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Evaluation of Drainage Enhancement for Vegetation Recovery in New England Salt Marshes Using Public Domain, High-Resolution Aerial Imagery

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ABSTRACT

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Paired stressors of sea-level rise and abandoned ditches and embankments from historic farming practices have exacerbated waterlogging and accelerated replacement of valuable interior high marsh with large pools throughout the United States Atlantic seaboard. High marsh loss has contributed to substantial population declines and the threat of future extinction of the Saltmarsh Sparrow (*Ammodramus caudacutus*), an endemic species of coastal wetlands. Creation of runnels and selective ditch maintenance has been promoted as short- and medium-term solutions to conserve and restore high marsh habitat and restore natural single-channel hydrology. A comprehensive monitoring program was launched in 2020 to evaluate the effect of runnels and maintenance of selective ditches on the hydrology, vegetation, and elevation of interior marshes across 17 marshes of Maine, Massachusetts, and Rhode Island, with the explicit goal of habitat conservation for the Saltmarsh Sparrow. The marsh surface was classified from 2010–21 with public aerial imagery to document the change in aerial extent of the vegetated marsh surface and unvegetated:vegetated ratio of tidal watersheds (mean size = 2.12 ± 0.18 ha) associated with specific management actions: runnelling, reference healthy marshes, and no-action pannes and pools. Runnels reversed the expansion of pools and pannes with annual declines of −0.037 unvegetated:vegetated ratio and gains of 1.55% vegetated area. Tidal watersheds gained an overall net 2.08 ha vegetated surface post-restoration, despite continued losses in reference and no-action tidal watersheds. Re-establishing hydrologic paths to allow regular tidal flooding and drainage promotes revegetation of shallow waterlogged pools—a first step toward rebuilding marsh elevation and conserving habitat for saltmarsh sparrows.

ADDITIONAL INDEX WORDS: *Hydrologic restoration, ditch plug, runnel, UVVR, tidesheds, high marsh.*

INTRODUCTION

Recent losses of salt marshes throughout the Eastern United States as a result of lateral shoreline erosion (Burns, Alexander, and Alber, 2020) and the inability of the systems to maintain elevation with sea-level rise is well documented (Crosby *et al.*, 2016; Gedan, Altieri, and Bertness, 2011; Hartig *et al.*, 2002). One of the main mechanisms of large-scale marsh degradation is the conversion of interior high marsh meadows (*i.e. Spartina patens* and *Distichlis spicata*) to short-form *Spartina alterniflora* pannes and unvegetated pools from excessive flooding and lack of drainage (Burns, Alber, and Alexander, 2021; Raposa *et al.*, 2017; Warren and Niering, 1993). In New England, increased flooding frequency and duration from sea-level rise is further exacerbated by the presence of historic agriculture relicts such as embankments and farming ditches and more recent 20th century mosquito management (Adamowicz *et al.*, 2020; Vincent, Burdick, and Dionne, 2013). Embankments on the landscape prevent full drainage of the marsh interior between tidal cycles, increase anoxia and sulfide stress on high marsh graminoids (Mora and Burdick, 2013b), and have been directly attributed to the conversion of interior marsh habitat to bare pannes or "waffle" pools (Adamowicz *et al.,* 2020; Mora and Burdick, 2013a; Smith *et al.*, 2021). Such hydrologic impairments decouple the natural negative feedback loop that maintains marsh elevation with sea-level rise (Cahoon *et al.*, 2019) and create conditions for vegetation dieback and pool formation (Kirwan and Guntenspergen, 2012; Vincent, Burdick, and Dionne, 2013, 2014). Over time, sea-level rise combined with the loss of precolonial hydrology has resulted in large, waterlogged basins, which lead to vegetation dieback and pool formation and expansion. Endemic marsh avian species, including *Ammodramus caudacutus* (Saltmarsh Sparrow), are increasingly threatened with the loss of nesting and foraging habitat (Gjerdum, Elphick, and Rubega, 2005; Roberts

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et al., 2019; Shriver *et al.*, 2016). Recent annual population declines of 9% across its Atlantic coast range make extinction of *A. caudacutus* possible by 2060 (Correll *et al.*, 2017).

To conserve and improve the habitat quality of interior marshes, coastal ecologists have implemented a strategy to restore surface hydrology over the past decade in New England (Adamowicz *et al.*, 2020). One technique to achieve regular tidal flooding and drainage is by using runnels (Wigand *et al.*, 2015). Runnels are designed as shallow swales (≤30 cm width and depth) that either meander through waterlogged interior marshes or connect to the nearest hydrologic channel (*e.g.*, a ditch, open hydrologic pathways through embankments) to address unstable, expanding pools. The restoration goal for runnels or similar drainage enhancements is improvement of surface drainage between tidal cycles and amelioration of biogeochemical stress for recovery of salt marsh vegetation (Wilson *et al.*, 2014). The eventual goal is the recovery of elevation and high marsh graminoid habitat; however, it may not be possible in microtidal systems or in subsidence basins too low in elevation. When coupled with an understanding of historic agricultural practices common in New England marshes (Adamowicz *et al.,* 2020; Smith, Hafner, and Niles, 2017), strategic runnelling can be effective at reducing waterlogging and building marsh capital (McKown *et al.*, 2023).

Preliminary studies of early runnel projects initiated in Rhode Island in the 2010s have shown success in reducing standing surface water between tidal cycles and lowering the groundwater table (Watson *et al.*, 2022) without over-aeration and oxidation of the peat substrate that could lead to marsh collapse (Perry, Ferguson, and Thornber, 2022; Raposa *et al.*, 2019). Revegetation of pannes and shallow pools with *S. alterniflora* and high marsh graminoids occurs within 5 years after an initial 1- to 2-year lag (Besterman *et al.*, 2022); however, the full re-colonization of high marsh graminoids may require rapid building of elevation for at least a decade (Wilson *et al.*, 2014). When applied to shallow, mega-pools in Massachusetts, runnels appear to enhance both drainage and vegetation cover of the surrounding high marsh platform (McKown *et al.*, 2023).

The rise of publicly available, high-resolution, multispectral imagery and accompanying statistical and classification software has allowed researchers to broaden ecological monitoring in temporal and spatial scales with limited personnel and funding (Haskins *et al.*, 2021; Shuman and Ambrose, 2003). From a restoration standpoint, historic imagery can document trends of pre-restoration conditions to better quantify rates of change and trajectories of habitat degradation compared with snapshots of 1 to 2 years with typical field-monitoring programs (Campbell *et al.*, 2017; Orth *et al.*, 2010). Remote sensing may be the most practical form of vegetation monitoring for large-scale natural events such as coastal flooding (Campbell and Wang, 2019) and climate change (Jorgenson *et al.*, 2018) or watershed-scale restoration efforts (Silverman *et al.*, 2019; Suir, Sasser, and Harris, 2020). The combination of remote sensing and field methods for post-restoration monitoring can address a broader suite of goals such as documenting landscape-scale vegetation contraction and expansion while describing species composition shifts or changes in the biogeochemical environment on plot-level scales (McKown *et al.*, 2021; Qi, MacGregor, and Gedan, 2020; Thomsen *et al.*, 2021).

Remote sensing has increasingly been adopted over the past two decades in tidal wetland research to document shifts in vegetation distribution and species composition (Campbell *et al.*, 2017; Schieder and Kirwan, 2019; Tuxen *et al.*, 2008), pool and creek geomorphology (Smith and Pellew, 2021; Wilson *et al.,* 2014), and sediment transport (Mariotti *et al.*, 2020; Moore *et al.*, 2021). Certain metrics have been developed to streamline and improve the evaluation of salt marsh health and vegetation condition. The normalized difference vegetation index (NDVI) or normalized difference water index are commonly used to monitor hydrology and vegetation of salt marsh and serve as foundations for more detailed cover classifications (Suir, Sasser, and Harris, 2020). For example, the unvegetated:vegetated ratio (UVVR) is a geospatially derived, pixel-based metric of the vegetation community on the tidal watershed scale that has been directly related to sediment budgets, marsh lifespans, and tipping points in the stability of the vegetation community (Ganju *et al.*, 2020; Ganju *et al.* 2017; Wasson *et al.*, 2019). Drainage enhancements projects are ideal for remote sensing monitoring and assessment with the UVVR because of the scale of hydrologic alterations at the smallest watershed and the need to minimize physical disturbance in unconsolidated soils of pannes for vegetation establishment. For example, Watson *et al.* (2022) documented the recovery of vegetation after runnel installation in Rhode Island for six growing seasons with color-infrared World View-2 satellite imagery.

