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Wetland loss patterns and inundation-productivity relationships prognosticate widespread salt for southern New England

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Abstract

Tidal salt marsh is a key defense against, yet is especially vulnerable to, the effects of accelerated sea level rise. To determine whether salt marshes in southern New England will be stable given increasing inundation over the coming decades, we examined current loss patterns, inundation-productivity feedbacks, and sustaining processes. A multi-decadal analysis of salt marsh aerial extent using historic imagery and maps revealed that salt marsh vegetation loss is both widespread, and accelerating, with vegetation loss rates over the past four decades summing to 17.3%. Seaward retreat of the marsh edge, widening and headward expansion of tidal channel networks, loss of marsh islands, and the development and enlargement of interior depressions found on the marsh platform contributed to vegetation loss. Inundation due to sea level rise is strongly suggested as a primary driver: vegetation loss rates were significantly negatively correlated with marsh elevation (r^2=0.96; p=0.0038), with marshes situated below mean high water (MHW) experiencing greater declines than marshes sitting well above MHW. Growth experiments with *Spartina alterniflora*, the Atlantic salt marsh ecosystem dominant, across a range of elevations and inundation regimes further established that greater inundation decreases belowground biomass production of *Spartina alterniflora* and thus negatively impacts organic matter accumulation. These results suggest that southern New England salt marshes are already experiencing deterioration and fragmentation in response to sea level rise, and may not be stable as tidal flooding increases in the future.

Keywords

climate change; sea level rise; anthropogenic impacts; wetlands; storms; *Spartina alterniflora*; elevation capital

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INTRODUCTION

Worldwide, coastal marshes are recognized as a key transition zone that is critical for buffering the coast from the effects of climate change, and they are also among one of the habitats most vulnerable to the effects of accelerated sea level rise (SLR) (Temmerman et al. 2012). Previous studies have demonstrated the strong role that coastal marshes play in protecting the shoreline against storm-induced erosion and flooding (Möller et al. 1999; Wamsley et al. 2010; Gedan et al. 2011), yet global forecasts of coastal wetland response to SLR predict a marsh loss rate of 20–60% during this century (Titus 1988; Nicholls et al. 2007). Studies are greatly needed to assess the current status and deterioration trajectory of salt marshes at local spatial scales that are relevant for coastal governance. Studies conducted at local scales may facilitate regulatory decisions or interventions that will lead to wetland preservation in place or facilitate the upland migration of these valuable habitats.

Previous assessments of marsh vulnerability to SLR have used predictive models based on future SLR rate estimates from the Intergovernmental Panel on Climate Change (e.g. Stocker et al. 2013) and other sources (Rahmstorf 2007). These studies have suggested that salt marshes are vulnerable to SLR only under the more extreme scenarios. For example, Morris and others found that at North Inlet, South Carolina, elevation-productivity and elevation-deposition feedbacks were expected to maintain marsh elevations under SLR scenarios including, and even exceeding, 1 cm yr^{-1} (Morris et al. 2002; Morris 2007). Similarly, scientists from both Northwestern Europe and California have used empirically verified elevation-deposition relationships to predict sustainability of even low-elevation marshes over the next century (Orr et al. 2003; Temmerman et al. 2004; Strahlberg et al. 2011; Fagherazzi et al. 2012).

In contrast, where historical data exists, reports show that significant marsh drowning has already occurred. A multi-decadal tidal wetlands assessment conducted using LANDSAT imagery for the Chesapeake and Delaware Bay regions established that more than half of all tidal wetlands were in a degraded condition, and that substantial declines have occurred in vegetated area over past decades (Kearney et al. 2002). Using higher resolution imagery, salt marsh deterioration has also been noted as being widespread on Cape Cod, MA (Smith 2009), in New York City (Hartig et al. 2002), and along the Long Island shore (Bowman 2015). Outside of the eastern U.S., high rates of marsh drowning have also been found for the Mississippi Delta region, in Europe and elsewhere (Day et al. 1998; Day et al. 2000; Hughes and Paramor 2004; Van Dyke and Wasson 2005). Studies of marsh change over time suggest that vulnerability and loss are quite widespread.

Here, we report on a series of assessments conducted in the Northeastern United States, where studies have shown ongoing salt marsh deterioration is due to multiple and interacting causes (Donnelly and Bertness 2001; Hartig et al. 2002; Smith 2009; Wigand et al. 2009; Deegan et al. 2012). Also, SLR rates have increased over the last century in this region (Donnelly et al. 2004), particularly over the past two decades due to climate-related changes in the speed and width of the Gulf Stream and other interacting dynamic processes (Sallenger et al. 2012; Ezer and Atkinson 2014).

Our principal objectives were first, to establish contemporary loss rates for coastal wetlands in Rhode Island, and second, to evaluate the role of inundation in current marsh loss patterns, with the ultimate goal of determining whether salt marshes in southern New England will be stable against inundation increases occurring over the coming decades. In our study, changes in vegetated marsh extent were measured through interpretation of historic and contemporary maps and aerial imagery. Plant growth experiments were used to establish a relationship between inundation and biomass production, to determine whether increased inundation would result in enhanced or reduced rates of organic matter accumulation and peat formation. Elevation and water level measures were performed to compare experimental treatments with field conditions, and to identify the role of inundation in current marsh loss patterns. Finally, sedimentary analyses were performed to estimate the relative contributions of potential sediment sources to marsh accumulation.

Study Sites

While less expansive than their mid-Atlantic and southeastern counterparts (Roman et al. 2000), tidal marshes of Long Island and southern New England abut a significant portion of the shoreline, and provide valuable services to human and wildlife populations, including nursery and foraging habitat for native species, recreational opportunities, and nutrient removal and flood abatement services (Bromberg and Bertness 2005). A total of thirty-six discrete areas of natural salt marsh in Rhode Island were studied, comprising a wide selection of geomorphic settings that included backbarrier and fringing marshes found along Narragansett Bay, coastal salt ponds, and estuarine rivers and creeks (Fig. 1; Table 1). Sites were chosen to encompass all twenty of the largest areas of contiguous coastal marshlands in Rhode Island, and also included sites where previous assessments have been conducted (Nixon and Oviatt 1973; Wigand et al. 1999; Wigand et al. 2010). Tides in this area are semi-diurnal (0.05-1.5 m; Roman et al. 2000) and variability in tidal range is a function of limited tidal exchange through coastal inlets and tidal wave resonance in upper Narragansett Bay.

Rhode Island marshes typically lack surface freshwater inputs, and are therefore relatively homogenous with respect to salinity, with typical soil and water salinities of 20-30 % (Raposa 2009). Characteristic vegetation is comprised of a narrow band of tall form *Spartina alterniflora* adjacent to the shoreline and tidal channels, with the higher portion of the salt marsh inhabited by a combination of short form *Spartina alterniflora, Distichlis spicata, Spartina patens*, and *Juncus gerardi* (Nixon 1982; Roman et al. 2000). Following the nomenclature of Haines (2011), typical forbs include *Salicornia depressa, Plantago maritima, Triglochin maritima*, and *Limonium carolinianum.* Where natural hydrology is intact, marshes are drained by tidal channels with a slightly bulbous planform, and feature frequent ponds and pannes (Adamowicz and Roman 2005). More commonly, natural drainage has been supplanted by recirculation ditches constructed to control the breeding of salt-water mosquitos (Kennish 2001). Marsh substrates consist primarily of peat soils (~≥50% organic matter; Burton and Hodgson 1987), as regional sediment supplies and soil temperatures are naturally low, and because plants are adapted for over-wintering.

METHODS

In our study, changes in vegetated marsh extent were measured through interpretation of historic and contemporary maps and aerial imagery. Plant growth experiments were used to establish a relationship between flooding and biomass production, to determine whether increased inundation would result in enhanced or reduced rates of organic matter production. In contrast with marshes of the U.S. Southeast and Pacific coasts, marshes in the southern New England region are typically comprised of peat soils, thus the inundation-biomass production relationship is key for predicting responses to increased flooding. Elevation and water level measures were additionally performed to compare experimental treatments with field conditions, and to identify the role of inundation in current marsh loss patterns. Finally, sedimentary analyses were performed to estimate the relative contributions of potential sediment sources to marsh accumulation.

Site characterization employed a nested study design. We measured elevation and recent rates of marsh vegetation loss (ca. 1972 to 2011) at a total of thirty-six Rhode Island coastal marshes (Fig. 1; Table 1) in order to identify marsh loss patterns and to identify potential relationships with between vegetation mortality and flooding. More detailed information was gathered for five focus sites (Mary Donovan Marsh, Narrow River, Nag Marsh, Brushneck Cove, and Hundred Acre Cove), where water levels were measured for several months, soil composition was analyzed, and a multi-decadal marsh vegetation loss assessment was performed using a combination of historic aerial imagery and maps spanning the nineteenth, twentieth, and twenty-first century. Field mesocosm experiments at three sites (Mary Donovan Marsh, Nag Marsh, and Hundred Acre Cove) were used to parameterize the relationship between elevation and productivity for *Spartina alterniflora*, the ecosystem dominant and most common species.

Field Mesocosms

Minerogenic marshes rely primarily on the build-up of inorganic sediment to aggrade (Temmerman et al. 2004), and inundation-deposition feedbacks allow marshes to respond to increased inundation through increased deposition (Orr et al. 2003). As a result, for minerogenic marshes, survival under a regime of rapid inundation increase is a function of sediment supply from local and watershed erosion (Kirwan et al. 2010). In contrast, for organic-rich marsh soils in southern New England, belowground productivity primarily determines whether marshes faced with accelerated SLR will aggrade or submerge. For organic marsh soils, soil volume is typically 90% organic material and associated porosity, suggesting that the supply of mineral sediment is secondary to peat formation (Turner et al. 2000). While some studies have suggested that biomass production will increase with inundation (Morris et al. 2002; Morris et al. 2013), other studies have come to the opposite conclusion (Kirwan and Guntenspergen 2012; Voss et al. 2013), and none have been conducted in the southern New England region.

To determine whether inundation increases will result in enhanced or reduced belowground biomass production, elevation-productivity relationships for *Spartina alterniflora* were determined using field mesocosms (‘marsh organs’) designed to isolate the effects of inundation on plant productivity (Kirwan and Guntenspergen 2012; Morris et al. 2013; Voss et al. 2013). Additionally, one marsh organ was directly adjacent to agricultural fields (Mary Donovan); as a result, nutrient availability at this site was much higher than the other two locations.

