Long-term outcomes of salt marsh tidal restorations in Rhode Island

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Abstract

Salt marshes are valuable coastal ecosystems, but many have been degraded by roads, railways, and other infrastructure that restrict tidal flow and impound watershed runoff, causing shifts in chemistry, physical structure, and vegetation and animal communities. Tidal restoration is a common ecological management practice that involves restoring daily tidal flow to tide-restricted salt marshes in order to restore native vegetation and habitat functions. Biological communities may take one or more decades to recover following tidal restoration, but outcomes are seldom assessed on that timescale. We assessed the long-term outcomes of eight tidal restorations in Rhode Island by sampling plant and nekton communities using the original methods and stations from pre-restoration monitoring. We additionally assessed the current conditions of the restoration marshes against surrounding unrestricted salt marshes using data from a rapid assessment method and a simple crab-burrow count. Vegetation recovery was variable at the restoration marshes, with some marshes gaining native species concurrent with the loss of the invasive reed *Phragmites australis*, signaling recovery, while others continued to lose native vegetation cover. However, relative to an unrestricted control marsh that, over the same time-period, showed vegetation shifts indicating inundation stress, our findings suggest that restoration actions promoted vegetation recovery at all of the restoration marshes. Fish density responded positively to tidal restoration, but overall, nekton communities shifted away from fish abundance and starkly toward domination by the grass shrimp *Palaemonetes pugio* at the restoration and control marshes, suggesting ambient habitat degradation. Together, the time-series vegetation and nekton data suggest that while restoration actions promoted biological recovery, ambient inundation stress has worked to counteract it. According to our rapid index of marsh integrity, current marsh integrity was highly variable across the restoration marshes. Integrity index scores were not significantly different on average between restoration and reference marshes, but the cover of *Phragmites* was higher and the cover of meadow high marsh was lower for restoration marshes, suggesting incomplete recovery. We found that integrity increased with the degree of adaptive management following restoration, as well as the age of restoration. Coupled with modest evidence that vegetation recovery also increased with restoration effort and age, this finding suggests the importance of adaptive management and recovery time in restoration success. Additionally, in order to ensure recovery, salt marsh restoration practitioners in Southern New England and elsewhere may need to shift their methods and expectations to accommodate prevalent, increasing inundation stress. Our study methods highlight the utility of strictly-standardized long-term vegetation and nekton monitoring in assessing salt marsh restoration outcomes, and demonstrate how rapid assessment data collected across a broad set of reference wetlands can add valuable context to restoration findings.

1. Introduction

Salt marshes perform a host of functions that are valuable to people and wildlife, including water pollution filtration and uptake, protection from riverine and coastal flooding, provision of critical habitat for fish and wildlife, and recreation and aesthetics. Salt marshes are at increasing risk of degradation due to stressors caused by ongoing coastal development and climate-change impacts, such as changing precipitation patterns and rising sea levels (Gedan et al. 2009, 2011, Raposa et al. 2016, Watson et al. 2016). Salt marsh restoration can restore functions and values that have been diminished by human-caused degradation (Burdick and Roman 2012).

Tidal restriction and impoundment of freshwater runoff are common salt marsh disturbances associated with the construction of roads, railways, raised farm trails, or other linear features across the marsh (Roman and Burdick 2012). State and federal environmental agencies also intentionally impounded salt marshes across the eastern U.S. in the mid-1900s for mosquito control and to create waterfowl habitat (Erwin 1986; Montague et al. 1987). Tidal restriction and impoundment (hereafter referred to as 'tidal restriction' for simplicity) can cause subsidence and submergence of the marsh platform, lower soil salinity, trap excessive watershed nutrients and toxins, and reduce or eliminate tidal flushing, often creating stagnant, eutrophic systems with altered plant and animal assemblages (Roman et al. 1984, Raposa 2002, Roman et al. 2002). These changes make tide-restricted marshes in the Northeastern United States prone to invasion and eventual domination by the non-native genotype of the common reed *Phragmites australis* which can largely displace native plant species on the high marsh (Meyerson et al. 2009). By the latter part of the 1900s, salt marsh managers began to recognize the ecological damage caused by tidal restrictions and the value to humanity and wildlife of restoring original conditions; tidal restorations have since been conducted across the eastern U.S.

In Rhode Island (RI), state agencies and their partners have been supporting, permitting, and funding coastal restoration projects for more than 30 years, but few restoration projects have been assessed on a long-term basis where the full benefits of restoration could be evaluated. Although tidal restorations may take one to two decades to mature (Broome et al. 1988, Warren et al. 2002), restoration projects conducted in the northeastern U.S. typically include funding for only one or two years of post-restoration monitoring, leaving the long-term outcomes of many restoration projects unknown.

Biological monitoring is often conducted to help assess the early outcomes of tidal restoration, usually by focusing on plant species cover and nekton (fishes and decapod crustaceans) community composition. Both groups measurably respond to changes in physical environmental conditions and are integral components of marsh functionality (Roman et al. 2002, Konisky et al. 2006, Raposa et al. 2018a). As a common target of salt-marsh restoration, recovery of native vegetation can be used as a direct measure of restoration success. Recent studies have also linked the density of burrowing marsh crab (*Uca* spp., *Sesarma reticulatum*) burrows with inundation stress (Crotty et al. 2017, Raposa et al. 2018b), suggesting they that may act as a simple indicator of marsh vulnerability to sea-level rise.

Rapid assessment may also be used to indicate restoration success. Rapid assessment methods are designed to assess site-level wetland conditions, such as disturbances, integrity, and support of ecosystem services, using data collected during a single site visit supported by simple geospatial analysis

(Fennessey et al. 2007). Rapid assessment can therefore quickly provide information about wetland ecological conditions across large areas of concern, such as states or regions, allowing for the efficient development of large multi-wetland reference samples, against-which individual marshes, including restoration marshes, can be compared (Kutcher and Forrester 2018).

Our goal was to use biological and rapid assessment methods to assess the long-term outcomes of salt marsh tidal restoration in RI. The combination of time-series information from biological monitoring and contextual information from rapid assessment allows us to characterize restoration outcomes relative to pre-restoration conditions, and compare resulting conditions relative to unrestricted (and unrestored) marshes. Additionally, we evaluate the monitoring strategies for their effectiveness at elucidating long-term restoration outcomes, and make recommendations for adaptive management on a time scale that is most relevant to the pursuit of sustainable restoration.

2. Methods

2.1 Study sites

Eight matured salt marsh restoration sites from across RI were selected for long-term assessment. We defined a matured restoration site as a tide-restored (formerly tide-restricted) salt marsh with ten or more growing seasons since the re-introduction of daily tidal flooding, even if other restoration actions were conducted later. We selected the mature tidal restoration sites (hereafter, Restoration marshes) based on age since tidal restoration (>10 years), availability of pre-restoration biological data, and even distribution across Narragansett Bay and coastal Rhode Island. The Restoration marshes were located in the Narragansett Bay upper estuary (3 marshes), mid estuary (2), lower estuary (1), and the Atlantic coast (2), and marsh size ranged from 0.9 to 38 ha. (Table 1, Fig. 1. App. A). Restoration age—i.e., the time elapsed between the first growing season following initial tidal restoration and the growing season of 2020—ranged from 10 to 22 years. And, the number of total management activities conducted per marsh ranged from one to nine (Table 2). All marshes showed evidence of impounding fresh water prior to restoration, and at all marshes except GOOS, *P. australis* had become a dominant feature of the marsh platform.

The number of management activities conducted, and the timespan of those activities, were variable across the Restoration marshes. All marshes had undergone tidal restoration, entailing opening a tidal restriction to restore tidal exchange with the surrounding estuary. Other treatments varied by marsh and included mulching, cutting, or removal of *P. australis*; herbicide treatment of *P. australis* (typically three seasons of Glyphosate spraying); planting of native marsh species (typically *Spartina alterniflora*, *Spartina patens*, *Iva frutescens*, and *Distichlis spicata*); substantial removal of fill, and others, as defined (Table 2). We define: *deep drainage* as the manipulation or creation of major conduits of tidal exchange across the marsh platform, *water control* as a manmade structure designed to limit the volume or duration of tidal flow, *shallow drainage* (or *runnels*) as shallow ditches designed to drain surface water from the marsh platform to larger tidal water features, *elevation enhancement* as the placement of sediments on the marsh surface to raise platform elevations, and *pool creation* as the physical excavation of water features on the high marsh surface. The timespan across-which adaptive management activities were conducted ranged from a single season to 22 years (Table 2).

Table 1. Locations and sizes of eight matured salt marsh restoration marshes and a control* marsh assessed in 2020.

