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Transient coastal landscapes: Rising sea level threatens salt marshes

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Abstract

Salt marshes are important coastal environments that provide key ecological services. As sea level rise has accelerated globally, concerns about the ability of salt marshes to survive submergence are increasing. Previous estimates of likely survival of salt marshes were based on ratios of sea level rise to marsh platform accretion. Here we took advantage of an unusual, long-term (1979-2015), spatially detailed comparison of changes in a representative New England salt marsh to provide an empirical estimate of habitat losses based on actual measurements. We show prominent changes in habitat mosaic within the marsh, consistent and coincident with increased submergence and coastal erosion. Model results suggest that at current rates of sea level rise,

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marsh platform accretion, habitat loss, and with the limitation of the widespread "coastal squeeze", the entire ecosystem might disappear by the beginning of the next century, a fate that might be likely for many salt marshes elsewhere.

Keywords: salt marsh loss, sea level rise, ecological services, vegetation mosaics, coastal squeeze.

Highlights:

Accelerated sea level rise increased submergence in a New England salt marsh.

Increased sea level collapsed creek banks and widened channels.

Low marsh vegetation cover increased while high marsh cover decreased.

Habitat, sediment stability, C burial, foraging and nursery services were affected.

Vegetation cover seems likely to disappear by the beginning of the next century.

1. Introduction

Global sea levels have been rising since the last glacial maximum, at rates of around 1 mm yr^{-1} during late Holocene (Kemp et al., 2011). These moderate sea level rise rates allowed development of coastal salt marshes, through vertical build-up of salt marsh platforms, by accumulation of underground organic matter and sediment deposition, at rates greater than or equal to sea level rise (Redfield, 1972).

Sea levels have variably increased across recent centuries (Church and White, 2011), but a significant recent acceleration of sea level rise is apparent across many coasts of the world (Kopp et al., 2016; Merrifield et al., 2009), with hotspots including the Atlantic coast of North America (Sallenger et al., 2012). Accelerated sea level rates are evident in Cape Cod (Fig. 1a), where recent rate of sea level rise is almost six times higher than the rates observed during the late Holocene. Such increases in sea level force a plethora of environmental effects on coastal ecosystems such as salt marshes (Morris et al., 2002; Valiela, 2006).

Accelerated sea level rise rate, if not matched by accretion of the marsh platform, should significantly increase submergence and shift the vegetation mosaic within salt marshes (Miller et al., 2001; Smith, 2014). Plant zonation within salt marshes is vertically stratified, with species of plants and algae constrained within certain elevation ranges (Valiela and Rietsma, 1995) (Fig. 1b, c). As sea level rises, the vegetation strata can only survive if they migrate into shallower sites in a landward direction. Rising sea levels also force accompanying changes in coastline position and re-distribute sediment within coastal wetlands (Kolker et al., 2009).

Concerns about sea level-driven alteration of vegetation and sediments in salt marsh ecosystems are based on potential loss of important ecological services that these coastal environments furnish—stabilization of sediment, shoreline protection, interception of nutrients, provision of habitat use for shellfish, birds, and other fauna, sequestration of atmospheric carbon, plus a number of other functions (Craft et al., 2009; Valiela, 2006). The plethora of ecological services furnished by salt marshes result from ecological, biogeochemical, and geological mechanisms that differ among the various vegetation and sediment habitats within salt marshes. Thus, as sea level alters marsh habitat mosaics, it seems reasonable to expect potential changes in services provided by marsh ecosystems, and loss of these services if the marsh disappears.

There have been differing conclusions about salt marshes facing rising sea levels, ranging from exceptionally vulnerable (Watson et al., 2017), to threatened (Crosby et al., 2016), to susceptible (Valiela, 2006), to over-estimated (Kirwan et al., 2016). These studies rely on meta-analyses that compare estimated sea level rise to estimated salt marsh platform accretion. Adaptability of salt marshes to rising sea levels depends on their ability to build vertically at rates greater than or equal to relative sea level rise, or else to migrate inland at rates faster than erosion at their seaward boundary. Moderate increases of flooding duration can increase mineral sediment deposition rates (Vandenbruwaene et al., 2011), and plant productivity (Kirwan and Guntenspergen, 2012), contributing to increased vertical accretion, and reduced erosion rates. In addition, spatial and temporal variations in accretion, and plant-mediated changes of hydrodynamics and sediment transport mechanics along the marsh platform, may also alter wetland stability during periods of elevated sea level rise rates (Rodriguez et al., 2017; Sandi et al., 2018; Alizad et al., 2016; Belliard et al., 2016). Such biogeomorphic feedbacks allowed marshes to keep pace with rising sea levels during recent centuries (Kirwan et al., 2016; Morris et al., 2002).

Current and likely future conditions forecast a more problematic future for salt marshes through this century. First, comparisons with historical fluctuations in global sea level reconstructions show that recent accelerated rates of global sea level rise are unprecedentedly high, and expected to continue accelerating in the future (Merrifield et al., 2009). Second, susceptibility of coastal wetlands to sea level rise might be greatest in coasts with lower fluvial sediment sources to support platform accretion (Kirwan et al., 2010). Third, in populated areas, salt marsh inland migration can be limited by the effects of the “coastal squeeze”, a phenomenon that occurs when structures built landward of coastal wetlands, or steep upland topography, prevent landward incursion of marsh vegetation as sea level rises (Doody, 2004).

Here we examine the effects of recent rising sea level on the habitat mosaic within a coastal salt marsh by taking advantage of an unusual, spatially detailed comparison of decadal changes in Great Sippewissett Marsh, on Cape Cod. Vegetation in the marsh is characteristic of New England and other regions along the North American Atlantic Coast, with a variety of plant species located at specific elevations within the tidal range according to their relative tolerance to submergence (Ewanchuck and Bertness, 2004; Nixon, 1982, and Fig. 1b, and c). This marsh is, as are many wetlands around the world, surrounded by human development. The marsh is exposed to mean tidal range of about 1.65 m, and sediment accretion takes place largely through accumulation of below ground plant biomass, rather than via riverine sediment inputs (Valiela, 2015).

In this paper we test whether 1) there have been recent decadal changes in the vegetation mosaic of a representative salt marsh in New England; 2) the changes in salt marsh vegetation were associated with parallel rise in sea level; and 3) knowledge of the links of recent vegetation and sea level changes could provide the means, combined with forecasted future sea level rise, to predict future trajectories of the marsh mosaic. Expanding scale from this empirically based synthesis, we then carry out a literature meta-analysis to elucidate the fate of salt marshes across the world's coasts to the end of this century in the context of accelerated sea level rise.

2. Materials and Methods

2.1 Sea level rise data and calculations.

Sea level data were obtained from a tide gauge station located eight km south of Great Sippewissett Marsh, in Woods Hole, MA (NOAA Station ID#: 8447930). 1972 to 2013 monthly sea level data were used to estimate sea level rise rates. To determine whether there had been a significant shift in sea level rise, and when it took place, we carried out a piecewise linear regression analysis of the sea level rise time series. The intent of this statistical analysis was to objectively determine the best fits of regression models to segments of the time series of sea level rise data, to estimate significant differences of sea level rise rates during different segments of the data series, as well as identify the hinge years when significant shifts may have taken place.

2.2 Field mapping and GIS processing.

The 1979 field-recorded mapping involved examination of the entire salt marsh. We subdivided an outline map of the marsh, obtained from a detailed aerial photograph, into sub-units of the marsh approximately of 100 m^2. These sub-units were then closely examined, and habitat data recorded while walking through each area and recording position and taxonomic make up of parcels with different vegetative cover, aiming at a resolution of at least 0.5 m.