Four rows of replicate 15-cm diameter plant pots were filled with native marsh sediments collected from ponds, and planted with sods of *S. alterniflora* collected early in the growing season (apart from one site where nursery stock was utilized) at three sites in Narragansett Bay. Pot row elevations differed by 20 cm, and nylon mesh squares held the space available for belowground biomass growth (i.e., 30cm) constant. At the end of the growing season, dry weights were obtained for plant roots and rhizomes were sieved from plant pot muds using a 0.5 mm sieve, washed, dried to constant weight, and weighed.

Porewater was collected from each pot monthly for analysis of salinity, pH, hydrogen sulfide, and dissolved nutrient concentrations using a permanently installed micropiezometer outfitted with a 0.5 μm nylon mesh frit. To clear the piezometer and collect fresh porewater, the maximum volume of water possibly contained by the piezometer (5mL) was discarded, and fresh sample obtained. These pore-waters were preserved in the field for hydrogen sulfide analysis. Samples were analyzed for pH using a calibrated Accumet portable pH meter, and for salinity using a refractometer. Samples were then preserved for nutrient analysis. Porewater samples were analyzed for hydrogen sulfide using a Genesys 2 spectrophotometer (Cline 1969; Strickland and Parson 1972), and for ammonium, phosphate, nitrite and nitrate using an Astoria Pacific A2 micro-segmented flow autoanalyzer (U.S. EPA methods 350.1, 353.2, 365.2).

At one site (Nag Marsh), core insert tubes were flooded and scanned post-harvest using a GE medical systems computed tomography (CT) scanner to visualize belowground biomass structure and volume of coarse roots and rhizomes (Davey et al. 2011; Wigand et al. 2014). Using calibration rods of water, silica, and air, voxel values were converted to Houndsfield density units, and known density windows were used to visualize three-dimensional belowground biomass structure.

Digitization of Marsh Vegetation Extent

To establish trends in marsh stability, marsh vegetation boundaries at study sites were photointerpreted using aerial imagery from 1972 and 2011 (ArcGIS, version 10). At one study site, located in Massachusetts, but part of the Narragansett Bay Estuary, 1974 and 2010 air photos were used. The 1972 imagery was georeferenced utilizing the 2011 orthoimagery as a base map, employing at least 15 control points per scene. The majority of these control points were hard structures, such as building or bridge edges, but a number of soft control points (landscape features) were also utilized to reduce distortion within marsh scenes. To ensure that differences found in the overlay analysis were not a function of image quality, both 2011 and 1972 imagery were resampled to a 0.75-m pixel dimension, and rendered in grayscale using a red-green-blue to hue-saturation-intensity conversion, with the intensity component interpreted (Van Dyke and Wasson 2005). Vegetation boundaries were digitized manually following previous studies (Civco et al. 1986; Halls and Kraatz 2006; McLoughlin et al. 2015), and the vegetated area for each site was calculated in ArcGIS using spatial analysis tools (Table 1). While this study was retrospective, and it was not possible to ground-truth vegetation classification during the time of image collection, we were able to obtain historical vegetation survey data from 84 plots collected in summer 2010 from the Narragansett Bay National Estuarine Research Reserve. This data was collected within 8 months of image acquisition, and was used to estimate classification accuracy. To estimate uncertainty in change rates resulting from image misalignment, we measured the area of 10 hard structures that were judged to have not changed significantly over time (e.g., buildings, parking lots, city blocks) for each imagery pair, and used the normalized root mean square deviation as an uncertainty estimate.

Additional qualitative and quantitative analyses of historic imagery and maps were conducted at focus sites to provide a longer-term perspective on recent changes in marsh vegetation extent. For these analysis, spatially referenced imagery available via mapserver from the Rhode Island State Geographic Information System (RIGIS) Database was utilized (Table 2). In addition to aerial imagery, background data on habitat change was provided by comparing digitized and georeferenced U.S. Coast Survey maps from the mid 1800s to more recent imagery. In addition, changes through time in channel width were measured. For each focus site, channel widths were measured at 20 randomly selected points.

Assessment of Elevation Capital

Elevation capital is a term that has been used to describe the elevation of a marsh relative to its potential growth range (Cahoon and Guntenspergen 2010). Coastal wetlands that exist near the upper limit of their potential growth range are said to possess elevation capital, while wetlands that are near the lower limit of their potential growth range are said to be lacking in elevation capital. To measure marsh elevation capital, we measured marsh elevations relative to elevation and water level datums. Topographic surveys of salt marsh elevations were performed along transects with a rotary laser. Topographic surveys included the establishment and survey of temporary benchmarks (1–4 per site), which consisted of marks atop rebar or PVC driven into the marsh to a depth of 0.5–1 m. Orthometric heights were measured for each temporary benchmark using static post-processed kinematic surveys utilizing a survey grade GPS receiver (Trimble 4700; accuracy <5 cm); these measures allowed calculation of marsh orthometric height distribution. On Prudence Island, surveys of Nag and Coggeshall Marshes were conducted using the established elevation control network, adjusted for consistency with the NAVD88 Geoid12A datum. We used median marsh elevation as an indicator of marsh platflorm height to reduce potential skew caused by channel and shore edges, which were up to 1.5 m lower than the marsh platform proper.

Water levels in tidal channels at four sites were measured with Solinst Model 3001 Levelogger Edge at five-minute intervals for a five-month period, and barometrically compensated using data logged with a Solonist Barrologger Edge. At Nag Marsh, water levels, temperature, and conductivity were monitored using an Aqua Troll 200 instrument. Tidal datums for each marsh (relative to the National Tidal Datum Epoch) were computed using the modified-range-ratio method, with Newport or Providence, RI as control stations (NOAA 2003). This method is generally associated with accuracy on the order of 2–3 cm (Swanson 1974).

Additional Site Assessments

To characterize soil properties among focus sites, three 50-cm sediment cores were collected from each of five marsh sites, and profiled at 10-cm intervals for wet and dry density, sediment organic content, and particle size distribution. Density and organic content measures were made by subsampling a known volume of sediment, drying it to constant weight, re-weighing, ashing the sub-sample at 550°C for four hours, and measuring a final weight (Heiri et al. 2001). Soil composition was analyzed on the basis of volume (Nyman et al. 1990). Soil subsamples were also analyzed for particle size distribution, as grain size varied between sites. Soil texture plays a key role in mediating the response of a coastal wetland to both erosive stresses (i.e., cohesive vs. noncohesive sediments) and the development of elevated porewater sulfide concentrations associated with poor water quality or inundated conditions. Multiple aliquots of heated hydrogen peroxide were used to oxidize organic material, and sodium hexametaphosphate was used as a dispersant (Gray et al. 2010) prior to introduction into a Malvern Mastersizer 2000S laser granulometer.

RESULTS

Field Mesocosms

Productivity-elevation relationships show that *Spartina alterniflora* belowground growth varied as a function of orthometric height (Fig. 2). Plant growth was least robust at the lowest elevation and greatest inundation times. Highest biomass values were found at the highest elevations and lowest inundation times, both within and across sites. This relationship is emphasized by analysis of CT scans of core inserts visualizing abundance of coarse roots and rhizomes of *S. alterniflora* grown across a range of elevations (Fig. 3). By calculating pot-specific inundation times using water level data, we find that optimum growth corresponded to elevations above site-specific mean high water (MHW) values. No substantial differences were found in the elevation-productivity relationships observed for nursery stock vs. field collected sods with respect to slope or intercept (ANCOVA; F=1.53, p=0.29; F=2.75, p=0.12). Aboveground biomass results were not analyzed because harvest dates occurred in late fall to allow for biomass translocation from senescing leaves to belowground tissue; at the time of harvest, material was in a partially decomposed state.

Porewater pH, salinity, and DIN varied across treatments. Porewater salinity covaried somewhat with elevation, suggesting concentration through evapotranspiration. At the highest elevations, pH was depressed, which is consistent with a transition from estuarine to upland soils, as southern New England glacial outwash soils are naturally acidic. Slightly higher DIN values were found at lower elevations, either through exposure to high water column nitrogen loads, or because plants growing under inundated conditions were unable to assimilate available DIN as efficiently as those growing under less inundated conditions (Koch et al. 1990). At Hundred Acre Cove, in particular, porewater suggested more brackish conditions than the other two sites, with lower salinity, pH, hydrogen sulfide concentration, and DIN relative to the other marshes (Fig. 4).

Porewater hydrogen sulfide concentration, dissolved inorganic nitrogen (DIN), salinity, and pH also varied with elevation and across sites (Fig. 4). A relationship between pot elevation and porewater hydrogen sulfide concentrations was found: lower elevation mesocosms (≤30 cm NAVD88) were consistently associated with porewater hydrogen sulfide concentrations known to produce detrimental impacts on marsh macrophytes (>1mM or 1,000 μM; Koch 1990). High elevation mesocosm pots (≥60 cm NAVD88) were consistently associated with low sulfide concentrations (<0.1 mM or 100 μM). Plotting sulfide concentrations as a function of DIN values also suggest a potential negative effect of high nitrogen loads on sulfide concentrations (Fig. 5). For relatively low porewater nitrogen values (<100 μM DIN), little covariability was apparent between porewater DIN and sulfides, but in this study the highest hydrogen sulfide values were found only where porewater nitrogen concentrations could be considered elevated (>500 μM DIN).

Changes in Wetland Vegetation Extent

Digitization of salt marsh aerial extent from historic imagery shows that fragmentation is common in the marshes surveyed. The aerial extent of vegetated salt marsh declined 17.3%, with values for individual marshes ranging from a gain of 7.4% to a decline of 40.8% (Table 1). Declines were evident at 35 of 36 sites, with only one site gaining in marsh aerial extent. Uncertainty estimates resulting from image misalignment ranged from 1.5% to 6.7% (Table 1), and there was no evidence either year had a significant bias towards a larger area based on a sign test (1972 polygons were larger 49.5% of the time in test polygons constructed by outlining hard structures; two-tailed probability = 0.86). Based on the 2010 historical data, classification accuracy was 91.5%. Differences in the position of marsh shore edges and the upland border (defined by the presence of *Iva)* in transect and digitization data was never more than 1 pixel (0.75m), however accurately mapping dieback patches on the marsh platform using panchromatic imagery proved more challenging. Patch locations were correct, but the extents mapped were generally different than observed in field surveys. There was a slight positive bias towards dieback mapping using aerial imagery with 10.9% of the historical plots mapped as unvegetated using 2011 aerial imagery vs. 8.5% mapped by field surveys in 2010.

We found that fragmentation fell into four general categories: shoreline erosion, loss of marsh in the bay head region of backbarrier lagoons and estuaries, widening and headward erosion of tidal channels, and the development and expansion of interior depressions or ponds (Fig. 6).

