects affect marshes and how best to design future projects to meet objectives and foster success. These issues are not new. There is a long history of digging in marshes, often in an attempt to reduce water levels, surface ponding, and mosquito populations. In the northeast US, these projects have progressed from colonial-era ditch digging to promote salt hay farming, to Depression-era ditching for mosquito control, recent open-marsh water management (OMWM) projects that augment ditching with pond creation, and current-day drainage enhancement projects aimed at building marsh resilience to SLR (See reviews in Crain et al. 2009 and Tonjes 2013). Many positive results have been reported, but these can be short-lived without regular maintenance to ensure that drainage channels remain functional. In Connecticut, for example, colonial farmers initially dug ditches that promoted salt hay farming, but the ditches were not well maintained and eventually began to fill in, leading to waterlogged conditions, pond development, and vegetation die-off within decades (Britton 1912). Subsequent re-digging of ditches in the early 1900s quickly improved marsh surface drainage and led to elevation gains, loss of marsh surface ponds, and eventual establishment of high marsh plants (Britton 1912). Results of OMWM are generally mixed (James-Pirri et al. 2012), but another recent example of a holistic approach to drainage enhancement involving channel and pool creation coupled with ditch filling (Integrated Marsh Management; IMM), was shown to reduce invasive speciescover and mosquito populations and increase high marsh vegetation and wildlife (Rochlin et al. 2012). These studies highlight beneficial outcomes from drainage enhancement projects, but others demonstrate adverse impacts including eventual channel slumping and erosion, loss of marsh surface pools and natural creeks and associated fauna, and impacts to resident birds, often in the absence of improved mosquito control (Dale and Hulsman 1990, Adamowicz and Roman 2005, Pepper and Shriver 2010, Tonjes 2013). Further, the long-term effects of drainage enhancement are still emerging (Rosza, pers. obs.), with Coverdale et al. (2013a) reporting a decrease in overall marsh resilience through interactions between historic ditching and other stressors.

The effects of drainage enhancement are complex and can vary in type and magnitude among different kinds of drainage channels (e.g., ditch vs runnel). They also depend on the time scale of observation, whether or not channels are maintained over time, and which parameters are chosen for monitoring. Controlled field experiments that evaluate the effects of drainage enhancement are needed, including in different regions (Crain et al. 2009). We therefore conducted one such experiment with the goal of assessing physical and biological effects of a drainage enhancement project in Narragansett Bay, RI. We quantified responses with before and after monitoring in the experimental marsh (i.e., the marsh where digging occurred) and in a nearby control marsh to isolate changes that were only due to drainage enhancement. We focused on monitoring parameters that are suitable for evaluating objectives common to any drainage enhancement project, including reducing marsh surface inundation, increasing drainage, strengthening soils, reducing marsh surface ponding, and increasing the extent and condition of high marsh vegetation. Results of our study will improve our understanding of how marshes respond to drainage enhancement, and should ultimately help restoration practitioners improve future project design.

**Methods**

**Study site**. This study was conducted in the 22.0-ha Coggeshall Marsh, located within the Narragansett Bay National Estuarine Research Reserve (NBNERR) on Prudence Island RI (Fig. 1). Coggeshall Marsh is a typical southern New England meadow/fringe marsh dominated by *Spartina alterniflora* in low-elevation regularly-flooded areas and a mix of high marsh (i.e., salt meadow) species (*Spartina patens, Distichlis spicata and Juncus gerardii*) in higher, irregularly-flooded areas (Neiring and Warren 1980). Much of the marsh lies at elevations below the level for optimal *S. alterniflora* growth, and it is currently undergoing rapid vegetation shifts from *S. patens* to *S. alterniflora*; all told, Coggeshall Marsh is predicted to lose approximately 40% of its current area with just 0.9 m of additional SLR (Cole Ekberg et al. 2017; Raposa et al. 2017b; Watson et al. 2017). During surveys to assess SLR impacts to RI marshes in 2012 (Cole Ekberg et al. 2017), the northern 3.1-ha section of Coggeshall Marsh was identified as waterlogged, with stressed marsh vegetation and patches of invasive common reed *Phragmites australis* (hereafter, *Phragmites*). New drainage channels were subsequently dug into this section of marsh to relieve waterlogging. In our study, we call these new channels ‘creeks’ because they were excavated with a sinuous morphology to try to mimic natural creeks, even though they are analogous with the ‘channels’ described by Rochlin et al. (2012) and their dimensions are similar to typical linear ditches. This is the focus of our study, which used the remaining 18.9-ha of the marsh an experimental control within a before-after-control-impact framework (BACI; Stewart-Oaten et al., 1986).

**Drainage enhancement.** The project at Coggeshall Marsh was primarily drainage enhancement via naturalized creek creation, but it also included invasive *Phragmites* removal. Initially, only 116 linear meters of creeks, totaling 128 m3 in volume at a density of 37 m ha-1, were present in the experimental marsh. It was estimated that a total creek volume of between 254-495 m3 would be needed to drop water level depths by 4 cm on average across the marsh. Ultimately, 645 m and 352 m3 of new tidal creeks were created in late fall/early winter 2013, resulting in a total creek length of 761 m, at a volume of 480 m3 and density of 242 m ha-1, in the experimental marsh (Fig. 1). Nearly all excavations were conducted with a low-pressure excavator (IHI model 35NX; 3600 kg, 2 psi, modified with 0.76 m-wide nylon polymer track pads), although some hand-digging with shovels also occurred in very soft areas that were difficult to reach with the excavator. Mean width and depth of new creeks was 1.6 m and 0.4 m, respectively, though this varied among different sections of the marsh due mainly to the ability of the soils to support and provide access for the excavator. Creek width and depth was largest near the mouth of the main creek (3.3 m, 0.5 m); creeks in the western section of the marsh were narrower and deeper (1.3 m, 0.4 m) than those in the eastern section (1.6 m, 0.3 m). Before excavations, the eastern section of the marsh was bisected west to east by a remnant linear ditch, which was filled in with *S. alterniflora*, without free-flowing water, although it remained lower in elevation (by approximately 10 cm) than the surrounding marsh and did appear to provide a minimal amount of drainage to this section of marsh. This remnant ditch was replaced with a new creek as part of this project. In addition to creek excavations, all *P. australis* present in the experimental marsh (0.83-ha total) was treated with herbicide via foliar spray in early fall, followed by mowing later in the season, annually from 2013-2016.

