Potential Effects of Sea-Level Rise on Salt Marsh Elevation Dynamics in a New Hampshire Estuary

Andrew R. Payne[1] · David M. Burdick[1] · Gregg E. Moore[2]

Abstract

Salt marsh survival in the face of sea-level rise (SLR) depends largely on a marsh's ability to compensate for increased flooding by building in elevation, but the rate of elevation gain depends on processes that are not well-understood (i.e., belowground productivity, sediment accretion, and subsidence). An array of planted and unplanted pots was installed in the field to examine the effects of tidal flooding on productivity and elevation change. We found that belowground plant volume increased linearly as elevation increased for both *Spartina patens* and *Spartina alterniflora*. Even though the volume of roots increased by 400% with elevation for *S. alterniflora* and > 200% for *S. patens*, there was no relationship between belowground volume and elevation change of the original soil surface, perhaps due to the infilling of porosity by roots. However, the soil in planted treatments subsided significantly less than in unplanted controls. Measurements from Surface Elevation Tables (SETs) indicated that local high marshes have been losing elevation relative to sea level at an average rate of 2.1 mm/year. The rate of vertical gain decreased at SET locations of greater marsh elevation, and high marshes did not keep pace with SLR even when sediment supply appeared to be high (TSS = 57 ± 7 mg/L). The high marsh accretion deficit, combined with a continuing trend of reduced belowground growth with increased flooding due to SLR, suggests that *S. alterniflora* will replace less flood-tolerant species over time, potentially causing loss of high marsh habitat.

Keywords Accretion · Marsh organ · Root volume · *Spartina alterniflora* · *Spartina patens* · Subsidence

Introduction

Salt marshes are valued for their ability to sequester carbon, attenuate coastal flooding, improve water quality, and provide habitat for fish and wildlife (Costanza et al. 1997; Barbier et al. 2011), but the survival of many marshes and their ecosystem services are threatened by climate change (reviewed in

✉ Andrew R. Payne
apayne870@gmail.com

[1] Department of Natural Resources and the Environment, Jackson Estuarine Laboratory, University of New Hampshire, 85 Adams Point Rd., Durham, NH 03823, USA

[2] Department of Biological Sciences, Jackson Estuarine Laboratory, University of New Hampshire, 85 Adams Point Rd., Durham, NH 03823, USA

Short et al. 2016). Marshes can be thought of as poised systems that depend on the balance between sea level rise (SLR) and marsh elevation gain through accumulation of sediment and organic matter. An increase in SLR due to global warming (Nicholls and Cazenave 2010) has altered this balance in many marshes, resulting in increased flooding and conversion of high marsh to low marsh and low marsh to mudflat when plant tolerances are exceeded (Watson et al. 2017a). Past salt marsh loss from direct human impacts is estimated at 37% in New England (Bromberg and Bertness 2005), but potential sea level rise of 1–2 m by 2100 (Vermeer and Rahmstorf 2009) may result in the loss of all marshes except those with high sediment loads and large tide ranges (Kirwan et al. 2010). To predict and manage the potential impacts of SLR, a better understanding is needed of how increased flooding will affect the complex processes controlling marsh elevation change.

Marshes in northern New England are characterized by thick accumulations of peat up to 4000 years old (Kelley et al. 1995). The low marsh is composed almost entirely of *Spartina alterniflora* while the high marsh is more diverse, consisting of *Spartina patens*, *Distichlis spicata*, *Juncus gerardii*, and less-abundant forbs and shrubs. The large tide

range in northern New England allows marshes to span a wide range of elevations and may increase resilience to SLR for those marshes with high "elevation capital" (marsh surface elevation relative to the lower limit of plant growth; Cahoon and Guntenspergen 2010). However, the more flood-tolerant *S. alterniflora* is beginning to replace high marsh species in some New Hampshire marshes as sea levels rise (Burdick unpub. data), potentially leading to a loss of species diversity. Additionally, resilience in New England marshes may be hindered by two factors that limit elevation gain: generally low sediment supply (Chapman 1960; Weston 2014) and lower productivity than southern marshes (Seliskar et al. 2002).

Elevation change is determined by the balance between processes that decrease elevation (subsidence from compaction and decomposition) and those that build it (accumulation of organic matter and sediment). Sediment accretion increases as flooding increases (Cahoon and Reed 1995), acting as a feedback on SLR where greater flooding results in greater elevation gain. However, many studies show accumulation of organic matter correlates more strongly with elevation gain (Bricker-Urso et al. 1989; Callaway et al. 1997; Nyman et al. 2006; Boyd and Sommerfield 2016). The main source of organic matter is marsh plant productivity (Wang et al. 2003), which may either increase or decrease with greater flooding, depending on marsh elevation and plant species (Morris et al. 2002; Watson et al. 2014; Kirwan and Guntenspergen 2015). While belowground productivity can directly build elevation by adding volume to the soil (Langley et al. 2009), the net effects of flooding on soil processes are not well-understood. A lab mesocosm study showed that increased flooding results in greater elevation gain through swelling of the soil and potentially slower decomposition of soil organic matter (Cherry et al. 2009), but relationships between flooding, productivity, and decomposition may differ under field conditions.

The rate of marsh elevation change is commonly measured by Surface Elevation Tables (SETs) paired with feldspar Marker Horizons (MHs) to determine the net effects of both aboveground and belowground processes (Cahoon et al. 1995; Anisfeld and Hill 2012; Raposa et al. 2016). SET measurements indicate that the rate of elevation gain is less than 1/3 the rate of relative sea level rise (RSLR) in Rhode Island marshes (Raposa et al. 2016). Although RSLR has been lower in New Hampshire due to post-glacial adjustment of Earth's crust (Karegar et al. 2016), transition of high marsh to low marsh at some locations suggests that elevation change is not keeping pace (Burdick unpub. data).

Since it is unknown how the processes controlling marsh elevation will be impacted by SLR regionally, we examined the effect of flooding on plant productivity, sediment accretion, and subsidence. We used a "marsh organ" experiment (Morris 2007) to isolate the effects of flooding rate on plant productivity. In a novel application of the marsh organ method, we examined the relationship between belowground plant growth and elevation change. Additionally, we used SETs to examine the rate of marsh-building at different elevations and to determine whether marsh elevation gain was keeping pace with SLR locally. The goal of this research was to better understand feedbacks associated with increased flooding and to contribute to regional assessments of marsh vulnerability to sea level rise, allowing managers to focus resources on marshes most appropriate to achieve management objectives.

Methods

Study Sites

The marsh organ experiment was located at Great Bay Farms (GBF), a riverine marsh roughly 2.7 ha in area in the southeast corner of the Great Bay Estuary, New Hampshire (Fig. 1). We chose this marsh because it is low-energy, relatively secluded, and contains SET-MH stations that allowed us to pair marsh organ data with elevation data from the marsh platform. The marsh vegetation spans an elevation range of about 1.2 m, with *S. alterniflora* dominating the low marsh, and *S. patens* and *D. spicata* dominating the high marsh along with less abundant species such as *J. gerardii*, *Solidago sempervirens*, and *Triglochin maritima*. MH transects were located at GBF and Crommett Creek (CRC), a submerging riverine marsh on the western side of Great Bay. SET-MH stations were located at GBF and Sandy Point, both part of the Great Bay National Estuarine Research Reserve (GBNERR).

Marsh Organ Experiment

Two marsh organ arrays were constructed to determine the effect of flooding rate on plant productivity and surface elevation change. A third organ was built as part of a separate study but held unplanted control pots that were used for CO_2 measurements. Marsh organs consisted of a wooden staircase structure that held four replicate pots at five elevations for each species (Fig. 2). The lower three elevations of *S. alterniflora* had two unplanted control pots each while the upper two elevations had four (also used as controls for *S. patens* comparisons). Pots were made of 10-cm diameter PVC pipes cut to lengths of 40 cm and capped on the bottom. Five holes were drilled in each cap, and two holes were drilled in the sides of the pots 20 cm down to allow for both vertical and horizontal drainage. The holes were covered with landscape fabric to prevent soil loss. An additional three holes were drilled at 1, 2, and 3 cm from the top to allow water to drain from the soil surface.

