

Salt marshes and eutrophication: An unsustainable outcome

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Abstract

Most plant production by emergent coastal marshes occurs belowground. This belowground production adds to the accumulation of organic matter sustaining salt marshes as sea level rises, thus preventing excessive flooding, eventual plant death, and habitat loss. The ubiquitous nutrient enrichment of coastal salt marshes stimulating aboveground plant growth may result in higher rates of inorganic matter accumulation that compensates for marsh flooding caused by sea level rise. Results from several short-term experiments, however, demonstrate that root and rhizome biomass and carbon accumulation is reduced with nutrient enrichment, suggesting that eutrophication of coastal waters may not be a compensatory counterbalance to the effects of global sea level rise on salt marshes. We show that the net effects of 36 yr of nutrient enrichment in replicated field experiments do not lead to higher organic or inorganic accumulation. Enrichment reduces organic matter belowground and may result in a significant loss in marsh elevation equivalent to about half the average global sea level rise rates. Sustaining and restoring coastal emergent marshes is more likely if they receive less, not more, nutrient loading.

Coastal ecosystems are essential habitats for fish, wildlife, and people, and are stressed by a variety of changes worldwide, including sea level rise, eutrophication, predator overharvesting, food web dysfunctions, and hydrologic change (Jackson et al. 2001; Kirby 2004; Millennium Ecosystem Assessment 2005). Some of these stressors may be compensatory, and others may be synergistically maladaptive. The results of experiments from Massachusetts to Louisiana established that the aboveground production of the salt marsh macrophyte *Spartina alterniflora* is, in general, stimulated by increased nitrogen loading (Valiela and Teal 1974; Morris 1991). The majority of the annual total biomass production, however, occurs belowground (Valiela et al. 1976; Blum 1993; Darby and Turner 2008a). The root and rhizome biomass, unlike the aboveground biomass, may be diminished with nutrient enrichment as plants adjust their nutrient foraging strategies (Valiela et al. 1976; Darby and Turner 2008b,c). These changes belowground may be accompanied by a lower soil organic carbon content and increased microbial activity (Morris and Bradley 1999; Sundareshwar et al. 2003; Caffrey et al. 2007). A possible decline in the belowground accumulation of organic matter, however, may be compensated for by higher accumulations of inorganic matter settling out in the denser aboveground vegetation. Any changes in the long-term net accumulation of organic and inorganic matter with nutrification will affect the vertical position of the marsh relative to sea level rise and plant health. Louisiana marshes that were identified as “unhealthy” before sampling, for example, have a relatively low belowground biomass in comparison to those defined as “healthy” before sampling (Turner et al. 2004), and the vertical accretion of organic-rich salt marshes is primarily determined by the accumulation of organics, not inorganics

(Turner et al. 2001). An implication of these observations is that the increases in nutrient loading to coastal systems, which are widespread (National Research Council 2000; Rabalais 2002), may, or may not, compromise the long-term stability of salt marshes. Here we report on how 30+ yr of experimental addition of nutrients to a salt marsh affects the net accumulation of organic and inorganic materials, and changes marsh elevation.

Methods

Sample site—We sampled a suite of ten 10-m-diameter salt marsh plots (314 m²) in the Great Sippewissett Salt Marsh, Falmouth, Massachusetts (Valiela et al. 1975, 1976; Giblin et al. 1980). Each circular plot is bisected by a single drainage creek. These tidal creeks were originally ditches, which had been dug many decades before the experiment began and which provided a relatively consistent “creek” structure. The 10 plots are located in close proximity along two branches of a main tidal creek of the marsh. Their tidal waters arrive at the plots from a common creek, which bifurcates <100 m from the first plot moving landward from creek to upland. The plots are located on separate tidal creeks such that the greatest direct distance between any two plots was ~200 m, with individual plot boundaries generally separated by more than 10 m. All of the plots are arranged so that each plot contains a single tidal creek and each tidal creek has only one plot.

The two replicated treatments consisted of various dosages of nitrogen and phosphorus amendments during the growing season. The treatments have been without interruptions every year since the experiment started. Control, low-fertilizer, and high-fertilizer (C, LF, and HF, respectively) sites were established in 1970, urea and urea + phosphate (U and UP, respectively) plots in 1971, and extra-high-fertilizer (XHF) plots in 1974. The U and

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UP treatments were two halves of the same 10-m-diameter plot, with the drainage feature separating them. The LF, HF, and XHF dosage rates consisted of a commercially available sludge fertilizer (Milorganite fertilizer; 10% N, 6% P_2O_5). The U and UP treatment plots were dosed with the same amount of nitrogen as urea and the UP plot received P as superphosphate (46% N and 18% P, respectively, by weight). The P doses in UP and HF plots were the same amounts, although of different form.