**Monitoring**. To assess responses of the experimental marsh to drainage enhancement, we monitored water levels, marsh platform elevations, soil strength, habitats, vegetation, and birds annually from 2013-2016 (one year before drainage enhancement, three after) in the experimental and control marshes. We began the first year of monitoring before the final design of the drainage enhancement component was completed. Without knowing exactly where the new creeks would be, we decided to randomly locate all monitoring locations throughout the experimental marsh. Much of the monitoring occurred along three transects in each marsh that stretched from the seaward marsh edge to the marsh/upland transition (i.e., the ecotone). To try to isolate the effects of just drainage enhancement, we designed our monitoring to exclude the *P. australis* areas when possible; for this study no transects in the experimental marsh extended into *Phragmites*. The only exception was bird monitoring because it was sometimes difficult to visually determine what habitat an individual bird was in (e.g., leading edge of *Phragmites* vs adjacent *Spartina* spp.) especially at greater distances, and because many individuals moved between both areas during a single survey.

Water levels. Water levels were monitored in the experimental and control marshes each year using Onset Hobo water level loggers. In the experimental marsh, four locations were randomly chosen from within all existing short-form *S. alterniflora* or bare areas (either unvegetated or covered with dead vegetation); in the control marsh, we used two existing logger locations, also in short-form *S. alterniflora*. At each location, a logger was deployed in a shallow PVC well (4-cm diameter; 70-cm deep) sunk into the marsh, with the upper 10 cm of the well extending above the marsh surface (Roman et al. 2001). Each well was capped on the top and bottom and drilled with 6-mm holes for drainage. Water level readings were taken every 15 minutes throughout the June through September growing seasons. All water level data were referenced to the level of the local marsh surface at the wells to calculate 1) percent of time the marsh surface was flooded (i.e., inundation) and 2) minimum daily water depth below the marsh surface (i.e., drainage).

Elevation. Marsh surface elevations were monitored each year using a Trimble R8 GPS enabled for real-time kinematic (RTK) surveys and referenced to the North American Vertical Datum of 1988 (NAVD88). Elevations were collected at approximately meter intervals along the lengths of each transect in the experimental marsh only, and from five locations within each vegetation plot each year in both marshes. The former data were used to create elevation profiles along the transects in the experimental marsh, with a focus on elevation changes within major habitat types (see ‘habitats’ below). The latter data were used to determine if drainage enhancement affected the elevation of the experimental marsh by comparing plot elevations among years, and doing the same in the control marsh.

Soils. Penetration depth, an indicator of soil strength, was measured at eight randomly-chosen locations along the vegetation monitoring transects in each marsh (in any habitat excluding creeks and ecotones). Penetration depth was measured using a capped PVC tube and a slide hammer penetrometer weighing 4.6 kg and exerting a force of 20 J per blow to a surface area of 38.5 cm2 (size of a typical 5-cm white PVC cap). Penetration depth is defined as the depth of penetration of the PVC tube into the soil after five blows from the hammer attached to the penetrometer.

Habitats. The extent of pre-defined habitat types (Supplemental Table 1) along each transect in each marsh was monitored annually with the RTK-GPS to track change over time and to couple with elevation profiles in order to examine elevation change within major habitat types. Habitat extent was quantified by collecting georeferenced points at every boundary between adjacent habitats along each transect; the width of each habitat band could then be calculated from these points and summed across all transects within each marsh.

Vegetation. Emergent marsh vegetation was monitored annually in each marsh around the end of the growing season (late August/September) following National Estuarine Research Reserve (NERR) protocols (Moore 2013), which in turn are similar to those in Roman et al. (2001). Monitoring occurred in vegetation plots spaced at equal intervals along the transects in each marsh, with 20 plots in the experimental marsh and 21 plots in the control marsh. Community composition and the percent cover of individual species and other cover types (e.g., bare ground) were quantified using the point intercept method at 50 points in each plot and, when present in a plot, up to 12 individuals of each target species (*S. alterniflora*, *S. patens*, *D. spicata*, and *J. gerardii*) were measured to assess canopy height.

Birds. Marsh birds were monitored biweekly from late June through early September (six dates each year) in each marsh using the Bird Area Survey protocols (USGS 2012). Surveys were conducted from four locations in the control marsh and two in the experimental marsh. All birds seen in any visible part of the marsh from each location were identified and counted; counts were later converted to densities (number of birds ha-1) using the area of marsh visible from each survey location.

**Statistical analyses**. All statistical analyses were run separately for the experimental and control marshes, and the general approach was to run tests among the four years of the study to compare data before (2013) and after (2014-2016) drainage enhancement. Marsh habitat composition was compared among the four years using analysis of similarity (ANOSIM, square-root transformed data, Bray-Curtis similarity resemblance matrices). For significant global tests, subsequent pairwise comparisons were conducted to identify pairs of years that were significantly different. This same approach was also used for comparing vegetation and bird communities over time. Individual monitoring parameters (soil strength; *S. alterniflora, S. patens*, and *D. spicata* cover and height; bare ground cover; and densities of the ten most abundant bird species) were compared among the four years with one-way analysis of variance (ANOVA, untransformed data); when the assumption of normality could not be met, Kruskall-Wallis ANOVA on ranks tests were used instead. Any significant differences from ANOVA were followed by Tukey pairwise comparison tests to identify which years were significantly different. All statistical tests were run using SigmaPlot version 14.0 and PRIMER version 7.0.13.

**Elevation change visualization.** In addition to statistical tests for elevation change, a visualization of changes between 2013 and 2016, by habitat, was developed. Profile elevations were smoothed using a local regression (i.e. loess) with a smoothing parameter set to 0.12. The portions of the transect, for both years, were assigned habitats based on the habitat classification in 2013. All data analysis for this visualization was conducted in R version 3.5.3 and data ingest with readr and readxl packages, data manipulation done with the dplyr, stringr, and tidyr packages and visualization built with ggplot2, hrbrthemes, and cowplot (Wickham 2016, Wickham et al. 2018, R Core Team 2019, Rudis 2019, Wickham 2019, Wickham and Bryan 2019, Wickham and Henry 2019, Wickham et al. 2019, Wilke 2019). Code and data for this part of the analysis are available at <https://github.com/jhollist/marsh_restore>.

**Results**

**Water levels**. Changes in water levels in the experimental marsh after drainage enhancement were not consistent among the four logger locations. At two locations, the marsh surface was initially very waterlogged and remained inundated >75% of the time. After drainage enhancement, surface inundation declined dramatically to 3-10%, and mean daily low water level dropped by approximately 20 cm (Fig. 2). These two locations were relatively far from existing creeks (61-122 m) in 2013; after excavations, distance to creek decline substantially (< 4 m). At the other two locations, waterlogging and surface inundation was initially less severe, and both parameters were relatively unchanged after drainage enhancement (Fig. 2). For these wells, distance to creek was very similar before (13-28 m) and after (10-13 m) excavations. Water levels remained relatively unchanged over time in the control marsh.

