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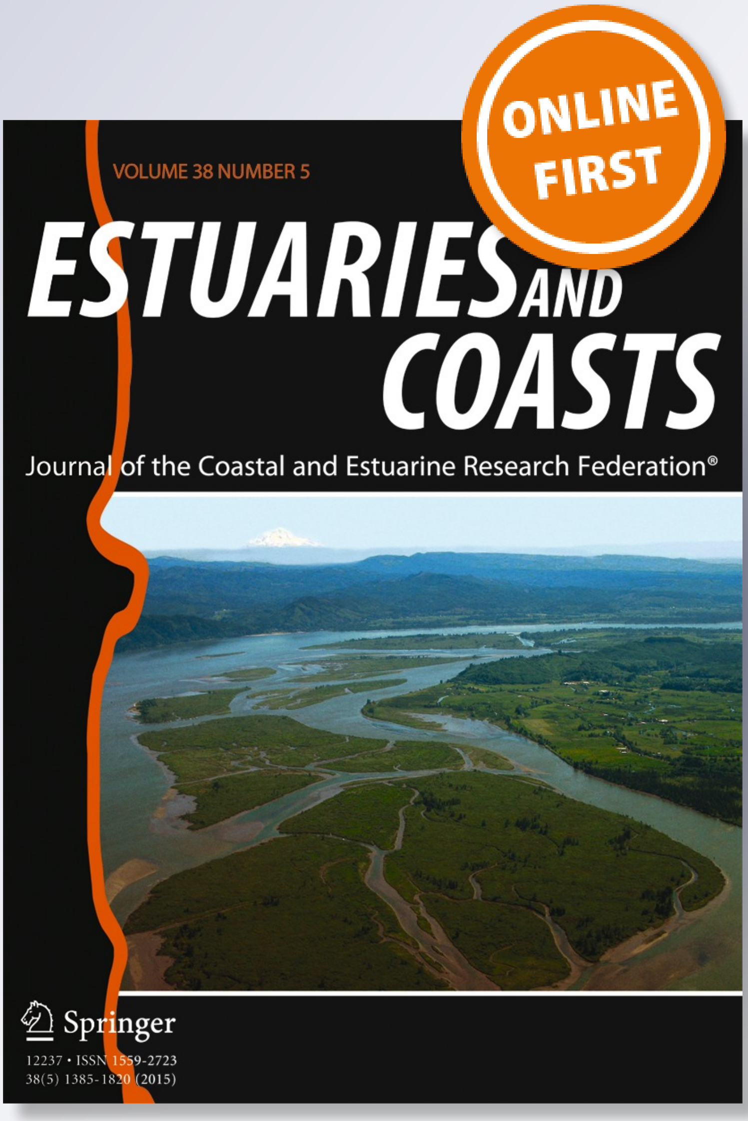

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Vegetation Dynamics in Rhode Island Salt Marshes During a Period of Accelerating Sea Level Rise and Extreme Sea Level Events

Kenneth B. Raposa[1] · **Robin L. J. Weber**[1] · **Marci Cole Ekberg**[2] · **Wenley Ferguson**[2]

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Abstract Sea level rise is a major stressor on many salt marshes, and its impacts include creek widening, ponding, vegetation dieback, and drowning. Marsh vegetation changes have been associated with sea level rise across southern New England, but most of these studies pre-date the current period of rapidly accelerating sea level rise coupled with episodic events of extreme increases in water levels. Here, we combine data from two salt marsh monitoring and assessment programs in Rhode Island that were designed to assess marsh responses to sea level rise and use these data to document temporal and spatial patterns in marsh vegetation during the current period of extreme water level increases. Vegetation monitoring at two Narragansett Bay salt marshes confirms the ongoing decline of the salt meadow species *Spartina patens* during this period as it becomes replaced by *Spartina alterniflora*. Bare ground resulting from vegetation dieback was significantly related to mean high water levels and led to the rapid conversion of mixed *Spartina* assemblages to *S. alterniflora* monocultures. A broader spatial assessment of RI marshes shows that *S. alterniflora* dominance increases at lower elevation marshes toward the mouth of Narraganset Bay. Our data provide additional evidence that *S. patens* continues to decline in southern New England marshes and show that losses can accelerate during periods of extreme high water levels. Unless adaptive management actions are taken, we predict that marshes throughout RI will continue to lose salt meadow habitat and eventually resemble lower elevation marshes that are already dominated by *S. alterniflora* monocultures.

Keywords Narragansett Bay · Salt marsh · *Spartina patens* · Sea level rise · Dieback

Introduction

New England salt marshes protect shorelines from storms and floods (Gedan et al. 2011a; Shepard et al. 2011), provide nursery and breeding habitat for fish and birds (Hanson and Shriver 2006; McKinney and Wigand 2006; Raposa and Roman 2001; Roman et al. 2000), filter nutrients and pollutants (Valiela and Cole 2002), and sequester carbon (Kirwan and Mudd 2012; Mcleod et al. 2011). These and other valuable ecosystem services, however, have often been impacted by historic human stressors such as mosquito ditches (Vincent et al. 2013), tidal restrictions (Burdick and Roman 2012), and marsh fill (Bromberg and Bertness 2005). Previously compromised salt marsh landscapes are more vulnerable to current threats including non-native species invasion (i.e., *Phragmites australis*; Roman et al. 1984), elevated nutrient inputs (Deegan et al. 2012), and creek bank grazing and erosion from overabundant crab populations (Coverdale et al. 2013). While the relative degree to which these current impacts occur can vary among individual marshes, a more universal threat is from increasing rates of sea level rise (Gedan et al. 2011b; Kirwan and Megonigal 2013).