Treatment additions were biweekly from April through November at low tide. The dose amounts of N for C, LF, HF, XHF, U, and UP treatment plots were 0, 2.46, 7.56, 22.68, 7.56, and 7.56 kg N km⁻² yr⁻¹, respectively. The P doses for the same treatments were 0, 0.64, 1.98, 5.94, 0, and 1.98 kg P km⁻² yr⁻¹, respectively. The nutrient load coming into the estuary from the watershed will be higher than the nutrient load to the marsh, because the marsh plain does not interact with all waters entering the estuary (Howes et al. 1996). Marsh nutrient loading estimates based on the watershed nutrient loads are, therefore, a potential marsh nutrient load, and will be higher than the actual nutrient load to the marsh. Given this caveat, the applied N loading rates at the HF treatment plots are one tenth of the estuarine water loading rate to salt marshes in Narragansett Bay (Wigand et al. 2003), but several times higher than in most (but not all) Gulf of Mexico estuaries (Turner and Rabalais 1999). The nitrogen and phosphorus loading rates at the XHF treatment plot (nine times the rate at the LF site), however, are among the highest loading rates known.

Nutrient amendments resulted in an increase of the aboveground biomass in all treatment plots (Valiela et al. 1976; Giblin et al. 1980). Sampling in 1977 and 1978 demonstrated that there was no difference in the aboveground biomass between the UP, LF, and HF treatment plots, for which biomass was approximately double that in the C plots, and that the aboveground biomass in the XHF plots was an additional 20% higher (Giblin et al. 1980). Sampling in 1981–1984 indicated that the aboveground biomass in the XHF sites was 4.1 times that of the C sites (Howes et al. 1986).

The plots originally contained areas of tall *S. alterniflora* along the creeks, short *S. alterniflora* on the marsh plain, and *Spartina patens* and *Distichlis spicata* on the marsh plain. The composition of the plots has changed very little. There was evidence of a shift towards more *D. spicata* in the 1978–1986 time frame and development of shrubs in the XHF plots. The present species cover, however, does not appear to be that different from the original cover described by Valiela et al. (1975), except that the fertilized plots have more tall *S. alterniflora*.

Sediment dating—Sediment cores were collected 2 m inside and outside of these treatments to investigate whether the organic and inorganic accumulations changed above the ¹³⁷Cs-dated 1963–1964 soil horizon. We collected 29 sediment cores in 2005, and dated them with ¹³⁷Cs radioisotopic methods. There were 13 samples taken from outside of the treatment plots that represented C samples. Some of these C samples were sufficiently close to two

different treatment plots so that they could be used as controls in a paired comparison of C and treatment plots. There were two, one, six, and four C samples for the UP, U, HF, and XHF treatments, respectively. These were paired with four samples taken from inside the treatment plots, with two samples each from two replicated treatments.

Sediment samples were collected in 10-cm-diameter × 50-cm-long seamless stainless steel core tubes. Cores were collected about 2 m inside the 10-m-diameter circle along a transect perpendicular to the creek bisecting the plot. The apparent soil compaction resulting from the core collection was assessed by measuring the soil surface (relative to the core end and to the marsh surface) inside the core tube, both before and after the core was removed from the ground. The cores were frozen upon return to the laboratory and their length measured again; they were then extruded using hot water on the steel core exterior and sectioned at 1-cm intervals. The thickness of each section was measured. The core sections were placed in polystyrene dishes (each subsection volume was ~90 cm³), weighed to the nearest 0.01 g, dried for 24 h at 60°C, and then weighed again. The dry weight was converted to a bulk density value (g dry weight cm⁻³) using the thickness of the individual core sections and the core diameter. The sections were homogenized using a Willey Mill® (with a No. 20 mesh screen) and then placed into labeled containers. A subsample (~1.0 g) of each of the homogenized core sections was taken to estimate inorganic and organic content by loss on ignition at 550°C. The dating methods, the quality control criteria, and the counting statistics applied are described in Milan et al. (1995). The accretion rate is the height above the peak in the ¹³⁷Cs activity, which is assumed to be circa 1963–1964. A Student's *t*-test was used to test for differences inside compared to outside of the treatment plots and for differences between treatments.

The soil volume represented by the organic and inorganic content in the dated sediments was determined using literature values (2.61 g cm⁻³ and 1.14 g cm⁻³, respectively; Turner et al. 2001).

Marsh elevation—We compared the marsh elevation 3 m outside to 3 m inside of the 10-m-diameter fertilized plots using a laser level with a relative accuracy of ±0.1 cm. Elevations were taken every 60 cm along a transect bisecting the circle and aligned perpendicular to the drainage ditch. Data from both sides of the circle were used, unless the treatment plot was divided into two treatments (U and UP treatments) or treatment plots were adjacent to the edges of other treatment plots. When the marsh surface elevation 3 m outside of the plot was higher than that 3 m inside on both sides of the circular plot, then the elevation change was negative and overland flow on the falling tide was towards the central creek.