**Elevation**. Elevations of monitoring plots were not significantly different among years in either marsh (ANOVA on ranks; experimental marsh, *H* = 0.35, *P* = 0.95; control marsh, *H* = 0.24, *P* = 0.97). However, a closer examination reveals that mean plot elevation in the experimental marsh increased by 5 mm the first year after drainage enhancement but dropped to 8 mm below initial conditions after three years (Fig. 3). After excluding one plot near the upland border where creek spoils were inadvertently deposited, the marsh decline was more pronounced (mean loss of 11.5 mm by 2016). Mean vegetation plot elevation in the control marsh remained unchanged over time (0.61-0.62 m NAVD each year). The magnitude of elevation decline in the experimental marsh varied markedly depending on the type of dominant vegetation that was initially present in each plot, ranging from a mean loss of <1 mm in high marsh habitats, to 6 mm in high marsh/*S. alterniflora* mixed habitats to 18 mm in plots dominated by *S. alterniflora* and/or bare ground (Fig. 3). Similar changes in elevation for these habitat types were also observed at a broader scale from elevation profiles across the entire length of the transects (Fig. 4).

**Soils**. Soil strength (i.e., penetration depth) trended firmer over time in the experimental marsh (Fig. 5), with a marginally significant difference among years (ANOVA on ranks, *H* = 1.42, *P* = 0.70). Soil strength in the control marsh remained virtually unchanged over time (ANOVA on ranks, *H* = 3.30, *P* = 0.35).

**Habitats**. There were no significant changes in overall habitat composition over time in either marsh (ANOSIM; *R* = -0.28, *P* = 0.99 for the experimental marsh; *R* = -0.28, *P* = 0.96 for the control marsh), but drainage enhancement did elicit some conspicuous changes for specific habitats. The area of creeks increased the first year after excavation, but then declined in subsequent years as they began to re-fill and as vegetated creekbanks converted to bare (Table 2)*.* Bare areas on the marsh platform increased the first year after drainage enhancement but completely revegetated after three years (58% as high marsh mix, 36% as *S. alterniflora*, and 6% as high marsh). Bare areas were minimal in the experimental marsh in 2013 (only 0.8 ha), but these too had completely revegetated by 2016 (100% with *S. alterniflora*). The amount of high marsh remained relatively constant over time in both marshes.

**Vegetation**. Vegetation communities in both marshes were dominated by *S. alterniflora*, *S. patens*, and *D. spicata*, with ecotones comprised mostly of *Iva frutescens* and *J. gerardii*, or *Spartina pectinata* and *Schoenoplectus americanus* in brackish areas of the experimental marsh (Table 3). Drainage enhancement had virtually no effect on vegetation; there was no significant difference in experimental marsh vegetation community composition before and after excavations (ANOSIM, *R* = -0.04, *P* = 0.99). The vegetation community also did not change over time in the control marsh (ANOSIM, *R* = -0.01, *P* = 0.70). There were no changes in *S. alterniflora*, *S. patens*, or *D. spicata* percent cover over time in either the experimental marsh or the control marsh (ANOVA on ranks, *P* > 0.05 for each species). Bare ground cover in the experimental marsh did not change over time (ANOVA on ranks, *H* = 5.06, *P* = 0.17), but it was significantly higher in 2015 and 2016 compared to 2013 in the control marsh (ANOVA on ranks, *H* = 13.71, *P* = 0.003; *P* < 0.05 for Tukey pairwise tests). Finally, mean heights of *S. alterniflora*, *S. patens* and *D. spicata* did not change across any years in either the experimental or control marsh (ANOVA for *S. alterniflora* and *S. patens*, ANOVA on ranks for *D. spicata*, *P* > 0.05 for each species). Temporal trends in vegetation cover and height in the experimental marsh appear unrelated to drainage enhancement and instead mirror those ongoing in the control marsh, namely the declines in cover and height of dominant species and the increase in bare ground cover over time (Figs. 6 and 7).

**Birds**. Bird communities in both marshes were dominated by Red-winged Blackbird, Tree and Barn Swallows, and Grassland Sparrows (Saltmarsh Sparrow *Ammodramus caudacutus* and Seaside Sparrow *Ammodramus maritimus*) (Table 4). Bird communities did not change over time in the experimental marsh (ANOSIM; *R* = 0.05, *P* = 0.20). They did change in the control marsh (ANOSIM; *R* = 0.19, *P* = 0.03), but the only pairwise difference was between two of the years after drainage enhancement (2014 v 2016, *R* = 0.42, *P* = 0.02); all pre- vs post-excavation comparisons in the control marsh were not significant (*P* > 0.05 in each case). Densities of the ten most abundant species in the experimental marsh (Grassland Sparrows, Red-winged Blackbird, American Robin, Tree Swallow, Barn Swallow, Cedar Waxwing, American Goldfinch, Flycatchers, Gray Catbird and Song Sparrow) did not change over time (ANOVA on ranks, *P* > 0.05 for each species); the same was true in the control marsh.

**Discussion**

Ecological responses to tidal marsh drainage enhancement vary widely, with pronounced differences among different types and sizes of created channels (i.e., less-intrusive channels such as shallow runnels generally elicit fewer impacts to marshes, even over the long-term; Dale 2005, Dale and Knight 2006). Our study provides an case study example of generally muted responses by a waterlogged marsh to the creation of new tidal creeks. Some of the parameters we monitored were clearly affected by drainage enhancement, but many were little changed over time. Most conspicuous changes were physical, including rapidly-improved hydrology, a loss of elevation that varied in degree among habitats, and marginally stronger soils; biological changes were less apparent. These results mirror those from marsh hydrological restoration projects that report rapid changes to physical conditions before biological responses emerge (Konisky et al. 2006, Raposa et al. 2018b). The changes that did occur in our study also varied spatially within the experimental marsh or over time after excavations. The duration of surface inundation was reduced, and drainage improved, but not everywhere in the marsh. Elevation initially increased after drainage enhancement, but then declined over time. Even more confounding is that some of the changes in the experimental marsh also occurred in the control marsh (e.g., complete revegetation of areas that were bare prior to excavations). This suggests that, for some parameters, larger-scale factors such as SLR may have played a more influential role in eliciting change than localized drainage enhancement. These findings collectively illustrate the highly variable and dynamic nature of marsh responses to drainage enhancement, and call for long-term monitoring of additional projects (including control marshes) from different regions.