Pots were filled with a mixture by volume of 45% sand, 40% peat moss, and 15% mud collected from a nearby mudflat. We determined this ratio by experimenting with different amounts of materials until we found a mixture that allowed
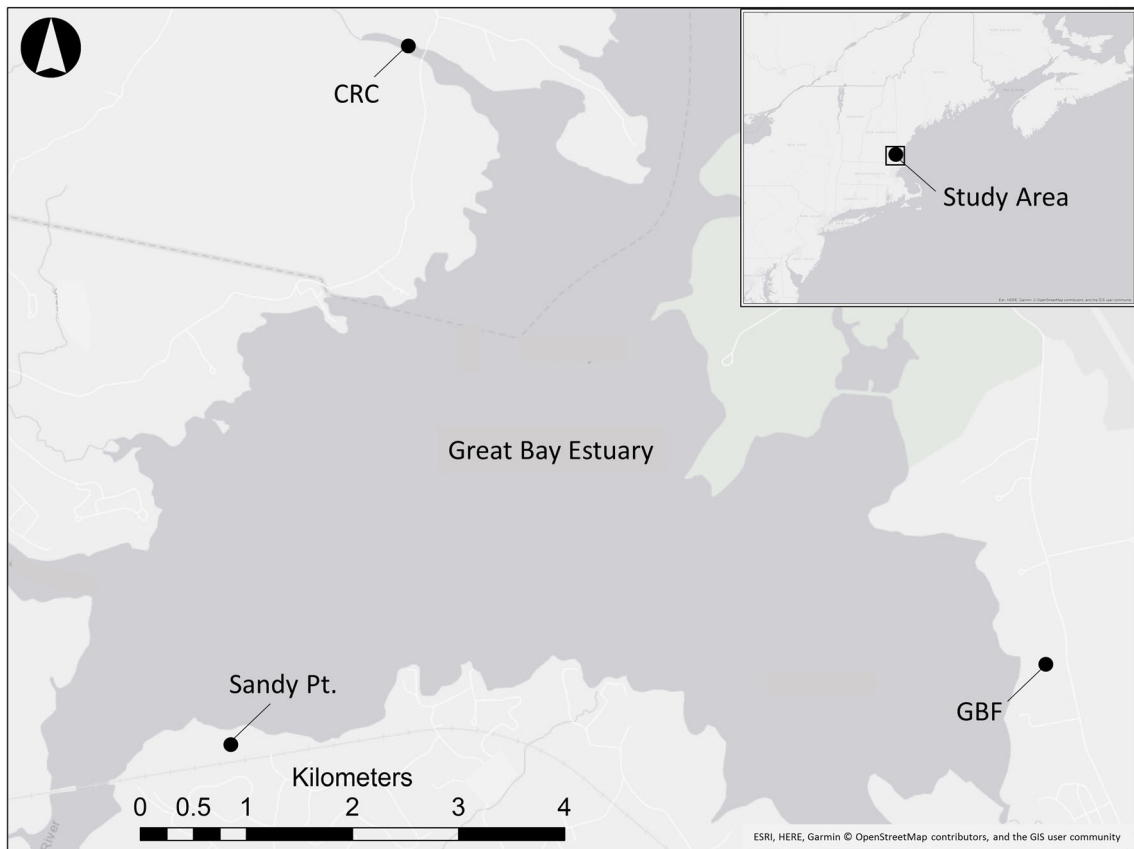
Fig. 1 Map of the study sites in the Great Bay Estuary, NH. Great Bay Farms (GBF) = site of marsh organ experiment, MH transects, and 3 SET-MH stations; Sandy Point = 5 SET-MH stations; Crommett Creek (CRC) = MH transects

water to drain from the surface over a 6-h period. Peat moss was chosen as a source of organic matter because it could be relatively easily washed from roots for biomass analysis, and

we believed that the rate of decomposition would be slow and similar to that of marsh peat. Pots were placed in a tank that was flooded and drained daily with salt water for 5–7 days to

Fig. 2 Diagram of the marsh organ set-ups with the *S. alterniflora* structure on the left and *S. patens* on the right. Color bands show elevation ranges of habitat types on the adjacent marsh
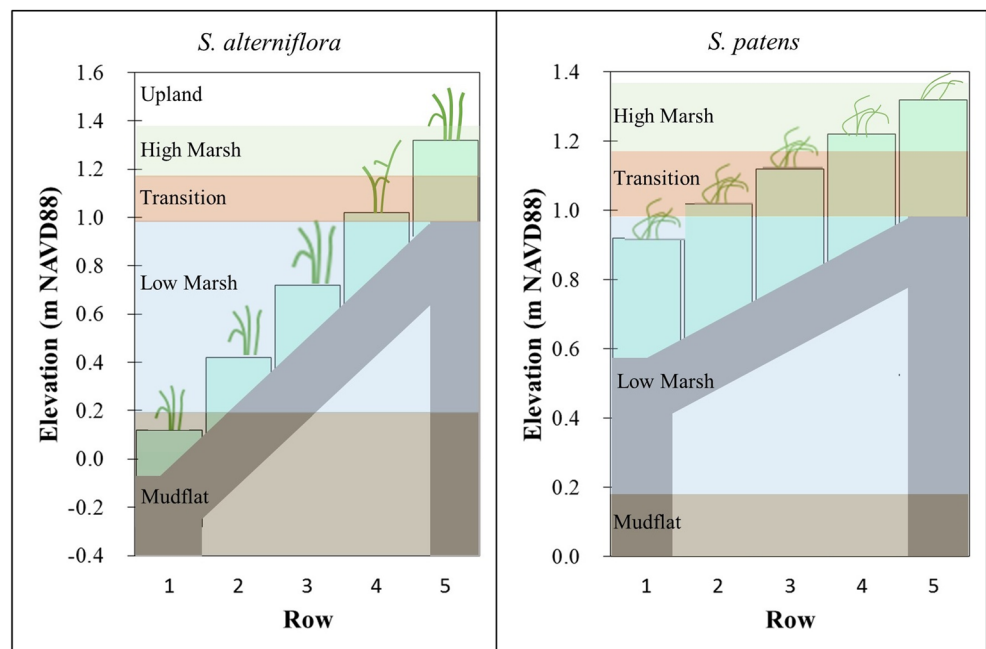
allow the soil to compact before planting. *Spartina* were sourced locally, and roots were rinsed of soil in salt water before planting bare-root culms in pots; control pots were left unvegetated. We kept aboveground biomass similar in each pot by planting 4–5 culms/pot for *S. alterniflora* and 8–9 culms/pot for *S. patens.* The average initial plant height (measured to the tip of the tallest leaf) was 24 ± 0.7 cm for *S. alterniflora* and 17 ± 0.6 cm for *S. patens.* Marsh organs were deployed in a tidal creek at GBF marsh in May 2017. The structures were oriented to maximize southern exposure (150° magnetic North) while taking into account the shape of the creek. To ensure marsh organs were deployed at the correct elevations, they were related to a benchmark elevation determined by a Leica GSSN Rover model GS14 Real Time Kinematic (RTK) GPS with $< \pm 1.5$-cm accuracy. Pots were transported to the marsh 1–2 days after planting, and randomly assigned numbered pots were fastened to the marsh organ. Two weeks after deployment, each pot was evaluated, and any dead plants were replaced to ensure that total biomass remained similar between pots.

Marsh Organ Soil Measurements

To isolate subsurface elevation change from sediment accretion, we created MHs by pinning two circles (diameter = 3 cm) of landscape fabric to the soil surface of each pot. We measured the change in height of MHs and the overall surface elevation change using a mini-SET that consists of a flat strip of metal holding 5 pins (modified from Cherry et al. 2009; Fig. 3). To measure soil height change, the mini-SET was placed on the rim of the pot and pins were lowered to the soil surface. The length of each pin was measured to determine the distance between the soil surface and the rim of the pot. The mini-SET was then rotated 90°, and the procedure was repeated for a total of 10 measurements/pot. Notches were cut into the rim of each pot marking the placement of the mini-SET to ensure consistency between measurements. To measure subsurface height change, the two outer pins on the mini-SET were replaced with knitting needles, allowing us to penetrate the deposited sediment and measure the height of buried MHs relative to the rim of the pot. Marsh organ mini-SET and MH measurements were taken in early June and mid-September of 2017. Sediment accretion was determined by subtracting the subsurface height change from the soil height change. To accommodate unexpectedly high sediment accretion that accumulated above the rim of low-elevation pots, we extended the pot height when taking measurements using a PVC collar that fit firmly on the top of the pot.

In this study, sediment that accumulated above MHs is termed *sediment accretion*. We use the term *subsurface height change* to describe the change in height of the original soil surface from belowground processes (i.e., belowground plant growth and subsidence). *Soil height change* refers to surface
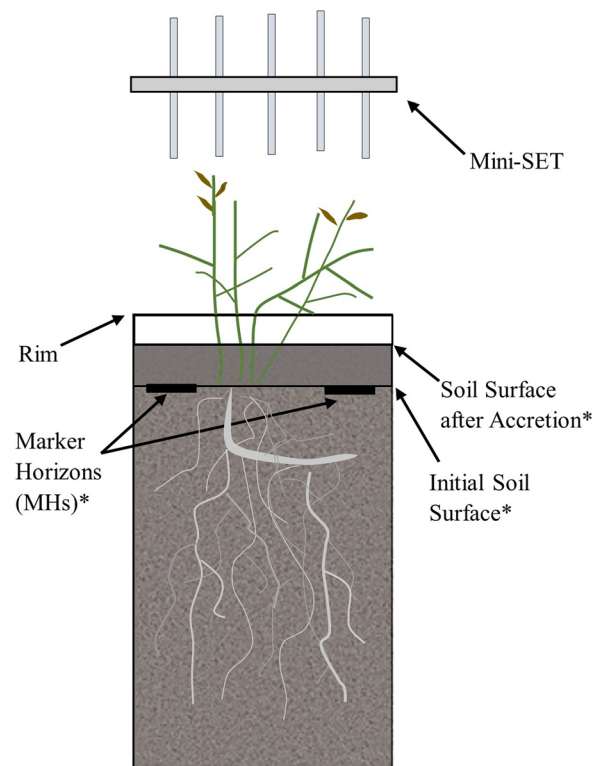
Fig. 3 Diagram showing mini-SET and marsh organ pot set-up

elevation change in marsh organ pots measured using the mini-SET. Hence, soil height change reflects both accretion of new sediment and soil compaction or expansion due to belowground processes. Since flooding frequency and duration decrease as elevation increases, the terms elevation and flooding are used interchangeably.