Fertilizer volume—The fertilizer contributes a significant volume to the applied soils. This volume was estimated for each treatment dose and then subtracted from the elevation differences measured inside and outside of the treatment plots. The potential volume of the applied Milorganite

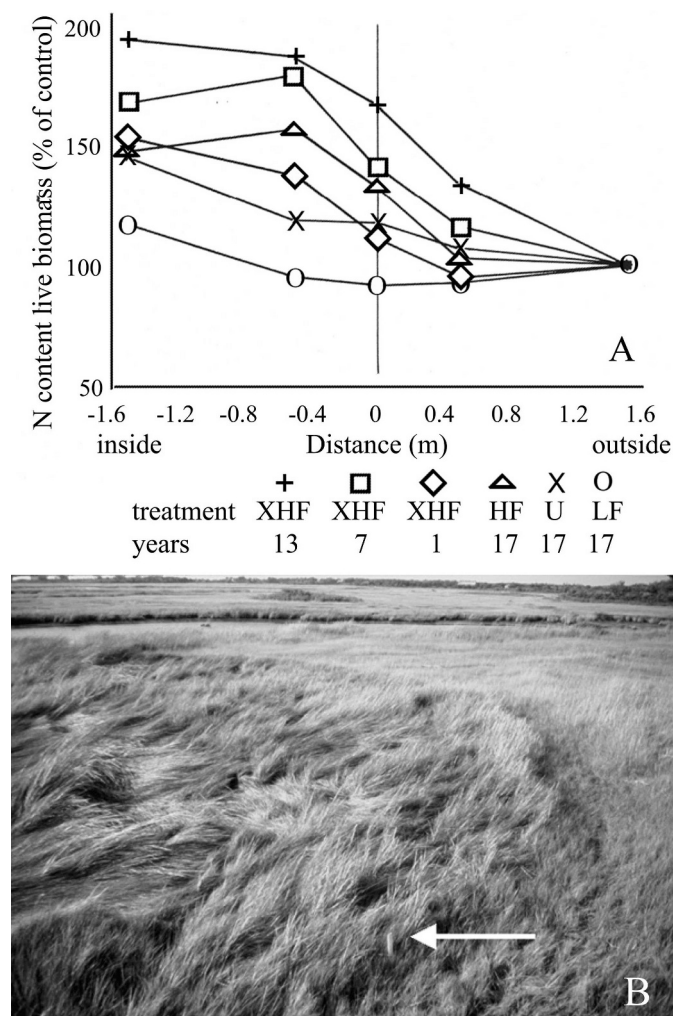


Fig. 1. The transition from fertilized to unfertilized marsh. (A) The percentage of nitrogen in live aboveground biomass at the end of the summer (1987) for six different treatment sites along a 3-m sampling line crossing the plot boundary. (B) The white arrow points at the top of a stake positioned at the original edge of the fertilized plot (left), and indicates a distinct break between the fertilized and unfertilized vegetation.

retained in the marsh was estimated using a bulk density of 0.70 g cm^{-3} for the applied material, which is more than three times the bulk density in the C plots. We made the assumption that the marsh retained 100% of the applied Milorganite amendments for several reasons. (1) A retention rate of 100% is based on an analysis of trace metals within these plots. These marshes retain up to 100% of the Cr, Cu, and Pb applied in the fertilizer, and a minimum of 20%, 55%, and 20% for Cd, Mn, and Zn, respectively (Giblin et al. 1980). These latter trace metal retention estimates are conservative, because soluble metals in pore water will be transported below the 25-cm-deep sampling horizon, where they may become trapped in sulfide complexes. (2) There is an abrupt discontinuity front in vegetation biomass quantity at the edge of the plot that has been maintained without “bleeding” beyond the marker stakes for more than three decades (Fig. 1; Howes

unpubl.). (3) Howes and White (1994) and Hamersley and Howes (2005) found there are very low losses of ^{15}N labeled N added to the plots. They calculated that there was a long-term N retention rate of 40% in the plots, with all but 6% of the nitrogen losses that were not taken up by the plants occurring through denitrification, which was confirmed by direct ^{15}N injections (Hamersley and Howes 2005). Equivalent measurements of retention for the U- and superphosphate-amended plots are not available.

Shear vane resistance and organic content—A transect line running 2.5 m parallel to the creek bisecting the plot was established in August 2008 so that five samples could be taken within the plot, spaced 1 m apart. Ten additional samples were taken along the transect, but located on either side outside of the plot, 2 m away from the apparent circumference, and with 0.5 m spacing between samples. The 15 samples consisted, therefore, of 5 samples outside the experimental treatment circle on the seaward end, 5 within the plot, and 5 outside the plot on the upland side. Samples were taken from each one of the C, U, UP, LF, and HF treatment areas. These were the only plots that could be sampled in this manner, because the other replicates were too close to either a creek edge or a transition to another plant community. We sampled the marsh soil between 25- and 30-cm depth using a 7.5-cm-diameter core tube with a sharpened end. The depth sampled was chosen to be before the beginning of the experiments (1970). The maximum depth of the 1963–1964-dated horizon for all C, U, P, LF, and HF plots was 19.2 cm. This means that the 25-cm depth is representative of a soil dated circa 1952 or older, but was also within the rooting zone when the experiments began. The bottom 2 cm of sediment was placed in a plastic bag, leaving exposed sediment at the bottom of the core tube. The torque required to shear the exposed soil surface remaining in the core tube was measured using a soil shear vane resistance device (Torvane H-24212 1, Humbolt Manufacturing) and then the remaining core material was returned to the marsh hole. The point of making these shear vane measurements was to obtain a relative value of plant decomposition, based on the presumption that the integrity of the root matrix is directly related to shear vane values. The organic and inorganic contents of the retrieved sediments were determined as described for the dated sediment cores. A Student's *t*-test was run on the data obtained inside compared to outside of the treatment plots.