**Water level changes**. There is a long history of digging in marshes to successfully induce drainage for farming, mosquito control, and other purposes. We therefore expected water levels to drop at each of the four logger locations in our study, but this only occurred at half of them. Why did hydrology change so dramatically at some, but not all, locations? Many studies report lowered water levels resulting from channel digging (Lesser et al. 1982, Tonjes 2013), although this is not universal (Vincent et al. 2013), and the effect diminishes in marshes that experience lower tide ranges (Taylor 1938). Hydrology can also vary at small spatial scales within a marsh depending on distance to drainage channels, depth of channels, soil characteristics, topography, and other factors (Lesser et al. 1982, Nuttle and Hemond 1988, Bradley and Morris 1990, Howes and Goehringer 1994). Our study was not designed to identify which factors affected hydrology, but we suspect the divergent responses among logger locations were driven by pre-existing conditions and proximity to creeks. Earlier studies have demonstrated that water level declines are more pronounced closer to drainage channels (e.g, Nuttle and Hemand 1988, Montalo et al. 2006). The two locations where water levels were greatly lowered in our study were initially very impounded, interior sections of the marsh located far from the existing creek; after excavations, distance to the nearest creek was greatly reduced. The two locations where water levels remained unchanged were initially less impounded than even the control marsh and were located close to creeks, both before and after excavations. Thus, hydrology was only improved when new creeks were dug into areas that were initially isolated and impounded. For future projects, this highlights the need to understand pre-existing patterns in hydrology at small scales, perhaps by deploying many loggers throughout a marsh. This would ensure that new drainage channels are only placed in locations with impaired hydrology, thereby minimizing unnecessary digging that could foster negative marsh impacts (e.g., Bourn and Cottam 1950, Adamowicz and Roman 2005, Coverdale et al. 2013, and see below).

**Elevation decline**. Another notable physical change induced by drainage enhancement was a net drop in elevation across the experimental marsh. Marsh elevation increased the first year after drainage enhancement, likely due to spoil deposition on the marsh surface from creek digging (spoils were initially deposited along new creek edges but were later dispersed across the marsh with the excavator). Over time, elevation dropped beyond initial conditions, with declines most pronounced in highly degraded habitats such as sparse, stunted *S. alterniflora* and bare areas with limited belowground biomass to support soils. It is not uncommon for surface elevations to decline after digging into a marsh due to soil compaction and subsidence and the diversion of sediment into ditches that would otherwise accumulate on the marsh surface (Lemay 2007, Corman et al. 2012, Vincent et al. 2013). In our study, the declines were likely due to water loss from unconsolidated pore space, leading to rapid soil compaction and subsidence. Some southern New England salt marshes increasingly rely on soil swelling from excess water retention to drive accretion in lieu of declining contributions from inorganic materials and organic matter (Carey et al. 2017). Highly waterlogged soils can expand like a sponge (i.e., ‘swell’) and temporarily inflate marsh surface elevation (Nuttle et al. 1990, Cahoon et al. 2011). This sequence is reversed when soils are allowed to dry out and compact in association with drainage enhancement projects.

A drop elevation after a drainage enhancement or other intervention project should raise concerns, but in our study, it likely led to increasingly stronger soils with compaction. In some cases, it may be appropriate to accept some degree of elevation loss in exchange for improving hydrology and soil strength, and helping vegetation recolonize. Given that RI marshes are increasing in elevation at a rate of only 1.4 mm yr-1 on average (Raposa et al. 2017a), our project resulted in approximately 8 years of elevation capital lost at the marsh-wide scale, and 13 years lost in *S. alterniflora* and bare habitats. Without drainage enhancement, waterlogged areas with minimal plant growth in the experimental marsh would have likely continued along a trajectory towards drowning. In the short term at least, our project led to improved hydrologic and soil conditions and helped bare areas revegetate. Over the longer-term, however, these areas may still eventually succumb to drowning because recolonization was primarily with stunted *S. alterniflora*, which is not able to keep up with SLR when growing below its optimal elevation (Watson et al. 2017).

**Biological responses**. Biological communities were largely unaffected by drainage enhancement, but the duration of our project was limited, and biological parameters can take longer to respond to restoration and intervention activities than physical ones. Vegetation shifts in response to earlier drainage enhancement projects are common, but more specific changes vary widely. An increase in woody vegetation associated with drier, high elevation areas is often reported; increases in high marsh grasses are less common, and in some cases vegetation was largely unaffected (see reviews in James-Pirri et al. 2012 and Tonjes 2013). These disparate findings demonstrate that vegetation responses to drainage enhancement are complex and can be affected by many interacting factors including project type, design, and execution, relative changes in water levels and marsh elevation, and the scale of observation. In our project, there were no significant differences in vegetation community composition, species cover, or canopy height after drainage enhancement at the marsh-wide scale, but at a smaller scale, areas that were initially bare had completely recolonized after three years. There was also no increase in high marsh vegetation, perhaps because any potential gains in high marsh from improved drainage alone could have been offset by the corresponding drop in marsh elevation, which would favor *S. alterniflora* over *S. patens* (Watson et al. 2016). The muted vegetation responses could also have been a consequence of the final placement and density of new creeks in the experimental marsh. Logistical issues (including soft soils) prevented the excavator from digging a new north-to-south creek through the large center section of the experimental marsh as planned, thereby ensuring that it remained relatively isolated from creeks even after drainage enhancement.

We did observe clear trends in the cover and height of some target species, but nearly identical trends were also seen in the control marsh, again indicating no effects from drainage enhancement. Instead, it could signal that vegetation patterns in the experimental marsh were influenced more by larger-scale factors, perhaps SLR, which is known to elicit declines in vegetation cover and height and increase bare ground (Watson et al. 2016, Raposa et al. 2017b), all of which were observed in both marshes in our study. Clearly, a universal and consistent response of vegetation to drainage enhancement does not exist; instead net responses are the result of complex interactions of many factors. Our study provides an example of vegetation remaining unaffected by drainage enhancement over the short-term at the scale of the entire marsh; future monitoring would help detect longer-term changes that are due to drainage enhancement and independent of ongoing large-scale regional drivers.

Similar to vegetation, bird responses to drainage enhancement techniques varies widely, including increased diversity and shorebird and waterfowl abundance from IMM (Rochlin et al. 2012), community composition shifts via the loss of surface pools from ditching (Clarke et al. 1984, Wilson et al. 1987), and minimal or potentially negative responses to OWMW (Pepper and Shriver 2010, James-Pirri et al. 2012). In our study, bird community composition was unaffected by drainage enhancement in the short-term. We suspect this is primarily due to the lack of natural pools in the experimental marsh at any point during the study, which limited the number of wading and shore birds and resulted in a simple community comprised mostly of small passerines that frequent ecotone habitats. Marsh sparrow density also did not change in our study, likely due to no concurrent change in high marsh habitat, which is used for nesting by these species *(*Marshall and Reinert 1990, Greenlaw and Rising 1994). In contrast, Red-winged Blackbird density declined, and the collective density of some insectivorous and frugivorous birds that forage along marsh edges (e.g, Tree Swallow, Cedar Waxwing, Flycatchers, and Song Sparrows) was higher in the experimental marsh after drainage enhancement without simultaneous changes in the control marsh. For Red-winged Blackbird, this too was not likely a response to drainage enhancement but rather to treatment and removal of *Phragmites*, which is a favored nesting habitat for this species (Benoit and Askins 1999). Drainage enhancement may have contributed to the higher density of the other marsh edge species (possibly via improved production of ecotone shrubs with more drainage), but even this is confounded by *Phragmites* removal, which led to ecotone shrub colonization in former *Phragmites* areas. However, from a management perspective, these collective results indicate that in some cases drainage enhancement does not adversely affect birds, at least in the short-term, especially when natural pools are absent before project implementation.