In a multistate conservation and research plan for *A. caudacutus* by the Atlantic Coast Joint Venture (ACJV) and affiliated federal and state agencies and conservation organizations, runnels were identified as a priority tool to restore and sustain high marsh meadow habitat in the short to medium term (0–15 years; ACJV, 2019; Hartley and Weldon, 2020). Over the past decade, numerous runnel and other drainage enhancement projects (*i.e.* ditch plug removal, ditch maintenance) were implemented across New England for expressed goals of removal of excess surface water and vegetation recovery. However, the effectiveness of runnels for habitat recovery has been evaluated only in several site-specific studies and not across an entire region. A multistate Before-After-Control-Impact field monitoring program in New England for the ACJV is being conducted to better understand the effects of runnels on hydrology, vegetation, marsh elevation, and avian community abundance and composition.

This study presents the results of the geospatial analysis component, which applied image classification and the UVVR to analyze pre- and post-restoration vegetation trends on tidal watershed scales. The goals of the analysis were twofold: (1) evaluate the effect of runnels on vegetation recovery relative to reference and no-action controls and (2) determine how the condition of the marsh prior to restoration may affect the restoration trajectory.

METHODS

The following section describes the process of image classification and creation of the subtidesheds, which were used to

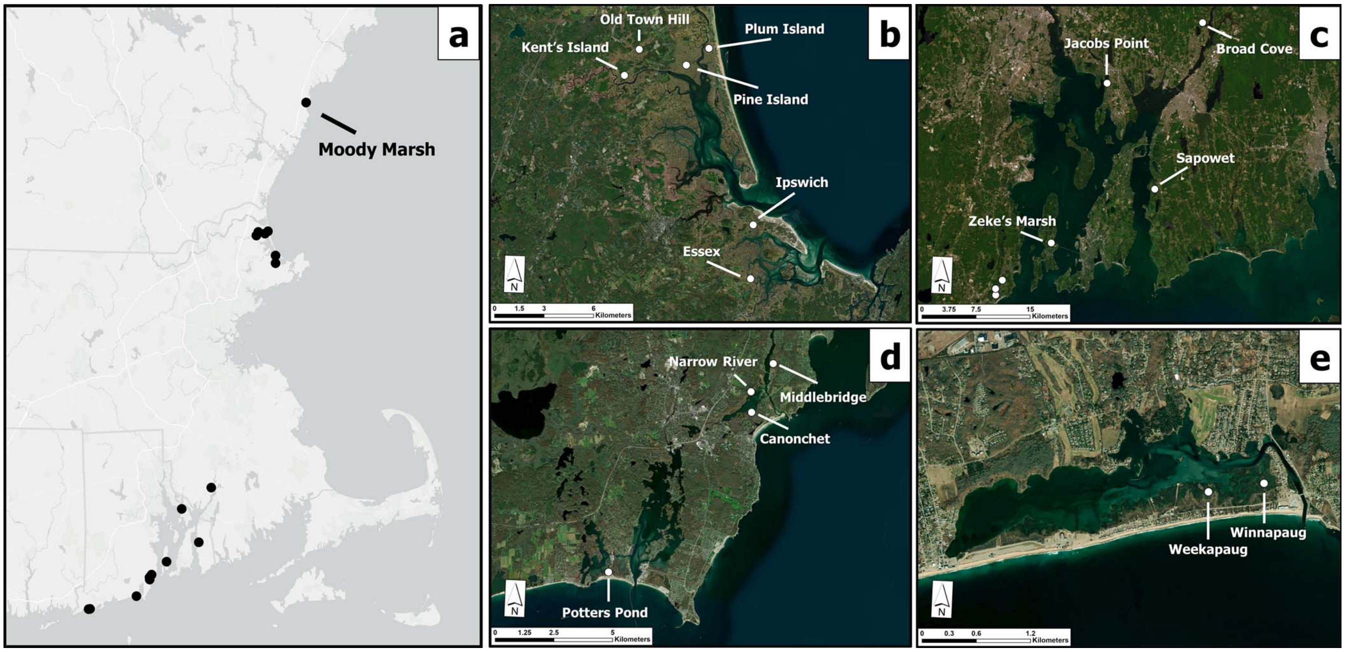

Figure 1. Site map of salt marsh sites with runnel restoration of geospatial analysis. (a) Distribution of sites throughout New England and insets of (b) north shore of Massachusetts, (c) northern Narragansett Bay, (d) southern Narragansett Bay, and (e) Winnapaug Pond.

calculate percent vegetation cover and UVVR for individual subtidesheds over the restoration timelines. Two linear mixed-spline models were then developed to evaluate the effect of drainage enhancement methods on the vegetation cover between treatments and marsh condition.

Study Area and Restoration History

Salt marshes with runnels or other drainage enhancements completed by spring 2023 were selected across New England (Figure 1), including Maine ($n = 1$), Massachusetts ($n = 8$), and Rhode Island ($n = 10$). Runnels were constructed as shallow swales (≤0 cm width, ≤30 cm depth) to improve drainage by either meandering through waterlogged pannes of short-form *S. alterniflora* and *Salicornia depressa* or serving as short connections from expanding shallow water pools to channels through ditch spoils or embankments (Besterman *et al.*, 2022: Winnapaug; McKown *et al.*, 2023: Pine Island; Perry, Ferguson, and Thornber, 2022: Narrow River; Watson *et al.*, 2022: Narrow River's Canonchet and Middlebridge sites). Runnels were constructed from 2010–23 either by low ground pressure excavator or by hand with shovels or post driver and angle iron (Table 1). At Moody Marsh, pools that formed behind intentionally installed ditch plugs were initially breached during coastal storm surge in 2010 and further improved in 2015. Plum Island–South had a series of drainage methods applied including ditch plug removal (2015–19), ditch maintenance (2015–19), and runnel construction (2019–21). At Old Town Hill, drainage was enhanced through the removal of sediment from filled-in ditches, which mimicked runnel implementation. Several large, connected pools slated for runnel installation at the southern end of Essex in 2022 were naturally breached in 2017. Because of the latitudinal extent from the Gulf of Maine to Narragansett Bay and Block Island Sound, sites varied in tidal range, salinity regime, distance upstream from estuary inlets, and surrounding landscape geomorphology and development.

Image Classification of Vegetated and Unvegetated Marsh Surface

Baseline data on marsh health before and after runnel construction was assessed using publicly available aerial imagery of the National Agriculture Imagery Program (NAIP; U.S. Department of Agriculture) from 2010–21. Aerial imagery (RBG, infrared) was acquired in the summer through early fall (July–October) with resolutions of 1 m (2010, 2012, 2014) and 0.6 m (2016, 2018, 2021; see Supplementary Material 1). The NDVI was calculated to provide a high-resolution measurement of plant health across the salt marsh surface (Figure 2). Vegetated and unvegetated areas (*i.e.* water, temporary wrack, bare mud, floating algae) were classified with vector support machine classification (supervised, pixel-based) using the Classification Wizard Tool in ArcGIS Pro (ESRI, Redlands, California). Pixel-based classification was conducted to adhere to the methodologies of Ganju *et al.* (2017) and Wasson *et al.* (2019) to better capture the matrix of scattered vegetation within shallow water of pannes. Training polygons were created for each year of each imagery raster in similar salt marshes outside of the study areas to account for the various imagery collection dates throughout the growing season. Depending on the availability of additional salt marsh within the imagery, roughly 10–20 training polygons were created for each classification group. The same training polygons were used for classification for the

Table 1. *Site characteristics for the study area, including runnel construction details, tideshed classification, and total marsh hectares analyzed. The number of tidesheds are reported for each treatment. Random and ditch remediation treatment tidesheds are not included in marsh size. Year 0 of the restoration timeline is assigned as the next full growing season post-restoration. Restoration timeline values of −10 to 0 years represent pre-restoration, and 0 to 8 years represent post-restoration. Tidal range represents the difference between mean higher high water levels and mean lower low water levels. Tidal range data is from the nearest NOAA tide gauges, Save The Bay tidal gauges, or unpublished water-level recorder data (Weekapaug and Winnapaug).*