Marsh geomorphic setting appears to exert a strong control on multi-decadal changes in vegetated marsh area at focus sites. For the backbarrier marshes profiled (Nag Marsh and Mary Donovan Marsh), the extent of marsh vegetation expanded from 1939 through 1985, and contracted from 1985 to 2011 (Fig. 7). For fringing marshes, however, vegetation loss appears gradual, and approximately linear. Extending this analysis to historic U.S. Coast Survey maps dating to the 1860s (Fig. 7; Table 2) shows that declines in wetland vegetation have been occurring since the 1860s.

At the focus sites, tidal creeks expanded by 29% between 1939 and 2011 at an approximately linear rate of change (Fig. 7A). The most dramatic change in channel width was found at Mary Donovan Marsh, with other marsh sites generally experiencing some stabilization in channel width during the late twentieth century. The exception to this overall pattern, however, is Nag Marsh, where channels have narrowed between 1939 and 2011, presumably related to the shift to a more restricted inlet and the abandonment of mosquito ditch maintenance.

Inundation due to SLR is strongly indicated as a driver of vegetation loss. Plotting vegetation loss as a function of median marsh height relative to MHW (NTDE) (Fig. 8; $r2=0.96$; $p=0.0038$), indicates that elevation, as a proxy for inundation, accounts for 96% of the variability in loss rates. Plotting site vegetation loss between 1972 and 2011 as a function of median orthometric height also indicates a significant relationship between elevation and loss rate (Fig. 8; $r2=0.20$; $p=0.0070$), although less variability is explained. These results suggest that higher loss rates are being found for lower elevation (more inundated) marshes, and lower loss rates are being found for higher elevation (less inundated) marshes.

Elevation Capital

Elevation surveys established marsh platform heights relative to the vertical datum NAVD 88 (computed using GEOID12A), and relative to MHW using data loggers. Marsh elevations ranged from −0.34 to 0.96 m relative to the NAVD88 datum, with the most common value in the 0.5 m range, and with an asymmetric (negatively skewed) bimodal distribution (Fig. 2B). Elevation also tended to covary with the elevation of MHW. Among focus sites, the location with the highest MHW elevation (Hundred Acre Cove; 2012 MHW= 0.74m) had the highest median orthometric height (0.59 m NAVD88), while the site with the most restricted high tides (Narrow River; 2012 MHW=0.35 m NAVD88) had the lowest median elevation at 0.34 m NAVD88. In addition, for marshes where elevations were measured (although water levels were not) the lowest elevations were found for marshes fringing Rhode Island’s coastal ponds (Table 1), where the tidal range is <0.40 m (Lee and Olson 1985).

Comparison of our field mesocosm data with the range of elevations present in Rhode Island salt marshes suggests that increased inundation will result in reduced belowground biomass production. Across the full range of elevations tested in field mesocosms, lower elevations were associated with reduced production. At an elevation of approximately 0.9 m NAVD88, end of season belowground biomass was greater than 5,000 g m^{-2}, while at an elevation of approximately 0m NAVD88, end of season biomass was only 1,000 g m^{-2}. Thus, we conclude that increased inundation will decrease belowground biomass production for salt marshes in this region.

Soil Composition

Marsh sediment profiles analyzed for organic content showed variability in dependence on organic vs. mineral aggradation (Table 3). At Mary Donovan Marsh, where marsh soil was most mineral, sediment organic content was typically 20% and bulk density 0.5 g cc^{-1}, while at Narrow River, where soils were most organic, sediment organic content was 50%, and sediment bulk density ~0.2 g cc^{-1}. At all sites, soils displayed a tendency to decline in organic matter with depth, reflecting peat decomposition. Soil organic content was highest at Nag Marsh, a backbarrier marsh, and Narrow River, an estuarine fringing marsh, suggesting that geomorphic structure does not exert a principal control on soil organic matter. However, both backbarrier marsh soils were found to be consistently higher in sand content than was found for fringing marshes. Soil volume of organic material and associated pore space ranged from 84 to 96%.

DISCUSSION

Marsh Loss Rates

This study has documented significant, although not extreme, shifts in the extent of emergent vegetation over the past four decades for Rhode Island coastal marshes. With the exception of one site, all marshes analyzed have seen declines in the extent of wetland vegetation, with an overall loss rate of 17.3% and rates for individual marshes ranging from 1.6% to 40.8%. Loss of vegetation has occurred through multiple mechanisms, including shoreline erosion, loss of marsh in the bay head region of backbarrier lagoons and estuaries, and widening and headward erosion of tidal channels. In addition, marshes have seen the formation and expansion of interior ponds in areas of poor drainage, behind blocked ditches, and in the center point of grid-ditched marsh islands. While not necessarily detected by imagery analysis, many marshes also contain expansive areas of fragmented vegetation (hummocks), and terraced creek edges where channel integrity has been compromised by mussel mortality, herbivory, crab burrows. A comparison with previous studies conducted for Delaware and Chesapeake Bay (Kearney et al. 2002), New York City (Hartig et al. 2002), Long Island (Browne 2011; Bowman 2015), and Cape Cod (Smith 2009) suggests that the patterns of marsh vegetation loss described here are part of a larger regional occurrence, rather than associated with a particular location or anthropogenic stressor, such as poor water quality (Deegan et al. 2012), inlet dredging and associated tidal range shifts (Swanson and Wilson 2008), ditching (Crain et al. 2009), or overfishing concentrated around boat launches (Bertness et al. 2014).

For marshes analyzed in Rhode Island, rates of vegetation loss for the past four decades have ranged from minimal (<2%) to substantial (40%). This degree of vegetation loss is exceeded by reports for Long Island, including New York City. Loss of marsh vegetation in Jamaica Bay between 1974 and 1999 summed to 36% with a range of 8 to 78% for different regions and islands within the Jamaica Bay system (Hartig et al. 2002). For Long Island, loss of marsh vegetation between 1974 and 1999 through the early 2000s was 11–79% for the northern shore of Long Island, 28 to 43% for the Peconic Bay system, and 18–36% along the South Shore (NYDEC 2012). For Cape Cod, vegetation losses are also apparent for virtually all tidal wetlands for the second half of the twentieth century with loss rates of less than 10% to greater than 30% (Smith 2009). Overall, these values for marsh vegetation loss are considerable; suggesting that the process of marsh deterioration reported here for Rhode Island is part of a larger regional pattern that extends to all of Long Island and southern New England. The results from our study, and from the literature, suggest that the extreme rates of vegetation loss seen at Jamaica Bay, which divides the boroughs of Brooklyn and Queens in New York City and which has been the focus of a multi-million dollar intervention and restoration, are matched or exceeded at other locations in the region.

Marsh Loss Patterns

Shoreline erosion—Marsh vegetation loss has occurred through a combination of processes. For fringing marshes, seaward retreat is a dominant mode of loss. Shoreline erosion is often a function of greater exposure to wind waves resulting from accelerated SLR; previous studies have found that in comparison with other intertidal landforms, marshes have a high sensitivity to wave erosion with SLR due to covariability between elevation and water depth, fetch, and wave-induced bottom shear stresses (Fagherazzi and Wiberg 2009; Mariotti et al. 2010). At no location in our study was progradation of shoreline vegetation observed between 1972 and 2011. Bank slumps, undercutting, and detachments have been and are occurring along marsh edges, and some marsh edges are also becoming pockmarked as vegetation becomes fragmented (Fig. 9A). For backbarrier marshes, shoreline erosion and marsh loss are also occurring where barriers are able to transgress landward (Fig 9C). For the U.S. as a whole, coastal marsh loss resulting from shoreline erosion accounted for less than one percent of total marsh loss in 1980 (Gosselink and Baumann 1980). However, shoreline erosion of wetlands is widespread phenomenon found in Rhode Island (Gordon and Bernd-Cohen 1999; Hapke et al. 2010) and in coastal wetlands, throughout the region. For Cape Cod, 65% of wetlands have experienced shoreline erosion between 1947 and 2005 (Smith 2009), while significant erosion has been noted on the south shore of Long Island (Leatherman and Allen 1985), and accounts for much of the loss of island marshes in the Jamaica Bay system (Hartig et al. 2002). In Delaware Bay, shoreline erosion of marsh vegetation has been recognized for several decades (Phillips 1986; Schwimmer 2001); in Connecticut, marshes on the eastern portion of Long Island Sound eroded between 1983 and 2006 (Stocker et al. 2014).

Expansion of tidal channel networks—Expansion of tidal channel networks is widespread and is occurring in both natural tidal channels and mosquito ditches, although at some sites first order mosquito ditches have narrowed as maintenance has been discontinued (Fig. 9E). Marsh vegetation is lost to creek widening and headward erosion. The pattern of expansion typically involves both channel edges and is exaggerated at bifurcation nodes. Erosion also results in losses of sinuosity and of point bar deposits. This pattern has also been noted for Cape Cod marshes, where expansion of high order channels has been described by Smith (2009).

Amplification of tidal channel networks to accommodate an increased tidal prism has been documented through modeling and flume experiments (Kirwan and Murray 2007; D’Alapaos et al. 2010). Tidal channel network expansion, channel cross-sectional area, and drainage density may respond rapidly to changes in the estuarine tidal prisms resulting from accelerated SLR (Stefanon et al. 2012). This suggests that the loss of marsh vegetation through expansion of tidal channel networks may be driven primarily by hydrodynamic shifts rather than by a specific vulnerability of plant biomass, rooting profile, or soil characteristics. Other studies, however, show that Northeastern marshes may be more vulnerable to erosion where an increased tidal prism co-occurs with other anthropogenic impacts (Deegan et al. 2012; Wigand et al. 2014).

Interior ponding—Interior ponds are common natural features and an ecologically valuable component of New England marsh landscapes (Adamowicz and Roman 2005). It is therefore important to develop a nuanced distinction between stable ponds and those contributing to directional marsh fragmentation. Although work in Plum Island Estuary, Massachusetts, suggests that marsh pools are drained through a natural cyclic process of channel incision (Wilson et al. 2014), results from lower tidal range sites suggest that development and expansion of interior ponds may be the largest contributor to contemporary marsh loss in the Northeast region (Kearney et al. 1988; Hartig et al. 2002; Smith 2009).