Marsh Name	Code	Town	Area (ha.)	Latitude	Longitude
Walker Farm	WALK	Barrington	6.2	41.758	-71.324
Jacobs Inner	JAIN	Warren	2.7	41.713	-71.288
Silver Spring	SILV	Bristol	4.4	41.679	-71.276
Coggeshall*	Control	Portsmouth	20	41.651	-71.343
Potter Pond	POTT	Portsmouth	0.9	41.639	-71.342
Duck Cove	DUCK	North Kingstown	2.4	41.556	-71.438
Gooseneck Cove	GOOS	Newport	8.5	41.458	-71.331
Sachuest Mid	SAMI	Middletown	3.5	41.870	-71.249
Galilee Inner	GAIN	Narragansett	38	41.379	-71.504

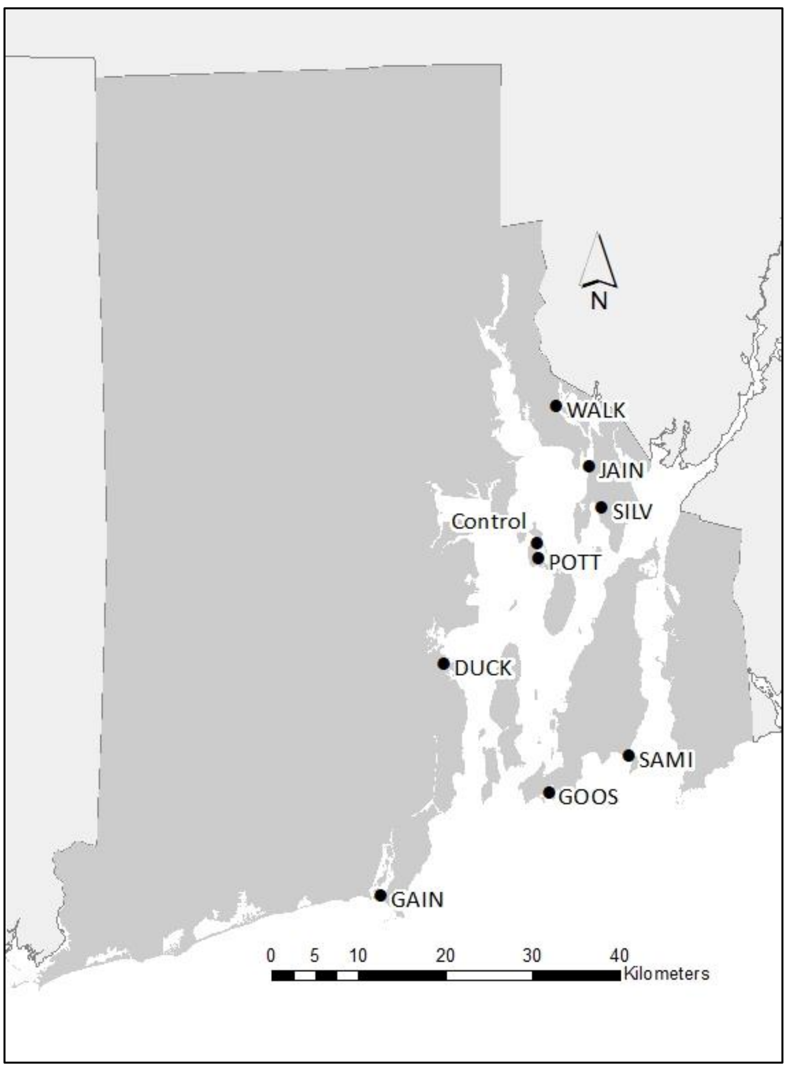

Figure 1. Locations of eight mature tidal restoration marshes and a control marsh in Rhode Island (Table 1).

Table 2. Restoration details of eight tidal restoration mashes in Rhode Island; X indicates the restoration activities undertaken for each marsh.

	Restoration Activities														
Site	Tidal	Mulch *P. australis*	Herbicide *P. australis*	Native Planting	Deep Drainage	Water Control	Shallow Drainage	Fill Removal	Elev. Enhancement	Pool Creation	First Growing Season	End of Rest. Activ.	# of Rest. Activities	Span of Rest. Activ.	Age in 2020
SAMI	X	X	X	X	X	X	X		X	X	1998	2020	9	22	22
GAIN	X	X		X	X	X		X			1998	1999	6	1	22
WALK	X	X			X	X		X			2006	2007	5	1	14
SILV	X	X	X	X				X			2009	2011	5	2	11
JAIN	X	X	X				X				2010	2016	4	6	10
POTT	X	X	X								2003	2003	3	0	17
GOOS	X						X				2010	2014	2	4	10
DUCK	X										2002	2002	1	0	18

2.2 Study design

We collected new vegetation and nekton data to compare with data collected prior to restoration, and new rapid assessment and crab-burrow data for comparison to data from current unrestricted marshes. We used a BACI (Before-After, Control-Impact) study design (Green 1979, Smith 2014) to analyze the time-series biological data. Vegetation and nekton community-composition data collected prior to restoration (i.e., Before Impact) were analyzed against newly-collected data (i.e., After Impact) to assess change over time for both groups. Coggeshall Marsh, a 20-ha marsh located in the approximate geographic center of the distribution of restoration marshes, was used as the tide-unrestricted (hereafter) Control marsh. Vegetation and nekton data collected at the Control marsh concurrent with pre-restoration and new data collection at the Restoration marshes were used to help differentiate restoration responses from background ecological change over time.

We used recent rapid assessment data from 30 unrestricted salt marshes distributed across Narragansett Bay and coastal Rhode Island as a reference sample (hereafter, Reference marshes) against-which to compare the new rapid assessment data from the Restoration marshes, and used recent crab-burrow data from 10 unrestored marshes (R. McKinney, 2016 unpublished data) as reference for our new crab-burrow data. We calculated the mean elevation of the vegetated marsh platform as an indicator of inundation stress, and used the number and timespan of restoration activities as rudimentary indicators of restoration effort to investigate variability in restoration outcomes (Table 2).

2.3 Data collection

2.3.1 Vegetation composition

For assessing changes in vegetation composition, we used the methods and sampling stations originally used in pre-restoration sampling from each Restoration marsh. All new sampling was conducted between July and September 2020. The sampling generally followed point-intercept methods described by Roman et al. (2002). For all but one marsh, square, 1-m^2 sampling plots were systematically arranged along 3 to 5 transects running from the water's edge to the upland interface and evenly spaced across the marsh. The total number of plots sampled per marsh varied according to the original data sampling (Table 3). A sampling grid of 50 evenly-distributed sampling points was laid across each plot and a 1-m-long by 3-mm-diameter rigid rod was lowered vertically at each point. Every plant touching the rod was recorded to estimate cover for the plot; for example if *S. alterniflora* touched the rod at 10 of the 50 points, its cover was estimated to be 20% for the plot. For sampling plots dominated by reeds or shrubs taller than the rod (making accurate estimation using point-intercept difficult), we visually estimated plant species cover as falling within one of six cover classes assigned according to the Braun-Blanquet approach (Westhoff and Van Der Maarel 1978). We used the mid-points of the cover classes as values for analyzing the visually-estimated data. Earlier work indicated that point intercept and visual-estimation methods produced similar results in salt marsh monitoring (Raposa et al. 2020). For DUCK, a larger number of 0.5-m^2 rectangular plots and visual estimation were used in the original monitoring program; we followed those methods for our new sampling, but only used half the number of plots as originally used (every other plot). We could not locate the original vegetation sampling stations for GAIN, and therefore did not assess vegetation change for that marsh.

Table 3. Details of vegetation, crab-burrow, and nekton sampling methods used to assess eight tidal restoration marshes in Rhode Island. *Visual estimation was only used in plots dominated by tall (>1m) vegetation.

Marsh	# Veg/crab Plots	Plot Size (m^2)	Veg Estimation Method	# Nekton Stations
DUCK	35	0.5	Visual	n/a
GOOS	21	1.0	Point-intercept	13
JAIN	13	1.0	Point-intercept / Visual*	6
POTT	28	1.0	Point-intercept	23
SAMI	14	1.0	Visual	7
SILV	21	1.0	Point-intercept / Visual*	16
WALK	25	1.0	Point-intercept	13
GAIN	n/a	n/a	n/a	20
Control	21	1.0	Point-intercept	15

2.3.2 Crab burrow counts

Burrowing-marsh-crab (*Uca* spp., *Sesarma reticulatum*) abundance was estimated roughly following methods detailed in Raposa et al. (2018b) and using the plots already established for vegetation sampling (Sec 2.2.1 above). All crab burrows ≥3mm in diameter, falling within each vegetation plot, were counted just following vegetation surveys. Earlier studies have found that the density of crab burrows can indicate marsh platform degradation associated with increasing inundation stress, as burrowing crabs are better able to burrow through peat that is partly softened and released from dense root networks of the high-marsh grass *S. patens* (Crotty et al. 2017, Raposa et al. 2018b).

2.3.3 Nekton composition

Nekton composition was estimated using a 1-m^2 throw-trap according to Raposa (2002). We located and used the original pre-restoration stations for our post-restoration sampling (Table 3). The throw-trap was thrown off the edge of the marsh platform during a mid-dropping tide to trap any nekton using the shallow water. The nekton were removed from the trap using a 1-m by 0.5-m dip net. All nekton were identified and counted, and 15 individuals of each species were measured for total length. The animals were processed quickly to avoid mortality, and released.

2.3.4 MarshRAM rapid assessment

We assessed marsh platform community composition, integrity, ecosystem functions and services, and human disturbances using the MarshRAM rapid assessment (Kutcher 2019). The relative cover of pre-defined salt marsh community types was estimated using a *walking transect* method, wherein the investigator takes even steps along eight evenly-spaced transects traversing the marsh from upland to water's edge. Each step across a prescribed marsh cover type (Table 4) was counted as a data point, and all data points were aggregated to estimate the relative cover of each type. For example, if 12 of 200 total steps across all transects traversed the *Salt Shrub* cover type, then *Salt Shrub* cover was estimated to be 6% for the marsh. In order to apply relative cover to assess marsh integrity, each cover type was preassigned a coefficient of community integrity. The coefficients range from 0 to 10, where coefficients approaching 0 reflect low sensitivity to inundation stress and other human disturbances (e.g., nutrient loading, impoundment, ditching) and low habitat value, whereas coefficients approaching 10 reflect high sensitivity to those disturbances and high habitat value. The average of the coefficients, weighted by the proportion of each community type across all transects, is used as an index of marsh integrity (IMI). An earlier application of IMI, using the same 30 unrestricted marshes as used in this study, suggests that marshes with higher IMI scores are less degraded and less vulnerable to inundation stress and loss than those with lower IMI scores (Kutcher 2019).

