Soil Shear Strength Losses in Two Fresh Marshes with Variable Increases in N and P Loading

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Abstract
We measured soil shear strength (SSS) from 2009 to 2018 in two hydrologically distinct freshwater marshes dominated by *Panicum hemitomon* after nitrogen (N) and phosphorous (P) were applied to the surface in spring. The SSS averaged over 100-cm depth in the floating and anchored marshes declined up to 30% throughout the profiles and with no apparent differences in the effects of the low, medium, and high N + P dosing. Plots with only N or P additions exhibited significant changes in SSS at individual depths below 40 cm for the anchored marsh, but not the floating marsh. The average SSS for the anchored marsh over the entire 100 cm profile declined when N and P were added separately or together. At the floating marsh, however, the SSS decreased when N and P were added in combination, or P alone, but not for the N addition. Increasing nutrient availability to these freshwater marsh soils makes them weaker, and perhaps lost if eroded or uplifted by buoyant forces during storms. These results are consistent with results from multi-year experiments demonstrating higher decomposition rates, greenhouse gas emissions, and carbon losses in wetlands following increased nutrient availability.

Keywords Fresh marsh · Eutrophication · Soil degradation · Soil shear strength · *Panicum hemitomon*

Introduction

The rising concentrations of nitrogen, phosphorus and sulfur in air, streams, and other waterbodies over the last 100 years (Meybeck 2003; Holtgrieve et al. 2011; Greaver et al. 2012; Stets et al. 2015) are in intimate contact with wetlands whose global extent is approximately 0.43 million km² in the conterminous US (Dahl 2000) and 12.8 million km² globally (Finlayson et al. 1999). The world's wetlands declined 87% since the 1700s, 83% since 1800, and 64–71% since 1900 (Davidson 2014). The remaining wetland area in 2000 was 6–7% of the global land mass (Melton et al. 2013) and holds a disproportionately larger portion of the terrestrial carbon storage, which is one-fifth of the world's soil carbon (Post et al. 1982). Wetlands produce 15–40% of the global methane and N_2O emissions (Melton et al. 2013; Turetsky et al. 2014), which are potent greenhouse gases that may compromise the climate benefits from burying carbon in wetland soils. The results from the few multi-year field experiments in tundra and coastal wetland ecosystems demonstrated that increased nutrient availability may cause a decline in live below-ground biomass, weaken soils, enhance greenhouse gas emissions, and decrease resilience to hurricanes (Mack et al. 2004; Bragazza et al. 2006; Darby and Turner 2008a, b; Deegan et al. 2012; Feller et al. 2015; Hollis and Turner 2018).

Temperate freshwater wetlands are 25% of all wetland area in the conterminous US (Dahl 2000), but there are few measurements in them that integrate how the increased availability of nitrogen and phosphorus affects carbon stocks. The experiments done include short-term mesocosm and

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greenhouse experiments that necessarily focus on a few factors isolated from the recognized many arising from climate, evolutionary pressures and anthropogenic influences. These results are sometimes contradictory, rarely investigate more than the top 30 cm of soil, and are usually less than 1-year long. The results may be a challenge to understand. Güsewell (2002), for example, demonstrated how 16 wetland plants in a floating fen enriched with N and P responded differently in year 1 than in year 2. The results from field experiments are consequences from complexly nuanced soil processes. The results from long-term field experiments in a New England salt marsh took years to express themselves (Deegan et al. 2012), and Eriksson et al. (2010) found that the combined effect of nitrogen and sulfate deposition on methane emission from a peatland experiment reversed after 6 years. Experiments measuring methane emissions from boreal peatlands sometimes showed inconsistent results. For example, peat sods with raised temperatures had increased CH_4 emissions from experiments lasting 1–3 years in the field (Turetsky et al. 2008) and large mesocosm (Updegraff et al. 2001), but not over 5–6 years in other mesocosm experiments (Keller et al. 2004), and reduced emissions over 12 years (Eriksson et al. 2010). Thus, long-term field experiments are advised but, as Bridgham et al. (2006) observed, they are few.

The use of different methodologies may confound interpretations. The experiments at Sippewisset, Massachusetts, USA, for example, showed that experimental additions of sewage sludge over 2 to 3 years had a > 50% decline in the root mass in the top 15–25 cm compared to control sites, whereas root growth more than doubled over several months inside hollowed-out 20 cm diameter ingrowth holes filled with sand (Valiela et al. 1976). Graham and Mendelssohn (2016) explained this apparent contradiction 40 years later by showing how adding nutrients to in-growth cores stimulated exploitation of the accommodation space, and that the consequences for biomass equilibrium in mature soil fertilization experiments is the down-regulation of the belowground standing crop. Furthermore, experiments vary by the kind and amounts of elements added, depth sampled, and vegetation type, and whether the biomass sampled is live or dead roots, rhizomes, or macro-organic matter. The variations in hydrology, soil type and salinity among sites are also important. The value of multi-year field and comparative experiments seems quite clear.

Here we describe the results of a decade-long (2009–2018) field experiment testing if soil shear stress (SSS) changes after adding N and P to two freshwater wetlands distinguished by whether or not they are capable of floating or if they are anchored to the soil. A standard measure of SSS was made to document the experimental effect because it is non-destructive and can be repeatedly made within a 100 cm deep soil profile.

Materials and Methods

Study Area

We sampled two freshwater marshes dominated by *Panicum hemitomon* in the Mississippi River Delta. These marshes cover about 25% of the area of fresh marsh vegetation in coastal Louisiana (807 km²; Chabreck 1972). *P. hemitomon* is a keystone species creating a belowground accumulation of living and dead roots and rhizomes within an organic soil matrix of about 1 m thickness and a density of <0.10 g cm⁻³. Water levels can sometimes fluctuate by more than 0.5 m in a week and are dependent on wind direction and strength that varies seasonally and with local storm conditions. The floating marsh surface rises directly with increases in water level that can reach more than 1 m and is never inundated (Swarzenski et al. 1991). The anchored marsh surface moves only a few cm when surface water levels increase and becomes submerged during high water events (Turner et al. 2018). Both areas are in interior marsh sites, are 1–3 km from the closest bayou, and were accessed by airboat across the solid marsh they were embedded in.

The first site is in the Joyce Wildlife Management Area (WMA). The site is an anchored marsh located in the Lake Pontchartrain Basin watershed, south of Hammond, Louisiana (USA). It is one of several marshes aligned east to west (Fig. 1) and overlies the mostly sand-clay remnants of a former embayment of the Gulf of Mexico. It is surrounded by forested wetlands and is comprised of a homogenous community of *P. hemitomon* (Fig. 1b). The plant composition is similar to a marsh within 3 km that was dated as 1100 years ±100 years old (Bodker et al. 2015).