Site	State	Drainage Type	Tidal Range (m)	Growing Season Post-restoration (Year 0)	Restoration Timeline (years)	Tidesheds (Run – Ref – NAC)	Marsh Size (ha)
Broad Cove	MA	Runnel	1.46	2017	−7 to 4	3 – 7 – 0	5.59
Essex	MA	Runnel	3.03	2017, 2022, 2023	−7 to 4, −10 to −1, −10 to −2	3 – 5 – 4	19.46
Ipswich	MA	Runnel	3.03	2023	−10 to −1	3 – 3 – 6	23.67
Kent's Island	MA	Runnel	2.89	2022, 2023	−10 to −1, −9 to −2	4 – 3 – 7	28.94
Old Town Hill	MA	Ditch maintenance	2.89	2020	−10 to 1	6 – 4 – 4	20.60
Pine Island	MA	Runnel	2.89	2015	−5 to 6	2 – 2 – 11	76.62
Plum Island–North	MA	Plug removal, Runnel	2.89	2015, 2017, 2019, 2020	−6 to 5, −7 to 4, −9 to 2, −9 to 1	5 – 4 – 1	73.90
Plum Island–South	MA	Ditch maintenance, Plug removal, Runnel	2.89	2019, 2022	−9 to 2, −10 to 0	7 – 3 – 1	43.80
Moody Marsh	ME	Plug removal	2.91	2011	0 to 11	1 – 1 – 8	20.59
Canonchet	RI	Runnel	0.47	2015	−5 to 6	2 – 1 – 1	7.54
Jacobs Point	RI	Runnel	0.86	2016	−5 to 6	4 – 4 – 2	9.20
Middlebridge	RI	Runnel	0.47	2016	−6 to 5	1 – 2 – 5	5.90
Narrow River	RI	Runnel	0.47	2016	−6 to 5	1 – 1 – 1	9.30
Potters Pond	RI	Runnel	0.32	2018	−8 to 3	3 – 1 – 2	3.03
Zeke's Marsh	RI	Runnel	0.96	2014	−4 to 7	4 – 1 – 1	7.79
Sapowet–North	RI	Runnel	1.20	2018	−8 to 3	2 – 1 – 1	12.22
Sapowet–South	RI	Runnel	1.20	2022	−10 to 0	1 – 1 – 2	13.45
Weekapaug Foundation	RI	Runnel	0.49	2017	−7 to 4	6 – 2 – 6	9.51
Winnapaug Town Land	RI	Runnel	0.49	2013	−3 to 8	10 – 2 – 9	7.87
					Total	70 – 48 – 70	398.98

Run = Runnel, Ref = Reference, and NAC = No Action

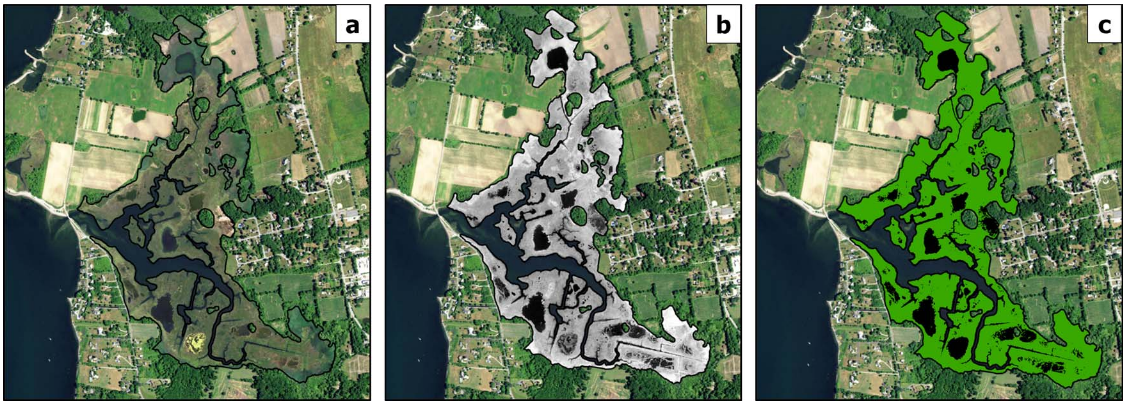

Figure 2. Image classification workflow to assess the unvegetated:vegetation ratio developed by Ganju *et al*. (2017), demonstrated for Sapowet Marsh site in 2016: (a) National Agriculture Imagery Program, (b) normalized difference vegetation index, (c) supervised pixel classification to unvegetated (black) or vegetated (green) marsh surface and manual reclassification of shadows and floating algae.

entire NAIP imagery collection with only minor edits if temporary changes in a given image would have mischaracterized the training polygon; the polygons typically included tree canopy shadows, wrack deposition, or floating green algae.

After classification, manual reinterpretation was completed to address three limitations of image classification of salt marshes: (1) tree canopy shadows, (2) floating green algae in pools, and (3) temporary wrack deposition. Shadows from the upland tree canopy absorb the near infrared band, decrease the NDVI value, and cause the marsh platform to be classified as unvegetated. Floating green algae mats on the surface of pools have similar NDVI signatures to marsh graminoids and are commonly misclassified as vegetated. Temporary wrack deposition from coastal storms is misclassified as unvegetated bare ground despite being vegetated later in the season because of the timing of the NAIP imagery. Manual reinterpretation was completed by consulting multiple other imagery datasets such as Google Earth and Leaf-Off imagery. It should be noted that the shape and size of tree shadows, floating green algae, and temporary wrack are easily identifiable in NAIP imagery. Existing primary tidal ditches (0.5–1 m width) were manually included for Essex, Ipswich, Kent's Island, Pine Island, Plum Island, and Old Town Hill as unvegetated into the imagery classification because of the prevalence of ditches on the landscape.

Spatial accuracy assessments were conducted for every year of each site based on 50–60 stratified random points between the two classification groups within each study area marsh. Points were visually verified based on Google Earth, Leaf-Off, and other imagery within 1 year of the NAIP imagery. Producer accuracy, User accuracy, and Kappa index of agreement were calculated to evaluate the vegetated–unvegetated classifications (Congalton, 1991). Producer accuracy is the probability a pixel was correctly classified. User accuracy is the probability a classified pixel actually represents the actual conditions of the image. Kappa index is the overall accuracy of the classification across all of the groups.

Tidal Watershed Delineation and Classification

Subtidal watersheds (henceforth tidesheds) within each salt marsh were manually drawn based on hydrology through the delineation of existing embankments, tidal creeks, and tidal ditches. Tidesheds serve as experimental replicates to represent both hydrologic units and effects of restoration activities. Watershed analysis was not used for tideshed delineation because public LIDAR data was too coarse in horizontal or vertical resolution (pixel size $\geq$ 0.5 m) to account for the influence of historic agricultural embankments on tidal hydrology. Tideshed delineations underwent multiple rounds of review and revision based on site visits, local knowledge from authors and stakeholders (Geoff Wilson, New England Wetland Restoration, and Nancy Pau, Parker River Wildlife Refuge), and surveys of historic agricultural embankments (Geoff Wilson). Tideshed boundaries were delineated with 2010 NAIP imagery and remained constant throughout the study (see Supplementary Material 2). Boundaries of the entire salt marsh sites were based on the National Wetland Inventory geospatial dataset and manually refined to the upland and shoreline extents of 2010 NAIP imagery (USFWS, 2023). Natural tidal channels and creeks greater than 10 m in width in 2010 were excluded from tideshed boundaries (Ganju *et al.*, 2017).