Table 4. Salt marsh communities and coefficients of community integrity (CCI) used to generate indices of marsh integrity (IMI) for 31 unrestricted and 10 tide-restored salt marshes in Rhode Island. Broad cover-types are listed in approximate order from upland interface to seaward edge, followed by typically less abundant features (from Kutcher 2019).

Marsh Habitat	CCI	Description
Salt Shrub	9	Infrequently flooded shrub community (>30% shrub cover) located at higher elevations on the marsh platform and at the upland interface; typically dominated by *Iva frutescens*, *Baccharis halimifolia*
Brackish Marsh Native	10	Emergent community where freshwater from the watershed dilutes infrequent flooding by seawater; typically dominated by non-halophytic, salt tolerant vegetation such as *Typha angustifolia*, *Schoenoplectus robustus*, *Spartina pectinata*
Phragmites	3	Areas where *Phragmites australis* cover >30%
Meadow High Marsh	10	Irregularly flooded emergent high marsh community dominated by any combination of *Spartina patens*, *Juncus gerardii*, *Distichlis spicata*; *S. alterniflora* absent
Mixed High Marsh	7	Irregularly flooded emergent high marsh community comprised of any combination of S. patens, *Juncus gerardii*, *Distichlis spicata*; *S. alterniflora* present
Sa High Marsh	5	Irregularly flooded emergent high marsh; typically monoculture of *S. alterniflora*, although *Salicornia* sp. may be present
Dieoff Bare Depression	1	Shallow gradual depression on marsh platform, irregularly flooded by tides but typically remaining flooded or saturated to the surface throughout the tide cycle; <30% vascular vegetation cover, or bare decomposing organic soil, typically with remnant roots of emergent vegetation; may have algal mat, filamentous algae, wrack, or flocculent matter present
Low Marsh	8	Regularly flooded, typically sloping emergent community located at the tidal edges of the marsh and dominated by tall-form *S. alterniflora*

Dieback Denuded Peat	0	Typically non-depressional marsh platform feature; marsh peat is exposed (vegetation <30%)and perforated from grazing, crab burrowing, and erosion; typically at or near tidal edge
Natural Panne	8	Shallow steep-sided depression on marsh platform with clearly defined edge; irregularly flooded, typically dry at low tide; species may include any cover of *Plantago maritima*, *Sueda maritima*, *Salicornia* sp., *J. gerardii*, *Aster* sp.
Natural Pool	6	Shallow steep-sided depression on marsh platform with clearly defined edge; irregularly flooded by tides but typically remaining flooded throughout the tide cycle; organic or sandy substrate lacking emergent vegetation and roots but may support *Ruppia maritima*
Natural Creek	8	Narrow, natural, unvegetated, regularly-flooded or subtidal feature cutting into the marsh surface; typically sinuous
Ditch	2	Manmade ditches and associated spoils on the marsh surface; typically linear
Bare Sediments	4	Irregularly or infrequently flooded; sandy or gravelly sediments on the marsh surface with <30% vegetation cover; typically from recent washover event or elevation enhancement project

2.4 Data analysis

2.4.1 Vegetation composition

Prior to analysis, we addressed inconsistencies in the original vegetation data to estimate pre-restoration unvegetated marsh cover (i.e., bare ground). We re-categorized the original cover values for any dead-standing vegetation as 'bare ground'. And, for any original plot where the sum-total of living vegetation cover and 'bare ground' was still less than 100%, we estimated bare groundcover by setting it at a value that would result in 100% total cover when summed with the cover of living vegetation.

Analysis focused on examining changes in vegetation community structure over time, before and after restoration, at each site individually, and examining community similarity with the Control marsh at each time period. We were interested in exploring effects of restorations on vegetation and whether restored marsh communities became more similar with reference conditions over the long-term. Pre-restoration similarity was examined using the closest corresponding year of available data in the Control marsh (e.g., 2000 from the Control for marshes restored early, 2011 data from the Control for marshes restored later); all data for post-restoration analyses were from 2020, except SAMI, for which data were from 2019.

First, we analyzed the cover of key vegetation species as indicators of restoration effects, where gains in the cover of *I. frutescens*, *S. alterniflora*, and *S. patens* (native species typically dominant in healthy marshes), and losses in the cover of *P. australis* and bare ground (typical signs of tidal restriction and impoundment) were used to indicate recovery, whereas the inverse would indicate failure to recover. We assessed the change over time for each indicator (1) directly and (2) adjusting for (by subtraction) changes in the Control marsh to account for changes caused by background (non-restoration) factors. We also aggregated the adjusted cover values of our indicators for each marsh to generate a measure of the overall magnitude of restoration-induced recovery for each marsh, where *Magnitude* = (∑ Δ%native species – Δ%*P. australis* – Δ%Bare)/2. Dividing the aggregated changes by two accounts for the redundancy of losing *P. australis* and bare, and gaining native species for any given area.

Next, for each marsh, we used one-way Analysis of Similarity (ANOSIM) on pre- and post-restoration data from the Restoration marshes and the Control marsh, to directly compare vegetation communities over time. Similarity Percentages (SIMPER) was then used to identify species that contributed to community similarity within each marsh each time period, and community dissimilarity between the restored marsh and the Control both before and after restoration. Prior to all analyses, data were square-root transformed to dampen the weight of dominant species and resemblance matrices created with Bray-Curtis similarities were used for all ANOSIM at each marsh. PRIMER version 7.0.17 (Clarke and Gorley 2006; Clarke and Warwick 2001).

2.4.2 Crab burrow counts

Site-wide crab-burrow tallies from the eight Restoration marshes were compared to each other using ANOVA, against tallies from the ten unrestored marshes using independent T-test, and against the site-wide proportion of MarshRAM cover types and IMI using Pearson correlation analysis. Prior to analysis, we adjusted the reference sample data, which represented an equal number of samples from the high marsh and the low marsh, to reflect the proportion of high marsh to low marsh in the Restoration sample (0.966 to 0.34, respectively). Pearson correlation analysis was again used in the Restoration marshes at the plot scale (n=155), to compare the density of burrows with plot-specific vegetation composition.

2.4.3 Nekton composition

We examined changes in nekton community structure over the study period, from before restoration to present, to assess the effects of restorations on nekton. We used the closest corresponding year of pre-restoration data available from the Control marsh (e.g., 2000 from the Control for marshes restored early, 2011 data from the Control for marshes restored later); all post-restoration data were from 2020. We used Pearson correlation to investigate associations among fish and crustacean density and richness and other environmental factors. Analyses included: 1) examining changes over time in fish density, fish species richness, crustacean density, and crustacean richness, and 2) using the ratio of grass shrimp (*Palaemontetes Pugio*) density to total finfishes density as an indicator of habitat integrity.

2.4.4 Rapid assessment data

IMI values for Reference marshes were sorted into quartiles to represent least-degraded (upper quartile), most-degraded (lower quartile), and intermediately-degraded (interquartile range) salt marshes. Bar graphs and independent T-tests were used to compare IMI scores, community composition, and other rapid-assessment attributes between Restoration and Reference marshes. To investigate variability in restoration outcomes, we used Pearson correlations to test for relationships between various attributes from MarshRAM and other methods; when conditions for equal variances or normality could not be met, we used Spearman rank correlation.

We correlated IMI scores to: 1) the degree of disturbance observed at the Restoration marshes (MarshRAM *Disturbance Index*), 2) mean elevation—representing inundation stress, 3) age of the tidal restoration, 4) the number of restoration activities undertaken and the number of years the activities spanned—indicators of restoration effort, and 5) MarshRAM's *Replacement Ratio* metric, which compares the estimated area of land that the marsh will theoretically migrate over with little or no active management to the size of the existing marsh platform.

We also compared additional relevant information on marsh functions derived from MarshRAM between Restoration and Reference marshes. These include: 1) MarshRAM *Disturbance* as an index of cumulative human disturbance, 2) the number of natural habitat types observed as an indicator of natural habitat richness, and 3) the density of all waterbirds observed (#birds/ha) and the number of marsh sparrows flushed along walking transects (#birds/m) as indicators of bird use. We also compared the values between Restoration and Reference marshes of a MarshRAM metric that ranks aggregate salt-marsh ecosystem functions and services that include storm protection of property, flood-flow alteration, habitat complex or corridor adjacency, sediment and toxin retention, nutrient uptake, carbon storage, support of endangered or threatened species, fish and shellfish habitat, wildlife habitat, hunting or fishing support, passive recreation, and educational or historical significance.

3. Results

3.1 Vegetation composition

Changes in key vegetation indicators from pre-restoration to present (2019-2020) varied across the study marshes, with only SAMI trending toward recovery across all indicators, and the Control marsh and GOOS declining in nearly every indicator (Table 5). However, adjusting for (subtracting) changes in the Control, all Restoration marshes trended toward recovery. Collectively, the Restoration marshes improved across all vegetation indicators, and one individual marsh (WALK) improved across all the indicators, when adjusted for the Control. There was modest evidence that the aggregate adjusted *magnitude* of net vegetation recovery per marsh (see Table 5) was correlated with the age of the restoration (Pearson, n=7, r=0.62, P=0.136), and with our proxies for restoration effort: the number of restoration activities (r=0.62, P=0.134) and the timespan of restoration activities (r=0.58, P=0.174), although with the low sample size, none of these findings were significant at an α of 0.05.

