

Evaluating Tidal Wetland Restoration Performance Using National Estuarine Research Reserve System Reference Sites and the Restoration Performance Index (RPI)

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Abstract Evaluations of tidal wetland restoration efforts suffer from a lack of appropriate reference sites and standardized methods among projects. To help address these issues, the National Estuarine Research Reserve System (NERRS) and the NOAA Restoration Center engaged in a partnership to monitor ecological responses and evaluate 17 tidal wetland restoration projects associated with five reserves. The goals of this study were to (1) determine the level of restoration achieved at each project using the restoration performance index (RPI), which compares change in parameters over time between reference and restoration sites, (2) compare hydrologic and excavation restoration projects using the RPI, (3) identify key indicator parameters for assessing restoration effectiveness, and (4) evaluate the value of the NERRS as reference sites for local restoration projects. We found that the RPI, modified for this study, was an effective tool for evaluating relative differences in restoration performance; most projects

achieved an intermediate level of restoration from 2008 to 2010, and two sites became very similar to their paired reference sites, indicating that the restoration efforts were highly effective. There were no differences in RPI scores between hydrologic and excavation restoration project types. Two abiotic parameters (marsh platform elevation and groundwater level) were significantly correlated with vegetation community structure and thus can potentially influence restoration performance. Our results highlight the value of the NERRS as reference sites for assessing tidal wetland restoration projects and provide improved guidance for scientists and restoration practitioners by highlighting the RPI as a trajectory analysis tool and identifying key monitoring parameters.

Keywords Reference network · Hydrologic restoration · Excavation · Elevation · Vegetation monitoring · NERR biomonitoring

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Introduction

Coastal habitat restoration remains a conservation priority in the USA and is guided in part by mandates set forth in the Estuary Restoration Act (ERA) of 2000 (Title 1 of the Estuaries and Clean Waters Act of 2000). The ERA directed a federal interagency council to prepare a National Strategy for Coastal Habitat Restoration and set a goal of restoring one million acres of estuarine habitat by 2010. A key objective identified in the ERA was to develop and enhance monitoring, data sharing, and research capabilities for coastal restoration projects. Restoration monitoring plans associated with all projects supported by ERA funds were required to select appropriate monitoring parameters, conduct baseline monitoring, draw comparisons with “least-disturbed” reference sites, and determine the appropriate sampling frequency and duration.

The ERA directed the National Oceanic and Atmospheric Administration (NOAA) to develop standard monitoring protocols for estuarine habitat restoration projects. Two NOAA programs directly affected by the ERA were the Restoration Center and the National Estuarine Research Reserve System (NERRS; part of the Office for Coastal Management). The Restoration Center supports research on the recovery of restored coastal habitats, the development and testing of innovative restoration methods, and the development of monitoring protocols and criteria to gauge restoration performance. The NERRS developed a system-wide Restoration Strategy (National Estuarine Research Reserve System 2002), which focuses on a set of priorities that includes assessing and monitoring restoration performance and evaluating the role of NERRS as local reference sites. Synergies between these two NOAA programs clearly exist for addressing the objectives outlined in the ERA.

Degraded tidal wetlands can be restored using a variety of approaches that differ in the degree of human modification. One example involving relatively low levels of intervention is the removal of dikes, undersized culverts, and other tide-restricting structures to reintroduce tidal flooding to degraded marshes, thereby initiating recovery processes towards mature marsh structure and function (Burdick and Roman 2012). At more degraded sites, direct physical manipulations of the marsh surface by excavating or adding fill can be used to create elevation and edaphic conditions amenable for the establishment and recovery of tidal wetland vegetation (Cornu and Sadro 2002). Zedler and Callaway (2000) conducted a literature review of different types of salt marsh restoration projects and documented differences in restoration effectiveness relative to the degree of human modification of the restored area. However, additional direct comparisons of alternate approaches to restoration (e.g., hydrologic reconnection vs. excavation/fill) using the same monitoring and data analysis methods are needed.

In order to comprehensively evaluate project results, quantitative monitoring should be used to compare restoration and least-disturbed reference sites and document the effectiveness of different restoration methods. Even though monitoring does not need to be expensive to adequately evaluate restoration projects, it is usually under-funded, often inadequately or inappropriately planned and implemented, and rarely based on a set of core variables relevant to restoration. Quantitative protocols are sorely needed to guide users in the uniform monitoring of restoration projects and to promote improved restoration monitoring in general (Neckles et al. 2002). For example, Konisky et al. (2006) demonstrated the value of a common set of monitoring standards for the Gulf of Maine and how this information could provide managers with a set of ecological benchmarks to help evaluate the status of restored salt marshes throughout the region. A concomitant need is to simultaneously identify key ecological parameters that drive restoration recovery so that they can be included in restoration monitoring protocols. Although some useful protocols already exist for monitoring a variety of ecological parameters associated with tidal wetlands (e.g., James-Pirri et al. 2012; Neckles et al. 2013; Roegner et al. 2008; Roman et al. 2001; Simenstad et al. 1991; Thayer et al. 2005), there is a need to further develop custom monitoring approaches using key indicator parameters to address the specific information needs outlined in the ERA and the NERRS Restoration Strategy (National Estuarine Research Reserve System 2002; Yozzo and Laska 2006).

A primary goal of tidal wetland restoration should be to reestablish the physical structure, biotic assemblages, feedbacks, and ecosystem functions that characterize comparable natural systems (Craft et al. 1999; Staszak and Armitage 2013). Ideally, tidal wetland attributes are compared over time between restoration wetlands and carefully selected reference sites that represent least-disturbed examples of the same wetland types undergoing restoration (Havens et al. 1995; Neckles et al. 2002; Roman et al. 2002). This approach provides an objective basis for judging the effectiveness of individual restoration projects while identifying meaningful and robust performance standards for application to future wetland restoration plans (Neckles et al. 2002; Short et al. 2000). Unfortunately, many tidal wetland restoration projects suffer from a lack of appropriate reference site information (Moore et al. 2009; Zedler and Callaway 2000). Locating appropriate reference sites can be challenging in highly altered coastal landscapes, and the spatial and temporal variability of both natural and restored tidal wetlands often makes meaningful comparisons difficult to achieve (Zedler and Callaway 2000). One way to address this is to develop baseline condition datasets from a network of reference sites (Brinson 1993; Diefenderfer et al. 2003; Thayer et al. 2005), but these datasets are generally lacking for tidal marsh habitats in most coastal regions of the USA. With its Biomonitoring and Sentinel Sites

programs (Moore 2012; National Estuarine Research Reserve System 2012; key components of its broader System-Wide Monitoring Program [SWMP]), the NERRS is uniquely situated to not only generate baseline condition datasets for tidal wetland habitats from multiple regions across the USA, but also to offer a network of least-disturbed examples of local tidal wetland types that can serve as reference sites.

There is a growing understanding that truly effective restoration monitoring includes evaluating multiple ecosystem functions over longer time scales (Craft et al. 1999; Konisky et al. 2006; Moreno-Mateos et al. 2012; Zedler and Callaway 2000). However, limited funds and narrowly focused program priorities can limit restoration performance assessments to monitoring a small number of parameters (e.g., vegetation characteristics, fish habitat) over very short time frames (Kentula 2000; Zedler and Callaway 2000). A potentially comprehensive and cost-effective approach for assessing restoration performance is to use reference site information to develop success criteria for comparison to restoration sites (Kentula 2000). Another strategy for evaluating restoration performance is the comparison of time series data for a suite of structural and functional attributes at restored and reference sites (e.g., Craft et al. 2002). This approach was advanced by Moore et al. (2009), who designed the restoration performance index (RPI) to document the performance of salt marsh restoration projects by converting data characterizing several structural and functional attributes at restoration and reference sites into indices used to score restoration performance relative to reference site conditions. The RPI has subsequently been used to evaluate ecosystem services provided by a restoring marsh relative to a reference marsh in New Hampshire (Chmura et al. 2012). Because it converts disparate ecosystem response data into a uniform scale, the RPI offers promise for facilitating meaningful comparisons among and between restoration and reference sites.

To further test the utility of the RPI, the NOAA Restoration Center and the NERRS entered into a partnership in 2007 to conduct a study in which NERR least-disturbed tidal wetlands were used as reference sites for evaluating tidal wetland restoration projects supported by ERA funding. Specifically, nine reference sites at five NERRs were used to evaluate the performance of 17 local tidal wetland restoration projects previously funded by the ERA through the NOAA Restoration Center. Restoration projects were categorized as either a hydrologic (breaching or removing tidal restrictions) or excavation (manipulation of marsh sediments to establish correct tidal wetland elevations) restoration technique. Quantitative monitoring was standardized and coordinated across all restoration and reference sites. Selected monitoring parameters that represent structural and functional indicators of ecosystem health were based on pilot tidal wetland

monitoring projects implemented under the NERRS Biomonitoring Program (Moore 2012) and NOAA's restoration monitoring manual (Thayer et al. 2005). The goals of this study were to (1) determine the level of restoration progress during our study period (2008–2010) at each project using the RPI, (2) evaluate the comparative restoration performance of hydrologic and excavation restoration approaches, (3) identify and evaluate key environmental parameters that drive variations in restoration performance, and (4) evaluate the use of long-term NERRS monitoring sites as reference wetlands for local restoration projects. Results will advance the science of tidal wetland restoration in general and help guide the planning, design, and evaluation of future restoration projects.

