2022

A rapid method to assess salt marsh condition and guide management decisions

Thomas E. Kutcher
University of Rhode Island

Kenneth B. Raposa

Charles T. Roman
University of Rhode Island

Citation/Publisher Attribution

Kutcher, T. E., Raposa, K. B., & Roman, C. T. (2022). A rapid method to assess salt marsh condition and guide management decisions. *Ecological Indicators, 138*, 108841. https://doi.org/10.1016/j.ecolind.2022.108841
Available at: https://doi.org/10.1016/j.ecolind.2022.108841

A rapid method to assess salt marsh condition and guide management decisions

Creative Commons License

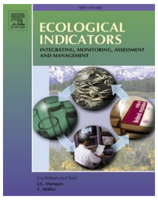
This work is licensed under a Creative Commons Attribution-Noncommercial-No Derivative Works 4.0 License.

Ecological Indicators 138 (2022) 108841

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind



A rapid method to assess salt marsh condition and guide management decisions

Thomas E. Kutcher [a,*], Kenneth B. Raposa [b], Charles T. Roman [c]

[a] *Rhode Island Natural History Survey, University of Rhode Island, Kingston, RI, USA*
[b] *Narragansett Bay National Estuarine Research Reserve, Prudence Island, RI, USA*
[c] *University of Rhode Island, Department of Natural Resources Science, Kingston, RI, USA*

ARTICLE INFO

Keywords:
Salt marsh
Rapid assessment
Ecological restoration
Marsh migration
Inundation stress
Sea-level rise

ABSTRACT

Salt marshes are increasingly vulnerable to degradation and loss from accelerating sea-level rise and other pervasive disturbances, spurring a need for broad, science-based information to guide management. The Salt Marsh Rapid Assessment Method, MarshRAM, was designed to address this need by documenting information characterizing salt marsh type, setting, ecological value, disturbance, integrity, and opportunity for landward migration at the site scale. We used the method to collect information from onsite and remote observations of thirty-one (31) salt marshes in Rhode Island, USA. MarshRAM's *Wetland Disturbance Index* is a checklist that ranks the intensity of individual and cumulative human disturbances, while the *Index of Marsh Integrity* (IMI) is generated using a novel walking-transect approach to rapidly characterize site-wide vegetation-community composition. The IMI was designed to reflect ecological response to direct disturbances and inundation stress, and our finding that IMI strongly correlates with cumulative disturbance + marsh platform elevation indicates it works as intended. A strong correlation between IMI components and historic marsh loss suggests that salt marsh community cover can also serve as an indicator of salt marsh resilience. Our study marshes diverge from accounts of historic New England salt marsh conditions in that meadow high marsh species no longer dominate the high marsh zone, *Spartina alterniflora* is now the dominant high marsh species, and severe edge erosion and invasion by *Phragmites australis* are ubiquitous. We demonstrate how MarshRAM data can be analyzed to inform restoration and conservation strategies and policy decision-making. For example, our findings suggest that inundation stress is strongly impacting marsh platform integrity, high-marsh vegetation loss is a strong indicator of degradation and vulnerability, and unassisted landward marsh migration may already be promoting resilience to inundation stress. We suggest adapting MarshRAM to meet the management needs of other regions or broader applications.

1. Introduction

Salt marshes are valuable to people and wildlife but are highly vulnerable to human disturbances. They are among the most productive ecosystems on earth, providing food and habitat for numerous species (Nixon 1980, Deegan et al., 2002, Gedan et al., 2009, Barbier et al., 2011), absorbing floodwater and wave energy to protect coastal properties (Shepard et al., 2011), and providing recreational opportunities and natural viewsheds. In developed regions, human disturbances, such as filling, hydrologic alterations, excessive nutrient loading, and invasive species have led to widespread salt marsh degradation and loss (e.g., Gedan et al., 2009, Gedan et al., 2011, Crotty et al., 2017, Wigand et al., 2014). More recently, inundation stress associated with accelerating sea-level rise has emerged as a key contributing factor to marsh degradation and loss, as evidenced by changes and declines of plant species composition and cover, and increases in unvegetated habitat and open water throughout northeastern US salt marshes (Donnelly and Bertness, 2001, Raposa et al., 2017b, Roman, 2017, Watson et al., 2017, Payne et al., 2019) and elsewhere, globally (Cahoon and Reed, 1995, Kirwan and Megonigal, 2013).

The many stressors facing salt marshes pose a challenge for management and conservation, requiring comprehensive science-based information capable of characterizing ecological status across various hydrogeomorphic settings and disturbance regimes. Salt marsh

---

* Corresponding author at: Postal Address: PO Box 1858, Kingston, RI 02881, USA.
E-mail address: tkutcher@rinhs.org (T.E. Kutcher).

https://doi.org/10.1016/j.ecolind.2022.108841
Received 27 January 2022; Received in revised form 1 April 2022; Accepted 1 April 2022

managers need to consider stress caused by multiple individual and cumulative human disturbances, response variability inherent in marsh type or setting, and the ecosystem functions and services that may be at risk from existing stressors and potential intervention actions. With sea-level rise simultaneously threatening marsh sustainability at broad spatial scales, managers may need to prioritize specific salt marshes where intervention actions are warranted, focusing on those that are most vulnerable, have the greatest potential for recovery, or offer the greatest value of ecosystem services. In this context, understanding the potential for landward migration at the site scale may help managers recognize to what extent conservation or alteration of adjacent lands may promote marsh sustainability. To evaluate this suite of possibilities for numerous marshes across a region, an efficient assessment method that can integrate diverse information could be a valuable tool for managers.

The Salt Marsh Rapid Assessment Method (MarshRAM), presented in this paper, provides managers with a single, comprehensive, efficient method to document and classify information on salt marsh physical and biological attributes, ecosystem functions and services, geomorphic and landscape setting, human disturbances, integrity and vulnerability, and landward migration potential. We designed MarshRAM to capture reliable site-level information in a single site visit, allowing numerous sites to be compared against each other or categorized based on condition, value, or other attributes (Fennessy et al., 2007). MarshRAM was developed and tested in Rhode Island, USA, but was designed with a format that can be adapted to salt marshes across broader geographic regions. The goal of this paper is to describe and evaluate MarshRAM, focusing on its ability to support salt marsh management.

2. Methods

2.1. MarshRAM overview and study design

MarshRAM builds on the New England Rapid Assessment Method (NERAM; Carullo et al., 2007, Wigand et al., 2011) and the Rhode Island Salt Marsh Assessment (RISMA; Ekberg et al., 2017), which were designed to characterize human disturbances and marsh-platform integrity, respectively. MarshRAM has six integrated parts, including checklists of observable characteristics and condition indicators, and models that estimate the condition of the marsh and its surrounding landscape. MarshRAM produces five indices reflecting (1) ecological and cultural value, (2) surrounding landscape condition, (3) the intensity of human disturbances, (4) marsh platform integrity, and (5) landward migration potential. We conducted MarshRAM at thirty-one (31) Rhode Island salt marshes (Fig. 1) in 2017 and 2018 during the peak of the growing season (mid-July through September). Data collection consisted of field and office components following the MarshRAM User's Guide (Kutcher, 2021) and using a dedicated field datasheet (Appendix A), as described below.

2.1.1. Observational checklists and models

Onsite and remotely-sensed observations are used to assess marsh characteristics, ecosystem functions and services, surrounding land use, and wetland disturbances. The *Marsh Characteristics* component documents, by discrete checklist categories, marsh area, position in the watershed, geomorphic setting and type, tide range, hydrology, edge exposure, and habitat diversity. This information helps categorize marshes by type and setting, which may affect analyses of how marshes respond to stressors. *Ecosystem Functions and Services* is used to estimate and rank the occurrence and importance of 12 ecosystem functions and services commonly cited in the literature (e.g. U.S. ACE, 2003), and the sum of those ranks is used as an aggregate measure of ecological and social value. As an opportunistic supplement to assessing ecosystem function, marsh-dependent and facultative birds are identified and tallied as they are observed when approaching sections of the marshes for the first time.