The second site in the Salvador WMA located in the Barataria watershed. It is bounded by Lake Cataouatche to the northeast, Bayou Couba to the east, and Lake Salvador to the south. *P. hemitomon* dominates a root mat that is mixed with *Sagittaria latifolia* (Fig. 1d). The 30–50 cm thick marsh mat floats freely with changes in water level except when water levels are too low (Swarzenski et al. 1991) and has as many as 20–22 species in a 0.5 m² plot (Fig. 1e, f). *P. hemitomon* makes up about 20–30% of aboveground biomass in September. Separating belowground biomass by species is impractical, but *P. hemitomon* likely makes up the majority (Holm 2006). The shrub-scrub wax myrtle (*Morella cerifera*) grows in the immediate area, and its spread is controlled by the occasional marsh burn.

Experimental Design

The experiments in the two marshes were done from boardwalks marked with 1- × 1-m square plots placed on alternate sides of the boardwalk and separated from each other by 1 m or more. The anchored marsh boardwalk was established

Fig. 1 The two study sites (anchored marsh, **a–c**; floating marsh, **d**, **e**, **f**) and a 60 cm soil core from the floating marsh. **a** Homogenous *P. hemitomon* ground vegetation at the anchored marsh; **b** Aerial imagery of the homogenous vegetative cover at the anchored marsh; **c** Open water formed from the marsh to the right of 'B' after partially-treated sewage was added in 2009; **d** Illustrative soil core from the floating marsh showing different soil horizons; **e** and **f** Mixed vegetative cover at the floating marsh. **g** and **h** Schematic of the sampling plots located along a boardwalk in the Salvador and Joyce Wildlife Management Areas (WMAs), respectively; C = control, L, M and H = low, medium and high Osmocote® plots, N = nitrogen only plots; P = phosphorous only plots. The arrows point to the general location of the study areas in the insert map
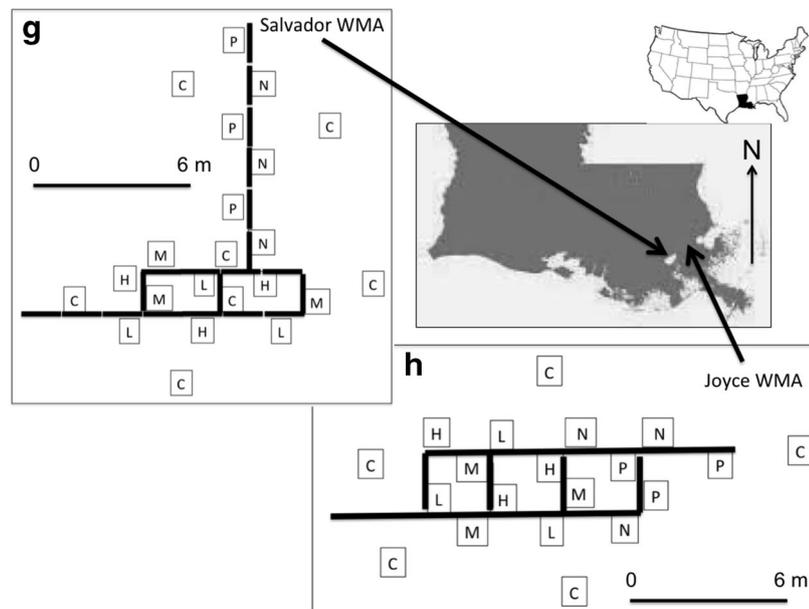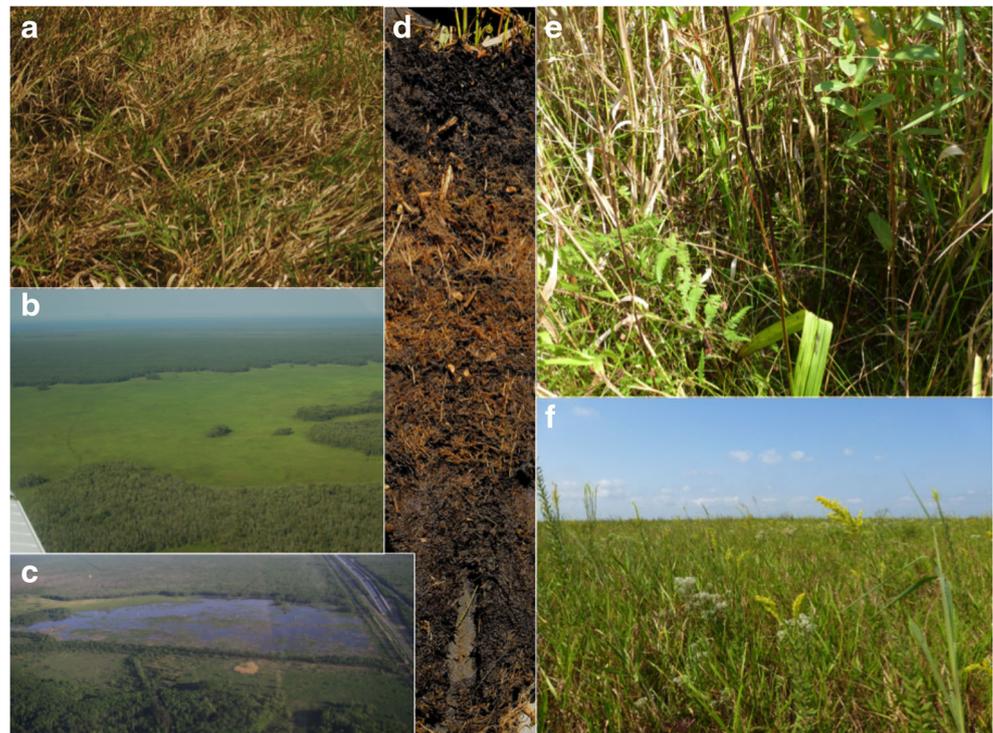
April 2009, and the initial N and P doses were made in June 2009. Subsequent doses were made in April 2010 and each year in the spring from 2013 to 2018. The dosing at the floating marsh began in July 2009 and lasted until March 2015. Each year there were three different levels of surficial additions of slow-release Osmocote® (0.45, 1.36 and 2.26 kg N ha^{-1}, and 0.14, 0.43 and 0.71 P ha^{-1}, respectively), and single doses of super phosphate (Ca(H$_2$PO$_4$)$_2$); 0.52 kg P ha^{-1}), or calcium nitrate (Ca (NO$_2$)$_2$; 0.25 kg N ha^{-1}). The three levels of Osmocote®, and single does of either superphosphate or calcium nitrate are abbreviated herein as Low, Medium High, P and N, respectively. Thus, there were 3 replicate plots each of a control, low, medium and high Osmocote® addition, 3 plots of a P addition, and 3 plots of N additions (Fig. 1g, h). Triplicate plots were spaced along alternating sides of a boardwalk in a sequence of control, low, medium and high nutrient treatments. Five other control plots were 3 m outward from the boardwalk at right angles in three directions (but not where the airboat docked). The total plot number equaled 23 plots. The timing of the

dosage each spring varied depending on weather and hunting restrictions; dosage dates are in the Supplemental Materials (Table S1). The doses used in these experiments ranged from 400 to 1000 kg N ha annually, which are comparable to the loading rates used elsewhere (Morris and Bradley 1999; Deegan et al. 2012; Graham and Mendelssohn 2015; Ratliff et al. 2015), and less than in some estuaries in the Gulf of Mexico (Turner et al. 2000).