To draw the habitat map for 2015, we developed an ortho-rectified map of Great Sippewissett Marsh using an unpiloted aerial vehicle flown at an altitude of 50m. A DJI Phantom III Professional quadcopter provided multiple overlapping aerial images using a built-in 12-

megapixel camera borne by the unpiloted aerial vehicle, and fitted with an f/2.8 lens with a 94-degree field of view. Photographs were captured in an overlapping grid pattern optimized for the generation of a composite orthomap. The application, Map Pilot 1.41, by DronesMadeEasy© automatically generated the photographic grid and photo sequence through a simple user interface where we defined the target area, altitude, and percentage overlap. We specified an 80% overlap to ensure a high-quality image. The unpiloted aerial vehicle automatically followed the specified flight path and captured over 1,000 individual, overlapping images. A web service provided by MapsMadeEasy.com allowed the resultant images to be uploaded for rendering. The individual images were then aligned into a composite orthorectified image with a resolution of 0.3 m, suitable for geospatial analysis, and comparable to the resolution of the 1979 map (Fig. SI 1).

We used QGIS Semi-Automatic Classification Plugin 4.9.5 (SACP) to classify the color-coded vegetation types present in the 1979 vegetation map to convert them to polygons in a shapefile. To create a corresponding 2015 vegetation map, the unpiloted aerial vehicle-obtained composite image was divided into ten sections of approximately equal size, each of which was then classified individually. Open water, creek bank, sand dune and beach, and upland vegetation categories were all manually classified and converted to polygons. High and low marsh vegetation were then isolated and classified using SACP, according to the differences in texture and coloration of the two vegetation types. This classification was then visually checked for accuracy against the stitched image, and then further verified by confirming category boundaries observed in situ. We then extracted vegetation type areas and point-by-point conversion data from the resulting spatially referenced shapefiles. We measured shoreline movement by averaging 10 measurements taken at equidistant points along the shore. Dune area was measured above 41°34'52.4"N and below 41°35'16.2"N. Marsh vegetation data were not measured above 41°59’28.6"N, and the lower boundary was defined by a road to the south (Fig. SI 1).

All GIS analyses were completed using the QGIS 2.14 software. Both the 1979 vegetation map and the 2015 unpiloted aerial vehicle-obtained composite image were geo-referenced against fully referenced aerial and satellite imagery corresponding to the years each was completed, using buildings, roads and other fixed structures and features as reference points. We used 23 control points for geo-referencing. The root mean square error during the process was 3.3 m.

2.3 Numerical modeling of future marsh habitat trajectories.

The modeling effort aimed to predict trajectories of the different vegetation habitats through the 21st century as sea level rose. The strategy was to use habitat and vegetation changes we observed in the real-time comparisons from the 1979 and 2015 maps, plus sea level and elevation constraints for each marsh habitat (Fig. 1c), as the basis for anticipating future changes throughout Great Sippewissett Marsh.

The modeling conservatively assumed that the current effective sea level rise as 2.93 $mm \cdot yr^{-1}$ [the difference between current sea level rise rate of 5.83 mm yr^{-1} (Fig. 1a) and accretion rate (~2.9 mm yr^{-1}, Kinney and Valiela, 2013)], would be maintained through the century. Some studies of salt marsh accretion assume increase in water depth favors accretion by increasing external sediment deposition (Fagherazzi et al., 2012), but in Great Sippewissett Marsh and other New England salt marshes, accretion is mainly from burial of roots and rhizomes, rather than from fluvial sediment supply (Drake et al., 2015; Valiela, 2015). We need to emphasize here that 1) fluvial transport of terrestrial sediment is minimal, and freshwater enters the marsh via groundwater flow through sand-gravel aquifers (Valiela and Teal, 1979), and 2) change in vegetation-stimulated accretion is captured in the model by use of the data on vegetation cover itself. We conservatively assumed that the constrained vertical distribution of the several vegetation zones (Fig. 1c), defined by different tolerance of plant species to inundation and salinity (Kirwan and Guntenspergen, 2010), will not change. The model applied current annual rates of change in area of vegetation type for each type of habitat, as a point of departure for the simulation.

We then defined, at annual time-steps, the area of each elevation-constrained habitat (Fig. 1c) that would be present as effective sea level rose through the years. This defined conservative future trajectories for the several habitats found within Great Sippewisett Marsh. Examination of the surrounding terrain, and the presence of a constructed embankment on the east side of Great Sippewissett Marsh (see Fig. SI 1) suggested that landward incursion by wetland vegetation would be prevented (the "squeeze" effect), a ceiling to landward incursion that was also included in the model.

From recent trajectories of sea level rise in the area (Fig. 1 a), we assumed that the areas of the various habitats were relatively constant in the period between 1979 and 1992, when sea level rise rate was relatively low, at 0.99 mm yr^{-1}. Long-term (1932-1992) rate of sea level rise in this location was 2.54 mm yr^{-1}. Both, long- and short-term pre-1992 rates fall below the average accretion rates in Great Sippewissett Marsh of around 2.9 mm yr^{-1} (Kinney and Valiela, 2013). We therefore concluded that observed habitat changes took place after the acceleration of rates of sea level rise to 5.83 mm yr^{-1} after 1992.

Observed rates of habitat change were used to numerically model possible post-2015 trajectories of these habitats in Great Sippewissett Marsh. For each year, changes in the different habitats were calculated as the balance between gains and losses of areas of the specific habitats, and propagated into the future as

$$A_{X,t} = A_{X,t-1} + Again_X - Aloss_X, \tag{1}$$

where $A_{X,t}$ is the area covered by habitat *X* in year *t*, $A_{X,t-1}$ the area in the previous year, and $Again_X$ and $Aloss_X$ the amount of area gained or lost by habitat *X* each year.

For un-vegetated sandy habitats and dunes, we assumed no gains in area as sea level rises. Our data revealed partial colonization by low marsh of sandy habitats and of the landward side of the dunes as these habitats are progressively inundated (see Fig. SI 2). We assumed this colonization to continue at current rates until the total colonization of available sandy habitat area. Changes in area of sandy habitats were calculated as

$$A_{SH,t} = A_{SH,t-1} - Col_{LMtoSH}, \tag{2}$$

where Col_{LMtoSH} the amount of area of sandy habitats colonized by low marsh each year.

For dunes, measured losses were due to low marsh colonization in the landward side and submergence in the seaward side. We assumed that initially both sides of the marsh were approximately equal so low marsh colonization will only occur until the complete eastern half was covered by this type of vegetation. Changes in dune habitats were calculated as

$$A_{DH,t} = A_{DH,t-1} - Col_{LMtoDH} - Subm_{DH}, \tag{3}$$

where Col_{LMtoDH} is the area of dunes colonized by low marsh, and $Subm_{DH}$ the area dunes lost by submergence each year.

We assumed, from knowledge of the local landscape (and Fig. SI 1) that the “squeeze” effect exerted on high marsh vegetation by the berm on the east side of the marsh, and *P. australis*, will continue into the future, preventing inland expansion of the marsh. Therefore, for high marsh vegetation, the coastal “squeeze” prevented up-gradient incursion, and hence constrained the gains in area in the upper range of topography,

$$A_{HM,t} = A_{HM,t-1} - Subm_{HM} - Col_{PhragtoHM}, \quad (4)$$

where $Subm_{HM}$ is the area of high marsh lost by submergence of its lower elevation limit and $Col_{PhragtoHM}$ is the area of high marsh colonized by *P. australis* each year.

Most of the 1978 area of high marsh lost in 2015 was invaded by low marsh. We assumed that the area of high marsh lost by submergence each year was initially colonized by low marsh,

$$Subm_{HM} = Col_{LMtoHM}, \quad (5)$$

where Col_{LMtoHM} is the area of high marsh colonized by low marsh each year.