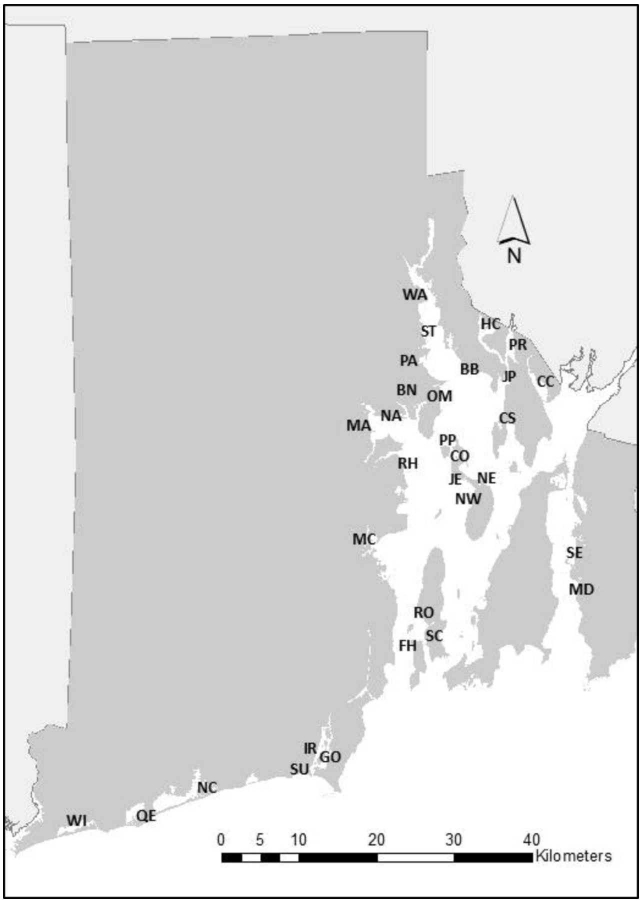


Fig. 1. Distribution across Rhode Island, USA, of 31 salt marshes assessed in 2017 and 2018 using MarshRAM. Refer to Table 2 for site codes.

Surrounding Land Use uses photointerpretation of recent aerial photography and field verification to estimate the proportion and intensity of human land uses within 150 m of the marsh-upland edge. Surrounding landscape integrity metrics are commonly applied in rapid assessments as coarse-but-reliable indicators of aggregate human disturbance (Fennessy et al., 2007), and prior studies have shown a strong relationship between wetland condition and surrounding land use (Bried et al., 2013, Kutcher and Bried, 2014, Kutcher and Forrester, 2018).

Wetland Disturbances estimates, categorizes, and ranks the intensity of 30-m buffer disturbances, tidal restriction, ditching and draining, anthropogenic nutrient inputs, filling and dumping, edge erosion, marsh crab burrowing, platform vegetation die-off, vegetation removal and soil disturbances, and the cover of invasive *Phragmites australis* as detailed in the field datasheet (Appendix A). *Wetland Disturbances* adapts observational NERAM metrics found to be effective in reflecting salt marsh platform condition in southern New England (Wigand et al., 2011), and adds metrics to assess crab-burrow density and edge erosion, which are degradation features thought to be related to recent inundation stress (Crotty et al., 2017, Ganju et al., 2017, Raposa et al., 2018). Ranking of intensity is coarse for most metrics (i.e., *None*, *Low*, *Moderate*, and *High* intensity categories, with decreasing scores assigned to each higher intensity level) to promote consistency among users. Additionally, checklists are included with most metrics for identifying specific and categorized sources of stress to support policy analysis.

We collected information for these components in the field and the office from direct observations, available reports, publications, and data, and through analysis of recent aerial images. To assess inter-user variability when completing the observational checklist parts of MarshRAM, two investigators separately completed the observational sections at nine marsh sites. In all cases, both investigators had been trained to understand the method and interpretation of all metrics.

2.1.2. Marsh community composition and Index of marsh integrity

This MarshRAM component adapts elements of RISMA (Ekberg et al., 2017) and floristic quality assessment (e.g., Bourdaghs et al., 2006), uses a novel sampling approach to estimate the relative abundance of typical salt marsh community cover types (*Marsh Community Composition*), and generates a biological index of salt marsh integrity (*Index of Marsh Integrity*, hereafter IMI). The relative proportion of marsh cover types typical in New England (Table 1) is quantified using eight transects traversing the marsh platform from the marsh-upland interface to the subtidal zone of a major water feature (bay, salt pond, major creek). Using the field maps as a guide, community types are sampled by walking each transect using repeatable, even paces. For every step across the marsh surface, the dominant cover type traversed is tallied as a single data point (Appendix A, Section E). For example, twelve steps through a salt shrub zone would be tallied as twelve *Salt Shrub* data points for analysis. The relative abundance of each cover type is then derived from the aggregate tallies of each type across all transects. The aim of this 'walking-transect' sampling approach is to efficiently and accurately characterize marsh community composition by estimating the relative abundance of the various marsh cover types across the marsh surface. Because the aim of the walking-transect approach is deriving relative abundance, step-length is inconsequential as long as it is consistent throughout the survey.

Table 1
Salt marsh community cover types (modified from Ekberg et al., 2017) and coefficients of community integrity (CCI) used to generate indices of marsh integrity (IMI) for 31 salt marshes in Rhode Island. CCI values indicate a gradient of low (0) to high (10) integrity. Cover types are listed in approximate order from upland interface to seaward edge, followed by typically less abundant features.

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 the invasive common reed *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

IMI assigns a coefficient to each salt marsh cover type based on its perceived indication of marsh degradation and habitat value. These *Coefficients of Community Integrity* (or CCI) were estimated through the consensus of a team of experienced salt marsh scientists using a standardized scoring system that rates each cover type by sensitivity to inundation and other stressors, and habitat value (Appendix B). Cover types with high sensitivity to stress and high habitat value were assigned coefficients approaching or equal to ten (10), whereas cover types indicative of stress and with lower habitat value were assigned coefficients approaching or equal to zero (0) (Table 1). For example, *Meadow High Marsh* was judged to be highly sensitive to both inundation stress and direct human disturbances, and to have high habitat value; this habitat was therefore assigned a coefficient of 10. In contrast, *Dieback Denuded Peat* was judged to be the outcome of stress and to have low habitat value; it was therefore assigned a coefficient of 0. The mean of the coefficients of all cover types, weighted by relative proportion of each type across all transects, constitutes the IMI, as below, where Tt (total tally) = the number of steps in each community type, tallied across all transects.

$$IMI = \frac{\sum(CCI \times Tt)}{\sum Tt}$$

We conducted the community composition sampling following transects drawn on field maps of recent aerial photography with the initial transect located from a random point and the remaining transects evenly spaced from the initial transect to span the entire marsh (Kutcher, 2021). We treated areas covered by wrack as the community type beneath the wrack at the time of the survey. As opportunistic information to supplement ecosystem-functions assessment, marsh-obligate sparrows (*Ammospiza* spp.) flushed while walking the transects were tallied to produce a coarse measure of sparrow density.

To test the precision of the *walking-transect* method across community types and users, the consistency of step-length and its effects on community-composition ratios and IMI scoring were evaluated for *Salt Shrub*, *Mixed High Marsh*, and *Phragmites* community types. Two investigators of varying step-length each walked transects of 20 steps across each community type, and the distance traversed was measured. This was replicated across five separate transects in each community type, and variability among the replicates and community types was analyzed for each investigator. Traversed distances were further applied to calculate relative proportions of each community type and IMI scores, for comparison against theoretical IMI scores that would result from taking perfectly-even steps across the various types.

2.1.3. Migration potential

The *Migration Potential* component (Appendix A, Section F) is designed to rapidly estimate and characterize landward marsh migration potential using a combination of remote-sensing data and field observations. This component evaluates geomorphic (e.g., slope and elevation), hydrologic (water features), vegetation-type (forested, grassland, etc.), and land-use (e.g., development type) features within a 60-m upland buffer from the marsh upland edge (to coincide with State of Rhode Island management authority of 200 feet). Information on the corridor condition is derived from interpretation of aerial imagery overlaid with high-resolution elevation data displaying 1-ft (30.5-cm) contours. Remote assessments are evaluated during the field surveys and adjusted if necessary. We used ESRI ArcMAP GIS software to generate and determine the area of sixty-meter buffers around each salt marsh assessment unit, and used the 5ft contour (approximately 1.5 m above mean high water) of the *RIGIS Contour Lines-2011 Statewide LiDAR* (available at https://www.rigis.org, accessed July-Sept 2018) to identify low-lying lands.