Historically, New England salt marsh development and maintenance within the tidal frame was accomplished through vertical accretion and lateral transgression into adjacent upland areas at rates commensurate with those of local sea level rise (Redfield 1972). Even through the 1980s–1990s, vertical salt marsh accretion was able to keep pace with sea level rise

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through a combination of inorganic sediment deposition and organic biomass accumulation (Bricker-Urso et al. 1989; Roman et al. 1997). However, there is growing concern that a combination of reduced sediment supplies and rapidly accelerating rates of sea level rise (which are three to four times higher than the global average) is now leading to marsh accretion deficits throughout the region (Sallenger et al. 2012; Weston 2014). In addition, extensive manmade structures such as dams, bulkheads, seawalls, or naturally steep topography will inhibit the lateral transgression of many New England marshes (Bozek and Burdick 2005; Titus et al. 2009). As a consequence, researchers and managers are increasingly documenting the waterlogging and degradation of vegetated salt marsh platforms (Smith 2009; Warren and Niering 1993).

Impacts to salt marshes from sea level rise include vegetation shifts, marsh dieback, marsh surface ponding, and marsh peat subsidence (Alber et al. 2008; Hartig et al. 2002; Kirwan and Megonigal 2013). Dramatic examples of these impacts have been reported from low elevation areas subject to rapid rates of relative sea level rise and declining sedimentation rates such as the Mississippi delta region (Blum and Roberts 2009) and Chesapeake Bay (Kearney et al. 2002). In southern New England, the first documented sea level rise impacts to salt marshes were provided by Warren and Niering (1993), who observed a loss of high marsh *Spartina patens* in favor of stunted *Spartina alterniflora* and other species in a Connecticut marsh over a 40-year period. A subsequent study in nearby Narragansett Bay, Rhode Island also attributed a decrease in *S. patens* and simultaneous increase in *S. alterniflora* over a 4-year period (1995–1998) to rising sea levels (Donnelly and Bertness 2001). More recently, Smith (2009) conducted a time series mapping analysis to document a loss of high marsh habitats from 1938 to 2005 over a broad area on Cape Cod, MA that was partially attributed to sea level rise.

These studies provided some of the first evidence of impacts to southern New England salt marshes from prolonged inundation and waterlogging associated with accelerating sea level rise. Evaluations of more recent impacts to salt marshes are needed, however, because sea levels are now rising even faster than when these studies were conducted (Boon 2012). In Narragansett Bay, for example, the long-term rate of sea level rise (2.7 mm year^{-1} from 1930 to 2012 at the Newport, RI tide station) is substantially lower than rates calculated from more current data over shorter time scales (e.g., 4.6 mm year^{-1} increase in mean high water (MHW) from 1985 to 2000 and 7.5 mm year^{-1} from 2000 to 2013, which is when vegetation monitoring occurred in our study). The dramatic rate increase in recent years is partly due to anomalously high water levels associated with a high number of cool-season extratropical storms (Sweet and Zervas 2011) and with an extreme rise in regional sea levels in 2009–2010 (Goddard et al. 2015). Similarly accelerating rates of sea level rise have been documented at tide stations across the northeastern US coast (Boon 2012). These trends are likely to continue because sea levels are predicted to rise even faster during the twenty-first century (Church et al. 2013). For these reasons, southern New England is an ideal region to examine ongoing salt marsh responses to accelerating sea level rise and anomalously high water levels.

Two new monitoring and assessment programs are currently underway in Rhode Island to quantify the status and response of salt marsh vegetation, habitats, and soils to multiple stressors, with a focus on impacts from sea level rise. The first effort is the National Estuarine Research Reserve System's Sentinel Sites Program, which is ongoing at two salt marshes in the Narragansett Bay National Estuarine Research Reserve (NBNERR) (NERRS 2012). This is an intensive, long-term monitoring program designed specifically to quantify impacts to salt marshes from sea level rise. The second effort is the Rhode Island Salt Marsh Assessment (RISMA) program, which is designed to characterize the current condition of Rhode Island's high marsh communities and soils. The NBNERR effort provides a temporal perspective on changes that are occurring to marsh vegetation and habitats, whereas the state-wide assessment provides a snapshot of marsh condition at a much broader scale across Narragansett Bay and coastal Rhode Island. The RISMA program data are also useful for placing any long-term changes observed at the Sentinel Sites marshes into a broader spatial perspective. The goals of this study are to (1) document temporal and spatial patterns in the vegetation composition of Rhode Island salt marshes during the current period of accelerating sea level rise, (2) assess vegetation change patterns in relation to marsh surface elevations, and (3) identify the effects of short-term extreme rises in sea level on dominant vegetation species. Our study provides a case study illustrating the ongoing decline in high salt marsh vegetation in southern New England and links rapid changes in vegetation to recent extreme increases in sea levels.

Methods

Study Sites

All Sentinel Sites data were collected from the Coggeshall and Nag salt marshes within the NBNERR on Prudence Island, RI (Fig. 1). Coggeshall Marsh is a 25-ha meadow marsh that exchanges tidal water with Narragansett Bay along a long vegetated marsh edge and via two main tidal creeks. Nag Marsh is a 15-ha back-barrier meadow marsh that lies between a vegetated sand dune and a road; tidal exchange with the Bay occurs through a single main tidal creek that bisects the marsh. Spring and neap tide ranges in Nag Marsh are approximately 0.75 and 0.35 m, respectively, while salinity ranges from 11 to