Table 5. Net changes from pre-restoration to 2020 (2019 for SAMI) in the % cover of key vegetation species and bare ground, unadjusted and adjusted for (subtracting) contemporaneous changes in the Control marsh. ALL refers to all plots aggregated across all Restoration marshes (n=156); control-marsh pre-restoration cover-values from 2000 and 2011 were averaged to determine net changes for ALL and Control. Magnitude = (∑Δ%native species–Δ%*P. australis*–Δ%Bare)/2, and roughly represents the aggregate of net vegetation recovery across the indicators. Green shading represents a shift toward recovery, whereas red shading represents a shift away from recovery. *Values were adjusted using Control-marsh data from 2000 or 2011, whichever were closest to the year of pre-restoration data for the restoration marsh.

	DUCK	GOOS	JAIN	POTT	SAMI	SILV	WALK	ALL	Control
Unadjusted									
I. frutescens	0.0	0.0	28.1	-1.1	37.6	12.2	0.3	6.8	-2.8
S. alterniflora	34.2	-4.2	10.9	13.7	8.3	11.5	48.6	20.7	3.1
S. patens	7.3	-13.4	-27.8	-19.6	32.9	-10.5	-0.2	-7.8	-30.5
P. australis	-47.5	3.8	-34.0	-12.1	-3.6	0.8	-0.9	-14.9	0.4
Bare	34.0	2.5	11.9	24.5	-33.8	NA	-44.4	1.2	15.9
Magnitude	20.6	-9.0	14.7	-8.5	45.0	3.6	38.9	13.4	-18.2
Adjusted for Control*									
I. frutescens	1.3	4.3	32.4	0.3	38.9	16.5	4.6	9.6	
S. alterniflora	21.8	1.9	17.0	1.3	-4.1	17.6	54.7	17.6	
S. patens	47.0	7.9	-6.5	20.2	72.6	10.8	21.1	22.7	
P. australis	-47.9	3.4	-34.3	-12.5	-4.0	0.4	-1.3	-15.2	
Bare	14.8	-10.2	-0.8	5.4	-53.0	NA	-57.2	-14.7	
Magnitude	51.6	10.4	39.0	14.4	82.2	22.2	69.4	39.9	

Overall similarities in vegetation composition between the Restoration marshes and the Control marsh over time are shown in Figure 2. Assuming that initial Control marsh conditions are the restoration target, all marshes became more similar to target conditions over the study period. However, most were even more similar to current conditions at the Control marsh (which has been continually degrading over time in response to sea-level rise), suggesting overriding influences of other external factors besides restoration. Only SAMI, and to a lesser extent, JAIN and SILV marshes, became more similar to the original restoration target.

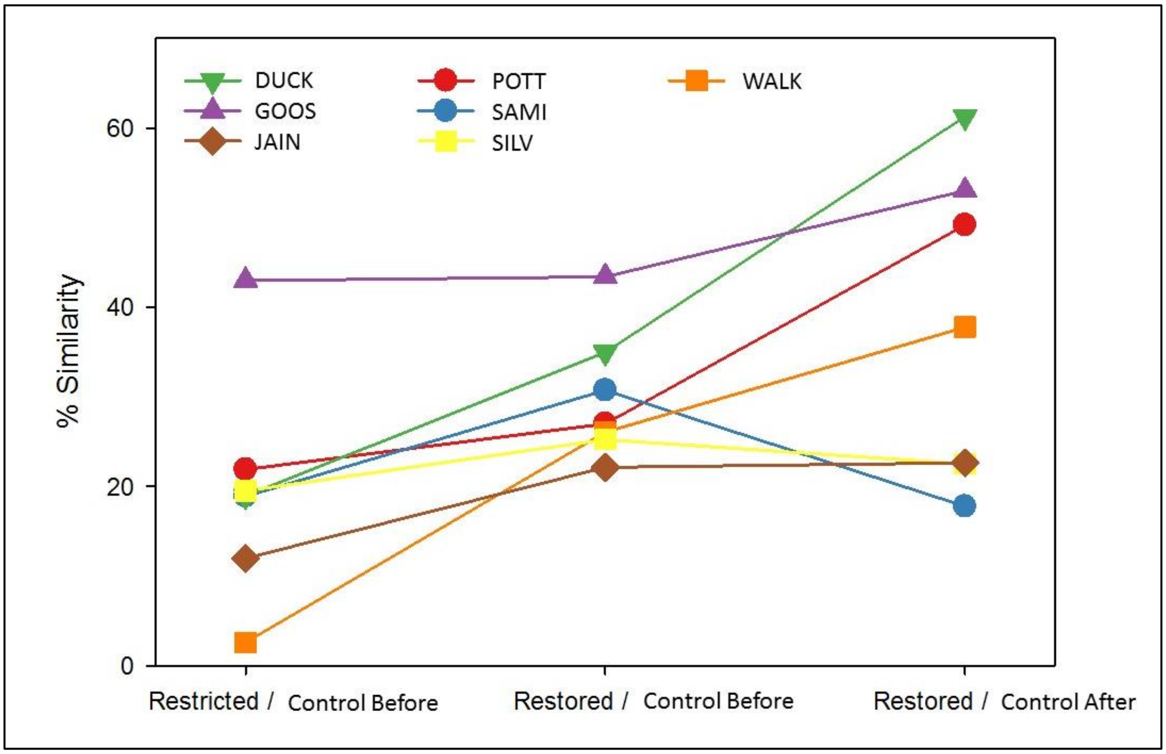

Figure 2. Vegetation community similarity between Restoration marshes and the Control marsh over time. LEFT: similarity between tide-restricted marshes and initial reference conditions (Control Before); MIDDLE: similarity between current restored marshes initial reference conditions (Control Before, i.e., the restoration target); RIGHT: similarity between current restored marshes and current reference (Control After).

3.2 Crab burrow counts

We detected no difference in current crab-burrow densities between the Restoration marshes and the 10 unrestricted baseline marshes (mean=10.67 versus 11.07 burrows/m^2, respectively; independent T-test, *df*=15, *T*=-0.06, *P*=0.952). Burrow densities were similar among Restoration marshes, except for Potter Pond, which had a significantly higher density than the others (ANOVA, *df*=6, *F*=5.94, *P*<0.001; Table 6). At the marsh scale (*n*=7), crab-burrow density was strongly correlated with the proportion of the community types *Dieback Denuded Peat* (Pearson correlation, *r*=0.96, *P*<0.001) and *Low Marsh* (*r*=0.80, *P*=0.029) derived from MarshRAM. And, at the plot scale (*n*=155), crab-burrow density was positively correlated with the cover of bare ground (*r*=0.43, *P*<0.001) and negatively correlated with *P. australis* cover (*r*=-0.29, *P*=0.008).

Marsh	Crab-burrow Density
POTT	37.32
DUCK	10.57
GOOS	10.55
WALK	9.68
SILV	3.55
JAIN	2.15
SAMI	0.86
GAIN	N/A

Table 6. Estimated crab burrow densities (m^{-2}) at eight tidal Restoration marshes in Rhode Island.

3.3 Nekton composition

Changes in nekton composition over time were highly variable among the Restoration marshes, although some patterns emerged. Fish density and richness declined on average across the Restoration marshes and at the Control marsh, whereas mean crustacean density increased by over 190 individuals per m^2, mostly due to increases in *P. pugio* (Table 7). Accordingly, the ratio of *P. pugio* density to aggregate fish density increased in every Restoration marsh, across all marshes, and in the Control marsh over the study period, although there was no indication that fish density declined with increasing *P. pugio* (r=-0.06, P=0.445). However, when adjusted for (subtracting) the Control values, fish density and nekton richness increased across marshes, and the ratio of *P. pugio* to fish declined.

The (unadjusted) increase in *P. pugio* was strongly correlated with the (unadjusted) increase in *S. alterniflora* we found across marshes (Pearson, n=6, r=0.94, P=0.005), but not with any other vegetation indicator (P>0.05 for all), whereas (unadjusted) fish density was most-closely correlated (+) with the aggregate (unadjusted) *magnitude* of vegetation recovery (Table 5; r=0.64, P=0.173), although with the low sample size, this finding was not significant at α=0.05. Of the nekton metrics (fish density, fish richness, crustacean density, crustacean richness, and *P. pugio* to fish ratio), only change in crustacean richness showed any sign of varying predictably with age of restoration (Spearman Rank, n=7, r_s=0.68, P=0.093) or our proxies of restoration effort (# restoration actions, r_s=0.60, P=0.152), and neither of these were significant at an α of 0.05.

Table 7. Long-term changes in the density (m^{-2}) of nekton species across eight tidal restoration marshes, unadjusted and adjusted for (subtracting) contemporaneous changes in the Control marsh. Changes among restoration marshes compare data from pre-restoration (1998-2010) to data from 2020, whereas changes in the control marsh compare the average of data from 2000 and 2011 to data from 2020. All represents the average value across marshes. For density and richness, green shading represents a shift toward recovery, whereas red shading represents a shift away from recovery, assuming that a gain in the ratio of *P. pugio* to fish indicates habitat degradation (James-Pirri et al. 2014).