Soil Shear Strength Measurements

We used a Dunham E-290 Hand Vane Tester equipped with a 19.5×40-mm vane to measure the SSS of marshes in situ at 10-cm intervals between the surface and 100 cm (Turner 2011). The skin friction without vanes was <0.1 kPa (kilo Pascal). Five replicate measurements were made at each interval within each plot for a total of 1050 individual measurements on each date. The shear vane measurements are reported as the mean ± 1 Standard Error ($\mu \pm 1$ SE). We describe these measurements as 'soil shear strength' (SSS), and recognize that there are many other important aspects of soil integrity involving elasticity, anchoring strength, root biomechanics, etc. (Niklas and Spatz 2012).

Control plots were sampled in April 2009 (anchored marsh) and June 2009 (floating marsh) before the first fertilizer treatments of the same day. Subsequent measurements were taken annually in the springtime (March to May). Five measurements for each 10-cm depth were averaged, and a five-sample average was computed for the 0–100 cm sampling depth. There were variations in the SSS in the control plots from year-to-year that were partially related to seasonal changes in temperature, water level, and the changes in the individual species and community. The responses in the first few years were not measured because of the anticipated differential responses by roots and rhizomes in year 1 and 2 (Holm 2006). The experimental plots at the end of the experiment were 6 and 9 years old for the floating marsh and anchored marsh, respectively. The variability from year-to-year was normalized by calculating differences as a 'response ratio' by measuring changes relative to the control site for springtime measurements made that year.

Porewater

We used a hand-operated syringe collector to draw porewater from 15-cm deep in each plot on four sampling trips. These samples were taken before fertilization on that sampling trip. Two samplings (2014 and 2015) were made in the anchored marsh after it was fertilized in the previous year. The floating marsh was sampled once (2015) after being fertilized in the previous year, and once (2018) when it was not fertilized in the previous year. The porewater samples were brought to our analytical laboratory chilled in an ice chest. All analyses were

checked and verified using standards, blanks and analytical replicates. We kept water samples frozen until determination of the concentration of dissolved forms of inorganic nitrogen (N) and inorganic phosphorus (P) using a Lachat Quick-Chem 8000 Flow Injection Analyzer and the Lachat Methods approved by United State Environmental Protection Agency (available at https://www.wbdg.org/FFC/EPA/EPACRIT/epa600_4_79_020.pdf): method 31–107–06-1-B for ammonium, method 31–107–04-1-C for nitrate/nitrite, and method 31–115–01-1-H for dissolved inorganic phosphate (DIP). A 5-point standard curve was used and QC standards were analyzed before, during and after each set of samples analyzed. The concentration of dissolved inorganic nitrogen (DIN) was calculated as the sum of the dissolved nitrate+nitrite and ammonium concentrations. The coefficient of determination for the standard curve was >0.98 for all nutrient analysis.

Statistics

The mean and standard error of the mean ($\mu \pm 1$ SE) of the porewater concentration of dissolved inorganic nitrogen and phosphate was calculated for each of the three treatment plots. The $\mu \pm 1$ SE of SSS were calculated for each depth in each plot, and in every year, and the average values for the complete 0- to 100-cm depth profile in various combinations of individual plots, treatments and experimental area. The average values for each depth, all depths, and plot treatments were normalized as a ratio of the average control plot value for each year for each site; equal values of SSS in control and experimental treatment are a response ratio of 1. The annual average for each treatment from 2014 through 2018 (inclusive) for the anchored marsh site ($n = 5$ years) and 2011–2015 for the floating marsh site was used to determine if there were significant differences between treatments and control plots ($n = 5$ years). The lowest average kPa value for any depth layer in each treatment for that year was also identified and compared to the lowest in the control plots for that year. A repeated measure ANOVA was performed to test for differences among experimental treatments, and if there were differences in the treatments compared to that in the control plots (degrees of freedom = 4). Each year of data represents a set of matched observations, and sphericity was assumed. The $p < 0.05$.

Results

Control Values

The SSS averaged over 100 cm in the anchored marsh control plots for all years sampled had values ranging between 8.6 and 17.5 kPa (Fig. 2). The average shear vane value for 23 sites at the floating marsh in June 2009, before treatments started, was
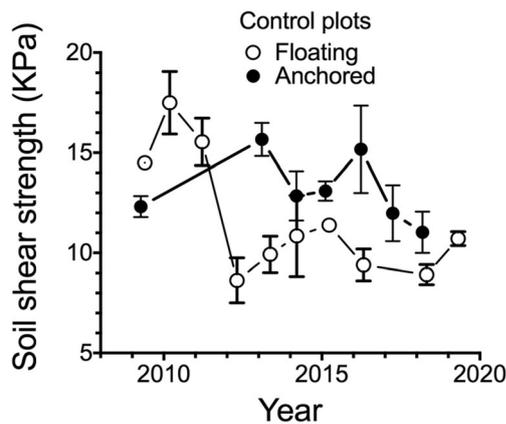
Fig. 2 The average (μ ± 1 SE) SSS in the control plots averaged over 100-cm depth at anchored and floating marshes for each year

14.5 ± 0.86 kPa (0 to 100 cm) and ranged from a low of 8.6 (2012) to 17.5 (2010) kPa. The average SSS over 0–100 cm in the floating marsh control plots from 2009 to 2019 was 11.74 ± 0.96 kPa ($n = 10$). The average pre-treatment SSS in the anchored marsh plots in April 2009 was 12.32 ± 0.53 kPa, and the average SSS from 2009 to 2018 was 13.16 ± 0.63 kPa ($n = 7$). A two-tailed unpaired t-test showed no significant difference between the average of all annual SSS values at the two sites ($p = 0.24$). The SSS in floating and anchored marshes in the same years ($n = 6$) was 10.83 ± 0.82 and 13.36 ± 0.72, respectively, which was not significantly different ($p = 0.09$). The variability of the SSS (kPa) in control marshes in spring at the two sites did not move synchronously; the values at one site were sometimes below or above the other site (Fig. 2).