Interior ponding occurs at both the heads of first order channels and in areas isolated from tidal creeks (Kearney et al. 1988). Where *S. patens* is experiencing dieback, peat collapse often leads to depressions and ponding (DeLaune et al. 1994). Short form *S. alterniflora* often colonizes such areas, but in some cases soils are waterlogged due to elevation loss, and remain un-vegetated. Ponds deepen and expand where sediment aggradation in ponds falls short of the rate of increase in mean high water, or edges are eroded during storms (Fig. 9F). Different from the Plum Island Estuary, drainage of ponds through channel incision occurred only rarely in our examination of RI aerial imagery (Fig. 9D), perhaps due the reduced tidal range in RI (<1 m vs. 2 m+ in the Gulf of Maine). The loss of drainage associated with mosquito ditching also appears to play a role in the formation of shallow depressions, as such depressions occur where marsh ditches have been blocked by peat. Ponding occurs through a variety of mechanisms, but in all cases, marsh vegetation loss can be significant.

Interactions between marsh loss and drainage patterns—Our study reports that vegetated area in RI backbarrier marshes expanded during the first part of the twentieth century, but declined during the last decades of the twentieth century, resulting in marsh vegetation loss since the 1970s for all but one of the backbarrier marshes studied. Closer examination of barrier features suggests that this pattern is linked to widening of inlets. Where tidal exchange occurs through narrow inlets that traverse coastal barriers, the effective tidal range experienced in a geographic unit can be quite restricted compared with that of adjacent ocean, sound, estuary, or bay. Backbarrier marshes are thus responding to changes in inundation that are related to SLR, but can be buffered or amplified by changes in inlet dimension, which ultimately determine inundation patterns.

Backbarrier estuaries are very sensitive to changes in inlet dimension, the sensitivity of which has important implications for regional marsh stability. Where a backbarrier marsh is low in elevation, inlet widening from increased storm frequency may lead to dramatic increases in inundation that far exceed high rates of SLR, due the rapid increase in tidal range. This phenomenon has contributed to vegetation loss rates of greater than 10% per decade for several locations on Long Island (e.g. Cedar Point, Frost Creek; NYDEC 2012). At Pleasant Bay (Cape Cod, MA) a barrier breach in 1987 caused a 40-cm increase in tidal range, and a resultant shift from high to low marsh vegetation (Smith 2009). This process can also be exacerbated by marsh loss itself, as inlet dimensions are a function of both coastal processes and the tidal prism (Behrens et al. 2009; D'Alpaos et al. 2010). Where marsh erosion results in increased estuary volume and an enlarged tidal prism, marsh loss may fuel a positive feedback progression: marsh loss drives inlet expansion, which drives further marsh loss as wetlands are exposed to both increases in inundation and the erosive power of wind and waves.

Examination of marsh loss patterns in marshes where drainage has been supplanted by recirculation ditches constructed for mosquito control shows that such ditches may increase marsh vulnerability to SLR. Mosquito control practices in southern New England date back to the Civil War as soldiers returning home carried malaria, and disease outbreaks led to widespread draining of swamps and wetlands (Rozsa 1995). Construction of ditches to facilitate drainage intensified during the Works Progress Administration (Corman et al. 2012) and over 90% of New England's tidal wetlands were ditched (Kennish 2001). Regular ditch maintenance was discontinued by most mosquito control agencies between the 1960s and 1980s as agencies began abandoning their policies of ditching and draining marshes. Ditching and draining were associated with declines in resident fish species that prey upon mosquito larvae, and had other detrimental impacts for wildlife (James-Pirri et al. 2012).

In many grid-ditched marshes, the combination of ditch spoils and sediment aggradation and peat build-up concentrated along well-drained ditch edges has resulted in channel-side berms. Poorly drained pools then form in areas where tidal channels are unable to traverse these berms (Fig. 9B). These impacts are most apparent in microtidal locations, such as Rhode Island's coastal salt ponds, where island ponds cannot drain as the hydraulic gradient is insufficient to trigger deep breaches through channel-side berms. In addition to ditches acting as berms and preventing drainage, ditches blocked by peat prevent, rather than facilitate, drainage. In many marshes, natural drainage has been completely supplanted by man-made ditches that in some cases are not self-maintaining, leading to saturated soil conditions and loss of high marsh vegetation.

Role of SLR in Marsh Habitat Loss

While a combination of anthropogenic factors impact marsh vulnerability, we posit the majority of the marsh loss documented here to accelerated SLR. First, marsh vegetation loss over the past four decades is a strong linear function of elevation relative to MHW (Fig. 8; $r2$=0.96). Marshes that are situated below MHW have lost more vegetation than marshes sitting well above MHW, suggesting inundation as a significant driver of loss. Secondly, evidence from mesocosm experiments shows that increased inundation is associated with a loss of productivity, suggesting that SLR is a destabilizing force for southern New England marshes, where aggradation is tied strongly to belowground biomass production and peat formation. Further, consistent vegetation loss among marshes, particularly backbarrier marshes, dates primarily to the late 1970s and early 1980s, when inundation rates increased (Carey et al., this volume). Lastly, forecast models predicting marsh survival as a function of SLR rates and sediment availability (Kirwan et al. 2010; Fagherazzi et al. 2012) indicate that southern New England marshes will not survive under current rates of SLR (>3.5 mm y^{-1}; Boon 2012) in combination with low turbidity (<5 mg L^{-1}; Morton 1972). This implies, at best, that current southern New England marshlands now exist in a state of metastability (Weston 2014). Taken jointly, these observations strongly suggest that increased inundation associated with accelerated SLR is a critical driver of ongoing marsh vegetation loss.

Our current paradigm suggests that salt marshes respond to SLR through three pathways tied to accretion: 1) when coastal submergence exceeds the capacity of a marsh to aggrade vertically, the marsh will drown; 2) if coastal submergence is matched by aggradation, the marsh will persist; or 3) if marsh aggradation exceeds coastal submergence, the marsh will expand (e.g., Orson et al. 1985). Data presented here identifies additional vulnerabilities not addressed by the current paradigm. First, even if marsh aggradation matches submergence rates on average, either a sudden inundation increase caused by barrier breaches, or a slow increase caused by inlet erosion, may cause marshes to drown. This supposition is supported by our observations of marsh loss linked to barrier erosion for Rhode Island marshes and by reports that describe increased inundation and marsh loss resulting from inlet dredging with harbor construction (Van Dyke and Wasson 2005; Smith 2009). Secondly, marshes aggrade differently depending on marsh zone. Where aggradation is concentrated on the banks of tidal channels, large areas of poor drainage may develop in the upper marsh and may persist for decades and enlarge in microtidal marshes. Such an occurrence may be exacerbated by the presence of a poorly developed drainage network (i.e., blocked mosquito ditches). Finally, current observations suggest that marsh edge loss may occur regardless of marsh accretion rates due to wave erosion during storm surges (Fagherazzi and Wiberg 2009).

Additional Factors Contributing to Marsh Loss

Although we ascribe the coastal marsh vegetation loss reported by this study principally to accelerated SLR, additional factors such as herbivory, fungal pathogens, poor water quality and altered hydrology are further noted as important stressors relevant to salt marsh survival. Coastal marshes in Rhode Island now exhibit clear indicators of damage from the purple marsh crab, *Sesarma reticulatum* (Bertness et al. 2014). Evidence of *Sesarma* damage is found along creek edges, where *Sesarma* clipping has removed channel edge vegetation. We interpret this herbivory as an additional factor that weakens tidal channel edges. However, given that *S. reticulatum* are most active when submerged (Seiple and Salmon 1982), increased tidal flooding may be an important factor in recent increases in Sesarma-damaged areas in Rhode Island coastal marshes. Recent studies into wetland dieback patterns have also identified an association with the pathogenic fungal species *Fusarium palustre.* Fungal incidence has been found to be higher in sites undergoing dieback than in healthy marshes (Elmer et al. 2012). Additionally, recent work has noted that drought, fungal infection, and herbivory may in combination lead to plant mortality (Elmer et al. 2014).

Further, data presented here on porewater nitrogen and sulfide concentrations, and supported by previous work (Watson et al. 2014), suggests that high nitrogen loads may fuel the microbially-mediated process of organic matter mineralization and sulfate reduction, leading to the decomposition of marsh peat and accumulations of the phytotoxin hydrogen sulfide in marsh soils. While not conclusive, these results suggest it is possible that SLR and high nitrogen loads may synergistically degrade marshes by cooperatively contributing to elevated hydrogen sulfide concentrations (>3mM; Fig. 4). Indeed, in many estuaries with extreme coastal marsh losses, including Venice Lagoon, Italy (Day et al. 1998), Jamaica Bay, New York City (Hartig et al. 2002) and Elkhorn Slough, Monterey Bay, California (Van Dyke and Wasson 2005), stressors related to increased inundation and nitrogen pollution co-occur. In addition, previously published research has suggested that long term nutrient additions can negatively impact channel banks, where inundation is presumably higher (Deegan et al. 2012), even while increasing growth on the marsh platform proper (Morris et al. 2013). Jointly, these observations suggest that inundation possibly mediates the effects of eutrophication on coastal marshes through redox sensitive biogeochemical reactions; additional studies are needed to confirm and extend these results.

Declines in sediment inputs may also be contributing to marsh deterioration in the Northeastern U.S. (Weston 2014). Studies from Massachusetts, Chesapeake Bay, and California have described marsh progradation in synchrony with settlement era land clearing and erosion (Pasternack et al. 2001; Kirwan et al. 2011; Watson and Byrne 2013). Additionally, studies of water clarity and sediment transport in the region find declines in sediment inputs and water column suspended sediment concentrations over recent decades (Borkman and Smayda 1996; Weston 2014;C. Wigand, unpublished data). However, suspended sediment concentrations in Narragansett Bay 40 years ago were only on the order of 4 mg L^{-1} (Morton 1972), below levels normally associated with long-term marsh stability (Kirwan et al. 2010). The decline of sediment availability from “low” to “very low” undoubtedly exacerbates marsh loss issues, but is unlikely to be the sole driver of coastal wetland fragmentation.