	WALK	JAIN	SILV	POTT	SAMI	GOOS	GAIN	ALL	Control
Fishes									
Fundulus heteroclitus	5.37	18.33	-6.10	-14.89	13.42	-4.91	-6.25	0.71	-15.94
Cyprinodon variegatus	0.38	2.75	1.52	-14.91	7.29	0.00	-3.80	-0.97	0.71
Lucania parva	0.67	2.88	-0.28	-51.04	0.00	1.79	-0.29	-6.61	-0.24
Menidia beryllina	-0.20	0.08	-3.21	-0.48	-0.05	2.46	0.01	-0.20	1.21
Brevoortia tyrranus	0.00	0.00	0.00	0.00	0.00	0.00	4.44	0.63	0.00
Fundulus majalis	0.00	0.00	0.45	0.28	1.07	-0.33	0.26	0.25	-1.39
Menidia menidia	0.00	0.00	0.07	-0.49	0.00	-0.10	0.11	-0.06	-2.32
Apeltes quadracus	0.00	0.00	-0.10	0.82	0.07	-3.35	-0.06	-0.38	-0.02
Anguilla rostrata	0.00	0.04	-0.14	-0.06	-0.05	-0.03	-0.08	-0.05	-0.20
Gobiosoma spp.	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	-0.14
Lepomis spp.	0.00	-0.08	-0.73	0.00	0.00	0.00	0.00	-0.12	0.00
Tautogolabrus adspersus	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00
Pungitius pungitius	0.00	0.00	-0.93	0.00	0.00	0.00	0.00	-0.13	-0.02
Pseudopleuronectes americanus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.06
Crustaceans									
Palaemonetes pugio	822.90	140.13	-15.76	274.51	34.67	3.46	77.97	191.13	203.17
Crangon septemspinosa	0.00	0.00	0.00	0.00	1.71	-2.79	0.03	-0.15	-1.34
Callinectes sapidus	0.06	0.00	0.00	0.10	0.21	-1.52	0.23	-0.13	0.12
Carcinus maenas	0.00	0.00	0.00	-0.23	0.00	-0.06	-0.78	-0.15	0.01
Pagarus longicarpus	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.02	-4.40
Libinia emerginata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02
Limulus polyphemus	0.00	0.00	0.00	0.00	0.00	0.00	-0.02	0.00	0.00
Panopeus spp.	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-1.10
	Unadjusted								
Fish Density	6.21	24.00	-9.24	-80.72	21.76	-4.46	-5.67	-6.87	-18.43
Fish Richness	-1	1	-1	1	0	-2	-1	-0.43	-2.00
Crustacean Density	822.96	140.13	-15.76	274.38	36.60	-0.91	77.58	190.71	196.50
Crustacean Richness	1.00	1.00	0.00	1.00	2.00	0.00	1.00	0.86	-0.50
Nekton Density	829.17	164.13	-25.21	193.67	58.36	-5.37	71.92	183.81	178.07
Nekton Richness	0	2	-1	2	2	-2	0	0.43	-2.50
P. pugio to Fish Ratio*	60.05	5.61	0.73	33.62	1.19	0.16	3.93	9.91	18.50
	Adjusted								
Fish Density	24.64	42.43	9.19	-62.29	40.19	13.97	12.76	11.56	
Fish Richness	1.00	3.00	1.00	3.00	2.00	0.00	1.00	1.57	
Crustacean Density	626.45	-56.38	-212.26	77.88	-159.90	-197.41	-118.92	-5.79	
Crustacean Richness	1.50	1.50	0.50	1.50	2.50	0.50	1.50	1.36	
Nekton Density	651.10	-13.95	-203.28	15.59	-119.71	-183.44	-106.15	5.74	
Nekton Richness	2.50	4.50	1.50	4.50	4.50	0.50	2.50	2.93	
P. pugio to Fish Ratio*	41.54	-12.90	-17.77	15.11	-17.31	-18.34	-14.57	-8.60	

3.4 Marsh platform integrity and community composition

Current (2020) marsh integrity according to IMI was variable across Restoration marshes, with IMI values for four Restoration marshes falling within the lower quartile of IMI Reference values, and values for one marsh falling in the upper Reference quartile (Figure 3). Overall, the difference in IMI values between Restoration and Reference marshes was not significant, although on average, the cover of *Phragmites* (as a rapid-assessment cover type) was higher and the cover of *Meadow High Marsh* was lower for Restoration marshes than for Reference marshes (Table 8). Bar graphs (Figure 3) reveal that SAMI and POTT, two marshes with higher IMI scores, had greater-than-average cover of *Salt Shrub*, whereas the five lowest-scoring marshes had greater-than-average cover of *Phragmites*. Gooseneck Cove, the lowest-scoring marsh, lacked *Salt Shrub* and *Meadow High Marsh* cover completely, and had above-average cover of both *Phragmites* and *Die-off Bare Depression*.

There was little indication that MarshRAM metrics for aggregate disturbances, aggregate ecosystem functions and services, natural habitat richness, or bird use differed between the Restoration and Reference marshes (Table 8). Among the eight Restoration marshes, IMI was strongly correlated with the number of growing seasons since tidal restoration (Pearson, r=0.75, P=0.033) and with the number of restoration activities identified in Table 2 (Spearman rank, r_s=0.83, P=0.011). Our findings further suggest (although not significant at α=0.05) that IMI may also relate to the time span of restoration activities (r=0.57, P=0.136) and the MarshRAM *Disturbance* index (r_s=0.55, P=0.160), but not with the mean elevation of the marsh platform (r=0.13, P=0.757), or MarshRAM's migration *Replacement Ratio* (r=0.12, P=0.779) (Table 9). There was modest evidence that the cover of *Phragmites* per marsh declined with restoration age (r=-0.69, P=0.056), but not with the number (r=-0.23, P=0.294) or span (r=-0.19, P=0.325) of restoration activities.

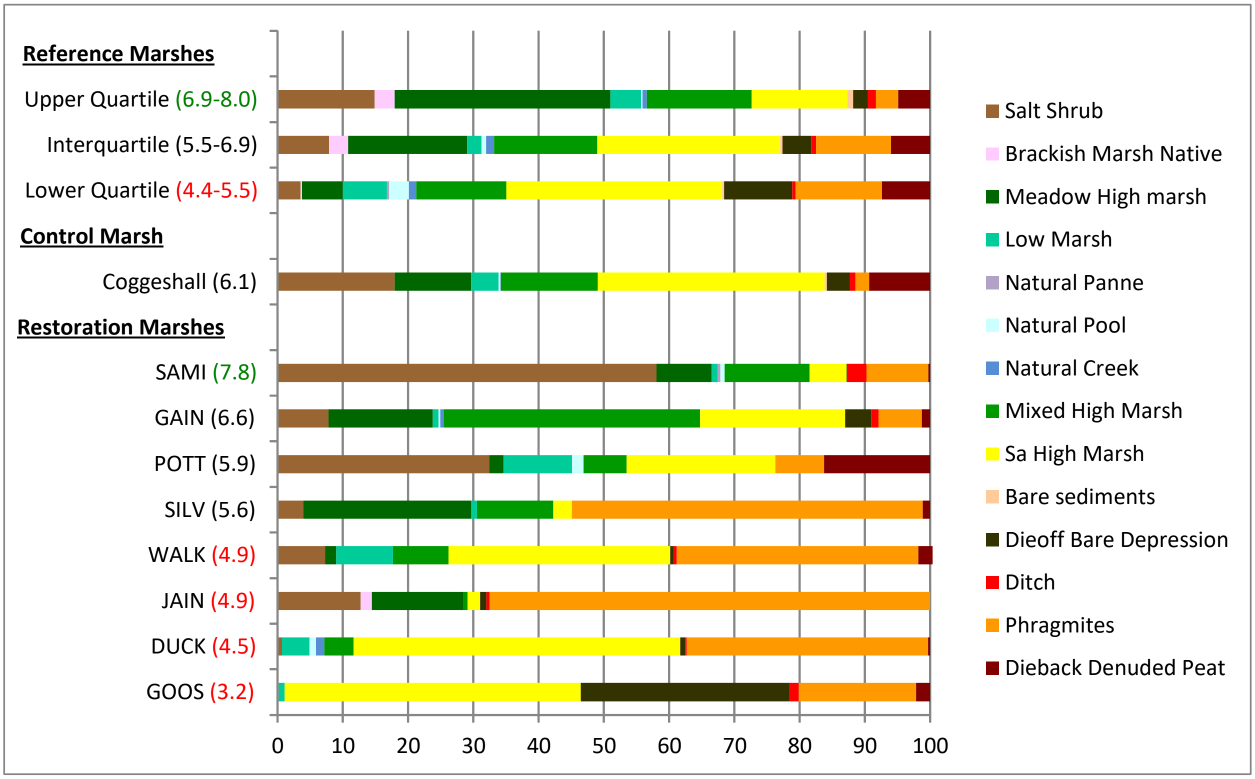

Figure 3. IMI scores (parenthetic) and relative proportions of IMI salt marsh cover types from 30 representative unrestricted salt marshes (Reference marshes), the marsh used as a Control marsh for vegetation and nekton analyses, and 8 Restoration salt marshes in Rhode Island. The IMI scores for Reference marshes are aggregated into quartiles, where the upper quartile (green scores) represents least-degraded salt marshes and the lower quartile (red scores) represents most-degraded salt marshes. The Restoration marshes are listed in descending order of IMI score.

Table 8. Mean values and independent t-test results for MarshRAM rapid assessment metrics across 8 Restoration marshes and 30 Reference marshes in Rhode Island. F & S = functions and services, Bird Density = sum of all birds counted/ha^2, # Sparrows Flushed = all salt marsh sparrows flushed during IMI transects, SD= standard deviation, and *Df* = degrees of freedom. Shading indicates a significant or nearly-significant difference.