To model changes in area of *P. australis* we assumed a constant rate of colonization while high marsh habitats are still present. After high marsh areas disappear, and in spite of tolerance to salinity of the new genetic variant, *P. australis* area was assumed to recede and be invaded by low marsh at the same rate as high marsh,

$$A_{Phrag,t} = A_{Phrag,t-1} + Col_{PhragtoHM} \quad if \ A_{HM,t} > 0, \quad (6)$$

$$A_{Phrag,t} = A_{Phrag,t-1} - Subm_{HM} \quad if \ A_{HM,t} = 0. \quad (7)$$

Future trajectories of low marsh vegetation where calculated as the balance between the areas gained by colonization of high marsh, sand and dune habitats, and the by the effects of sea level rise as

$$A_{LM,t} = A_{LM,t-1} + Col_{LMtoHM} + Col_{LMtoSH} + Col_{LMtoDH} - Subm_{LM}, \quad (8)$$

where $Subm_{LM}$ is the area of low marsh lost by submergence of its lower elevation limit each year.

To solve equation 8, two different periods were defined. In the first period, where high marsh and sandy habitats were still present, low marsh upland colonization was still possible. The processes of colonization of new areas and accretion would redistribute much of the area covered by low marsh in the optimum central part of its elevation range (see Fig. 1b, c). We assumed that low marsh area cover follows a normal distribution (Tabak et al., 2016), with 99.9% of the area of low marsh distributed in the 30 cm of its elevation range, and much of its area distributed around the central part of this range. The losses caused by the effects of sea level rise would then occur in the lower elevation limit of low marsh distribution. We assumed that the amount of low marsh area at this lower elevation limit would be minimum and, therefore, we expect minimum losses in terms of area per year as sea level rises during this initial period,

$$Subm_{LM} = minSubm_{LM} \; if \; t < T_{Sq}, \qquad (9)$$

where T_{Sq} is the year where the sum of high marsh, *P. australis* and sandy habitats is null, and upland colonization of low marsh is not possible.

After the total loss of higher elevation areas, the “squeeze” effect will be exerted on low marsh. The cumulative losses of the normally distributed low marsh area would accelerate each year until half of the elevation range has been inundated and then decrease until the entire range is inundated. To simulate the changes in the rate of low marsh loss in this period, as the normal distribution of areas are inundated, we assumed that

$$Subm_{LM} = minSubm_{LM} + \frac{maxA_{LM}}{\sqrt{2\sigma^2\pi}} e^{-\frac{(t-\mu)^2}{2\sigma^2}} \; if \; t > T_{Sq}, \qquad (10)$$

where $maxA_{LM}$ is the area of low marsh in the year T_{Sq}. The parameters for the calculation of losses of a normal distribution of low marsh area were calculated as

$$\mu = \frac{1}{2}\frac{Range_{LM}}{SLR - Accr_{LM}}, and \tag{11}$$

$$\sigma = \frac{1}{6}\frac{Range_{LM}}{SLR - Accr_{LM}}, \tag{12}$$

where $Range_{LM}$ is the elevation range covered by low marsh, SLR the rate of sea level rise, and $Accr_{LM}$ is the rate low marsh accretion each year.

For the case of the creek banks, the gains in area by colonization of low marsh in the higher elevation limit of its distribution, and the losses by inundation of its lower limit, both caused by sea level rise, were considered equal during the first period of low marsh expansion. During this period, observed losses were assumed to be caused only by edge slumping,

$$A_{CB,t} = A_{CB,t-1} - Slump_{CB} \; if \; t < T_{Sq}, \tag{13}$$

where $Slump_{CB}$ is the rate of creek bank edge slumping each year.

After low marsh starts to recede, further losses of creek bank took place at a rate proportional to low marsh losses, and to the relative areas of both habitats. Creek bank vegetation area was then calculated as

$$A_{CB,t} = A_{CB,t-1} - Slump_{CB} + \left(A_{LM,t} - A_{LM,t-1}\right)\frac{A_{CB,t-1}}{A_{LM,t-1}} \; if \; t > T_{Sq}, \tag{14}$$

The calculation of the different terms used in the previous equations were based on observed changes in areas resulting from the comparison of 1979 and 2015 datasets, the elevation ranges of the various Great Sippewissett Marsh habitats (Valiela and Rietsma, 1995), measured accretion rates in Great Sippewissett Marsh (Kinney and Valiela, 2013), and the post-1992 local rate of sea level rise of 5.83 mm yr^{-1}. Values for these variables are in Table SI 1.

To assess uncertainty in sea level rise effects on the different habitat covers, the standard error of the post-1992 sea level rise estimates was used to recalculate model parameters, assuming the various habitat parameter responses are proportional to the difference between sea level rise and habitat accretion rates,

$$P_{X,SLR \pm S.E.} = P_{X,SLR} \frac{SLR \pm S.E - Acrr_X}{SLR - Accr_X}. \qquad (15)$$

3. Results and discussion

3.1 Sea level rise changes.

Sea level rise was variable throughout the study period (Fig. 1a), but remained at about 1 mm yr^{-1} up to the hinge year of 1992, and then rose to 5.83 mm yr^{-1} after that year. This difference appears large enough to expect significant effects on the vegetation and habitats of Great Sippewissett and similar salt marshes.

3.2 Overall changes in Great Sippewissett Marsh, 1979-2015.

The total area of Great Sippewissett Marsh system changed slightly between 1979 and 2015 (Fig. 2a, b, and Table SI 2). The marsh includes areas of vegetated cover, sand deposits (beach and dunes), and open water (Fig. 2a). The vegetated area was about twice that of open water, a proportion that suggests that Great Sippewissett is a reasonably mature marsh, where platform accretion historically exceeded, or kept up with sea level rise, vegetated areas predominated, and where exports of particulate organic materials to the adjacent sea were likely (Valiela, 1983).

Between 1979 and 2015 there was an almost 6% increase in open water area, and 3% and 19% decreases in vegetated cover and sand deposits, respectively (Fig. 2b). These recent changes suggest modest but detectable changes for the marsh as a whole, effects consistent with increased vegetation submergence and erosion of beach and dunes.

3.3 Alteration of the vegetation mosaic.

There were substantial changes in the specific units of the vegetation and habitat mosaic within Great Sippewissett Marsh between 1979 and 2015 (Fig. 2c, d and Fig. 3).

3.3.1 Loss of creek banks.

There was a 14% loss of creek banks and low-lying habitats that supported tall *Spartina alterniflora* (Fig. 2d). Creek bank loss was by slumping following increased submergence (Fig. SI 3a, b). Slumping of creek banks has been attributed to joint effects of nitrogen-related eutrophication and sea level rise (Deegan et al., 2012); both these drivers favor conversion of stands of short into taller *S. alterniflora*, changes that reduces marsh sediment cohesion (Deegan et al., 2012; Valiela, 2015). We found, however, that concentrations of dissolved inorganic nitrogen in waters of Great Sippewissett Marsh, while variable, have not increased across recent decades (Table SI 3), during which sea level has risen substantially in excess of local accretion of salt marsh platform (Valiela, 2015). This suggests that sea level effects might be the prime driver of creek bank loss in Great Sippewissett Marsh.

3.3.2 Increase in low marsh and decrease in high marsh.

The shift in vegetation mosaic included a 43% increase in area of low marsh vegetation, dominated by short *S. alterniflora* (Fig. 2d, Fig. 3a, and Table SI 2). During the same period, area covered by high marsh vegetation (mainly *Spartina patens*) decreased by nearly 60% (Fig. 2d, Fig. 3b, and Table SI 2). Increased submergence owing to recent sea level rise therefore shifted much of what was high to low marsh species cover, accounting for 66.5% of the gain in low marsh area cover; the remaining gain in low marsh cover was at the expense of newly submerged sand habitats. These major changes in the vegetated mosaic took place within the marsh even though there were only minor changes in total area cover of vegetation and open water (Fig. 2b, and Table SI 2). The increase in lower-lying low marsh at the expense of high marsh and sand habitats is clearly consistent with increased submergence as sea level rose, as reported for century-scale and decadal- and shorter-scale effects of sea level rise (Kolker et al., 2009; Raposa et al., 2017; Rietsma et al., 2011; Watson et al., 2016).