Methods

Study Sites

This study was conducted in 17 tidal restoration wetlands located in or near five NERR sites in Maine (Wells NERR), Rhode Island (Narragansett Bay NERR), Virginia (Chesapeake Bay VA NERR), North Carolina (North Carolina NERR), and Oregon (South Slough NERR) (Fig. 1). All wetlands were classified as emergent tidal marsh under the Biotic Component of the Coastal and Marine Ecological Classification Standard (Federal Geographic Data Committee 2012) except South Slough NERR's Yaquina study sites (Y-27 and Y-28), which were classified as tidal scrub-shrub wetlands. Each restoration wetland was paired with a least-disturbed reference wetland (nine total reference wetlands). Seven of nine reference wetlands were located within NERR boundaries; the Jacob's Point (RI) reference site lies outside but in close proximity to the Narragansett Bay NERR boundary and is an ongoing NERR biomonitoring data collection site. The Yaquina 28 (OR) reference site also lies outside the South Slough NERR boundary, about 145 km to the north, but less than 0.8 km from its paired reference site. Least-disturbed reference sites were paired with restoration sites based on the degree to which they were similar to the expected wetland characteristics of the restoration sites once fully recovered. The restoration wetlands included ten hydrologic restoration sites located in the Wells, Narragansett Bay, and South Slough NERRs and seven excavation restoration sites located in the North Carolina, Chesapeake Bay, and South Slough NERRs. Wetland characteristics varied considerably among restoration wetlands (e.g., vegetation composition, wetland size, geomorphology, restoration age), which was in part a reflection of the multiple biogeographic regions represented in this project. Detailed descriptions of paired restoration and reference wetlands are shown in Table 1.

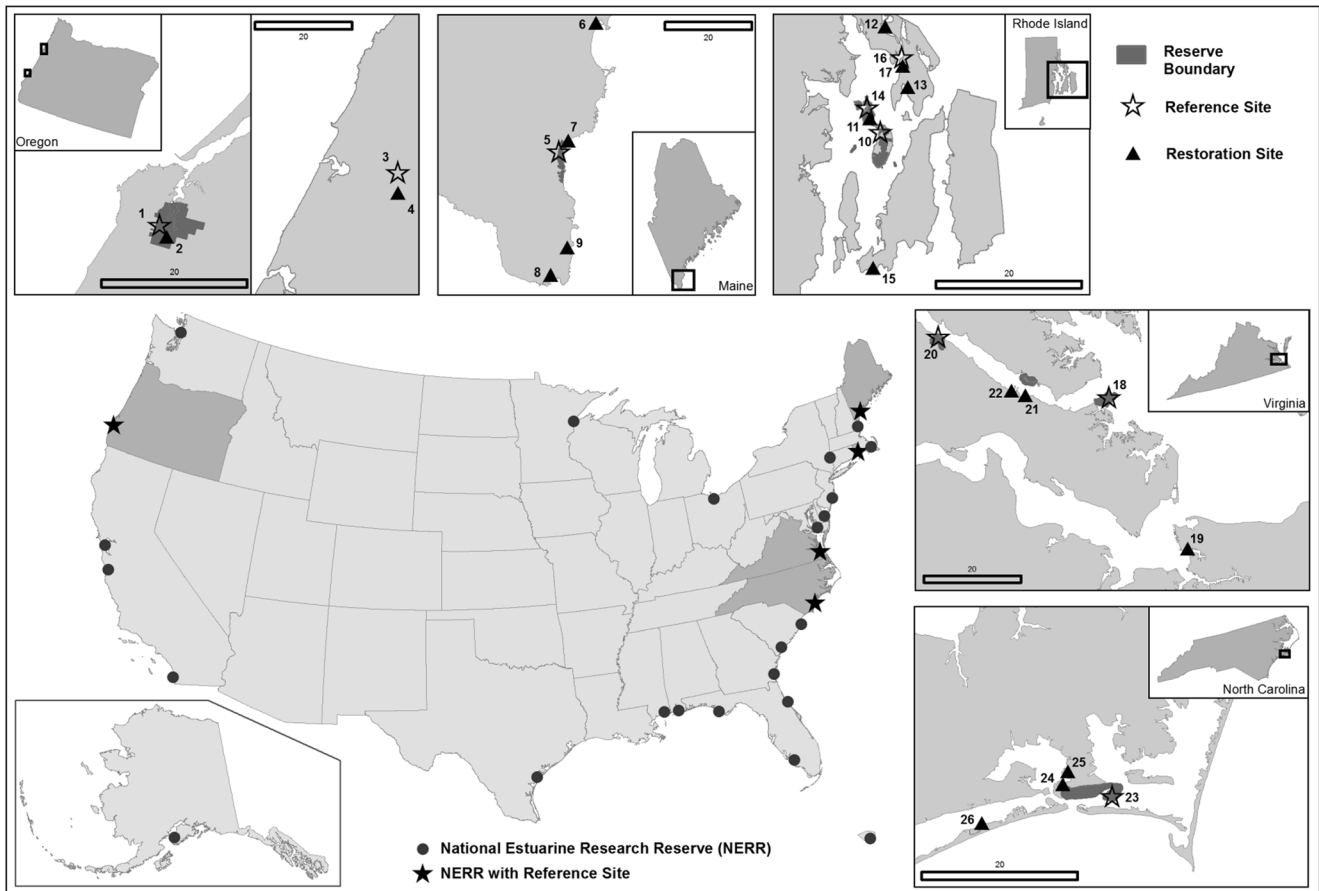


Fig. 1 Locations of the 17 restoration and 9 reference wetlands included in this study. Wetlands were located in or near the Wells (ME), Narragansett Bay (RI), Chesapeake Bay (VA), North Carolina (NC), and South Slough (OR) NERRs. Numbers correspond to site names as listed in Table 1

Table 1 Restoration and reference study sites in each of the five participating NERRs

Reserve	Reference wetland (salinity)	Restoration wetland	Type	Restoration date	Area (ha)	Proximity (km)	Salinity (ppt)	
South Slough, OR	1. Danger Point (5–28 ppt)	2. Kunz Marsh	E	1996	2.8	0.4	28	
	3. Yaquina 28 (0.5–5 ppt)	4. Yaquina 27	H	2002	3.2	1.1	5	
Wells, ME	5. Webhannet Marsh (25 ppt)	6. Cascade Brook	H	2004	36	32.3	11	
		7. Drakes Island	H	2005	31	3.1	16	
		8. Spruce Creek	H	2004	8	25.5	20	
		9. Wheeler Marsh	H	2005	6.9	21.8	25	
		11. Potter Pond	H	2003	2.3	2.2	23.8	
Narragansett Bay, RI	10. Nag Marsh (23 ppt)	12. Walker Farm	H	2005	6.5	14.7	20	
		13. Silver Creek	H	2009	5.6	7.2	17.3	
		14. Coggeshall Marsh (27 ppt)	15. Gooseneck Cove	H	2005	22.8	21.7	29.9
		16. Jacob's Point (28 ppt)	17. Jacob's Point	H	2010	6.7	0	11.3
Chesapeake Bay, VA	18. Goodwin Islands (16–23 ppt)	19. Hermitage Museum	E	2007	0.2	35	17–23	
	20. Taskinas Creek (6–16 ppt)	21. Naval Weapons Stn.	E	2006	0.4	22	1–23	
North Carolina		22. Cheatham Annex	E	2007	0.2	18	2–23	
	23. Middle Marsh (15–38 ppt)	24. Duke Marine Lab	E	2002	0.1	6	15–38	
		25. NC Maritime Museum	E	2001	0.1	6.4	15–38	
		26. Pine Knoll Shores	E	2002	0.1	20	15–38	

Information for each restoration site includes type (hydrologic or excavation), restoration date, wetland area, proximity to the paired reference site, and salinity. Numbers preceding each site name correspond to map locations on Fig. 1

Table 2 Monitoring parameters included in this study

Category	Parameter	Field method	Frequency	RPI	ANOSIM/SIMPER	BEST	Regression
Vegetation	Community composition	Vegetation plots	Annually		x	x	
	Percent cover (5 dominants)	Vegetation plots	Annually	x			x
	Percent cover (invasives)	Vegetation plots	Annually	x*			
	Species richness	Vegetation plots	Annually	x			x
Hydrology	Pore water salinity	Spot checks	Annually	x		x	
	Marsh surface inundation	Loggers	Annually	x			
	Groundwater depth	Loggers	Annually	x		x	x
	Maximum high tide	Loggers	Annually	x			
Elevation	Marsh surface elevation	RTK GPS; leveling	One time			x	x
Soils	Bulk density	Soil cores	One time			x	
	Percent organic matter	Soil cores	One time			x	

For each parameter, the field method and sampling frequency across study years are shown. An “x” denotes the parameters that were included in each statistical test (asterisk denotes that the percent cover of invasives was not included in the RPI in our study, but we recommend its inclusion moving forward). Statistical test acronyms include the restoration performance index (RPI), analysis of similarity (ANOSIM), similarity percentages (SIMPER), and biota-environment stepped (BEST) analysis

Field Sampling

Standardized monitoring was used to collect data describing specific biotic and abiotic parameters at all reference and restoration sites (Table 2). Parameter selection was guided in part by NOAA’s reference manual for restoration monitoring (Thayer et al. 2005) and in consultation with NOAA Restoration Center staff. Data collection was stratified across three distinct wetland elevation zones: (1) low intertidal (“low marsh”), (2) high intertidal (“high marsh”), and (3) wetland-upland transition (“transition”). Tidal wetland zones were classified in order to stratify major vegetation communities by elevation (Roegner et al. 2008). In reference and restoration sites with established plant communities subject to tidal flooding, the delineation of tidal wetland zones was guided by local knowledge of tidal wetland vegetation communities and then confirmed by the acquisition of elevation data. All data were collected from 2008 to 2010.

Vegetation data were collected from multiple monitoring plots located at regular intervals along three transects established at each project site (Supplementary Table 1). All transects were established following the NERRS Biomonitoring protocol (Moore 2012; largely based on Roman et al. 2001). Transects extended from the lowest intertidal emergent wetland elevations up through the wetland/upland transition. Each plot consisted of a 1.0-m² quadrat in which percent cover of all emergent vegetation species (including invasive species such as *Phragmites australis* at east coast sites and *Phalaris arundinacea* at west coast sites) and other structural attributes (e.g., woody debris, bare ground) was quantified using the point-intercept method at 50 points per quadrat (Godinez-Alvarez et al. 2009). Vegetation species richness (number of species per quadrat) was also quantified

by visually identifying all emergent vegetation species present in each quadrat. All vegetation data were collected during the middle or end of the growing season each year (typically July through September) to capture peak biomass and to facilitate taxa identification.