The shear vane values in control plots after experimental treatments began were higher at the surface and lowest at the 30- to 40-cm depth in the anchored marsh, and lowest at 60- to 70-cm in the floating marsh (Fig. 3). The values in the 90–100 cm layer were 23.74 ± 3.53 kPa in the anchored marsh compared to 13.84 ± 1.32 kPa in the floating marsh. The minimum value was 9.08 ± 0.081 kPa in the anchored marsh compared to a weaker 7.27 ± 0.23 kPa in the floating marsh. This minimum value in the floating marsh compared to in the anchored marsh is consistent with the idea that the lower layer was filled with detrital material that was not anchored to the firmer underlying layer unlike an anchored plant. If soil layer was composed of only water, then the SSS would equal zero.

Treatment Effects

Porewater Concentrations

The concentration of porewater DIN and DIP in plots with the low, medium and high fertilizer treatments the previous year were lowest in the control plots and highest in the high-fertilizer treatment plots (Fig. 4a–c). The 2018 porewater

concentrations in the floating marsh with no fertilizer added in 2016 or 2017 were about the same in all plots (Fig. 4d).

Individual Depth Layers

The response ratios with depth for all treatments in the anchored marsh ranged between 0.5 and 1.2 and were visually similar down core, with a noticeable increase up to 30 cm, and then lower values beneath (Fig. 5a, b). But the differences between treatments and control were only significant in the layer from 0 to 10, 40 to 50, 50 to 60, 60 to 70 and 70 to 80 cm for all treatments in the anchored marsh (F = 3.2, 9, 21, 4.7 and 6.2, respectively; df = 4). There were no differences within a 10 cm thick layer in any of the depth intervals between control and treatment effects in the floating marsh (Fig. 5c, d).

Annual Changes

There were no changes in the response ratio for the anchored marsh by 2013, but there was a decline afterwards in a similar pattern for all low, medium and high treatments and N and P treatments (Fig. 6a,b). The response ratios for the low, medium and high treatments in the floating marsh also declined overall after the first year of treatment, except for a one-year increase in 2011 for the high treatment response ratio. Response ratios for all treatments in the floating marsh increased from 2013 onward (Fig. 6c). The response ratio did not change after adding either N or P separately except during 2013 (Fig. 6d).

Overall Changes

The mean response ratio for the 0–100 cm layer in 5 years at both marshes and all treatments ranged between 0.69 and 1.08 and was between 0.56 and 0.82 for the minimum values in the same data set (Fig. 7). The response ratio in the anchored marsh was lower than 1.0 in all treatment plots for the average of all years (Fig. 7a; F = 14.2, $p < 0.05$, df = 4), and also lower than 1.0 for the minimum response ratio in the treatment plots (Fig. 7b; F = 5.5, $p < 0.05$, df = 4). The anchored marsh had no difference in the response ratio among treatments for the average values (F = 3.04, respectively; $p < 0.05$, df = 4), or minimum values (F = 6.79; $p < 0.05$, df = 4).

There was no difference in the average response ratio in the floating marsh between treatments and the control when comparing all treatment plots, except for the N treatment (Fig. 7c; F = 4.0, $p > 0.05$, df = 4). The response ratio of the minimum values, however, was lower in treatment plots compared to the control plots (Fig. 7d; F = 11.6, $p < 0.05$, df = 4), except for the N treatment which was not different.

Fig. 3 Average SSS values (kPa; μ ± 1 SE) in the control plots from 0- to 100-cm depth for each 10-cm soil depth layer. The values are the average of multiple years at the anchored and floating marshes

Anchored (2014-2018)
Floating (2011-2015)
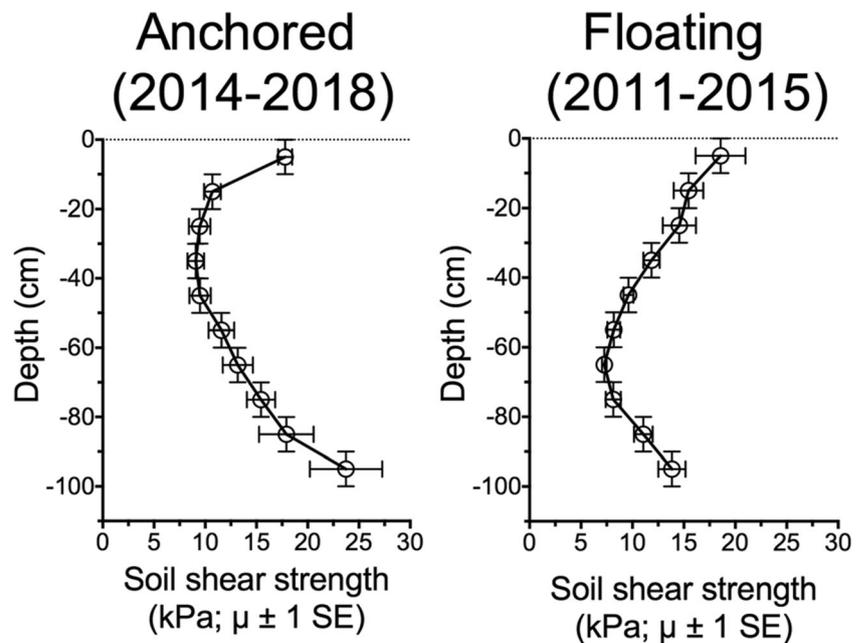
Discussion

Compared to the floating marshes, the anchored marshes had stronger soils over the 100-cm profile. The SSS of anchored and floating marshes were equal in the surface layer, but strongest in the anchored marshes at the bottom. Anchored marshes were weaker between 30 and 40 cm deep, compared to between 60 and 70 cm in the floating marsh. Increasing the nutrient availability in the anchored marsh resulted in a weaker soil profile as a whole, and at the weakest point, whereas the

SSS of the floating marsh declined only at the weakest point. The response ratios among individual treatments were below 1 in both marshes, but not for the N treatment in the floating marsh. A side-by-side comparison increasing either N or P in the floating marsh showed no difference on the average values of SSS for all years, but could be distinguished as lower in the P addition treatment for the minimum values. Increasing nutrient availability to a floating and anchored marsh reduced the SSS in the weakest layer throughout the 100 cm profile, but in different amounts and depths.