An obvious question is why high marsh plants did not simply migrate towards upland areas as sea level rose? We already mentioned two possible mechanisms in the discussion of the model approach. First, such a movement was made difficult by the presence of an embankment or berm, built during the 1870s to support a railroad (now a bicycle path) along the eastern boundary of the marsh (Fig. SI 1). Second, during the nearly four decades of our study, the landward edges of Great Sippewissett Marsh have been subject to invasion (Fig. 2d, and Table SI 2) by a genetic variant of common reed, *Phragmites australis*, as reported for other NE US coasts (Hazelton et al., 2014).

Instead, it is thought that the new variant is favored by its increased salinity tolerance and affinity for uptake of increased concentrations of dissolved inorganic nitrogen in freshwater flowing into salt marshes (Valiela, 2015). This invasive form of *P. australis* prevents landward movement of high marsh (Smith, 2013) owing to its characteristic growth as a narrow but aggressive phalanx along landward margins of salt marshes (Fig. SI 3c). *P. australis* is taller than high marsh plants, has robust belowground roots and rhizomes, and easily shades and excludes high marsh grasses.

3.4 Consequences for salt marsh ecological services.

Shifts in the vegetation mosaic following sea level rise may also alter the many other ecological services provided by salt marshes, including stability of underlying sediments, sequestration of carbon, and provision of food and nursery roles for animals.

3.4.1 Marsh sediment stability.

The changes recorded in the distribution of creek banks, low, and high marsh species are likely important for maintenance of stability of underlying sediments, an important ecological service provided by marsh vegetation. *S. alterniflora*, the dominant species in most Western Atlantic salt marshes, grows in taller, sparser stands low in the intertidal, and as dwarf, dense stands higher in the tidal range (Fig. 1b). The interwoven mass of roots and rhizomes of salt marsh plants is largely responsible for the consolidation of the otherwise loose sediments in marshes (Deegan et al., 2012; Valiela, 2015), but this stabilizing service is not uniform across the tidal range. The short *S. alterniflora* confers considerably greater binding to the underlying sediment because its root and rhizome mass is much denser than the tall form. Sediments in low-lying creek banks, usually supporting tall *S. alterniflora,* are less stable, and more likely to suffer erosion and slumping (Fig. SI 3a, b). On the whole, therefore, increased submergence weakens stability of salt marsh platforms.

3.4.2 Carbon sequestration.

Sea level-influenced vegetation shifts alter sequestration of carbon (Kirwan and Mudd, 2012; Morris et al., 2002). C burial rates in creek banks is about 4.7 times the rates measured in high marsh; carbon burial in low marsh is about 1.9 times those recorded in high marsh sediments (Ouyang and Lee, 2014). The stimulation of carbon sequestration associated with stands of *S. alterniflora* is consistent with observations in China and the British Isles (Gao et al., 2012; Ouyang and Lee, 2014).

3.4.3 Foraging and nursery services.

Sea level-related shifts of area of low and high marsh have consequences for consumers that feed and reproduce in creek bank, low marsh, and high marsh habitats.

Loss of creek bank habitat will impair certain species with major ecological and biogeochemical roles within salt marsh ecosystems. Creek bank and low-lying habitats are essential for a number of key marsh species. Fiddler crabs (*Uca pugnax*) play fundamental roles in sediment biogeochemistry and turnover (Wang et al., 2010), and their numerous burrows are most abundant within low-lying marsh sediments, mainly where tall *S. alterniflora* grows (Bertness and Miller, 1984; Krebs and Valiela, 1978). A number of bivalves, including oysters (*Crassostrea virginica*) and ribbed mussels (*Geukensia demissa*) live largely on salt marsh creek banks (Evgenidou and Valiela, 2002). Ribbed mussels by themselves were numerous enough in Great Sippewissett marsh in the 1980s to filter more than the whole volume of tidal exchange of water daily (Jordan and Valiela, 1982), and hence maintained water clarity. The narrow and patchy strips of tall stands of *S. alterniflora* on salt marsh creek banks are the sole nesting habitat of seaside sparrows (*Ammodramus caudacutus*), whose nests are extremely vulnerable to even slight increases in sea level (Gjerdrum et al., 2005) and whose populations are declining (Berry et al., 2015).

Expansion of low marsh at the expense of high marsh favors feeding by *Fundulus heteroclitus* and other fish species, predators that forage within the low marsh canopy during high tide (Werme, 1981), but are unable to penetrate into and feed within the much denser stands of high marsh vegetation (Vince et al., 1976). Sharp-tailed sparrows (*Ammodramus maritimus*) nest in high marsh sites, and nesting success of this vulnerable species is highly susceptible to flooding regimes (Gjerdrum et al., 2005), so that reproduction is reduced by rising sea level (Rush et al., 2009).

3.5 Erosion, transport, and biological effects of changes in sediments.

Beach and dunes along the seaward margin of Great Sippewissett Marsh (Fig. SI 1) receded to the East by a mean of 22 m between 1979 and 2015 (Fig. 3c, and Fig. SI 4). The removed sand, transported by storm over-wash and net tidal exchange into the marsh, seems likely to be responsible for the 82% increase in area of sand-covered bare bottom across the period (Fig. 2c, Fig. 3d, and Table SI 2). This extension of sand cover was synchronous with a 47% decrease in mud-covered bottom (Fig. 2c, Fig. 3d, and Table SI 2).

Continued reduction of beach and dune areas will have effects on a number of species, including nest success and population recovery by piping plover (*Charadrius melodus*) and least tern (*Sterna albifrons*), federally listed species that use these limited habitats as nesting areas. Reduction of nesting habitat changes are therefore of avian conservation concern in the region. The transition of mud to sand bottoms under open water (Fig. 2c,d) should have had other biological effects, such as increased supply of food for consumers entering marshes to feed, since production of benthic invertebrate biomass in sandy bottoms of Great Sippewissett Marsh is 2.5 to 16 times that reported for muddy sediment (Sarda et al., 1995). The shift in benthic sediment cover towards sandy sediments therefore has likely increased food supply for commercially important fish [flounder species, menhaden (*Brevoortia tyrannus*), bluefish (*Pomatomus saltatrix*), and other species], as well as for shrimp and crabs, that use open water habitats within salt marshes as early-life history foraging areas in New England and elsewhere in the world (Hampel et al., 2005; Sá et al., 2006).

3.6 Future of Great Sippewissett Marsh.

The modeled potential future trajectories of the total area and the vegetation mosaic of Great Sippewissett, yielded a forecast of the future of the marsh.

We assumed no changes in vegetated habitats of Great Sippewissett Marsh (creek bank, low marsh, and high marsh) between 1978 and 1992, a period during which platform accretion rates approximately matched long-term sea level rise rates (left side of Fig. 4, whole lines). As sea level rise rates accelerated by about six-fold after 1992, trajectories were set to match measured habitat changes that took place in the marsh until 2015 (right side of Fig. 4, whole lines). After 2015, and as sea level continues to rise, the model anticipates major changes in vegetated habitats (Fig. 4a, dashed lines), and predicts a progressive transformation of the salt marsh into open water areas (Fig. 4b, dashed lines).

