2016

Elevation dynamics in a restored versus a submerging salt marsh in Long Island Sound

Shimon C. Anisfeld
Yale University, shimon.anisfeld@yale.edu

Troy D. Hill
Yale University

Daonald R. Cahoon
United States Geological Survey, dcahoon@usgs.gov

Estuarine, Coastal and Shelf Science 170 (2016) 145–154

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

Elevation dynamics in a restored versus a submerging salt marsh in Long Island Sound

Shimon C. Anisfeld [a,*], Troy D. Hill [a], Donald R. Cahoon [b]

[a] School of Forestry & Environmental Studies, Yale University, 370 Prospect St., New Haven, CT 06511, USA
[b] Patuxent Wildlife Research Center, United States Geological Survey, 10300 Baltimore Avenue, c/o BARC-East, Building 308, Beltsville, MD 20705, USA

ARTICLE INFO

Article history:
Received 23 June 2015
Received in revised form
5 January 2016
Accepted 9 January 2016
Available online 13 January 2016

Keywords:
Restoration
Submergence
Sea-level rise
Tidal marsh
Long Island Sound

ABSTRACT

Accelerated sea-level rise (SLR) poses the threat of salt marsh submergence, especially in marshes that are relatively low-lying. At the same time, restoration efforts are producing new low-lying marshes, many of which are thriving and avoiding submergence. To understand the causes of these different fates, we studied two Long Island Sound marshes: one that is experiencing submergence and mudflat expansion, and one that is undergoing successful restoration. We examined sedimentation using a variety of methods, each of which captures different time periods and different aspects of marsh elevation change: surface-elevation tables, marker horizons, sediment cores, and sediment traps. We also studied marsh hydrology, productivity, respiration, nutrient content, and suspended sediment. We found that, despite the expansion of mudflat in the submerging marsh, the areas that remain vegetated have been gaining elevation at roughly the rate of SLR over the last 10 years. However, this elevation gain was only possible thanks to an increase in belowground volume, which may be a temporary response to waterlogging. In addition, accretion rates in the first half of the twentieth century were much lower than current rates, so century-scale accretion in the submerging marsh was lower than SLR. In contrast, at the restored marsh, accretion rates are now averaging about 10 mm yr^{-1} (several times the rate of SLR), much higher than before restoration. The main cause of the different trajectories at the two marshes appeared to be the availability of suspended sediment, which was much higher in the restored marsh. We considered and rejected alternative hypotheses, including differences in tidal flooding, plant productivity, and nutrient loading. In the submerging marsh, suspended and deposited sediment had relatively high organic content, which may be a useful indicator of sediment starvation.

1. Introduction

Tidal marshes are critical components of the coastal landscape, providing valuable habitat, shoreline protection, and biogeochemical processing. In order to survive in place during periods of rising seas, marshes must accrete vertically at roughly the rate of local relative sea-level rise (SLR). Over the last 100–150 years, the rate of relative SLR in Connecticut has been higher than at any other time in the last 2200 years (Kemp et al., 2015). There is evidence that global eustatic SLR is continuing to accelerate (Watson et al., 2015) and is likely to accelerate further in the coming decades (Church et al., 2013).

Marshes accrete vertically by gaining organic matter and inorganic sediment, as well as pore space (Reed, 1990). Models (and limited field data) suggest that marshes can adjust their accretion to varying rates of SLR through a set of feedbacks in which higher water levels lead to greater rates of both plant productivity and sediment deposition. However, these feedbacks operate only within certain ranges of critical biophysical parameters such as SLR, suspended sediment concentration, and tidal range (D'Alpaos et al., 2011; Kirwan and Temmerman, 2009; Kirwan et al., 2010; Kirwan and Megonigal, 2013; Morris et al., 2002).

Marsh vegetation loss through the process of interior marsh breakup and conversion to mudflat – referred to as "marsh submergence" – has been observed at many locations in the last few decades. At some of these locations, SLR rates are relatively low, and explanations other than excessive SLR have been sought for the failure of the marsh to maintain adequate accretion rates. These explanations include herbivory (Bertness et al., 2014; Holdredge et al., 2009), excessive nitrogen loading (Deegan et al., 2012;

* Corresponding author.
E-mail address: shimon.anisfeld@yale.edu (S.C. Anisfeld).

http://dx.doi.org/10.1016/j.ecss.2016.01.017

Turner, 2010), hydrologic changes (Swanson and Wilson, 2008; Turner, 1997), subsidence (Rybczyk and Cahoon, 2002; Turner, 2004), and reductions in suspended sediment (Day et al., 2000; Kirwan et al., 2011). In addition, several studies have suggested that lateral erosion of the marsh edge, rather than vertical submergence of the marsh interior, is the primary mechanism of vegetation loss, and that this process is controlled by sediment supply, along with wave energy and geomorphology (Fagherazzi et al., 2013; Mariotti and Carr, 2014; Mariotti and Fagherazzi, 2013).

During the same time period when concern over marsh submergence has been growing, managers in many locations have continued to implement marsh restoration programs. These efforts often target hydrologically-restricted marshes that have been invaded by *Phragmites australis* (hereafter Phragmites). Restoration of tidal flow (e.g., through removal of dikes and/or tide gates) often leads to successful restoration of a healthy marsh ecosystem (Warren et al., 2002). In order to fully eliminate Phragmites, managers often "aim low" in terms of target elevation, with the goal of creating a relatively wet system that will be colonized by low-marsh *Spartina alterniflora*. However, the restoration goal of creating low-lying marsh systems may need to be re-evaluated in the face of accelerating SLR and the specter of marsh submergence (Anisfeld, 2012).

In this paper, we explore the similarities and differences between two low-lying marsh systems in Long Island Sound (LIS) – one that has undergone successful restoration, and one that is undergoing submergence. We seek to understand the different trajectories of these systems and to learn lessons that apply to both submergence and restoration. In particular, we address the following questions:

1. Do the restored and submerging marshes differ in their current and historic rates of accretion? That is, are the differences in marsh lateral change (at one site, expansion of low marsh; at the other, loss of low marsh) reflected in point measurements of vertical change?
2. To what extent are the differences in fate between these two marshes associated with the presence of marsh stressors such as low productivity, high nitrogen (N), unfavorable hydrology, or low suspended sediment?

In addressing these questions, we measure marsh accretion using a number of different methods that span a range of timescales and measure different combinations of processes. These methods are each commonly used individually in the literature, but few studies have used all these approaches in the same system. In doing so, we also address a third question:

3. How similar are the results from different methods of assessing marsh sedimentation?

2. Site description

Both of our study sites are tidal salt marshes located on the northern shore of LIS. Tides in LIS are semi-diurnal, with a mean tidal range (MTR) that increases from east to west: 0.78 m in New London, CT, to 2.05 m in Bridgeport, CT (data from http://tidesandcurrents.noaa.gov; for tidal ranges at our sites, see Results). Both sites have grid-ditches, which were dug for mosquito control in the first half of the 20th century at almost all marshes in LIS (Rozsa, 1995).

Our submerging-marsh study site was at Sherwood Island State Park in Westport, CT (Fig. 1A). Currently most of the tidal flow originates from New Creek, although before the deepening of this channel in 1929, this connection to LIS was probably minor. In 1956–1957, in a fill project designed to create a new parking area and to store gravel for highway construction, approximately 20 ha of marsh (out of an original 46 ha) was converted into upland. Since 1957, there have been no significant physical or hydrological changes to the marsh, and the marsh is protected from further development by its location in the state park, as well as by the Connecticut Tidal Wetlands Act of 1969. However, aerial photograph analysis shows that submergence has been taking place in this system: over the period 1974–2010, 22% of the vegetated marsh was converted to mudflat Hill (unpublished). The pattern of vegetation loss includes both expansion of the pre-existing channels and mosquito ditches, as well as development of new interior mudflats. Our sampling locations at this site were on the marsh platform adjacent to the newly formed interior mudflat (Fig. 1A). Vegetation at sampling locations was dominated by short-form *S. alterniflora*.