High-frequency groundwater-level data were measured with Onset HOBO and/or In Situ Aqua Troll 2000 loggers deployed in groundwater monitoring wells located along one transect in each project site, with one well established in each primary wetland zone. Wells were typically 1-m deep × 3-cm diameter and fitted with risers of sufficient length to prevent the well from filling with water during high tides (Sprecher 2000). Data were collected every 6 min for at least a 2-week spring to neap tide period during the growing season. For each of the three wetland zones at each site, these data were subsequently used to calculate (1) mean low-tide water depth below the wetland surface (hereafter, groundwater depth), (2) maximum high tide, and (3) percent of time that the wetland surface was inundated by the tide (Neckles et al. 2013).

Pore water salinity was measured in the shallow root zone using methods that varied among NERR sites to conform with ongoing monitoring following established regional protocols at some sites. At the Chesapeake Bay and North Carolina reserves, pore water salinity was measured in the 5 to 30-cm root zone adjacent to groundwater wells with porous PVC sippers (Davies 2004; Montgomery et al. 1979). In the Wells and Narragansett Bay reserves, pore water salinity was measured next to multiple vegetation plots along one transect following the Roman et al. (2001) protocols, which utilize a stainless steel probe and a syringe to extract water samples from within the root zone. At the South Slough reserve, pore water salinity was measured at each vegetation

plot by extracting small, replicate soil samples to a depth of 10 to 15 cm. Although specific collection methods differed among sites, all field sampling was standardized by tide stage, season, and habitat to ensure that all data were comparable. All pore water salinity sampling (also referred to as “spot checks”) was conducted within 2 h of low tide on several dates throughout the June to September sampling period at each site (Supplementary Table 2).

Finally, soil cores were collected and wetland surface elevations were surveyed and tied to the NAVD88 geodetic datum at each site to help characterize ecosystem drivers that affect restoration performance. At least one soil core was collected from the three wetland zones along each site’s vegetation monitoring transects. Soil cores were collected by hand with a sharpened, thin-walled, stainless steel tube (3.5-cm diameter) inserted to a depth of 20 cm. Cores were sectioned longitudinally or horizontally, and these sections were used to measure soil bulk density and percent organic content in the root zone by loss on ignition following standard procedures (Ball 1964; Dean 1974; Soil Survey Staff 2014). Marsh platform elevation data were collected at each site using survey grade leveling, total survey stations, or a Trimble R8 GPS enabled for real-time kinematic surveys. Elevation data were collected from at least four locations within each vegetation plot along the length of each vegetation transect. All soil and elevation data were collected once at each site within the June to September sampling period.

Data Analyses

The RPI was used to quantify restoration site performance from 2008 to 2010 relative to each site’s paired reference wetland conditions and to compare the relative effectiveness of the two restoration approaches (hydrologic and excavation). In addition, restoration and reference site vegetation communities were directly compared to further characterize structural restoration performance and help validate RPI scores. Finally, relationships between specific abiotic parameters and vegetation communities were analyzed to help interpret vegetation data results and shed light on the use of abiotic data as indicators for assessing restoration performance.

RPI scores were calculated for each restoration wetland relative to its paired reference wetland (Moore et al. 2009) using multiple vegetation and hydrologic parameters that were collected across all study years. Vegetation parameters included the percent cover of the five dominant reference wetland plant species and species richness (the mean number of species per m²). Hydrological parameters included pore water salinity, wetland surface inundation duration, groundwater depth, and maximum high tide. Data for all parameters except species richness were averaged across the three wetland zones within each project site before inclusion into the RPI (species richness data were only used from high marsh vegetation plots, while salinity

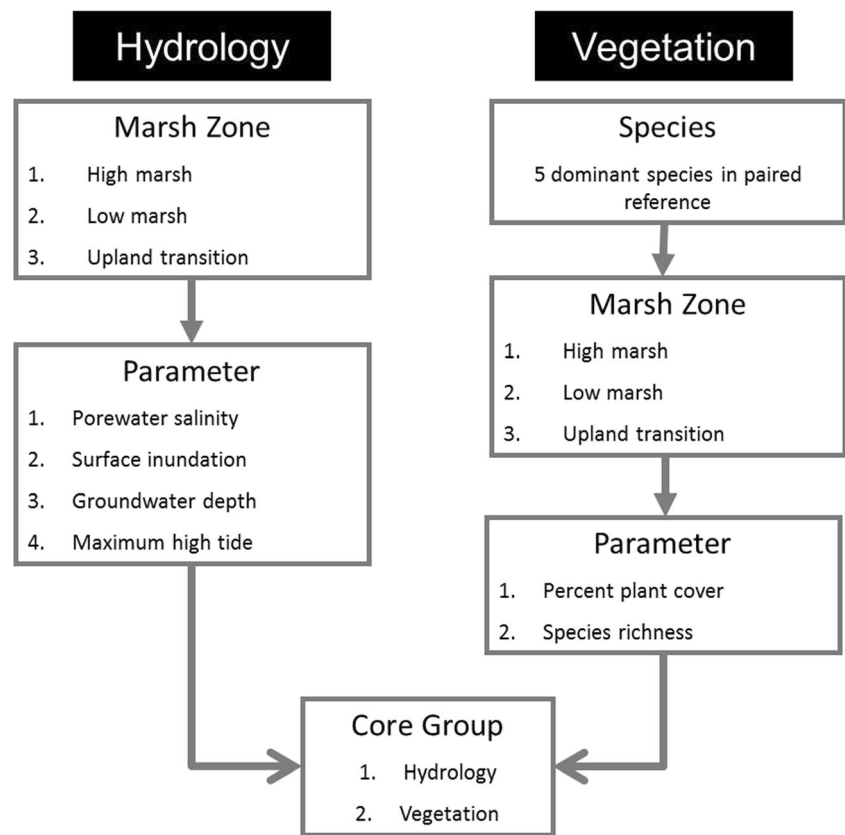
was also averaged across dates within each year). Scores were calculated using the following formula:

$RPI = (T_{\text{present}} - T_0)/(T_{\text{ref}} - T_0)$, where T_0 = initial conditions in the restoration wetland, T_{ref} = current reference site condition, and T_{present} = current conditions in the restoration wetland.

Typically, RPI analyses will incorporate pre-restoration data to serve as the initial condition to allow for a complete evaluation of restoration performance. This was not possible in our study because all restoration actions had already been initiated prior to our study; instead, we modified the RPI to use the first year of our study (2008) as a baseline for tracking restoration progress over the latter 2 years of the study (2009–2010). Interpretation of the modified RPI output is slightly different than the original index (Moore et al. 2009). More attention must be given to potential computational nuances when comparing sites of widely varying age, where data are not available for each year of restoration monitoring or where the restoration parameters differ in their trajectory (e.g., linear versus non-linear asymptotic change). This approach still provides insight into the trajectory of restoration performance over a discrete period of time and produces resulting RPI scores that can be used to evaluate relative restoration performance among regions and restoration types. Data weighting for the RPI occurred at three different levels for hydrology and four levels for vegetation (Fig. 2). For hydrology, the RPI weights by (1) wetland zone (low marsh, high marsh, upland transition), (2) parameter (salinity, surface inundation, groundwater depth, maximum high tide level), and (3) core group (hydrology, vegetation) in this order. For vegetation, the RPI weights by (1) species, (2) wetland zone, (3) parameter (percent cover, species richness), and (4) core group in this order (except species richness, which was neither weighted by zone nor plant species). At each level, the RPI only weights by present items, which are equally weighted. For example, the score range for a core group (hydrology, vegetation) is 0–0.5 when both core groups are available because they are equally weighted. However, if only one core group (e.g., hydrology) was available for the RPI, then hydrology would now have a score range of 0–1. The total RPI scores always range from 0 to 1, with a score of 1 indicating identical conditions in the restoration and reference wetlands for the parameters included in the score. Theoretically, RPI scores in a restoring wetland will increase over time and eventually approach or equal a value of 1.

Analysis of similarity (ANOSIM) was used to make direct comparisons of vegetation communities between paired restoration and reference sites; this was then coupled with similarity percentages (SIMPER) to determine which species contributed most to the observed differences. Bray-Curtis resemblance matrices were created using data from all species present prior to all ANOSIM analyses, and all data were square-root transformed prior to ANOSIM and SIMPER analyses to down-weight dominant species. For these multivariate analyses, the choice of data transformation is largely a biological

Fig. 2 Schematic of the process for weighting hydrology and vegetation parameters included in the restoration performance index (RPI). For example, vegetation is weighted first by all five dominant species, then sequentially by marsh zone, vegetation parameter, and core group. Each level is equally weighted



consideration (Clarke et al. 2014). Our conclusions therefore reflect the choice of an intermediate level of transformation that emphasizes the contributions of all species on community patterns.

SIMPER results were subsequently used to calculate vegetation species scores to identify (1) which species contributed most to overall dissimilarity between all restoration and reference wetlands and (2) which species were most closely associated with restoration or reference sites. When calculating scores, only the top five species that contributed to overall SIMPER dissimilarity within each wetland pair were included. Those species were assigned a number based on their dissimilarity status such that the most “dissimilar” species received a 5, the next species received a 4, and so on. For example, in a given restoration/reference site pair, if *Spartina alterniflora* contributed the most to overall dissimilarity and was more abundant in the reference wetland, it scored a 5 in the reference wetland category; if *P. australis* was the fifth largest contributor and was more abundant in the restoration wetland, it scored a 1 in the restoration category. For each species, these scores were then summed across all 17 wetland pairs. Scores were calculated separately for east and west coast sites.