With regard to vegetated habitats, modeled trajectories predict that creek banks will continue to slump, and the tall stands of *S. alterniflora* growing on the banks will largely disappear from Great Sippewissett Marsh by about the end of the century. Increased submergence will initially foster expansion of low marsh vegetation to almost twice its original area by 2030 (Fig. 4a). Most of the low marsh expansion will be at the expense of high marsh areas (Fig. 4a). After disappearance of high marsh around 2030, low marsh extent will decrease rapidly as it is further submerged, and has no available elevated areas to invade. If the land margin adjacent to the marsh were not steep or populated and protected from flooding by people, the high marsh could have grown landward, and could have survived considerably longer.

Our simulation did not allow high nor low marsh species to expand onto upland areas because of the coastal "squeeze" against the built environment (such as the embankment supporting the bicycle path to the East of the marsh, Fig. SI 1), and because of the dominant stand of *P. australis* along the landward margin of Great Sippewisset Marsh (Fig. SI 3c). Both low and high marsh will likely be gone by 2100—with an uncertainty of 30-40 years (Fig. 4). Great Sippewissett Marsh will then have largely disappeared, and the coast will show a shallow embayment in its place (Fig. 4b).

A recent evaluation of salt marsh vulnerability to current sea level rise rates showed that, based on the quantification of the marsh's sediment budget, many other North American marshes are likely to disappear in a similar time period (Ganju et al., 2017). According to the calculations of that same study, and with a ratio of open water to vegetated marsh of 0.59 in 2015, the likely lifespan of the marsh complex in Great Sippewissett could be just slightly more than 100 years, a result that matches almost exactly the calculations from our modeled trajectories.

An earlier study of sediment cores taken from Great Sippewissett Marsh suggested that about 2500 years ago, the area that is now Great Sippewissett Marsh included a shallow coastal embayment with a tidal mudflat, a rather narrow salt marsh rim, and some freshwater swamp towards land (Treggor, 1983). That description matches what the model anticipates will be the case after 2100. These conclusions point out that salt marshes such as Great Sippewissett are indeed transient coastal environments at geological scales, with lifetimes of a few thousand years.

3.7 Future trajectories of salt marshes in general.

The future of salt marshes and other coastal wetlands across the world is a fraught with uncertainty; the only guarantee is that major changes are inevitable. Although sea level rise is a major driver of changes (Tabak et al., 2016), the future of salt marshes also depend on the relative magnitude of sea level rise vs. sediment accretion, two variables that differ greatly among coastal wetlands (Fig. 5). The local variability in such variables is reflected in the diverse results reported. Kirwan et al. (2016) reviewed ratios of sea level rise and accretion in salt marshes, and suggested that marsh loss might be overestimated. Other meta-analyses (Blankespoor et al., 2014; US Fish and Wildlife Service, 2012; Watson et al., 2017) concluded that marsh loss may indeed be large.

The comparison of sea level rise vs accretion pointed out many points above and below the 1:1 line that indicates a balance between the two processes. Enumeration of the points above and below the 1:1 line suggested that 40.3% of salt marshes around the world could be fated to be submerged (Fig. 5). The meta-analyses included sites where riverine sediment inputs support accretion, a condition that may not be true of all salt marsh sites, and did not at all consider the widespread occurrence of coastal squeeze effects owing to human settlement of coastal areas (Valiela, 2006). The meta-analyses of Fig. 5 also assumed no further increases in sea level rise. Sea level rise, however, has accelerated during recent decades. Since the early 1990s, current rates of global sea level rise have almost doubled (Chen et al., 2017; Dangendorf et al., 2017; Merrifield et al., 2009). The acceleration will, conservatively, result in a doubling of rise of global sea level by 2100, forcing a rise of 65 cm by 2100, relative to sea level in 2005 (Nerem et al., 2018). Salt marshes, much like other coastal wetlands, are therefore facing a threatening future rise in sea level. To project a probable future, we re-drew the 1:1 line in Fig. 5, to a position likely to exist at the end of this century, assuming that accretion will remain unchanged, but sea level will rise as presently forecasted (Nerem et al., 2018). Under these conditions, by 2100, 94.9% of the 315 salt marsh sites—particularly those where platform accretion is largely through below-ground biomass accumulation—would likely become submerged. Although this oversimplifies conditions and does not account for future changes in accretion, these numbers still suggest that salt marshes are, and will continue to be, a significantly threatened major coastal environment.

The magnitude of changes forced by sea level rise on coastal wetlands is and will be significant, with greater submergence followed by slumping creek banks, less stable sediments, more low marsh, changing un-vegetated sediments, and likely decreased ability of high marsh to migrate landward. Further sea level rise will thoroughly alter the way salt marsh ecosystems convey their important ecological services—stabilization of coastlines, carbon sequestration, provision of wildlife habitats, and other services such as interception of nitrogen (Valiela, 2006).

Sea level rise will have measurable effects on the functioning of these ecosystems, with consequences for ecological services, and will make developing management measures a serious challenge. Moreover, the habitat mosaic and the topographic and hydrodynamic setting of different salt marshes will differ. Predictions of future trajectories in function and services, and planning strategies toward marsh preservation, will therefore have to be locality-specific. There is little doubt, however, that the best and most broadly meaningful preservation plan would include preventive global-scale measures aimed at diminishing sea level rise by decreasing atmospheric warming.

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Figure legends

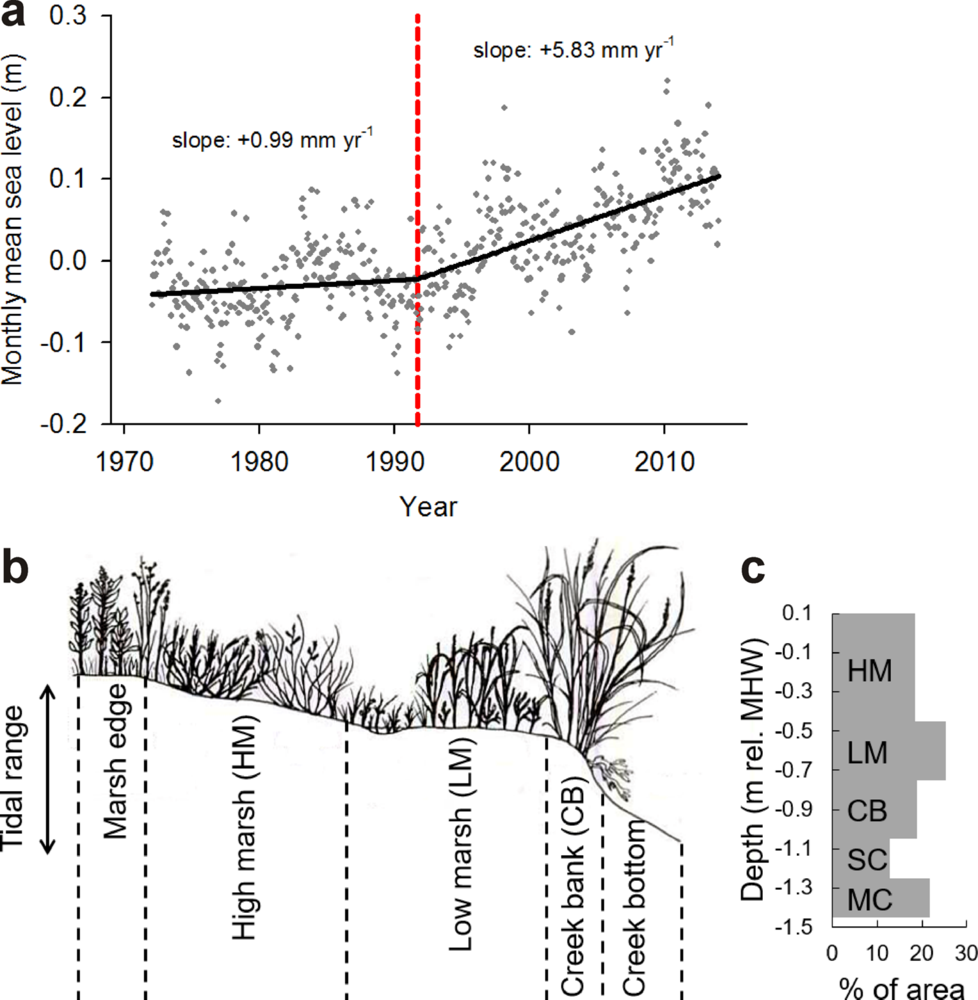
Fig. 1 Recent rates of sea level rise and habitat distribution across the elevation gradient. **a**) Record of sea level rise, 1972-2013, for the Woods Hole, Cape Cod, MA US, station. Plotted values are sea levels relative to NAVD88. Dashed red line represents the hinge year of 1992, when slopes changed significantly according to the piecewise regression analysis of data. **b**) Diagrammatic section through a New England salt marsh, from the upper marsh edge (on the left) to the tidal creek bottom (on the right); figure adapted from Valiela (2015). In sequence, from left to right: the upper tidal margin of the marsh is defined by a narrow band of marsh elder, *Iva frutescens*, and black rush, *Juncus maritima*; the high marsh platform is covered by marsh hay, *S. patens*, with some spikegrass, *Distichlis spicata*; intermediate and short forms of *S. alterniflora* grow on the low marsh platform, with the glasswort, *Salicornia europaea* growing in temporary open gaps; tall form of cordgrass, *S. alterniflora* occurs on creek banks—with a limited understory of brown algae; at the lower limit of the tidal range, there are un-vegetated bare sediments. **c**) Elevation range for high marsh, low marsh, and creek bank vegetation, and for un-vegetated areas covered by sand (SC) or mud (MC) sediments; from Valiela and Rietsma (1995).