Productivity and CO_2 Measurements

In mid-September, plant height was recorded as the distance between the soil surface and the tip of the longest leaf when held vertically. Plants were clipped, rinsed, and oven-dried for ≥ 2 days at 70 °C before biomass was recorded in grams of dry weight. To measure soil respiration, the clipped shoot bases were filled with silicone to limit gas exchange through the shoots (Wigand et al. 2016), and CO_2 emission was measured in each pot at the lab. Measurements were taken over 5 min using a portable greenhouse gas analyzer with a dark chamber attachment that fit securely over the pot (Los Gatos Research Model# 915-0011). Pots were then stored at 5 °C until belowground biomass could be assessed. After removing soil and dead roots, we measured belowground plant volume using a

gravimetric water displacement technique based on Archimedes Principle (Harrington et al. 1994; Ford et al. 1999). Briefly, a water-filled graduated cylinder is placed on a scale with a tube held inside using a ring stand. Belowground material is inserted into the tube, and the change in mass shown on the scale is equal to the volume of water displaced. We chose to focus on belowground volume as a metric because we believed it would more directly relate to elevation change. Belowground biomass was also measured after drying to constant weight at 70 °C.

Marsh Surface Elevation Change

To determine whether Great Bay marshes are keeping pace with sea level rise, SET-MH stations were installed in high marsh areas at GBF (three SETs in 2011) and Sandy Point (four SETs in 1994 and one in 2013). SETs consist of a benchmark pipe or rod driven into the marsh to the point of refusal, combined with a horizontal attachment that holds a series of pins (Cahoon 2015). The pins are gently lowered to the marsh surface to track elevation changes. At each SET station, two 0.09-m^2 plots of feldspar (750 mL) served as MHs where the thickness of sediment that accreted above the feldspar was measured yearly (although some years were missed). MHs show only sediment accretion whereas SETs also incorporate the effects of belowground processes such as root growth and shallow subsidence.

To compare the rate of marsh elevation change to RSLR, water level data were obtained from NOAA at https://tidesandcurrents.noaa.gov/. The rate of RSLR was determined using simple linear regression of monthly Mean High Water (MHW) beginning in 1993, when the current higher rate of SLR began (Nicholls and Cazenave 2010). MHW was used because it is the closest tidal datum to our SET elevations. Rates of RSLR from Portland, ME, and Boston, MA, were averaged because they are the closest tide gauges to our SETs and are roughly equidistant.

Rapid sediment accumulation in the lower marsh organs prompted us to add a field study to quantify sediment accretion on the marsh. In August 2017, three transects of feldspar MHs were installed at GBF surrounding the marsh organs to determine the relationship between elevation and sediment accretion rate. MHs were placed at 15-cm elevation increments ranging from 0.37 to 1.12 m NAVD88. We followed a similar procedure for the submerging CRC marsh, allowing us to compare accretion in two Great Bay marshes with suspected differences in sediment supply. Measurements of accretion above the feldspar layer were taken in December 2017 and May 2018. To measure total suspended solids (TSS), water samples were collected near MH transects at both sites half a meter below the surface during both flood and ebb tides (GBF: $n = 16$, CRC: $n = 8$). The samples were filtered using 1.5-μm pre-weighed glass fiber filters, and the mass of sediment was determined from oven-dried filters (70 °C).

Data Analysis

Relationships between elevation and plant responses, accretion rate, and CO_2 emission were analyzed using simple linear regression (see online supplementary material for linear equations). CO_2 data from four *S. alterniflora* pots were omitted from analysis because the chamber was not properly sealed during measurement. ANOVA was used to test the difference in subsidence between planted and unplanted pots, and a *t* test was used to compare TSS between GBF and CRC. Data were log transformed to meet assumptions of parametric tests for the following responses: *S. patens* stem density, *S. alterniflora* belowground volume, subsurface height change (*S. patens*, *S. alterniflora*, and controls), control CO_2 emission, and marsh surface elevation change. All analyses were performed in JMP Statistical Analysis Software, and means are reported ± standard error.

Results

## Plant Responses to Flooding

There was a strong relationship between belowground biomass and volume for both species (*S. alterniflora* $r^2 = .963$, $p < .0001$; *S. patens* $r^2 = .954$, $p < .0001$), although the slope appeared slightly steeper for *S. alterniflora* (Fig. 4). Belowground volume increased as elevation increased for both *S. alterniflora* ($r^2 = .788$, $p < .0001$) and *S. patens* ($r^2 = .818$, $p < .0001$; Fig. 5). *S. alterniflora*, which has considerably larger diameter roots and rhizomes, showed fourfold higher belowground volume than *S. patens* for overlapping elevations.

Overall, aboveground vegetation responses to elevation were not as strong as belowground, but still showed significant effects. Plant height decreased linearly with elevation for *S. patens* ($r^2 = .305$, $p < .05$), but formed a parabolic relationship for *S. alterniflora*, with the tallest plants growing at middle elevations ($r^2 = .400$, $p < .05$; Fig. 6). In contrast, biomass increased linearly with elevation for both *S. patens* ($r^2 = .264$, $p < .05$) and *S. alterniflora* ($r^2 = .398$, $p < .01$). There was also a significant positive linear relationship between elevation and number of stems for *S. alterniflora* ($r^2 = .602$, $p < .0001$), but not for *S. patens* ($r^2 = .167$, $p = .07$).

CO$_2$ Emission

The relationship between elevation and soil respiration differed between *S. alterniflora* and unplanted controls. Overall, the rate of CO_2 emission was significantly higher
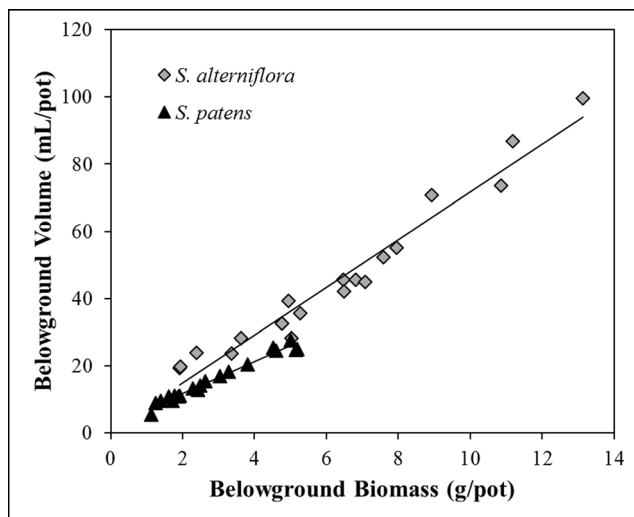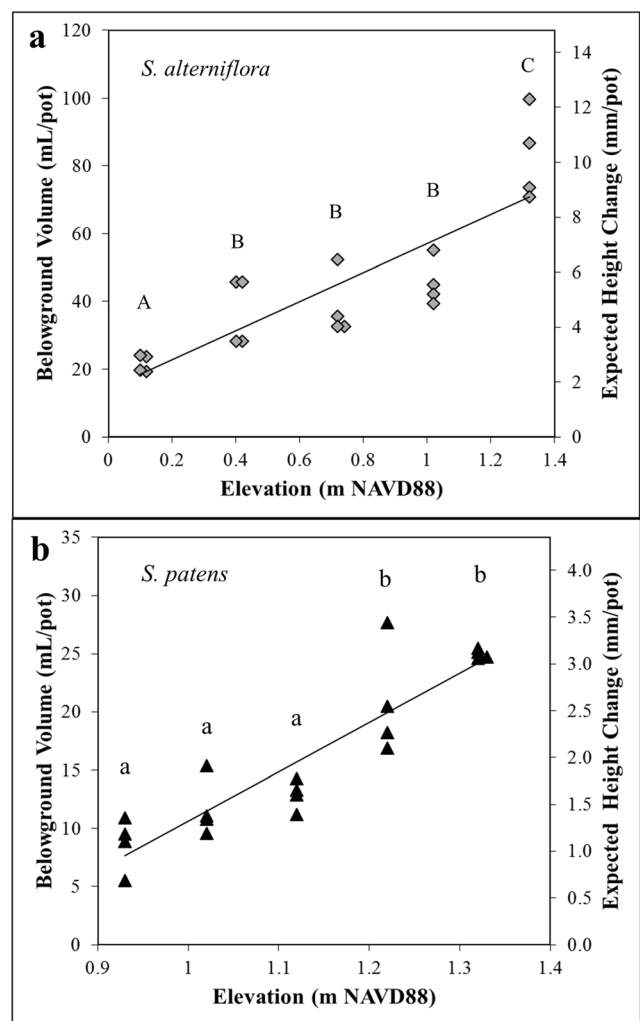
Fig. 4 Belowground volume as a function of belowground biomass (dry wt.) for *S. alterniflora* ($r^2 = .957$, $p < .0001$) and *S. patens* ($r^2 = .974$, p < .0001)

for *S. alterniflora* than for controls ($p < .0001$; Fig. 7a). There was a significant effect of elevation on CO_2 emission for *S. alterniflora* pots ($r^2 = .784$, $p < .0001$), but not for controls ($p = .100$). CO_2 emission increased in *S. alterniflora* pots with elevation, with the largest increase between the lowest two elevations. The relationship was similar between belowground biomass and CO_2 emission ($r^2 = .797$, $p < .0001$; Fig. 7b). *Spartina patens* showed no trends in CO_2 emission (not shown) relative to elevation or biomass, but these data appeared unreliable due to periodic CO_2 spikes.