Migration Potential uses a preassigned coefficient of migration potential for each prescribed land-cover/elevation type. The coefficients range from zero (no migration potential) to 10 (high potential), based on the best professional judgement of a group that included salt-marsh scientists, restoration practitioners, and government regulators. For example, elevated developed land was judged to have no migration potential (0), whereas low-lying active farmland was judged to have moderate potential (5), and low-lying abandoned farmland was judged to have high migration potential (10). The *Migration Potential* score uses the weighted average of those coefficients to characterize the relative potential of land abutting the wetland to support landward migration. Two additional metrics are also estimated: *Migration Area*, defined as the area of surrounding land with moderately-high and high migration potential (i.e., land that would require little or no management action to facilitate migration), and *Replacement Ratio*, which relates *Migration Area* to the area of the existing marsh (estimating what proportion of the marsh might persist without intervention). These site-scale migration metrics can be used in conjunction with higher-resolution, regional-scale migration data, such as those produced using a Sea Level Affecting Marshes Model (Warren Pinnacle Consulting, Inc.), to inform management planning.

2.1.4. MarshRAM scoring

MarshRAM generates two separate condition indices reflecting marsh disturbances (*Wetland Disturbance Index*) and marsh integrity (IMI) (Appendix A). Scores for each metric and index range from 0 to 10, where scores approaching 10 indicate no observed indications of disturbance or degradation, and scores approaching zero indicate severe disturbance or degradation. The *Ecosystem Functions and Services*, *Migration Potential*, and *Surrounding Land Use* metrics are not incorporated into the MarshRAM condition indices, but are instead designed to be interpreted in conjunction with the disturbance and marsh integrity scores to inform management decisions. MarshRAM keeps size, setting, diversity, functions and services, and migration potential information separate from disturbance and integrity scoring because some of the former factors are inherent and can confound the effective assessment of wetland condition (Fennessy et al., 2007, Kutcher and Forrester, 2018).

2.2. Data analysis

We used Winstat (R. Fitch Software, 2008) for statistical analyses, except where noted. Pearson correlation was used to analyze IMI against MarshRAM tally data and surrounding land use data, and historic loss, elevation, and cover data from prior studies (Berry et al., 2015, Ekberg et al., 2017, Watson et al., 2017). Spearman rank correlation was used to detect correlations of IMI and its components with MarshRAM observational data to compensate for the ordinal nature of the observational metrics. Kruskal-Wallace H-test was used for inter-user variability analysis of the walking transects where assumptions could not be met for ANOVA analysis. We used Bonferroni adjustment of the alpha value in analyses where multiple comparisons increased the probability of an erroneous positive outcome. IBM SPSS Statistics (IBM Corporation) multiple regression analysis was used to test for interactive and additive effects of disturbance and median marsh elevation on IMI.

3. Results

3.1. MarshRAM logistics

Each MarshRAM assessment took one investigator and a field assistant a single day or less to complete. Office-based preparation of field maps and GIS investigations took less than one hour per marsh, and field surveys generally took between two and five hours, depending on the size of the site and difficulty in traversing the transects and perimeter of the marsh. Community-type transects ranged in length from 10 m to 417 m ($n = 248$, $\bar{x}=108$) and averaged 861 m total per eight transects per marsh (range = 123 to 2200 m), and the number of data points tallied (i.e., the number of steps traversed during transect surveys) averaged 973 per marsh.

3.2. Marsh characteristics and stressors

Study marshes ranged in size from 0.6 to 93 ha ($n = 31$, $\bar{x}=14.8$) and were distributed across back-barrier marsh (10 sites), open embayment (8), valley marsh (6), coastal lagoon (4), and open estuarine coast (3) geomorphic settings. The tidal water of 28 sites was polyhaline (>18 ppt.), one was mesohaline (5-18ppt.), and two were not measured for salinity. All sites were interpreted as having potential or evident value as wildlife habitat, fish and shellfish habitat, and carbon storage, and 17 were characterized as having potential or evident value for storm protection of property. The most common stressors in the surrounding landscape within the 150 m buffer were residential development (27 of the 31 sites), raised roads (19 sites), trails (11 sites), and recreational development (10 sites).

3.3. MarshRAM Index values

Wetland Disturbance Index scores ranged from 4.2 to 8.1 ($\bar{x} = 6.3$, $SD = 0.91$), and IMI scores ranged from 4.4 to 8.0 ($\bar{x} = 6.2$, $SD = 0.96$) (Table 2). IMI scores reflect relative community composition as depicted in Fig. 2. *Meadow High Marsh*, *Dieoff Bare Depression*, *Salt Shrub*, and *Natural Pool* most strongly influenced the IMI scores (Table 3). Lower-quartile IMI scores ranged from 4.5 to 5.5 and upper-quartile scores ranged from 6.9 to 8.1; these were used to assign categories of degradation in Table 4, which demonstrates a decision-support matrix relating the IMI categories to observed disturbance intensities and other management information. *Migration Area* ranged from 0.0 to 12.6 ha ($n = 31$, median = 2.5, $\bar{x}=3.4$, $SD = 3.3$), and *Replacement Ratio* ranged from 0.0% to 136% (median = 25%, $\bar{x}=35\%$, $SD = 34\%$).

3.4. Analysis of MarshRAM properties

IMI was modestly correlated with median marsh platform elevation (+) and weakly correlated with the *Wetland Disturbance Index* (+), but together, elevation and *Wetland Disturbance* had a strong additive influence on IMI (Table 5), and there was no indication of interaction between those two variables ($P = 0.781$). IMI was negatively correlated with the historic loss of vegetated marsh area reported by Berry et al. (2015), and the combined cover of *Meadow High Marsh* (−) and *Die-off Bare Depression* (+) predicted 79% of historic loss (stepwise regression, $F(2, 7) = 13.09$, $P = 0.004$, $R^2 = 0.79$, $R^2_{adj.} = 0.73$). In contrast, the *Wetland Disturbance Index* was not correlated with historic loss values (Pearson, $P > 0.05$). IMI was correlated with the MarshRAM observational metric *ponding and dieoff depressions* (+) (Spearman Rank, $r_s = 0.52$, $P = 0.002$, $n = 31$) but not with any other individual *Wetland Disturbance* metric ($P >$ Bonferroni-adjusted α of 0.005).

The % cover of *Meadow High Marsh* decreased with increasing cover of *Sa High Marsh* (Pearson, $r = -0.54$, $P < 0.001$, $n = 31$), and both *Meadow High Marsh* and *Sa High Marsh* decreased with increasing cover of *Phragmites* ($r = -0.54$, $P < 0.001$, $n = 31$ and $r = -0.54$, $P < 0.001$, $n = 31$, respectively). The cover of *Phragmites* was also correlated with the

Table 2
Site codes, MarshRAM index scores, marsh loss, and median elevation of 31 RI salt marshes. IMI and *Wetland Disturbance* scores span a 0–10 scale, where scores approaching 10 indicate little or no observed disturbance or marsh degradation, and scores approaching zero indicate multiple, strong observations of disturbance and degradation.

Site	Code	*Wetland Disturbance*	IMI	% Loss[1]	Median Elevation[2]
Barrington Beach	BB	5.9	5.7	ND	0.74
Brush Neck Cove	BN	7.5	6.6	ND	0.29
Chase Cove	CC	6.4	7.8	ND	0.69
Coggeshall	CO	6.6	6.1	ND	0.62
Colt State Park	CS	5.3	6.9	ND	0.70
Fox Hill	FH	7.7	6.7	5.9	0.45
Galilee Outer	GO	6.5	5.9	ND	0.60
Hundred-acre Cove	HC	6.0	6.5	8.9	0.59
Island Road North	IR	6.1	5.5	ND	0.49
Jacob's Point Outer	JP	5.7	7.9	ND	0.70
Jenny	JE	6.2	5.9	ND	0.53
Mary Donovan	MD	5.9	6.4	ND	0.33
Mary's Creek	MA	4.2	5.3	ND	0.54
Mill Creek	MC	7.3	7.2	ND	0.53
Nag East	NE	5.9	6.0	ND	0.64
Nag West	NW	6.4	6.1	ND	0.64
Nausauket	NA	7.4	5.9	ND	ND
Ninigret Control	NC	7.3	5.6	12.9	0.09
Old Mill Cove	OM	5.3	5.3	ND	0.38
Palmer River	PR	6.1	6.0	1.7	0.56
Passeonquis	PA	6.1	7.1	ND	0.75
Providence Point	PP	8.1	7.8	ND	0.64
Quonnie East	QE	5.5	4.6	21.2	0.23
Rocky Hill	RH	6.3	6.3	5.5	0.55
Round	RO	6.5	6.1	9.3	0.54
Seapowet	SE	4.8	4.9	10.6	0.65
Sheffield Cove	SC	7.4	8.0	ND	ND
Stillhouse Cove	ST	4.9	6.9	ND	0.57
Succotash	SU	5.7	5.3	11.3	0.30
Watchemoket	WA	5.7	4.4	ND	0.40
Winnapaug	WI	7.1	4.7	26.7	0.13

[1] Annualized loss of vegetated marsh area from 1981 to 2008 estimated using aerial photo-interpretation, derived from Berry et al. (2015);

[2] median elevation relative to NAVD88 from Watson et al. (2017); ND = no data available.