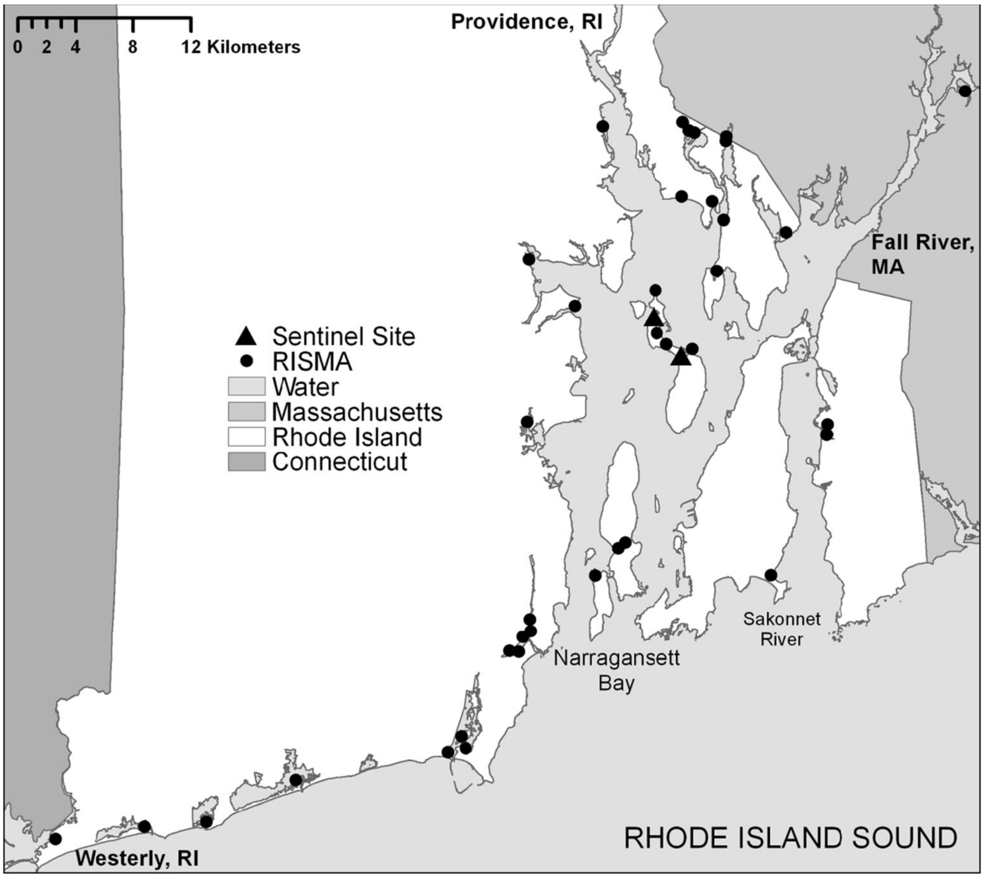


Fig. 1 Locations of the NBNERR Coggeshall and Nag Sentinel Sites salt marshes and the additional 27 RISMA marshes located across coastal Rhode Island and nearby Massachusetts (38 marsh units were sampled from these 27 marshes)

35 ppt, with a mean value of 27 ppt. These data are not available for Coggeshall Marsh but are likely similar to Nag due to the close proximity of the marshes to each other.

Coggeshall and Nag marshes are both dominated by typical New England salt marsh vegetation (e.g., *S. alterniflora*, *S. patens*, *Distichlis spicata*, *Juncus gerardii*, *Iva frutescens*, etc.) predictably distributed along an elevation gradient (Niering and Warren 1980). Both marshes are comprised of networks of tidal creeks, ponds, pools, and historic mosquito ditches and both remain relatively un-impacted by common anthropogenic stressors such as tidal restrictions, invasive species (i.e., the common reed *P. australis*), or elevated terrestrial nutrient inputs.

Vegetation data were collected from 38 discrete marsh units at 27 salt marshes in 2012–2013 as part of RISMA (Fig. 1). These marshes were located within the Narragansett Bay watershed and south coastal Rhode Island, including seven in the southern Rhode Island coastal ponds, five in the Narrow River (a sub-estuary of Narragansett Bay), and one along the Taunton River in Massachusetts (a tributary to Narragansett Bay). Marsh units ranged in size from 0.8 to 22.7 ha and were generally dominated by the same vegetation species as described above for Coggeshall and Nag marshes. Tide and salinity data were not available for each of these individual sites, although they are likely similar to Coggeshall and Nag because they are all within the same general geographic area.

Sentinel Sites Monitoring

Vegetation data were collected during 9 years between 2000 and 2013 at Coggeshall Marsh and every year from 2008 to 2013 at Nag Marsh. All data were collected around the end of the growing season (late August to early September) following the protocols described by Roman et al. (2001). In brief, this involved establishing permanent vegetation monitoring transects that stretched across the marsh platform from the upland edge to the estuary, with multiple permanent plots spaced along each transect. Three transects were established in each marsh with a total of 21 monitoring plots in Coggeshall Marsh and 24 plots in Nag Marsh. Community composition and percent cover of individual species (and other marsh features) were quantified within each plot each year using the point intercept technique at 50 points within each 1-m^2 plot. Elevations of the marsh platform in Coggeshall and Nag marshes were also collected with a Trimble R8 GPS enabled for real-time kinematic surveys and referenced to the North American Vertical Datum of 1988 (NAVD88). Elevations were collected at every meter interval along the lengths of each transect and from five locations within each vegetation plot in 2013.

In order to relate marsh elevations to inundation, local water level data were collected from a tidal creek in each marsh with an Onset Hobo logger that was installed to a known elevation. Pressure data were collected every 10 min for approximately 1 month in fall 2014, converted to water depth after compensating for barometric pressure, and referenced to NAVD88. Mean high tide was then calculated for each marsh during this time period. Additional water level data were obtained from the closest NOAA tide station (located in Newport, RI, approximately 13 and 16 km from Nag and Coggeshall marshes, respectively) to help explain any temporal patterns that emerged from vegetation monitoring. Monthly MHW data were obtained for the 2000–2013 period, averaged across all months for each year, and referenced to NAVD88 (NOAA CO-OPS 2014).

RISMA

RISMA involves the assessment of marsh vegetation, habitats, and soils in each marsh, but here, we only use data from the belt transect habitat monitoring component. Within each RISMA marsh, data were collected along transects that extended across the marsh platform from the upland border to the estuary. The number of transects per marsh ranged from one to seven, depending on marsh size. The composition and extent of habitats were quantified by laying out a meter tape across the entire length of each transect and then measuring all locations (in meters) where habitat composition changed. In this study, we are only concerned with pure salt meadow habitat and habitats dominated by *S. alterniflora* (i.e., pure *S. alterniflora* and/or *S. alterniflora* with lesser amounts of salt meadow species).

