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Beneath the Salt Marsh Canopy: Loss of Soil Strength with Increasing Nutrient Loads

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Abstract Although the broadly observed increase in nutrient loading rates to coastal waters in the last 100 years may increase aboveground biomass, it also tends to increase soil metabolism and lower root and rhizome biomass—responses that can compromise soil strength. Fourteen different multiyear field combinations of nutrient amendments to salt marshes were made to determine the relationship between soil strength and various nitrogen, phosphorus, and nitrogen+phosphorus loadings. There was a proportional decline in soil strength that reached 35% in the 60- to 100-cm soil layer at the highest loadings and did not level off. These loading rates are equivalent to those in the flow path of the Caernarvon river diversion, a major wetland restoration project near New Orleans; 12% of the wetlands in the flow path were converted to open water in 2005. The increased nutrient loading from the Mississippi River watershed this century has also driven the formation of the low oxygen zone (the "Dead Zone") that forms off the Louisiana–Texas shelf each summer. These results suggest that improving water quality in the watershed will aid the restoration of both offshore waters and coastal wetland ecosystems.

Keywords Wetland · Soil strength · Eutrophication · Sustainability · Louisiana

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Introduction

Among the attributes of coastal wetlands that society values is the ability to reduce wave energy and the height of storm surges, including those from hurricanes. The annual benefits of coastal wetlands in the form of storm surge reductions for all US coastal states are estimated to be about \$8,236 ha^{-1}, for example, and \$1,749 ha^{-1} for Louisiana (Costanza et al. 2008). Plants and soil remain in place only if they have sufficient mechanical strength to anchor them when strong currents and winds push against the resisting aboveground biomass. If the erosive forces are too strong, then the soil layer either degrades in pieces or lifts off of the underlying basement material and an open-water habitat takes its place. One consequence of hurricanes in the microtidal Louisiana marshes is the creation of 2-m high folds in the marsh (the "accordion effect") and "marsh-balls" (Lovelace and McPherson 1998), and to the re-distribution of mineral and organic sediments, as well as the surface debris (Turner et al. 2006).

It may seem that adding nutrients is a worthy management objective if the amount of aboveground biomass is a key factor determining whether wetlands are sustainable. Several studies, however, demonstrate that the soil ecosystem is affected in two important ways when the nutrient load is increased in coastal wetlands. One way is that the belowground biomass may be inversely related to nutrient loading. Root and rhizome biomass declines with increased nutrient loading in many freshwater, brackish, and saltwater coastal wetlands (Holm 2006; Darby and Turner 2008a, b; Langley et al. 2009). This result is consistent with the hypothesis that nutrient foraging is eased as nutrient supply rises in wetland plants. Under these conditions, plants can expend more of their carbon fixation on the accumulation of aboveground biomass (Levin et al. 1989).

A second effect of increased nutrient loading is that the organic decomposition rate of litter lying at the surface and soil organic matter rises, not only for coastal wetlands, but also for a variety inland wetlands (Harris et al. 1962; Eggelsmann 1976; Morris and Bradley 1999; Coûteaux et al. 1995; Bragazza et al. 2006; Laursen 2004; Mack et al. 2004; Franzen 2006; Wigand et al. 2009). Organic-rich freshwater marsh (*Panicum hemitomon*) soil exposed to a chronic influx of nutrient-rich river water was more decomposed, and the soil strength was reduced by 50%, compared with marsh soil substantively identical except that nutrient-poor rain was the source of freshwater (Swarzenski et al. 2008). An analysis of a 30-year nutrient addition experiment in an east coast (US) salt marsh showed that organic soil accumulation did not increase in the fertilized plots and that soil strength and elevation declined (Turner et al. 2009). Some of this carbon loss is because the bacteria use carbon as the electron donor to form nitrogen gas (denitrification) under anaerobic conditions (Hamersley and Howes 2005). Anaerobic metabolism may also result in the increased formation of hydrogen sulfide, nitrous oxide, and methane. Wigand et al. (2009) describe, for example, a pattern of rising soil respiration in a variety of Rhode Island salt marshes as nitrogen loading increased among locations even though the standing stock of belowground biomass declined. The result of nutrient enrichment may be, therefore, that soil carbon is depleted because less belowground organic matter is produced and because a higher proportion of belowground production is metabolized.