Tidesheds were classified at each site into treatments of runnel, ditch remediation, reference, no action, or "other" based on locations of restoration activities and vegetation community assessments (unpublished data; Table 1). Runnel tidesheds represent areas where regular flooding was restored through runnel creation, ditch plug removal, or ditch maintenance. Ditch remediation is a second tool used to restore single-channel hydrology, along with runnels, and addresses long-term elevation loss in overdrained areas by reducing the density of deep (>0.5 m) ditches (Burdick *et al.*, 2020). Reference tidesheds are marsh platforms that remained relatively unchanged over time and may serve as valuable sparrow nesting habitat. No-action tidesheds are degraded high marshes, including *S. alterniflora* and *Salicornia* pannes, pools, and interior mudflats. Other tidesheds comprise low marsh, upland edge, or stands of *Phragmites* *australis*, which are not representative of habitats targeted by runnel installation. Ditch remediation and other tidesheds were excluded from further data analysis because they were not the focus of this study.

Restoration Timeline and Vegetation Metrics

NAIP imagery of each tideshed was arranged along a restoration timeline ranging −10 to +8 years, whereas pre-restoration is defined from −10 to 0 years and post-restoration from 0 to 8 years. The near infrared component of NAIP images was not included until 2010, which equated to roughly −10 in the restoration timeline providing most sites with substantial pre-restoration monitoring. Year 0 was designated as the growing season (May–September) immediately after drainage enhancement activities because a demonstrated response in vegetation cover had not been found within the first growing season (Besterman *et al.*, 2022; McKown *et al.*, 2023). The timelines of reference and no-action tidesheds of a given marsh site were based on the timelines of the respective runnel tidesheds.

Restoration activities were completed in stages at Kent's Island, Essex, and Plum Island (Table 1). Runnel activities were staged in 2022 and 2023 for Kent's Island and Essex. A natural breach to a pool in 2017 at tideshed 7 in Essex led to drainage and revegetation and was considered a natural runnel project. Various drainage enhancement methods were conducted at Plum Island sites from 2015 to 2021, including runnelling and ditch plug removal. The restoration timeline of runnel tidesheds at these sites was assigned based on the timing of the specific drainage activities. At sites where activities were staggered over time between tidesheds (Table 1), reference and no-action tidesheds were grouped with the closest runnel tideshed (see Supplementary Material 3).

The UVVR and change in percentage of vegetated area were calculated for each tideshed to better understand the baseline effects of historic management actions and current sea-level rise on marsh stability. The UVVR is the ratio of the number of unvegetated pixels to the number of vegetated pixels:

$$UVVR = \frac{\sum Unvegetated\ Pixels}{\sum Vegetated\ Pixels} \tag{1}$$

The UVVR ranges from zero (fully vegetated tideshed) to near infinity for nearly unvegetated areas (Ganju *et al.*, 2017). The vegetated area for each tideshed was calculated based on the summation of vegetated pixels. The change in percentage of vegetated area was calculated over time for each tideshed based on the earliest date of the restoration timeline, providing a unitless description of progress, relatively irrespective of initial marsh degradation conditions:

$$Change\ in\ Vegetated\ Area\ (\%) = \frac{Vegetated\ Area_{Year\,n} - Vegetated\,Area_{Earliest\ Year}}{Tideshed\ Area} \tag{2}$$

where, Year n is the imagery analysis monitoring date along the restoration timeline and Earliest Year is the first date in the restoration timeline of a given tideshed.

Question 1: Impact of Runnels on Vegetation Cover (Treatment Model)

To estimate baseline vegetation conditions of the treatments before restoration, one-way mixed analysis of variances (random effect = site) with post-hoc Tukey's tests were conducted on the UVVR scores and percentage of vegetated area immediately before restoration (−2 to 0 years) between treatments (n = 188). From a remote sensing perspective, degraded tidesheds were defined as those with unvegetated area similar to no-action control sites, most likely associated with pannes and shallow pools on the landscape. Site name was included as a random intercept effect to consider the differences in tidal regime, site history, and broader environmental factors between marshes.

Linear mixed-spline models were fitted for UVVR score and the percentage of vegetated area over the restoration timeline (Brown, 2021; Harrison *et al.*, 2018; Zuur and Ieno, 2016). Maximum likelihood tests found the spline regression explained the variance in the model better than linear regressions for UVVR and percentage of vegetated area based on Akaike information criterion values (see Supplementary Material 3). A linear spline regression was selected over piece-wise linear regressions to force pre- and post-restoration regressions to the same intercept. The effect of treatment on the trajectory of UVVR and percentage of vegetated area at the tideshed scale was analyzed with an analysis of covariance (ANCOVA, referred to as treatment model; n = 1043), whereas timeline, tideshed treatment, and their interaction were included as fixed effects. The ANCOVA was computed to discern the effects of treatment and marsh condition on the trajectory of the vegetation cover. Tideshed was included as a random intercept effect in both models to account for the repeated measures of each tideshed over time. Tidesheds were uniquely labelled, removing the need for a nested structure in the models. The knot of the spline models was selected at year 0 to discern the effect of restoration activities. The north and south sites of Sapowet and Plum Island were aggregated, respectively, before analysis because they would experience similar environmental and tidal conditions (n = 17 sites).

The total change in vegetated area for each treatment was calculated for pre- and post-restoration to better understand the magnitude of loss and gains across the salt marsh systems. The baseline for changes for pre-restoration was at the earliest timeline year and for the post-restoration analysis the initial baseline was the latest date before restoration activities (−2 to 0 years). Losses and gains were calculated across site and the total study area of the project.

Question 2: Effect of Initial Conditions on Vegetation Recovery (Condition Model)

Exploratory analysis revealed a clear division between runnel tidesheds based on prerestoration marsh condition: Those with relatively high vegetation cover are best described as pannes of short-form *S. alterniflora* (*e.g.,* Jacob's Point, Broad Cove, Zeke's Marsh) *vs.* interior marshes with high bare and open water cover. To assess a possible different trajectory of runnel restoration on marsh condition, no-action control and runnel tidesheds were divided into degraded pannes (well-vegetated) and

Table 2. *Summary statistics of salt marsh health immediately prior to restoration activities (−2 to 0 years). Values reported as mean ± standard error. Lower case letters denote significant differences for comparisons of the post-hoc Tukey test (p < 0.05).*

Treatment	UVVR	Vegetated Area (%)
No action	0.29 ± 0.03^{bc}	79.9 ± 1.5^{b}
Reference	0.06 ± 0.01^{ab}	94.5 ± 0.8^{a}
Runnel	0.45 ± 0.10^{c}	77.3 ± 2.2^{b}

degraded pools (high bare cover) groupings based on the individual UVVR score immediately before restoration (−2 to 0 years; see Supplementary Material 1). Degraded pannes had UVVR scores of less than 0.13 based on the threshold of stability found by Ganju *et al*. (2017). Reference tidesheds were excluded from the marsh condition model because all tidesheds had UVVR values less than 0.13 immediately before restoration of respective runnel tidesheds. A linear spline-mixed ANCOVA model was applied to UVVR and percentage of vegetated area over the restoration timeline (referred to as condition model; $n = 773$) with timeline, tideshed treatment (runnel and no action only), condition (pannes and pools), and their interactions (*i.e.* three-way interaction) as fixed effects and site and tideshed as random effects.

Model Assumptions and R Packages

Tidesheds served as the experimental unit for all statistical analyses. All model assumptions were visually verified by plotting residuals *vs.* fitted values and each covariate in the model. Fixed effects of the mixed models were evaluated with the Satterthwaite approximations of degrees of freedom (Luke, 2017). Slopes were calculated for the linear and pre- and postrestoration of both spline models (treatment and condition model) using predicted values of the regressions. Mixed-effect models were created and evaluated using *lme4*, *afex*, and *splines2* packages; slopes were predicted using *ggpredict* and visualized with *ggplot2* and *patchwork* in R version 4.2.1 (Bates *et al.*, 2015; Lüdecke, 2018; Pedersen, 2022; R Core Team, 2023; Singmann *et al.*, 2016; Wang and Yan, 2021; Wickham, 2016). Tideshed boundary shapefiles, UVVR scores and vegetated area, post-classification data analysis R code, and project metadata can be accessed in Figshare, a public data repository (McKown *et al*., 2024).

RESULTS

In the following section, the accuracy and considerations with the imagery classification are highlighted. The two goals of the data analysis are addressed individually: (1) evaluate the effect of runnels on vegetation recovery relative to reference and no-action controls and (2) determine how the condition of the marsh prior to restoration may affect the restoration trajectory.