We also determined the organic:inorganic ratios in the ^{137}Cs -dated cores taken within the treatment plots and 2 m outside the plot. These samples came from the one U, two UP, and two HF treatment plots that had at least 5 cm of sample immediately below the 1963–1964 ^{137}Cs -dated soil horizon. These samples had no Milorganite in them because the sample was formed before the experimental treatments began. We calculated the organic:inorganic ratios in this 5-cm layer, which is dated before the experiments started.

Results

Soils—Data on the ^{137}Cs inventory, the accretion rate, and the accumulated organic and inorganic material above

Table 1. The ^{137}Cs inventory (pCi cm^{-2}), accretion rate (cm yr^{-1}), and accumulation of organic and inorganic matter (g cm^{-2}) above the 1963 horizon in the experiment and control plots. Treatment abbreviations: XHF, extra high fertilizer; HF, high fertilizer; U, urea; U+P, urea+phosphate. The ^{137}Cs profile downcore is in Fig. 2.

Treatment	Core no.	Plot	Experiment	^{137}Cs inventory (pCi cm^{-2})	Accretion rate (cm yr^{-1})	Sum organic above 1963 (g cm^{-2})	Sum inorganic above 1963 (g cm^{-2})
XHF	103	8	Control	0.48	0.25	1.25	0.51
XHF	216	8	Control	0.48	0.26	1.30	0.47
XHF	210	8	Treatment	0.61	0.64	3.13	2.18
XHF	102	8	Treatment	0.69	0.61	2.84	2.23
XHF	11	6	Control	0.59	0.43	1.27	1.19
XHF	44	6	Control	0.69	0.43	1.55	1.15
XHF	204	6	Treatment	1.13	0.59	2.77	2.28
XHF	201	6	Treatment	0.91	0.48	2.33	2.66
U	109	4	Control	0.58	0.37	1.11	0.64
U	208	4	Control	0.77	0.22	0.91	0.47
U	205	4	Treatment	0.73	0.31	1.13	1.01
U	111	10	Treatment	0.64	0.34	1.39	0.93
U	217	10	Treatment	0.28	0.36	1.26	0.86
U+P	206	11	Control	0.53	0.33	1.16	0.62
U+P	213	11	Treatment	0.59	0.37	1.19	0.86
U+P	214	11	Treatment	0.55	0.35	1.36	1.08
U+P	105	12	Control	0.68	0.30	1.43	0.93
U+P	218	12	Treatment	0.85	0.31	1.56	0.80
U+P	212	12	Treatment	0.61	0.31	1.51	0.84
HF	107	9	Control	0.24	0.32	1.26	0.56
HF	207	9	Control	0.31	0.46	1.66	1.18
HF	209	9	Control	0.85	0.44	1.88	1.53
HF	32	9	Control	0.19	0.30	1.16	0.47
HF	47	9	Treatment	0.37	0.40	1.29	1.16
HF	215	9	Treatment	0.48	0.35	1.74	1.21
HF	200	9	Treatment	0.75	0.36	1.63	1.20
HF	12	2	Control	0.74	0.19	0.96	0.65
HF	202	2	Control	0.56	0.15	0.80	1.03
HF	211	2	Treatment	0.35	0.33	1.34	0.78

the 1963–1964 horizon are in Table 1. The average accretion rate for all samples was $0.36 \pm 0.11 \text{ cm yr}^{-1}$ ($\mu \pm 1 \text{ SD}$), and the average organic and inorganic accumulation rates above the 1963–1964 horizon were 1.52 ± 0.57 and $1.09 \pm 0.58 \text{ g cm}^{-2}$, respectively. The ^{137}Cs inventories of individual core segments are shown in Fig. 2. Each soil profile has a distinct peak in the ^{137}Cs activity sandwiched between layers with little or no ^{137}Cs activity. There is no evidence from the ^{137}Cs inventory downcore to suggest that the sites have had significant erosion of the surface layer, storm deposition, or extensive mixing. Erosion would reduce the relative size of the ^{137}Cs peak or remove it, deposition would be revealed by a secondary peak or unusually high deposition rates, and either would cause the ^{137}Cs inventory to vary across the landscape. Extensive mixing would smear the ^{137}Cs peak. The ^{137}Cs inventory among the 13 C sites had a coefficient of variance of 36%, and the absence of these indicators in the ^{137}Cs profile supports the conclusion that the sampled area has neither received a recent deposition of new sediments nor undergone significant erosion of the previously deposited sediments.