MarshRAM observational metric *anthropogenic nutrient inputs* (r_s = −0.48, P = 0.003, n = 31) but not with *filling and dumping*, *buffer encroachment*, or *Surrounding Land Use* (r_s > Bonferroni-adjusted α of 0.007).

The linear density (sparrows/m) of marsh-obligate sparrows flushed during IMI transects was not correlated with the *Wetland Disturbance Index*, IMI, or any singular observational disturbance metric, but a positive association of sparrow linear density with the number of ditch data points tallied along the transects was significant (*Pearson*, r = 0.58, P < 0.001, n = 31) considering a *Bonferonni*-adjusted α of 0.002.

The sum of ranks ascribed to *Ecosystem Functions and Services* was correlated with marsh area (r_s = 0.59, P = 0.0003, n = 31), but not with *Wetland Disturbance* or IMI (P > Bonferroni-adjusted α of 0.013 for both). Marsh area was not correlated with % historic loss (Berry et al., 2015), median elevation (Watson et al., 2017), MarshRAM *natural habitat diversity*, IMI, or the *Wetland Disturbance Index* (P > *Bonferroni*-adjusted α of 0.007 for all).

Migration Potential was strongly correlated with *Buffer Encroachment* scores (Pearson, r = 0.66, P < 0.001, n = 31) but only weakly associated with *Surrounding Land Use* (Pearson, r = 0.35, P = 0.056, n = 31), both of which decrease with increased disturbance. *Replacement Ratio* was inversely correlated with historic loss values (*Pearson*, r = -0.66, P = 0.039, n = 10).

3.5. *Inter-user variability analysis*

No differences were detected between investigators assessing the *Wetland Disturbance Index* (*dependent T-test*, T = -1.2, P = 0.26, n = 9), the sum of *Ecosystem Functions and Services* ranks (T = -0.13, P = 0.90, n = 9) or any of the component metrics/ranks of either index (*dependent T-test*, P > 0.05 for all that met statistical criteria; those not meeting criteria were identically-scored across sites by users). Mean inter-user differences for the *Wetland Disturbance Index* were<3% of the potential metric range of 10, and for the sum of *Ecosystem Functions and Services* ranks, mean differences were 6% of the potential range of 36.

Likewise, no differences were detected in the length of walking-transect steps across three structurally-distinct community types (*Salt Shrub*, *Meadow High Marsh*, *Phragmites*) for each of two investigators (*Kruskal-Wallace H-test*, $n = 3 \times 5$; H = 4.2, P = 0.117 for User 1 and H = 4.1, P = 0.127 for User 2). The two users had different overall step lengths (*Mann-Whitney U test*, Z = −2.6, P = 0.009), but because IMI is based on relative cover, inter-user variability of IMI values generated using the measured step lengths was < 1% of the IMI range (0–10).

4. Discussion

4.1. *MarshRAM as an indicator of Wetland condition*

MarshRAM's *Wetland Disturbances Index* and *Index of Marsh Integrity* (IMI) were designed to reflect different aspects of salt marsh condition. *Wetland Disturbances* quantifies aggregate observable marsh disturbances, whereas IMI was designed to reflect marsh-platform integrity as it responds to disturbances. In freshwater wetlands, several studies have documented a strong relationship between observational disturbances and integrity indices that are based on vegetation sensitivity to disturbances, such as Floristic Quality Assessment, upon-which IMI is partly based (e.g., Miller et al., 2006, Kutcher and Forrester, 2018). In contrast, only a weak relationship was evident between IMI and the MarshRAM disturbance index. However, IMI was designed to weight the stress of increased inundation period equally with the aggregate of observable disturbances (Appendix B). The response of IMI to marsh-platform median elevation (as a proxy for inundation stress), and the markedly-stronger response of IMI to elevation + disturbance, suggest that MarshRAM reflects the cumulative effects of inundation stress and other disturbances (nutrient stress, ditching, filling, etc.), as designed. IMI's strong response to elevation + disturbance supports earlier studies, which suggest that the impacts of certain disturbances (nutrient loading, ditching, crab over-grazing of *S. alterniflora*) on marsh integrity can be exacerbated or catalyzed by sea-level rise (Wigand et al., 2003, Wigand et al., 2014, Kirwan et al., 2016, Crotty et al., 2017). But the stronger correlation of IMI with elevation than with any other disturbance suggests that marsh communities may be responding primarily to an increased duration of marsh platform flooding that may be an outcome of accelerating sea-level rise (Raposa et al., 2017b, Watson et al., 2017, Payne et al., 2019). A more-precise measure of inundation stress, such as marsh elevation capital (i.e., elevation in relation to the tide frame), could further clarify the relationship between inundation, disturbance, and marsh integrity (Watson et al., 2017, Cahoon et al., 2019), but such precise measurements may be onerous to collect across multiple marshes for rapid assessments.

The significant correlation of IMI with marsh loss suggests that IMI may in-turn reflect overall marsh vulnerability to sea-level rise. IMI explains 61% of historic marsh loss among Rhode Island salt marshes assessed by Berry et al. (2015), and the most efficient model, comprising only the cover of *Meadow High Marsh* (−) and *Die-off Bare Depression* (+), explains 73% of marsh historic loss among those sites. Although the sample size for this trend is low (n = 10 marsh sites), this finding suggests that changes in high marsh community types may be a strong indication of marsh vulnerability, and is consistent with earlier findings that the ratio of unvegetated to vegetated salt marsh (UVVR) can signal