The effect of increased nutrient loadings on wetland ecosystem trajectories, of course, may take years to become apparent. Qualls and Richardson (2008), for example, found that decomposition of leaves from the dominant freshwater marsh macrophyte (*Cladium jamaicense* Pers.) was greater with phosphorus enrichment after 1 year, whereas Newman et al. (2001) report no change in litter decomposition with nutrient enrichment after 3 years. Holm (2006) found that nitrogen+phosphorus additions in a *P. hemitomon* marsh did not stimulate decomposition in litter bags placed on the soil surface. Organic matter decomposition in in-growth chambers placed in the soil, however, was faster in the nutrient-enriched chambers after 2 years when compared to organic decomposition in the control chambers. The responses to increased nutrient loadings may also be nonlinear. Penton and Newman (2007, 2008), for example, suggested that there was a P load threshold above which decomposition was triggered and where slower peat accumulation and elevation changes occurred.

These results support the conclusion that multiyear experiments are needed to understand the relationships between soil sustainability and increased nutrient loading to wetlands. Some significant questions need to be addressed, therefore, about how much and how rapidly soil strength is affected by less belowground biomass and higher organic decomposition rates. Is there a significant change in soil strength and, if there is, how does the trajectory of soil strength decline as nutrient loading increases? How do the results of an increased loading of nitrogen, the primary limiting nutrient of the aboveground biomass (Morris 1991), compare to an increased loading of phosphorus, which can have the effect of lowering belowground biomass and increasing soil metabolism (Darby and Turner 2008a, b; Sundareshwar et al. 2003)?

I report herein on how soil strength changes in the upper 100 cm of salt marshes after 14 different combinations of N and P were added to replicated plots. These experiments lasted from 1 to 5 years. I then discuss the implications for marsh restoration and conservation, with attention to some wetland restoration projects near New Orleans, LA.

Methods

The strength of salt marsh soils in coastal Louisiana salt marshes treated with and without different nutrient additions was measured with a Dunham E-290 Hand Vane Tester. This handheld instrument measures the maximum resistance (torque) when a rod fitted with vanes is rotated at different soil depths. The in situ readings of the peak shear vane strength are recorded on a calibrated scale built into a head assembly. The head assembly is directly attached to a steel rod fitted with 25.4×50.8-mm vanes at the terminal end. A handle is used to both insert the vane to the desired depth and to apply the shearing torque. No correction for the skin friction of the extension rods (done with a shaft without vanes) was routinely performed on individual measurements, but selected measurements indicate that that the skin friction was always less than 0.1 kPa (kilopascal). Measurements within the control and treatment plots at 10-cm intervals were compared to obtain the percent shear vane strength of the plots with nutrient additions. There were a minimum of three and a maximum of six vertical profiles for each plot. The mean ±1 standard error ($\mu \pm 1$ SE; $p < 0.1$) was calculated for each 10-cm increment.

Shear vane measurements were made in treatment plots fertilized with various amounts and kinds of nitrogen, phosphorus, or nitrogen+phosphorus treatments (+N, +P, +N+P, respectively; Table 1). The plot locations were in a salt marsh within 1 km of the Louisiana Universities Marine Consortium (LUMCON) at Cocodrie, LA, where the percent organic matter in the upper 30 cm was 21.9± 2.4% (unpublished results). All sampled locations are at least 10 m inland of the streamside levee marsh where the marsh soils are at least 1 m deep and overlay deltaic

Table 1 The nutrient addition treatments, including nutrient dose, sample size, and plot size

Site	Treatment (number)	Plot size(m^2)	Sample size		N source	P source	Start	Cumulative N (1,000 $kgha^{-1}$)	Cumulative P (1,000 $kgha^{-1}$)
			Control	Experimental					
LA									
1	N+ series (5)	0.25	3	3	Ammonium sulfate	None	2004	1.2, 2.3, 4.7, 9.3, 18.6	0
1	P+ (1)	0.25	3	3	None	Superphosphate	2004	0	1.2
1	N+P+ (1)	0.25	3	3	Ammonium sulfate	Superphosphate	2004	18.6	1.2
1	P+ series (3)	0.25	3	3	None	Superphosphate	2006	0.0	0.07, 0.14, 0.26
1	High N+ (1)	0.25	3	3	Blood meal	None	2003	13.0	0
1	High P+ (1)	0.25	3	3	None	Superphosphate	2003	0	28.7
1	High N+P+ (1)	0.25	3	3	Blood meal	Superphosphate	2003	13.0	28.7
2	N+P+ (1)	0.25	9	3	Osmocote	Osmocote	2008	3.6	1.1
MA									
3	N+P+ (2)	78.54	12	1	Milorganite	Milorganite	1970	9.6, 28.7	2.5, 7.5
3	N+ (1)	78.54	12	1	Urea	None	1971	28.0	0.0
3	N+P+ (1)	78.54	12	1	Urea	Superphosphate	1971	28.0	7.3