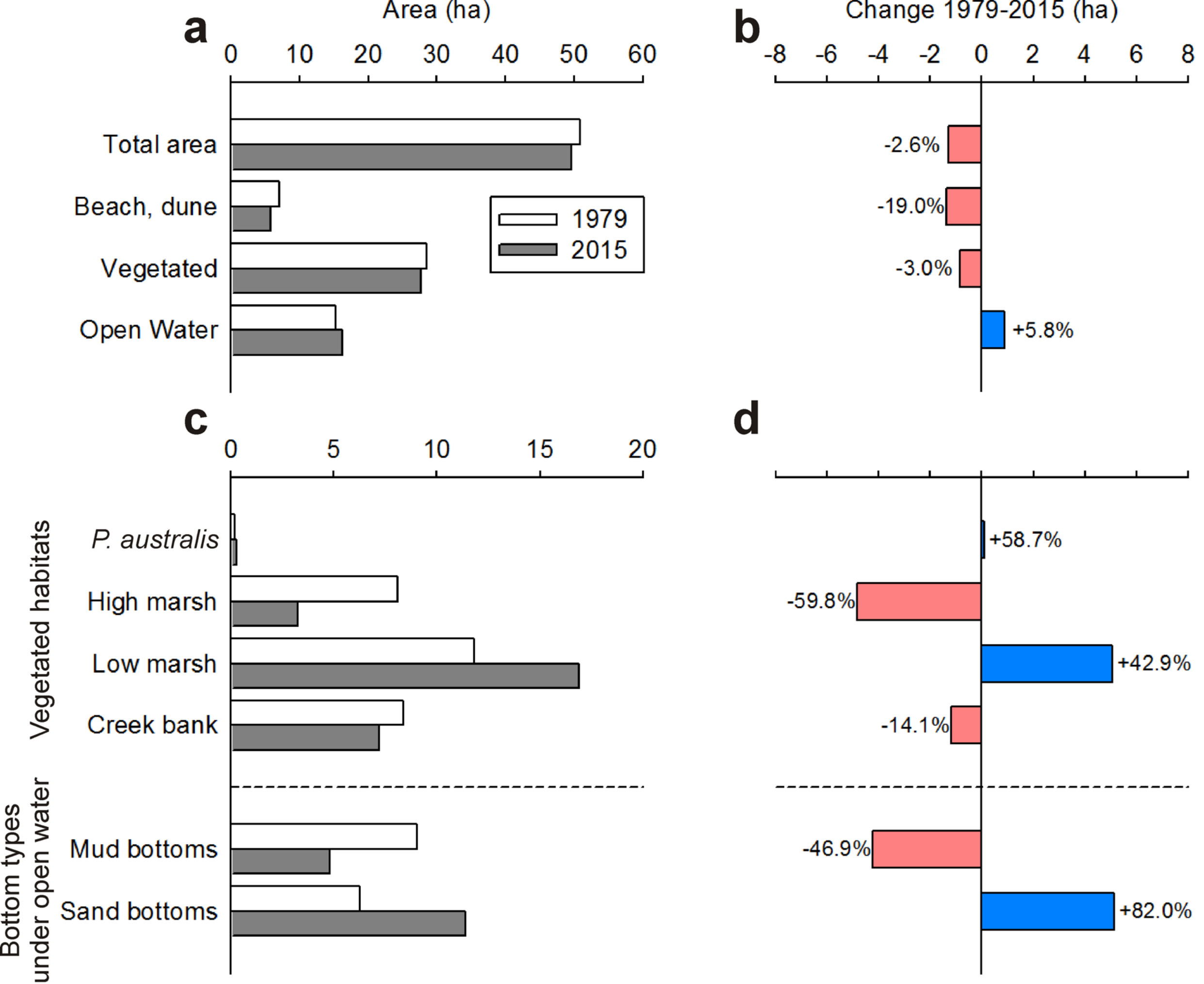
Fig. 2 Changes in areas of major habitats of Great Sippewissett Marsh between 1979 and 2015. **a**) Areas±s.e. of total marsh and major habitats. **b**) Percent changes between 2015 and 1979, for the same categories. **c**) Area±s.e. of major vegetation types. **d**) Percent changes between 2015 and 1979, for the same categories, where blue shows gains, and red shows losses.