## Soil Elevation Responses to Flooding and Plant Growth

MHs showed that soil in the root zone subsided across all elevations and pot types. Subsidence in *S. alterniflora* pots was significantly lower than in unplanted controls ($p < .01$; Fig. 8). The same was true for *S. patens* pots that shared elevations with controls ($p < .01$). Despite the effect of plants in reducing subsidence, there was no relationship between belowground volume and subsurface height change for either species. Subsurface height change was negative for all elevations with an average of $-3.0 \pm .84$ mm for *S. alterniflora* and $-2.2 \pm .33$ mm for *S. patens*. There was no significant trend between elevation and subsidence for controls, but there was more subsidence at the lowest elevation than at the highest elevation ($p < .05$). Sediment accretion greatly outweighed subsidence for lower elevations, resulting in a positive net soil height change. However, sediment accretion was low at higher elevations, resulting in an overall decrease in soil height for upper *S. patens* elevations (1.22 m and 1.32 m) and the uppermost *S. alterniflora* elevation (1.32 m).

Fig. 5 Belowground volume as a function of elevation for **a** *S. alterniflora* ($r^2 = .788$, $p < .0001$) and **b** *S. patens* ($r^2 = .817$, $p < .0001$). The secondary axis shows expected change in soil height based on belowground volume input. Different letters denote significant differences between elevations determined by Tukey's HSD test

Local Sediment Accretion and Elevation Change

Sediment accretion in the natural marsh followed a similar pattern to the marsh organs, with the highest rates recorded at lower elevations along the creek edge (Fig. 9). Accretion decreased as elevation and distance from the creek increased for both sites. The first measurement in December 2017 showed no difference in sediment accretion between GBF and CRC, despite significantly higher TSS at GBF (57.5 ± 6.9 mg/L vs. 25.5 ± 4.4 mg/L at CRC, $p < .001$), but the second measurement in May 2018 showed significantly higher rates of accretion at GBF relative to the percent of time flooded ($p < .001$; Fig. 9c). Three plots at GBF and two plots at CRC could not be relocated in May, perhaps due to erosion or winter ice scour. Although they were measured at different times, accretion on the natural marsh was about half that found in the marsh organs.
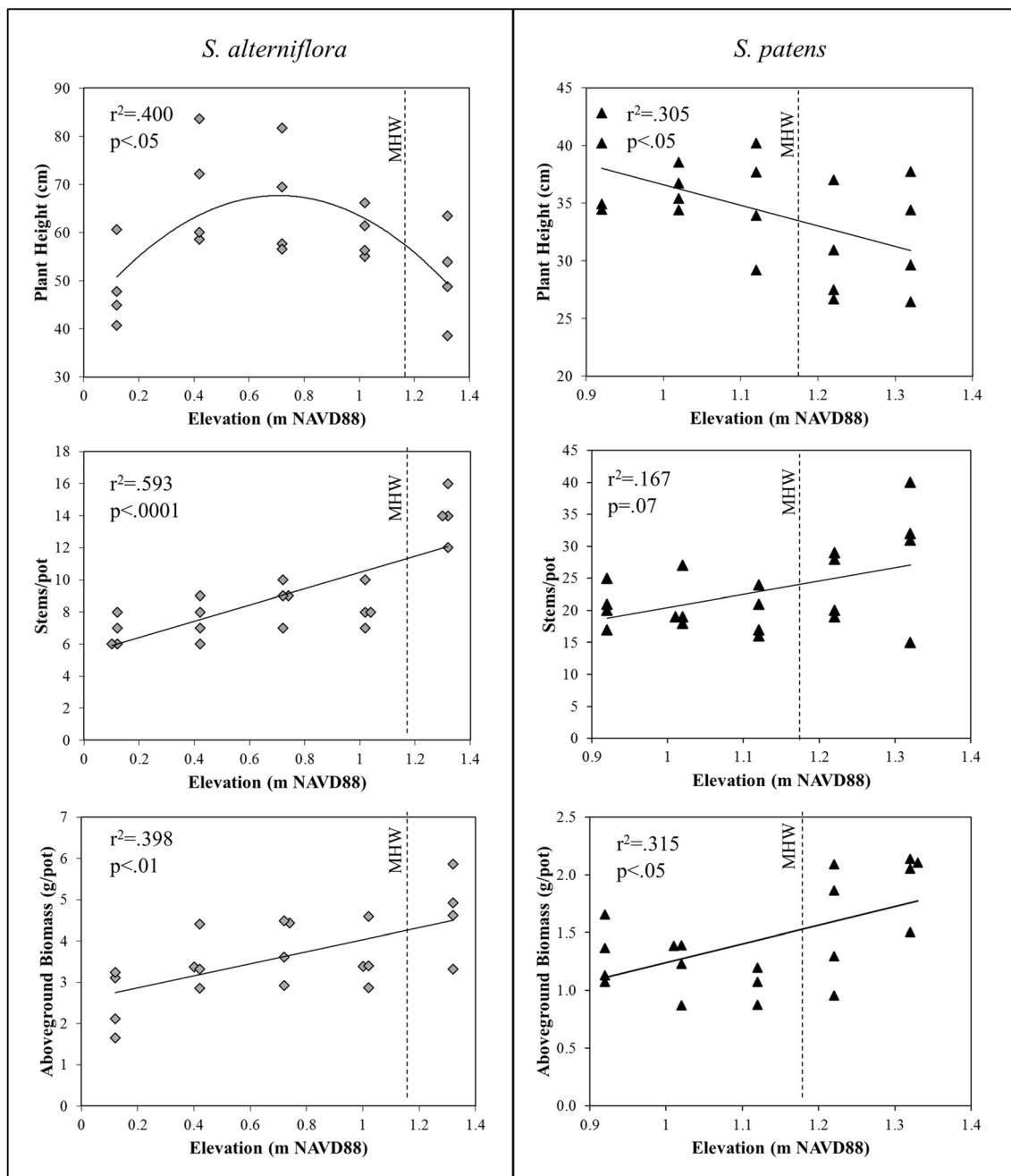
Fig. 6 Aboveground responses as a function of elevation for *S. alterniflora* on the left and *S. patens* on the right. Vertical dashed line shows mean high water

The rate of surface elevation gain depended on the initial elevation of the marsh. SETs showed the rate of elevation gain increased as initial marsh elevation decreased ($r^2 = .658$, $p < .05$; Fig. 10). Sediment accretion at SET stations showed a similar pattern with elevation, but the relationship was not significant ($r^2 = .182$, $p = .29$). Surface elevation gain was lower than the rate of RSLR at both marshes; elevation grew at an average rate of 2.07 mm/year (Table 1) whereas MHW increased at an average rate of 4.17 mm/year since 1993. Sediment accretion was higher than elevation gain for both marshes, indicating that the soil below MHs is subsiding.

However, the difference between accretion and elevation gain was only significant at GBF ($p < .05$).
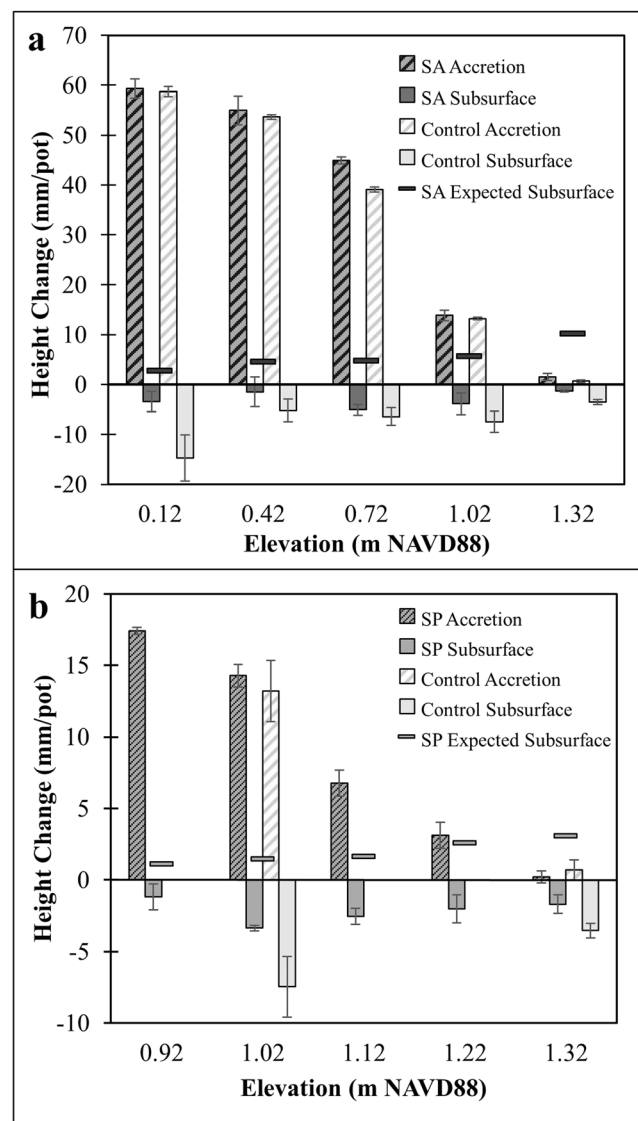
Discussion