Biota-environment stepped (BEST) analysis (with the BIO-ENV algorithm) was used to examine relationships between plant community structure and abiotic parameters by testing for correlations (based on Spearman’s rank correlations) between vegetation community composition and a subset of

relevant abiotic data (Clarke and Ainsworth 1993), which included soil bulk density, soil percent organic content, pore water salinity (averaged across all sample dates), groundwater depth, and wetland platform elevation. BEST analysis tests were run individually for each NERR site using Bray-Curtis resemblance matrices of square-root-transformed vegetation community data and Euclidian distance resemblance matrices of abiotic parameters based on data from individual monitoring plots. To determine if these abiotic parameters could also then be used to help predict restoration performance, linear regression analysis was used to test for significant relationships between RPI scores (total and vegetation component only) and important parameters identified in the BEST analyses. Prior to the regression, elevation data were referenced to a local tidal datum (mean higher high water (MHHW)) to facilitate valid comparisons among sites located in different geographic regions. All ANOSIM, SIMPER, and BEST analyses were performed with PRIMER v.6.1.9 (PRIMER-E Ltd). All regression tests were performed using JMP 9.0.1 (SAS Institute Inc.).

Results

Comparison of Wetland Pairs Based on RPI

Modified RPI scores for tidal wetland restoration sites were compared across sites, restoration type, and age. Total RPI

Table 3 Restoration performance index (RPI) scores for each of the 17 wetland restoration sites

Reserve	Restoration site	Type	Year 1			Year 2			Mean		
			Hydro	Veg	Total	Hydro	Veg	Total	Hydro	Veg	Total
Wells, ME	Cascade Brook	H	0.30	0.28	0.58	0.24	0.30	0.54	0.27	0.29	0.56
	Drakes Island	H	0.29	0.18	0.47	0.26	0.06	0.32	0.28	0.12	0.40
	Spruce Creek	H	0.25	0.15	0.40	0.35	0.32	0.67	0.30	0.24	0.54
	Wheeler Marsh	H	0.41	0.33	0.74	0.37	0.35	0.72	0.39	0.34	0.73
Narragansett Bay, RI	Gooseneck Cove	H	0.44	0.32	0.76		0.23	0.23	0.44	0.28	0.50
	Jacobs Point	H				0.14	0.28	0.42	0.14	0.28	0.42
	Potter Pond	H		0.69	0.69	0.50	0.37	0.87	0.50	0.53	0.78
	Silver Creek	H	0.25	0.32	0.57	0.20	0.36	0.56	0.23	0.34	0.57
	Walker Farm	H	0.50		0.50	0.40	0.09	0.49	0.45	0.09	0.50
Chesapeake Bay, VA	Chaatham Annex	E	0.38	0.22	0.60	0.39	0.08	0.47	0.39	0.15	0.54
	Hermitage Museum	E	0.37	0.20	0.57	0.34	0.20	0.54	0.36	0.20	0.56
	Naval Weapons	E	0.25	0.33	0.58	0.34	0.16	0.50	0.30	0.25	0.54
North Carolina	Duke Marine Lab	E	0.50	0.38	0.88	0.50	0.38	0.88	0.50	0.38	0.88
	NC Marine	E		0.33	0.33	0.50	0.17	0.67	0.50	0.25	0.50
	Pine Knoll	E	0.00	0.48	0.48	0.33	0.43	0.73	0.17	0.46	0.61
South Slough, OR	Kunz Marsh	E		0.11	0.11	0.17	0.03	0.20	0.17	0.07	0.16
	Yaquina 27	H		0.11	0.11	0.50	0.14	0.64	0.50	0.13	0.38

For each wetland, scores are shown for the hydrologic and vegetation subcomponents along with overall scores for each year; mean scores across the two study years are also provided. Year 1 represents 2009, year 2 represents 2010, and these 2 years are comparisons to the baseline year of 2008

scores (averaged across 2009 and 2010) were highly variable and ranged from 0.16 (Kunz Marsh, OR) to 0.88 (Duke Marine Lab, NC) (Table 3). Averaging total RPI scores across all wetlands within each NERR shows that restoration wetlands in ME, RI, and VA generally achieved an intermediate level of restoration performance over our study period (mean RPI = 0.54 in VA and 0.55 in RI and ME) (Fig. 3). Total RPI

scores were somewhat higher in North Carolina (mean = 0.66) and substantially lower in Oregon (mean = 0.25). In 12 out of the 17 wetland pairs, scores for the RPI vegetation component were lower than for the corresponding hydrology component. Most of the relatively low RPI scores for the vegetation component were driven by large differences in the percent cover of dominant species parameter between paired wetlands, which

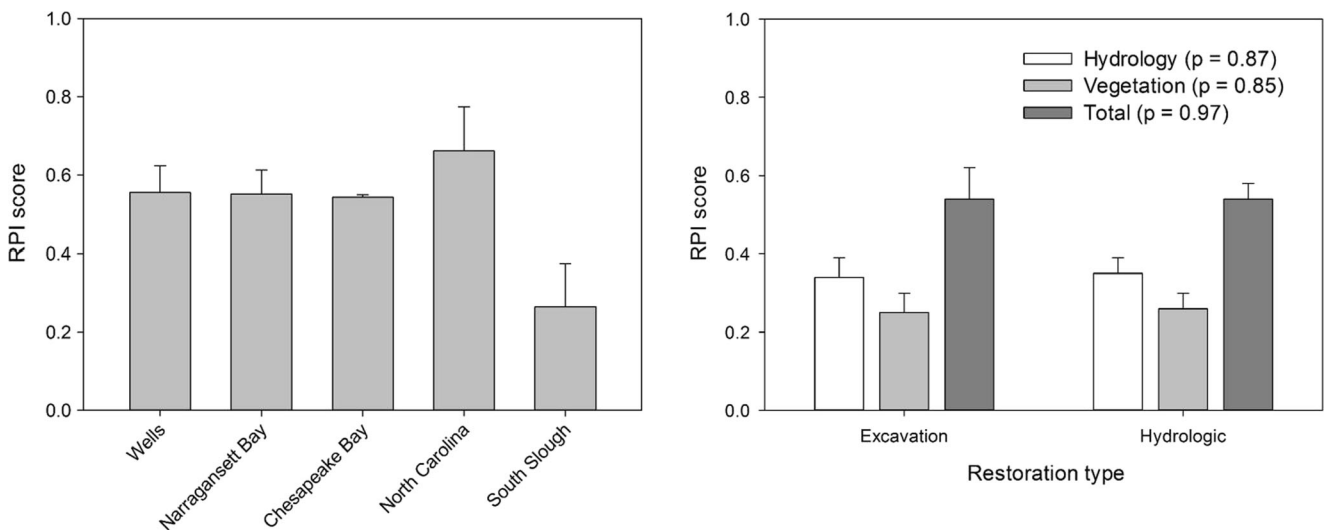


Fig. 3 Comparison of mean RPI scores among participating NERR sites (left) and between excavation and hydrologic restoration types (right). Total RPI scores are shown for both; scores are also shown for the hydrologic and vegetation components of RPI for the restoration types.

Statistical tests to compare scores across NERR sites were not run due to low sample sizes. No significant differences in RPI scores were found between the two types of restoration projects based on Students *t* test. Error bars are 1 SE

Table 4 Percent differences in mean annual parameter values (2008–2010) between paired reference and restoration sites

NERR	Site	Type	Salinity (ppt)			Inundation (%)			Water depth (m)			Max. high tide (m)			% cover dominants			% cover invasives			Richness (no/m ²)		
			08	09	10	08	09	10	08	09	10	08	09	10	08	09	10	08	09	10	08	09	10
ME	CABR	H	12.2	17.4	13.6	0.0	0.0	15.4	0.0	0.0	0.0	0.0	0.1	10.0	9.1	12.0	20.1	53.2	39.3	0.0	0.0	0.0	0.0
	DRIS	H	8.6	4.8	4.5	0.0	0.0	16.2	0.0	0.0	0.1	0.3	0.3	7.2	8.5	5.8	11.7	18.1	0.0	1.1	0.4	0.7	
	SPCR	H	3.4	7.2	5.2	0.0	0.0	7.6	0.0	0.0	0.0	0.2	0.2	6.7	4.0	3.6	6.1	6.7	7.2	0.4	0.4	0.0	
	WHMA	H	6.7	0.0	0.0	0.0	0.0	18.4	0.0	0.0	0.0	0.2	0.0	5.2	5.4	2.4	0.0	0.0	0.0	0.0	0.0	0.0	
RI	GOCO	H	1.9	0.0				0.0					0.0	4.1	4.9	14.4	16.0	9.0	5.5	0.1	0.0	0.1	
	JAPO	H	18.2	24.0	12.1								20.8			23.5	75.0		47.3	0.7	0.6		
	POPO	H	0.0	0.0		0.0	0.0		0.0	0.0	0.0	0.3	0.0	9.7	8.5	7.1	22.0	19.2	18.0	0.3	0.0	0.0	
	SICR	H	8.1	5.1	5.7								0.0	8.0	7.2	7.2	31.5	26.6	30.7	0.0	0.0	0.0	
	WAFR	H	9.6	3.2	0.0	47.5			0.0	0.2	0.0	0.3	0.0	18.7		17.7	74.5	16.2		0.0			
VA	CHAN	E	5.4	5.7	4.9	5.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	14.8	11.0	3.2	0.0	0.0	0.0	0.6	0.3	0.8	
	NAWE	E	6.9	6.2	3.4	49.5	7.4	20.9	0.6	0.1	0.1	0.6	0.2	11.2	8.9	0.3	1.6	2.2	0.1	0.8	0.0	0.7	
	HERM	E	3.4	4.7	3.4	0.0	0.0	0.0	0.2	0.0	0.0	0.2	0.0	4.0	3.8	0.0	0.0	0.0	0.0	0.8	0.5	0.3	
NC	DUMA	E	0.0	0.0	0.0			0.0			0.1	0.3	19.3	12.9	17.0		0.0	0.0	0.0	0.0	0.0	0.0	
	NCMA	E	0.0	0.0	0.0			0.0		0.0	0.0	0.0	4.7	3.4	16.0		0.0	0.0	0.0	0.0	0.1	0.1	
	PIKN	E	0.0	0.0	0.0			0.0		0.0	0.0	0.3	24.0	13.3	23.8		0.0	0.0	0.0	0.0	0.0	0.0	
OR	KUMA	E	0.0	0.0	0.0			0.0		0.3	0.6	23.5	27.3	25.4		0.0	0.0	0.0	0.0	3.5	4.3	3.8	
	YA27	H	0.0	0.0	0.0			0.0		0.0	0.0	0.0	15.2	15.2	15.7		0.0	0.0	0.0	1.7	1.4	1.2	