The sampling was within 1 km of the LUMCON Marine Laboratory, at Cocodrie, LA. The experiment at location 2 is within 0.5 km of site 1 and used for the soil decomposition experiment (cotton strip tensile strength). Experiments at location 3 started in 1970 and 1971 in Massachusetts and are reported in Turner et al. (2009). A listing of the fertilizer components (chemical constituents (by weight) of the fertilizers added) is as follows—ammonium sulfate: 33% N, by weight; 0% P; superphosphate: 18% P, by weight, measured as phosphate (P_2O_5), in the form of 3 [$CaH_4(P_2O_5)2{\cdot}H_2O$]; 0% N. Derived by acidification of fluorapatite (phosphate rock) to form mono-calcium phosphate monohydrate, calcium sulfate, and hydrogen fluoride; blood meal: 12% N and 0% P by weight, respectively, with 0.5% Fe, by weight. Derived from blood meal and ferrous sulfate; Osmocote: a commercially available fertilizer released over 4 months; 10% ammonium N and 9% nitrate N, by weight; 6% available phosphate (P_2O_5) by weight; 12% soluble potash (K_2O), by weight. The manufacturer's description is that the N, P, and potash sources provide 16% slow-release N, 5% slow-release P, and 10% slow-release potash. Percent sulfur, by weight; Milorganite: a commercially available sludge fertilizer using secondary treatment plant sludge from the city of Chicago; 10% N, and 6% P_2O_5, respectively, by weight. Urea: 46% N by weight as a crystalline solid ($(NH_2)_2CO$); 0% P; 5% sulfur mineral soils. Measurements were made in 24 of the thirty-six 0.25-m^2 plots described by Darby and Turner (2008b), which were started in 2004 and continued with 3 to 4 equal doses annually to arrive at the cumulative doses shown in Table 1. The doses in these 24 plots include triplicate treatments of 5 different levels of nitrogen additions (+N), 1 phosphorous treatment (+P), 1 +N+P treatment, and 1 control set (in triplicate). Darby and Turner (2008a) harvested plants from 24 of these plots in 2005 (21 treatments and 3 control plots). Measurements of plant canopy height and stem density in 2006 and 2008 showed no difference in the same plot compared to the 2005 values. The location of the shear vane measurements did not overlap where Darby and Turner (2008a) extracted soil cores in 2005 to measure root density. All plots have had no intentional disturbances by our field survey team.

There were twelve 0.25-m^2 plots of triplicate treatments of control, +N, +P, and +N+P treatments that began in 2003. These latter plots had similar +N treatment as in the plots started in 2004, but with higher +P treatments. Three additional replicated +P treatments (plus controls) were started in 2006 for a total of 48 plots at one marsh, of which 9 were control plots and 39 (13 different treatments) were experimental plots. The total number of triplicate treatments was, therefore, six different +N treatments, five different +P treatments, and two different +N+P treatments. The range of the cumulative dose of N and P was 1,200 to 18,600 kg N ha^{-1} and 70 to 28,700 kg P ha^{-1}.

Vertical profiles of shear vane strength were made at 10-cm increments in each of the control and in the triplicate treatment plots in November 2008. A one-way analysis of variance with a Tukey's multiple comparisons test was used to determine whether there were significant differences in soil strength between the nine control and triplicate treatment plots and among different kinds of treatment plots ($p<0.05$) (Graphpad Software Inc. 2005). The values for the treatment plots were expressed as a percent of the values in the control plots ($\mu\pm1$ SE; $n=3$). A regression of the percent of the control plot versus cumulative N or cumulative P addition was conducted where $\log_{10}Y = a + bx$, and $p<0.05$. Shear vane values obtained from the 13 treatments located in the 1 contiguous marsh were compared to the data described in Turner et al. (2009) for 4 different nutrient enrichment treatments conducted for 30 years at the Great Sippewissett Marsh, Sippewissett, MA.

An additional +N+P treatment was started in 2008 (plus control plots) located within 0.5 km of the experimental treatments near LUMCON. These treatment plots were treated with a cumulative nitrogen and phosphorus dose (Osmocote) of 3,600 and 1,100 kg ha^{-1}, respectively. This site was within 30 m of where soil decomposition measurements were made in 0.5×2-m plots. The loss of the tensile strength of canvas strips buried between the surface and 80 cm was used as a surrogate measure of soil decomposition. The methods of Slocum et al. (2009) were used. Briefly, three 80-cm×10-cm strips of artist's canvas were inserted into the marsh soil within 2×0.5-m plots in September 2009 and retrieved after 2 weeks. Additional strips were inserted and removed at the beginning of the experiment to serve as "disturbed control" treatments. A 2-cm×10-cm strip was at 10-cm intervals from each of three replicates from both control and treatment plots, and its tensile strength measured with a Mecmesin MultiTest 1-D digital motorized test standard and a basic force gage equipped with spring-loaded roller grips. Measurements were made at room temperature. The tensile strength of the disturbed control strips was subtracted from the tensile strength of the canvas strips left in the treatment plots to calculate the percent daily change in tensile strength.