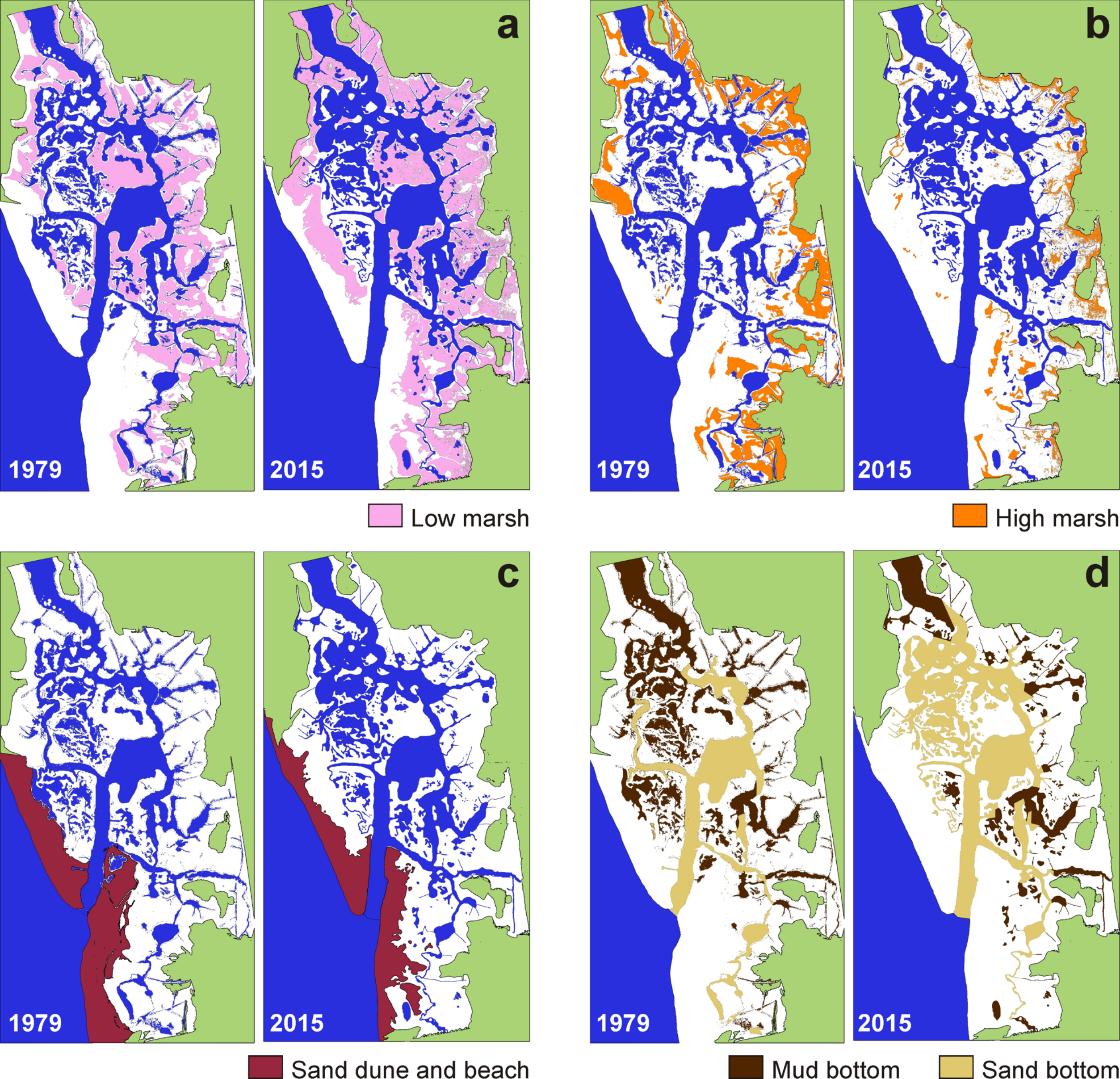
Fig. 3 Spatial changes in major habitats of the Great Sippewissett Marsh between 1979 and 2015, for **a**) low marsh, **b**) high marsh, **c**) beaches and dunes, and **d**) un-vegetated mud and sand sediments.

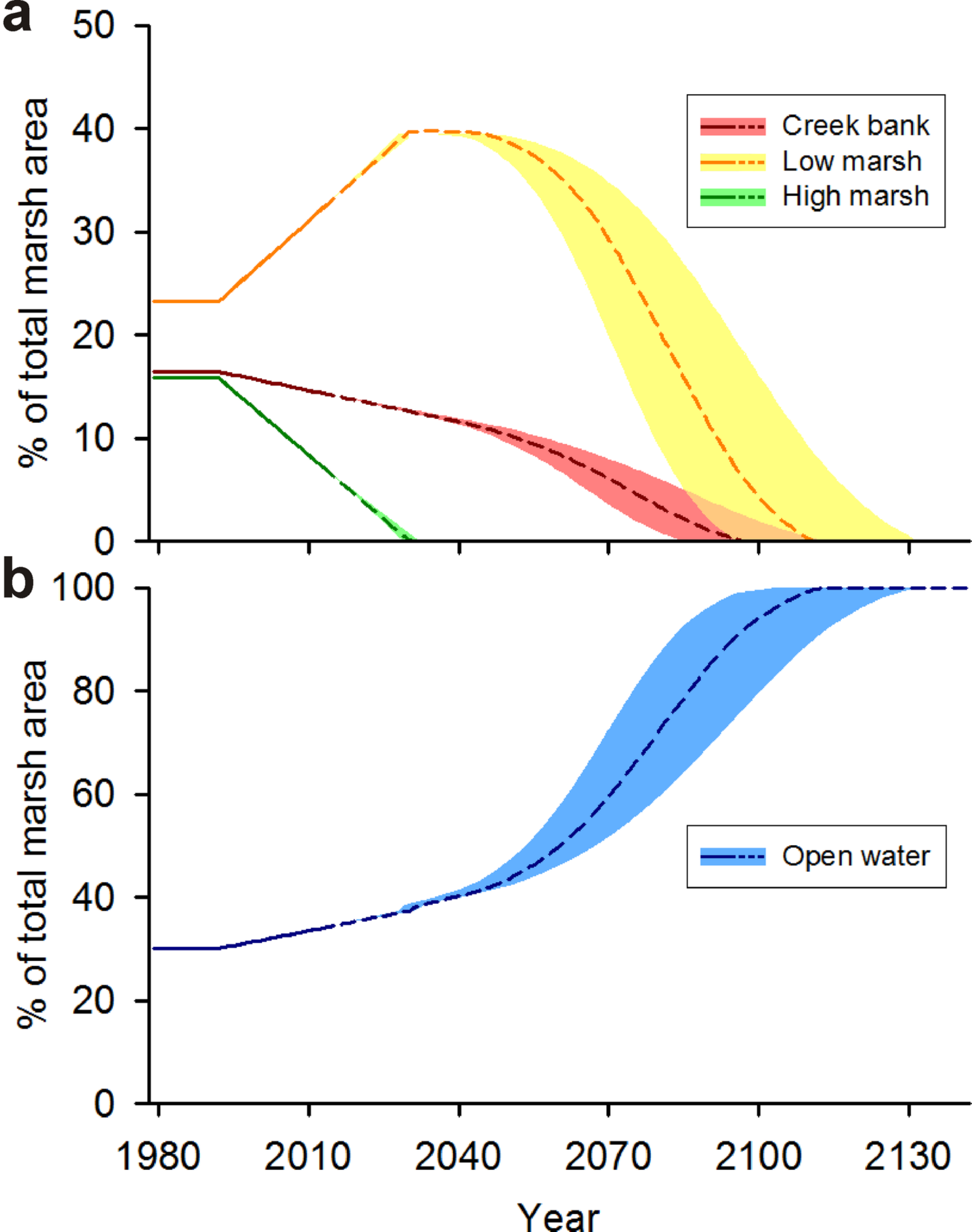
Fig. 4 Modeled trajectories of Great Sippewissett Marsh for **a**) vegetated habitats, and **b**) open water area. Values expressed as percentages of total area of the marsh in 1979. Solid lines represent changes in areas between 1979 and 2015. Dashed lines represent modeled changes in areas into the future. Shaded areas represent the propagated uncertainty associated with the estimated areas of each habitat.