**Unanticipated effects of excavations**. One unanticipated biological response to our project was an apparent dramatic increase in marsh crab abundance after drainage enhancement (the Atlantic salt marsh fiddler *Uca pugnax* was largely absent from the experimental marsh before the project, but later was conspicuously abundant along the edges of newly excavated creeks; Raposa pers. obs.). This finding is anecdotal because we did not directly monitor crabs, but indirect evidence comes from the change in the extent of bare creekbanks before and after drainage enhancement. This habitat, which is often caused by crab herbivory and/or burrowing (e.g., Holdredge et al., 2009; Coverdale et al. 2012; Wilson et al. 2012), was absent from the experimental marsh before the project began, but increased rapidly after. Crabs and their impacts are typically concentrated along or near creekbanks (Wasson et al. in press) and have been shown to increase in association with the creation of ditches and runnels (Lesser et al. 1976, Breitfuss et al. 2004; Dale 2005). It follows then that any project designed to alleviate waterlogging by expanding the network of drainage channels in a marsh will also allow crabs greater access to that marsh. In regions such as southern New England, where impacts to marshes from crabs are intense and pervasive (Wasson et al. in press, Coverdale et al. 2013b), any positive effects from drainage enhancement may be at least partially mitigated by negative impacts from crabs. Possible linkages between drainage enhancement and increasing crab abundance is an area ripe for additional research, including comparisons among different types of drainage channels, vegetation types, and degrees of waterlogging.

**Perspectives from other projects***.* Our study provides results from just one study; additional results, sometimes unanticipated and largely unpublished, are now emerging from other drainage enhancement projects across southern New England. For example (Ron). Combined, these projects paint a picture of current drainage enhancement activities eliciting a complex set of effects on marshes. In some cases, benefits can be realized, and marsh resilience to SLR improved, but unanticipated results in other cases reinforces the need for caution and careful planning before any drainage enhancement project is initiated. And it again highlights the need for additional controlled experiments centered around ongoing drainage enhancement projects.

**Summary and recommendations**. Understanding the core effects of drainage enhancement requires, at a minimum, monitoring water levels and marsh elevations. But what is the best way to do that, and should other parameters also be considered? From our study, we recommend broader spatial coverage for elevation and water level monitoring beyond just along transects or at a few locations within a marsh. Using just one water level logger or tide station with a known elevation should be adequate for calculating surface inundation, but more loggers should be deployed within a marsh (using a method similar to our project) to get a better understanding of drainage across a broader scale. Perhaps loggers deployed evenly across an entire marsh, or at regular intervals from new drainage channels, would be appropriate. Comprehensive water level data should also be collected prior to any project to guide the overall design and proper placement of drainage channels. For elevation data collected in the field, points collected using a dense grid network (e.g., Neil et al. 2017) would allow researchers to examine elevation change over time across an entire marsh to identify areas impacted by drainage enhancement and where further interventions might be necessary. Projects should also monitor key vegetation parameters, which we recommend augmenting with additional monitoring of crab abundance and/or impacts, especially in areas where crabs are a concern.

In summary, our project provides a case study to help researchers and managers better understand potential marsh responses to drainage enhancement. It also provides cautionary evidence of impacts to elevation and possible confounding effects from crabs and large-scale drivers of change. In some cases, drainage enhancement as described here bay be effective for alleviating waterlogging and building resilience, but in severely degraded marshes it may lead to drops in elevation of a magnitude that could render revegetation unsustainable over the long-term. We advocate for additional case studies by allocating resources for building before and after monitoring programs into future drainage enhancement projects.

**Acknowledgements**

Christopher Haight and Roger Wolfe for providing information on projects included in Table 1; all students and interns for help with field work; CRMC and NERRS for funding.

**Literature Cited**

Adamowicz, S.C. and C.T. Roman. 2005. New England salt marsh pools: a quantitative analysis of geomorphic and geographic features. Wetlands 25:279-288.

Anisfeld, S.C., T.D. Hill, and D.R. Cahoon. 2016. Elevation dynamics in a restored versus a submerging salt marsh in Long Island Sound. Estuarine, Coastal and Shelf Science 170:145-154.

Benoit, L.K. and R.A. Askins. 1999. Impact of the spread of *Phragmites* on the distribution of birds in Connecticut tidal marshes. Wetlands 19:194–208.

Bradley, P.M. and J.T. Morris. 1990. Physical characteristics of salt marsh sediments: ecological implications. Marine Ecology Progress Series 61:245-252.

Breitfuss, M.J., R.M. Connolly, P.E.R. Dale. 2004. Densities and aperture sizes of burrows constructed by *Helograpsus haswellianus* (Decapoda: Varunidae) in saltmarshes with and without mosquito control runnels. Wetlands 24:14-22.

Britton, W.E. 1912. The mosquito plague of the Connecticut coast region and how to control it. Connecticut Agricultural Experiment Station Bulletin 173:3-14.

Bourn, W.S., and C. Cottam. 1950. Some biological effects of ditching tidewater marshes. Research Report 19, U.S. Fish and Wildlife Service, Washington, DC.

Cahoon, D.R., B.C. Perez, B.D Segura, and J.C. Lynch. 2011. Elevation trends and shrink-swell response of wetland soils to flooding and drying. Estuarine, Coastal and Shelf Science 91:463-474.

Cahoon, D.R., J.C. Lynch, C.T. Roman, J.P. Schmidt, and D.E. Skidds. 2019. Evaluating the relationship among wetland vertical development, elevation capital, sea-level rise, and tidal marsh sustainability. Estuaries and Coasts 42:1-15.

Carey, J.C., S.B. Moran, R.P. Kelly, A.S. Kolker, and R.W. Fulweiler. 2017. The declining role of organic matter in New England salt marshes. Estuaries and Coasts 40:626-639.

Clarke, J.A., B.A. Harrington, T. Hruby, and F.E. Wasserman. 1984. The effect of ditching for mosquito-control on salt-marsh use by birds in Rowley, Massachusetts*.* Journal of Field Ornithology 55:160–180.

Cole Ekberg M.L., K.B. Raposa, W.S. Ferguson, K. Ruddock, and E.B. Watson. 2017. Development and application of a method to identify salt marsh vulnerability to sea level rise. Estuaries and Coasts 40:694-710.