The accretion rate (cm yr^{-1}), organic accumulation (g cm^{-2}), inorganic accumulation (g cm^{-2}), bulk density (g cm^{-3}), and ^{137}Cs inventory (pCi cm^{-2}) in the C samples

(outside the treatment plots) were equal to each other for all C, UP, U, HF, and XHF treatment plots (Fig. 3). There was no difference between any of the values for accretion, organic and inorganic accumulation, and bulk density between the paired C and treatment samples (UP, U, and HF), even though the aboveground biomass was very different. There was, however, a higher accretion rate, organic and inorganic accumulation, and ^{137}Cs inventory in the XHF treatment plot compared to in the paired C samples (Fig. 3). The average organic content is $>62\%$ (dry weight) in all soil samples and the bulk density in the upper 10 cm is about 0.16 g cm^{-3} . The volumetric equivalent of the inorganic content, therefore, was calculated to average 6%. This result means that the sediment volume and elevation is determined by the amount of organics accumulating, not the inorganics, and that there is 0.1 g cm^{-3} of organic matter in each 1 cm of marsh accumulation (Fig. 4).

Surface elevation—The marsh surface elevation is deflected downward toward the creeks along a transect perpendicular to the creek bisecting the circular plots. The drop in marsh elevation along this transect 3 m inside the LF and HF treatment plots, compared to 3 m outside, was greater than that in the C plots. Only in the XHF treatment

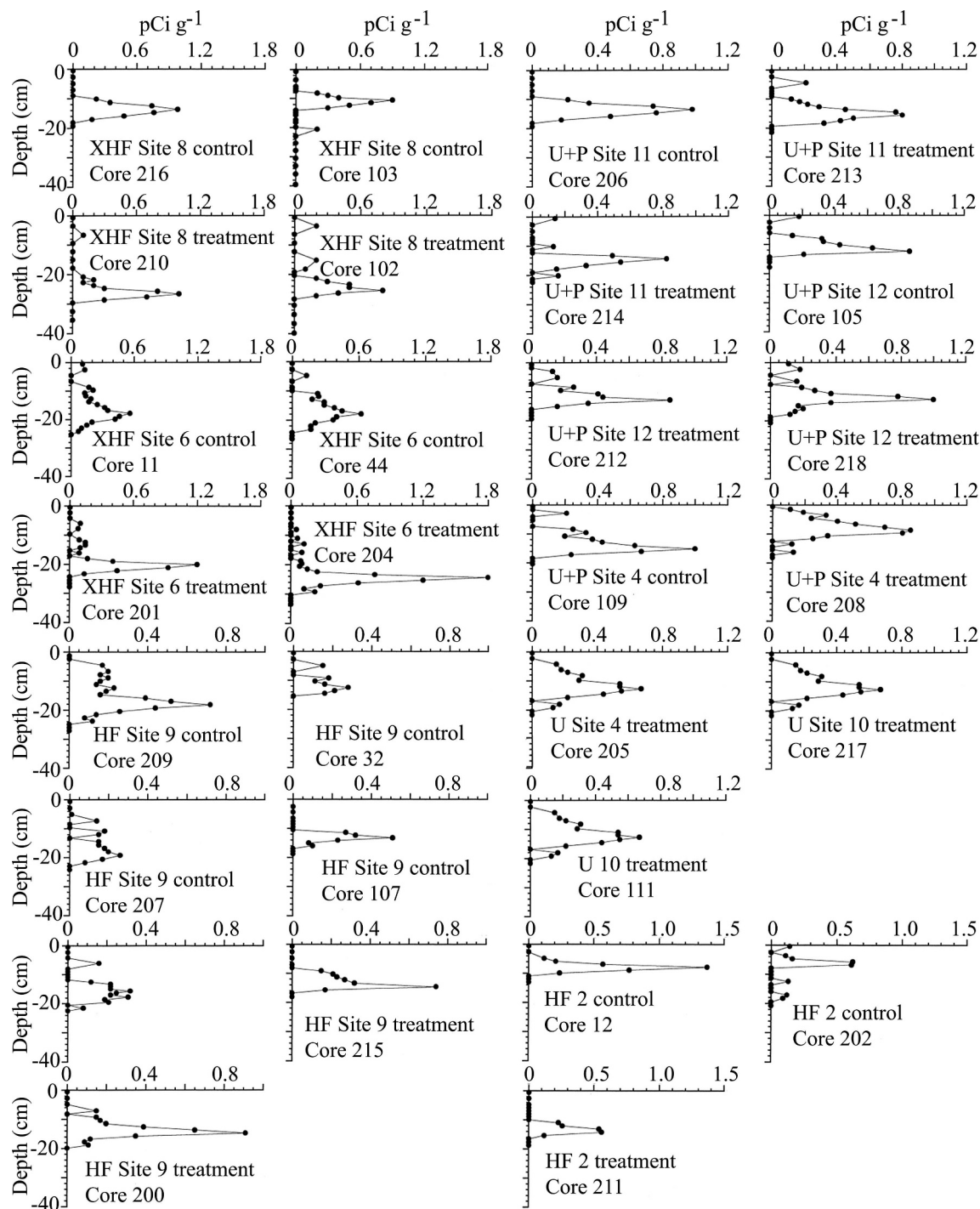


Fig. 2. The ^{137}Cs activity downcore is shown for each core. The abbreviations, core identification number, plot number, and treatment (control or experimental) are the same as in Table 1, which has the ^{137}Cs inventory.

plots was the elevation higher inside of the treatment plot compared to outside of the plot. The XHF treatment plot received the highest amount of fertilizer of all plots, and its volume amounted to about 14 cm over the entire XHF plot over 32 yr. The volume of fertilizer added was subtracted from the observed elevation changes to estimate a potential subsidence rate in the absence of this fertilizer volume (Fig. 5). The average potential subsidence rates at the LF,

HF, and XHF treatment plots were 0.61, 1.5, and 0.80 mm yr^{-1} , respectively, and the average (0.97 mm yr^{-1}) is about half the global sea level rise (1.8 mm yr^{-1} ; IPCC Working Group I Fourth Assessment Report 2007), and 37% of the 1921–1990 record for nearby Boston Harbor (2.65 mm yr^{-1} ; NOAA 2008).