Tideshed Delineation and Classification Accuracy

Overall, 277 tidesheds were delineated with a mean area of 2.53 ± 0.16 ha across the 19 sites, comprising 700 ha of salt marsh (Table 1). The focus area of the three treatments included 188 tidesheds of 399 ha with a mean area of 2.12 ± 0.18 ha per tideshed, comprising 48 reference, 68 runnel, and 72 no-action tidesheds (110 ha, 130 ha, and 155 ha, respectively). Image classification created a dataset of 1043 individual UVVR scores for the three treatment tidesheds. Accurate image classification could not be confidently completed for Moody Marsh in 2021 because of poor NAIP image quality or for Potters Pond in 2010 because of the inability to distinguish between algae cover and salt marsh vegetation despite use of additional imagery. Spatial accuracy assessments demonstrated that 92% of imagery classifications were highly accurate (Kappa > 0.80) for each site and year. The only classification with a Kappa score less than 0.75 was Canonchet Marsh in 2010 (see Supplementary Material 4). Misclassifications of pixels were mostly found along creek and ditch edges and within shallow water pannes (*i.e.* distribution of small patches of vegetation within large expanses of water). Overall, imagery classification was successful in delineating pool areas and shallow water pannes on the marsh landscapes.

Question 1: Impact of Runnels on Salt Marsh Health (Treatment Model)

Salt marsh condition immediately before restoration activities was different across treatments for UVVR ($F_{2,180.4} = 6.32$, $p = 0.002$) and percentage of vegetated area ($F_{2,175.3} = 23.15$, $p < 0.001$). The runnel treatment tidesheds had greater unvegetated area than reference according to both metrics (UVVR: $p = 0.001$; vegetated area: $p < 0.001$) and similar to no action (UVVR: $p = 0.299$; vegetated area: $p = 0.536$). The runnel tidesheds had mean UVVR scores 0.39 greater than the references and 17% less vegetated area than references before runnel construction (Table 2). Across the study site, runnels were created in areas with high bare and open water cover similar to the no-action control tidesheds.

Treatment had an effect on the trajectory of the UVVR score ($F_{4,839.2} = 8.75$, $p < 0.001$) and percentage of vegetated area ($F_{4,838.1} = 39.82$, $p < 0.001$; Table 3) based on the significant interaction terms between time and treatment in the treatment model (see Supplementary Material 3 for mixed-model

Table 3. *Analysis of covariance (ANCOVA) of the treatment spline-mixed models for UVVR and vegetated area metrics across all three treatments with restoration timeline, treatment, and their interaction as fixed effects.*

Metric	Term	Numerator df	Denominator df	Sum of Squares	Mean Square	F	p
UVVR	Timeline	2	834.7	0.40	0.20	8.59	<0.001
	Treatment	2	241.6	0.25	0.12	5.23	0.006
	Timeline × treatment	4	839.2	0.82	0.21	8.75	<**0.001**
Vegetated area	Timeline	2	830.5	813.3	406.6	28.28	<0.001
	Treatment	2	233.7	592.5	296.3	20.60	<0.001
	Timeline × treatment	4	838.1	2290.8	572.7	39.82	<**0.001**

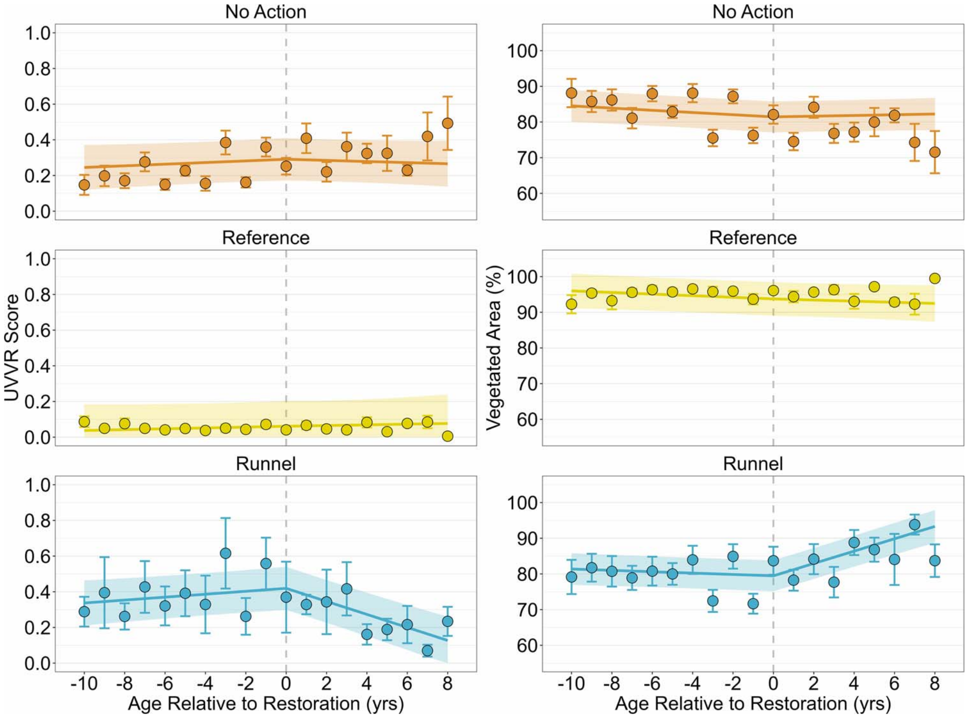

Figure 3. Spline-mixed treatment model of (left) mean UVVR score and (right) percentage of vegetated area across tidesheds over the restoration timeline. The mean and standard error for each year is provided with points and error bars. Confidence intervals (95%) represent the variance of the fixed effects with random effects held constant.

summaries). Before restoration, all treatments showed trends of marsh degradation based on both metrics (Figure 3, see Supplementary Material 3). The runnel tidesheds pre-restoration had the greatest annual increase of UVVR score at 0.008 y^{-1} yet the slowest loss of vegetated area at -0.19% y^{-1}. The trends reversed post-restoration at the runnel tidesheds with magnitudes greater than experienced before restoration. The UVVR score decreased 0.037 y^{-1}, and the vegetated area increased 1.73% y^{-1}. The trajectory of vegetation cover of the no-action control also reversed after runnel installation at -0.003 y^{-1} UVVR and 0.10% y^{-1}. Marsh degradation continued in the reference tidesheds yet at the slowest rate during post-restoration.

Vegetated salt marsh area declined pre-restoration for all treatments for a total loss of 8.36 ha or 2.3% of total marsh area across 188 tidesheds (Figure 4a, Table 4; see Supplementary Material 5 for breakdown by site). Vegetation losses in the pre-restoration timeframe were greatest for the runnel tidesheds (-3.78 ha), followed by no-action tidesheds (-3.24 ha) and reference tidesheds (-1.34 ha). After restoration activities, the study area gained a net of 3.44 ha or 1.1% from year 0 across 133 tidesheds, primarily driven by gains of 5.58 ha in the runnel tidesheds. Salt marsh losses continued yet slowed in the no-action (-1.84 ha) and reference tidesheds (-0.29 ha). It should be noted that possible post-restoration vegetation gains cannot be accounted for because several sites had no post-restoration data by 2021 (*e.g.*, Essex, Kent's Island, Ipswich marshes in Massachusetts, and Sapowet–South marsh in Rhode Island). Furthermore, numerous sites have been monitored only for 1 to 3 years after runnelling.