Coastal Marsh Survival Under Future Climate Change Scenarios

The results of this study suggest that wetlands in southern New England are more vulnerable to the impacts of SLR than previously recognized. High rates of SLR due to changes in ocean circulation, in combination with extremely low sediment availability and a low tidal range suggest that threshold rates of SLR, above which marshes will convert to tidal flats, have already been crossed for wetlands in Rhode Island. The negative trend in vegetated wetland habitat extent identified in our study in combination with other recent studies from the U.S. Northeast (Hartig et al. 2002; Smith 2009; Basso et al. 2015; Berry et al. 2015; Cameron Engineering and Associates 2015), and research data that shows an association between high inundation and high loss rates, suggests that wetlands in region are already being impacted by SLR. However, this observation does not necessarily mean that coastal wetland disintegration is inevitable globally. Coastal marsh vulnerability to loss driven by SLR is dependent on very specific local factors, including but not limited to, existing marsh elevation, macrophyte growth range and rooting profile, sediment availability, soil composition, local inundation regime, altered hydrology, and wave climate. We assert that development of a rigorous remote sensing monitoring program is key to detection of changes in coastal wetland habitats, and should be an integral part of coastal wetland monitoring and assessment programs. Coastal wetland assessment programs have typically focused on analyzing adjacent and catchment area land use (e.g., Nestlerode et al. 2014), and the managers of the National Wetlands Inventory classification system caution against using data from different years for identifying trends in wetland extent because classification methods have changed over time (e.g., Tiner et al. 2014). As a model of such a program, we can describe the work currently underway for Rhode Island by the NOAA Coastal Services Center. An informatics method was developed to classify color/near infrared aerial photography and LiDAR using an object-based method and decision rules for object identification (Robinson et al. 2015). Once developed and verified using training sets and ground-truthing, the automated analysis can then be re-run using subsequent years of imagery at minimal expense. With the application of such a monitoring method, site-level vulnerabilities can be identified and conservation strategies appropriately employed; without such data, understanding of marsh habitat change trajectory is largely conjectural.

CONCLUSIONS

A multi-decadal analysis of salt marsh aerial extent conducted using historic and recent aerial imagery suggests that marsh vegetation loss has summed to 17.3% over the past four decades for Rhode Island, USA. Habitat conversion of vegetated marsh platforms to bare tidal flat has been driven by seaward retreat of the marsh edge, widening and headward expansion of tidal channel networks, loss of marsh islands, and the development and enlargement of interior depressions found on the marsh platform. The response of southern New England coastal marshes to additional inundation increases associated with future SLR is intrinsically linked to the ability of salt marshes to respond through increased sediment trapping and enhanced productivity. Across the range of elevations found for Rhode Island coastal marshes, our mesocosm experimental results show a strong positive relationship between elevation and belowground productivity (Fig. 2A; Fig. 3). Considering elevation as a surrogate for inundation, our results suggest that significant productivity declines will accompany increased inundation. While enhanced deposition of mineral sediment will occur with increased flooding, soil composition analysis finds that less than 8% of marsh soil volume for southern New England marshes is composed of mineral sediment (Table 3). Given that soil volume consists primarily of marsh peat, this finding prognosticates a continuation of the widespread marsh loss reported here.

Multi-faceted coastal adaptation strategies are currently being developed to aid in coping with coastal marsh loss as a result of SLR, including a combination of informed coastal planning and a new generation of habitat-focused coastal protection solutions. For instance, SLAMM modeling (Sea Level Affecting Marshes Model) is being used in several northeastern states in combination with newly available coastal LiDAR to better understand where marsh migration opportunities exist. Priority parcels can be targeted for acquisition by land trusts, municipalities, or conservation organizations, and management strategies may also be adapted to facilitate marsh migration. Marsh protection intervention strategies include beneficial use of dredge spoils to build marsh elevation, drainage enhancements, and soft shoreline protection strategies such as sand nourishment along eroding inlets and the installation of living shorelines (Wigand et al., this volume). However, despite the promise of these techniques, cost will probably constrain the implementation of effective protection strategies. Our analyses suggest that it may not be realistic to expect that our U.S. Northeastern coasts will support extensive or widespread coastal wetlands in the future unless proactive steps are taken to reduce greenhouse gas emissions and stabilize global sea levels.

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Glossary

Backbarrier marsh

marsh that has formed on the leeward side of a barrier beach or spit

Elevation relative to MHW

Elevation relative to the average of all the high water heights, smoothed for the impact of waves, observed over the NTDE

Ellipsoidal datum

Ellipsoid datums are based on a geometric model that approximates the earth's surface without topography. NAD83 and WGS84 are ellipsoidal datums

Datum

A fixed reference for elevations determined by geodetic leveling

GEOID12A

An ellipsoid model of the earth's surface that roughly corresponds to sea level and is used as a reference surface. A geoid model is used to calculate orthometric heights from GNSS data, and to convert between ellipsoidal and orthometric height. The model currently used in the U.S. is GEOID12A. Geoid models are updated periodically by the National Geodetic Survey. Orthometric heights calculated using different geoid models can vary substantially

GNSS

Global navigation satellite system

Mean High Water (MHW)

The average of all the high water heights observed over the present NTDE

NAVD88

is the vertical control datum established in 1991 by the minimum-constraint adjustment of the Canadian-Mexican-United States leveling observations

National Tidal Datum Epoch (NTDE)

The specific 19-year period adopted by the National Ocean Service as the official time segment over which tide observations are taken and reduced to obtain mean values (e.g., mean lower low water, etc.) for tidal datums. The present NTDE is 1983 through 2001

Orthometric height

Leveled heights above a fixed elevation datum

Tidal datum

A tidal datum is a standard elevation defined by a certain phase of the tide relative to the NTDE

Tidal height

Elevation relative to a tidal datum, which could include mean lower low water (MLLW), mean tide level (MTL), mean high water (MHW) or any number of tidal datums

Tidal inlet

A channel which is maintained by bidirectional tidal flow, and which connects the ocean with a back-barrier tidal creek system. Tidal inlets (and/ or lagoon hypsometry) frequently, although not always, restrict tidal exchange relative to adjacent marine shores, and barrier inlets may be prone to closure or new breaches during storms

Tidal prism

The difference between the volume of water in an estuary or lagoon at high tide and at low tide, or the amount of water that flows into and out of an estuary or lagoon over a tidal cycle

Tidal range

The difference in height between mean high water and mean low water

Tidal resonance

In the open ocean, tides are typically <2 m. When a tide wave enters a bay, river mouth, or continental shelf, a propagating tide may meet a reflection from a previous tide, amplifying the tidal range

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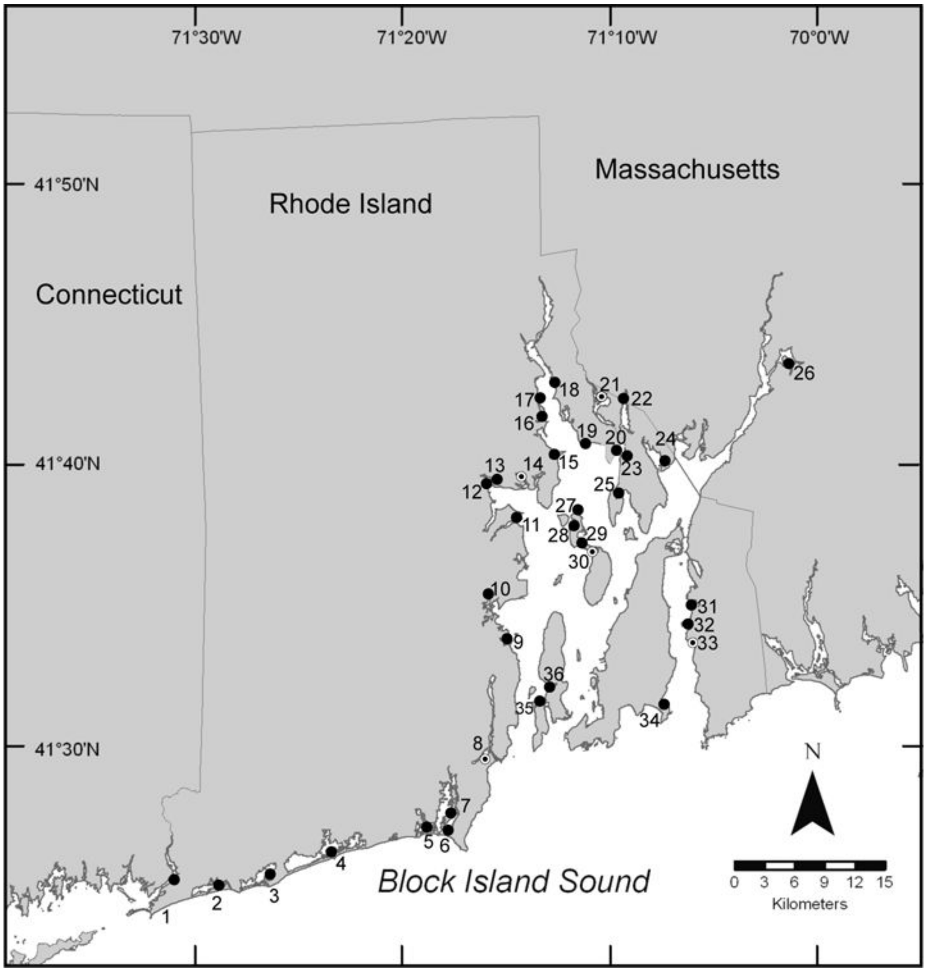

Fig. 1.

Map showing distribution of study sites. Elevation and marsh vegetation loss were measured at a total of thirty six sites. Site numbers are defined in Table 1. At a subset of five sites denoted by fisheye symbols a higher resolution assessment was conducted. Field mesocosms were deployed at three of these sites: (21) Hundred Acre Cove, Barrington, RI, 41°46′N, 71°19W, (30) Nag Marsh, Prudence Island, RI, 41°37′N, 71°19′W, and (33) Mary Donovan Marsh, Little Compton, RI, 41°32′N, 71°13′W

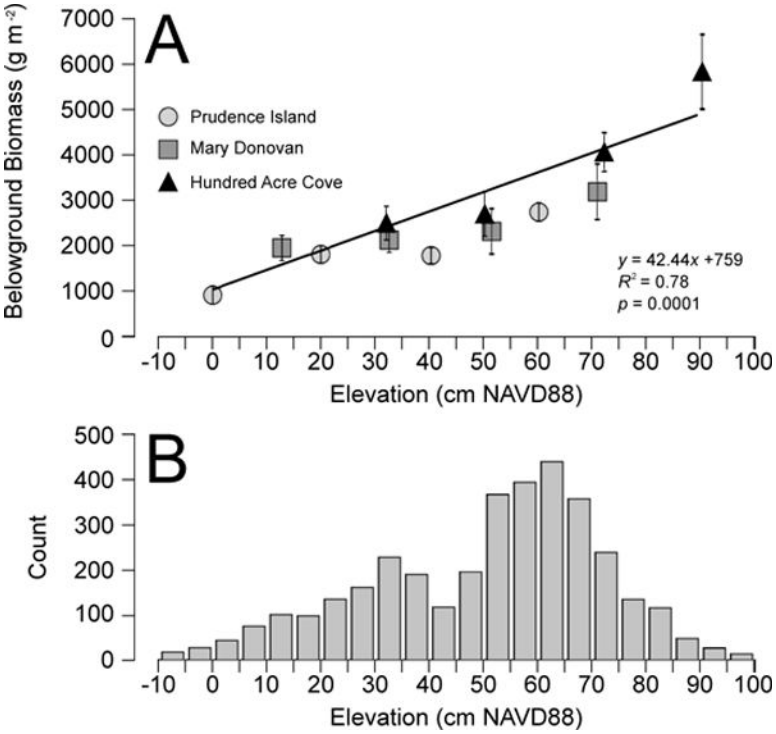

Fig. 2.