Data Analysis

For Sentinel Sites point intercept data, Analysis of Similarity (ANOSIM) was used to statistically compare vegetation community composition among time periods that had been previously identified using cluster analysis. In Coggeshall Marsh, the time periods were 2000–2004, 2008–2009, and 2010–2013. In Nag Marsh, the time periods were 2008–2009 and 2010–2013. Similarity Percentages (SIMPER) was then used to identify species that contributed to community similarity within each time period and community dissimilarity between any time periods that were found to be significantly different with ANOSIM. All data were square-root transformed prior to ANOSIM and SIMPER analyses to dampen the weight of dominant species. Resemblance matrices were created using Bray-Curtis similarities prior to all ANOSIM analyses. All cluster, ANOSIM, and SIMPER analyses were run separately for Coggeshall and Nag marshes because the data were collected over different periods of time, and all analyses were run using PRIMER version 6.1.2 (Clarke and Gorley 2006; Clarke and Warwick 2001). In addition, annual mean percent cover for each species was fitted with best-fit regression (using SigmaPlot version 12 and SigmaStat version 3.5) to identify any significant changes in cover over time. All elevation data were adjusted to a common NAVD88 orthometric reference system (Geoid03) and compared between the two marshes with a Mann–Whitney rank sum test. Elevation frequency distributions were also generated for each marsh and compared with a Kolmogorov-Smirnov test.

For RISMA, the relative composition of salt meadow and *S. alterniflora*-dominated habitats within each marsh unit was calculated by dividing the total length of each habitat by the total length of all transects. The relative composition of each habitat was then regressed against the latitude of each marsh unit to explore spatial patterns along the north–south gradient of coastal Rhode Island.

Results

Vegetation Plots

From point intercept vegetation monitoring, 17 vegetation species and 3 additional marsh features (bare ground, wrack, and stone) were documented in Coggeshall and Nag marshes across the study period (Table 1). Salt marsh vegetation community composition in Coggeshall Marsh changed significantly over time based on ANOSIM (global $R=0.04$, $p=0.018$). Specifically, a significant shift in vegetation composition was found between the 2000–2004 and 2010–2013 time periods (pair-wise test, $R=0.063$, $p=0.002$; vegetation did not change between the other time periods). A subsequent SIMPER test showed that almost half (46 %) of the overall dissimilarity in vegetation communities between the two time periods was due to a decrease in *S. patens* (comprised 25.3 % of overall community dissimilarity) and a concomitant increase in *S. alterniflora* (20.8 %). In Nag Marsh, vegetation communities did not change significantly between the 2008–2009 and 2010–2013 time periods (ANOSIM, global $R=-0.016$, $p=0.70$). However, a SIMPER analysis showed that the contributions of *S. alterniflora* and *S. patens* to community structure changed over time in a manner consistent with Coggeshall Marsh. In 2008–2009, Nag Marsh community similarity was mostly driven by *S. patens* (47 %), followed by *S. alterniflora* (34 %). The opposite was observed in 2010–2013, when *S. alterniflora* and *S. patens* contributed 41 and 36 % to community similarity, respectively.

When considering the percent cover of individual species, a statistically significant decrease in *S. patens* over time was found in both Coggeshall (polynomial regression, $R^2=0.96$, $p<0.001$) and Nag (linear regression, $R^2=0.96$, $p<0.001$)

Table 1 Vegetation composition and percent cover in Coggeshall and Nag marshes during three time periods between 2000 and 2013 from point-intercept monitoring. For each time period, data are means (and standard errors) across years. Species are listed in order of decreasing mean cover across all sampling years

Species	Coggeshall			Nag	
	2000–2004	2008–2009	2010–2013	2008–2009	2010–2013
Spartina alterniflora	48.35 (4.69)	59.53 (2.95)	67.67 (1.73)	43.83 (0.26)	48.18 (0.95)
Spartina patens	47.83 (1.42)	42.20 (1.91)	29.44 (2.94)	57.35 (0.74)	48.40 (2.24)
Distichlis spicata	14.19 (2.32)	13.34 (0.28)	17.87 (0.63)	20.18 (0.00)	21.58 (0.62)
Juncus gerardii	4.47 (0.63)	12.29 (0.19)	10.31 (0.99)	6.70 (0.88)	7.56 (0.17)
Iva frutescens	8.57 (1.32)	9.34 (1.72)	10.33 (0.72)	4.22 (0.04)	3.35 (0.18)
Bare ground	0.62 (0.00)	2.24 (1.00)	4.74 (0.96)	1.73 (0.18)	3.60 (0.63)
Salicornia spp.	2.91 (0.57)	0.10 (0.10)	2.81 (1.93)	1.78 (1.78)	3.31 (1.99)
Limonium nashii	0.91 (0.27)	3.05 (0.95)	0.93 (0.25)	0.49 (0.04)	0.27 (0.13)
Scirpus sp.	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	2.57 (0.66)	1.42 (0.49)
Wrack	0.37 (0.26)	0.57 (0.57)	0.07 (0.07)	1.05 (1.05)	0.02 (0.02)
Stone	0.00 (0.00)	1.48 (0.23)	0.55 (0.29)	0.00 (0.00)	0.00 (0.00)
Agalinis maritima	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.53 (0.08)	0.02 (0.02)
Solidago sempervirens	0.27 (0.09)	0.05 (0.05)	0.19 (0.05)	0.00 (0.00)	0.00 (0.00)
Panicum virgatum	0.38 (0.03)	0.00 (0.00)	0.05 (0.05)	0.00 (0.00)	0.00 (0.00)
Aster tenuifolium	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.23 (0.12)
Baccharis halimifolia	0.14 (0.14)	0.00 (0.00)	0.02 (0.02)	0.00 (0.00)	0.00 (0.00)
Setaria sp.	0.11 (0.11)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Unknown grass	0.00 (0.00)	0.05 (0.05)	0.05 (0.05)	0.00 (0.00)	0.00 (0.00)
Morella pensylvanica	0.10 (0.10)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Cyperus sp.	0.02 (0.02)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)

(Fig. 2). *S. patens* decreased faster in Coggeshall than in Nag, and if the current trend continues, it could be lost from these areas in Coggeshall within 5 years. *S. alterniflora* increased linearly in both Coggeshall ($R^2=0.71$, $p<0.01$) and Nag ($R^2=0.75$, $p=0.03$). Finally, *J. gerardii* increased in Coggeshall only ($R^2=0.56$, $p=0.02$), whereas *I. frutescens* decreased in Nag only ($R^2=0.86$, $p=0.08$). The cover of all other vegetation species did not change significantly over time in either marsh.