Data on plant aboveground biomass (gram per square meter) and stem density (number per square meter) at these 0.25-m^2 plots were collected at the Louisiana study site in September 2005. The aboveground data are described in Darby and Turner (2008b). The stem density in these same plots and five additional ones at the same location were re-sampled in November 2008, together with shear vane measurements as described above.

Results

The down core profiles of shear vane strength in the control plots show that soil strength at the surface is approximately 11 kPa and declines to about 4.5 kPa between 20 and 50 cm and then rises to 11 kPa at 100 cm (Fig. 1). The data analyses were restricted to the 0- to 100-cm layer because stiff resistance to the penetration of the shear vane begins at around 120 cm in these marshes.

The downcore shear vane strength measurements in soils receiving the +N+P treatments are shown in Fig. 1. These treatment plots received a cumulative N dose that is the highest of all the Louisiana marsh treatments and is in the middle of the range of the cumulative P dose (Table 1). There was no difference in the shear vane strength between

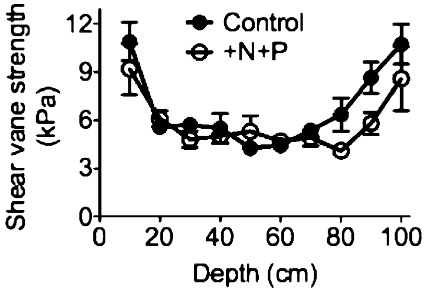

Fig. 1 The shear vane strength (kilopascal; $\mu\pm1$ SE) for replicated control and +N+P treatment plots for the 10- to 100-cm soil depth