(A) The relationship between elevation, as a surrogate for inundation, and belowground biomass production for *Spartina alterniflora;* (B) The range of elevations found in 36 Rhode Island coastal marshes

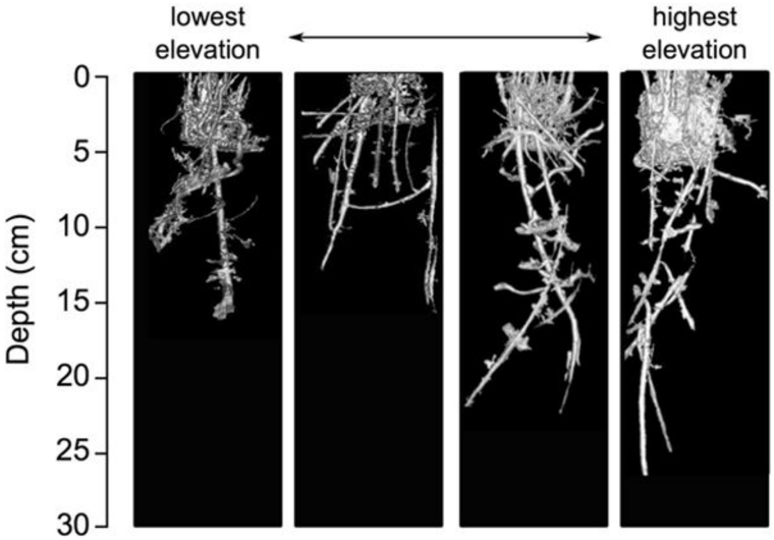

Fig. 3.

Rhizome and root visualization for select mesocosm pots grown under varying inundation at Nag Marsh, Prudence Island as shown by CT scan

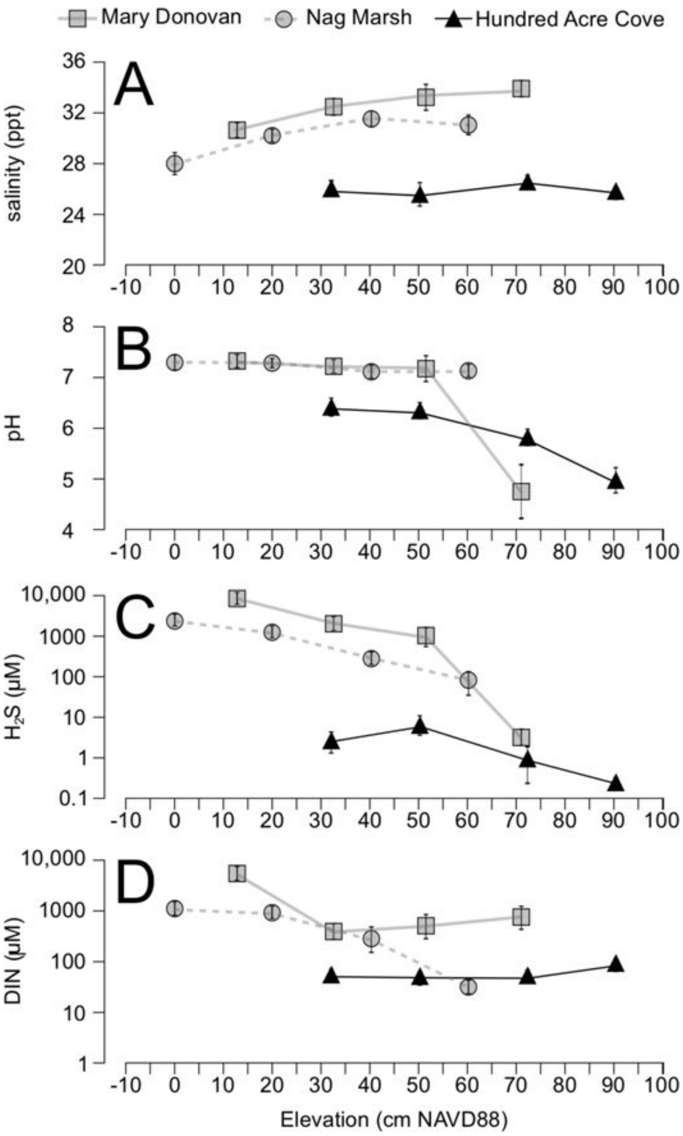

Fig. 4.

(A) Porewater salinity, (B) pH, (C) Hydrogen sulfide, and (D) Dissolved Inorganic Nitrogen (DIN) concentrations as a function of elevation and site, for field mesocosms

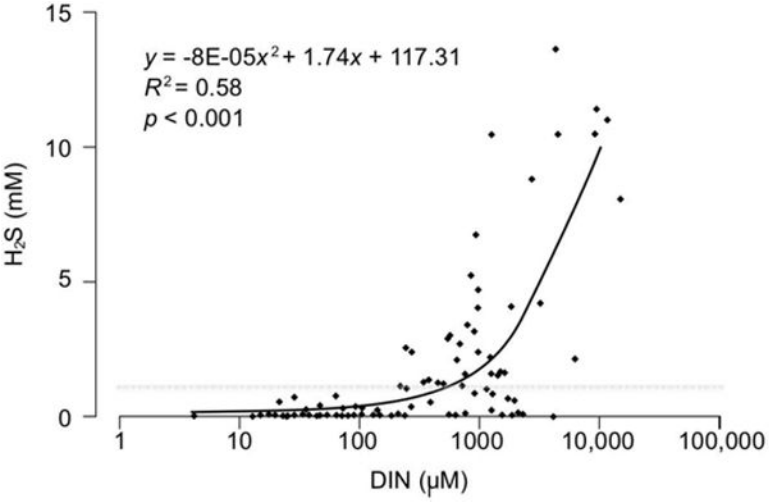

Fig. 5.

Porewater hydrogen sulfide concentration as a function of porewater DIN. Porewater sulfide values above 1mM or 1,000 μM (denoted by a dashed line) are known to reduce growth and nitrogen uptake (Koch et al. 1990)

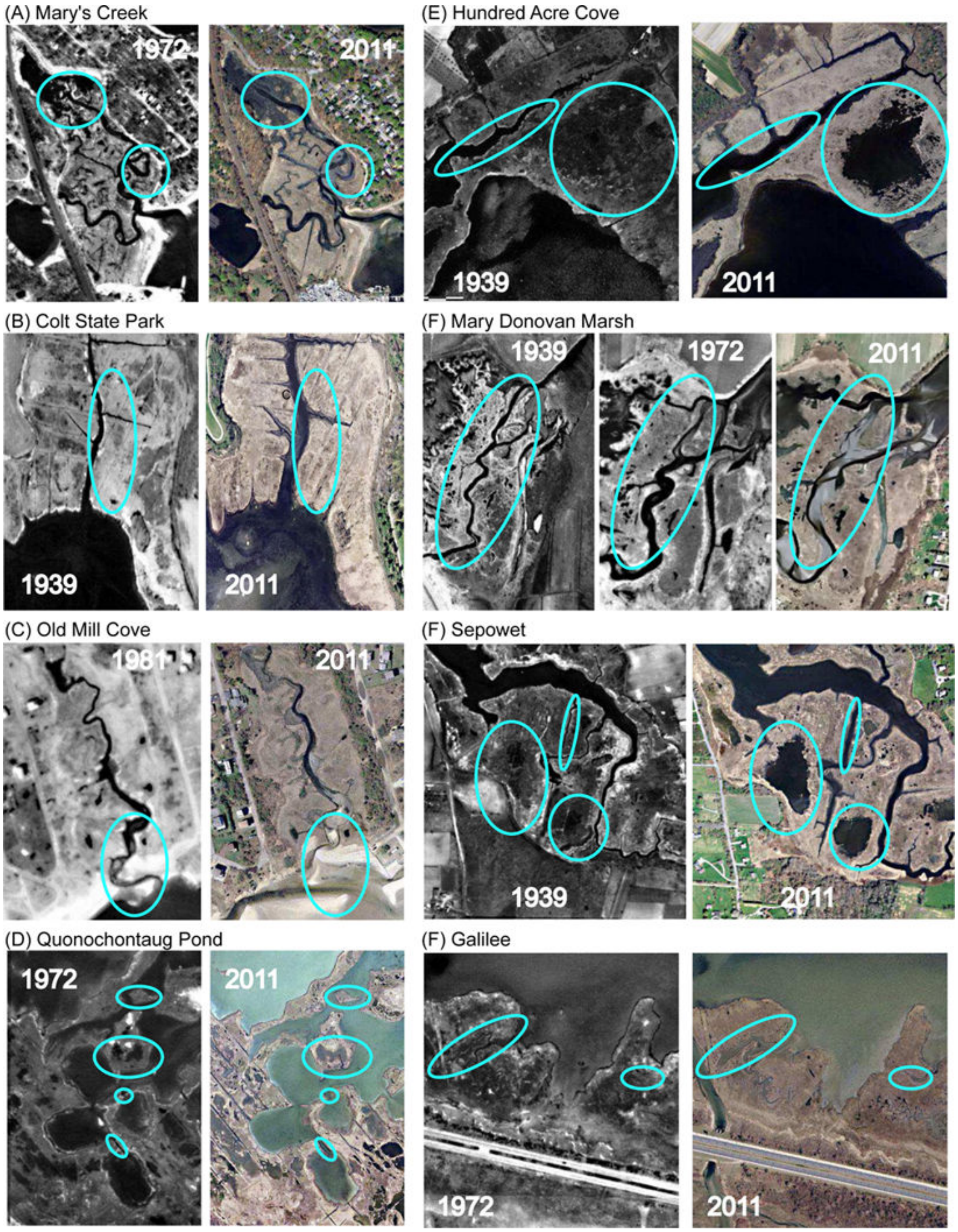

Fig. 6.