Fig. 4 Porewater concentrations of dissolved inorganic nitrogen and dissolved inorganic phosphate in the control, low, medium and high plots on four dates. No fertilizer treatment was made in the previous year in the floating marsh (D)
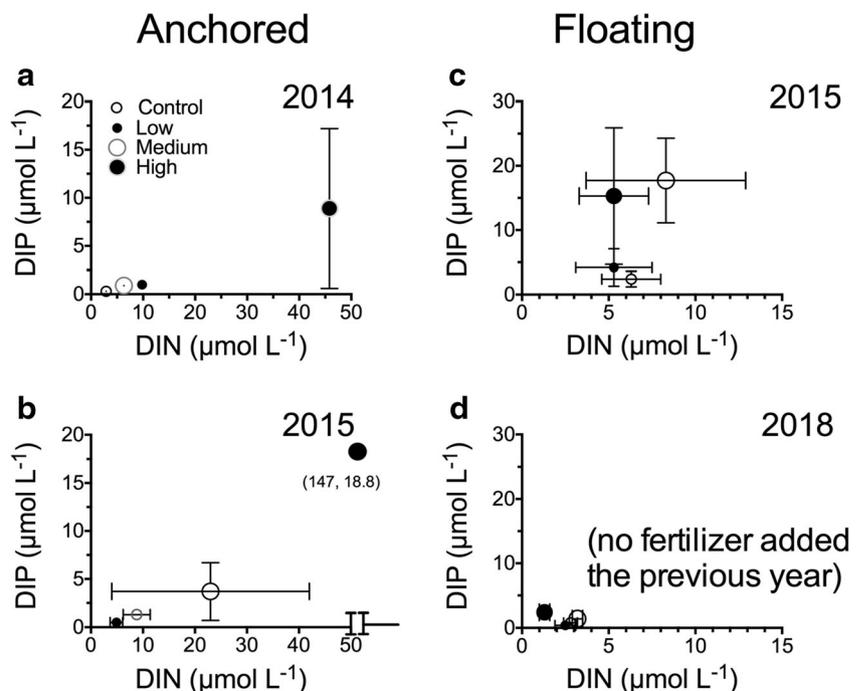
Fig. 5 Average SSS (μ ± 1 SE) with depth for different treatments shown as a response ratio. **a** anchored marsh low, medium and high treatments. **b** anchored marsh nitrogen (N) and super phosphate (P) treatments. **c** floating marsh low, medium, and high treatments. **d** floating marsh nitrogen (N) and phosphate (P) treatments. Significant differences between the control ($p < 0.05$) values are shown with an asterisk to the left in each of the four panels
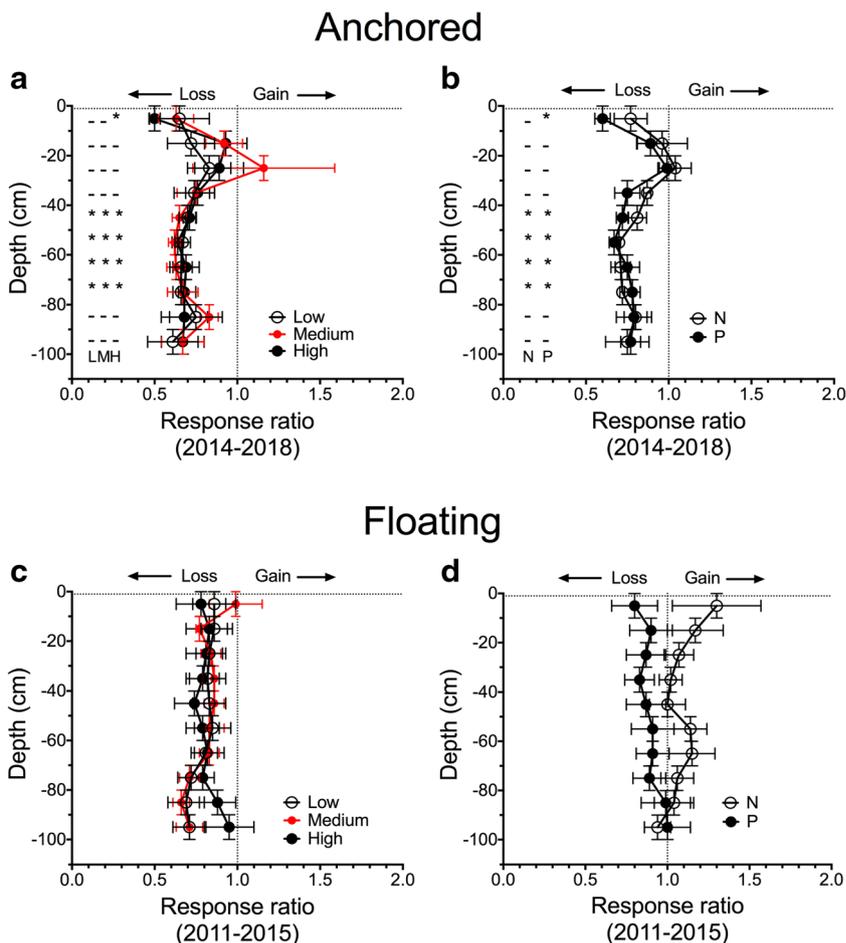
Weakening of the entire profile is indicative of complex belowground interactions reaching a depth of at least 100 cm that involves plants, microbes and biogeochemistry. There are at least two implications for marsh health. First,