Our restored-marsh study site was the inner (northern) portion of Jarvis Creek in Branford, CT (Fig. 1B). In the early 1930s, a tide gate was installed seaward of this site, resulting in restricted tidal flow, lowered water levels, and Phragmites invasion; the tide gate was removed in 1979 to restore tidal flow (Paul Capotosto, CT DEEP, pers. comm.). Aerial photos from 1951, 1965, and 1974 show extensive Phragmites coverage, especially on the slightly higher ground near the mosquito ditches (Fig. 1B). By the 1986 aerial photo, Phragmites coverage had started to decline, and by the time we began visiting this site in 2004, the marsh was dominated by

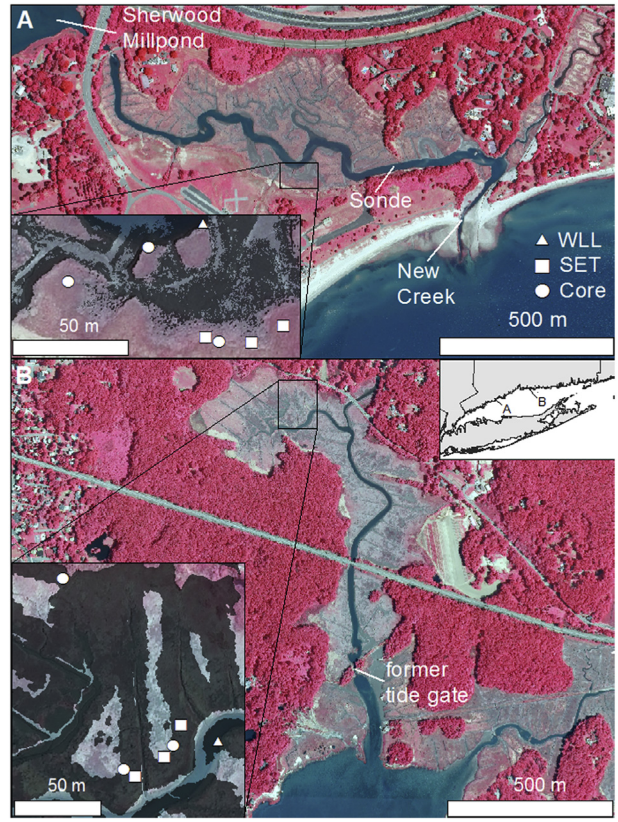

Fig. 1. Site map: Aerial photos from 2010 (CT DEEP 2014). A: Sherwood (submerging marsh); B: Jarvis (restored marsh). Shown are the locations of water level loggers (WLL), Surface Elevation Table (SET) benchmarks (n = 3), and sediment cores (n = 3) at each site, as well as the turbidity sonde that was deployed at Sherwood only. In the insets, the dark overlay shows areas that converted from vegetated to mudflat (at Sherwood, A) or from Phragmites to other (at Jarvis, B) over the period 1974–2010, based on aerial photography.

tall-form *S. alterniflora*, with no Phragmites visible except near the upland edges. The marsh does contain patches that are too wet to support vegetation, but our sampling locations were fully vegetated with tall-form *S. alterniflora*. Thomas and Varekamp (2012) used foraminifera and plant macro-fossils in a dated core from this site to confirm that Phragmites marsh transitioned to low marsh starting around 1979, with accretion rates of ~5.5 mm/yr for the low marsh (post-1979) peat. Ownership of this marsh is a mix of private and public (including the Branford Land Trust and Yale University), but the site is fully protected from development by the Tidal Wetlands Act.

3. Methods

To assess accretion in these marshes, we used three approaches, each of which operates at a different time scale and captures a different aspect of marsh sedimentation. *Sediment traps* covered day- to month-long periods in 2008 and 2013; this method focuses exclusively on short-term surface processes, specifically sediment deposition. *Surface-elevation tables (SETs) with marker horizons* covered the period 2005–2014 with sub-annual resolution; marker horizons are used to assess surface processes, such as sediment deposition and erosion, while SETs also incorporate sub-surface processes, such as belowground production and decomposition. *Radiometric dating of sediment cores* covered the period 1900–2005 with sub-decadal resolution; this method primarily captures surface processes, but also incorporates some belowground processes (e.g., belowground production above the ^{137}Cs peak).

3.1. Sediment traps

Triplicate plots were installed at each site (Fig. 1). Plots were separated horizontally by at least 16 m and were dominated by *S. alterniflora*. To avoid any impact to the marsh surface, plots were accessed using removable aluminum planks placed on permanent wooden supports.

To assess short-term deposition on the marsh surface, sediment traps were deployed at each of the 6 plots in both 2008 and 2013. Sediment traps consisted of 47 mm diameter 0.45 μm cellulose nitrate filters, which were rinsed with DI water, dried at 105 °C, and weighed prior to deployment. In the field, traps were placed on upside-down petri dishes set flush with the sediment surface, and secured with metal braces. In June–July 2008, 4 traps were deployed in each plot for 29 days. In July 2013, one trap was deployed in each plot for 24 h (though one of the traps at Jarvis was lost before retrieval). Traps were retrieved, dried, weighed, and ashed at 500 °C to determine organic and inorganic deposited sediment. In 2008, the 4 traps from each plot were combined before statistical analysis. Mass deposition rates were expressed in g m^{-2} yr^{-1}. To convert these mass deposition rates to volumetric change (mm yr^{-1}), we calculated a bulk density for each sample based on its organic matter content. This was possible because of the strong negative correlation between carbon and bulk density that was found for the samples from the sediment cores ($n = 58$, $r^2 = 0.67$).

3.2. SET-MH

At each of the 6 plots, we installed one deep-rod surface-elevation table (SET) benchmark and two feldspar marker horizon (MH) sub-plots (Cahoon et al., 1996, 2002; Callaway et al., 2013). The former provides a measure of change in elevation relative to the deep benchmark (referred to as “elevation change”), while the latter provides a measure of material accreted at or near the surface (referred to as “accretion”).

SET benchmarks were installed in November 2004 (Sherwood) and June 2005 (Jarvis) by pounding survey rods to refusal (average depth = 6.9 m). Benchmark elevations were measured with a total station and converted to NAVD88 based on RTK GPS measurements. Plots were sampled periodically through August 2014; sampling generally took place in summer, although we also sampled in fall 2011 (Sherwood only) and fall 2012 (both sites) to assess the effects of the Irene and Sandy storms, respectively.

SET sampling at each plot consisted of 9 pin readings in each of 4 directions, while MH sampling consisted of 3–6 readings from one cryo-core from each of the two markers. Cryo-cores were not collected during the first sampling periods, when the feldspar layer was still visible at the surface. Beginning in 2008, surface sediment from cryo-cores was carefully sliced off of the underlying feldspar and returned to the lab, where the sample was dried and organic matter content was determined by loss on ignition at 500 °C. In 2012, we installed new marker horizons at Jarvis, as the original marker horizons were becoming unusable due to their depth. The final measured depths of the original marker horizons were added to the cryo-cores that sampled the new marker horizons, so that all accretion data are expressed relative to the original (2005) horizons.

SET replication (re-setting the entire arm as well as the pins) was carried out with a frequency of 11% (32 out of 295 total directions), and had a mean standard error (SE) of 1.0 mm. Marker horizon replication (collecting a second cryo-core from the same marker horizon sub-plot) was carried out with a frequency of 10% (12 out of 117 cryo-cores), and had a mean SE of 1.6 mm.

For each sampling period at each plot, elevation change was averaged over the 4 directions (each of which was an average of 9 pins) and accretion was averaged over the 2 cryo-cores (each of which was an average of 3–6 readings). Linear regressions of accretion and elevation change against time were carried out for each plot. At Sherwood, the average r^2 for the 6 regressions was 0.88, while at Jarvis it was 0.98.