Spatial patterns of marsh vegetation loss at eight Rhode Island marshes: (A) imagery of Mary’s Creek shows creek widening, point bar loss, reduction of sinuosity, and loss of marsh in the bay-head region of the backbarrier estuary; (B) creek widening at Colt State Park; (C) barrier erosion and creek widening at Old Mill Cove; (D) channel widening and island shrinkage at Quonochontaug Pond; (E) channel widening, shoreline erosion, and pond development at Hundred Acre Cove; (F) tidal channel enlargement at Mary Donovan Marsh;

(G) pond development and channel widening at Sepowet; and (H) tidal channel headward erosion at Galilee

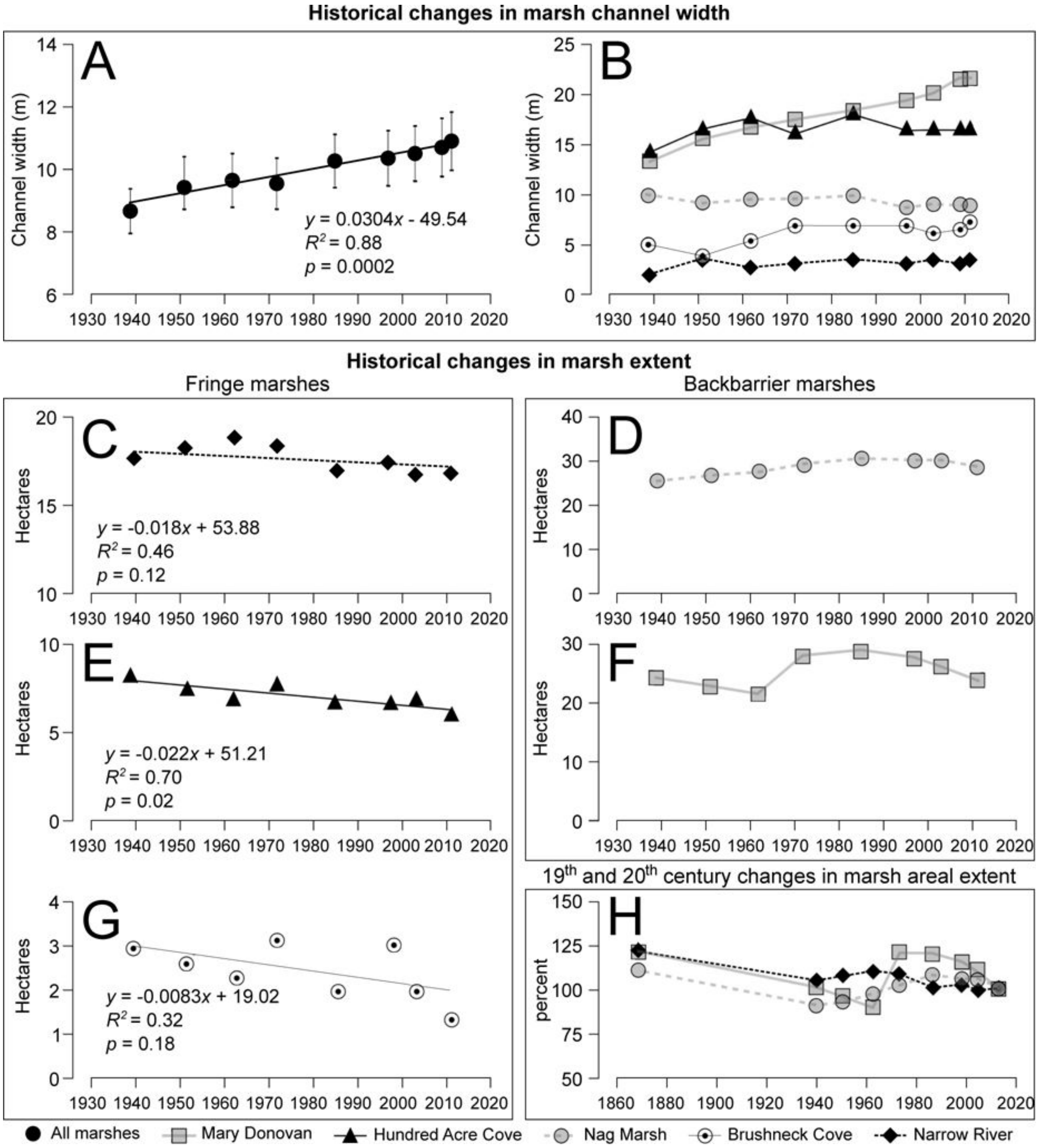

Fig. 7.

(A) Mean cross sectional tidal channel width at 100 locations at five focus sites, and (B) for individual sites from 1939 to 2011. (C) Change in vegetation area from 1939 to 2011 at Narrow River; (D) Nag Marsh; (E) Hundred Acre Cove; (F) Mary Donovan Marsh, and (G) Brushneck Cove. The left panels (C,E,G) are fringing marshes; right panels (D,F) are backbarrier marshes. (H) Trends in vegetation area for Narrow River, and Mary Donovan Marsh including data from early U.S. Coast Survey charts. Charts for Brushneck Cove do not depict tidal wetlands, and Hundred Acre Cove was not surveyed

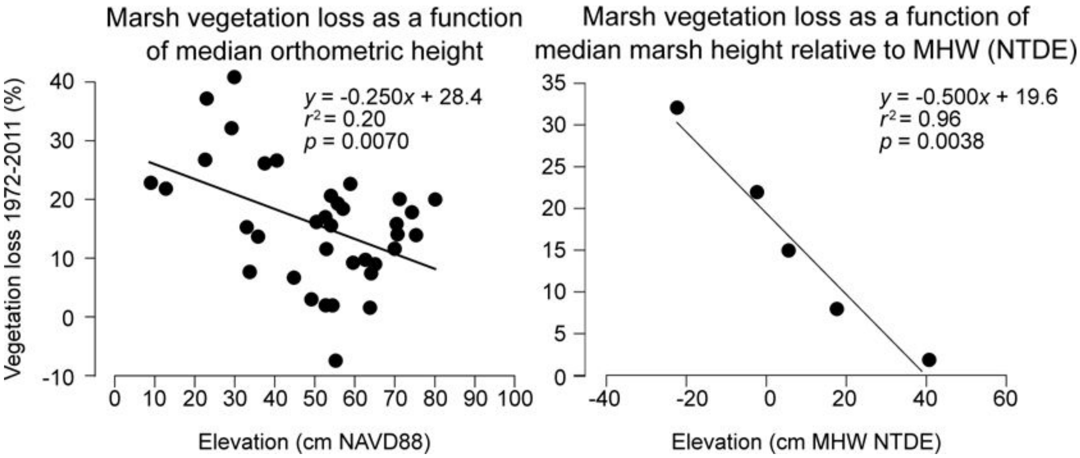

Fig. 8.

(A) Percent vegetation loss from coastal wetlands between 1972 and 2011 relative to 1972 area as a function of median orthometric height and (B) median marsh elevation relative to mean high water (MHW NTDE) for five focus sites. Values for MHW were calculated using empirical water level measures during 2011 and 2012 and referenced to the NTDE using the modified range ratio method (NOAA 2003). Values of MHW from VDATUM or interpolated from tide stations were found to be poor estimates of site-specific MHW values.

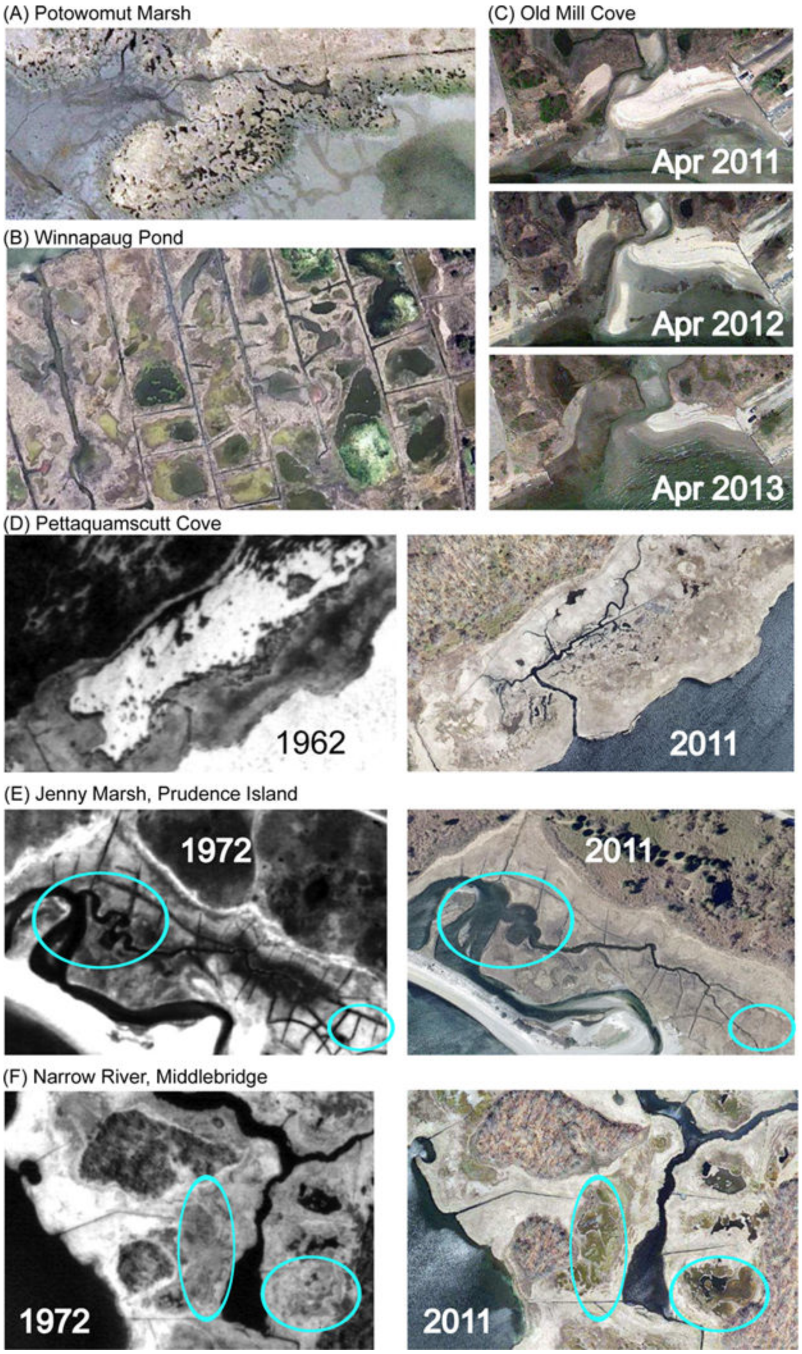

Fig. 9.