During our study, areas of vegetation dieback (defined here as the death of live vegetation within consecutive years, leaving behind mostly exposed peat) greatly expanded throughout both marshes beginning in 2009–2010. In fact, a significant peak in bare ground resulting from dieback was observed in 2011 in both Nag (polynomial regression, $R^2=0.99$, $p=0.02$) and Coggeshall (polynomial regression, $R^2=0.86$, $p=0.05$) marshes (Fig. 2). These changes in bare ground cover over time were directly related to water levels in Narragansett Bay. There was a significant positive relationship between annual bare ground cover (averaged between the two marshes) and annual MHW over the course of this study (linear regression, $R^2=0.85$, $p<0.001$; Fig. 3a). When new outbreaks of vegetation dieback affected individual monitoring plots, bare ground cover increased by an average of 26 % in 1 year in the affected plots. These vegetation dieback events differentially affected *S. patens* and *S. alterniflora*. *S. patens* cover decreased by 41 % on average during the year in which a dieback event occurred and did not recover over the next 2 years (Fig. 3b). *S. alterniflora* decreased much less than *S. patens* during dieback events, and it rapidly recovered to almost 100 % cover over the next 2 years.

Marsh Elevations

Temporal trends in *S. patens* cover varied depending on monitoring plot elevation. *S. patens* cover declined significantly in monitoring plots at a mean elevation of 0.59 m NAVD88 but was stable in plots at a higher mean elevation of 0.63 m NAVD88. Elevations between these two group of monitoring plots were significantly different (*t* test, $t=-3.482$, $p=0.002$). At a broader scale, *S. patens* declined faster in Coggeshall Marsh, which was significantly lower in elevation than Nag Marsh (0.59 vs. 0.63 m; Mann–Whitney rank sum test, $p<0.001$). The distribution of elevations also differed between the two marshes (Kolmogorov-Smirnov test, $D=0.33$, $p=0.028$), with the distribution at Coggeshall skewed toward lower elevations relative to Nag Marsh (Fig. 4). In addition, high tide levels were on average 4 cm higher in Coggeshall compared to Nag (0.65 and 0.61 m, respectively, based on Hobo loggers). This translates to greater tidal inundation at Coggeshall Marsh

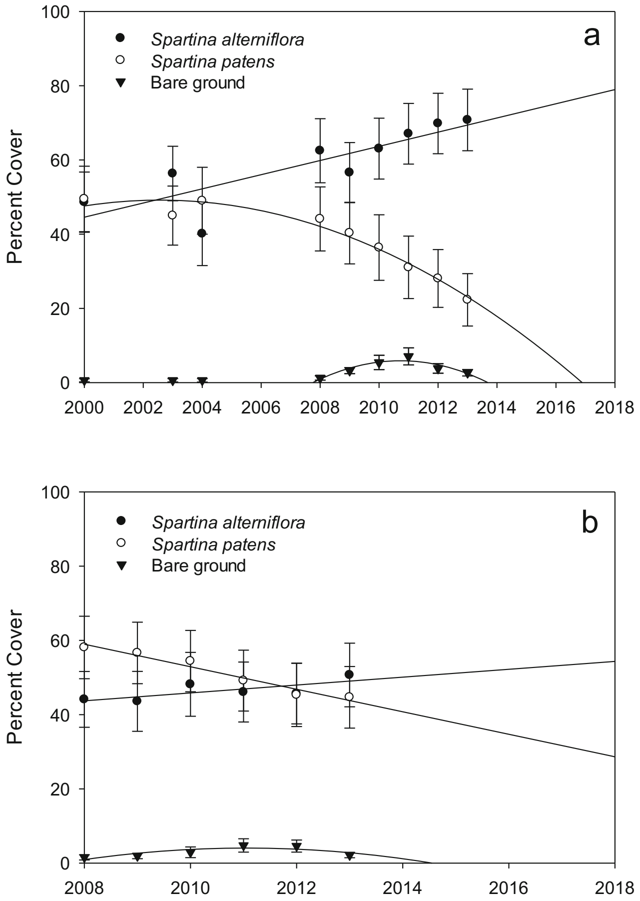


Fig. 2 Changes in the cover of *S. alterniflora*, *S. patens*, and bare ground over time in Coggeshall (**a**) and Nag (**b**) marshes. *Error bars* are 1 SE and *curves* were fit with linear and polynomial regressions (the regression for bare ground in Coggeshall was fit using data from 2008 to 2013 only)

where 79 % of the marsh surface was lower than the local mean high tide level compared to only 38 % of the surface of Nag Marsh.

Spatial Context of Sentinel Sites Marshes

From the initial round of RISMA, a significant increase in the relative cover of *S. alterniflora*-dominated habitats was found with a concomitant significant decrease in salt meadow while moving south within the study area (Fig. 5). Both of these latitudinal trends were statistically significant (linear regression, R^2=0.59, p<0.001 for *S. alterniflora* habitats; R^2=0.43, p<0.001 for salt meadow). Latitude explained less of the variability in relative habitat cover when considering marshes only in Narragansett Bay (i.e., by excluding data from the microtidal coastal ponds and the Narrow River), but the trends remained significant (R^2=0.54, p<0.001 for *S. alterniflora*-dominated habitats; R^2=0.29, p=0.004 for salt meadow). The actual amount of *S. alterniflora* and salt meadow habitats in Nag Marsh were approximately equal to the amount predicted by the regression (Fig. 5). The cover of *S. alterniflora*-dominated habitats was higher than predicted in Coggeshall Marsh, however, and salt meadow cover was lower than predicted.