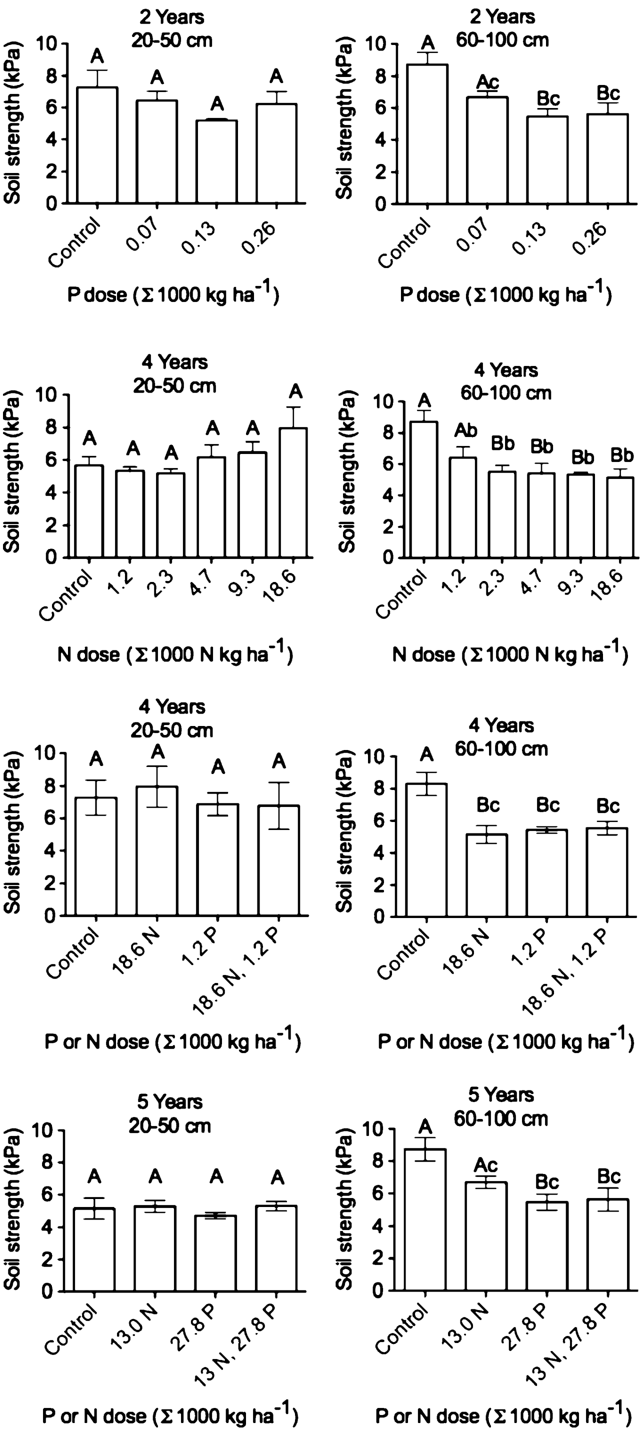

Fig. 2 The average shear vane strength (kPa; $\mu \pm 1$ SE; $n=3$) at 20 to 50 (*left side*) and at 60- to 100-cm soil depth (*right side*) versus the treatment conditions in different plots (C = control; $+N$ = nitrogen; $+P$ = phosphorus). *Means with the same lettering* are not statistically different from each other (level of significance <0.05). The cumulative years of treatment and cumulative N, P, or NP added are indicated in each graph. Note that the shear vane in the control sites are the same in all upper and all lower soil layers and that the shear vane strength in the 4-year experiment is from the same plot (18.6×1,000 kgN ha^{-1} treatment) in two adjoining panels (*second* and *third rows*)

control and +N+P treatments in the upper soil profile (20–50 cm), but the soil strength was reduced by about 21% in the 60- to 100-cm soil layer of the +N+P treatment plots.

The focus of the following discussion is about how soil strength in the upper (top) and deeper (bottom) soil layer differs between control and nutrient-amended plots. The shear vane strength in the 60- to 100-cm layer, but not the layer above, declines with higher loading of either N or P (Fig. 2). There was no statistically significant difference between soil strength in the control and treatment plots in the surface layer (Fig. 2, left, four panels). The shear vane strength in the bottom layer, in contrast, was always lower than that in the control plots when P and N loadings were at least 130 kgPha^{-1} P or 2,000 kgNha^{-1} (Fig. 2, right, four panels). The effect of N and P additions was not significantly different than if P was added separately in the 5-year-old treatment plots, but not in the 4-year-old treatment plots. The relationship between the cumulative +N, +P, or +N+P loading and the relative change in shear vane strength is in Fig. 3. There was a significant relationship between the cumulative nutrient loading and the loss of soil strength, amounting to a maximum of about 35%.

The soil strength in the upper layer of the Louisiana marshes remained relatively constant compared to that in

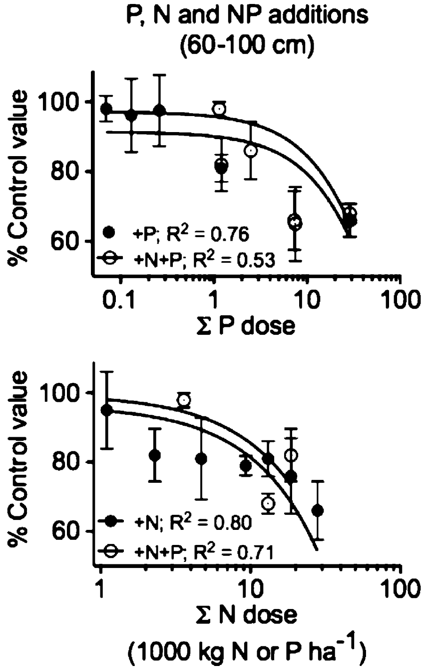

Fig. 3 The average shear vane strength at 60- to 100-cm soil depth as a percent of the value in the control plots versus the cumulative N and P load (Σ 1,000 kgha^{-1}) for plots fertilized with +P or with +N+P (*upper panel*) or plots fertilized with +N and +P+N (*lower panel*). The data are fit to a regression equation of the form log $Y = a + bx$. The coefficient of determination (R^2) is shown in each graph, where *n.s.* = not significant

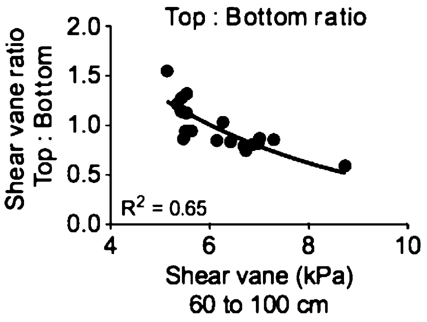

Fig. 4 The ratio of the shear vane strength at the top (20 to 50 cm) and bottom (60 to 100 cm) of a soil profile versus the shear vane strength at the bottom. The data are fit to a regression equation of the form $\log Y = a + bx$. The coefficient of determination (R^2) is 0.65 and $p<0.05$

the soil beneath. As nutrient amendments caused the shear vane strength to decrease in the lower layer, the ratio of soil strength in the two layers (top/bottom) doubled from around 0.7 to 1.5 (Fig. 4).