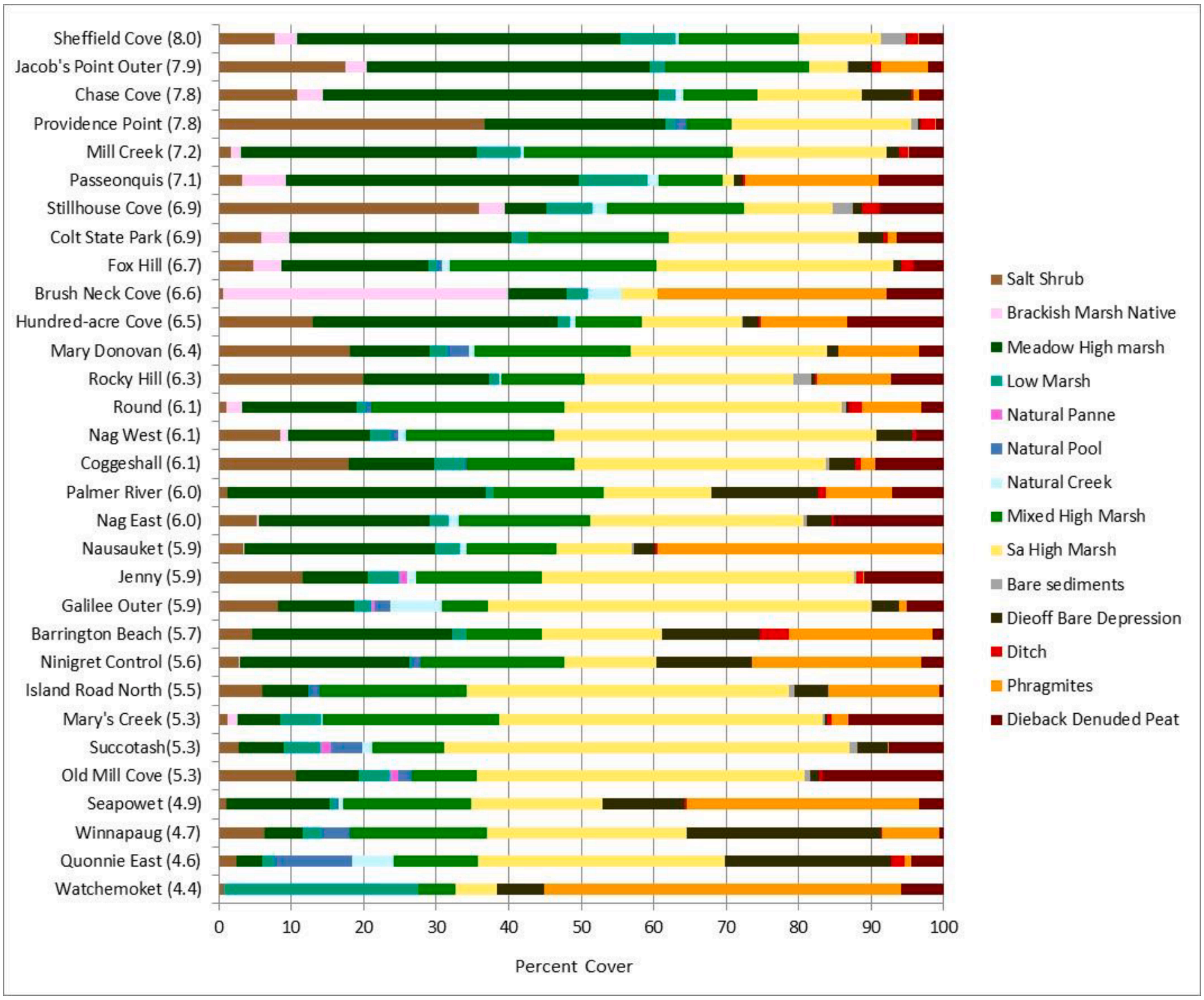


Fig. 2. IMI scores (parenthetic) and relative proportions of IMI salt marsh cover types from 31 salt marshes in Rhode Island; salt marshes are listed in descending order of marsh integrity according to IMI scores.

Table 3
MarshRAM community types and their mean cover across 31 RI salt marshes, sorted by Pearson correlation coefficients (*r*), indicating the relative influence of each type on IMI values across the marshes; i.e., the highest (+) and lowest (−) *r* coefficients indicate cover types that most strongly increase (+) or decrease (−) IMI scores.

MarshRAM Community	% Cover	IMI	
		r	*P*
Meadow High marsh	19.3	0.73	<0.01
Salt Shrub	8.7	0.46	0.01
Bare sediments	0.5	0.32	0.08
Ditch	0.8	0.24	0.19
Brackish Marsh Native	2.4	0.23	0.22
Mixed High Marsh	15.4	0.08	0.66
Dieback Denuded Peat	6	−0.11	0.54
Natural Creek	1.1	−0.11	0.54
Low Marsh	3.9	−0.18	0.32
Natural Panne	0.1	−0.24	0.18
Phragmites	9.8	−0.37	0.04
Sa High Marsh	25.7	−0.38	0.04
Natural Pool	1.1	−0.43	0.02
Dieoff Bare Depression	5.2	−0.53	<0.01

salt marsh vulnerability to increased inundation (Ganju et al., 2017, Wasson et al., 2019). Our findings further suggest that loss of high marsh vegetation from ponding and dieoff is a more influential mechanism of marsh loss than edge dieback (including creek expansion), which did not significantly contribute to our model explaining loss. Analysis of MarshRAM community data against rigorous historic loss data across a larger set of wetlands may help clarify these apparent trends.

Meadow High Marsh most-strongly influenced the IMI, further indicating its sensitivity to human disturbances, particularly increased inundation. And, *Die-off Bare Depression* and *Natural Pool* had the strongest negative correlations with IMI, even as their average cover across our study marshes was only 5.2% and 1.1% of total marsh area, suggesting that even a minor occurrence of marsh platform die-off may be an early indicator of declines in marsh integrity, and supporting recent evidence that runaway pond expansion may be an important mechanism of salt-marsh loss in the Northeastern U.S. (Mariotti et al., 2020). However, it is noted that natural marsh ponds are a valued marsh cover type, providing habitat for marsh fishes and birds.

In contrast to *Meadow High Marsh*, *Sa High Marsh*, which was the dominant cover-type overall in the sample (25.7%), only modestly influenced IMI variability (negatively) and did not contribute significantly to the historic loss model, even as it has been shown that *S. alterniflora* is often the initial marsh species to invade the meadow high marsh community under a regime of elevation deficits in relation to the tide frame (Warren and Niering, 1993, Raposa et al., 2017a). This may point to die-off in the high marsh as a more indicative tipping point in marsh degradation.

Table 4
Matrix depicting IMI marsh degradation categories (IMI Bin) in relation to categories of MarshRAM functions and services, marsh migration potential, intensity of human disturbances, and mean elevation (from Watson et al., 2017); MD = most-degraded, ID = intermediately-degraded, LD = least-degraded; AA = above average, A = average, B = below average summed ranks of MarshRAM *Ecosystem Functions and Services*; *Migration Area* = ha of adjacent land with moderately-high migration potential; *Replacement Ratio* = *Migration Area*/area of site; disturbance categories: X = low-intensity, XX = moderate-intensity, XXX = high-intensity; green, yellow, and red shading represent, respectively, upper-quartile, interquartile range, and lower-quartile categories of marsh resiliency or value.

SITE CODE	IMI Bin	Disturbance	Elevation	Functions and Services	Migration Potential	Migration Area (ha)	Replacement Ratio	Buffer	Impoundment	Ditching	Nutrients	Fill	Erosion	Crabs	Die-off	Mowing	Phragmites
Sheffield Cove	LD	Low	ND	A	High	1.5	92%	X		XX		XX	XXX				X
Jacob's Point, Outer	LD	High	High	A	Low	0.5	6%	XX		XX	XX	XX	XX	XX	X		XX
Chase Cove	LD	Mod	High	A	High	4.1	80%		X	XX	X	X	XXX	XX	X		X
Providence Point	LD	Low	Med	B	High	2.5	53%			XX			X	X	X		X
Mill Creek	LD	Low	Med	B	Mod	1.4	29%			XX	X		XXX	XX			X
Passeonquis	LD	Mod	High	A	Low	2.3	75%	X		X	XXX		XXX	XX		X	XX
Stillhouse Cove	LD	High	Med	B	Low	0.0	0%	XXX		XX	XX	XX	XXX	X	XX	X	X
Colt State Park	LD	High	High	A	Mod	8.2	39%	X		XXX	XX	X	XXX	XXX	X	X	X
Fox Hill	ID	Low	Low	A	Mod	3.9	25%	X		X		X	XX	X	X		X
Brush Neck Cove	ID	Low	Low	A	Mod	3.2	114%				XXX		XX		X		XX
Hundred-acre Cove	ID	Mod	Med	AA	Mod	1.3	20%			X	XXX		XXX	XXX	X	X	X
Mary Donovan	ID	Mod	Low	A	Mod	5.4	15%	X		X	XXX	X	XX	XXX	X	X	X
Rocky Hill	ID	Mod	Med	AA	High	5.0	29%	XX	XX	X	XX	X	X	X	X	X	X
Round Marsh	ID	Mod	Med	A	High	11.7	37%	X	X	XX	XX	X	XX	X	X		X
Nag West	ID	Mod	Med	AA	Mod	2.9	22%			XX		X	XXX	XXX	X	X	X
Coggeshall	ID	Mod	Med	A	Mod	7.7	38%			XX	X		XXX	XXX	X		X
Palmer River	ID	Mod	Med	AA	High	5.2	27%			XX	XX		XXX	XXX	XX		X
Nag East	ID	Mod	Med	AA	Mod	3.9	18%	X		XX	X	X	XXX	XXX	X	X	X
Nausauket	ID	Low	ND	B	Low	1.0	13%	X		XX	XX			X	X		XX
Jenny	ID	Mod	Med	A	Mod	3.8	30%	X		XXX		X	XXX	XXX		X	X
Galilee	ID	Mod	Med	B	Low	1.4	13%	XX		X		XXX	XXX		X	X	X
Barrington Beach	ID	Mod	High	AA	Mod	1.1	18%	X	X	XX	XXX	XX		X	XX		XX
Ninigret Control	ID	Low	Low	A	Mod	0.0	0%				XX		XXX		XX		XX
Island Road North	MD	Mod	Med	B	Low	0.4	29%	XXX			XXX	XX	XX		X		XX
Mary's Creek	MD	High	Med	B	Low	0.0	0%	XXX		XX	XX	XXX	XXX	XXX	XX	X	X
Succotash	MD	High	Low	A	Mod	6.5	16%	XX	X	X	XX	XX	XX	XXX	X		X
Old Mill Cove	MD	High	Low	B	Mod	2.0	73%	X		X	XXX	XX	XXX	XXX	XX		X
Seapowet	MD	High	Med	AA	Mod	12.6	14%	XX	X	XX	XX		XXX	XXX	XX	X	XX
Winnapaug	MD	Low	Low	A	Mod	0.0	0%	X		X	XX	X	XX		XX		X
Quonnie East	MD	High	Low	AA	High	5.3	19%			XXX	XX	XX	XXX	XX	XX		X
Watchemoket	MD	High	Low	B	Low	0.8	136%	XX	X		XXX	XX	XX	XX			XXX

Table 5
Pearson correlation coefficients (r) and probability values comparing MarshRAM IMI values with loss and elevation estimates from prior studies, and with latitude—Bonferroni adjusted $\alpha = 0.013$; *Wetland Disturbance* + Median Elevation represents the additive effect of the two prior metrics analyzed against IMI using stepwise regression (r reported rather than r^2 for comparison to other metrics); note that the *Wetland Disturbance* index decreases with increased disturbance. Values from Stillhouse Cove were removed from this analysis because a prior marsh-platform restoration may have affected how IMI values related to the Reference Indicators compared with the other non-restored sites.