3.3. Cores

In 2004/2005, three sediment cores (30–40 cm deep) were obtained from each marsh (Fig. 1), using a Russian peat corer (for 5 out of the 6 cores) or a PVC coring device (1 core); compaction was minimal in all cases. Cores were sectioned into slices (1–3 cm in thickness), which were weighed wet, dried, re-weighed, and ground to a fine powder. Dry sediment weight was corrected for salt content based on water loss. C and N content were measured on >60% of the samples using a LECO CN analyzer; C content for unmeasured samples was interpolated from the adjacent measured samples. Organic matter content was estimated as twice the C content, based on a large number of Connecticut and New York tidal marsh samples that were analyzed for both parameters ($n = 287$, $r^2 = 0.98$, $p < 0.01$; Hill and Anisfeld, 2015). Dry bulk density was calculated from water loss by assuming that the soil was saturated (as observed in the field) and that the particle densities for inorganic and organic material were 2.6 and 1.2 g cm^{-3}, respectively (Kolker et al., 2009).

Finely-ground sediment samples were sealed in 10 mL scintillation vials, and activities of ^{210}Pb, ^{137}Cs, and ^{226}Ra (through equilibration with its daughter ^{214}Pb) were determined on a Canberra high-purity germanium well detector, at energies of 46.5, 661.7, and 352.7 keV, respectively. Excess ^{210}Pb was calculated by subtracting ^{226}Ra activity from ^{210}Pb. Counting error was estimated as twice the square root of the number of net counts. Corrections for ^{210}Pb and ^{226}Ra self-absorption were determined for a subset of samples by counting a ^{210}Pb/^{226}Ra standard in a pipet inserted into the center of the sample vial and applying the formula of Cutshall et al. (1983). At Sherwood, self-absorption corrections were relatively large (up to 30%) and varied significantly with bulk density, so calculated corrections were applied to samples based on their bulk density. At Jarvis, self-absorption corrections were much lower (average 11% for ^{210}Pb and 7% for ^{226}Ra), with no relationship to bulk density, so no correction was applied.

Accretion rates for cores were calculated using both ^{137}Cs and ^{210}Pb dating methods. ^{137}Cs was used to calculate average accretion rates since 1963 by assigning the peak in ^{137}Cs activity to the year of peak ^{137}Cs emissions (1963). ^{210}Pb was used to calculate varying accretion rates over time using the Constant Rate of Supply (CRS) model (Appleby and Oldfield, 1978) following the approach described in Hill and Anisfeld (2015). For the Jarvis cores, there was still some excess ^{210}Pb at the bottom of the core, so we extrapolated downwards using exponential decay from the bottom three points in order to calculate complete excess ^{210}Pb inventories; this contributed <10% to the total inventory.

3.4. Hydrology and elevation

Water level loggers (Global Water) were deployed at both marshes (see Fig. 1 for locations) for various time periods between 1/29/2005 and 2/18/2007. During this time, water level (corrected for barometric pressure changes) was recorded every 5 min over a total of 866 tides at Sherwood (5/13/05–3/8/06; 5/18/06–10/12/06) and 877 tides at Jarvis (1/29/05–3/7/05; 9/14/05–12/7/05; 12/19/05-5/17/06; 6/20/06–2/18/07). High- and low-tide times and water levels were extracted from this dataset. We assessed the tidal asymmetry by calculating the average length of ebb and flood tides at each marsh. We also compared high-tide times and water levels with values for the corresponding high tides at the nearby National Oceanic and Atmospheric Administration (NOAA) tidal gauge in Bridgeport (verified water level from NOAA; station 8467150). The hydrologic data were also used to evaluate the tidal flooding regime at each site. To eliminate any bias introduced by the different time periods covered, we focused on the period during which loggers were simultaneously in place at both marshes (6/20/2006-10/12/2006). This dataset was used to calculate mean high water (MHW) at each site and to calculate two tidal flooding parameters as a function of elevation at each site: flooding frequency (percentage of high tides that reached a given elevation) and flooding duration (percentage of the time that the water level was at or above a given elevation). Tidal flooding parameters were calculated for each plot based on its average elevation. Average sediment surface elevation for each plot was determined from the elevation of the SET benchmark, the geometry of the SET instrument, and the initial pin readings.

3.5. Productivity and biomass

Net aboveground primary production (NAPP) was measured for each plot annually from 2005 to 2008 using the peak standing crop method (Kirby and Gosselink, 1979). Small quadrats (0.01–0.09 m^2, depending on the year) were placed on the plot and all vegetation within the quadrat was clipped at the sediment surface, returned to the lab, rinsed with tap water followed by DI water, dried at 60–70 °C, and weighed.

Belowground net primary production (BNPP) was measured in 2007 using ingrowth chambers, consisting of a 5 cm diameter PVC pipe in which most (87%) of the PVC material was cut out from the pipe and replaced with 0.95 cm nylon mesh (Anisfeld and Hill, 2012). This permitted the chamber to retain its rigidity, but still allowed for unimpeded ingrowth. Chambers were filled with mud collected from the tidal creek at each site, which was pre-sieved to remove any roots and rhizomes. Chambers were deployed from April to October of 2007. Chambers were extracted by cutting around each with a saw before removal from the peat. Chamber contents were wet-sieved (1 mm), and the sieve-retained material (macro-organic matter, MOM) was dried at 60 °C; this material was assumed to represent productivity of one growing season. One plot at each site received duplicate chambers. Coefficients of variation for replicate chambers were 0.4% for Jarvis and 3.6% for Sherwood.

In addition, we collected a second set of cores (referred to as "sieved cores") in 2005 in order to assess the contribution of belowground biomass and fine organic matter to the peat. Cores (0–25 cm) were collected with a Russian peat corer from each SET plot (n = 3 per site), returned whole to the lab, weighed, and wet-sieved (1 mm). The sieve-retained material (MOM) was dried at 60 °C and weighed, while the sieve-passing material (mud, along with all the water used to perform the sieving) was dried at 80 °C and weighed. Concentrations of C and N in the mud and the MOM from the sieved cores – as well as in the NAPP and BNPP samples – were measured using a LECO CN analyzer. The organic component of the mud (referred to as micro-organic matter or micro-OM) was estimated as twice the mud C content.

Soil CO_2 flux (microbial and root/rhizome respiration) was measured using a LI-COR 6200 and a customized respiration chamber deployed between culms, as described in Anisfeld and Hill (2012). In 2007, CO_2 flux was measured once per month in June, July, September, and October; these values were converted to an annual flux by interpolating August flux and assuming that respiration in May was equal to the value in June and that respiration from November to April was negligible. In 2008, CO_2 flux was measured once per month in May, June, July, and August; these values were converted to an annual flux by assuming that respiration in September and October were each half of the value in August and that respiration from November to April was negligible.

3.6. Suspended sediment

Samples of water flooding the marsh platform (19 at Sherwood, 14 at Jarvis) were collected in July 2008 over a 3-h period around high tide. In addition, in July 2013, samples of water flooding the marsh platform (3 per site) were obtained using one-way sampling bottles designed to collect a sealed water sample when the water reached a fixed elevation (Nalgene Storm Water Sampler, ThermoFisher Scientific). Water samples were filtered (0.45 μm glass fiber filters) to assess total suspended sediment (TSS); filters were then ashed to determine organic content, which allowed us to calculate the concentration of organic suspended sediment (OSS) and inorganic suspended sediment (ISS).

In addition, a Eureka Manta water quality sonde was deployed at Sherwood for two weeks in May 2013, during which turbidity and water level were recorded at 10 min intervals. Of the 1897 values recorded, 3 were deleted as obvious outliers. To convert turbidity readings to TSS values, seven water samples were collected by hand and measured for both turbidity and TSS. The correlation between the two parameters had an r^2 of 0.95 and covered the range 2.5–6.7 NTU.

3.7. Statistics

Differences between the two sites were evaluated using t-tests. When data did not meet assumptions of normality and equal variance, t-tests were carried out on transformed data that did meet these assumptions, as noted in the results. In one case (suspended sediment data from 2008), no adequate transformations were found, so a Mann–Whitney test was carried out instead.