All parameters included in the RPI analysis are included, as was the percent cover of invasive species. Any difference between paired wetlands of between 10 and 20% is shaded in light gray; differences of >20% are shaded dark gray

differed by more than 10% in 22 out of 49 annual wetland pairs (Table 4). Although it was not included in the RPI analyses, the percent cover of invasive species also differed greatly between paired wetlands (different by more than 10% in 15 of 49 wetland pairs). In contrast, species richness and all hydrologic parameters were much more similar between paired wetlands with groundwater depth, maximum high tide, and species richness never differing by more than 10% between any pair of wetlands. After averaging across years, there were no significant differences in RPI scores (total RPI, vegetation component only, and hydrology component only) between excavation and hydrologic restoration projects, indicating that both types of projects achieved a similar level of restoration progression in our study (Fig. 3). Finally, a series of linear regressions indicated that RPI scores were not significantly related to restoration site age (age \times total RPI, $R^2 = 0.002$, $p = 0.836$; age \times RPI hydrology, $R^2 < 0.001$, $p = 0.919$; age \times RPI vegetation, $R^2 < 0.001$, $p = 0.97$).

Comparison of Wetland Pairs Based on Vegetation Communities

Vegetation communities remained significantly different between paired restoration and reference wetlands for 15 of 17 cases tested (ANOSIM; Table 5), corroborating the relatively low scores for the vegetation component of the RPI. In fact, restoration wetland vegetation communities achieved

similarity to reference wetland communities at only the Wheeler Marsh (ME) and Hermitage Museum (VA) sites. Based on SIMPER, average vegetation community dissimilarity between paired wetlands ranged from 52 to 89%, and in 14 of the 17 wetland pairs at least 50% of community dissimilarity was driven by only five species (Table 5).

S. alterniflora (score of 70 after combining tall and short forms of this species) and *Spartina patens* (53) contributed by far the most to overall dissimilarity between east coast wetland pairs and were subsequently followed by *Distichlis spicata* (30), bare ground (18), and *P. australis* (18) (Fig. 4). Percent cover of *S. alterniflora*, *S. patens*, *D. spicata*, and bare ground was generally greater in reference wetlands, while the opposite was true for *P. australis*. Dissimilarity between west coast wetland pairs was mostly driven by higher abundances of *Agrostis stolonifera*, *Carex lyngbyei*, *Triglochin maritimum*, and *Deschampsia caespitosa* in restoration wetlands and a higher abundance of *P. arundinacea* in reference wetlands.

Abiotic Parameters Affecting Vegetation Communities

Based on BEST analyses, groundwater depth and wetland elevation were included in the overall model that most explained restoration wetland vegetation communities in every case tested (Table 6). Pore water salinity was identified as an

Table 5 Results of ANOSIM tests comparing vegetation communities between paired reference and restoration sites

NERR	Restoration site	Type	ANOSIM		SIMPER	
			R	p	Avg. dissim	Top 5 spp. %
Wells, ME	Cascade Brook	H	0.36	0.001	76.0	41.9
	Drakes Island	H	0.11	0.012	63.1	54.8
	Spruce Creek	H	0.14	0.013	69.5	51.0
	Wheeler Marsh	H	0.06	0.106	69.5	48.2
Narragansett Bay, RI	Gooseneck Cove	H	0.11	0.005	60.3	70.4
	Jacobs Point	H	0.66	0.001	79.6	74.0
	Potter Pond	H	0.16	0.003	66.4	71.2
	Silver Creek	H	0.27	0.001	72.4	65.5
	Walker Farm	H	0.54	0.001	80.0	73.3
Chesapeake Bay, VA	Cheatham Annex	E	0.2	0.003	66.0	92.2
	Hermitage Museum	E	0.06	0.069	59.9	92.3
	Naval Weapons	E	0.14	0.002	65.7	78.7
North Carolina	Duke Marine Lab	E	0.09	0.001	58.5	59.0
	NC Maritime Museum	E	0.09	0.001	51.7	71.4
	Pine Knoll Shores	E	0.1	0.001	60.1	66.8
South Slough, OR	Kunz Marsh	E	0.34	0.001	64.8	53.0
	Yaquina 27	H	0.61	0.001	89.1	49.1

Also shown is the percent dissimilarity between reference and restoration wetland vegetation communities (all species and contribution of the top 5 species) as determined with SIMPER

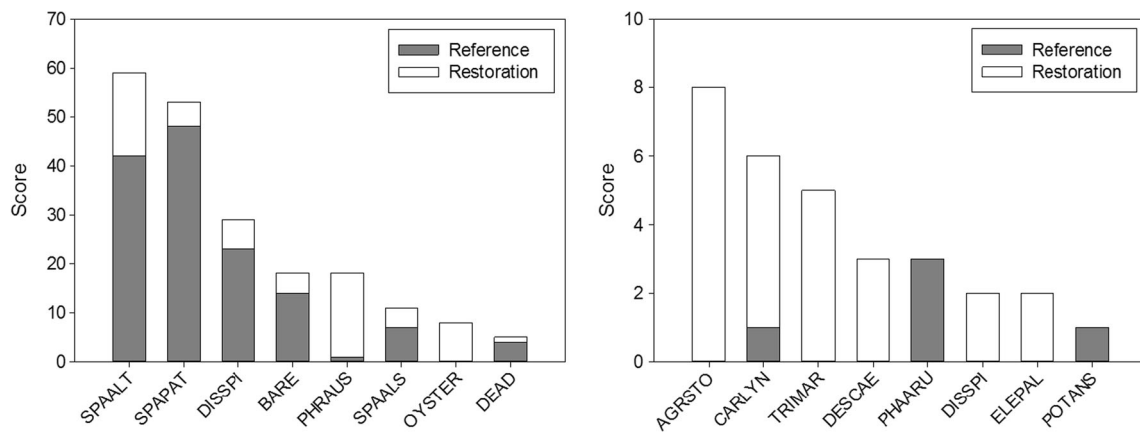


Fig. 4 Contribution of species to vegetation community dissimilarity between restoration and reference wetlands based on scores derived from SIMPER results (see text for explanation on how scores were calculated). Only species with total scores of 5 or more are shown for east coast sites (left panel); all contributing species are shown for west coast sites (right panel). SPAALT *Spartina alterniflora* (tall), SPAPAT

Spartina patens, DISSPI *Distichlis spicata*, BARE bare ground, PHRAUS *Phragmites australis*, SPAALS *S. alterniflora* (short), OYSTER oyster shell, DEAD dead vegetation, AGRSTO *Agrostis stolonifera*, CARLYN *Carex lyngbyei*, TRIMAR *Triglochin maritimum*, DESCAE *Deschampsia caespitosa*, PHAARU *Phalaris arundinacea*, ELEPAL *Eleocharis palustris*, POTANS *Potentilla anserina* ssp. *pacifica*

important explanatory parameter in Maine and Oregon, but not at the other reserves. Soil parameters were not included in any of the final models that most explained vegetation communities. Subsequent linear regression analyses show that modified RPI scores increased as groundwater depth increased (Fig. 5). In contrast, RPI scores were not significantly related to wetland surface elevation even though total RPI scores trended slightly higher with elevation.

Discussion

Moreno-Mateos et al. (2012) conducted a review of over 600 wetland restoration sites worldwide and found that structural and functional metrics in restored wetlands remained approximately 25% lower than in reference sites even after a century of restoration in some cases. Our

study, while conducted on much smaller spatial and time-scales and evaluating a snapshot of restoration (2008–10), appears to corroborate these results. We found that almost all of the 17 tidal restoration wetlands examined in our study achieved only about half of the expected restoration levels based on the monitoring parameters included in the RPI, which reiterates the difficulties inherent with trying to restore degraded wetlands to fully functional natural systems. More optimistically, our study also further highlights the value of using the RPI to evaluate restoration progress over time compared to reference conditions, identifies key abiotic parameters for assessing restoration performance, and highlights the NERRS as a national source of local reference sites and relevant long-term ecological monitoring datasets.