Reference Indicators	IMI		
	n	r	P
Historic Loss	10	−0.78	0.008
Latitude	30	0.37	0.044
Median Elevation	28	0.53	0.004
MarshRAM *Wetland Disturbance*	30	0.44	0.016
Wetland Disturbance + Median Elevation	28	0.75	0.004

4.2. Condition of the study marshes; a historical comparison

Our study marshes diverge in community composition from historic accounts of southern New England salt marshes. In their seminal study, Miller and Egler (1950) detailed vegetation communities at a salt marsh complex surrounding Barn Island at the eastern-most border of Connecticut, USA (directly bordering Rhode Island). The un-ditched portions of the Barn Island marshes were reported to be dominated by *Juncus gerardii* and *S. patens* high marsh with fringing bands of low marsh and marsh-upland transition communities. The authors estimated that "as much as" 20% of those un-ditched areas comprised circular or nearly-circular *Pannes and Potholes*; the occurrence of *S. alterniflora* on the high marsh was reported to be restricted to those pannes. Considering 20% cover of *Panne and Pothole* evenly split among the authors' *Pothole* and *Panne* types, a fringing band of *S. alterniflora* low marsh (set conservatively at 10%), a fringe of marsh-upland interface (arbitrarily set at 10%), and the remaining 60% split among meadow high marsh *J. gerardii* and *S. patens* types (both classified as *Meadow High Marsh* by MarshRAM), an un-ditched late-1940s Barn Island marsh without invasive *P. australis* would have an IMI value of approximately 8.9. Similarly, in his historic description of plant zonation in New England salt marshes, Chapman (1938) presents a plan-view map of a representative back-barrier marsh, depicting approximately one third of the marsh covered by low-marsh *S. alterniflora* and the bulk of the remaining two-thirds covered by *S. patens*, *Distichlis spicata*, and *J. gerardii*, all three of which would be classified as *Meadow High Marsh* by MarshRAM. Small areas (~5%) of fringing "freshwater marsh" depicted on the platform are described as sometimes being dominated by *P. australis*. Applying IMI to the distribution of plant communities in Chapman's New England salt marsh, assuming 5% *P. australis*, would generate an IMI score of 9.0.

We acknowledge that the accounts by Miller and Egler (1950) and Chapman (1938) may not be fully representative of undisturbed historic marsh conditions; however, a comparison with these earlier conditions may offer perspective on the current conditions of our study marshes. Applying an IMI of 8.9 (or 9.0) as a reference value of historic marsh integrity would suggest that the integrity of marshes in our study sample ($n = 31$) is considerably degraded on average. There is no way, however, to determine if the study marshes that approach an IMI of 8.9 are within natural variation of an undisturbed marsh, or are in-fact degraded by human activities, including sea-level rise. For example, Sheffield Cove (IMI = 8.0) has a nearly-representative distribution of historic communities, except for a moderate occurrence (16.6%) of *Mixed High Marsh* (*Meadow High Marsh* and *S. alterniflora* mix), which was not a type described by Miller and Egler or Chapman, and 3.4% representation of *Dieback Denuded Peat* (edge die-back) a feature also not described by those authors. *S. alterniflora* was reported as occurring “rarely” in both of Miller and Egler’s high marsh communities (*J. gerardii* and *S. patens* dominated), suggesting that a substantial mixed community did not occur at Barn Island Marsh at that time. Nearly every marsh in our study had some *Mixed High Marsh*, but its occurrence was low (5–10%) at some sites.

In contrast to Sheffield Cove, several sites in the lower range of IMI scoring clearly diverge from the historic marshes. For example, Quonnie East, has only 3.5% cover of *Meadow High Marsh*, 34% *Sa High Marsh*, 11.7% *Mixed High Marsh*, and 23% *Dieoff Denuded Peat*. Several marshes with intermediate IMI values have very little *Dieoff* (<5%), but have low representation of *Meadow High Marsh* (<20%) and high representation *Mixed* and *Sa High Marsh* (>40% combined), indicating a vegetation shift from *S. patens* to *S. alterniflora* that is theorized to precede dieoff in the process of marsh drowning (Warren and Niering, 1993, Raposa et al., 2017b).

4.3. Management implications

Understanding how sea-level rise and other disturbances are contributing to marsh degradation is critical for identifying conservation approaches (Roman, 2017, Wasson et al., 2019), and a decision-support matrix based on MarshRAM and marsh-elevation data, such as presented in Table 4, may be a useful tool to help managers visualize and interpret this complex information. Table 4 demonstrates how collecting the full suite of MarshRAM data across multiple sites can establish a range of salt marsh conditions, against which individual wetlands can be evaluated (i.e. “reference gradient”, Faber-Langendoen et al., 2009). Assigning management categories, based on upper and lower quartiles and interquartile ranges of metric and attribute values, can be used to clarify the relationships among ecosystem services, human disturbances, elevation, marsh integrity, and migration potential by simplifying interpretation of each metric (a central purpose of categorization) and reducing the chance of overestimating metric-value precision and meaningfulness (Barbour et al., 1996, Miller et al., 2006). MarshRAM disturbance information (*Wetland Disturbances*), categories of marsh integrity (IMI), relative value (sum of *Ecosystem Functions and Services*), and migration potential (*Migration Area* and *Replacement Ratio*) have been used together in Rhode Island to help managers prioritize salt marshes for restoration and marsh migration facilitation activities (Kutcher and Chaffee, 2021).

MarshRAM data can also identify general management considerations for marshes. The negative association between historic loss and *Replacement Ratio* (the proportion of existing marsh that, with minimal or no management, will theoretically replace marsh losses from sea-level rise as marshes migrate inland) suggests that marshes with larger migration corridors relative to the size of the marsh are losing vegetated area more slowly than those with smaller corridors; this implies that unassisted marsh migration may already be contributing to marsh sustainability. But, the median *Replacement Ratio* of 25% implies that, without active management, only about a quarter of existing marsh area will be replaced through landward migration as marshes succumb to inundation stress. Management within existing marsh footprints, such as thin-layer sediment placement (Raposa et al., 2020), and management of the surrounding landscape, such as removal of barriers to migration and protection of migration corridors, may be needed to improve the prospects of salt marsh sustainability.

Phragmites was present at every marsh in the study. *Phragmites* can outcompete native species, diminish habitat value, and impede landward migration of native marsh species (Farnsworth and Meyerson, 1999, Meyerson et al., 2000, Benoit and Askins, 2002, Smith, 2013). *Phragmites* is a disturbance to marsh function, but is also a vegetative response to other disturbances such as filling, land clearing, hydrologic alteration, and nutrient loading associated with coastal development (Roman et al., 1984, Silliman and Bertness, 2004, Meyerson et al., 2009). Our rapid data clarify the pervasiveness of *Phragmites* in the region and provide managers with a way to compare its abundance across individual marshes in relation to other disturbances captured by MarshRAM.

Marsh edge erosion was another pervasive disturbance (29 of 31 sites), assessed as severe (>60% of the marsh edge) at 18 of 31 sites. Coincident occurrence of *Dieback Denuded Peat*, indicating burrowing-crab damage (Holdredge et al., 2009), at many of these highly-eroded sites suggests a positive interaction between crab overabundance and sea level rise (Crotty et al., 2017, Raposa et al., 2018). The prospect of accelerating sea-level rise, often coupled with crab herbivory and burrowing precipitating marsh loss along the seaward edge, further emphasizes the need for promoting marsh migration along the landward edge.