There is a strong positive relationship between aboveground biomass and stem density (R^2=0.84; Fig. 5, left panel), indicating that either biomass or stem density is a relative surrogate for nitrogen loading, because the accumulation of aboveground biomass is primarily limited by nitrogen (Darby and Turner 2008b). There was an inverse relationship between shear vane strength (at 60- to 100-cm soil depth; relative to the control plots) and stem density that is strongly linear (R^2=0.66; Fig. 5, right panel). This result is consistent with the results in Fig. 2, which show a declining soil strength with higher nutrient loading.

The tensile strength of canvas strips placed in a nearby marsh was 16% to 48% lower in the fertilized plot, compared to the control plot (Fig. 6), implying that the nutrient amendment caused a significantly higher decomposition down to 60 cm, and perhaps deeper. These results are from one experiment and there are, undoubtedly, strong variations during the year.

Discussion

It appears that the effects of the nutrient additions to these salt marshes are to increase the aboveground biomass (Darby and Turner 2008b), lower belowground biomass (Darby and Turner 2008a, b), increase decomposition rates (Fig. 6), and reduce soil strength (Figs. 1, 2, 3, and 4). These are interrelated consequences embedded within a suite of consequences. If soils oxidize and there are few roots and rhizomes, then the soil will become weaker. Some of these results could be inferred as a possibility based on the results of Morris and Bradley's (1999) experiments in a mineral-rich South Carolina salt marsh. Morris and Bradley (1999) found that +N+P fertilization (ammonium nitrate and phosphate; Huang and Morris 2005) for 12 years increased soil respiration by 36%. Morris et al. (2005), however, also describe a rise in surface elevation in the fertilized plots because of the accumulation of inorganic material in the mineral-rich marsh that they sampled. Elevation changes from soil decomposition (subsidence) seem more likely to occur in organic-rich sediments with a relatively small inorganic source for deposition. Other factors besides soil inorganic content will modify the effects of higher nutrient loading on soil organics, e.g., tidal range occupied, climate, and flooding cycles. Temperature, for example, has a well-established direct effect on soil organic decomposition and root turnover (Gill and Jackson 2000), and salt marsh plants have some physiological abilities to adapt to flooding and salt stress (Mendelssohn et al. 1981). It is unlikely, therefore, that all marshes will respond equally to similar increases in nutrient loads.

Nutrient additions caused significant changes in the soils of these Louisiana marshes between 60- and 100-cm soil depth. It is interesting to ask why there is an effect at this deeper layer, and not in the surface layer. The results of sediment dating of these marshes (^{137}Cs dating method;

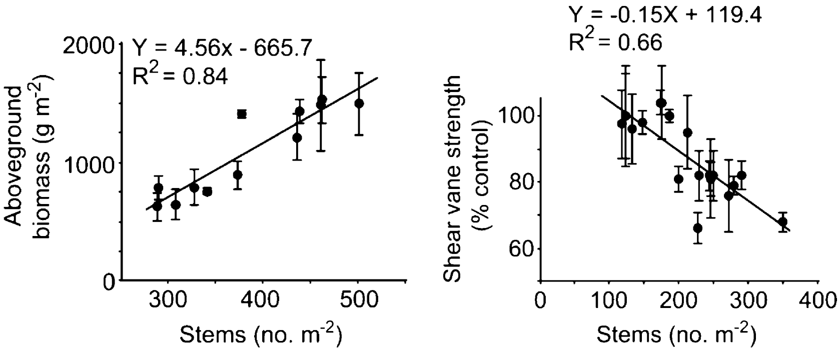

Fig. 5 Relationships between stem density (number per square meter) and both aboveground biomass (*left*; gram per square meter; $\mu\pm1$ SE) and shear vane strength at 60- to 100-cm soil depth (*right*). The shear vane data were converted to the value in the treatment plot as a percent of that in the control plots ($\mu\pm1$ SE). The *data in the left panel* are from 11 different nutrient treatments (in triplicate) sampled in September 2005 and described in Darby and Turner (2008a). The *data in the right panel* are from 16 triplicate plots sampled in November 2008. $p<0.01$ in both graphs