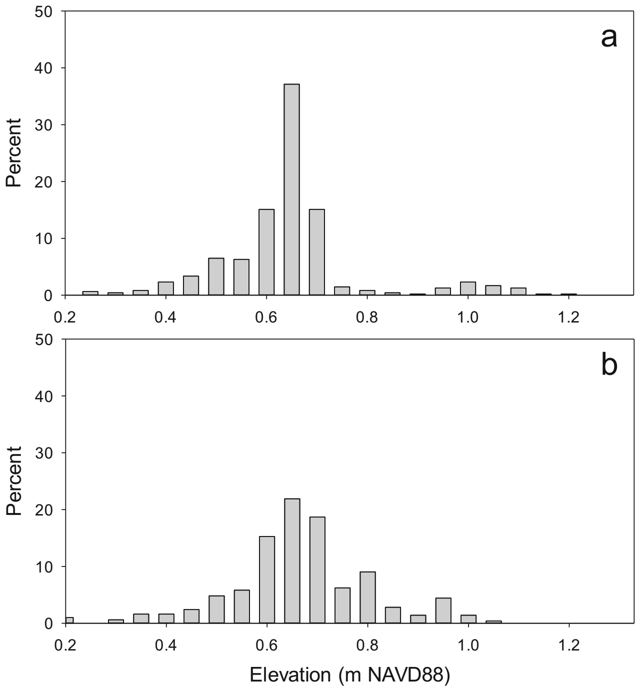
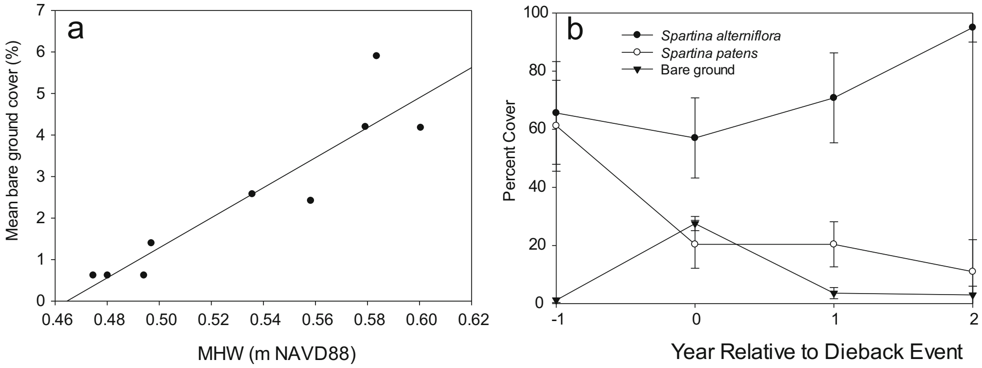


Fig. 4 Elevation frequency distributions for Coggeshall (**a**) and Nag (**b**) salt marshes. All data were collected with an RTK GPS every meter along three monitoring transects in each marsh in 2013

Fig. 3 Relationship between annual bare ground cover (a proxy for dieback; averaged across Coggeshall and Nag marshes) and annual MHW (**a**). Changes in *S. alterniflora*, *S. patens*, and bare ground cover over 4 years standardized to dieback events (=year 0) (**b**)



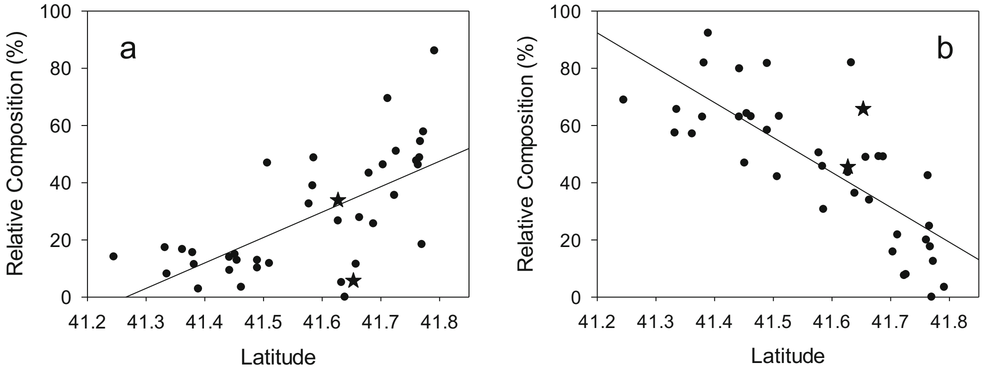


Fig. 5 Relative composition of salt meadow (*S. patens* or *D. spicata* or both) (**a**) and *S. alterniflora* habitats (pure *S. alterniflora* and/or *S. alterniflora* mixed with salt meadow) (**b**) across latitude within coastal Rhode Island. The two NBNERR Sentinel Sites are denoted with *stars*

Discussion

The replacement of *S. patens* salt meadow by encroaching *S. alterniflora* was first documented in Rhode Island by Donnelly and Bertness (2001). Our study shows that this vegetation shift is ongoing in Narragansett Bay salt marshes and confirms that *S. patens* salt meadow is declining in multiple marshes across the southern New England region (Smith 2009; Smith et al. 2012; Warren and Niering 1993). Our long-term data from Coggeshall Marsh also demonstrate that these vegetation changes in Rhode Island salt marshes have accelerated in recent years coincident with extreme increases in sea levels during 2009–2010 (Goddard et al. 2015). However, whereas the *S. patens* decline in our study was primarily due to replacement by *S. alterniflora*, there was less evidence of impacts from encroaching *S. alterniflora* on outer Cape Cod where flooding stress and crab herbivory both contributed to high marsh dieback (Smith et al. 2012). This indicates that multiple stressors may be acting synergistically to drive the loss of *S. patens* salt meadow from southern New England salt marshes.