The increasing number of marsh-dependent sparrows flushed along the IMI transects with the increasing cover of ditches suggests that marsh sparrows may be opportunistically using salt marsh ditches for nesting or foraging. Reinert and Mello (1995) found that seaside sparrows (*A. maritima*) in southern New England focused their nesting and foraging activities in the medium-height high-marsh *S. alterniflora* bordering ditches and creeks, whereas saltmarsh sparrows (*A. caudacuta*) more-often used salt-meadow plants (e.g., *S. patens*, *J. gerardii*), which frequently colonize the raised spoils associated with historic ditching. We did not differentiate between the two sparrow species during our MarshRAM surveys. Still, this finding raises the concern that both the benefits and potentially harmful effects of managing historic ditches should be considered in management planning (Corman et al., 2012). More intensive study into sparrow use of historic ditches may be warranted for clarifying the full ecological effects of ditch remediation, particularly given the recent decline of marsh sparrows and their critical dependence on specific attributes of salt marsh vegetation for survival (Correll et al., 2017).

Studies have long theorized that vegetation dieoff can occur between the natural creek levees and between linear spoils of ditches (Nichols, 1920, Miller and Egler, 1950, Smith and Niles, 2016, Watson et al., 2017). Our study found no relationship between the cover of natural creeks or ditching intensity versus IMI or versus the cover of marsh dieoff (even as ditching was observed at 27 of 31 sites), suggesting that the inter-levee panne-formation process is not currently a main driver of marsh degradation at our study marshes. More recent studies implicate increased rates of sea-level rise and accretion deficits in relation to the tide frame in widespread dieoff of the historic peat platform (Ekberg et al., 2017, Watson et al, 2017); findings from our study, that relatively small amounts of dieoff and ponding have a strong negative influence on IMI and historic loss, more-closely support these recent conclusions. Documenting changes over time in dieoff features, pool size, vegetation shifts, and marsh platform elevation in relation to the tide frame will clarify the role of sea-level rise in the process of marsh degradation and loss.

4.4. Logistics and transferability

Our findings indicate that MarshRAM observational data and our novel, rapid *walking-transect* data were efficient to collect and consistent across users and habitat types in Rhode Island, USA. We recognize that some MarshRAM attributes and metrics may need to be modified for application in other regions, but the utility of the RAM— e.g., categorizing marshes by attributes for analysis, identifying specific disturbances and their individual and aggregate influences on marsh integrity, comparing individual marshes against a "reference gradient" of condition for management planning—can be preserved. For example, the *walking transects* may be difficult to conduct at the larger marshes found in some regions. In such cases, investigators may be able to instead use a geographic information system and remote classification to determine the relative proportions of the meaningful community types; applying those proportions to the CCI could then be used to calculate IMI scores. Specific recommendations for adapting MarshRAM to other regions are offered in Appendix C.

4.5. Conclusion

MarshRAM has provided valuable information for salt-marsh management in Rhode Island, and it could be adapted for application in other states, across regions, or nationwide. Although rapid assessment methods for estuarine wetlands in other states exist (Jacobs, 2003, Carullo et al., 2007, CWMW, 2013), MarshRAM may offer benefits not provided by others, such as setting and classification information, a ranking method for functions and values, opportunistic waterbird and marsh bird tallies, a tested surrounding-landscape evaluation model, comprehensive disturbance metrics, community composition information that can generate metrics of degradation/vulnerability, and metrics characterizing site-level landward migration potential. Also, MarshRAM keeps function and value information separate from disturbance and degradation information, which can be important for analysis and decision support (Table 4). The inclusive, yet rapid framework of MarshRAM may be attractive to applied scientists and managers beyond Rhode Island because, with a single visit per marsh, it provides reliable site-level information that may be useful for characterizing marsh condition and value, better understanding the relationships between disturbances and marsh integrity, prioritizing sites for restoration and conservation, and assessing restoration success.

CRediT authorship contribution statement

Thomas E. Kutcher: Conceptualization, Methodology, Formal analysis, Writing – original draft. **Kenneth B. Raposa:** Methodology, Validation, Writing – review & editing. **Charles T. Roman:** Methodology, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This work was supported by a contract agreement between the State of Rhode Island Department of Environmental Management and the Rhode Island Natural History Survey. The agreement was funded in part by funds provided by a U.S. Environmental Protection Agency Clean Water Act Section 104(b)3 Wetland Program Development Grant. Although the information in this document has been funded in part by the U.S. Environmental Protection Agency, it may not necessarily reflect the views of the Agency and no official endorsement should be inferred. Susan Kiernan, Caitlin Chaffee, Carolyn Murphy, David Gregg, and Kira Stillwell administered this work, and Jennifer Beck assisted with field work and data summary. Cathleen Wigand helped develop coefficients of community integrity, field tested the methods, and provided valuable feedback. Wenley Ferguson, James Turek, and Caitlin Chaffee field tested and provided valuable feedback on the methods. The Rhode Island Natural History Survey is generously housed by the University of Rhode Island College of the Environment and Life Sciences, USA.

Appendices. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2022.108841.

References

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81 (2), 169–193.

Barbour, M.T., Gerritsen, J., Griffith, G.E., Frydenborg, R., McCarron, E., White, J.S., Bastian, M.L., 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. J. North Am. Benthol. Society 15 (2), 185–211.

Berry, W.J., Reinert, S.E., Gallagher, M.E., Lussier, S.M., Walsh, E., 2015. Population status of the seaside sparrow in Rhode Island: a 25-year assessment. Northeastern Naturalist 22 (4), 658–671.

Bourdaghs, M., Johnston, C.A., Regal, R.R., 2006. Properties and performance of the floristic quality index in Great Lakes coastal wetlands. Wetlands 26 (3), 718–735.

Bried, J.T., Jog, S.K., Matthews, J.W., 2013. Floristic quality assessment signals human disturbance over natural variability in a wetland system. Ecol. Ind. 34, 260–267.

Cahoon, D.R., Reed, D.J., 1995. Relationships among marsh surface topography, hydroperiod, and soil accretion in a deteriorating Louisiana salt marsh. J. Coastal Res. 357–369.

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

Carullo, M., Carlisle, B.K. and Smith, J.P., 2007. A New England rapid assessment method for assessing condition of estuarine marshes: A Boston Harbor, Cape Cod and Islands pilot study. Massachusetts Office of Coastal Zone Management, Boston, USA.

Chapman, V.J., 1938. Coastal movement and the development of some New England salt marshes. Proc. Geol. Assoc. 49 (4), 373–IN5.

Corman, S.S., Roman, C.T., King, J.W., Appleby, P.G., 2012. Salt marsh mosquito-control ditches: sedimentation, landscape change, and restoration implications. J. Coastal Res. 28 (4), 874–880.

Correll, M.D., Wiest, W.A., Hodgman, T.P., Shriver, W.G., Elphick, C.S., McGill, B.J., O'Brien, K.M., Olsen, B.J., 2017. Predictors of specialist avifaunal decline in coastal marshes. Conserv. Biol. 31 (1), 172–182.

Crotty, S.M., Angelini, C., Bertness, M.D., Armas, C., 2017. Multiple stressors and the potential for synergistic loss of New England salt marshes. PLoS ONE 12 (8), e0183058. https://doi.org/10.1371/journal.pone.0183058.

CWMW (California Wetlands Monitoring Workgroup). 2013. California Rapid Assessment Method (CRAM) for Wetlands, Version 6.1, 67pp.

Deegan, L.A., Hughes, J.E., Rountree, R.A., 2002. Salt marsh ecosystem support of marine transient species. In: Weinstein, M.P., Kreeger, D.A. (Eds.), Concepts and Controversies in Tidal Marsh Ecology. Kluwer Academic Publishers, Dordrecht, pp. 333–365. https://doi.org/10.1007/0-306-47534-0_16.

Donnelly, J.P., Bertness, M.D., 2001. Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. Proc. Natl. Acad. Sci. 98 (25), 14218–14223.

Ekberg, M.L.C., Raposa, K.B., Ferguson, W.S., Ruddock, K., Watson, E.B., 2017. Development and application of a method to identify salt marsh vulnerability to sea level rise. Estuaries Coasts 40 (3), 694–710.

Faber-Langendoen, D., Lyons, R., Comer, P., 2009. Developing options for establishing reference conditions for wetlands across the lower 48 states. NatureServe, Arlington, VA, USA.