Fig. 6 The response ratio for each year in all treatment plots. **a** anchored marsh low, medium and high treatments. **b** anchored marsh N and P treatments. **c** floating marsh low, medium and high treatments. **d** floating marsh N and P treatments
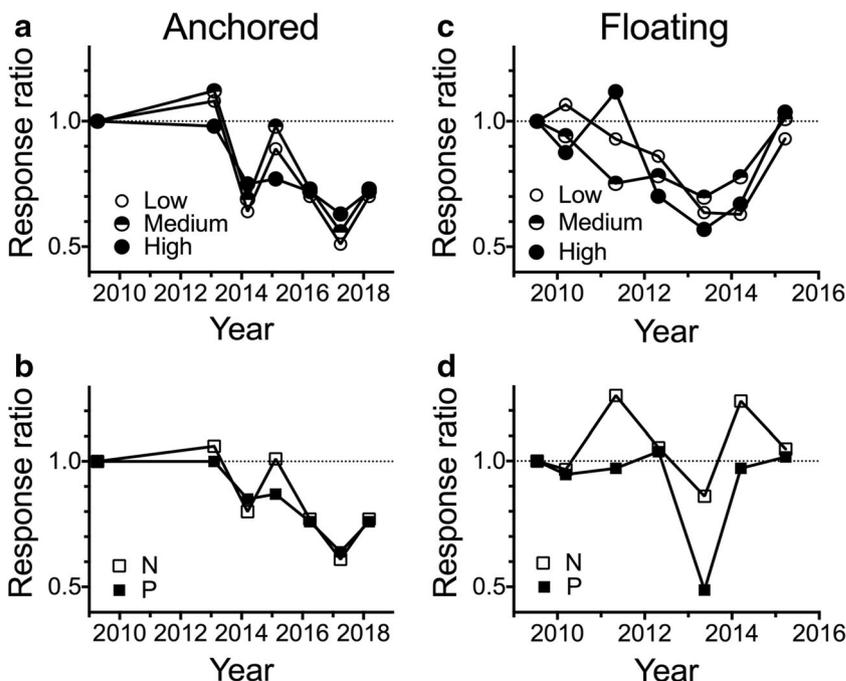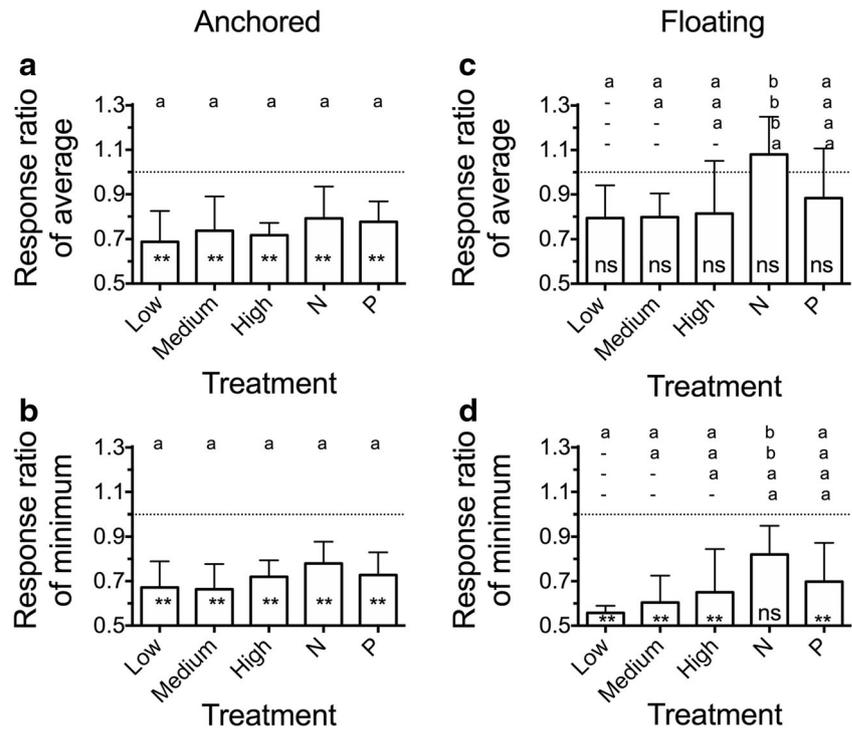
weakening increases the marsh susceptibility to erosion at the marsh edge and vertically if the marsh buoyancy exceeds the anchoring strength to sediments. The discontinuities in SSS, for example, contributed to bank failure and erosion in salt marshes enriched over many years with fertilizer (Wigand et al. 2018). The shearing zone was about 50-cm deep in the *P. hemitomon* marshes in a nearby anchored marsh (Turner et al. 2018), and in the flow path of Mississippi River water diverted into marshes (Kearney et al. 2011). Buoyant forces lift a healthy marsh mat upwards without adverse consequences when the mat movement horizontally is constrained by the mat itself or by boundaries like levees. Second, a faster soil decomposition rate is likely with greater nutrient availability (Grasset et al. 2017; Song et al. 2019). Swarzenski et al. (2008), for example, concluded that the entire 50-cm soil profile at the Penchant freshwater marsh in nearby Terrebonne Parish was affected by more than 40 years of exposure to nutrient-rich water from the Atchafalaya River. Organic matter decomposition was enhanced and the upper 50 cm of soil mats were degraded compared to identical marshes whose source of freshwater was rain. The more decomposed mat made the affected marshes more susceptible to erosion during infrequent high-energy events (e.g., hurricanes) and regular low-energy events, such as tides and during weather front passages.

The variable SSS response to nutrient additions down core and between sites illustrates some of the conceptual fragility of extrapolating results from a shallow core sample < 30 cm to deeper layers, or using shallow cores in greenhouse experiments. Increasing nutrient availability at these two sites

resulted in a decreased SSS within the 100-cm profile, but not equally at all depths at a site or among sites. This finding is important because field studies that do not sample beneath the rooting depth, which is usually 30 cm, miss detecting important experimental effects. Many of the vegetative strands of the anchored marsh are dead and co-mingled within the peat mass that is stronger at the bottom of the profile. There is a possibility that the roots and rhizome of the perennial community below this rooting depth are physiologically active and responding directly to the enhanced nutrient availability. Dead roots contribute to soil strength by themselves and contribute to soil strength. Hollis and Turner (2018), for example, found that dead roots were stronger than live roots of the same size. If the root matrix is bound to the soil mostly through an interweaving with dead roots, then their decomposition may irreparably compromise mat integrity.

A declining SSS may also be because of the consequences to the soil flora and fauna, and not just from changes to roots, per se. It has long been recognized that the relatively more numerous, longer and long-lived roots of plants living in sparse nutrient environments have more mycorrhizal masses than in nutrient-enhanced environments (Chapin 1980). Microbial communities may also be affecting the strength of dead material quite differently throughout the soil profile through decomposition (Craine et al. 2007; Dickinson et al. 1993; Fox et al. 2012; Kominoski et al. 2015; Song et al. 2019), root growth and root quality. The effect of increasing nutrient availability, then, will also vary depending on the effect on live and dead belowground biomass of different ages, hence depth. We recognize, therefore, that these SSS

measurements are indicative of many possible influences, and are integrative, but not process-specific.

Soil Shear Strength over Time

Holm (2006) fertilized a *P. hemitomon* marsh at western (Francis Camp) and eastern (Lake Bouef) Terrebonne Parish, Louisiana, USA, over 2 years and compared how root and rhizome biomass in the first 30 cm changed compared to the control plots. The belowground biomass of fertilized plots in year 1 was 161% of the control value, but 75% of that in the control plot in year 2. The root mass and SSS were strongly correlated in the surface layer for a variety of coastal plants in Louisiana (Sasser et al. 2018), and particularly for *P. hemitomon* live belowground biomass (y) which is well-correlated with SSS (x) in the upper 15 cm ($y = 18.7x$-183.8, $R^2 = 0.81$, $n = 40$; Sasser et al. 2013). Fluctuations in SSS in the upper layer can be considered a surrogate for a changing root biomass, but perhaps only a partial correspondence. The measurements of SSS after several years of fertilization in our experiments showed a similar percent decline as observed by Holm (2006) in year 2. The rhizome: root biomass ratio in Holm's study was 1.9: 1 in the first year of fertilization compared to less than 0.5: 1 in the control, but then declined to that in the control plots by year 2. The rhizomes responded quickly in year 1 by producing more biomass, but declined in year 2, which is a cautionary outcome to consider when applying the results from 1- year long experiments to field situations lasting decades. One implication is that a response to an increase in accommodation space of mesocosms and greenhouse experiments may take more than a few years to be expressed.