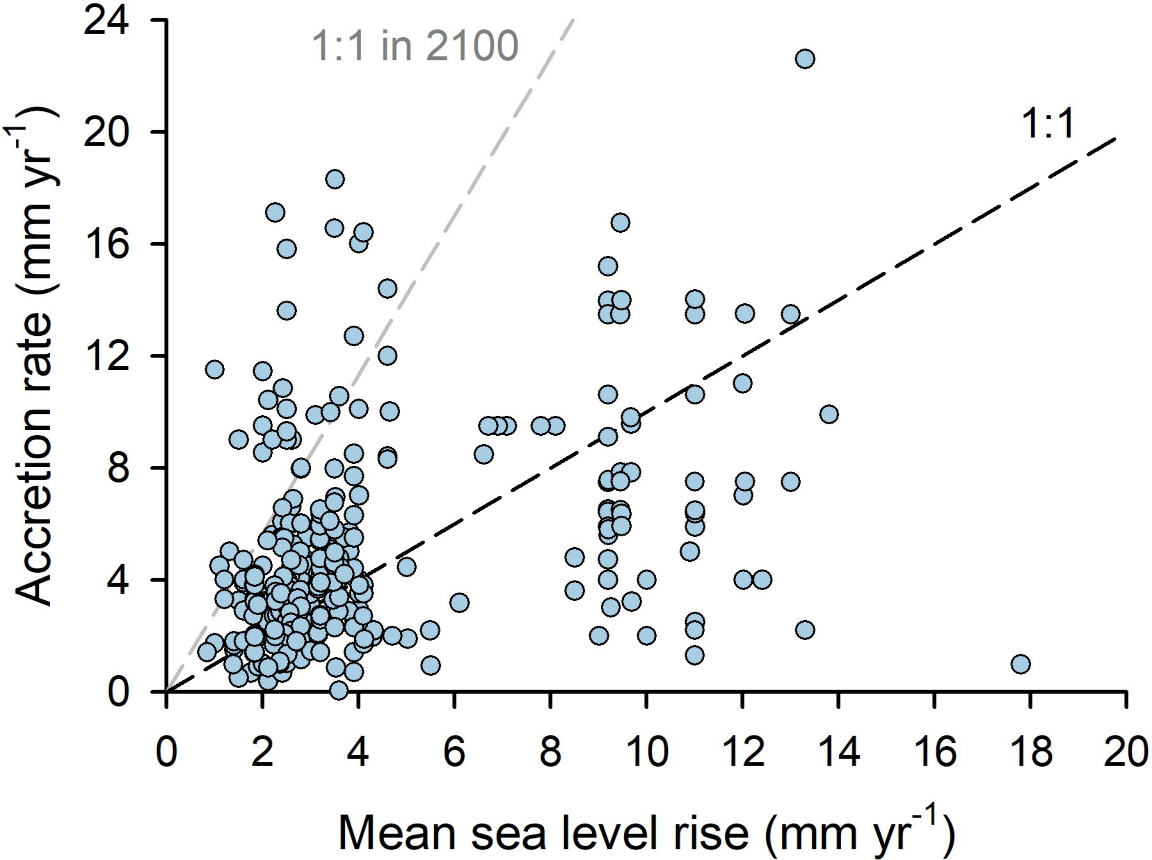
Fig 5 Sediment accretion rates versus mean sea level rise in salt marshes around the world. The graph includes data from 315 locations. Data from compilations in Crosby et al. (2016), Kirwan et al. (2016), Raposa et al. (2016), and Valiela (2006). The black dashes show the 1:1 line indicating agreement between accretion and sea level rise during recent historical times. The grey dashes show the same agreement line, modified to show position in 2100, assuming no rise in accretion, and increased sea level to levels predicted for the end of this century.


a
0.3
0.2
0.1
0.0
-0.1
-0.2
Monthly mean sea level (m)
slope: +0.99 mm yr-1
slope: +5.83 mm yr-1
1970
1980
1990
2000
2010
Year
b
Tidal range
Marsh edge
High marsh (HM)
Low marsh (LM)
Creek bank (CB)
Creek bottom
c
0.1
-0.1
-0.3
-0.5
-0.7
-0.9
-1.1
-1.3
-1.5
Depth (m rel. MHW)
HM
LM
CB
SC
MC
0
10
20
30
% of area

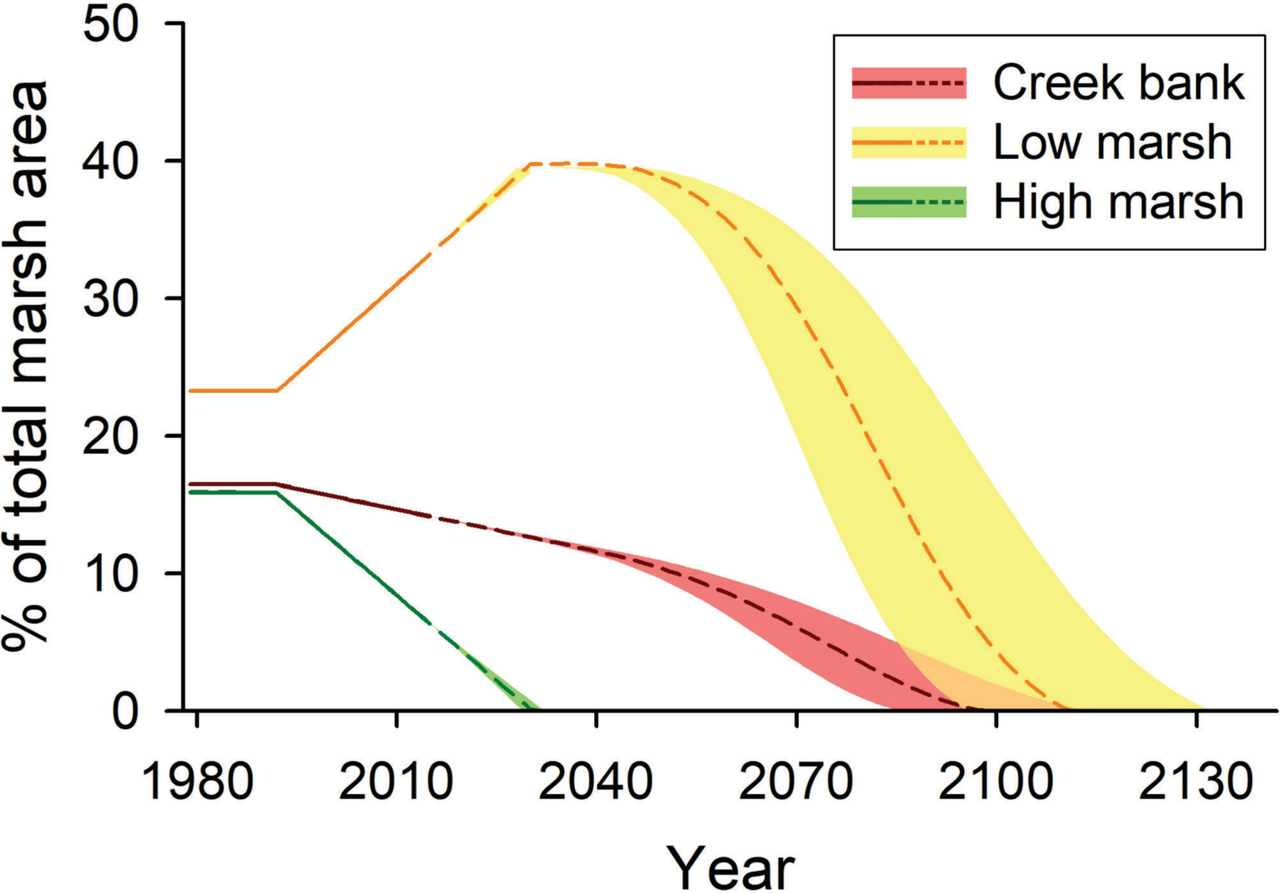

a
Area (ha)
0
10
20
30
40
50
60
Total area
Beach, dune
Vegetated
Open Water
1979
2015
b
Change 1979-2015 (ha)
-8
-6
-4
-2
0
2
4
6
8
-2.6%
-19.0%
-3.0%
+5.8%
c
0
5
10
15
20
Vegetated habitats
P. australis
High marsh
Low marsh
Creek bank
Bottom types under open water
Mud bottoms
Sand bottoms
d
+58.7%
-59.8%
+42.9%
-14.1%
-46.9%
+82.0%


a
1979
2015
Low marsh
b
1979
2015
High marsh
c
1979
2015
Sand dune and beach
d
1979
2015
Mud bottom
Sand bottom


a
b
50
40
30
20
10
0
100
80
60
40
20
0
% of total marsh area
% of total marsh area
Creek bank
Low marsh
High marsh
Open water
1980
2010
2040
2070
2100
2130
Year



1:1 in 2100
1:1
Accretion rate (mm yr^{-1})
0
4
8
12
16
20
24
0
2
4
6
8
10
12
14
16
18
20
Mean sea level rise (mm yr^{-1})



50
40
30
20
10
0
% of total marsh area
1980
2010
2040
2070
2100
2130
Year
Creek bank
Low marsh
High marsh