This decline in elevation is equivalent to 10% of the soil volume accumulated above the 1963–1964-dated horizon in

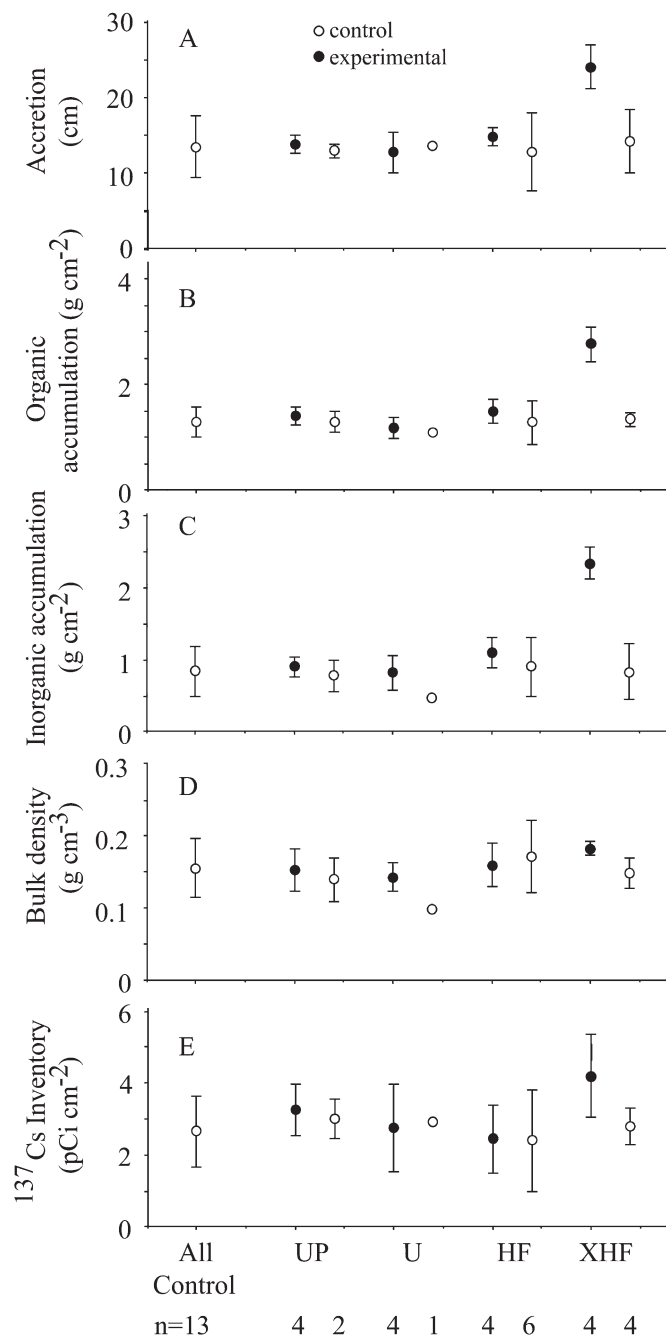


Fig. 3. The variations in measured soil core parameters in all control plots, and the paired control and treatment plots: urea + phosphorus (UP), urea (U), high fertilizer (HF), and extra-high fertilizer (XHF). Values are for the amount above the 1963–1964 soil horizon, except for bulk density, which is the amount in the 0–10 cm surface layer. (A) Accretion (cm), (B) organic matter (g cm^{-2}), (C) inorganic matter (g cm^{-2}), (D) bulk density (g cm^{-3}), and (E) average ^{137}Cs inventory (pCi cm^{-2}). The mean ± 1 SD are shown. n is the number of samples used to compute $\mu \pm 1$ SD.

the C plots. This 10% change is masked in the upper profile of the marsh soil profile because of the volume of the Milorganite fertilizer amendment, which has an ash content of 27% by weight.

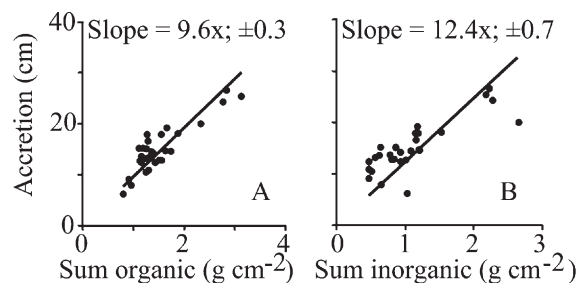


Fig. 4. The relationship between the accretion (cm) above the 1963–1964 horizon and the accumulation of (A) organic matter and (B) inorganic matter in 29 dated sediment cores. A simple linear regression of each data set is shown with the slope forced through the origin (0,0). The slope ± 1 SD is shown.