Nutrient Loading Amounts

The SSS in soils treated with three different amounts of nutrients did not respond proportionally with the increase in nutrient availability. Instead, the SSS declined to the same amount in the low dose, as in the medium and high dose plots. It is possible that there was some nutrient leakage from one plot to another, to cause a smearing of effects. But between every high and low plot there was a control plot that it was compared to, which should minimize the effect of smearing. Further, the porewater concentrations of DIN and DIP were different from the controls and increased with dosing amount. A non-proportional response (equal decline) was measured for the individual root strength of *Spartina patens* exposed to 3 different nutrient additions in a greenhouse study using separate containers for each treatment (Hollis and Turner 2019). An effect on live roots could be understood as being a consequence to the cellular structure occurring as foraging demands are eased when soil porewater becomes replete with nutrients. But there are also changes below this rooting zone, where microflora slowly decompose dead material in an anaerobic environment. That slow decomposition rate is one reason why there is a disproportionately larger carbon pool per m^2 in wetlands than in aerobic soils. It seems, therefore, that the consequences of enhanced nutrient availability include changes in both root structure and amount, and detrital quality.

Marshes with different mineral content and plant communities, or deeper depths to the Pleistocene-aged deposits might react quite differently. Plants have diverse root structures, which will have a consequential effect on root strength, root density, root size distribution, and, eventually, SSS values. Increasing nutrient loading to marshes will change plant community composition, including encouraging invasive species (Green and Galatowitsch 2002; Tyler et al. 2007), although we did not see that effect in these experiments.

Conclusions

This study indicates that increasing the availability of nitrogen and phosphorus is not helpful for organic marshes– even when the extra nutrients are at a low concentration. This is an issue affecting marsh stability (Turner et al. 2009, 2018). The additive effect of flooding stress, decomposition and reduced SSS make the marsh more vulnerable to storms and shoreline erosion, and (perhaps) to other disturbances like fire, grazing and trampling. An important consideration is that keeping carbon in wetlands, not in the atmosphere, counteracts pressures leading to a warmer world. The implication is that less nutrient loading, not more, is the guidance to successfully sustain and restore soil marsh health.

References

Bodker JE, Turner RE, Tweel AW, Schulz C, Swarzenski C (2015) Nutrient-enhanced decomposition of belowground biomass of a freshwater wetland. Aquatic Botany 127:44–52. https://doi.org/10.1016/j.aquabot.2015.08.001

Bragazza L, Freeman C, Jones T, Rydi H, Limpens J, Fenner N, Ellis T, Gerdol R, Hájek M, Hájek T, Iacumin P, Kutnar L, Tahvanainen T, Toberman H (2006) Atmospheric nitrogen deposition promotes carbon loss from peat bogs. Proceedings National Academy Sciences USA 103:19386–19389

Bridgham SD, Megonigal JP, Keller JK, Bliss NB, Trettin C (2006) The carbon balance of north American wetlands. Wetlands 26:889–916

Chabreck RH (1972) Vegetation and soil characteristics of the Louisiana coastal region. Louisiana State University agricultural Experiment Station bulletin 664

Chapin FS (1980) The mineral nutrition of wild plants. Annual Review of Ecology and Systematics 11:233–260

Craine JM, Morrow C, Fierer N (2007) Microbial nitrogen limitation increases decomposition. Ecology 88:2105–2113

Dahl TE (2000) Status and trends of wetlands in the conterminous United States 1986 to 1997. Department of the Interior, Fish and Wildlife Service, Washington, D. C, 82 pp

Darby FA, Turner RE (2008a) Effects of eutrophication on salt marsh roots, rhizomes, and soils. Marine Ecology Progress Series 363:63–70

Darby FA, Turner RE (2008b) Below- and aboveground biomass of *Spartina alterniflora*: response to nutrient addition in a Louisiana salt marsh. Estuaries and Coasts 31:326–334. https://doi.org/10.1007/s12237-008-9037-8

Davidson NC (2014) How much wetland has the world lost? Long-term and recent trends in global wetland area. Marine Freshwater Research 65:934–941

Deegan LA, Johnson DS, Warren RS, Peterson BJ, Fleeger JW, Fagherazzi S, Wollheim WM (2012) Coastal eutrophication as a driver of salt marsh loss. Nature 490:388–392

Dickinson RE, Hauglustaine DA, Heinze C, Holland E, Enríquez S, Duarte CM, Sand-Jensen K (1993) Patterns in decomposition rates among photosynthetic organisms: the importance of detritus C:N:P content. Oecologia 94:457–471

Eriksson T, Öquist MG, Nilsson MB (2010) Effects of decadal deposition of nitrogen and sulfur, and increased temperature, on methane emissions from a boreal peatland. Journal Geophysical Research 115:G04036. https://doi.org/10.1029/2010JG001285

Feller IC, Dangremond EM, Devlin DJ, Lovelock CE, Proffitt CE, Rodriguez W (2015) Nutrient enrichment intensifies hurricane impact in scrub mangrove ecosystems in the Indian River lagoon, Florida, USA. Ecology 96:2960–2972

Finlayson CM, Davidson NC, Spiers AG, Stevenson NJ (1999) Global wetland inventory – current status and future priorities. Marine Freshwater Research 50:717–727

Fox L, Valiela I, Kinney E (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 and Coasts 35:445–458

Graham SA, Mendelssohn IA (2015) Coastal wetland stability maintained through counterbalancing accretionary responses to chronic nutrient enrichment. Ecology 95:3271–3283

Graham SA, Mendelssohn IA (2016) Contrasting effects of nutrient enrichment on below-ground biomass in coastal wetlands. Journal of Ecology 104:249–260

Grasset C, Levrey LH, Delolme C, Arthaud F, Bornette G (2017) The interaction between wetland nutrient content and plant quality controls aquatic plant decomposition. Wetlands Wetlands Ecology Management 25:211–219

Greaver TL, Sullivan TJ, Herrick JD, Barber MC, Baron JS, Cosby BJ, Deerhake ME, Dennis RL, Dubois J-JB, Goodale CL, Herlihy AT, Lawrence GB, Liu L, Lynch JA, Novak KJ (2012) Ecological effects of nitrogen and sulfur air pollution in the US: what do we know? Frontiers Ecology Environment 10(7):365–372. https://doi.org/10.1890/110049

Green EK, Galatowitsch SM (2002) Effects of *Phalaris arundinacea* and nitrate-N addition on the establishment of wetland plant communities. Journal Applied Ecology 39:134–144

Güsewell S (2002) Time-dependent effects of fertilization on plant biomass in floating fens. Journal Vegetation Science 13:705–718

Hollis LO, Turner RE (2018) The tensile root strength of five emergent coastal macrophytes. Aquatic Botany 146:39–46

Hollis LO, Turner RE (2019) The tensile root strength of *Spartina patens*: response to atrazine exposure and nutrient addition. Wetlands 39:759–775. https://doi.org/10.1007/s13157-019-01126-1

Holm GO Jr (2006) Nutrient constraints on plant community production and organic matter accumulation of subtropical floating marshes. Ph.D. dissertation, Louisiana State University, Baton Rouge