Plant Responses to Flooding

The impact of flooding on belowground productivity was a primary focus of this study because of the direct implications for marsh elevation gain. We found that belowground biomass
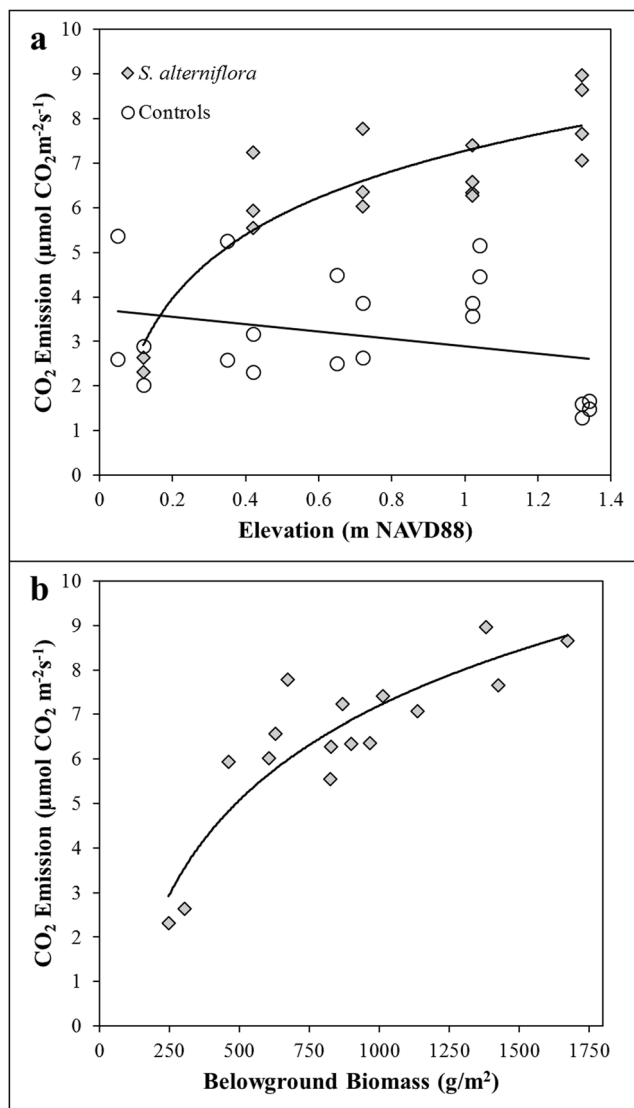
Fig. 7 **a** Relationship between elevation and CO_2 emission for controls and *S. alterniflora*. (Controls $r^2 = .143$, $p = .100$, *S. alterniflora* $r^2 = .784$, $p < .0001$). **b** Belowground biomass and CO_2 emission for *S. alterniflora* ($r^2 = .797$, $p < .0001$)


Fig. 8 Change in soil height due to sediment accretion and subsurface processes for **a** *S. alterniflora* (SA) and controls and **b** *S. patens* (SP) and controls. Patterned bars show sediment accumulation above marker horizons. Solid bars show subsurface change (change in marker horizon height). Dashes show expected subsurface change based on input of belowground plant volume. Error bars represent standard error

increased linearly with elevation (i.e., decreased with flooding) for both *S. alterniflora* and *S. patens*. This linear relationship was shown in similar marsh organ studies on *S. patens* (Kirwan and Guntenspergen 2015; Watson et al. 2016), and one study on *S. alterniflora* (Watson et al. 2017a; Table 2). Voss et al. (2013) also showed an overall increase in belowground biomass of *S. alterniflora* with elevation, but unlike our study, biomass decreased at the top elevation. In a natural marsh in Virginia, Blum (1993) reported greater root growth of *S. alterniflora* in the higher, more-oxidized marsh than the low-lying creekside site. Marsh grasses, rushes, and forbs also had higher shoot/root ratios at lower elevations in a transplant experiment in Oregon, suggesting that flooding stress impairs belowground growth more than aboveground (Janousek and Mayo 2013).

Overall, aboveground growth was also impaired at lower elevations in this study. The response of plant height differed from this pattern, showing a weak negative relationship with elevation for *S. patens* and a parabolic relationship for *S. alterniflora*. Although the parabolic response of *S. alterniflora* plant height suggests that flooding rate is optimal at middle elevations, aboveground biomass showed a different pattern; biomass increased linearly with elevation because plants at higher elevations grew more shoots. Contrasting with our results, marsh organ studies in South Carolina and Massachusetts found a parabolic relationship between aboveground biomass and elevation (Morris et al. 2013; Wigand et al. 2016), as well as a negative linear

Fig. 9 Sediment accretion rate as a function of elevation for **a** Great Bay Farms and **b** Crommett Creek with overall trends shown by exponential curves. Gray squares show data from May 2018 and black squares show December 2017. Graph **c** shows sediment accretion rate as a function of percent of time flooded for GBF ($r^2 = .718$, $p < .001$) and CRC ($r^2 = .609$, $p < .01$) measured in May 2018

*Rate from 2012-2017
**Rate from 2014-2017
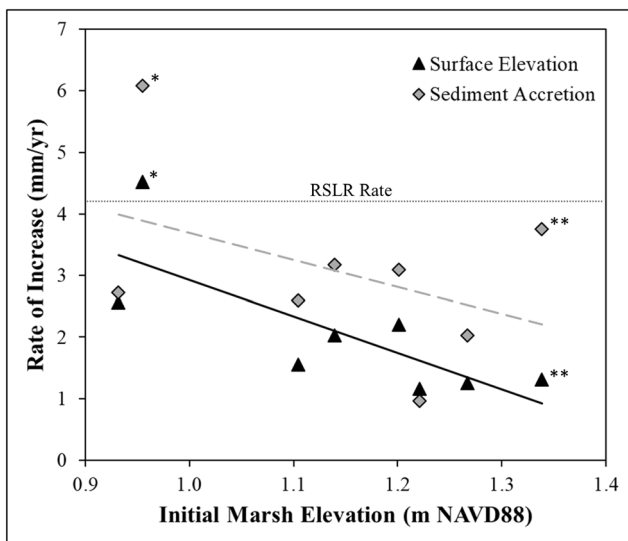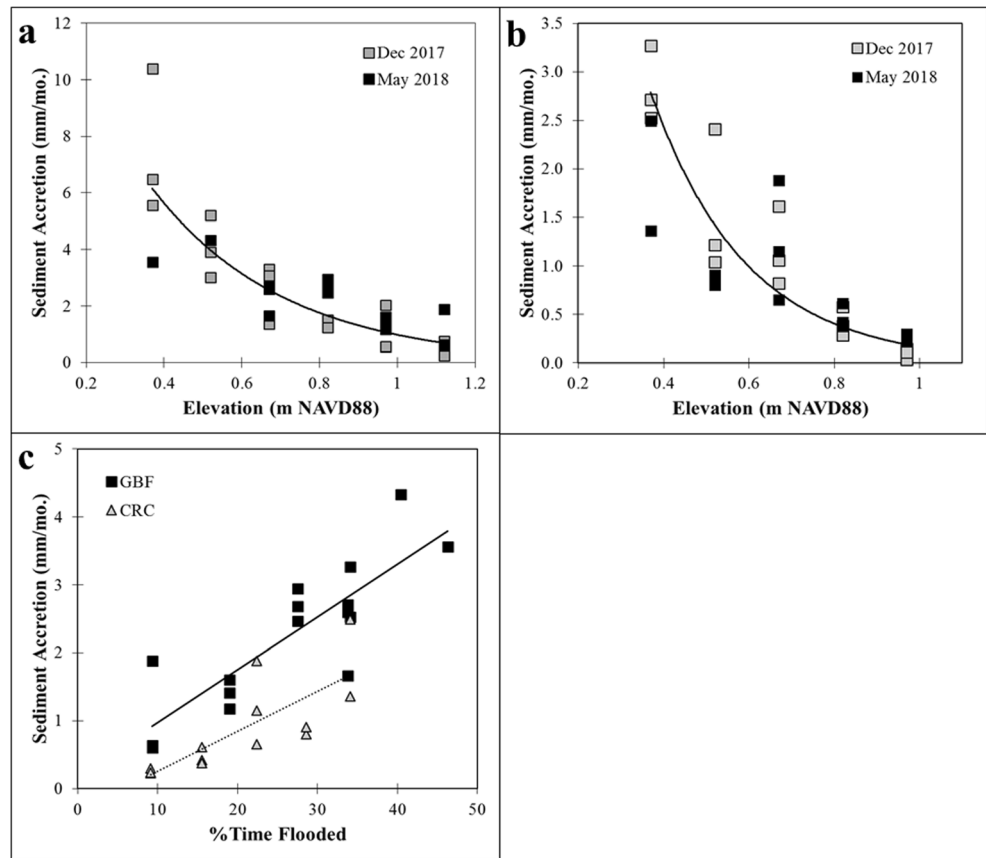
Fig. 10 Rates of marsh surface elevation gain and sediment accretion as a function of elevation for Great Bay SET-MH stations. Rates were calculated from 2011 to 2017 data except for points marked with asterisks. The horizontal dashed line shows RSLR (relative sea level rise) calculated from the change in mean high water 1993–2018. Elevation vs. surface elevation change: $r^2 = 0.658$, $p < .05$; elevation vs. sediment accretion: $r^2 = 0.182$, $p = .29$