Shear vane and organic: inorganic content—The organic: inorganic ratio in the 5-cm layer beneath the 1963–1964 soil horizon was $31\% \pm 7\%$ ($\mu \pm 1$ SD) lower inside the five treatment plots (U, UP, and HF treatments), compared to the dated cores taken outside the treatment area.

The shear vane resistance and the organic: inorganic ratio in the horizon deposited before the start of the experiments were directly related, and were lower in the soil beneath the treatment plots compared to the marsh immediately outside the plots (Fig. 6). The C plot had the same shear vane strength inside and outside of the plot. The LF treatment plot had a lower shear vane strength inside the plot compared to outside, but the difference was not statistically significant, whereas the shear vane strength in the HF, U, and UP treatments was lower inside the plots compared to outside the plots. The average decline in the shear vane strength in the LF, HF, U, and UP plots was 36%. The average changes of the organic: inorganic ratio in the C, LF, HF, U, and UP experimental plots were +1%, -9%, -10%, -3%, and -7%, respectively. The average change of the organic: inorganic ratio in plots with N+P added was, therefore, -8.6%, which is less than observed in the five plots sampled higher in the soil profile, which were, therefore, closer to the soil surface where the fertilizer was applied. We assumed that the amount of inorganic matter after burial is conservative and concluded that there was a loss of organic matter in the experimental plots, which was coincidental with a loss in shear vane strength.

Discussion

These results unequivocally demonstrate that nutrient amendments did not increase marsh soil accretion rates. The experimental addition of various nutrients led to less organic matter accumulation in the deeper layers and no change in the upper 25 cm, and occurred without an enhanced mineral accretion rate despite the increase in the aboveground plant production. The analysis of the elevation data demonstrated that nutrient enrichment may also result in a significant loss in marsh elevation. The loss of shear vane strength as organic matter declines suggests that a marsh becomes less resistant to erosion by wind, waves, or storms as nutrient loading increases. The size of the nutrient additions in these experiments is not

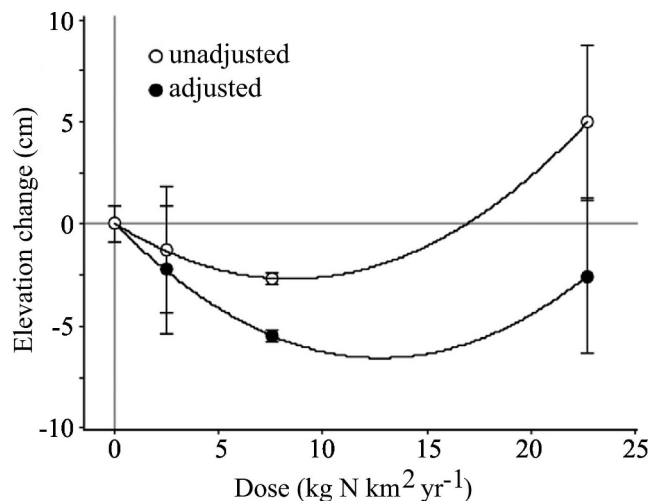


Fig. 5. The average changes in elevation (cm) within the first 3 m inside of the LF, HF, and XHF treatment plots compared to 3 m immediately outside the treatment plots. Negative values indicate a lower elevation inside the treatment plot. The unadjusted data are shown as open circles. The values adjusted for the volume of fertilizer amendments are filled circles. A hyperbolic regression analysis was performed on the relationship between the elevation change and the fertilizer loading rate. The $\mu \pm 1$ SD for two paired plots is shown. Note that the error bars may be as small as the symbol.

unusually higher than that already occurring in many coastal wetlands. There is, therefore, a strong basis to conclude that nutrient enrichment of these salt marshes diminishes, and does not enhance, the chances for marsh sustainability or restoration.

The estimates of soil organic matter changes can be compared to two less-direct measures of organic matter losses resulting from nutrient enrichment of marsh soils. The fertilization of a mineral-rich South Carolina marsh resulted in a $40 \text{ g C m}^{-2} \text{ yr}^{-1}$ loss in soil carbon (Morris and Bradley 1999). This amount of carbon equals about $100 \text{ g m}^{-2} \text{ yr}^{-1}$ loss of organic matter. If the same proportions of carbon lost per nitrogen applied occurred in the Sippewissett marshes we studied, then the elevation change would be 2.7 mm yr^{-1} , which compares to the average 1.1 mm yr^{-1} in the LF and HF plots at Sippewissett, which have similar nitrogen loading rates. Some carbon loss occurs because of the conversion of nitrogen to N_2 (denitrification), which requires carbon as an electron receptor. Most of this carbon loss, however, occurs because there is a tight coupling between soil carbon metabolism and the nitrogen cycle (Howes and White 1994; Hamersley and Howes 2005), which will be stimulated by the addition of a fertilizer amendment. Nitrogen additions, for example, cause an increase in aerobic and N_{ox} respiration, and these high-energy-yield respiratory pathways allow carbon compounds to be respired that are not well-degraded in sulfate-reducing environments. These carbon losses can be estimated using literature values. We assumed (conservatively) that 25% of the nitrogen applied was incorporated into the soil nitrogen cycle, and that 6.1 moles of CO_2 were released for each nitrogen atom (Hamersley and Howes 2005). We used the slope shown in