Landscape images of marsh vegetation patterns: (A) Pockmarked low marsh vegetation at Potowomut, Warwick; (B) grid-ditched marsh and ponding at Winnapaug pond, Weekapaug, RI; (C) Barrier inlet changes at Old Mill Cove, Conimicut before and after storms Irene and Sandy on 20 August 2011 and 28 October 2012, respectively; (D) Only two instances of natural pond draining through channel incision have been found through extensive examination of Rhode Island historic aerial imagery: shown here is a pond at Narrow River which drained via channel incision in the mid 1980s; (E) First order channels have narrowed, but larger order channels have expanded between 1972 and 2011 at Jenny Marsh, Prudence Island; (F) Interior ponding has caused vegetation die-back at the Narrow River Estuary, Narragansett, RI

Table 1

Rhode Island marshes assessed in this study. Numerical code refers to Fig. 1 locations. Elevation indicates median elevation relative to the NAVD88 Geoid12A datum. Loss mode is indicated as (*S*) shoreline erosion, (*C*) creek widening, (*I*) interior ponding, or *(B)* bay head lagoon expansion.

	Site	Marsh Type	Location	Median Elevation (m)	1972 Area (ha)	2011 Area (ha)	Percent Change (%)	Type
1	Avondale	Fringing	41°19′N, 71°50′W	0.23	4.93	3.10	−37.2 ± 1.8	*C,I*
2	Winnapaug	Fringing salt pond	41°20′N, 71°46′W	0.13	59.8	46.7	−21.9 ± 4.0	*S,C,I*
3	Quonochontaug	Fringing salt pond	41°20′N, 71°43′W	0.23	28.7	21.2	−26.1 ± 2.5	*S,C,I*
4	Ninigret Pond[1]	Fringing salt pond	41°22′N, 71°38′W	0.09	82.8	64.0	−22.7 ± 4.1	*S,C,I*
5	Succotash	Fringing salt pond	41°23′N, 71°31′W	0.30	37.5	22.2	−40.8 ± 4.2	*S,C,I*
6	Galilee	Fringing salt pond	41°23′N, 71°30′W	0.60	8.5	7.7	−9.2 ± 3.1	*S,I*
7	Island Rd.	Fringing salt pond	41°23′N, 71°31′W	0.49	0.84	0.81	−3.0 ± 1.9	*C*
8	Narrow River[2]	Fringing estuarine	41°26′N, 71°28′W	0.34	18.3	16.9	−7.7 ± 4.0	*S,I*
9	Bissel Cove	Fringing cove	41°16′N, 71°26′W	0.36	3.8	3.3	−13.8 ± 1.5	*S,B*
10	Mill Cove	Fringing cove	41°35′N, 71°27′W	0.53	4.9	4.1	−17.0 ± 7.0	*S,C*
11	Potowomut	Back-barrier	41°40′N, 71°25′W	0.55	7.2	7.7	+7.4 ± 1.6	*C*
12	Mary’s Creek	Back-barrier	41°41′N, 71°27′W	0.54	5.7	4.5	−20.7 ± 2.4	*S,C,I,B*
13	Apponaug	Fringing	41°42′N, 71°27′W	0.53	1.8	1.8	−1.6 ± 2.2	*S*
14	Brushneck Cove	Fringing	41°41′N, 71°25′W	0.29	3.0	2.1	−32.0 ± 1.8	*S,C*
15	Old Mill Cove	Back-barrier	41°43′N, 71°20′W	0.38	1.5	1.1	−26.4 ± 1.6	*S, B*
16	Passeonkquist	Fringing cove	41°46′N, 71°24′W	0.75	4.2	3.6	−13.7 ± 2.2	*S,C*
17	Stillhouse Cove	Fringing cove	41°46′N, 71°23′W	0.57	0.84	0.68	−18.4 ± 3.9	*S*
18	Watchemoket	Fringing cove	41°48′N, 71°22′W	0.40	0.92	0.67	−26.3 ± 4.2	*S*
19	Barrington Beach	Back-barrier	41°44′N, 71°19′W	0.74	7.5	6.2	−17.5 ± 2.1	*I,B*
20	Smith Cove	Fringing cove	41°43′N, 71°18′W	0.71	8.2	6.6	−19.7 ± 2.0	*S*,C
21	100 Acre Cove	Fringing	41°46′N, 71°19′W	0.59	7.7	6.0	−22.4 ± 2.6	*S,C,I*
22	Palmer River	Fringing river	41°46′N, 71°17′W	0.56	82.9	67.0	−19.1 ± 6.2	*S,C,I*
23	Jacob’s Point	Back-barrier	41°43′N, 71°17′W	0.70	8.4	7.2	−14.0 ± 3.8	*C, B*
24	Chace Cove	Back-barrier	41°42′N, 71°14′W	0.69	5.1	4.5	−11.6 ± 2.0	*C*
25	Colt State Park	Fringing cove	41°41′N, 71°17′W	0.70	22.5	18.9	−15.8 ± 2.0	*C*
26	Assonet	Fringing cove	41°47′N, 71°5′W	0.80	3.9	3.2	−19.8 ± 3.1	*C*
27	Providence Pt.	Back-barrier	41°39′N, 71°20′W	0.64	3.3	3.0	−7.4 ± 3.2	*S,C*,I,B

	Site	Marsh Type	Location	Median Elevation (m)	1972 Area (ha)	2011 Area (ha)	Percent Change (%)	Type
28	Coggeshall	Fringing	41°39′N, 71°20′W	0.62	22.0	19.9	−9.6 ± 2.5	*S,C,I*
29	Jenny	Back-barrier	41°38′N, 71°20′W	0.53	9.5	8.5	−11.4 ± 2.4	*S,C,I,B*
30	Nag Marsh	Back-barrier	41°37′N, 71°19′W	0.64	29.3	28.8	−1.8 ± 3.5	*S,C,I*
31	Sepowet	Back-barrier	41°35′N, 71°13′W	0.65	55.8	50.7	−9.0 ± 3.6	*S,C,I*
32	Fogland	Back-barrier	41°34′N, 71°13′W	0.50	2.2	1.8	−16.2 ± 3.0	*S,B*
33	Mary Donovan	Back-barrier	41°32′N, 71°13′W	0.33	28.3	24.1	−14.9 ± 1.7	*S,C,I,B*
34	Satchuest	Back-barrier	41°29′N, 71°15′W	0.54	4.6	3.9	−15.2 ± 4.2	*I*
35	Fox Hill	Fringing cove	41°29′N, 71°23′W	0.45	9.8	9.2	−6.4 ± 1.9	−*S,C*
36	Round Marsh	Fringing cove	41°31′N, 71°23′W	0.54	22.8	22.4	−1.8 ± 3.6	*S,C,I*
TOTAL					609.2	504.0	−17.3 ± 3.3%	

[1] Including Green Hill Pond

[2] Canonchet Farms section of Pettaquamscutt Cove only

Table 2

Maps and imagery used to digitize salt marsh extent and channel width. Rhode Island imagery was obtained courtesy of the Rhode Island Geographic Information System (RIGIS), Massachusetts imagery was obtained courtesy of the U.S. Geological Survey, via Earth Explorer, and U.S. Coast survey charts were obtained from the National Archive

Year	Type	Resolution	Title/Origin	Source
1866	Coast survey chart	1:10,000	T-1054 Prudence Island U.S. Coast Survey	National Archive
1867	Coast survey chart	1:10,000	T-912 North of Quonset to Greenwich Bay U.S. Coast Survey	National Archive
1869	Coast survey chart	1:10,000	T-1118 Narrow River to Saunderstown U.S. Coast Survey	National Archive
1870	Coast survey chart	1:10,000	T-1156 East side Sakonnet River U.S. Coast Survey	National Archive
1939	panchromatic	1.3 m pixel^{-1} 1:24,000	RI Statewide Planning Program	RIGIS
1951–2	panchromatic	1.5 m pixel^{-1} 1:24,000	RI Statewide Planning Program	RIGIS
1962	panchromatic	1.1 m pixel^{-1} 1:24,000	RI Statewide Planning Program	RIGIS
1972	panchromatic	0.60 m pixel^{-1} 1:4,800	RI Statewide Planning Program	RIGIS
1974	panchromatic	1.5 m pixel^{-1} 1:24,000	U.S. Geological Survey	USGS
1985	color infrared	2.11 m pixel^{-1} 1:58,000	USDA National Resource Conservation Service	RIGIS
1997	color ortho	0.61 m pixel^{-1} 1:5,000	RI Dept. of Transportation	RIGIS
2003–4	color digital ortho	0.61 m pixel^{-1} 1:12,000	USDA National Agricultural Inventory Program	RIGIS
2008	panchromatic	0.61 m pixel^{-1} 1:12,000	RI Dept. of Environmental Management	RIGIS
2010	color near infrared digitial	1.0 m pixel^{-1}	USDA National Agricultural Inventory Program	USGS
2011	color near infrared digitial ortho	0.15 m pixel^{-1}	RI Dept. of Environmental Management	RIGIS

Table 3

Bulk density, percent organic matter by weight, volume of organic matter, water and gas, mineral matter, and soil texture in the upper 50 cm of soil, from five southern New England tidal marshes.

		Bulk density ($g\ cc^{-1}$)		Organic content (% dry wt)		Percent by volume		Soil composition (%)		
Marsh type	**No. of Cores**	**Mean**	**(SD)**	**Mean**	**(SD)**	**Organic matter, water, gas**	**Mineral matter**	**sand**	**silt**	**clay**
Back-barrier	6	0.35	(0.25)	34.85	(18.61)	90.0	10.0	22.6	69.7	7.8
Nag Marsh	3	0.22	(0.05)	49.14	(13.54)	95.6	4.4	19.7	71.4	8.9
Mary Donovan	3	0.48	(0.31)	20.55	(9.81)	84.4	15.6	25.4	67.9	6.7
Fringing	9	0.26	(0.13)	40.94	(12.83)	93.6	6.4	11.1	79.1	9.8
Narrow River	3	0.20	(0.03)	50.26	(7.90)	96.1	3.9	7.3	82.3	10.4
100 Acre Cove	3	0.31	(0.18)	41.71	(12.37)	92.4	7.6	11.6	78.5	9.8
Brushneck Cove	3	0.28	(0.11)	30.85	(9.96)	92.3	7.7	14.5	76.4	9.1
All sites	15	0.39	(0.19)	38.51	(15.58)	92.2	7.8	15.7	75.3	8.9