4. Results

4.1. Sediment traps, SET-MH, and cores

Sediment traps revealed large differences in short-term sedimentation between the two sites (Table 1A). In 2008, sedimentation at Jarvis was 19 times as high as at Sherwood, while, in the 24-h experiment in 2013, sedimentation at Jarvis was 66 times as high as at Sherwood. These differences in mass deposition translated into somewhat smaller – though still large – differences in vertical accretion (Table 2), due to the higher organic content (and lower bulk density) of the deposited sediment at Sherwood.

Accretion and elevation change in our SET-MH plots are shown in Fig. 2 and summarized in Table 2. Jarvis had significantly higher values than Sherwood for both surface accretion (MH) and elevation change (SET).

The difference between SET and MH measurements (Table 2) suggests that the sum of belowground processes was slightly net negative at Jarvis and net positive at Sherwood. In other words, some subsidence occurred at Jarvis, perhaps due to sediment compaction caused by the high inorganic content. Conversely, at Sherwood, belowground processes contributed positively to marsh elevation.

Profiles of ^{137}Cs and excess ^{210}Pb for our 6 cores (Fig. 3) also showed large differences between the two sites: Cs peaks were deeper in Jarvis, indicating more rapid sedimentation. ^{210}Pb profiles also showed deeper penetration at Jarvis, but neither site showed log-linear decay, a finding that is not surprising given the variable sedimentation history of these sites and that supports our choice of the CRS model.

CRS and ^{137}Cs dating were reasonably consistent: the CRS-calculated depth of the 1963 horizon was within 1 core section of the ^{137}Cs peak (expected to occur in 1963) for 4 out of the 6 cores, and within 2–3 core sections for the other two cores (Fig. 3). Inventories of ^{210}Pb at Sherwood (650 ± 10 mBq cm^{-2}) were slightly higher than would be expected based on atmospheric deposition (370–630; Graustein and Turekian, 1986; Turekian et al., 1983), while inventories at Jarvis (1140 ± 110) were considerably higher, suggesting delivery of sediment-bound ^{210}Pb (Cochran et al., 1998), possibly as a result of rapid sediment deposition. Inventories of ^{137}Cs at both Sherwood (80 ± 30 mBq cm^{-2}) and Jarvis (150 ± 20) were somewhat lower than expected (170–260; Graustein and Turekian, 1986, decayed to 2004), perhaps indicating some mobility and loss; however, the sharpness of the ^{137}Cs peaks for 5 of the 6 cores (Fig. 3) suggests that such mobility has been relatively minor.

CRS-based histories of accretion rates over time (Fig. 4) suggest that both Sherwood and Jarvis had low accretion rates (~2 mm yr^{-1}) in the first half of the 20th century. At Sherwood, a period of relatively rapid accretion (~6 mm yr^{-1}), dating to the 1960s or 1970s (depending on the core), was followed by a decline back to lower levels (~3 mm yr^{-1}). This feature also appears in an additional core from this site, described in Hill and Anisfeld (2015). The period of higher accretion may be related to the fill dumped on portions of the marsh (though not our coring sites) in the late 1950s. At Jarvis, the cores all showed an increase in accretion around 1950 and a second increase around 1980. This second increase – to accretion rates as high as 10 mm yr^{-1} – presumably reflects the effects of tidal restoration in 1979.

Average accretion rates for different time periods, calculated from the core data, showed similar results (Table 2). Near-surface accretion rates were about three times higher at Jarvis than at Sherwood, while accretion rates averaged over longer time periods (40 or 70 years) were still higher at Jarvis, but by a smaller margin.

4.2. Hydrology and elevation

Hydrologic parameters for the two marshes are summarized in Table 3. The sites had identical tidal ranges, but differed in other parameters. High tides at both sites were well-correlated with high tides at the Bridgeport NOAA station (Fig. 5). High tides at Sherwood had variation that was almost identical to Bridgeport (as indicated by a slope very close to 1 in Fig. 5), while the variation in high tides at Jarvis was dampened (a slope of about 0.75). That is, high tides at Jarvis varied over a range that was only about 75% of the range among high tide elevations at Sherwood. The lower variation at Jarvis may be due to the greater hydrologic distance between that site and the open waters of LIS (Fig. 1). This difference in connectivity is also reflected in the time lag between Bridgeport high tides and marsh high tides, a lag that was considerably longer at Jarvis (Table 3). Both marshes appeared to be flood-dominated, with ebb tides taking up more than half of the time (implying greater velocities on the flood), but this effect was larger at Sherwood. Lastly, the marsh platform (as reflected in our plots) was about 8 cm higher relative to MHW at Sherwood than it was at Jarvis.

Flooding-elevation curves for the two sites are shown in Fig. 6, along with the initial (2004–05) sediment surface elevations. Compared to Sherwood, Jarvis was clearly wetter (flooded more frequently and for longer duration). This was in part simply a function of its lower elevation, but it was also affected by the steeper flooding-elevation curves at Jarvis. In turn, the steeper curves are a consequence of the lower variability in high tides noted in the previous paragraph.

4.3. Productivity and biomass

Aboveground production ranged from ~300 to ~600 g C m^{-2} yr^{-1}, with no difference between the two sites (Fig. 7). Belowground production was considerably higher than aboveground (although the ingrowth method provides a relative, not absolute, measure), and did not differ by site. Annualized rates of soil CO_2 respiration (plant+microbe) were ~300–500 g C m^{-2} yr^{-1}

Table 1
Differences between the two sites in sediment parameters. Mean ± SE (n = 3 except where noted). Asterisks indicate significant differences between sites:*$p < 0.05$; **$p \leq 0.01$; ***$p \leq 0.001$.

	Sherwood	Jarvis
A. Deposited sediment (sediment traps)		
	deposition (g m^{-2} yr^{-1})	
2008*[a]	291 ± 53	5600 ± 2600
2013***[a]	1360 ± 70	90,000 ± 40,000[b]
B. Soil composition (sieved cores)		
	mass (kg m^{-2})	
MOM**[a]	10.0 ± 0.1	4.6 ± 0.7
micro-OM	9.5 ± 1.0	7.6 ± 0.4
IM*	53 ± 9	89 ± 8
C. Suspended sediment (water samples)		
	TSS (mg/L)	
2008***	24.8 ± 1.1[d]	63 ± 5[e]
2013*	6.7 ± 0.6	117 ± 36
	organic matter (%)	
2008***	41.1 ± 0.9[d]	30 ± 2[e]
2013*[c]	27 ± 2	11.4 ± 0.6

[a] Data were ln-transformed for equal variance.
[b] n = 2.
[c] Data were x^{-5}-transformed for equal variance.
[d] n = 19.
[e] n = 14.

Table 2
Summary of sedimentation measurements used in this study (mean ± SE). “Sig.” indicate significant differences between the two sites (*p < 0.05; **p ≤ 0.01; ***p ≤ 0.001; t-tests on untransformed data except as noted; n = 3 except as noted.). NA: not applicable.

Method	Time scale	Accretion/elevation change (mm yr^{-1})			Organic matter (%)		
		Sherwood	Jarvis	sig.	Sherwood	Jarvis	sig.
Sediment traps 2013	1 day	4.0 ± 0.3[a]	154 ± 14[ab]	***	17.7 ± 1.2	9 ± 7[b]	
Sediment traps 2008	1 month	2.0 ± 0.4[a]	13 ± 6[a]		48.4 ± 0.5	13 ± 2	***
SET	9 years[c]	3.0 ± 0.6	9.9 ± 1.1	**	NA	NA	
MH	9 years[c]	2.3 ± 0.1	12 ± 2	*	32.7 ± 1.2[ad]	12.0 ± 0.7[ae]	***
Cores (surface, Pb CRS)	2-10 years[f]	3.1 ± 0.6	8.9 ± 0.7	**	25 ± 11	13 ± 3	
Cores (^{137}Cs)	40 years[f]	2.8 ± 0.1	6.5 ± 0.4	***	21 ± 8	16 ± 3	
Cores (70 year avg, Pb CRS)	70 years[f]	2.8 ± 0.3	4.3 ± 0.1	**	21 ± 9	16 ± 3	

[a] t-test was carried out on ln-transformed data.
[b] n = 2.
[c] starting in 2004/5.
[d] n = 21.
[e] n = 15.
[f] ending in 2004/5.