	Mean Value (SD)				
Metric	Restoration	Reference	*t*	*Df*	*P*
IMI	5.4 (1.4)	6.1 (1.0)	1.17	36	0.095
% *Phragmites*	29.5 (22.9)	10.1 (13.2)	-0.31	36	0.003
% Meadow HM	8.5 (9.3)	19.8 (13.3)	2.37	36	0.032
% Sa HM	23.1 (18.9)	26.2 (15.3)	0.48	36	0.635
Disturbance	6.4 (0.9)	6.3 (0.9)	-0.15	36	0.882
Habitat Richness	5.0 (0.8)	5.0 (1.2)	-0.07	36	0.942
Ecosystem F & S	18.1 (3.6)	16.2 (3.8)	-1.27	36	0.211
# Birds Observed	17.3 (22.4)	19.1 (20.2)	0.2	36	0.839
# Sparrows Flushed	2.0 (2.9)	2.8 (4.6)	0.46	36	0.647

Table 9. Values for metrics describing 8 salt marsh Restoration marshes in Rhode Island. Metrics and their sources are described in the Methods.

Marsh	IMI	Disturbance Index	Replacement Ratio	Mean Elev. (m)
SAMI	7.8	7.8	45%	0.62 (0.15)
GAIN	6.6	6.6	38%	0.32 (0.15)
POTT	5.9	7.4	82%	0.45 (0.23)
SILV	5.6	5.5	46%	0.81 (0.23)
WALK	4.9	5.1	92%	0.65 (0.21)
JAIN	4.9	6.2	44%	1.01 (0.15)
DUCK	4.5	5.7	78%	0.49 (0.17)
GOOS	3.2	6.5	25%	0.25 (0.27)

4. Discussion

4.1 Conditions at the Restoration marshes compared with Reference marshes

If we consider current marsh integrity as the endpoint of restoration, restoration outcomes were highly variable, although condition and functionality was similar among Restoration and Reference marshes on average. According to our index of marsh integrity (IMI), half of the Restoration marshes fell among the most-degraded marshes in our Reference sample, suggesting modest restoration outcomes. The integrity of one marsh (GOOS), in fact, was lower than any of the Reference marshes, reflecting its failure to regain cover of native vegetation after tidal restoration. The Restoration marshes with integrity scores indicating modest (3) or least-degraded (1) condition suggest much better outcomes. Earlier work has found that a minimally-disturbed historic New England salt marsh, such as described by Chapman (1938), would have an IMI of approximately 9.0 (Kutcher 2019), suggesting that one marsh, SAMI, may even be approaching historic integrity, which is often viewed as a definitive restoration benchmark. Tidal restoration projects typically aim to reduce invasive plant species cover and restore native marsh species (Roman and Burdick 2016 and citations within), and the higher cover of *Phragmites* and the lower cover of *Meadow High Marsh* across the Restoration marshes in comparison to the Reference marshes therefore suggests that, on average, restoration efforts fell somewhat short of full recovery. Still, the lack of evident differences in our measures of integrity, disturbance, ecosystem functions and services, bird use, or burrowing-crab density between the Restoration and Reference marshes indicates that they are now functioning similarly to surrounding unrestricted marshes, suggesting overall restoration success.

4.2 Inundation stress confounds recovery

Our vegetation analysis suggests that restoration efforts promoted recovery across all the Restoration marshes, but inundation stress ultimately dampened or prevented full recovery in most cases. The Restoration marshes varied in vegetation recovery, with most gaining *I. frutescens* and *S. alterniflora*, and losing *P. australis*, indicating recovery, while on average they lost *S. patens* and gained bare ground, a trend away from recovery. However, in relation to the Control marsh, all marshes trended toward recovery in most indicators. Our Control marsh lost *S. patens* and *I. frutescens*, and gained *P. australis* and bare ground, trending toward degradation in all indicators except *S. alterniflora* over the study period. But even the gain in *S. alterniflora* concomitant with the loss of *S. patens* and gain of bare ground on the high marsh has been identified as a sign of inundation stress thought to be associated with sea-level rise (Raposa et al. 2015); therefore, gains in *S. alterniflora* may not be a sign of vegetation health for the Control marsh nor an entirely positive sign of recovery for the Restoration marshes. Four of the Restoration marshes gained *S. alterniflora* and lost *S. patens*, and two of those also gained bare, signaling a trend away from recovery. However, when adjusted for contemporaneous changes at the Control marsh, all marshes but one gained *S. alterniflora*, all but one gained (or lost less) *S. patens*, and all marshes gained *I. frutescens*, indicating vegetation recovery promoted by restoration. And, aggregating the indicators (using our index, *Magnitude*), all Restoration marshes trended toward recovery when adjusted for the Control marsh. Altogether, these findings suggest that restoration actions generally promoted vegetation recovery, while background factors, such as inundation stress related to sea-level rise, were working against recovery and toward salt marsh degradation, consistent with recent findings in Southern New England (Watson et al. 2016).

This trend was also apparent in our similarity analysis of the vegetation data. All marshes trended toward original target conditions (Control Before), but most (six of eight) ended up even more similar to current conditions at the Control marsh (Control After), which was characterized less by *S. patens* and more by bare ground than Control Before. SAMI, a marsh with highly-manipulated hydrology, was the only marsh that was considerably more similar to the initial reference target than to current conditions. JAIN and SILV resembled Control Before conditions similarly to Control After; these marshes were the highest in elevation. These factors suggest less susceptibility to inundation stress and therefore less convergence with the stressed Control marsh over time.

Nekton also seemed to respond to both restoration efforts and inundation stress. Only crustacean richness increased over time at the Restoration marshes, but when accounting for losses at the Control marsh, the Restoration marshes showed relative improvements across all measured indicators: fish density, fish and crustacean richness, and the ratio of *P. pugio* to fish, indicating restoration effects. Although modest, our finding of increasing fish density with the magnitude of overall vegetation recovery suggests that fish responded positively to native vegetation growth when coupled with a loss of *P. australis* and bare ground, suggesting a positive response to vegetation recovery advanced by restoration. But a pervasive increase in the density of *P. pugio* overshadowed any gains in fish density, as evidenced by the increased ratio of *P. pugio* to fish at every Restoration marsh and at the Control. *P. pugio* was correlated with increasing *S. alterniflora* and did not correlate with a decline in fish density, indicating that *P. pugio* was not competing with the fish but rather reacting to an increase in the low marsh grass *S. alterniflora*, which could be explained by vegetation recovery, inundation stress (Warren and Niering 1993, Raposa et al. 2015), or another factor such as eutrophication (Nixon and Oviatt 1973, Deegan et al. 2012). Earlier work has linked an increase in the ratio of *P. pugio* to fish with human disturbances in the surrounding watershed, suggesting that ambient habitat degradation over the study period may have contributed to gains in both *S. alterniflora* and *P. pugio* (James-Pirri et al. 2014).

Burrowing crabs also apparently benefitted from inundation stress. Burrow density was strongly correlated with the proportions of dieback denuded peat at the site level, and with the proportion of bare ground at the plot level, both indications of vegetation loss from inundation stress. This is consistent with recent findings that burrowing crab abundance is promoted by inundation stress on the marsh platform associated with sea-level rise (Crotty et al. 2017, Raposa et al. 2018b). POTT had a higher density of burrows than the other marshes, even as it was in higher-than-average condition according to IMI. But, POTT had the highest proportion of low marsh, the main habitat used by burrowing crabs, and the highest proportion of dieback denuded peat, a cover-type associated with crab overabundance and edge erosion, among the Restoration marshes (Altieri et al. 2012).

Considering recent predictions of pervasive and increasing inundation stress (Watson 2016, Raposa et al. 2015), our findings suggest that hydrologic management may be an important factor in salt marsh restoration. SAMI had the highest integrity, was the only marsh to trend toward recovery in all vegetation indicators, and had the lowest crab-burrow density. Rigorous adaptive restoration work at SAMI included the installation of a highly-manipulated network of deep channels designed to quickly transport water through the marsh, which may have reduced tidal inundation stress and thus allowed native marsh species to expand. Similarly, the restoration at GAIN, another marsh with a higher integrity score, (vegetation and crab-burrow data were unavailable), included the installation of tide gates to limit the amount of water entering the marsh, and a perimeter ditch to quickly drain water from the marsh surface and soils. In contrast, GOOS had the lowest integrity, trended away from recovery in every vegetation indicator present at the site, and had relatively high crab-burrow density. The marsh platform at GOOS was subsided from decades of impoundment (resulting in the lowest elevation of all the marshes) and was exposed to the full extent of the tides upon restoration, suggesting that inundation stress from the reintroduction of tidal water prevented vegetation recovery. Our findings suggest that for low elevation marshes, manipulation of tidal flow across the marsh aiming to reduce inundation stress may be necessary to promote vegetation recovery and achieve expected restoration outcomes.

4.3. Restoration factors

4.3.1 Adaptive management

Our findings suggest that restoration outcomes were dependent on factors beyond the reintroduction of tidal flow. Tidal flow was restored at all of our marshes, yet vegetation recovery was highly variable. If we use marsh integrity in relation to our Reference marshes to gauge ultimate restoration outcomes, the strong correlations of our integrity index (IMI) with the number of adaptive management actions taken suggests that active and ongoing manipulations after initial restoration are strong determinants of restoration success. Earlier work found IMI to increase with elevation and migration potential among the Reference marshes (Kutcher et al. 2019), but there was no sign of these correlations among the Restoration marshes, further suggesting overriding site-specific influences of restoration. SAMI, a marsh that has been hydrologically modified using deep drainage and has seen multiple other adaptive management actions spanning 22 years, has exceeded the condition of the Control marsh and is among the highest-integrity salt marshes in Rhode Island, according to IMI. Restoration success at SAMI is supported by our other parameters, showing the highest magnitude of overall vegetation recovery, an increase in fish density and crustacean richness, and the lowest crab-burrow density, even as its elevation was medial among marshes.