relationship (Morris et al. 2013). In Chesapeake Bay marshes, Kirwan et al. (2012) found the relationship between elevation and *S. alterniflora* biomass is sometimes parabolic, but the shape of the relationship depends on variables such as marsh slope, precipitation, and groundwater depth. Marsh organ studies on *S. patens* align with our results more closely, showing that aboveground biomass increases linearly with elevation (Langley et al. 2013; Kirwan and Guntenspergen 2015), but may decrease at high elevations (25 cm above MHW) (Watson et al. 2016). At GBF, the upland edge began at 21 cm above MHW (1.38 m NAVD88) and lay above our experimental range.

The relationship between elevation and productivity has strong implications for marsh resilience to SLR. In this study, the positive linear relationship between elevation and biomass suggests aboveground and belowground productivity will decrease with SLR in northern New England for all elevations of *S. patens* and *S. alterniflora*. Other studies have shown lower productivity could result in less elevation gain due to the smaller contribution of marsh plants to soil organic matter (Cherry et al. 2009; Langley et al. 2009), as well as less sediment accretion due to reduced sediment trapping by leaves and stems (Gleason et al. 1979; Morris et al. 2002; Leonard and Croft 2006). Since the highest elevations produced the

Table 1 Rates of surface elevation change and sediment accretion with standard error

Marsh	Number	Initial Elevation (m NAVD88)	Elevation Gain (mm/year)	Accretion (mm/year)	Subsidence (mm/year)
Great Bay Farms	3	1.15 ± 0.03	1.93 ± 0.19	2.96 ± 0.18	1.03
Sandy Pt. (edge)	2	0.92 ± 0.002	3.54 ± 0.98	4.41 ± 1.69	0.87
Sandy Pt. (platform)	2	1.36 ± 0.06	1.24 ± 0.04	2.25 ± 0.81	1.01
X̄		1.14 ± 0.05	2.07 ± 0.39[**]	3.05 ± 0.53[***]	0.99[*]

Subsidence rates were calculated by subtracting elevation change rate from accretion rate

[*] Significant difference from zero at 0.05 (*), 0.01 (**), 0.0001 (***)

most biomass, our results are consistent with studies showing that upper species boundaries are governed not by lack of flooding, but by competition with less flood-tolerant plants (Bertness 1991). Productivity at low elevations is likely hindered by suboptimal soil conditions such as low oxygen availability (Mendelssohn et al. 1981) or high sulfide concentrations that can interfere with nutrient uptake (Koch et al. 1990).

Soil Elevation Responses to Flooding and Plant Growth

Although the presence of plants resulted in less subsidence, there was no relationship between belowground volume and subsurface height change in our study. Because our plants were not grown in natural peat, the relationship may have been confounded by roots growing into existing pore spaces rather than creating new ones by displacing soil (e.g., Day et al.

2011). The average soil porosity before plants were added was 68%. By the end of the experiment, roots occupied a mere 3% of the soil volume, indicating that ample pore space was available for roots to grow into. Subsidence, caused by compaction or decomposition in the soil may have also obscured the relationship between root growth and subsurface height change. The positive relationship we found between belowground biomass and CO_2 emission suggests roots were the main source of soil respiration, but roots may have also enhanced respiration in the rhizosphere by influencing microbial communities (e.g., Chen et al. 2012).

Previous research suggests the relationship between belowground growth and elevation change would have emerged over time, once the soil had fully consolidated and more of the existing pore space became occupied by roots (e.g., Cahoon et al. 2018). Studies of peat cores show a strong relationship between elevation gain and organic matter accumulation

Table 2 The relationship between elevation and above and belowground biomass found in other marsh organ studies for *S. patens* (*SP*) and *S. alterniflora* (*SA*)

Study	Species	Location	Elevation Range (relative to MHW) and % time flooded	Aboveground biomass	Belowground biomass
Langley et al. 2013	SP*	Edgewater, MD	MHW not stated 3 to 95%	Positive linear	Positive linear
Kirwan and Guntenspergen 2015	SP	Blackwater, MD	MHW not stated 0 to 100%	Positive linear	Positive linear
Watson et al. 2016	SP	Greenhouse Study	− 15 to + 25 cm %Flooding not stated	Parabolic	Positive asymptotic
Payne et al. (this study)	SP	Great Bay, NH	− 26 to + 14 cm 2 to 22%	Positive linear	Positive linear
Morris et al. 2013	SA	North Inlet, SC	− 102 to + 40 cm %Flooding not stated	Parabolic	Not measured
Morris et al. 2013	SA	Plum Island, MA	− 60 to + 60 cm %Flooding not stated	Negative linear	Not measured
Voss et al. 2013	SA	Pine Knoll Shores, NC	MHW not stated 0.5 to 95%	Parabolic	Positive asymptotic
Wigand et al. 2016	SA	Plum Island, MA	− 49 to + 18 cm %Flooding not stated	Parabolic	Positive asymptotic (not significant)
Watson et al. 2017a	SA	Narragansett Bay, RI	Not stated	Not measured	Positive linear
Payne et al. (this study)	SA	Great Bay, NH	− 106 to + 14 cm 2 to 58%	Positive linear	Positive linear

[*] Belowground biomass mixed with *Schoenoplectus americanus*

(Bricker-Urso et al. 1989; Callaway et al. 1997; Nyman et al. 2006; Boyd and Sommerfield 2016). Since most aboveground litter is lost through decomposition or export (White and Trapani 1982), belowground growth is the key component of soil organic matter. The importance of belowground growth is also supported by mesocosm studies showing that soil elevation can increase solely through root production (Cherry et al. 2009; Langley et al. 2009). While sediment accretion may be an important factor in building low marsh elevation, sediment-limited areas of the high marsh appear to build in elevation primarily through belowground growth.

Sediment Accretion and Surface Elevation Change

Similar to the marsh organ results, MH transects at both sites showed a decrease in sediment accretion as elevation increased. Sediment accretion rates measured from feldspar MHs are likely more accurate than marsh organ rates because they were taken from the actual marsh surface. However, in the feldspar plots, elevation covaries with distance from the creek, making it impossible to isolate the effects of each variable. Other studies have similarly shown that most suspended sediment is deposited close to creeks (Stumpf 1983; Kastler and Wiberg 1996; Leonard 1997; Christiansen et al. 2000), and mineral accretion correlates with percent flooding (Cahoon and Reed 1995). Sediment accretion can also vary significantly between nearby marshes, as shown at GBF and CRC. Variability in sediment accretion may relate to differences in wind and wave energy that affect resuspension and transport of sediment (Allen and Duffy 1998; Day et al. 1999).

SETs indicate that low-lying marsh areas of the Great Bay are building vertically at a faster rate than higher areas. The two lowest rates of elevation gain were measured at stations farthest from a major creek or the marsh edge, suggesting distance from sediment sources is also important (also found by Christiansen et al. 2000). We found that marsh elevation relative to sea level has been decreasing by 2.1 mm/year on average—about 5 cm loss in elevation capital over the past 25 years. Raposa et al. (2016) reported even lower rates of marsh building, and combined with a higher rate of RSLR, showed why high marshes in Rhode Island are rapidly drowning (Watson et al. 2016). Although New Hampshire salt marshes may be less vulnerable to SLR than marshes in southern New England due to isostatic adjustment (Karegar et al. 2016), the loss of elevation capital is cause for concern. At one of our Sandy Point stations, the dominant species has transitioned from *S. patens* to *S. alterniflora* over 18 years (Burdick unpub. data), despite being the only station that appears to be building at a rate similar to that of RSLR. High sediment supply may not prevent loss of high marsh, since the highest accretion deficit of 2.2 mm/year was measured at GBF, a marsh where the TSS of 57 mg/L is higher than found in most salt marshes (Leonard 1997; Kirwan et al. 2010). For

comparison, the median TSS reported in the nearby Plum Island Estuary was less than 30 mg/L for most sampling locations, and the highest median TSS was only ~40 mg/L (Hopkinson et al. 2018). Although our small number of samples (n = 16 at GBF) only provide a rough estimate of sediment supply, TSS data combined with rapid accretion measured in the low marsh suggest GBF elevation gain is not limited by sediment availability from tidal creeks.