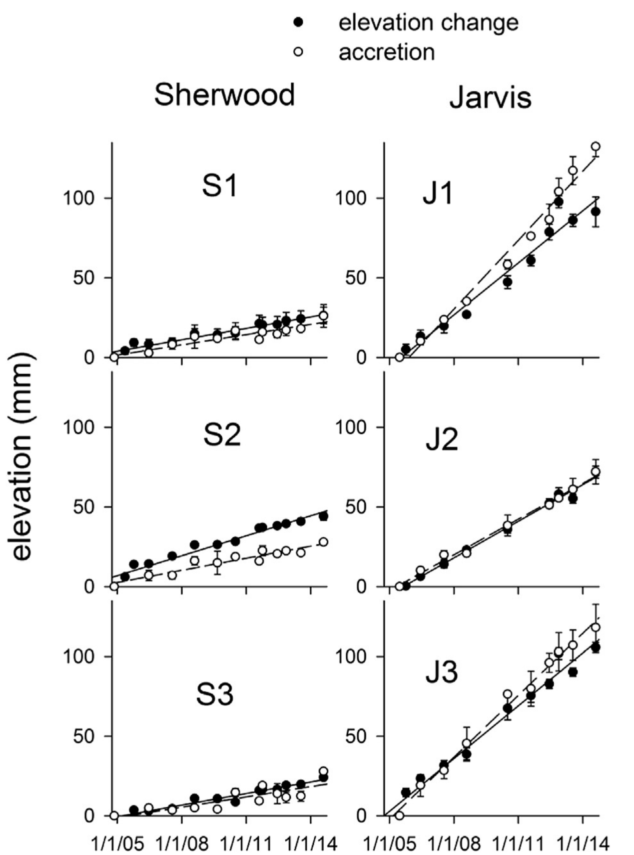

Fig. 2. Accretion (open symbols, dashed best-fit line) and elevation change (solid symbols, solid best-fit line) over time at our 6 SET-MH plots, beginning in November 2004 (Sherwood) and June 2005 (Jarvis). Mean and SE are shown for each point (n = 2 MH sub-plots for accretion, n = 4 directions for elevation change).

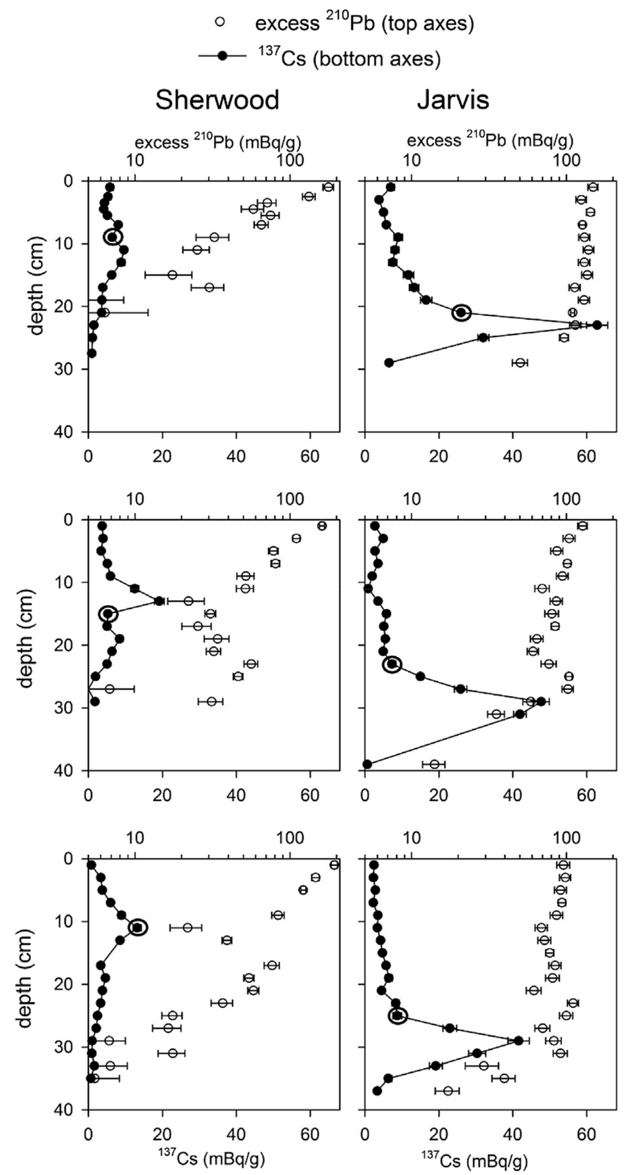

Fig. 3. Profiles of excess ^{210}Pb (top axes, log scale, open circles) and ^{137}Cs (bottom axes, linear scale, solid circles) for our 6 cores. Circled ^{137}Cs points indicate the expected depth of the ^{137}Cs peak based on ^{210}Pb CRS dating.

at both sites.

There were differences between the two sites in the composition of the soils (Table 1B). Peat at Sherwood had higher MOM and lower inorganic matter (IM) than at Jarvis; it also had lower bulk density (0.29 ± 0.04 vs. 0.40 ± 0.03 g cm^{-3}). This is consistent with a conceptual model in which Jarvis is rapidly accreting inorganic sediment, diluting the contribution of in-situ production.

Differences in nitrogen content between the two marshes were relatively small (Fig. 8). For most of our measurements, Jarvis had slightly higher N content than Sherwood (contrary to the nutrient-

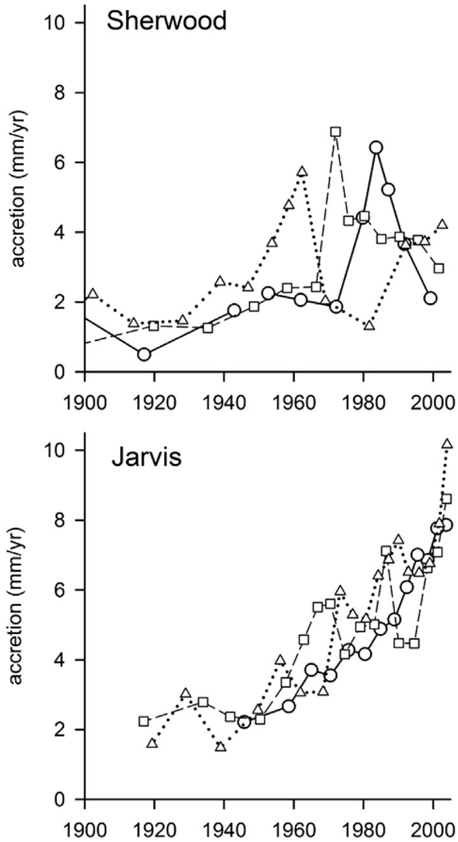

Fig. 4. Accretion rates over time, based on ^{210}Pb CRS dating. Sherwood shows peak accretion in the 1960s and 1970s, while Jarvis shows an increase in accretion around 1980.

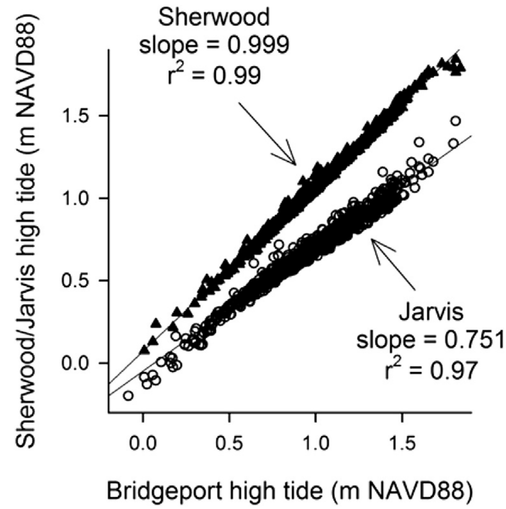

Fig. 5. Height of Sherwood and Jarvis high tides plotted against the height of the corresponding Bridgeport high tide (all in m NAVD88). Data from 2005 to 2007; n = 866 for Sherwood, 877 for Jarvis, $p < 0.001$ for both correlations.