Assessing Restoration with the RPI

Evaluations of tidal wetland restoration projects suffer from a lack of field protocols and analysis tools that provide results that are directly comparable over multiple spatial scales. To this end, our project adds to other case studies (Chmura et al. 2012; Moore et al. 2009) by showing that the RPI can serve as an effective trajectory analysis tool for comparing multiple tidal wetland restoration projects. The modified RPI is not an absolute metric of restoration success, but it does provide a valuable measure of relative differences in restoration site performance. When carefully applied and considered, its output can challenge traditional thinking about the dynamics of ecological restoration and how comparisons can be made across disparate restoration sites. Within this context, we were able to use the RPI to quantify the relative effectiveness of 17 wetland restoration projects across five different states and between two dramatically different types of restoration

Table 6 Results from BEST analyses to identify the abiotic parameters that most influence restoration wetland vegetation communities

NERR	r	Salinity	Water depth	Elevation	Soil bulk density	Soil % organic
Wells, ME	0.38	x	x	x	x	x
Narragansett Bay, RI	0.18	x	x	x	x	x
Chesapeake Bay, VA	0.42	x	x	x	x	x
North Carolina	0.25	x		x		x
South Slough, OR	0.42	x		x	x	x

Tests were run separately for each NERR site using data from all restoration wetlands within each site. An “x” indicates the parameters that were included in model tests for each NERR; shaded cells show those parameters that explained the most variability in vegetation communities

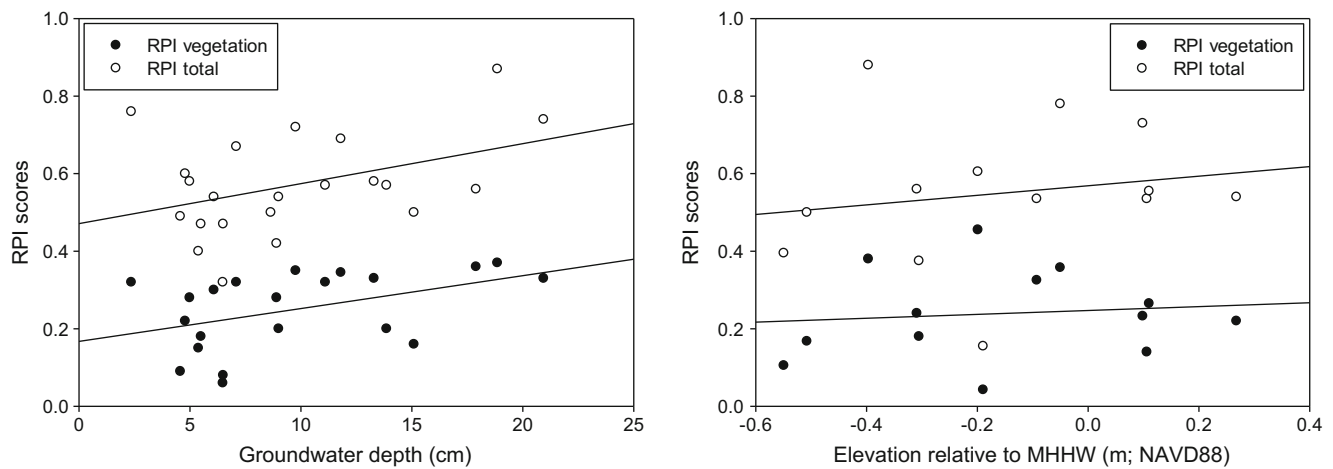


Fig. 5 Linear regressions of RPI scores (total scores and scores for vegetation component only) against influential abiotic parameters identified with BEST analyses. *Left*: vegetation RPI scores are significantly related to groundwater depth in restoration wetlands ($R^2 = 0.20$, $p = 0.04$), and total RPI scores tend to increase with

increasing groundwater depth ($R^2 = 0.16$, $p = 0.06$). *Right*: total RPI and vegetation RPI scores are not significantly related to mean restoration wetland elevation ($R^2 = 0.03$, $p = 0.58$, and $R^2 = 0.01$, $p = 0.72$, respectively)

projects. It is relevant to note that the RPI was originally developed using a selection of restoration projects in New England for which pre-restoration data were available. The intent was to capture the maximum potential change over time along each site's individual restoration trajectory. However, relatively few restoration projects are fortunate to have this level of data available. While pre-restoration data may help to more fully illustrate where the most significant restoration responses occur along the project's life and yield absolute progression, these data are not required for evaluating the system's change over time. Our application of the RPI provides insight into how marshes moved towards reference conditions during our study period (2008–2010) instead of progression from pre-restoration. Since the RPI compares restoration trajectory in relative terms with a paired reference site for each monitoring event to provide site-specific context, the index is flexible enough to be effectively applied to any point in time up to the present monitoring year. This was evident in our study, where the RPI proved useful for directly comparing the level of restoration performance among multiple wetland restoration projects and regions, as indicated by similar results yielded from more traditional approaches (e.g., plant community analysis, percent differences).

The results of the RPI depend heavily on the parameters that are included in the index. Among the RPI's hydrologic parameters, there were few large relative differences between reference and restoration wetlands. These results indicate positive hydrologic restoration performance and that core physical processes are in place to support continued structural and biological recovery. In contrast, larger relative differences in vegetation parameters remained between reference and restoration wetlands. This was driven by vegetation community composition, which remained different at 15 of 17 wetland pairs, and by vegetation species richness, which was highly

variable during this study and often led to marked changes in the RPI vegetation component across consecutive years. However, species richness variability is likely explained more by our sampling design than by measurable ecological changes. The low number of species present in most monitoring plots (generally one to three species on the east coast but ranging from three to seven in Oregon) means that a small change in species number could induce a large change in vegetation RPI scores given that richness was weighted as 25% of the total RPI score. This could be remedied by using a larger quadrat or by collecting data across more years to help put short-term inter-annual variability into temporal perspective or by including additional vegetation parameters (e.g., invasive species cover) into the RPI.

Our study reinforces the notion that hydrologic processes develop and recover more quickly than plant communities at hydrologic restoration sites (e.g., Burdick et al. 1997; Konisky et al. 2006). Tidal wetland plant community development typically requires natural recruitment of plant propagules (unless plant communities are established through active planting), development of favorable edaphic conditions, and progress through initial large-scale facilitative succession (where early colonizing species create habitat conditions that favor later successional species) followed by similar successional processes at smaller scales in response to disturbance events (Pennings and Bertness 2001). The pace of plant recovery may cause RPI scores to linger in the lower ranges for some time after restoration plan implementation. Interpretation of the RPI scores without the benefit of a solid understanding of restoration processes has the potential to inspire unnecessary or premature adaptive management actions on the part of less experienced restoration practitioners and land owners. The sometimes slower recovery of conspicuous wetland biological monitoring parameters (such as plant community

response) needs to be articulated to all project partners early on. This again highlights the need for prolonged post-restoration monitoring to document comprehensive changes that may not occur in immediate post-restoration years.

A critical component of evaluating restoration status is to understand that restoration outcomes may not exactly mirror reference conditions for many years, if ever. Therefore, it is not always realistic to expect RPI scores of 1 (i.e., 100% restored) within timescales typically used to evaluate restoration projects (i.e., 5–10 years). This is especially true for restoration efforts that address major site degradation (e.g., wetland subsidence at tide-restricted sites) or at sites where full restoration of site hydrology is not possible due to financial, practical, or social constraints placed on restoration design and implementation. In addition, restoration outcomes may not match reference conditions (RPI scores plateau below 1) because of imperfect reference site selection. For example, a restoration wetland in a polyhaline tidal river system will never fully converge with a back-barrier mesohaline reference wetland, especially if salinity and other parameters affected by salinity (e.g., vegetation) are included in the RPI. It is therefore important for restoration scientists and practitioners to incorporate realistic restoration outcomes into project goals during project conceptual planning and design and to take great care in selecting one or more reference sites that closely represent achievable target conditions for their restoration sites (Clewell et al. 2005). Of course, finding high-quality “least-disturbed” reference sites that closely match historic restoration site conditions can be a significant challenge. Even more challenging can be finding adequate funding for truly informative effectiveness monitoring from funding sources that typically prioritize restoration implementation over evaluation. When sites and funding can be found, however, the use of multiple reference sites to establish a “reference domain” can be a powerful restoration evaluation tool (Thayer et al. 2005). A reference domain provides for restoration scientists and practitioners the range of reference conditions and associated natural variability typical of the site being restored. The use of multiple reference sites and associated reference domains would greatly enhance the confidence with which scientists and practitioners could use the RPI.

Assessing Restoration Based on Vegetation Communities

One potential goal of tidal wetland restoration projects is for vegetation communities to become similar between the reference and restoration wetlands. Based on ANOSIM, this was only achieved at the Wheeler Marsh (ME) and Hermitage Museum Marsh (VA) restoration sites. Restoration and reference wetland vegetation remained significantly different at the other 15 sites. These results are consistent with our RPI results, which showed that Wheeler Marsh had the highest RPI

score among all the sites in Maine and that the Hermitage Museum Marsh had slightly higher RPI scores compared to the other sites in Virginia. It is illustrative that restoration site vegetation did not converge with reference conditions at the vast majority of sites, some of which were restored almost 10 years prior. This finding provides further support for the notion that vegetation communities recover over longer time periods or that they will likely never resemble reference conditions for various reasons including permanent establishment of invasive species or response to sea-level rise (e.g., Moreno-Mateos et al. 2012; Mossman et al. 2012). As noted when discussing the RPI, restoration practitioners may need to recalibrate expectations on vegetation recovery rates at some sites or continue regular monitoring over much longer periods of time than is common to detect desired changes.

The ANOSIM results were augmented by SIMPER analyses, which quantified the contribution of individual species to overall vegetation dissimilarity in restoration/reference site pairs. In nearly all cases, overall dissimilarity exceeded 50% and ranged as high as 89%, providing another quantitative indicator of the degree to which each restoration site still needs to change to better resemble its reference site. On the east coast, three plant species from reference wetlands (*S. alterniflora*, *S. patens*, and *D. spicata*) accounted for the vast majority of overall dissimilarity between restoration-reference pairs in this study. This result flags key species in some regions whose distribution and abundance could be increased through adaptive management measures to become more similar to reference vegetation communities.

P. australis and bare ground also contributed heavily to restoration/reference wetland vegetation dissimilarity. *P. australis* is clearly a concern as it was the fifth most important species contributing to overall restoration/reference site dissimilarity in this study and was found almost entirely in hydrologic restoration sites. This species can invade and dramatically alter tidal wetland plant community structure and function (Bertness et al. 2002; Burdick and Konisky 2003), and its resistance to eradication can reduce overall levels of restoration performance in some wetlands. Bare ground was an important contributor to dissimilarity in Maine, Rhode Island, and North Carolina due to its greater abundance at reference sites, which likely reflects normal disturbance regimes of natural wetlands that may be lacking in restoring wetlands. Disturbed patches can recover fully through a successional process (Pennings and Bertness 2001) or shift to an altered state such as pools or forb pannes (Ewanchuk and Bertness 2003; Ewanchuk and Bertness 2004; Griffin et al. 2011; Wilson et al. 2009); they could therefore be considered as important features of mature wetland vegetation assemblages. In other sites, bare patches can be indicators of invasive species activity (e.g., nutria “eat outs” in tidal marshes) or responses to waterlogging and sea-level rise (Raposa et al. 2015). Care must therefore be taken when interpreting

differences in bare ground cover between restoration and reference tidal wetlands.