Similar to subsidence shown in marsh organ pots, we found that marshes in the Great Bay are also subsiding. The highest Great Bay subsidence rate of 2.4 mm/year is ~10% of the highest rate reported by Cahoon et al. (1995) in southeastern marshes. Subsidence in New Hampshire may be less because colder temperatures slow rates of decomposition (Chmura et al. 2003; Kirwan and Blum 2011; Crosby et al. 2017), but other factors could include nutrient availability (Wigand et al. 2009), peat thickness, and bulk density (van Ardenne et al. 2018). Although rates are lower in this study, the average subsidence of 1 mm/year is enough to cause significant loss of elevation capital over time.

Implications

The results of this study have implications at both local and regional scales. Using a numerical model, Kirwan et al. (2010) found that marshes with a sediment supply similar to that of GBF can survive SLR of up to 45 mm/year. Based on their model and the high marsh accretion deficit measured at GBF, the marsh may survive even the most extreme SLR scenarios but with a conversion of high marsh to low marsh. At more regional scales, our finding that increased flooding results in less productivity but greater accretion indicates that sediments will play a larger role in building marsh elevation as sea level rises. Marshes with high sediment supplies may be less vulnerable to SLR because increased sediment accretion may compensate for decreased productivity, but high marsh areas farther from the sediment source may face an accretion deficit even when sediment supply is high (as seen at GBF marsh). Our finding that planted pots subside less than unplanted controls suggests marshes will lose elevation at a higher rate after plants drown. The absence of belowground production following plant death can result in sudden marsh collapse and conversion to mudflat or open water (DeLaune et al. 1994; Day et al. 2011). Our results may also have implications for restoration projects by suggesting a lag time exists between marsh revegetation and elevation gain (e.g., Cahoon et al. 2018). Planting a restored marsh may reduce subsidence, but roots may have to fill existing pore space before soil expansion can occur.

Predictions of marsh loss are complicated by additional feedbacks associated with climate change (Watson et al. 2017b). Higher temperatures and CO_2 concentration can accelerate plant growth (Langley et al. 2009), but productivity gains may be offset by faster rates of decomposition, perhaps

even resulting in a net loss of soil organic matter (Chmura et al. 2003; Kirwan and Blum 2011; Crosby et al. 2017). Additionally, an increase in the strength and frequency of storms may augment sediment accretion (Stumpf 1983; Scheurch et al. 2013) while exacerbating erosion of the marsh edge (Priestas and Fagherazzi 2011). Although feedbacks make it difficult to predict the impact of climate change, SETs show clear, cumulative effects on elevation relative to sea level. Elevation gain is lower than the rate of RSLR for about 58% of salt marshes in the USA (Cahoon 2015), and SLR has caused extensive marsh loss in Louisiana (Blum and Roberts 2009) and replacement of high marsh species by *S. alterniflora* in New England (Warren and Niering 1993; Donnelly and Bertness 2001; Smith 2015; Raposa et al. 2017). Further study is needed to better understand the complex processes affecting marsh elevation to make management strategies more efficient and effective.

References

Allen, J.R.L., and M.J. Duffy. 1998. Medium-term sedimentation on high intertidal mudflats and salt marshes in in the Severn Estuary, SW Britain: the role of wind and tide. *Marine Geology* 150 (1–4): 1–27.

Anisfeld, S.C., and T.D. Hill. 2012. Fertilization effects on elevation change and belowground carbon balance in a Long Island Sound tidal marsh. *Estuaries and Coasts* 35 (1): 201–211.

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

Bertness, M.D. 1991. Zonation of *Spartina patens* and *Spartina alterniflora* in a New England salt marsh. *Ecology* 72 (1): 138–148.

Blum, L.K. 1993. *Spartina alterniflora* root dynamics in a Virginia marsh. *Marine Ecology Progress Series* 102: 169–178.

Blum, M.D., and H.H. Roberts. 2009. Drowning of the Mississippi Delta due to insufficient sediment supply and global sea-level rise. *Nature Geoscience* 2 (7): 488–491.

Boyd, B.M., and C.K. Sommerfield. 2016. Marsh accretion and sediment accumulation in a managed tidal wetland complex of Delaware Bay. *Ecological Engineering* 92: 37–46.

Bricker-Urso, S., S.W. Nixon, J.K. Cochran, D.J. Hirschberg, and C. Hunt. 1989. Accretion rates and sediment accretion in Rhode Island salt marshes. *Estuaries* 12 (4): 300–317.

Bromberg, C.D., and M.D. Bertness. 2005. Reconstructing New England salt marsh losses using historical maps. *Estuaries* 28 (6): 823–832.

Cahoon, D.R. 2015. Estimating relative sea level rise and submergence potential at a coastal wetland. *Estuaries and Coasts* 38 (3): 1077–1084.

Cahoon, D.R., and G.R. Guntenspergen. 2010. Climate change, sea-level rise, and coastal wetlands. *National Wetlands Newsletter* 32 (1): 8–12.

Cahoon, D.R., and D.J. Reed. 1995. Relationships among marsh surface topography, hydroperiod, and soil accretion in a deteriorating Louisiana salt marsh. *Journal of Coastal Research* 11 (2): 357–369.

Cahoon, D.R., D.J. Reed, and J.W. Day Jr. 1995. Estimating shallow subsidence in microtidal salt marshes of the southeastern United States: Kaye and Barghoorn revisited. *Marine Geology* 128 (1-2): 1–9.

Cahoon, D.R., J.C. Lynch, C.T. Roman, J.P. Schmit, and D.E. Skidds. 2018. Evaluating the relationship among vertical wetland development, elevation capital, sea-level rise, and tidal marsh sustainability. *Estuaries and Coasts* 42 (1): 1–15.

Callaway, J.C., R.D. DeLaune, and W.H. Patrick Jr. 1997. Sediment accretion rates from four coastal wetlands along the Gulf of Mexico. *Journal of Coastal Research* 13 (1): 181–191.

Chapman, V.J. 1960. *Salt marshes and salt deserts of the world*. New York: Interscience.

Chen, J., L. Wang, Y. Li, W. Zhang, X. Fu, and Y. Le. 2012. Effect of *Spartina alterniflora* invasion and its controlling technologies on soil microbial respiration of a tidal wetland in Chongming Dongtan, China. *Ecological Engineering* 41: 52–59.

Cherry, J.A., K.L. McKee, and J.B. Grace. 2009. Elevated CO_2 enhances biological contributions to elevation change in coastal wetlands by offsetting stressors associated with sea level rise. *Journal of Ecology* 97 (1): 67–77.

Chmura, G.L., S.C. Anisfeld, D.R. Cahoon, and J.C. Lynch. 2003. Global carbon sequestration in tidal, saline, wetland soils. *Global Biogeochemical Cycles* 17 (4): 1–12.

Christiansen, T., P.L. Wiberg, and T.G. Milligan. 2000. Flow and sediment transport on a tidal marsh surface. *Estuarine, Coastal and Shelf Science* 50 (3): 315–331.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387 (6630): 253–260.

Crosby, S.C., A. Angermeyer, J.M. Adler, M.D. Bertness, L.A. Deegan, N. Sibinga, and H.M. Leslie. 2017. *Spartina alterniflora* biomass allocation and temperature: implications for salt marsh persistence with sea-level rise. *Estuaries and Coasts* 40 (1): 213–223.

Day, J.W., J. Rybczyk, F. Scarton, A. Rismondo, D. Are, and G. Cecconi. 1999. Soil accretionary dynamics, sea-level rise and the survival of wetlands in a Venice lagoon: a field and modelling approach. *Estuarine, Coastal and Shelf Science* 49 (5): 607–628.

Day, J.W., G.P. Kemp, D.J. Reed, D.R. Cahoon, R.M. Boumans, J.M. Suhayda, and R. Gambrell. 2011. Vegetation death and rapid loss of surface elevation in two contrasting Mississippi delta salt marshes: The role of sedimentation, autocompaction, and sea-level rise. *Ecological Engineering* 337: 229–240.

DeLaune, R.D., J.A. Nyman, and W.H. Patrick Jr. 1994. Peat collapse, ponding, and wetland loss in a rapidly subsiding coastal marsh. *Journal of Coastal Research* 10 (4): 1021–1030.

Donnelly, J.P., and M.D. Bertness. 2001. Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. *Proceedings of the National Academy of Sciences* 98 (25): 14218–14223.

Ford, M.A., D.R. Cahoon, and J.C. Lynch. 1999. Restoring marsh elevation in a rapidly subsiding salt marsh by thin-layer deposition of dredged material. *Ecological Engineering* 12 (3-4): 189–205.

Gleason, M.L., D.A. Elmer, and N.C. Pien. 1979. Effects of stem density upon sediment retention by saltmarsh cordgrass, *Spartina alterniflora* Loisel. *Estuaries* 2 (4): 271–273.