The high integrity of GAIN further suggests the importance of both informed planning and adaptive management for restoration success. Although only a single year of adaptive management was conducted at GAIN, extensive planning preceded both the initial restoration and subsequent adaptive management (Golet et al. 2012). The initial restoration included intensive monitoring and hydrologic modeling, fill removal and grading, reconstruction of the historic stream network, mulching of Phragmites, planting of native marsh grasses, and the installation of tide gates to limit the influx of tidal water on the marsh, effectively clipping the upper limit of the tide in perpetuity. Based on findings from a year of hydrologic modeling and monitoring, the tide gates were re-adjusted, and a deep perimeter ditch network was installed to quickly drain the outer edges of the marsh to reduce inundation period and quickly carry away freshwater and nutrient inputs that might promote *Phragmites* expansion. Our findings indicate that, in the case of GAIN, intensive monitoring, modeling, and adaptive management for a year following the reintroduction of tidal water, in addition to long-lasting hydrological manipulation, were effective factors in promoting recovery.

4.3.2 Age of restoration

Our restoration age analyses provided insights on rates of marsh recovery and sustainability. Our findings that older restorations had higher community-cover integrity, lower *Phragmites* cover, and higher overall vegetation recovery than younger restorations support earlier studies suggesting that tidal marsh restoration may take upwards of 20 years to recover (Warren et al. 2002). Although we did not compare our findings to short-term restoration outcomes, we are confident that short-term monitoring data would not have adequately characterized the recovery trajectories of our long-term assessment. Habitat self-sustainability is a key tenet of ecological restoration (Gann and Lamb 2006), and our long-term assessment was conducted across a timescale sufficient to provide a clear picture of the opposing influences of restoration activities and ambient stressors that may threaten the salt marshes persistence in the future. The findings support our initial assumption that marsh restorations should be assessed on a longer timescale than typical project grants allow, and demonstrate the need for long-term state programming positioned to assess the outcomes of salt marsh restoration at this important timescale so that management strategies can adapt as environmental conditions change.

4.4 Phragmites management implications

Phragmites management remains problematic for restoration practitioners. Even as *P. australis* declined in six of seven Restoration marshes we assessed for vegetation, five of those marshes still have a higher proportion of *Phragmites* community cover than the average most-degraded marsh in our Reference sample. For example, DUCK and JAIN remain among the lowest integrity marshes among the Restoration marshes, even as their *P. australis* cover declined 47.5% and 34.0%, respectively, according to our plot data. And, with low cover of Sa High Marsh, Die-off Bare Depression, and Die-back Denuded Peat, indicating lower inundation stress, the low integrity scores of JAIN and SILV almost entirely stem from continued *Phragmites* domination. These findings indicate that, while restoration actions helped to mitigate *P. australis* and support native species, most were unable to bring *Phragmites* community cover back to ambient (unrestricted) levels.

The ability of restoration activities to reduce the cover of *P. australis* was variable and was not predictably responsive to restoration effort. For example, according to plot data, *P. australis* did not decline at SILV despite mulching and herbicide treatment, whereas DUCK saw the greatest decline in *P. australis* even though no mulching or herbicide was used. Variability in *P. australis* response to tidal restoration may be better explained by ambient factors than restoration effort in these cases. SILV is a valley marsh, terminating a perennial stream that drains a highly-developed suburban watershed, where excessive nutrient inputs have been documented (Herron and Green 2012). As a fast-growing species with high above-ground-biomass, the efficiency of *P. australis* to outcompete native marsh species is greatly enhanced under high-nutrient conditions, particularly in areas where salinity is below 20 ppt. where salt stress to *P. australis* is limited (Minchinton and Bertness 2003, Meyerson et al. 2009, Uddin and Robinson 2018), such as on the banks of the freshwater stream at SILV. Under this same mechanism, the persistence of *P. australis* at WALK might be explained by runoff from an adjacent, uncovered municipal composing facility, along with eutrophic conditions in the surrounding estuary. In contrast, the watershed of DUCK is comparatively small without substantial freshwater and nutrient inputs to magnify the competitive advantage of *P. australis*, allowing the full salinity of the mesohaline tidal water to more-effectively stress and kill the *P. australis*. But even at DUCK, the concurrent gain in bare ground indicates that while a regular influx of mesohaline water may suppress and kill *P. australis*, it may also inhibit the re-establishment of native vegetation at lower elevations of the marsh, suggesting a tradeoff that may require a nuanced approach to restoring hydrology that introduces high-salinity tidal water for an inundation period that is simultaneously detrimental to *P. australis* and tolerable for native species.

Overall, our findings suggest that reintroducing mesohaline water to a salt marsh system can reduce the foothold of *P. australis* in areas of direct, regular tidal inundation, but in many cases, successful *P. australis* management may require a broader multi-prong approach that includes a nuanced reintroduction of tidal water, efficient export of freshwater, and a reduction in nutrient inputs. Even if *P. australis* is mulched or treated with herbicide, there is little hope of native species sustainably recolonizing the marsh if conditions do not favor their competitive advantage and survival over the long term. In such cases that such longstanding physical and chemical environmental conditions cannot be restored, restoration of native salt marsh vegetation may not be viable.

4.5 Efficacy of parameters for long-term restoration assessments

4.5.1 Vegetation community composition and crab-burrow density

Vegetation composition was a highly informative parameter for assessing restoration outcomes. Vegetation predictably responds to management activities and stress, and can indicate impacts of both episodic and gradual events (Lopez and Fennessy 2002, Kutcher and Forrester 2018, Raposa et al. 2018a); this made it particularly useful for assessing the outcomes of restoration activities and long-term changes in inundation duration. Vegetation is also the foundation of many salt marsh functions and values such as overall and soil productivity, accretion, habitat cover, and shelter, making changes in its structure and composition directly indicative of salt marsh health and, thus, restoration success. Our vegetation indicators captured major shifts in dominant vegetation types that are the foci of the common restoration goals of mitigating *P. australis* and restoring native vegetation. And, the BACI design allowed us to clearly separate vegetation responses to restoration from long-term ambient effects.

Although the vegetation data were effective for assessing restoration effects, inconsistencies in the collection and documentation of the original data complicated analysis and interpretation. Most marshes generally followed Roman et al. (2001) protocols, but pre-restoration vegetation data could not be located for GAIN, while DUCK used smaller plots and unconventional ocular estimation categories, JAIN and SAMI had few (13, 14) plots, and plots at JAIN were unevenly distributed across the restoration area. DUCK and SAMI used ocular categorization whereas five other projects used point-intercept to estimate species cover. There was no documentation to determine whether 'bare ground' was consistently collected across marshes, requiring us to estimate bare ground for each plot at some marshes. Additionally, we were unable to find station markers for some of the original plots, requiring us to estimate plot locations from various field notes, coordinates, and maps. For long-term assessments, permanently marking stations would require markers that are highly-resistant to weathering and ice scour, such as tall, deeply-driven PVC stakes with identification markings that will not disappear over time. We interpreted and adjusted the data as necessary to address these issues, and are confident that the trends in our indicators are real and indicative of restoration effects. Still, data collected following clearly-defined and carefully-documented protocols would have made the analysis easier and more robust. We recommend strict application of the Roman et al. (2001) protocol and point-intercept method for assessing long-term salt marsh restoration outcomes. We further recommend tallying crab burrows during vegetation surveys; this takes a small amount of additional time per plot and can provide useful additional information on edge and inundation stress.

4.5.2 Nekton community composition

Collecting data on nekton communities can also provide useful information on long-term marsh responses to restoration, but it requires training, substantial effort, and long-term sampling rigor to be useful. Nekton are an important component of salt marsh and estuarine food webs, as they transfer energy from primary producers to higher consumers within and beyond the salt marsh (Kneib et al., 1986). Thus, the monitoring of nekton communities can increase our overall understanding of marsh functional responses to restoration. Our pre- and post-restoration data collection was conducted by the same experienced team, using the same protocols, collecting the data two times per year across 6-23 stations at each marsh. We were able to demonstrate a clear pattern of increasing *P. pugio* across all marshes, but few other trends were clear. Nekton are highly mobile, and variability was typically high across stations per marsh. Additionally, the successful use of nekton traps is sensitive to technique and training, requiring a long, accurate throw of the trap without disturbing the nekton. This can be difficult on windy days, where the wind can change the trajectory of the trap, and in shallow water, where nekton are particularly sensitive to surrounding movements. Although we strongly support the collection of nekton data for long-term monitoring of salt-marsh restoration projects, we highly recommend that the collection of useful data requires training and practice with using throw traps.

4.5.3 Rapid assessment data

Rapid assessment data were useful for placing marsh-recovery information from our before-after biological monitoring into perspective. Understanding the condition of recovered marshes relative to surrounding unrestored marshes revealed the extent to which restoration returns the marsh to pre-restriction conditions. Coupling our index of marsh integrity with vegetation indicators resulted in a more holistic picture from which it was possible to assess restoration success and evaluate effects of restoration effort and time, and consistent agreement across the methods (wherein marshes with higher vegetation recovery were assessed to have higher integrity) increases confidence in our findings. Additionally, the rapid assessments provided data on ecosystem functions and services, disturbances within the marsh and surrounding landscape, and bird use that was analyzed against other restoration data to provide additional information on restoration response. We recommend that salt marsh restoration programs collect rapid assessment and integrity data across multiple salt marshes to establish a reference baseline against-which to compare restoration marshes in any given region.