Comparisons of Hydrologic and Excavation Projects

In our study, excavation and hydrologic restoration projects performed equally well as measured by total RPI scores as well as the vegetation and hydrology subcomponents of the RPI. These results are encouraging for restoration practitioners because it shows that different structural and functional tidal wetland components might be expected to recover at similar rates despite often dramatic differences in both pre-restoration condition and in site-specific restoration activities. Despite these general similarities, we also found that invasive species cover (primarily *P. australis*) varied dramatically between restoration types in our study. *P. australis* was only abundant at hydrologic restoration sites because it often invades as a result of prior tidal restrictions and associated tidal wetland freshening (Roman et al. 1984) and can be difficult to remove entirely from such sites. Although it was not quantified in our study, evidence for *P. australis* stunting after hydrologic restoration was observed at several sites in Rhode Island (i.e., Potter Pond, Walker Farm, and Silver Creek) and has been documented elsewhere (Buchsbaum et al. 2006; Roman et al. 2002). We therefore recommend adding invasive species cover as a third parameter of the vegetation component in future RPI analyses. This makes the RPI score more ecologically meaningful in light of the results shown here, and it down-weights the contribution of the species richness component, which can be highly variable across years.

Important Abiotic Parameters Affecting Restoration

Elevation and groundwater level were identified as the two primary abiotic factors most affecting plant communities. Elevation contributed to the highest correlation at all five reserves when examining abiotic/plant relationships at restoration sites, and groundwater level contributed to the highest correlation at all sites where it was tested. These findings agree with the work of tidal wetland restoration scientists and practitioners over the past decade (e.g., Kirwan et al. 2009; Morris et al. 2002; Morris 2006; Morris 2007; Mudd 2011; Mudd et al. 2009) who have quantified critical relationships between wetland surface elevation, tidal inundation regime, and wetland plant communities (i.e., tidal wetland plant species vary in their tolerance to tidal inundation frequency and duration). The relationship between RPI scores (overall and vegetation subcomponent) and groundwater level across all sites in our study further emphasizes the degree to which this abiotic factor determines plant community structure and, by extension, the level of performance in recovering tidal wetlands. This also indicates that restoration performance, as measured with the RPI at our sites, tends to improve in restoration marshes

that are well drained (i.e., have water depths well below the marsh surface).

The NERRS as Long-Term Reference Sites

The use of reference sites, especially those permanently protected within the NERRS, generally provided appropriate benchmarks to evaluate the relative success of the 17 local tidal wetland restorations included in this study. The NERRS as a national network of protected sites provides favorable and stable settings with which to evaluate and contrast restoration activities across diverse biogeographic regions. An added benefit is that science staffs at many NERR sites are engaged in ongoing NERR Biomonitoring and Sentinel Sites monitoring (Moore 2012; NERRS 2012) in which data useful for helping assess tidal wetland restoration projects are routinely collected from least-disturbed reference sites. Emergent tidal wetland vegetation attributes, wetland platform elevation, sedimentation/vertical accretion dynamics, and wetland hydrology are all included in Sentinel Sites program monitoring. Restoration practitioners may find NERRS Sentinel Sites data particularly useful because (1) reference site data have already been collected, thereby lessening the reference monitoring workload and expense, and (2) the program is designed to establish long-term datasets, which inherently provide valuable temporal perspective for any restoration project. Ironically, NERR Sentinel Sites data may also prove useful for identifying unsuitable reference sites due to ongoing degradation of otherwise least-disturbed wetlands from large-scale stressors such as sea-level rise (Watson et al. 2014). An example is from the Narragansett Bay NERR in RI, where long-term Sentinel Sites monitoring documented the accelerating loss of high marsh vegetation and linked these impacts to increased tidal flooding associated with sea-level rise (Raposa et al. 2015). Instead of serving as reference sites towards which restoration projects should aspire, sites like this might now be thought of in the opposite vein as new restoration and adaptation projects seek to slow or even reverse these ongoing impacts. Thus, we might expect that the goal of some future projects will be to increase the dissimilarity between reference sites that continue to degrade and restoration/adaptation sites that have increased resilience to change.

Recommendations for Future Restoration/Adaptation Projects

Our study provides insights into tidal wetland restoration monitoring practices that are routinely applied and adapted to multiple regions around the USA. In terms of the RPI, we recommend its continued use (following Chmura et al. 2012 and Moore et al. 2009) as a trajectory tool for assessing trends in wetland restoration performance over time. To strengthen the interpretive ability of the RPI (modified or otherwise), we

also suggest that data should be collected before and after restoration activities to quantify absolute trajectories and to capture dynamic wetland responses that can occur during early stages of post-restoration wetland recovery. When using the RPI, we recommend the continued use of the percent cover of the five most abundant reference site species in the vegetation RPI component because it correlates with ANOSIM and SIMPER results. This should be combined with vegetation species richness (especially when data are to be collected for multiple years) and the percent cover of invasive species. When considering hydrology, we concur with Neckles et al. (2013) who recommend using water level loggers at multiple sites along the full elevation gradient in each restoration and reference wetland to collect high-frequency time series data (two weeks minimum) to quantify groundwater levels and tidal inundation regime. Water-level sensors in groundwater wells accurately track water levels above and below the marsh surface and, when water level data from local tide gauges are not available, can be very useful for estimating marsh surface inundation duration. We also recommend collecting elevation data from multiple transects that traverse the full extent of the wetland platform both before and after restoration. In summary, our study highlights the utility of using the NERRS as protected reference sites with robust long-term monitoring datasets across the USA for future restoration and adaptation projects. Although specific NERR wetlands may not represent appropriate reference sites for all projects, in many cases these wetlands can provide ideal target conditions for restoration sites or a picture of ongoing degradation of natural sites for assessing the results of adaptation projects aimed at building wetland resilience to sea-level rise and other stressors.

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References

- Ball, D.F. 1964. Loss-on-ignition as an estimate of organic matter and organic carbon in non-calcareous soils. *Journal of Soil Science* 15: 84–92.
- Bertness, M.D., P.J. Ewanchuk, and B.R. Silliman. 2002. Anthropogenic modification of New England salt marsh landscapes. *Proceedings of the National Academy of Sciences* 99: 1395–1398.
- Brinson, M.M. 1993. A hydrogeomorphic classification for wetlands. Wetlands Research Program Technical Report WRP-DE-4. U.S. Army Corps of Engineers, Waterways Experimental Station, Vicksburg, Mississippi. 101 pp.
- Buchsbaum, R.N., J. Catena, E. Hutchins, and M.J. James-Pirri. 2006. Changes in salt marsh vegetation. *Phragmites australis, and nekton in response to increased tidal flushing in a New England salt marsh Wetlands* 26: 544–557.
- Burdick, D.M., and R. Konisky. 2003. Determinants of expansion for *Phragmites australis*, common reed, in natural and impacted coastal marshes. *Estuaries* 26: 407–416.
- Burdick, D.M., and C.T. Roman. 2012. Salt marsh responses to tidal restriction and restoration: a summary of experiences. In *Tidal marsh restoration: a synthesis of science and practice*, ed. C.T. Roman and D.M. Burdick, 373–382. Washington: Island Press.
- Burdick, D.M., M. Dionne, R.M. Boumans, and F.T. Short. 1997. Ecological responses to tidal restorations of two northern New England salt marshes. *Wetland Ecology and Management* 4: 129–144.
- Chmura, G.L., D.M. Burdick, and G.E. Moore. 2012. Recovering salt marsh ecosystem services through tidal restoration. In *Tidal marsh restoration: a synthesis of science and practice*, ed. C.T. Roman and D.M. Burdick, 233–251. Washington: Island Press.
- Clarke, K.R., and M. Ainsworth. 1993. A method of linking multivariate community structure to environmental variables. *Marine Ecology Progress Series* 92: 205–219.
- Clarke, K.R., R.N. Gorley, P.J. Somerfield, and R.M. Warwick. 2014. Change in marine communities: an approach to statistical analysis and interpretation, 3rd edition. PRIMER-E: Plymouth.
- Clewell, A., J. Reiger, and J. Munro. 2005. Guidelines for Developing and Managing Ecological Restoration Projects (2nd edition). Society for Ecological Restoration online publication: <http://www.ser.org/resources/resources-detail-view/guidelines-for-developing-and-managing-ecological-restoration-projects#7>. Accessed May 23, 2016.
- Cornu, C.E., and S. Sadro. 2002. Physical and functional responses to experimental marsh surface elevation manipulation in Coos Bay's south slough. *Restoration Ecology* 10: 474–486.
- Craft, C.B., S.W. Broome, and C.L. Campbell. 2002. Fifteen years of vegetation and soil development following brackish-water marsh creation. *Restoration Ecology* 10: 248–258.
- Craft, C.B., J.M. Reader, J.N. Sacco, and S.W. Broome. 1999. Twenty five years of ecosystem development of constructed *Spartina alterniflora* (Loisel) marshes. *Ecological Applications* 9: 1405–1419.
- Davies, S. 2004. Vegetation dynamics of a tidal freshwater marsh: long-term and inter-annual variability and their relation to salinity. M.S. Thesis. College of William and Mary, Virginia Institute of Marine Science. Gloucester Point, VA. USA. 75 pp.
- Dean, W. Jr. 1974. Determination of carbonate and organic matter in calcareous sediments and sedimentary rock. *Journal of Sedimentary Research* 44: 242–248.
- Diefenderfer, H.L., R.M. Thom, and J.E. Adkins. 2003. Systematic approach to coastal ecosystem restoration. Prepared for National Oceanic and Atmospheric Administration Coastal Services Center by Battelle Marine Sciences Laboratory, Sequim, WA and NOAA Coastal Services Center, Charleston, SC. 54 pp.
- Ewanchuk, P.J., and M.D. Bertness. 2003. Recovery of a northern New England salt marsh plant community from winter icing. *Oecologia* 136: 616–626.
- Ewanchuk, P.J., and M.D. Bertness. 2004. Structure and organization of a northern New England salt marsh plant community. *Journal of Ecology* 92: 72–85.
- Federal Geographic Data Committee (FGDC). 2012. Coastal and marine ecological classification standard. Marine and Coastal Spatial Data Subcommittee. FGDC-STD-018-2012.
- Godinez-Alvarez, H., J.E. Herrick, M. Mattocks, D. Toledo, and J. Van Zee. 2009. Comparison of three vegetation monitoring methods: their relative utility for ecological assessment and monitoring. *Ecological Indicators* 9: 1001–1008.