The replacement of *S. patens* by *S. alterniflora* can be driven by nitrogen enrichment (Bertness et al. 2002; Emery et al. 2001) and sea level rise (Donnelly and Bertness 2001; Warren and Niering 1993). While our study was forensic rather than deterministic, nitrogen-driven vegetation shifts appear unlikely for the most recent decade because nitrogen inputs to the marshes have likely decreased or remained unchanged, depending on the source. Terrestrial nitrogen inputs have probably not changed over time because they highly correlate with the amount of developed land surrounding a marsh (Bertness et al. 2002), and both Sentinel Sites have been continuously surrounded by natural habitats since at least the early twentieth century. Inputs from atmospheric sources are also stable (Schmidt 2014), while estuarine sources have likely declined due to improvements to Narragansett Bay's largest wastewater treatment facilities, which have resulted in an overall 30 % reduction in dissolved inorganic nitrogen inputs into the Bay since the early 2000s (Krumholz 2012). We did not track in situ nitrogen concentrations as part of this study, but monthly nutrient monitoring in Nag Marsh as part of the NBNERR System-wide Monitoring Program indicates that nitrogen concentrations have remained low in this marsh since at least 2002 (Durant and Raposa 2011). Clearly, the ongoing decline in *S. patens* shown here is not associated with increasing nitrogen inputs to the marshes. This agrees well with Donnelly and Bertness (2001) who also rejected nutrients as a driver for *S. alterniflora* encroachment into the salt meadow in a nearby Prudence Island marsh.

Instead, evidence suggests that the changes documented here are primarily due to accelerating sea level rise and recent extreme high tide events (Goddard et al. 2015; Sweet and Zervas 2011). Elevations where *S. patens* declined were significantly lower than elevations where it was stable and *S. patens* declined faster in Coggeshall Marsh, where marsh platform elevations are significantly lower and tidal inundation is higher relative to Nag Marsh. This is in agreement with Warren and Niering (1993), who also documented vegetation changes at low marsh elevations that are first exposed to rising sea levels. We expect these changes to continue because complementary research in Nag Marsh shows that a deficit now exists between salt meadow accretion rates and recent rates of sea level rise (Carey et al. 2015) and that current inundation patterns are limiting the productivity of common Narragansett Bay salt marsh taxa including *S. patens* (Watson et al. 2014, 2015). Taken together, this suggests that sea level rise is driving the loss of *S. patens* salt meadow in our Sentinel Sites marshes, which agrees well with studies in other southern New England marshes that have experienced similar changes (Donnelly and Bertness 2001; Warren and Niering 1993).

In our marshes, there were two general mechanisms by which *S. patens* salt meadow was replaced by *S. alterniflora*. The first is the relatively slow landward encroachment of *S. alterniflora* into intact *S. patens* salt meadow that is likely driven by long-term sea level rise. The second is the more rapid, yet episodic, replacement via the formation and recovery of dieback patches. In our Sentinel Sites monitoring plots, dieback areas were largely absent from 2000 to 2008, but a multi-year event occurred in both marshes beginning around 2009 and peaking in 2011, which coincided with the dramatic increase in sea levels throughout the region in 2009–2010 (Goddard et al. 2015). The bare patches produced by dieback recovered within 2–3 years almost exclusively with stunted *S. alterniflora* (often preceded by a 1-year initial pulse of *Salicornia* spp.), even in areas that were originally equal mixtures of *S. alterniflora* and *S. patens*. This agrees well with Hartman (1988) who found that *S. alterniflora* was the only species that recolonized natural bare patches in a Massachusetts marsh, even in areas of high marsh that were originally dominated exclusively by *S. patens* and *D. spicata*. In the absence of inundation stress, *S. alterniflora* is competitively inferior to *S. patens* in high marsh habitats (Bertness 1991) but readily colonizes and becomes established in the high marsh when *S. patens* is absent (i.e., in bare patches).

Our study was not designed to identify the specific cause(s) that result in dieback formation, but evidence suggests that soil waterlogging resulting from excessive tidal flooding was a primary factor. We did not observe any evidence of crab herbivory (Smith et al. 2012) or wrack formation prior to the onset of dieback on the mid-marsh platform. We documented dieback that was likely caused by crab herbivory, but this was along creek edges, while wrack was generally found at higher elevations on the marsh near the *I. frutescens* shrub zone. We did, however, observe persistent soil waterlogging in many areas prior to the onset of dieback and suspect that this was related to anomalously high tide levels. In fact, we found that 85 % of the interannual variability in bare ground resulting from dieback was explained by changes in annual MHW levels during this study. In addition, total annual precipitation in Rhode Island was 22 % higher on average when dieback was ongoing compared to when it was mostly absent or recovering (NOAA NCDC 2014). Although a number of factors can cause marsh vegetation dieback (Alber et al. 2008), we believe that the multi-year pulse documented in our marshes was a direct response to extreme high tides and elevated precipitation levels that led to prolonged periods of soil waterlogging. If sea level rise continues to accelerate as expected (Church et al. 2013), and if climate change increases the annual amount of precipitation in New England (NECIA 2006), we can expect marsh dieback to persist or even accelerate, which in turn should accelerate the conversion of intact salt meadow habitat into monocultures of stunted *S. alterniflora*.