4.6 Conclusions

Multiple interacting factors contribute to the outcomes of salt marsh restoration, and the variability in results across Restoration marshes in this study demonstrates the need to investigate the efficacy of restoration activities in the context of overarching ambient factors that may influence the trajectory of recovery. The combination of time-series (BACI) and rapid assessment data allowed us to assess the recovery of individual marshes to restoration in the context of conditions at a set of representative marshes in the region. We found that restoration outcomes depended on restoration effort and time of recovery. Marshes with low planning and adaptive management were less likely to achieve restoration goals than those with rigorous planning and adaptive management. Ambient environmental conditions of increasing sea-level rise largely counteracted vegetation recovery unless adaptive management efforts were used at a marsh after restoration to reduce inundation stress, which may not be possible at some marshes. Only two marshes with highly-manipulated hydrology, multiple other restoration actions, and longest restoration time (22 years) surpassed the integrity of our Control marsh and the average integrity of our Reference marshes, according to the IMI. For future tidal restoration projects, we suggest the use of hydrologic modeling and adaptive hydrologic modification, coupled with other strategies to alleviate pervasive inundation stress and reduce *P. australis* proliferation in order to best facilitate the establishment and sustainability of native high marsh species.

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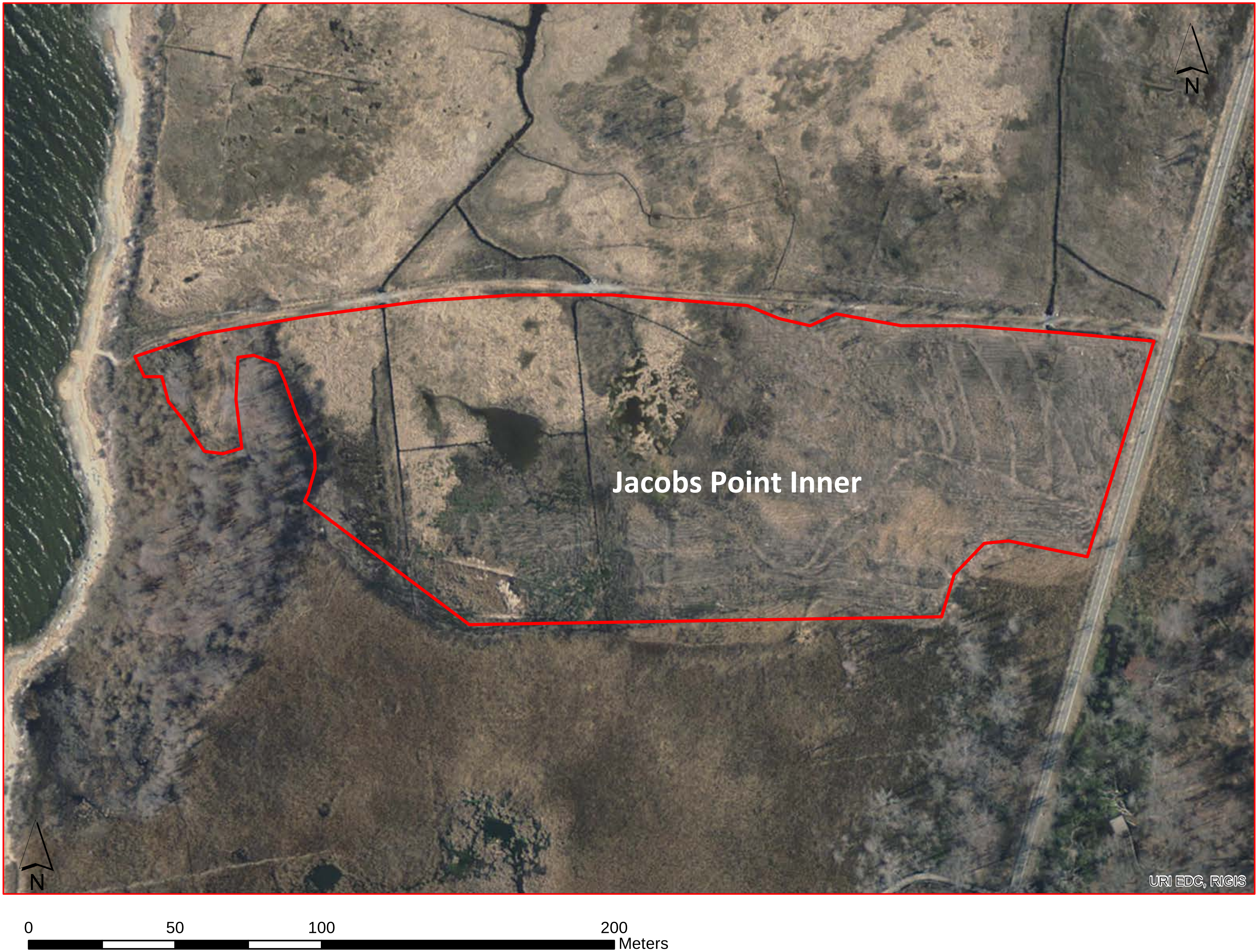
Appendix A

Eight mature Restoration marshes and a Control marsh assessed in 2020

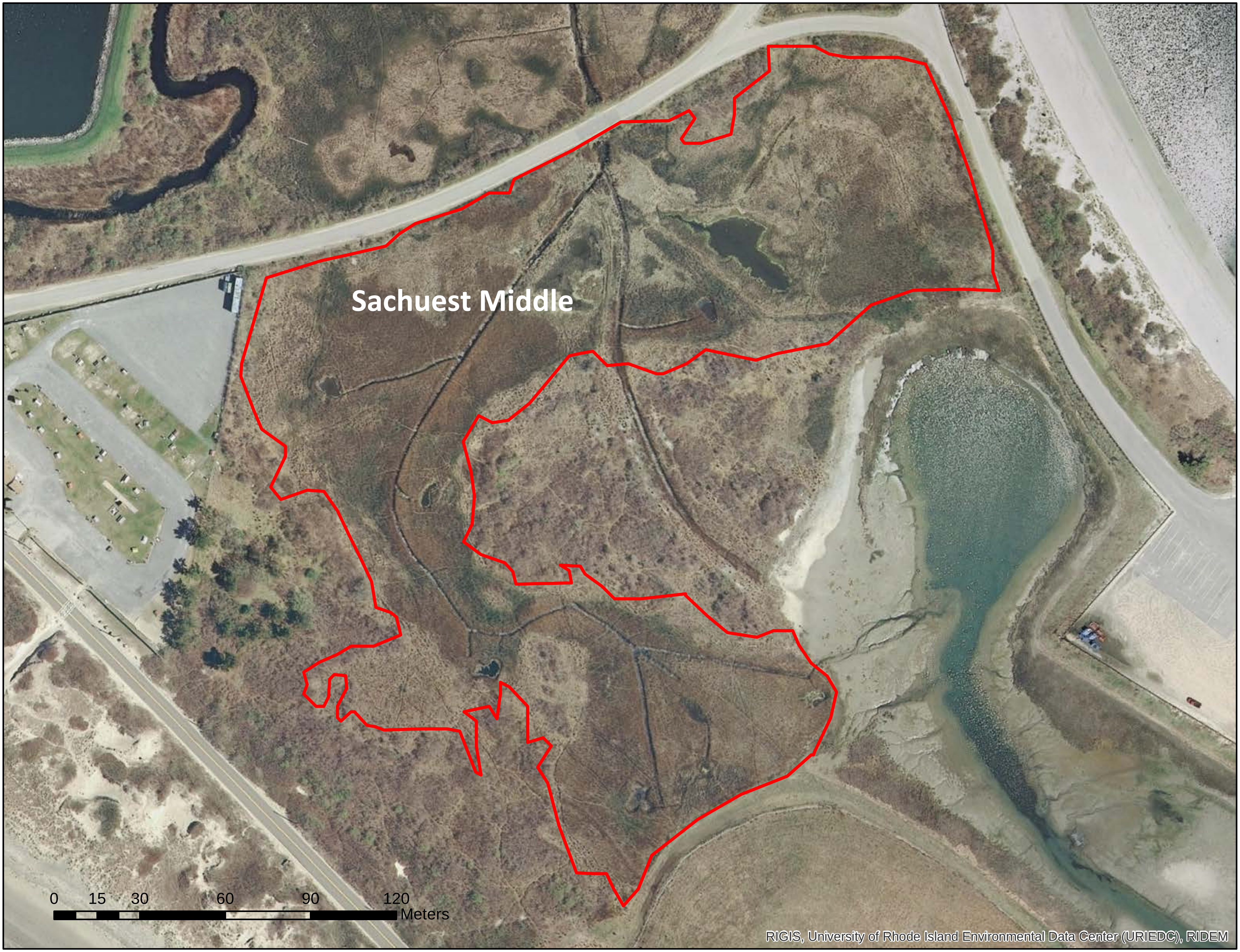
Duck Cove
0 12.5 25 50 75 100
Meters
RIGIS, University of Rhode Island Environmental Data Center (URIEDC), RIDEM


Galilee Inner
0 50 100 200 300 400
Meters
RIGIS, University of Rhode Island Environmental Data Center (URIEDC), RIDEM

Gooseneck Cove
0 40 80 160 240 320
Meters
RIGIS, University of Rhode Island Environmental Data Center (URIEDC), RIDEM


N
Jacobs Point Inner
N
URI EDC, RIGIS
0
50
100
200
Meters

Potter Pond
0 15 30 60 90 120
Meters
RIGIS, University of Rhode Island Environmental Data Center (URIEDC), RIDEM


Sachuest Middle
0 15 30 60 90 120
Meters
RIGIS, University of Rhode Island Environmental Data Center (URIEDC), RIDEM


Silver
Creek
0 30 60 120 180 240
Meters
RIGIS, University of Rhode Island Environmental Data Center (URIEDC), RIDEM


Walker
Farm
N
0
50
100
200
300
400
500
Meters

Coggeshall
Control Marsh
N
N
0 50 100 200 300 400 500 600
Meters