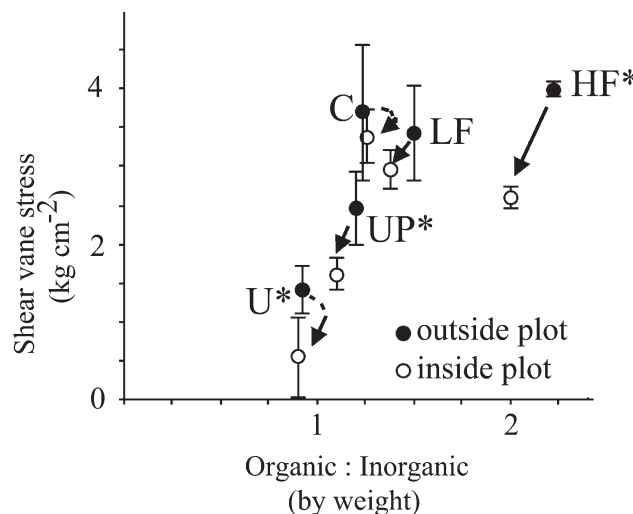


Fig. 6. The relationship between shear vane stress (kg cm^{-2}) on the marsh soil at 25–30 cm below the surface and the organic:inorganic ratio (by weight) for samples taken along a transect parallel to the creek bisecting the plot. The results for five treatments are shown for inside (filled circle) and outside (open circle) the plots, with an arrow indicating the change from outside to inside the plot. C=control; LF=low fertilizer; HF=high fertilizer; U=urea; UP=urea+phosphate. The means ± 1 SD are shown. An asterisk indicates a significant statistical difference between the values inside and outside of the plots.

Fig. 4 to estimate that the average elevation change in the LF and HF treatments resulting from this carbon loss would be at least 1.5 mm yr^{-1} . The estimated change in elevation between C and treatment plots at our study sites (1.1 mm yr^{-1}) is, therefore, in the range of what can be estimated using literature values.

There are no continuous measurements of decomposition rates or belowground production rates at these sites, and so we cannot estimate the relative influence of each in the overall belowground carbon balance. We know of no comparable in situ salt marsh belowground biomass studies with treatments lasting more than a few years. Measurements of root and rhizome biomass in experimental plots receiving nutrient additions for 1–3 yr suggest that root foraging for nutrients is eased, because the amount of live belowground biomass decreases as nutrient loading increases (Valiela et al. 1976; Darby and Turner 2008a,b).

Although the aboveground biomass in the treatment plots was more than twice that in the C plots, there was no additional accumulation of inorganics or organics in any treatment plot except for the XHF plots. The XHF treatment plots have the dead trunks of shrubs in them, indicating a change in belowground biomass quality. Shrubs, for example, have more lignin than grasses, and lignin is resistant to decomposition (Gill and Jackson 2000). The higher ^{137}Cs inventory in the XHF plots, but not in the other plots that also had enhanced aboveground biomass, supports the idea that the retention of the Milorganite may have contributed to this higher ^{137}Cs inventory. An alternative explanation is that perennial shrubs, which grew after the fertilization treatment started

at the XHF sites, are a more efficient collector of the atmospheric sources of ^{137}Cs than are annuals (grasses).

Our principal observation, that nutrient enrichment of these salt marshes did not enhance soil accumulation but did reduce carbon storage to cause a consequential increase in soil subsidence, should apply to other wetlands and even other soil types. This is an important observation because wetlands store around 10% of the world's soil carbon (Watson et al. 2000). Nutrient additions increase the rate of organic decomposition, for example, in wet agricultural soils (Harris et al. 1962), in dry agricultural soils (Khan et al. 2007), in the Everglades (Qualls and Richardson 2008), in long-term experiments in tundra soils (Mack et al. 2004), and in bogs (Bragazza et al. 2006). Furthermore, Swarzenski et al. (2008) observed a decline in soil shear vane strength and increases in various other indices of organic decomposition that they attributed to the nutrient loading for a fresh marsh in south Louisiana. The salt marshes most vulnerable to changes in elevation will be those organic-rich salt marshes at the low end of their elevation range and exposed to relatively high nutrient loading. The expression of these effects, however, might take decades to appear, by which time marsh health is on an irreversible trajectory to an undesirable demise. An example of this situation might be in Jamaica Bay, New York, an estuary that has lost much of its salt marsh to fragmentation, and has had significant marsh dieback (Hartig et al. 2002). Nitrogen and phosphorus concentration in the bay have increased over the last 20 yr to the point where it has some of the highest indicators of eutrophication in the New York metropolitan area (O'Shea and Brosnan 2000). Further, the additional synergistic effects arising from eutrophication, described here, and sea level rise are not likely to be a compensatory influence on marsh stability.

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