Question 2: Effect of Initial Conditions on Rate of Vegetation Recovery (Condition Model)

The trajectory of vegetation cover was then further tested using the baseline condition to help explain the variability of the vegetation response to runnels in the treatment model. There was a differential trajectory of the UVVR score between treatments ($F_{2,622.7} = 4.65$, $p = 0.010$) and baseline condition ($F_{1,621.5} = 3.71$, $p = 0.025$) over time separately based on the two-way interaction terms in the condition model. The three-way interaction term was not significant ($F_{2,622.5} = 2.32$, $p = 0.099$; Table 5, see Supplementary Material 3). The regression of the two marsh conditions differed within the runnel treatment but not for the no-action treatment. The degraded pannes of runnel tidesheds had low UVVR scores that did not markedly change over time. The degraded pool tidesheds of the runnel treatment improved after restoration. Both marsh conditions of no-action tidesheds showed little improvement after restoration based on UVVR (Figure 5). For percentage of vegetated area, the three-way interaction term was significant ($F_{2,622.8} = 7.06$, $p = 0.001$), meaning the vegetation recovery trajectory was dependent on both tideshed treatment and baseline condition. Degraded pool tidesheds of the runnel treatment experienced the lowest rates of vegetation losses pre-restoration and conversely the greatest rate of vegetation gains post-restoration (Figure 5). Both conditions of no-action tidesheds experienced

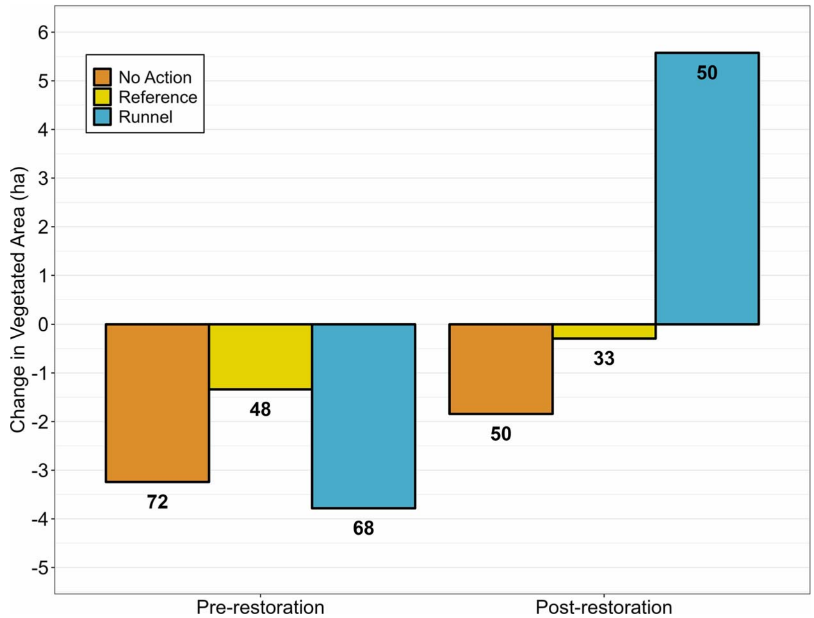

Figure 4. Change in vegetated marsh area (ha) before and after restoration activities across all sites for tideshed treatments. The number of tidesheds for each treatment and timeframe are reported with the bars.

vegetation loss pre-restoration yet diverged slightly post-restoration.

DISCUSSION

In the following section, the vegetation recovery after hydrologic restoration is discussed in the context of site history, tidal regime, and marsh conditions before restoration. Adaptive management considerations and additional restoration methods are highlighted. Broader considerations on the potential uses and limitations of remote sensing analysis for salt marsh monitoring and restoration are additionally addressed.

Marsh Degradation before Restoration Activities

Coastal ecologists oftentimes must be selective of timing, location, and size of restoration activities because of limited budgets and the regulatory need to minimize disturbance to salt marshes. The ability to document trends in marsh health with public aerial imagery can strengthen restoration planning. Over the past decade, drainage enhancement treatments (including runnels) were applied to tidesheds that exhibited excessive waterlogging with extensive pannes and pools. Before restoration, runnel tidesheds lost 3.78 ha or 3.80% of their original vegetated area, comparable with the no-action tideshed loss of 3.24 ha or 1.90%. The UVVR scores of both the runnel and no-action treatment tidesheds were five to nine times greater than respective reference tidesheds. The majority of runnel and no-action tidesheds had passed a theoretical tipping point of stability (UVVR > 0.13–0.15; Ganju *et al.*, 2017) and were on a trajectory of greater waterlogging, subsidence, and continued losses of high marsh habitat, as seen by others (Mariotti, 2016).

The trajectory of the vegetation cover of the runnel tidesheds before restoration can be divided between tidesheds with consistently dense vegetation or those with expanding mudflats and pools. It is possible the well-vegetated tidesheds (*e.g.*, Broad Cove, Jacob's Point, Zeke's Marsh) experienced a shift in dominance from high marsh graminoids to *S. alterniflora* before restoration because the remote sensing analysis does not differentiate

Table 4. *Pre- and post-restoration changes in vegetated salt marsh area (ha) for each treatment across the study area.*

Treatment	Timeline	Tidesheds	Total Marsh Size (ha)	Change in Vegetated Area (ha)	Percentage of Change of Marsh Size (%)
No action	Pre-restoration	72	166.3	−3.24	−1.95
	Post-restoration	50	118.4	−1.84	−1.56
Reference	Pre-restoration	48	99.5	−1.34	−1.34
	Post-restoration	33	75.6	−0.29	−0.39
Runnel	Pre-restoration	68	99.5	−3.78	−3.80
	Post-restoration	50	131.9	5.58	4.23
	Pre-restoration	188	365.30	−8.36	−2.29
	Post-restoration	133	325.90	3.44	1.06

Table 5. *Analysis of covariance (ANCOVA) of the condition spline-mixed model for UVVR and vegetated area metrics across no-action control and runnel treatments with restoration timeline, treatment, baseline condition, and their interactions as fixed effects (Bonferonni corrected α = 0.025).*

Metric	Term	Numerator df	Denominator df	Sum of Squares	Mean Square	*F*	*p*
UVVR	Timeline	2	618.38	0.35	0.17	5.62	0.004
	Treatment	1	193.03	0.04	0.04	1.14	0.288
	Baseline Condition	1	129.00	0.41	0.41	13.28	<0.001
	Timeline × treatment	2	622.27	0.29	0.14	4.65	**0.010**
	Timeline × condition	2	621.51	0.23	0.12	3.71	**0.025**
	Treatment × condition	1	196.28	0.03	0.03	0.84	0.361
	Three-way interaction	2	622.52	0.14	0.07	2.32	0.099
Vegetated Area	Timeline	2	619.27	807.06	403.53	23.29	<0.001
	Treatment	1	197.80	51.18	51.18	2.95	<0.001
	Baseline condition	1	127.22	1158.92	1158.92	66.89	<0.001
	Timeline × treatment	2	622.64	942.67	471.34	27.20	<0.001
	Timeline × condition	2	622.02	366.08	183.04	10.56	<0.001
	Treatment × condition	1	197.77	26.57	26.57	1.53	0.217
	Three-way Interaction	2	622.79	244.67	122.33	7.06	**0.001**

between species nor consider habitat quality for *A. cauducatus*. The process of high marsh graminoid replacement is a decadal process (Carey *et al.*, 2017; Raposa *et al.*, 2017; Smith, 2015) and a lagging indicator of long-term shifts in hydrology, elevation, and soil biogeochemistry (Himmelstein *et al.*, 2021; Watson *et al.*, 2016). Evaluation of the marsh vegetation community using remote sensing captures only the end results of long-term physical and hydrologic stressors, whereas field-based methods for hydrology, elevation, and pore water chemistry are valuable tools for detecting marsh vulnerability before vegetation losses (Cole Ekberg *et al.*, 2017).

Effect of Runnels on Marsh Health

The enhancement of drainage through runnel installation led to the recovery of vegetation in severely degraded high marsh habitats, despite continued vegetation losses of many surrounding no-action and reference tidesheds. The UVVR score declined by 0.04, and vegetated area improved by 1.7% on an annual basis post-restoration. The runnel tidesheds across the study area showed a net gain of more than 2.8 ha of vegetated marsh area than had been lost beforehand. From a restoration success perspective, as a whole, vegetated area in runnel tidesheds increased from 82% to 92% after 8 years. Reference tidesheds averaged 94% vegetated area when restoration activities were commenced, so runnel tidesheds had achieved almost 100% success. Additionally, only three sites (Winnapaug, Moody Point, and Zeke's Marsh) had runnels older than 6 years by 2021, highlighting substantial vegetation recovery within the first 5 years. *Spartina alterniflora* and *S. depressa* are typically the primary species recolonizing pannes and pools initially given the stressful biogeochemical soil conditions. A 3- to 5-year lag before substantial recolonization by high marsh graminoids into the interior of pannes and pools has been documented in studies in Rhode Island and Massachusetts (Besterman *et al.*, 2022; McKown *et al.*, 2023). Observational studies of natural pool breaches have shown