Table 3
Hydrologic parameters for the two sites. MTR and MHW are based on 6/20/06–10/13/06; all other hydrologic data are based on the period of record for each site (see Methods). "High tide lag" refers to the time lag in high tide at a site relative to high tide at Bridgeport. "Ebb fraction" indicates what percentage of the time is made up of ebb tides (as opposed to flood tides). Plot elevation is based on our first SET measurements in 2004/2005. Values shown are means ± SE.

	MTR (m)	Bridgeport slope (see Fig. 5)	High tide lag (min)	Ebb fraction (% of time)	Initial plot elevation (m MHW)
Sherwood	1.42	0.999	17.0 ± 0.4	66.7 ± 0.2	−0.19 ± 0.02
Jarvis	1.42	0.751	41.8 ± 0.6	55.9 ± 0.2	−0.27 ± 0.02

submergence hypothesis), although this difference was only statistically significant for aboveground vegetation in 2008.

4.4. Suspended sediment

Suspended sediment concentrations differed dramatically between the two marshes (Table 1C), with Jarvis having higher concentrations, especially of inorganic sediment. Although the number of suspended sediment samples was small, our conclusion that Sherwood was low in suspended sediment was supported by the automated sonde measurements (not carried out at Jarvis), which gave an average turbidity of 1.9 NTU (equivalent to 5.6 mg/L, based on our site-specific correlation). Of the 1894 automated turbidity measurements, 95% were under 5.7 NTU (equivalent to 10.8 mg/L).

5. Discussion

5.1. Differences between the sites: lateral and vertical change

Our results clearly demonstrate that the restored marsh (Jarvis) had a much higher sedimentation rate than the submerging marsh (Sherwood) across a variety of time scales and methods (Table 2). This suggests that Question 1 (see Introduction) can be answered at least partly in the affirmative: the different trajectories of vegetation change at these sites (Fig. 1) are associated with differences in accretion rates on the marsh platform (Table 2).

At the same time, however, even the submerging site is gaining elevation: despite the areal loss of vegetation, our point measurements illustrate that the marsh platform is keeping up with SLR. Specifically, our SET-measured elevation change of 3.0 ± 0.6 mm yr^{-1} is close to – but not lower than – the multi-decadal rate of SLR at the Bridgeport tide gauge (2.9 ± 0.5 mm yr^{-1}, 1964–2014; NOAA). Indeed, vegetation in our plots has been relatively stable over the 9 years of the study, although with complete loss of the limited *Spartina patens* that was mixed in with the dominant *S. alterniflora* at the beginning of the study. Yet within 10 m of our plots, there are previously-vegetated areas that are now unvegetated mudflats (Fig. 1A), so calling this a submerging marsh is not inaccurate.

One explanation for this phenomenon would invoke the importance of wave erosion at the marsh–mudflat interface, a process that can destroy marshes even when relative elevation change on the marsh platform is positive (Fagherazzi et al., 2013; Leonardi and Fagherazzi, 2015). Our study lacks the data to directly assess this explanation. However, several pieces of indirect evidence suggest that vertical, rather than horizontal, processes may be in large part responsible for submergence at Sherwood.

First, the mudflat at Sherwood is much smaller than the basins that have been identified as susceptible to wave-induced erosion (Mariotti and Fagherazzi, 2013). With a maximum fetch of ~50 m

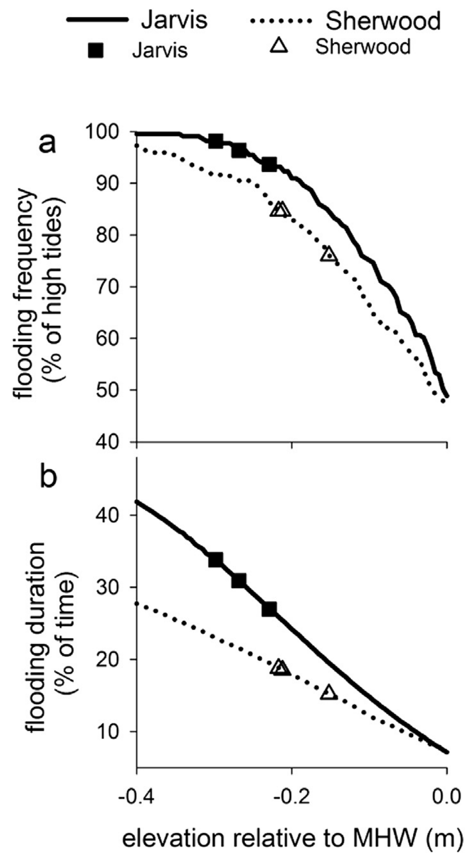

Fig. 6. Elevation-flooding curves for the two marshes: (a) flooding frequency (% of possible high tides); (b) flooding duration (% of the time). Plot average elevations (n = 3 per site) are shown for both sites. MHW and flooding parameters are based on 6/20/2006-10/12/2006.

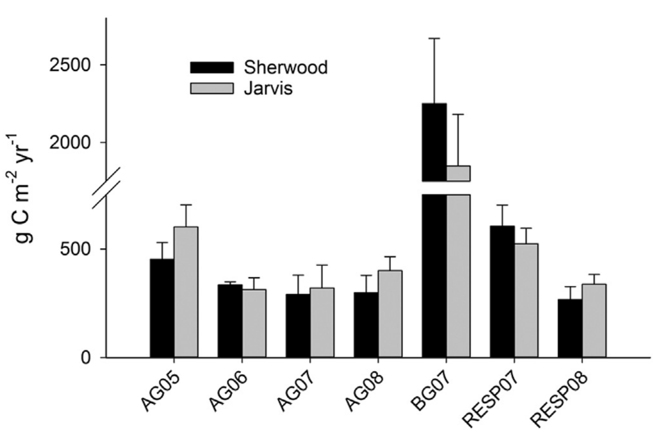

Fig. 7. Productivity and respiration at the two sites: AG = aboveground productivity; BG = belowground productivity; RESP = respiration rate; numbers indicate the year of the measurement. No significant differences were found between sites.

(Fig. 1), this mudflat is unlikely to generate sufficient wind-induced wave energy to result in lateral erosion, even at low suspended sediment concentrations.

Second, the rate of surface accretion (as opposed to elevation change) at our plots (2.3 ± 0.1 mm yr^{-1}, as measured by the marker horizon method) is lower than SLR, suggesting that without sub-surface peat expansion, which is likely to be a temporary response to increased waterlogging, our plots would in fact be submerging.

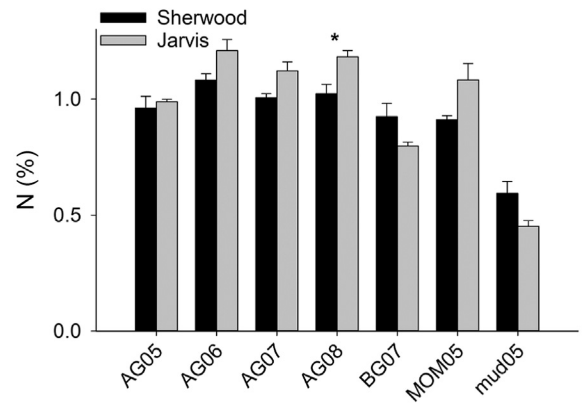

Fig. 8. Nitrogen content (mean and SE) of aboveground vegetation (AG), belowground production in ingrowth chambers (BG), and MOM and mud from sieved cores; numbers indicate the year of the measurement. Asterisk indicates a significant difference between sites (t-test, $p < 0.05$).