Holtgrieve GW, Schindler DE, Hobbs WO, Leavitt PR, Ward EJ, Bunting L, Chen G, Finney BP, Gregory-Eaves I, Holmgren S, Lisac MJ, Lisi PJ, Nydick K, Rogers LA, Saros JE, Selbie DT, Shapley MD, Walsh PB, Wolfe AP (2011) A coherent signature of anthropogenic nitrogen deposition to remote watersheds of the northern hemisphere. Science 334:1545–1548

Kearney MS, Riter CA, Turner RE (2011) Freshwater diversions for marsh restoration in Louisiana: twenty-six years of changing vegetative cover and marsh area. Geophysical Research Letters 38:L16405. https://doi.org/10.1029/2011GL047847

Keller JK, White JR, Bridgham SD, Pastor J (2004) Climate change effects on carbon and nitrogen mineralization in peatlands through changes in soil quality. Global Change Biology 10(7):1053–1064. https://doi.org/10.1111/j.1365-2486.2004.00785.x

Kominoski JS, Rosemond AD, Benstead JP, Gulis V, Maerz JC, Manning DWP (2015) Low-to-moderate nitrogen and phosphorus concentrations accelerate microbially driven litter breakdown rates. Ecological Applications 25:856–865. https://doi.org/10.1890/14-1113.1

Mack MC, Schuur EAG, Bret-Harte MS, Shaver GR, Chapin FS (2004) Ecosystem carbon storage in arctic tundra reduced by long-term nutrient fertilization. Nature 431:440–443

Melton JR, Wania R, Hodson EL, Poulter B, Ringeval B, Spahni R, Bohn T, Avis CA, Beerling DJ, Chen G, Eliseev AV, Denisov SN, Hopcroft PO, Lettenmaier DP, Riley WJ, Singarayer JS, Subin ZM, Tian H, Zürcher S, Brovkin V, van Bodegom PM, Kleinen T, Yu ZC, Kaplan JO (2013) Present state of global wetland extent and wetland methane modelling: conclusions from a model inter-comparison project (WETCHIMP). Biogeosciences 10:753–788

Meybeck M (2003) Global analysis of river systems: from earth system controls to Anthropocene syndromes. Philosophical Transactions Royal Society London B 358(1440):1935–1955

Morris JT, Bradley P (1999) Effects of nutrient loading on the carbon balance of coastal wetland environments. Limnology Oceanography 44:699–702

Niklas KJ, Spatz HC (2012) Plant physics. University of Chicago Press, Chicago

Post WM, Emanuel WR, Zinke PJ, Stangenberger AG (1982) Soil carbon pools and world life zones. Nature 298:156–159

Ratliff KM, Braswell AE, Marania M (2015) Spatial response of coastal marshes to increased atmospheric CO_2. Proceedings National Academies Sciences USA 112:15580–15584

Sasser CE, Evers-Hebert E, Milan B, Holm GO Jr. (2013) Relationships of marsh soil strength to vegetation biomass. Final Report to the Louisiana Coastal Protection and Restoration Authority through State of Louisiana Interagency Agreement No. 2503-11-45; Baton Rouge, LA

Sasser CE, Evers-Hebert E, Holm GO Jr, Milan B, Sasser JB, Peterson F, DeLaune RD (2018) Relationships of marsh soil strength to below-ground vegetation biomass in Louisiana coastal marshes. Wetlands 38:401–409

Song Y-Y, Song C-C, Ren J-S, Zhang X-H, Jiang L (2019) Nitrogen input increases *Deyeuxia angustifolia* litter decomposition and enzyme activities in a marshland ecosystem in Sanjiang plain, Northeast China. Wetlands 39:549–577

Stets EG, Kelly VJ, Crawford CG (2015) Regional and temporal differences in nitrate trends discerned from long-term water quality monitoring data. Journal American Water Resources Association (JAWRA) 51(5):1394–1407. https://doi.org/10.1111/1752-1688.12321

Swarzenski CM, Swenson EM, Sasser CE, Gosselink JG (1991) Marsh mat flotation in the Louisiana delta plain. Journal of Ecology 79:999–1011

Swarzenski CM, Doyle TW, Fry B, Hargis TB (2008) Biogeochemical response of organic-rich freshwater marshes in the Louisiana delta plain to chronic river water influx. Biogeochemistry 90:49–63. https://doi.org/10.1007/s10533-008-9230-7

Turetsky MR, Treat CC, Waldrop MP, Waddington JM, Harden JW, McGuire AD (2008) Short-term response of methane fluxes and methanogen activity to water table and soil warming manipulations in an Alaskan peatland. Journal Geophysical Research 113:G00A10

Turetsky MR, Kotowska A, Bubier J, Dise NB, Crill P, Hornibrook ERC, Minkkinen K, Moore TR, Myers-Smith IH, Nykanen H, Olefeldt D, Rinne J, Saarnio S, Shurpali N, Tuittila ES, Waddington JM, White JR, Wickland KP, Wilmking M (2014) A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands. Global Change Biology 20:2183–2197. https://doi.org/10.1111/gcb.12580

Turner RE (2011) Beneath the wetland canopy: loss of soil marsh strength with increasing nutrient load. Estuaries and Coasts 33:1084–1093

Turner RE, Stanley D, Brock D, Pennock J, Rabalais NN (2000) A comparison of independent N-loading estimates for U.S. estuaries. In: Valigura RW, Alexander RB, Castro MS, Meyers TP, Paerl HW, Stacey PE, Turner RE (eds) (2000) nitrogen loading in coastal water bodies: an atmospheric perspective. Coastal and Estuarine Studies Volume No. 57. American Geophysical Union, Washington, D.C., pp 107–118

Turner RE, Howes BL, Teal JM, Milan CS, Swenson EM, Goehringer-Toner D (2009) Salt marshes and eutrophication: an unsustainable outcome. Limnology Oceanography 54:1634–1642

Turner RE, Bodker JE, Schulz C (2018) The belowground intersection of nutrients and buoyancy in a freshwater marsh. Wetlands Ecology Management 26(2):151–159

Tyler AC, Lambrinos JG, Grosholz E (2007) Nitrogen inputs promote the spread of an invasive marsh grass. Ecological Applications 17:1886–1898

Updegraff K, Bridgham SD, Pastor J, Weishampel P, Harth C (2001) Response of CO2 and CH4 emissions from peatlands to warming and water table manipulation. Ecological Applications 11:311–326

Valiela I, Teal JM, Persson NY (1976) Production dynamics of experimentally enriched salt marsh vegetation: belowground biomass. Limnology Oceanography 21:245–252

Wigand C, Watson DEB, Martin R, Johnson DS, Warren S, Hanson A, Davey E, Johnson R, Deegan L (2018) Discontinuities in soil strength contribute to destabilization of nutrient-enriched creeks. Ecosphere 9(8):e02329. https://doi.org/10.1002/ecs2.2329