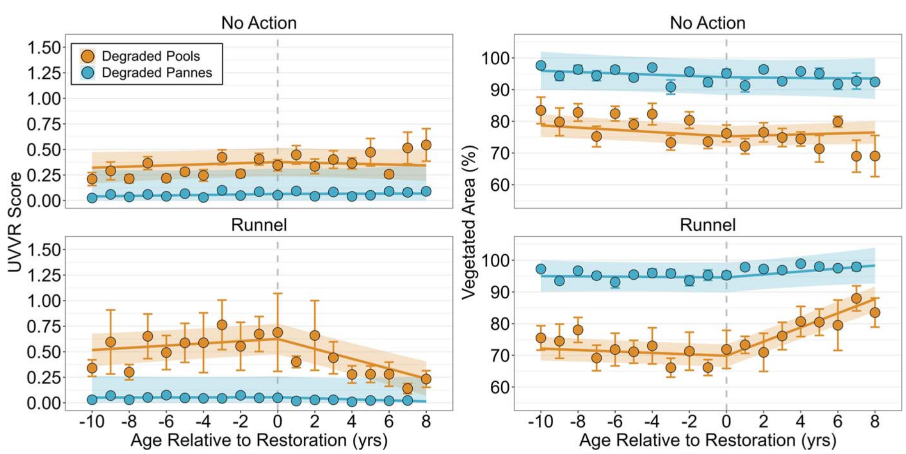

Figure 5. Spline-mixed condition model of UVVR score (left) and percent vegetated area (right) of healthy and degraded tidesheds of runnel and no-action control treatments over the restoration timeline. Healthy tidesheds were defined as UVVR scores less than 0.13 immediately prior to restoration (−2 to 0 years). The mean and standard error of each year are reported.

that full recovery of vegetation and, especially, elevation capital requires multiple decades (Smith and Pellew, 2021; Wilson *et al.*, 2014).

The high variability in the recovery of the vegetation after runnel installation (Figure 3) suggests that restoration follows a variety of pathways, depending on site-specific circumstances and intensity of subsidence. Vegetation cover improved from runnel installation in both less degraded (pannes) and more degraded (pools) tidesheds; however, as might be expected, revegetation by aerial assessment was more pronounced in the degraded tidesheds. When viewed through medium-term recovery periods (3 to 8 years), the more degraded a tideshed, generally, the greater the recovery rate. Revegetation may be relatively quick (1 to 5 years) in waterlogged pannes with sparse vegetation, whereas a lag may be required before substantial revegetation in hectare-sized pools (McKown *et al.*, 2023). Shifts in soil and biogeochemical properties to more suitable conditions after recent drainage enhancements are relatively undocumented and should be further explored (Perry, Ferguson, and Thornber, 2022). Other factors may also influence the rate of vegetation recovery, including tidal range, location within the estuary, site history (*e.g.*, mosquito ditching, agricultural infrastructure, abandonment), and sediment supply (Liu, Fagherazzi, and Cui, 2021; Mariotti, 2016; Schepers *et al.*, 2020; Smith, Hafner, and Niles, 2017). For example, the tidal range of the marshes in this study varied from microtidal in Rhode Island (<2 m) to mesotidal in Massachusetts and Maine (2–4 m), and it has been proposed that panne and pool recovery after breaching may differentiate between tidal regimes (Kearney and Turner, 2016; Mariotti, 2016, 2020). Overall, the strong reversal of UVVR and vegetated area in the spline regression speaks to the utility of runnels for drainage and revegetation across a myriad of site histories, hydrologic, and degradation conditions in New England.

Interestingly, a handful of no-action control tidesheds were found at Weekapaug and Winnapaug, where vegetation cover increased after drainage enhancement in neighboring tidesheds. The improvement in the no-action tidesheds at the two sites drove the minor improvement in the slopes in the treatment model (Question 1) and for the degraded pool condition in the condition model (Question 2). Two possible explanations for the counterintuitive improvement are the relative effect of the metonic cycle and the proximity of the tidesheds to drainage. The postrestoration timeline for Weekapaug and Winnapaug overlapped with the decline in tidal amplitude of the 18.6-year metonic cycle from 2016–21. The metonic cycle results in a change of 10–15 cm in tidal amplitude for Narragansett Bay, which equates to roughly 10% of the daily tidal range for the microtidal system (Figure 6). Second, the tidesheds are relatively narrow at Weekapaug and especially Winnapaug because of numerous waffle pool basins on the landscape (Smith *et al*., 2021). McKown *et al*. (2023) observed that drainage enhancement lowered the groundwater table up to 20 m away in the Plum Island Estuary. It is possible that the groundwater table was affected in adjacent small tidesheds at both sites, leading to vegetation recovery.

Limitations of Drainage Enhancement for Long-Term Habitat Recovery

The effect of runnels may have been only short-term (<10 years) at certain tidesheds where vegetation recovery in pannes and pools was partial and temporary. This study extended the remote sensing monitoring of Watson *et al.* (2022) at Canonchet marsh by an additional 2 years. The formation of new pannes 6 years post-restoration in tideshed 4 between 2019–21 was documented (see Supplementary Material 1), which would support the observations of increasing groundwater table by Watson and colleagues. Early vegetation recovery was followed by increasing waterlogging and panne expansion. Staffing constraints at the state level prevented full restoration in 2019 and adaptive management, including maintenance of runnels, in the southern section of tideshed 4. Continued monitoring by Save the Bay observed expansion of water coupled with vegetation declines in the same timeframe. McKown *et al.* (2023) recommended annual runnel inspection and maintenance for at least 5 years post-restoration after finding that a clogged runnel prevented complete drainage of pool in tideshed 29 in Pine Island.

Winnapaug tidesheds 11, 16, 17, and 18 had initial recoveries within the first 3 years; however, vegetation gains stagnated. The UVVR scores of Winnapaug remained relatively high (>0.10–0.15), indicating the vegetation community remained unstable and may yet fall back onto a trajectory of pool and panne expansion. Winnapaug and Weekapaug are located seaward of a large back barrier pond with diurnal tidal ranges less than 0.50 m (NOAA tidal gauge #8452600). Significant subsidence within the waffle pool basins has resulted in certain pools being too low in elevation to support salt marsh vegetation (Besterman *et al.*, 2022).

In the short and medium term, improved drainage from runnel installation effectively lowers the groundwater table and stimulates vegetation regrowth (McKown *et al*., 2023). In essence, runnels allow for a shift in stable states from pool to salt marsh by reestablishing vegetation and tidal hydrology, upon which the negative feedback loop of elevation maintenance is based (Baustian, Mendelssohn, and Hester, 2012; Wang and Temmerman, 2013). Over time, regrowth, primarily of *S. alterniflora*, should build elevation capital through sediment capture and belowground biomass (Cahoon *et al.*, 2019; Watson *et al*., 2017) and facilitate the recolonization of high marsh graminoids. However, runnels may be able to buy only a limited amount of time for the marsh platform as the rate of sea-level rise accelerates and the 18.6-year metonic cycle will begin to increase tidal ranges through 2034 (Figure 6).