Third, early 20th-century rates of accretion at Sherwood were lower than recent rates (Fig. 4), resulting in a substantial loss of relative elevation when averaged over the past century. Based on our cores, the average accretion rate over the period 1895–2005 at Sherwood was 2.1 ± 0.2 mm yr^{-1}, lower than the average rate of SLR over the same period (2.9 mm yr^{-1}; based on NOAA data for the Battery, the only regional tide gauge extending back to the 19th century). This is similar to results from a recent regional analysis (Hill and Anisfeld, 2015), which found a general loss of relative elevation in Connecticut and New York marshes over the past century.

Thus, we believe that the most likely explanation for the vegetation loss that has taken place at Sherwood is that the lowest-lying vegetated areas have succumbed to the cumulative effects, over several decades, of inadequate sedimentation and consequent loss of elevation relative to sea level. The remaining vegetated areas are surviving for now, as they were presumably somewhat higher to start with and may be receiving more sediment due to the distribution of the limited sediment supply over a smaller area of vegetated marsh. Still, the low rate of surface accretion, the continuing decline in that rate relative to the 1970s peak (Fig. 4), and the continuing acceleration in SLR all suggest that the remaining vegetated marsh is likely to submerge in the next decade or two.

5.2. Reasons for accretion differences

What caused the large difference in sedimentation between these two systems? On one level, this might be interpreted simply as the expected difference in accretion with elevation: Jarvis was somewhat wetter (Table 3; Fig. 6), so subject to greater accretion (e.g., Morris et al., 2002). Indeed, it seems clear that the very high recent accretion rates at Jarvis (several times the rate of SLR) are possible largely because of the relatively low-lying conditions created by restoration.

However, if this were the entire story, one would expect that the lower elevations at Sherwood would benefit from this effect. Instead, the comparable elevations at Sherwood (i.e., ~27 cm below MHW, the average elevation of our SET plots at Jarvis) are unvegetated mudflat, having become submerged in the past 40 years. This suggests that there are important differences between the two systems that are not simply functions of relative elevation.

Potential explanations for this difference in accretionary processes include productivity, hydrology, nutrients, and suspended sediments (Question 2 in the Introduction). We address each in turn.

Productivity differences between the two systems are unlikely to drive the difference in sedimentation. All our measures of productivity (aboveground and belowground) and respiration were similar between the two marshes (Fig. 7).

There were important hydrologic differences between the sites, but these do not appear likely to explain the lower sedimentation rates at Sherwood. The data in Table 3 suggest that Sherwood may be more flood-dominated than Jarvis, which implies more, not less, of an opportunity for sediment import. On the other hand, it is possible that the less sheltered nature of the Sherwood marsh resulted in greater velocities on both the flood and the ebb, with greater possibility for erosion (though potentially also greater sediment advection); additional measurements would be necessary to address this possibility.

The nitrogen-submergence linkage is not supported by this case study, as the submerging marsh (Sherwood) generally had equal or lower nitrogen levels compared to the restored marsh (Fig. 8). We used tissue N concentrations as our metric of N availability, as have others (e.g., Bertness et al., 2014), since N concentrations in pore water and tidal flooding water are often quite variable. Our data add to a growing body of literature on the nitrogen-submergence linkage that is quite inconsistent: some studies have shown a major impact of nitrogen loading on marsh belowground biomass and soil strength (Deegan et al., 2012; Wigand et al., 2014), while others have found that nitrogen loading is unlikely to have significant detrimental effects on marsh elevation (Anisfeld and Hill, 2012; Fox et al., 2012; Langley et al., 2013; Morris et al., 2013).

The most dramatic and relevant difference between the two sites was in the suspended sediment concentration. The higher TSS concentration at Jarvis compared to Sherwood (Table 1C) appeared to be closely tied to the higher sedimentation rate at both short and long time scales (Table 2). Our data support the idea that high sediment availability is a prerequisite both for marsh restoration and for marsh survival in a time of accelerated SLR.

A related difference between the two sites was the higher organic content of suspended and deposited sediment at Sherwood compared to Jarvis (Tables 1 and 2). This is similar to results from Louisiana (Day et al., 2011) and Chesapeake Bay (Ganju et al., 2015), where submerging marshes had sediment with high organic content. This suggests that, of the little suspended sediment that exists at Sherwood, a substantial portion may be recycled from the nearby submerging portion of the marsh (Ganju et al., 2015).

5.3. Comparison of different methods

Each of the commonly-used approaches to understanding marsh sedimentation captures a different combination of aboveground and belowground processes. We found remarkably similar results among these different methods (Question 3, Introduction). In particular, there was close correspondence between the core-top results (~1996–2005) and the SET results (2004–2014), with values at Jarvis of 8.9 and 9.9 mm yr^{-1}, respectively, and values at Sherwood of 3.1 and 3.0 mm yr^{-1}, respectively (Table 2). The slightly higher values for the more recent decade at Jarvis may suggest that the accretion rate has continued the upward trajectory shown in Fig. 4.

Differences between SET and MH results suggest that belowground processes are net positive at Sherwood and net negative at Jarvis, as discussed above. This is also consistent with the 1-month sediment trap results, which produced results lower than other methods at Sherwood and higher than other methods at Jarvis (Table 2), implying that processes other than surface sedimentation must be positive at Sherwood and negative at Jarvis. The 1-day sediment trap results, in contrast, were unusually high at both Sherwood and (especially) Jarvis, suggesting that this was an unusual event, or that the sediment deposited would have been eroded in subsequent tides.

Acknowledgments

The EPA Long Island Sound Study provided major funding to support this study (Grant LI-97100801). Additional financial support was provided by the Connecticut Sea Grant program, a Carpenter-Sperry Environmental Research Grant, and a Sounds Conservancy Grant from the Quebec-Labrador Foundation. Field assistance was provided by James Lynch (USGS) and Yale students Joanna Carey, Jessica Darling, and Azalea Mitch. Ivan Valiela, Patrick Megonigal, Rebecca Schultz, and two anonymous reviewers provided helpful comments. Lab assistance was provided by Helmut Ernstberger. We are grateful to the Branford Land Trust for access to Jarvis, and to the staff at Sherwood Island State Park, especially James Beschle, for access to Sherwood. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the US Government.