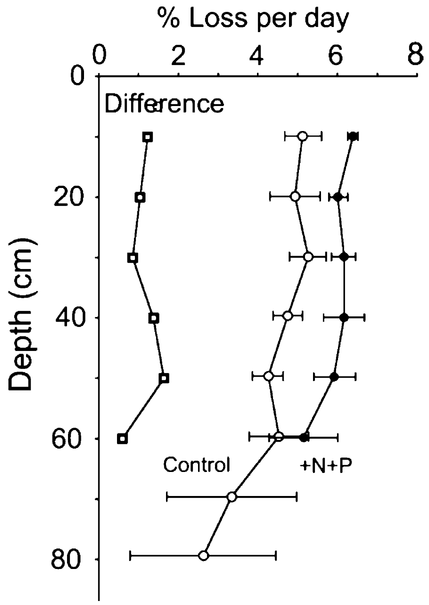

Fig. 6 The change in tensile strength of buried canvas (percent loss per day; ±1SE)

Milan et al. 1995) indicate that the accretion rate since 1963/1964 was 0.60 cmyear^{-1}. If we assume that accretion below 60 cm is nearly the same as above this layer, and disregarding compaction, then the 60-cm soil depth is at least 100 years old and contains organic matter formed before major water quality changes occurred (Parsons et al. 2006; Broussard and Turner 2009). Perhaps the exposure to a new source of phosphorus speeds up the soil microbes enough to cause the loss of soil strength, as organic matter is decomposed (e.g., Sundareshwar et al. 2003). Perhaps a key issue is how metabolism in the rhizosphere is affected by higher nutrient loads. Plant-mediated phosphorus metabolism is a potentially significant factor at these depths, even if the vertical distribution of biomass declines with depth. Although the density of rhizomes declines quickly below 30 cm in these marshes, for example, the fine roots penetrate much deeper and may have relatively higher metabolic activity per weight for nutrient uptake than at the surface. Reimold (1972), for example, demonstrated that the maximum phosphorus uptake of ^{32}P-labeled inorganic phosphate inserted into Georgia salt marsh soil was at 100-cm soil depth. Nutrient loading in recent years, therefore, might have an effect on the salt marsh organic matter that accumulated over several hundred years.

The results of these 5-year-long experiments in Louisiana might be compared to the results from the 30+ year experiments conducted at the Sippewissett marshes (MA). The nutrient amendments were qualitatively different at the two locations (Table 1), and the shear vane strength was measured at only 50 cm in the Sippewissett marshes, and not over the 0 to 100 soil profile measured in Louisiana. The direction and scale of the changes at Sippewissett (Fig. 7) is similar to what was observed in Louisiana: the shear strength declined with increasing nutrient load and by significant amounts for the same amount (20% to 30% at the highest rate).

Deegan et al. (2010) added N and P to the flooding tides of a salt marsh in Massachusetts to create a cumulative load over 2 years of 1,200 kgkm^2 N and 90 kgkm^2 P. These loading rates are towards the low end of what is described in the results reported herein. They found significant changes in the microbial, infaunal, and benthic microalgal community, as well as altered food web interactions. The loading rates Deegan et al. used were within the range they anticipated to result from the "addition of 1,000 houses in the upland watershed of each of the nutrient-enriched creeks". One thousand houses will be there for decades, of course, thus the effects of nutrient loading is appropriately examined using the cumulative long-term loading rate, as well as the annual rate.

The seemingly mild or nonexistent differences in soil strength in the bottom layer measured in the +N or +P treatment plots, or in combination (Fig. 2), are an interesting observation that deserves further scrutiny. It may be that metabolic activity at 60 to 100 cm is so intensely constrained by either N or P that a synergistic response is not inevitable. White and Reddy (2000) described some of the effects of increased P loading on N mineralization in the Everglades, but I am unaware of relevant studies of the effect of N loading on P cycling or how either affects soil strength.

One strong implication of these observations is that the widespread increases in nutrient loadings to coastal waters (NRC 2000) may alter marsh ecosystem functions and perhaps compromise the long-term stability of coastal marshes by increasing belowground organic matter decomposition rates and by reducing root production resulting in a net decline in soil strength. In addition, the increased

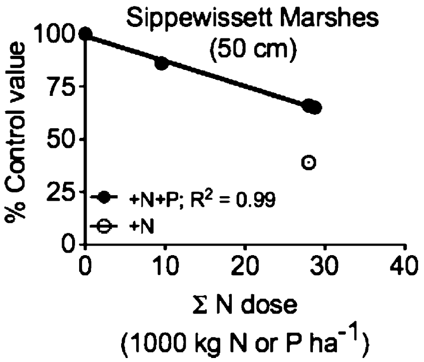

Fig. 7 Changes in soil strength as a percent of that in the control marsh at the long-term sewage-amended salt marsh at Sippewissett, MA. The data are described in Turner et al. (2009), but not presented graphically. $p<0.01$

biomass aboveground will also result in more “tug” on the belowground biomass during storms or from strong tidal currents. As a result of these changes, the chronic degradation of marsh organic matter may not be visually identified or expressed until the aboveground portion of the plant is ripped out of its substrate in a quick transition from marsh to mud flat or open water.