Impaired drainage due to severely clogged mosquito ditches may have compounded the effects of waterlogging on dieback and the loss of *S. patens*. The current conditions in our Sentinel Sites marshes (and most Rhode Island marshes) are similar to those observed in Connecticut approximately a century ago (Britton 1912). Then, linear ditches that were dug for historic salt hay farming had subsequently filled with sediment and vegetation; this was the suspected cause of shallow depression formation and standing water on the marsh surface, dying vegetation, and nuisance mosquito populations. The results after excavating these clogged ditches were gains in marsh elevation, firmer soils, loss of standing open water, and high marsh expansion. Similarly, Vincent et al. (2013) showed that blocking ditches (ditch plugging) results in high water levels on the marsh surface, saturated and weaker soils, and marsh subsidence. In most RI marshes, mosquito ditches have not been maintained for at least 60 years, and many of these ditches have now filled with sediment and re-vegetated. It is therefore possible that these clogged ditches are inhibiting drainage and exacerbating the current increases in *S. patens* loss and dieback. In response, multiple experiments are now underway in Rhode Island to excavate clogged mosquito ditches and monitor the responses of marsh hydrology, elevation, and vegetation. The findings of these projects should augment encouraging results seen from other areas (Rochlin et al. 2012) and help determine if management actions that restore drainage will enhance marsh resiliency against future sea level rise.

The composition of Coggeshall and Nag marshes is generally consistent with other marshes at similar latitudes within the Narragansett Bay region during the same time period based on RISMA. The two Sentinel Sites marshes can therefore be considered as representative of conditions within mid-Narragansett Bay, which in turn suggests that the trends we observed are probably occurring elsewhere in Narragansett Bay (e.g., in nearby Nag Pond Marsh; Donnelly and Bertness 2001). In fact, we believe that many salt marshes lower in the Bay have already largely converted to *S. alterniflora* because they are more susceptible to sea level rise relative to marshes in upper Narragansett Bay. Lower Bay vegetated marsh platforms slope less steeply toward the estuary (and therefore have less elevation capital; Cahoon and Guntenspergen 2010; Wigand et al. 2003) and are subject to higher relative rates of sea level rise compared to upper Bay marshes (long-term rates of 2.73 and 2.24 mm year^{-1} at the Newport and Providence tide stations, respectively; NOAA CO-OPS 2014). We can then also predict that the changes currently occurring in the mid-Bay Sentinel Sites marshes will eventually migrate northward into salt marshes in the upper Bay if sea level continues to rise as expected. In short, our monitoring and assessment data are timely in that they likely demonstrate what recently happened to vegetation communities in lower Narragansett Bay salt marshes and provide a preview of what might soon happen to marshes closer to the head of the bay. This should be useful for managers and restoration practitioners who might need to prioritize between trying to restore salt marshes that are already degraded or protecting marshes that remain less impacted by sea level rise.

The functional ability of New England salt marshes to provide habitat for fish and wildlife will be affected (in beneficial as well as deleterious ways) if accelerating sea level rise continues to induce excessive soil waterlogging, vegetation shifts, and dieback. For example, foraging habitat for resident nekton such as the mummichog (*Fundulus heteroclitus*) should improve with increased access to flooded marsh surface habitats (McIvor and Rozas 1996; Weisberg and Lotrich 1982) and declines in vegetation stem densities associated with the shift from *S. patens* to *S. alterniflora*, prolonged inundation, and dieback (Vince et al. 1976; Voss et al. 2013; Wigand et al. 2003). Nursery and overwintering habitat may also improve if dieback and waterlogging result in more shallow open water habitats on the marsh surface (Raposa 2003; Talbot and Able 1984). Indeed, short-term benefits for nekton have already been documented in degraded and submerging salt marshes in the Gulf of Mexico (Rozas and Reed 1993; Zimmerman and Minello 1984). The short-term benefits to nekton should in turn translate into improved foraging opportunities for wading birds that prey on these species on flooded marsh surfaces and in shallow open water habitats (Erwin et al. 2009; Raposa et al. 2009). However, birds use *S. patens* salt meadow more frequently than any other salt marsh habitat in southern New England marshes (McKinney and Wigand 2006), and its loss will likely result in declines in species of concern such as the seaside sparrow (*Ammodramus maritimus*) and the saltmarsh sparrow (*Ammodramus caudacutus*) (Bayard and Elphick 2011; Erwin et al. 2009). Of course, any short-term enhancement of the functional ability of marshes to provide habitat (or any other ecosystem services) will eventually become lost if sea level rise and its associated impacts to vegetated marshes leads to net losses of marsh area (Craft et al. 2009; Kearney et al. 2002).

In summary, our data confirm that the loss of *S. patens* salt meadow is ongoing in Narragansett Bay and that recent extreme high water levels may have accelerated its replacement by low marsh *S. alterniflora*. This adds further evidence that these changes are occurring at multiple marshes across southern New England and that many marshes are on a trajectory of *S. alterniflora* dominance. Identifying the stressors that drive these patterns was beyond the scope of this study, but evidence suggests that impacts were primarily from a combination of anomalously high water levels superimposed on long-term sea level rise potentially coupled with poor drainage. To this end, the concomitant impacts of sea level rise and impaired drainage on marsh structure and function need to be isolated and quantified. There is also a need to use historical aerial photography (similar to Smith 2009) and to expand long-term monitoring efforts to quantify past and future rates of vegetation change and dieback throughout New England. Ultimately, however, scientists and managers will need to take action to help maintain marsh resiliency in the face of sea level rise and other anthropogenic stressors (Wigand et al. 2015). Previous efforts to increase marsh elevations through sediment slurry additions (Stagg and Mendelssohn 2011) and to excavate clogged drainage channels to alleviate waterlogging (Rochlin et al. 2012) offer promise for helping marshes deal with accelerating sea level rise. Both of these approaches strive to reduce the frequency and duration of marsh surface inundation. Paradoxically, curtailing current *P. australis* removal and management practices might also help because *Phragmites* invasion can dramatically increase marsh accretion rates (Rooth and Stevenson 2000). Here we provide further evidence that southern New England salt marshes are changing rapidly and dramatically in response to accelerating sea level rise. In the absence of targeted marsh adaptation projects, we expect further deterioration of southern New England marshes as sea level rise continues to accelerate.

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