Elevation deficits incurred during marsh degradation may be too large to overcome within constrained timeframes by restoration and natural processes alone because of loss of sediment and belowground biomass inputs as well as peat degradation. Microtidal marshes or those without adequate sediment supply may be likely to experience only short-term gains with runnels as belowground biomass inputs may be insufficient to maintain pace with sea-level rise (Day *et al.*, 2011; Kearney and Turner, 2016; Kirwan and Guntenspergen, 2010). Based on this study's findings and previous in-field monitoring (Besterman *et al.,* 2022; McKown *et al.*,

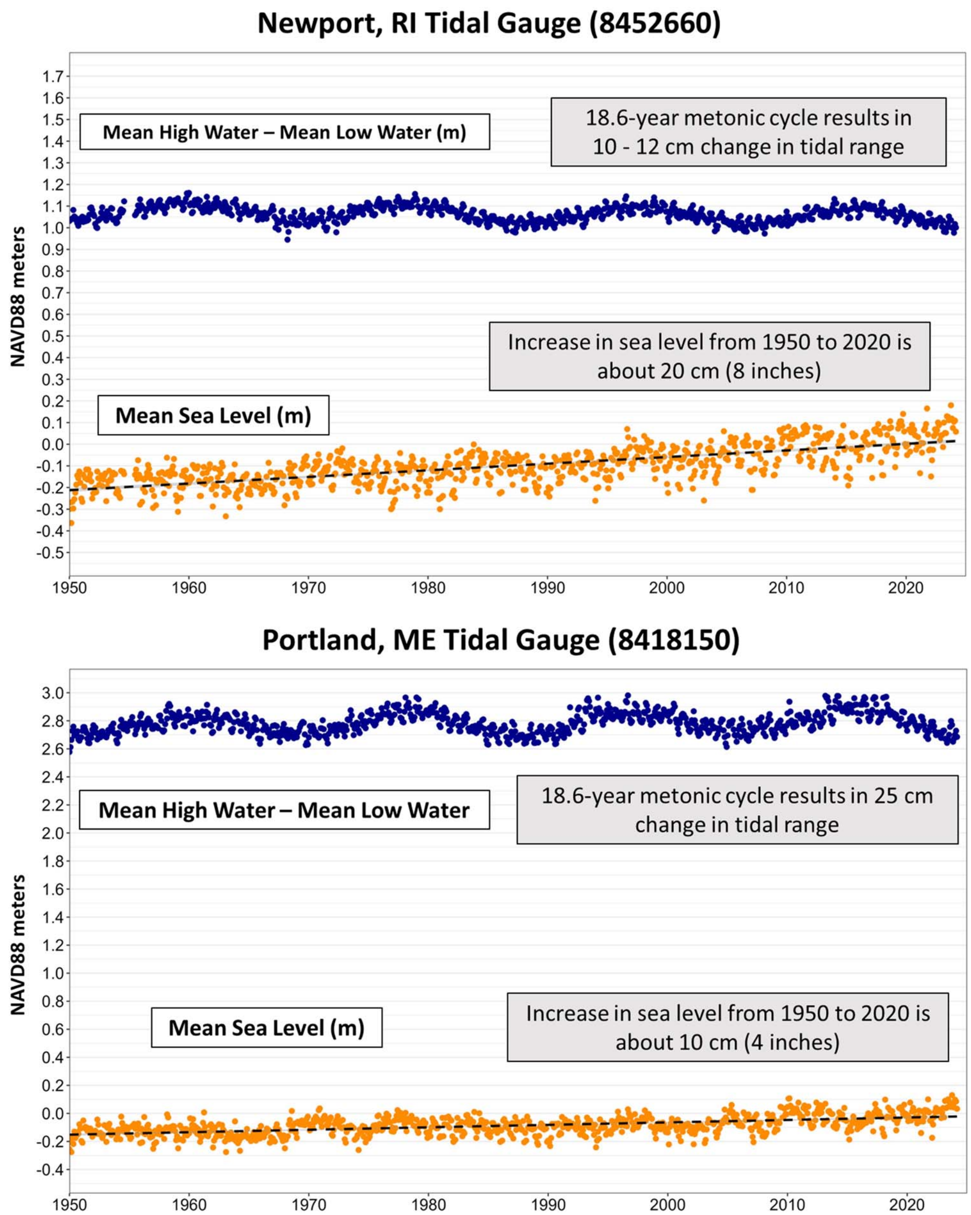

Figure 6. Mean sea level (orange) and difference in mean high water and mean low water (blue) of monthly tidal elevations from NOAA tidal gauges in (top) Newport, Rhode Island (8452660), and (bottom) Portland, Maine (8418150).

2023; Perry, Ferguson, and Thornber, 2022), implementation of runnels as early as extensive panne formation (*i.e.* short-form *S. alterniflora*) is observed is recommended to avoid unnecessary elevation capital loss and promote high marsh habitat. For marshes with severe elevation loss, repeated thin-layer sediment deposition could be considered if a suitable source of sediment is available in conjunction with hydrologic improvements to overcome elevation deficits and sustain the high marsh platform in the medium to long term (La Peyre, Gossman, and Piazza, 2009; Moore *et al.*, 2021; Raposa *et al.*, 2022). For salt marshes in southern New England, additional research is needed to understand whether runnels may provide additional habitat and avenues of expansion for *Sesarma reticulatum* (Purple marsh crab), which could lead to intense herbivory and marsh dieback in the interior marsh platform (Smith, 2024).

Limitations and Opportunities of Remote Sensing Coastal Wetlands

Remote sensing with public aerial imagery possesses limitations that should be understood before incorporation into coastal wetland–monitoring programs. Public aerial imagery may not be captured at low tides during the limited growing season (*e.g.*, July–September) to document peak vegetation cover in salt marshes. Differentiation of vegetation from floating algae in pools or dense *Vaucheria* ground cover was time consuming at sites dominated by shallow water pannes, such as Winnapaug and Weekapaug (Huang *et al.*, 2021). In degraded salt marshes, the image capture window for accurate classification is reduced because waterlogged and sulfide-stressed graminoids senesce earlier in the season. The NAIP imagery was not captured in Massachusetts and Maine until October 2021. As a result, classification training and supervision was time consuming for Massachusetts sites and impossible for Maine sites because senesced vegetation had similar NDVI signatures as algae or bare ground. Image collection during high tide or immediately following spring tides may lead to an overestimation of panne formation and loss of marsh by erosion, especially with water-based indices (*e.g.*, NWI, NMWI).

Aggregation of the marsh surface into two classifications loses valuable information of the vegetation community and provides only a conservative estimate about the recovery. Mapping of specific dominant graminoids and forbs in the salt marsh have been achieved with remotely sensed data (Campbell and Wang, 2019; Qi, MacGregor, and Gedan, 2020; Smith and Pellew, 2021). However, studies typically focus only on a few growing seasons of one to three sites and are captured at optimal times with high-resolution (<50 cm), unpiloted aerial vehicle (UAV) equipment. In this study, aggregation of vegetation cover was necessary, yet it prevented quantifying changes of high marsh habitat for *A. caudacutus* conservation efforts. Limitations from pixel resolution, seasonal and tidal timing, constrained budgets and personnel, and marsh condition pose substantial obstacles to species classification with public aerial imagery for discerning specific vegetation communities or preferred habitats of avian species. Adoption of UAVs with multi- or hyperspectral cameras should be considered for more detailed habitat analysis of specific sites when warranted by research or restoration goals (Doughty *et al.*, 2021; Haskins *et al.*, 2021). Incorporation of field monitoring with remote sensing is key to understanding the shifts in the vegetation community pre- and post-hydrologic manipulation, especially across numerous sites. The increasing availability of high-resolution imagery, classification software, and UAV deployment provides additional opportunities for coastal restoration research and monitoring in the future.

CONCLUSION

Anthropogenic alterations to tidal hydrology in East Coast salt marshes (*e.g.*, agriculture, mosquito ditching, open marsh water management) in conjunction with sea-level rise has led to waterlogging, subsidence, and the formation of large pools and bare pannes on the interior marsh platform. The decline of valuable high marsh habitat has been attributed to population declines of specialized endemic avian species. Through improved hydrology (regular flooding and drainage of waterlogged pannes and shallow pools), drainage enhancement measures reversed losses of high marsh habitat and stimulated recovery of vegetation coverage in 19 marsh sites across New England. Historical aerial imagery showed that high marsh graminoid habitat had been converting to panne habitat for at least 5 years before restoration, with many sites experiencing active conversion of interior marshes to shallow pools. The rate of vegetation recovery was a magnitude greater for highly degraded marshes compared with the rates of pre-restoration decline. Although the remote sensing analysis could not quantify the gains of nesting habitat for the Saltmarsh Sparrow, revegetation of pools and pannes by *S. alterniflora* is a first step to the recovery of the high marsh platform with a mix of graminoids preferred by sparrows as nesting sites. The full potential of drainage enhancement as a restoration tool may not be realized for another decade because most of the projects in this study were less than 5 years old. Across New England, runnels re-established drainage paths to allow regular flooding and draining. Revegetation of pannes and pools demonstrate runnels have the potential to enhance high marsh habitat to support sparrow populations. Runnels appear to be an effective tool to improve habitat and promote resilience and could be part of a holistic approach to restore hydrology and increase elevation within an adaptive management framework with an effective monitoring and maintenance program.

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