- Griffin, P.J., T. Theodose, and M. Dionne. 2011. Landscape patterns of forb pannes across a northern New England salt marsh. *Wetlands* 31: 25–33.
- Havens, K.J., L.M. Varnell, and J.G. Bradshaw. 1995. An assessment of ecological conditions in a constructed tidal marsh and two natural reference tidal marshes in coastal Virginia. *Ecological Engineering* 4: 117–141.
- James-Pirri, M.J., C.T. Roman, and E. L. Nicosia. 2012. Monitoring nekton in salt marshes. A protocol for the National Park Service's Long-Term Monitoring Program, Northeast Coastal and Barrier Network. Natural Resources Report NPS/NCBN/NRR-2012/579. U.S. Department of the Interior, National Park Service. Fort Collins, CO. 128 pp.
- Kentula, M.E. 2000. Perspectives on setting success criteria for wetland restoration. *Ecological Engineering* 15: 199–209.
- Kirwan, M.L., G.R. Guntenspergen, and J.T. Morris. 2009. Latitudinal trends in *Spartina alterniflora* productivity and the response of coastal marshes to global change. *Global Change Biology* 15: 1982–1989.
- Konisky, R.A., D.M. Burdick, M. Dionne, and H.A. Neckles. 2006. A regional assessment of salt marsh restoration and monitoring in the Gulf of Maine. *Restoration Ecology* 14: 516–525.
- Montgomery, J., C. Zimmermann, and M. Price. 1979. The collection, analysis and variation of nutrients in estuarine pore water. *Estuarine and Coastal Marine Science* 9: 203–214.
- Moore, G.E., D.M. Burdick, C.R. Peter, A. Leonard-Duarte, and M. Dionne. 2009. Regional assessment of tidal marsh restoration in New England using the Restoration Performance Index. Final report submitted to NOAA Restoration Center. 237 pp.
- Moore, K. 2012. Draft revised NERRS SWMP vegetation monitoring protocol: long-term monitoring of estuarine submersed and emergent vegetation communities. National Estuarine Research Reserve System Technical Report. 25 pp.
- Moreno-Mateos, D., M.E. Power, F.A. Comín, and R. Yockteng. 2012. Structural and functional loss in restored wetland ecosystems. *PLoS Biology* 10 (1): e1001247. doi:10.1371/journal.pbio.100124.
- Morris, J.T. 2006. Competition among marsh macrophytes by means of geomorphological displacement in the intertidal zone. *Estuarine and Coastal Shelf Science* 69: 395–402.
- Morris, J.T. 2007. Ecological engineering intertidal saltmarshes. *Hydrobiologia* 577: 161–168.
- Morris, J.T., P.V. Sundareshwar, C.T. Nietch, B. Kjerfve, and D.R. Cahoon. 2002. Responses of coastal wetlands to rising sea level. *Ecology* 83: 2869–2877.
- Mossman, H.L., A.J. Davy, and A. Grant. 2012. Does managed coastal realignment create saltmarshes with 'equivalent biological characteristics' to natural reference sites? *Journal of Applied Ecology* 49: 1446–1456.
- Mudd, S.M. 2011. The life and death of salt marshes in response to anthropogenic disturbance of sediment supply. *Geology* 39: 511–512.
- Mudd, S.M., S.M. Howell, and J.T. Morris. 2009. Impact of dynamic feedbacks between sedimentation, sea-level rise, and biomass production on near-surface marsh stratigraphy and carbon accumulation. *Estuarine, Coastal and Shelf Science* 82: 377–389.
- National Estuarine Research Reserve System. 2002. Restoration science strategy: a framework. National Estuarine Research Reserve System Technical Report. National Oceanic and Atmospheric Administration, Silver Spring, Maryland. 33 pp.
- National Estuarine Research Reserve System. 2012. Sentinel Sites program guidance for climate change impacts. NERRS final program guidance. 24 pp.
- Neckles, H.A., M. Dionne, D.M. Burdick, C.T. Roman, R. Buchsbaum, and E. Hutchins. 2002. A monitoring protocol to assess tidal restoration of salt marshes on local and regional scales. *Restoration Ecology* 10: 556–563.
- Neckles, H.A., G.R. Guntenspergen, W.G. Shriver, N.P. Danz, W.A. Wiest, J.L. Nagel, and J.H. Olker. 2013. Identification of metrics to monitor salt marsh integrity on National Wildlife Refuges in relation to conservation and management objectives. Final report to U.S. Fish and Wildlife Service, Northeast Region. USGS Patuxent Wildlife Research Center, Laurel, MD. 226 pp.
- Pennings, S.C., and M.D. Bertness. 2001. Salt marsh communities. In *Marine community ecology*, ed. M.D. Bertness, M.E. Hay, and S.D. Gaines, 289–316. Sunderland, MA: Sinauer.
- Raposa, K.B., R.L. Weber, M.C. Ekberg, and W. Ferguson. 2015. Vegetation dynamics in Rhode Island salt marshes during a period of accelerating sea level rise and extreme sea level events. *Estuaries and Coasts*. doi:10.1007/s12237-015-0018-4.
- Roegner, G.C., H.L. Diefenderfer, A.B. Borde, R.M. Thom, E.M. Dawley, A.H. Whiting, S.A. Zimmerman, and G.E. Johnson. 2008. Protocols for monitoring habitat restoration projects in the Lower Columbia River and Estuary. PNNL-15793. Report by Pacific Northwest National Laboratory, National Marine Fisheries Service, and Columbia River Estuary Study Taskforce submitted to the U.S. Army Corps of Engineers, Portland District, Portland, Oregon.
- Roman, C.T., M.J. James-Pirri, and J.F. Heltshe. 2001. Monitoring salt marsh vegetation: a protocol for the long-term Coastal Ecosystem Monitoring Program at Cape Cod National Seashore. 47 pp.
- Roman, C.T., W.A. Niering, and R.S. Warren. 1984. Salt marsh vegetation change in response to tidal restriction. *Environmental Management* 8: 141–150.
- Roman, C.T., K.B. Raposa, S.C. Adamowicz, M.J. James-Pirri, and J.G. Catena. 2002. Quantifying vegetation and nekton response to tidal restoration of a New England salt marsh. *Restoration Ecology* 10: 450–460.
- Short, F.T., D.M. Burdick, C.A. Short, R.C. Davis, and P.A. Morgan. 2000. Developing success criteria for restored eelgrass, salt marsh and mud flat habitats. *Ecological Engineering* 15: 239–225.
- Simenstad, C.A., C.D. Tanner, R.M. Thom, and L. Conquest. 1991. Estuarine habitat assessment protocol. EPA 910/9–91-037, Puget Sound Estuary Program, U.S. Environmental Protection Agency-Region 10, Seattle, WA. 191 pp., plus appendices.
- Soil Survey Staff. 2014. Soil Survey Field and Laboratory Methods Manual. Soil Survey Investigations Report No. 51, Version 2.0. R. Burt and Soil Survey Staff (ed.). U.S. Department of Agriculture, Natural Resources Conservation Service.
- Sprecher, S.W. 2000. Installing monitoring wells/piezometers in wetlands. WRAP Technical Notes Collection, ERDC TN-WRAP-00-02, U.S. Army Engineer Research and Development Center, Vicksburg, MS.
- Staszak, L.A., and A.R. Armitage. 2013. Evaluating salt marsh restoration success with an index of ecosystem integrity. *Journal of Coastal Research* 29: 410–418.
- Thayer, G.W., T.A. McTigue, R.J. Salz, D.H. Merkey, F.M. Burrows, and P.F. Gayaldo, (Eds.). 2005. Science-based restoration monitoring of coastal habitats, volume two: tools for monitoring coastal habitats. NOAA Coastal Ocean Program Decision Analysis Series No. 23. NOAA National Centers for Coastal Ocean Science, Silver Spring, MD. 628 pp., plus appendices.
- Watson, E.B., A.J. Oczkowski, C. Wigand, A.R. Hanson, E.W. Davey, S.C. Crosby, R.L. Johnson, and H.M. Andrews. 2014. Nutrient enrichment and precipitation changes do not enhance resiliency of salt marshes to sea level rise in the northeastern U.S. *Climatic Change* 125: 501–509.
- Wilson, K.R., J.T. Kelley, A. Croitoru, M. Dionne, D. Belknap, and R. Steneck. 2009. Salt pools are secondary and dynamic features of the Webhannet estuary, wells, Maine, USA. *Estuaries and Coasts* 33: 855–870.
- Yozzo, D.J., and M.S. Laska. 2006. Restoration program assessment of the National Estuarine Research Reserve System. *Ecological Restoration* 24: 13–21.
- Zedler, J.B., and J.C. Callaway. 2000. Evaluating the progress of engineered tidal wetlands. *Ecological Engineering* 15: 211–225.