References

Anisfeld, S.C., 2012. Biogeochemical responses to marsh restoration. In: Roman, C.T., Burdick, D.M. (Eds.), Tidal Marsh Restoration: a Synthesis of Science and Management. Island Press, Washington, pp. 39–58.
Anisfeld, S.C., Hill, T.D., 2012. Fertilization effects on elevation change and belowground carbon balance in a Long Island Sound tidal marsh. Estuaries Coasts 35, 201–211.
Appleby, P., Oldfield, F., 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported ^{210}Pb to the sediment. Catena 5, 1–8.
Bertness, M.D., Brisson, C.P., Bevil, M.C., Crotty, S.M., 2014. Herbivory drives the spread of salt marsh die-off. Plos One 9, e92916.
Cahoon, D.R., Lynch, J.C., Knaus, R.M., 1996. Improved cryogenic coring device for sampling wetland soils. J. Sediment. Res. 66, 1025–1027.
Cahoon, D.R., Lynch, J.C., Perez, B.C., Segura, B., Holland, R.D., Stelly, C., Stephenson, G., Hensel, P., 2002. High-precision measurements of wetland sediment elevation: II. The rod surface elevation table. J. Sediment. Res. 72, 734–739.
Church, J.A., Clark, P.U., Cazenave, A., Gregory, J.M., Jevrejeva, S., Levermann, A., Merrifield, M.A., Milne, G.A., Nerem, R.S., Nunn, P.D., Payne, A.J., Pfeffer, W.T., Stammer, D., Unnikrishnan, A.S., 2013. Sea level change. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: the Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.
Callaway, J.C., Cahoon, D.R., Lynch, J.C., 2013. The surface elevation table – marker horizon method for measuring wetland accretion and elevation dynamics. In: DeLaune, R.D., Reddy, K.R., Richardson, C.J., Megonigal, J.P. (Eds.), Methods in Biogeochemistry of Wetlands. Soil Science Society of America. Madison, WI, pp. 901–917.
Cochran, J.K., Hirschberg, D.J., Wang, J., Dere, C., 1998. Atmospheric deposition of metals to coastal waters (Long Island Sound, New York USA): evidence from saltmarsh deposits. Estuar. Coast. Shelf Sci. 46, 503–522.
Cutshall, N.H., Larsen, I.L., Olsen, C.R., 1983. Direct analysis of ^{210}Pb in sediment samples: self-absorption corrections. Nucl. Instrum. Methods 206, 309–312.
D'Alpaos, A., Mudd, S.M., Carniello, L., 2011. Dynamic response of marshes to perturbations in suspended sediment concentrations and rates of relative sea level rise. J. Geophys. Res. Earth Surf. 116, F04020.
Day, J.W., Kemp, G.P., Reed, D.J., Cahoon, D.R., Boumans, R.M., Suhayda, J.M., Gambrell, R., 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. Ecol. Eng. 37, 229–240.
Day, J.W., Shaffer, G.P., Britsch, L.D., Reed, D.J., Hawes, S.R., Cahoon, D., 2000. Pattern and process of land loss in the Mississippi Delta: a spatial and temporal analysis of wetland habitat change. Estuaries 23, 425–438.
Deegan, L.A., Johnson, D.S., Warren, R.S., Peterson, B.J., Fleeger, J.W., Fagherazzi, S., Wollheim, W.M., 2012. Coastal eutrophication as a driver of salt marsh loss. Nature 490, 388–392.
Fagherazzi, S., Mariotti, G., Wiberg, P.L., McGlathery, K.J., 2013. Marsh collapse does not require sea level rise. Oceanography 26, 70–77.
Fox, L., Valiela, I., Kinney, E.L., 2012. Vegetation cover and elevation in long-term experimental nutrient-enrichment plots in Great Sippewissett Salt Marsh, Cape Cod, Massachusetts: implications for eutrophication and sea level rise. Estuaries Coasts 35, 445–458.
Ganju, N.K., Kirwan, M.L., Dickhudt, P.J., Guntenspergen, G.R., Cahoon, D.R.,

Kroeger, K.D., 2015. Sediment transport-based metrics of wetland stability. Geophys. Res. Lett. 42, 7992–8000.
Graustein, W.C., Turekian, K.K., 1986. Pb-210 and Cs-137 in air and soils measure the rate and vertical profile of aerosol scavenging. J. Geophys. Res. Atmos. 91, 14355–14366.
Hill, T.D., Anisfeld, S.C., 2015. Coastal wetland response to sea level rise in Connecticut and New York. Estuar. Coast. Shelf Sci. 163, 185–193.
Holdredge, C., Bertness, M.D., Altieri, A.H., 2009. Role of crab herbivory in die-off of New England salt marshes. Conserv. Biol. 23, 672–679.
Kemp, A.C., Hawkes, A.D., Donnelly, J.P., Vane, C.H., Horton, B.P., Hill, T.D., Anisfeld, S.C., Parnell, A.C., Cahill, N., 2015. Relative sea-level change in Connecticut (USA) during the last 2200 yrs. Earth Planet. Sci. Lett. 428, 217–229.
Kirby, C.J., Gosselink, J.G., 1976. Primary production in a Louisiana Gulf Coast Spartina-alterniflora marsh. Ecology 57, 1052–1059.
Kirwan, M., Temmerman, S., 2009. Coastal marsh response to historical and future sea-level acceleration. Quat. Sci. Rev. 28, 1801–1808.
Kirwan, M.L., Guntenspergen, G.R., D'Alpaos, A., Morris, J.T., Mudd, S.M., Temmerman, S., 2010. Limits on the adaptability of coastal marshes to rising sea level. Geophys. Res. Lett. 37, L23401.
Kirwan, M.L., Megonigal, J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504, 53–60.
Kirwan, M.L., Murray, A.B., Donnelly, J.P., Corbett, D.R., 2011. Rapid wetland expansion during European settlement and its implication for marsh survival under modern sediment delivery rates. Geology 39, 507–510.
Kolker, A.S., Goodbred, S.L., Hameed, S., Cochran, J.K., 2009. High-resolution records of the response of coastal wetland systems to long-term and short-term sea-level variability. Estuar. Coast. Shelf Sci. 84, 493–508.
Langley, J.A., Mozdzer, T.J., Shepard, K.A., Hagerty, S.B., Megonigal, J.P., 2013. Tidal marsh plant responses to elevated CO_2, nitrogen fertilization, and sea level rise. Glob. Change Biol. 19, 1495–1503.
Leonardi, N., Fagherazzi, S., 2015. Effect of local variability in erosional resistance on large-scale morphodynamic response of salt marshes to wind waves and extreme events. Geophys. Res. Lett. 42, 5872–5879.
Mariotti, G., Carr, J., 2014. Dual role of salt marsh retreat: long- term loss and short-term resilience. Water Resour. Res. 50, 2963–2974.
Mariotti, G., Fagherazzi, S., 2013. Critical width of tidal flats triggers marsh collapse in the absence of sea-level rise. Proc. Natl. Acad. Sci. U. S. A. 110, 5353–5356.
Morris, J.T., Shaffer, G.P., Nyman, J.A., 2013. Brinson Review: perspectives on the influence of nutrients on the sustainability of coastal wetlands. Wetlands 33, 975–988.
Morris, J.T., Sundareshwar, P.V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of coastal wetlands to rising sea level. Ecology 83, 2869–2877.
Reed, D.J., 1990. The impact of sea-level rise on coastal salt marshes. Prog. Phys. Geogr. 14, 465–481.
Rozsa, R., 1995. Human impacts on tidal wetlands: history and regulations. In: Dreyer, G.D., Niering, W.A. (Eds.), Tidal Marshes of Long Island Sound: Ecology, History and Restoration. Connecticut College Arboretum, New London, Ct, pp. 42–50.
Rybczyk, J.M., Cahoon, D.R., 2002. Estimating the potential for submergence for two wetlands in the Mississippi River Delta. Estuaries 25, 985–998.
Swanson, R.L., Wilson, R.E., 2008. Increased tidal ranges coinciding with Jamaica Bay development contribute to marsh flooding. J. Coast. Res. 24, 1565–1569.
Thomas, E., Varekamp, J.C., 2012. Long Island Sound salt marshes: accretion and relative sea level rise. In: Geological Society of America, Abstracts with Programs, vol. 44, p. 117.
Turekian, K.K., Benninger, L.K., Dion, E.P., 1983. Be-7 and Pb-210 total deposition fluxes at New-Haven, Connecticut and at Bermuda. J. Geophys. Res. Oceans Atmos. 88, 5411–5415.
Turner, R.E., 1997. Wetland loss in the northern Gulf of Mexico: multiple working hypotheses. Estuaries 20, 1–13.
Turner, R.E., 2004. Coastal wetland subsidence arising from local hydrologic manipulations. Estuaries 27, 265–272.
Turner, R.E., 2010. Beneath the salt marsh canopy: loss of soil strength with increasing nutrient loads. Estuaries Coasts 34, 1084–1093.
Warren, R.S., Fell, P.E., Rozsa, R., Brawley, A.H., Orsted, A.C., Olson, E.T., Swamy, V., Niering, W.A., 2002. Salt marsh restoration in Connecticut: 20 years of science and management. Restor. Ecol. 10, 497–513.
Watson, C.S., White, N.J., Church, J.A., King, M.A., Burgette, R.J., Legresy, B., 2015. Unabated global mean sea-level rise over the satellite altimeter era. Nat. Clim. Change 5, 565–568.
Wigand, C., Roman, C.T., Davey, E., Stolt, M., Johnson, R., Hanson, A., Watson, E.B., Moran, S.B., Cahoon, D.R., Lynch, J.C., Rafferty, P., 2014. Below the disappearing marshes of an urban estuary: historic nitrogen trends and soil structure. Ecol. Appl. 24, 633–649.