Corman, S.S., C.T. Roman, J.W. King, and P.G. Appleby. 2012. Salt marsh mosquito-control ditches: sedimentation, landscape change, and restoration implications. Journal of Coastal Research28:874-880.

Coverdale, T.C., A.H. Altieri, and M.D. Bertness. 2012. Belowground herbivory increases vulnerability of New England salt marshes to die-off. Ecology 93:2085-2094.

Coverdale, T.C., N.C. Herrmann, A.H. Altieri, and M.D. Bertness. 2013a. Latent impacts: the role of historical human activity in coastal habitat loss. Frontiers in Ecology and the Environment 11:69-74.

Coverdale, T.C., M.D. Bertness, and A.H. Altieri. 2013b. Regional ontogeny of New England salt marsh die-off. Conservation Biology27:1041-1048.

Crain, C.M., K.B. Gedan, and M. Dionne. 2009. Hydrologic alteration of New England tidal marshes by mosquito ditching and tidal restriction. *In* Human impacts on salt marshes: A global perspective(B.R. Silliman, T. Grosholtz, and M.D. Bertness eds.), pp. 149–169. Berkley: University of California Press.

Croft, A.L., L.A. Leonard, T.D. Alphin, L.B. Cahoon, and M.H. Posey. 2006. The effects of thin layer sand renourishment on tidal marsh processes: Masonboro Island, North Carolina. Estuaries and Coasts 29:737-750.

Crotty, S.M., C. Angelini, and M.D. Bertness. Multiple stressors and the potential for synergistic loss of New England salt marshes. PLoS ONE 12(8):e0183058

Dale, P. 2005. Long term impacts of runnelling on an intertidal saltmarsh. Arbovirus Research in Australis 9:80-85

Dale, P.E.R. and K. Hulsman. 1990. Critical review of salt marsh management methods for mosquito control. Reviews in Aquatic Science 3:281-311.

Dale, P.E.R. and J.M. Knight. 2006. Managing salt marshes for mosquito control: impacts of runnelling, open marsh water management and grid-ditching in sub-tropical Australia. Wetlands Ecology and Management 14:211-220.

Dale, P.E.R., P.T. Dale, K. Hulsman, and B.H. Kay. 1993. Runnelling to control saltmarsh mosquitoes: long-term efficacy and environmental impacts. Journal of the American Mosquito Control Association 2:174–181.

Ewanchuk, P.J. and M.D. Bertness. 2004. The role of waterlogging in maintaining forb pannes in northern New England salt marshes. Ecology 85:1568–1574.

Greenlaw, J.S. and J.D. Rising. 1994. Sharp-tailed sparrow, Ammodramus caudacutus. In The Birds of North America, No. 112 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, and The American Ornithologists' Union, Washington, D.C.

Hill, T.D. and S.C. Anisfeld. 2015. Coastal wetland response to sea level rise in Connecticut and New York. Estuarine, Coastal and Shelf Science 163B:185-193.

Holdredge C., M.D. Bertness, and A.H. Altieri. 2009. [Role of crab herbivory in die-off of New England salt marshes](https://doi.org/10.1111%2Fj.1523-1739.2008.01137.x). Conservation Biology 23:672-679.

Howes, B.L. and D.D. Goehringer. 1994. Porewater drainage and dissolved organic carbon and nutrient loss through the intertidal creek banks of a New England salt marsh. Marone Ecology Progress Series 114:289-301.

James-Pirri M.J., R.M. Erwin, D.J. Prosser, and J.D. Taylor. 2012. Responses of salt marsh ecosystems to mosquito control management practices along the Atlantic Coast (U.S.A.). Restoration Ecology 20:395-404.

Kirwan, M.L., S. Temmerman, E.E. Skeehan, G.R. Guntenspergen, and S. Fagherazzi. 2016. Overestimation of marsh vulnerability to sea level rise. Nature Climate Change 6:253-260.

Konisky, R.A., D.M. Burdick, M. Dionne, and H.A. Neckles. 2006. A regional assessment of salt marsh restoration and monitoring in the Gulf of Maine. Restoration Ecology 14:516-525.

LeMay, L.E. 2007. The impact of drainage ditches on salt marsh flow patterns, sedimentation and morphology: Rowley River, Massachusetts. Master's thesis, College of William and Mary, Williamsburg, Virginia. 230 pp.

Lesser, C.R. 1982. A study of the effects of three mosquito control marsh management techniques on selected parameters of the ecology of a Chesapeake Bay tidewater marsh in Maryland. Maryland Department of Natural Resources, Annapolis, MD. 116 pp.

Lesser, C.R., F.J. Murphy, and R.W. Lake. 1976. Some effects of grid system mosquito control ditching on salt marsh biota in Delaware. Mosquito News 36:69-77.

Marshall, R.M. and S.E. Reinert. 1990.  Breeding ecology of seaside sparrows in a Massachusetts salt marsh. Wilson Bull. 102:501-513.

Montalto, F.A., T.S. Steenhuis, and J-Y. Parlange. 2006. The hydrology of Piermont marsh, a reference for tidal marsh restoration in the Hudson river estuary, New York. Journal of Hydrology 316:108-128.

Moore, K. 2013. Draft revised NERRS SWMP vegetation monitoring protocol: long-term monitoring of estuarine submersed and emergent vegetation communities. National Estuarine Research Reserve System Technical Report. 25 pp.

Neil, A., S. Rasmussen, M. Bradley, and S. Stevens. 2017. On-the-ground collection of high resolution elevation data in salt marsh environments: A Northeast Coastal and Barrier Network methods document. Natural Resource Report NPS/NCBN/NRR—2017/1371. National Park Service, Fort Collins, Colorado.

Nuttle, W.K. and H.F. Hemond. 1988. Salt marsh hydrology: implications for biogeochemical fluxes to the atmosphere and estuaries. Global Biogeochemical Cycles 2:91-114.

Nuttle, W.K., H.F. Hemond, and K.D. Stolzenbach. 1990. Mechanisms of water storage in salt marsh sediments: the importance of dilation. Hydrological Processes 4:1-13.

Pepper, M.A., and W.G. Shriver. 2010. Effects of open marsh water management on the reproductive success and nesting ecology of seaside sparrows in tidal marshes. Waterbirds 33:381–388.

Raposa, K.B., K. Wasson, E. Smith, J.A. Crooks, P. Delgado, S.H. Fernald, M.C. Ferner, A. Helms, L.A. Hice, J.W. Mora, B. Puckett, D. Sanger, S. Shull, L. Spurrier, R. Stevens, and S. Lerberg. 2016. Assessing tidal marsh resilience to sea-level rise at broad geographic scales with multi-metric indices*.* Biological Conservation 204:263-275.

R Core Team. 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: <https://www.R-project.org/>.