Harrington, J.T., J.G. Mexal, and J.T. Fisher. 1994. Volume displacement provides a quick and accurate way to quantify new root production. *Tree Planter's Notes* 45 (4): 121–124.

Hopkinson, C.S., J.T. Morris, S. Fagherrazzi, W.M. Wollheim, and P.A. Raymond. 2018. Lateral edge erosion as a source of sediments for vertical marsh accretion. *Journal of Geophysical Research: Biosciences* 123: 2444–2465.

Janousek, C.N., and C. Mayo. 2013. Plant responses to increased inundation and salt exposure: interactive effects on tidal marsh productivity. *Plant Ecology* 214 (7): 917–928.

Karegar, M.A., T.H. Dixon, and S.E. Engelhart. 2016. Subsidence along the Atlantic Coast of North America: insights from GPS and late Holocene relative sea level data. *Geophysical Research Letters* 43: 1–8.

Kastler, J.A., and P.L. Wiberg. 1996. Sedimentation and boundary changes of Virginia salt marshes. *Estuarine, Coastal and Shelf Science* 42 (6): 683–700.

Kelley, J.K., W.R. Gehrels, and D.F. Belknap. 1995. Late Holocene relative sea-level rise and and the geological development of tidal marshes at Wells, Maine, USA. *Journal of Coastal Research* 11 (1): 136–153.

Kirwan, M.L., and L.K. Blum. 2011. Enhanced decomposition offsets enhanced productivity and soil carbon accumulations in coastal wetlands responding to climate change. *Biogeosciences* 8 (4): 987–993.

Kirwan, M.L., and G.R. Guntenspergen. 2015. Response of plant productivity to experimental flooding in a stable and a submerging marsh. *Ecosystems* 18 (5): 903–913.

Kirwan, M.L., G.R. Guntenspergen, A. D'Alpaos, J.T. Morris, S.M. Mudd, and S. Temmerman. 2010. Limits on the adaptability of coastal marshes to rising sea level. *Geophysical Research Letters* 37 (23): 1–5.

Kirwan, M.L., R.R. Christain, L.K. Blum, and M.M. Brinson. 2012. On the relationship between sea level and *Spartina alterniflora* production. *Ecosystems* 15 (1): 140–147.

Koch, M.S., I.A. Mendelssohn, and K.L. McKee. 1990. Mechanism for the hydrogen sulfide-induced growth limitation in wetland macrophytes. *Limnology and Oceanography* 35 (2): 399–408.

Langley, A.J., K.L. Mckee, D.R. Cahoon, J.A. Cherry, and J.P. Megonigal. 2009. Elevated CO_2 stimulates marsh elevation gain, counterbalancing sea-level rise. 2009. *Proceedings of the National Academy of Sciences* 106 (15): 6182–6186.

Langley, A.J., T.J. Mozdzer, K.A. Shepard, S.B. Hagerty, and J.P. Megonigal. 2013. Tidal marsh plant responses to elevated CO_2, nitrogen fertilization, and sea-level rise. *Global Change Biology* 19 (5): 1495–1503.

Leonard, L.A. 1997. Controls of sediment transport and deposition in an incised mainland marsh basin, southeastern North Carolina. *Wetlands* 17 (2): 263–274.

Leonard, L.A., and A.L. Croft. 2006. The effect of standing biomass on flow velocity and turbulence in *Spartina alterniflora* canopies. *Estuarine, Coastal and Shelf Science* 69 (3-4): 325–336.

Mendelssohn, I.A., K.L. McKee, and W.H. Patrick. 1981. Oxygen deficiency in *Spartina alterniflora* roots: metabolic adaptation to anoxia. *Science* 214 (4519): 439–441.

Morris, J.T. 2007. Estimating net primary production of salt marsh macrophytes. In *Principles and standards for measuring net primary production in long-term ecological studies*, ed. T.J. Fahey and A.K. Knapp, 106–119. New York: Oxford University Press.

Morris, J.T., P.V. Sundareshwar, C.T. Nietch, B. Kjerfve, and D.R. Cahoon. 2002. Responses of coastal wetlands to rising sea level. *Ecology* 83 (10): 2869–2877.

Morris, J.T., K. Sundberg, and C.S. Hopkinson. 2013. Salt marsh primary production and its responses to relative sea level and nutrients in estuaries at Plum Island, Massachusetts, and North Inlet, South Carolina, USA. *Oceanography* 26 (3): 78–84.

Nicholls, R.J., and A. Cazenave. 2010. Sea-level rise and its impact on coastal zones. *Science* 328 (5985): 1517–1520.

Nyman, J.A., R.J. Walters, R.D. DeLaune, and W.H. Patrick Jr. 2006. Marsh vertical accretion via vegetative growth. *Estuarine, Coastal and Shelf Science* 69 (3-4): 370–380.

Priestas, A.M., and S. Fagherazzi. 2011. Morphology and hydrodynamics of wave-cut gullies. *Geomorphology* 131 (1-2): 1–2): 1–13.

Raposa, K.B., M.L. Cole Ekberg, D.M. Burdick, N.T. Ernst, and S.C. Adamowicz. 2016. Elevation change and the vulnerability of Rhode Island (USA) salt marshes to sea-level rise. *Regional Environmental Change* 16 (5): 389–397.

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

Scheurch, M., A. Vafeidis, T. Slawig, and S. Temmerman. 2013. Modeling the influence of changing storm patterns on the ability of a salt marsh to keep pace with sea-level rise. *Journal of Geophysical Research: Earth Surface* 118 (1): 84–96.

Seliskar, D.M., J.L. Gallagher, D.M. Burdick, and L.A. Mutz. 2002. The regulation of ecosystem functions by ecotypic variation in the dominant plant: a *Spartina alterniflora* salt-marsh case study. *Journal of Ecology* 90 (1): 1–11.

Short, F.T., S.K. Kosten, P.A. Morgan, S. Malone, and G.E. Moore. 2016. Impacts of climate change on submerged and emergent wetland plants. *Aquatic Botany* 135: 3–17.

Smith, S.M. 2015. Vegetation change in salt marshes of Cape Cod National Seashore (Massachusetts, USA) between 1984 and 2013. *Wetlands* 35 (1): 127–136.

Stumpf, R.P. 1983. The process of sedimentation on the surface of a salt marsh. *Estuarine, Coastal and Shelf Science* 17 (5): 495–508.

Van Ardenne, L.B., S. Jolicouer, D. Bérubé, D.M. Burdick, and G.L. Chmura. 2018. The importance of geomorphic context for estimating the carbon stock of salt marshes. *Geordarma* 330: 264–275.

Vermeer, M., and S. Rahmstorf. 2009. Global sea level linked to global temperature. *Proceedings of the National Academy of Sciences* 106 (51): 21527–21532.

Voss, C.M., R.R. Christian, and J.T. Morris. 2013. Marsh macrophyte responses to inundation anticipate impacts of sea-level rise and indicate ongoing drowning of North Carolina salt marshes. *Marine Biology* 160 (1): 181–194.

Wang, X.C., R.F. Chen, and A. Berry. 2003. Sources and preservation of organic matter in Plum Island salt marsh sediments (MA, USA): long-chain n-alkanes and stable carbon isotope compositions. *Estuarine, Coastal and Shelf Science* 58 (4): 917–928.

Warren, R.S., and W.A. Niering. 1993. Vegetation change on a northeast tidal marsh: interaction of sea-level rise and marsh accretion. *Ecology* 74 (1): 96–103.

Watson, E.B., A.J. Oczkowski, C. Wigand, A.R. Hanson, E.W. Davey, S.C. Crosby, R.L. Johnson, and H.M. Andrews. 2014. Nutrient enrichment and precipitation changes do not enhance resiliency of salt marshes to sea level rise in the Northeastern U.S. *Climatic Change* 125 (3-4): 501–509.

Watson, E.B., K. Szura, C. Wigand, K.B. Raposa, K. Blount, and M. Cencer. 2016. Sea level rise, drought and decline of *Spartina patens* in New England marshes. *Biological Conservation* 196: 173–181.

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

Watson, E.B., K.B. Raposa, J.C. Carey, C. Wigand, and R.S. Warren. 2017b. Anthropocene survival of southern New England's salt marshes. *Estuaries and Coasts* 40 (3): 617–625.

Weston, N.B. 2014. Declining sediments and rising seas: an unfortunate convergence for tidal wetlands. *Estuaries and Coasts* 37 (1): 1–23.

White, D.A., and J.M. Trapani. 1982. Factors influencing the disappearance of *Spartina alterniflora* from litterbags. *Ecology* 63 (1): 242–245.

Wigand, C., P. Brennan, M. Stolt, M. Holt, and S. Ryba. 2009. Soil respiration rates in coastal marshes subject to increased watershed nitrogen loads in southern New England, USA. *Wetlands* 29 (3): 952–963.

Wigand, C., K. Sundberg, A. Hansen, E. Davey, R. Johnson, E. Watson, and J. Morris. 2016. Varying inundation regimes differentially affect natural and sand amended marsh sediments. *PLoS One* 11 (10): e0164956.