Farnsworth, E.J., Meyerson, L.A., 1999. Species composition and inter-annual dynamics of a freshwater tidal plant community following removal of the invasive grass, Phragmites australis. Biol. Invas. 1 (2), 115–127.

Fennessy, M.S., Jacobs, A.D., Kentula, M.E., 2007. An evaluation of rapid methods for assessing the ecological condition of wetlands. Wetlands 27 (3), 543–560.

Ganju, N.K., Defne, Z., Kirwan, M.L., Fagherazzi, S., D'Alpaos, A., Carniello, L., 2017. Spatially integrative metrics reveal hidden vulnerability of microtidal salt marshes. Nat. Commun. 8 (1), 1–7.

Gedan, K.B., Silliman, B.R., Bertness, M.D., 2009. Centuries of human-driven change in salt marsh ecosystems. Marine Sci. 1 (1), 117–141.

Gedan, K.B., Altieri, A.H., Bertness, M.D., 2011. Uncertain future of New England salt marshes. Mar. Ecol. Prog. Ser. 434, 229–237.

Holdredge, C., Bertness, M.D., Altieri, A.H., 2009. Role of crab herbivory in die-off of New England salt marshes. Conserv. Biol. 23 (3), 672–679.

Kirwan, M.L., Megonigal, J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504 (7478), 53–60.

Kirwan, M.L., Temmerman, S., Skeehan, E.E., Guntenspergen, G.R., Fagherazzi, S., 2016. Overestimation of marsh vulnerability to sea level rise. Nat. Clim. Change 6 (3), 253–260.

Kutcher, T.E. and Chaffee, C., 2021. A framework for prioritizing salt marsh restoration and conservation activities in Rhode Island. Rhode Island Department of Environmental Management, Office of Water Resources, Providence, RI.

Kutcher, T.E., Bried, J.T., 2014. Adult Odonata conservatism as an indicator of freshwater wetland condition. Ecol. Ind. 38, 31–39.

Kutcher, T.E., Forrester, G.E., 2018. Evaluating how variants of floristic quality assessment indicate wetland condition. J. Environ. Manage. 217, 231–239.

Kutcher, T.E., 2021. MarshRAM user's guide: detailed instructions on how to conduct and interpret the Salt Marsh Rapid Assessment Method, MarshRAM, working draft. Rhode Island Department of Environmental Management Office of Water Resources, Providence, RI, 40pp.

Mariotti, G., Spivak, A.C., Luk, S.Y., Ceccherini, G., Tyrrell, M., Gonneea, M.E., 2020. Modeling the spatial dynamics of marsh ponds in New England salt marshes. Geomorphology 365, 107262. https://doi.org/10.1016/j.geomorph.2020.107262.

Meyerson, L.A., Saltonstall, K., Windham, L., Kiviat, E., Findlay, S., 2000. A comparison of Phragmites australis in freshwater and brackish marsh environments in North America. Wetlands Ecol. Manage. 8 (2), 89–103.

Meyerson, L.A., Saltonstall, K., Chambers, R.M., Silliman, B.R., Bertness, M.D., Strong, D., 2009. *Phragmites australis* in eastern North America: a historical and ecological perspective. Human impacts on salt marshes: a global perspective. University of California Press, Berkeley, USA, pp. 57–82.

Miller, W.R., Egler, F.E., 1950. Vegetation of the Wequetequock-Pawcatuck tidal-marshes, Connecticut. Ecol. Monogr. 20 (2), 143–172.

Miller, S.J., Wardrop, D.H., Mahaney, W.M., Brooks, R.P., 2006. A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. Ecol. Ind. 6 (2), 290–312.

Nichols, G.E., 1920. The vegetation of Connecticut VII. The associations of depositing areas along the seacoast. Bull. Torrey Bot. Club 47 (11), 511. https://doi.org/10.2307/2479953.

Nixon, S.W., 1980. Between coastal marshes and coastal waters—a review of twenty years of speculation and research on the role of salt marshes in estuarine productivity and water chemistry. In Estuarine and wetland processes (pp. 437-525). Springer US.

Payne, A.R., Burdick, D.M., Moore, G.E., 2019. Potential Effects of Sea-Level Rise on Salt Marsh Elevation Dynamics in a New Hampshire Estuary. Estuaries Coasts 42 (6), 1405–1418.

Raposa, K.B., Cole Ekberg, M.L., Burdick, D.M., Ernst, N.T., Adamowicz, S.C., 2017a. Elevation change and the vulnerability of Rhode Island (USA) salt marshes to sea-level rise. Reg. Environ. Change 17 (2), 389–397.

Raposa, K., Wasson, K., Nelson, J., Fountain, M., West, J., Endris, C., and Woolfolk. A., 2020. Guidance for Thin-Layer Sediment Placement as a Strategy to Enhance Tidal Marsh Resilience to Sea-Level Rise. Published in collaboration with the National Estuarine Research Reserve System Science Collaborative. 35pp.

Raposa, K.B., Weber, R.L.J., Ekberg, M.C., Ferguson, W., 2017b. Vegetation dynamics in Rhode Island salt marshes during a period of accelerating sea level rise and extreme sea level events. Estuaries Coasts 40 (3), 640–650.

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

Reinert, S.E., Mello, M.J., 1995. Avian community structure and habitat use in a southern New England estuary. Wetlands 15 (1), 9–19.

Roman, C.T., 2017. Salt marsh sustainability: challenges during an uncertain future. Estuaries Coasts 40, 711–716.

Roman, C.T., Niering, W.A., Warren, R.S., 1984. Salt marsh vegetation change in response to tidal restriction. Environ. Manage. 8, 141–150.

Shepard, C.C., Crain, C.M., Beck, M.W., Clifton, J., 2011. The protective role of coastal marshes: a systematic review and meta-analysis. PLoS ONE 6 (11), e27374.

Silliman, B.R., Bertness, M.D., 2004. Shoreline development drives invasion of Phragmites australis and the loss of plant diversity on New England salt marshes. Conserv. Biol. 18 (5), 1424–1434.

Smith, J.A., 2013. The role of Phragmites australis in mediating inland salt marsh migration in a mid-Atlantic estuary. PLoS ONE 8 (5), e65091.

Smith, J.A., Niles, L., 2016. Are Salt Marsh Pools Suitable Sites for Restoration? Wetland Sci. Pract. 33 (4), 102–109.

U.S. ACE (U.S. Army Corps of Engineers), 2003. The Highway Methodology Workbook. US Army Corps of Engineers New England Division. NEDEP-360-1-30. 28pp.

Warren, R.S., Niering, W.A., 1993. Vegetation Change on a Northeast Tidal Marsh: Interaction of Sea-Level Rise and Marsh Accretion. Ecology 74 (1), 96–103.

Wasson, K., Ganju, N.K., Defne, Z., Endris, C., Elsey-Quirk, T., Thorne, K.M., Freeman, C. M., Guntenspergen, G., Nowacki, D.J., Raposa, K.B., 2019. Understanding tidal marsh trajectories: evaluation of multiple indicators of marsh persistence. Environ. Res. Lett. 14 (12), 124073. https://doi.org/10.1088/1748-9326/ab5a94.

Watson, E.B., Wigand, C., Davey, E.W., Andrews, H.M., Bishop, J., Raposa, K.B., 2017. Wetland loss patterns and inundation-productivity relationships prognosticate widespread salt marsh loss for southern New England. Estuaries Coasts 40 (3), 662–681.

Wigand, C., McKinney, R.A., Charpentier, M.A., Chintala, M.M., Thursby, G.B., 2003. Relationships of nitrogen loadings, residential development, and physical characteristics with plant structure in New England salt marshes. Estuaries 26 (6), 1494–1504.

Wigand, C., Carlisle, B., Smith, J., Carullo, M., Fillis, D., Charpentier, M., McKinney, R., Johnson, R., Heltshe, J., 2011. Development and validation of rapid assessment indices of condition for coastal tidal wetlands in southern New England, USA. Environ. Monit. Assess. 182 (1-4), 31–46.

Wigand, C., Roman, C.T., Davey, E., Stolt, M., Johnson, R., Hanson, A., Watson, E.B., Moran, S.B., Cahoon, D.R., Lynch, J.C., Rafferty, P., 2014. Below the disappearing marshes of an urban estuary: historic nitrogen trends and soil structure. Ecol. Appl. 24 (4), 633–649.