Raposa, K.B., M.L. Cole Ekberg, D.M. Burdick, N.T. Ernst, and S.C. Adamowicz. 2017a. Elevation change and the vulnerability of Rhode Island (USA) salt marshes to sea-level rise. Regional Environmental Change 17:389-397.

Raposa, K.B., R.L.J. Weber, M.C. Ekberg, and W. Ferguson. 2017b. Vegetation dynamics in Rhode Island salt marshes during a period of accelerating sea level rise and extreme sea level events. Estuaries and Coasts 40:640-650.

Raposa, K.B., R.A. McKinney, C. Wigand, J.W. Hollister, C. Lovall, K. Szura, J.A. Gurak, Jr., J. McNamee, C. Raithel, and E.B. Watson. 2018a. Top-down and bottom-up controls on southern New England salt marsh crab populations. PeerJ 6:e4876.

Raposa, K.B., S. Lerberg, C. Cornu, J. Fear, N. Garfield, C. Peter, R.L.J. Weber, G. Moore, D. Burdick, and M. Dionne. 2018b. Evaluating tidal wetland restoration performance using National Estuarine Research Reserve System reference sites and the restoration performance index (RPI). Estuaries and Coasts 41:36-51.

Rochlin, I., M.J. James-Pirri, S.C. Adamowicz, M.E. Dempsey, T. Iwanejko, and D.V. Ninivaggi. 2012. The effects of integrated marsh management (IMM) on salt marsh vegetation, nekton, and birds. Estuaries and Coasts 35:727-742.

Roman, C.T., M.J. James-Pirri and J.F. Heltshe. 2001. Monitoring salt marsh vegetation. Long-term Coastal Ecosystem Monitoring Program, Cape Cod National Seashore, Wellfleet, MA. 47 pp.

Rudis, B. 2019. hrbrthemes: Additional Themes, Theme Components and Utilities for 'ggplot2'. R package version 0.6.0. https://CRAN.R-project.org/package=hrbrthemes

Schrift, A.M., I.A. Mendelssohn, and M.D. Materne. 2008. Salt marsh restoration with sediment-slurry amendments following a drought-induced large-scale disturbance. Wetlands 28:1070-1085.

Stagg, C.L. and I.A. Mendelssohn. 2010. Restoring ecological function to a submerged salt marsh. Restoration Ecology 18:10-17.

Stewart-Oaten, A., W.M. Murdoch, and K.R. Parker. 1986. Environmental impact assessment: “pseudoreplication” in time? Ecology 67:929-940.

Taylor, N. 1938. A preliminary report on the salt marsh vegetation of Long Island. New York State Bulletin 316.

Tonjes, D.J. 2013. Impacts from ditching salt marshes in the mid-Atlantic and northeastern United States. Environmental Reviews 21:116-126.

US Geological Survey. 2012. Bird area survey standard operating procedures. Unpublished protocols. USGS, Western Ecological Research Center, San Francisco Bay Estuary Field Station, Vallejo, CA.

Vincent, R.E., D.M. Burdick, and M. Dionne. 2013. Ditching and ditch-plugging in New England salt marshes: effects on hydrology, elevation, and soil characteristics. Estuaries and Coasts 36:610-625.

Warren, R.S. and W.A. Niering. 1993. Vegetation change on a northeast tidal marsh: interaction of sea-level rise and marsh accretion. Ecology 74:96-103.

Wasson, K., K. Raposa, M. Almeida, K. Beheshti, J.A. Crooks, A. Deck, N. Dix, C. Garvey, J. Goldstein, D.S. Johnson, S. Lerberg, P. Marcum, C. Peter, B. Puckett, J. Schmitt, E. Smith, K. St. Laurent, K. Swanson, M. Tyrrell, and R. Guy. In Press.Pattern and scale: evaluating generalities in crab distributions and marsh dynamics from small plots to a national scale. Ecology.

Watson, E.B., K. Szura, C. Wigand, K.B. Raposa, K. Blount, and M. Cencer. 2016. Sea level rise, drought and the decline of *Spartina patens* in New England marshes. Biological Conservation 196:173-181.

Watson, E.B., C. Wigand, E.W. Davey, H.M. Andrews, J. Bishop, and K.B. Raposa. 2017. Wetland loss patterns and inundation-productivity relationships prognosticate widespread salt marsh loss for southern New England. Estuaries and Coasts 40:662-681.

Weston, N.B. 2014. Declining sediments and rising seas: an unfortunate convergence for tidal wetlands. Estuaries and Coasts 37:1-23.

Wickham, H. 2016. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York.

Wickham, H. 2019. stringr: Simple, Consistent Wrappers for Common String Operations. R package version 1.4.0. https://CRAN.R-project.org/package=stringr

Wickham, H., and J. Bryan. 2019. readxl: Read Excel Files. R package version 1.3.1. https://CRAN.R-project.org/package=readxl

Wickham, H., and L. Henry. 2019. tidyr: Easily Tidy Data with 'spread()' and 'gather()' Functions. R package version 0.8.3. https://CRAN.R-project.org/package=tidyr

Wickham, H., R. François, L. Henry and K. Müller. 2019. dplyr: A Grammar of Data Manipulation. R package version 0.8.0.1. https://CRAN.R-project.org/package=dplyr

Wickham, H., J. Hester and R. Francois. 2018. readr: Read Rectangular Text Data. R package version 1.3.1. https://CRAN.R-project.org/package=readr

Wigand C., T. Ardito, C. Chaffee, W. Ferguson, S. Paton, K. Raposa, C. Vandemoer, and E. Watson. 2017. [A climate change adaptation strategy for management of coastal marsh systems](https://doi.org/10.1007%2Fs12237-015-0003-y). Estuaries and Coasts 40:682-693.

Wilke, C.O. 2019. cowplot: Streamlined Plot Theme and Plot Annotations for 'ggplot2'. R package version 0.9.4. https://CRAN.R-project.org/package=cowplot

Wilson, C.A., Z.J. Hughes, and D.M. Fitzgerald. 2012. The effects of crab bioturbation on mid-Atlantic saltmarsh creek extension: geotechnical and geochemical changes. Estuarine, Coastal and Shelf Science106:33-44.

A close up of a map

Description generated with very high confidence

Figure 1. Maps of the Coggeshall Marsh study sites. Upper locus map shows location of marshes on Prudence Island RI; lower locus map shows experimental (red outline) and control (white outline) sections of Coggeshall Marsh. Aerial photographs are from 2012 (upper panel) and 2016 (lower panel) and show changes in tidal creeks before and after drainage enhancement. Diagonal hashes indicate areas of invasive *P. australis*.