The long-term effects of increased nutrient loading on coastal soils are also of concern because the anticipated climate changes will result in an acceleration of sea level rise. Coastal marshes maintain their vertical position within the upper portion of the local tidal range through the accumulation of both inorganic and organic materials. The accumulation of organic matter at the present rate of sea level rise controls the vertical accretion rate in organic-rich salt marshes (Turner et al. 2000). Soil elevation may be lowered and to the detriment of the plant’s existence if either the root production is decreased or the decomposition of accumulated organic matter is accelerated enough so that the net organic accumulation is less than the relative sea level rise or, in particular, with the predicted acceleration in sea level rise. The anticipated additional rise in atmospheric CO_2 may, however, increase belowground biomass (Langley et al. 2009), whereas accelerated sea level rise will be an additional stressor. Teasing out the interrelationships is an appropriate challenge to address with field studies.

A decline in soil strength with nutrient enrichment suggests, but does not prove, that hurricanes will more easily tear the marsh apart at its weakest area, which is about 40 to 70 cm deep in the plots studied here. This may be the cause of the loss of 12% of the wetlands (106 km^2; Turner 2010) in the flow path of the Caernarvon river diversion, located downstream from New Orleans, and on the western side of where Hurricanes Katrina and Rita passed by this city in 2005. The nutrients in the diverted river water provide 60% to 71% of the total nitrogen load to all of Breton Sound (Hyfield et al. 2008) and bring the cumulative N load for the entire area to within range of the 10% soil strength loss shown in Fig. 3. There would be regional variations in loading that would be many times higher closer to the diversion structure. The Caernarvon diversion was constructed to strengthen the marsh and prevent further losses, but the reference area to the east, separated from the diversion water flow path by LA Highway 300, and also in the path of Hurricane Katrina, had much less marsh loss from these two hurricanes (Turner 2010). On the west bank of the Mississippi River, wetland loss from Hurricane Katrina was higher near the Naomi and W. Pt. la Hache Mississippi River diversions and absent a few kilometers away from them (Turner 2010). These observations suggest that this part of Louisiana’s wetland restoration program may have had an unexpected negative outcome and support the suggestion of Swarzenski et al. (2008) that the widespread decline of freshwater marshes in the Lake Penchant area is related to the inflow of nutrient-rich Atchafalaya riverwater during periods of high discharge.

The societal costs of building and operating these diversions are considerable. A diversion costs several hundred million dollars (U.S. Government Accountability Office 2007), the cost of restoring or creating wetlands is at least \$179,000 ha^{-1} (Turner 2010), and one estimate of the storm protection values specific to Louisiana is \$1,479 $ha^{-1}year^{-1}$ (Costanza et al. 2008). An estimate of the economic value of these wetland losses from Hurricane Katrina and Rita over 20 years, therefore, is in the range of several hundred million, to several billion dollars, and so I agree with Swarzenski et al. (2008) that a quantitative evaluation of the relations between river diversions and wetland soil ecosystems is desirable. Indeed, the consequences of implementing a proposed wetland restoration action that is based on unsupportable assumptions may be to lose wetlands, not sustain them.

The modern day water quality in the Mississippi River may, therefore, compromise attempts to use diversions of the Mississippi River to restore coastal wetlands where the receiving area has organic soils, which is most of Louisiana’s coastal wetlands (Chabreck 1972). Whatever benefit results from increases in the inorganic sediment supply should be weighed against the impact of also adding nutrients that weaken the soils. With that in mind, it is worth pointing out that reducing the size of the largest low oxygen zone in the western Atlantic (the “Dead Zone”) is also dependent on reducing nitrogen loading by the Mississippi River to the offshore zone (Rabalais et al. 2002, 2007). The primary determinant of the rise in nitrate concentration in the Mississippi River is land use changes in the watershed since the 1850s, and particularly since the 1950s (Turner and Rabalais 2003; Broussard and Turner 2010). Restoration of Louisiana’s coastal wetlands, shrinking the Dead Zone, and sustaining New Orleans are, therefore, intimately linked through improvements in the Mississippi River water quality—something that must take place inland, far from the coast.

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