Figure 2. Hydrology in the experimental and control marshes before (2013) and after (2014-2016) drainage enhancement. Top row: percent of the time the marsh surface was inundated at each station. Bottom row: mean minimum daily water level relative to the marsh surface at each station. Note the large changes in both hydrology parameters at stations 3 and 4 in the experimental marsh after drainage enhancement. All data were collected every 15 minutes for approximately 2 months during summer each year. Gaps in experimental and control marsh station 1 data are due to logger failure during deployment. Error bars are 1 SE.



Fig. 3. Elevations of vegetation monitoring plots. A. Annual mean elevation across all plots in the experimental marsh. B. Annual mean elevation across all plots in the control marsh. C. Mean change in elevation from 2013 to 2016 in experimental marsh plots dominated by different vegetation types (High marsh = high marsh vegetation only, Mix = mixture of high marsh and *S. alterniflora*, *S. alt*/bare = any combination of *S. alterniflora* and bare ground. Error bars are 1 SE.

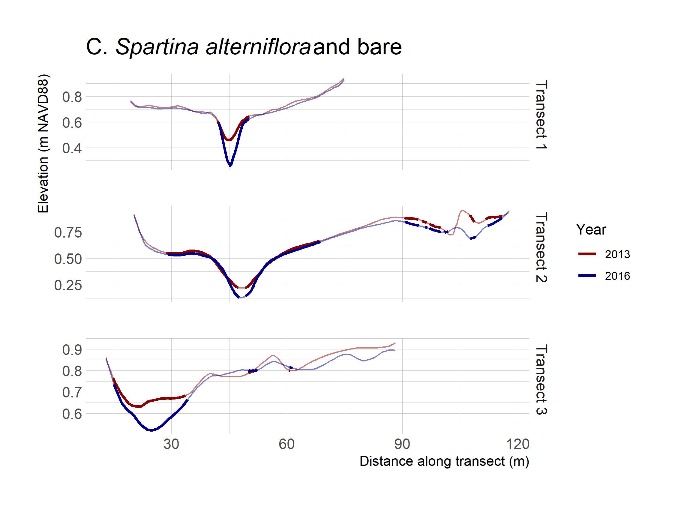
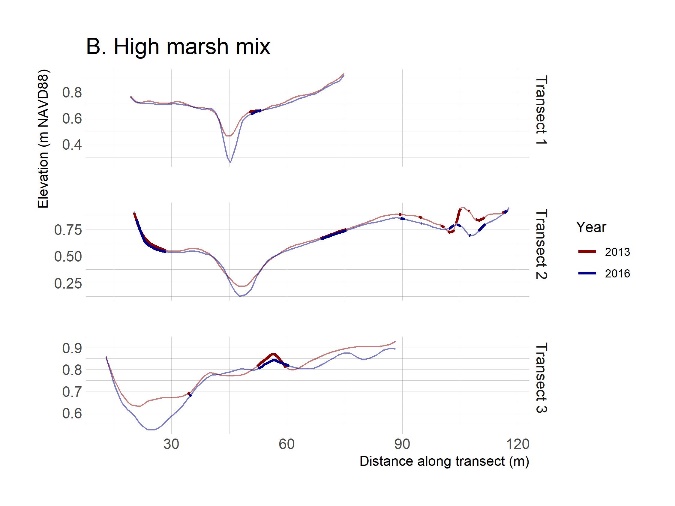
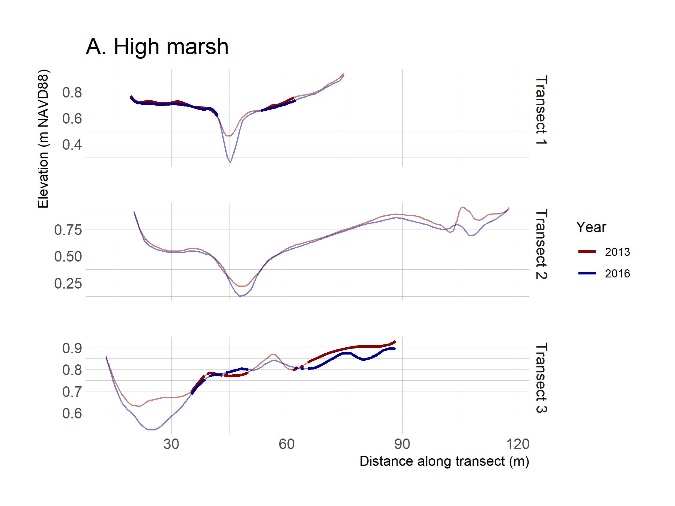
 

Figure 4. Elevation profiles along the three transects in the experimental marsh, showing data before drainage enhancement (2013) and three years after (2016). Elevation profiles were smoothed using a loess regression with a smoothing parameter of 0.12. Transects run across the entire marsh platform, from the seaward to landward marsh edges, left to right. In each plot, entire transects are shown, with thicker lines highlighting focal habitats (A. high marsh; B. high marsh mixed with *Spartina alterniflora*; C. *Spartina alterniflora* and/or bare ground).



Figure 5. Mean marsh platform soil bearing capacity in the experimental and control areas of Coggeshall Marsh over time, before (2013) and after (2014-2016) drainage enhancement. Linear trendlines and error bars (1 SE) are shown.



Figure 6. Trends in mean percent cover of *S. patens*, *S. alterniflora*, *D. spicata* and bare ground in the experimental and control marshes over time, before (2013) and after (2014-2016) drainage enhancement. Linear trendlines and error bars (1 SE) are shown.



Figure 7. Mean height of dominant vegetation species in the experimental and control marshes over time, before (2013) and after (2014-2016) drainage enhancement. Linear trendlines and error bars (1 SE) are shown.

Table 1. Drainage enhancement projects across the southern New England/Long Island Sound region between 2009-2018. Projects are grouped by State. Area is of the immediate enhancement work, not of each entire marsh. Years are when drainage features were excavated. Features types include sinuous naturalized creeks, linear ditches and shallow runnels, excavated either by hand or with an excavator. During this time, no drainage enhancement projects were conducted along the north shore of Long Island, NY.



Table 2. Relative composition of habitat types in the experimental and control marshes before (2013) and after (2014-2016) drainage enhancement. Each value represents the percent composition of each habitat for each marsh, each year, and is calculated by summing the extent in meters of each habitat across three transects, dividing by the total combined length of the transects, and multiplying by 100. Data are sorted in descending order by experimental mean. Refer to Supplemental Table 1 for habitat definitions.



Table 3. Annual mean percent cover of vegetation in the experimental marsh (n=20 plots) and control marsh (n=21 plots) from point-intercept sampling. Experimental and control means are across all four sampling years within each marsh.



Table 4. Annual mean bird density (no. ha-1) in the experimental and control marshes. Annual means are calculated sequentially across 1) all stations on each date, and 2) all dates; means for each marsh are further averaged across all years.



Supplemental Table 1. Definitions of habitat types that were monitored with the RTK-GPS during this project.

