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Nutrient and Sediment Effects on Oligohaline Marsh Plant Community in Mississippi River Delta

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NUTRIENT AND SEDIMENT EFFECTS ON OLIGOHALINE MARSH PLANT
COMMUNITY IN MISSISSIPPI RIVER DELTA

A Thesis

Submitted to the Graduate Faculty of the
Louisiana State University and
Agriculture and Mechanical College
in partial fulfillment of the
requirements for the degree of
Master of Science

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by

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ABSTRACT

The Mississippi River Delta has experienced more than 80% of wetland loss in the conterminous United States due partially to flood control levees that have hydrologically isolated the lower River from adjacent wetlands. Reconnecting the River to the wetlands through diversions is used to reduce wetland loss rates. However, diversion structures were originally designed to deliver freshwater rather than sediment. Furthermore, nutrient concentrations have increased in the upper watershed due to agricultural intensification. The goals of this study were to evaluate the effects of nutrients on oligohaline wetland plant community structure and function, to identify the major nutrient responsible for significant impacts, and to determine if the current diversion-sediment loading rates effectively reduce any impacts. Sixty intact marsh sods from a *Sagittaria lancifolia* dominated wetland near Madisonville, Louisiana were transported to a greenhouse at Louisiana State University. The sods were assigned one of six nutrient additions (NO_3 , NH_4 , PO_4 , SO_4 , $\text{NO}_3+\text{NH}_4+\text{PO}_4+\text{SO}_4$ [combo], and control) with or without sediment in a completely randomized design. After 25-months, the NH_4 , NO_3 , and combo treatment-levels increased aboveground biomass while SO_4 decreased it; PO_4 had no significant effect. Although belowground standing crop was not affected by nutrients, belowground biomass accumulation was significantly lower with SO_4 ; the other nutrients had no significant effects. Sediment addition significantly ($p=0.06$) increased aboveground biomass by the end of the second year, but had no effect in year 3. Sediment had no significant effect on belowground biomass or biomass accumulation nor was there a significant interaction with nutrients, even though sediment increased soil bulk density. NH_4 , SO_4 and Combo decreased decomposition rate, and soil shear strength increased with NH_4 . None of the treatments had significant effects on the number of new species or total species richness. The negative effects of SO_4 suggest that it is potentially a problematic ion, although the concurrent application of nitrogen and phosphorus

ameliorated these negative impacts. Also, sediment can be beneficial to coastal wetlands experiencing high subsidence rates by increasing soil bulk density. Thus, the optimization of diversions to carry sediments, along with continued surveillance of nutrient effects, would be wise management decisions.

INTRODUCTION

Wetlands globally are undergoing natural and human-induced disturbances resulting in their degradation and loss through conversions to open water or other landforms (Dahl 1990, Silliman et al. 2009). About half of the estimated 5.3-12.8 million km² of global wetland area was degraded during the first half of the twentieth century; the remaining wetlands now cover less than 9% of the earth's land surface (Zedler and Kercher 2005). Similar trends have occurred in the conterminous United States, where wetland area decreased by 53% between the 1780s and the 1980s (Dahl 1990). Consequently, many important functions and services provided by wetlands, including water quality improvement, flood control, biodiversity maintenance, wildlife conservation, and carbon sequestration, have been jeopardized (Zedler and Kercher 2005).

Wetland loss is particularly apparent in the deltas of the world (Overeem and Brakenridge 2009). In the Mississippi River Delta Complex (MRDC) (Louisiana, USA), for example, conversion of coastal habitat to water accounted for approximately 90% (~ 4,877 km²) of the total wetland loss in the continental United States between 1932 and 2010 (Couvillion et al. 2011). Currently, wetland loss in the MRDC continues at a rate of approximately 65 km² yr⁻¹ (Costanza et al. 2013), driven by natural processes such as delta abandonment, subsidence, sea level rise, wave erosion, herbivory, hurricanes, geologic faults, as well as human activities such as leveeing and hydrologic isolation, land reclamation for agriculture and industry, canal dredging, boat-induced shoreline erosion, invasive species-induced herbivory, and pollution, among other factors (Boesch et al. 1994, Turner 1997, Day et al. 2000). One of Louisiana's most challenging environmental issues is wetland submergence and shoreline erosion, which render the coastal zone and its natural and build capital more vulnerable to hurricanes.

One suggested approach to maintain, restore, and create coastal wetlands within the MRDC is to reconnect the Mississippi River to its former floodplain through river diversions (CPRA 2012). The Caernarvon and Davis Pond freshwater diversions (currently the largest of these diversions) attempt to mimic historic spring floods, which are now prevented by a network of levees along the lower Mississippi River. In doing so, these diversions provide a controlled flow of fresh water, sediments, and nutrients from the Mississippi River to adjacent coastal marshes and bays to promote coastal wetland restoration by theoretically stimulating marsh accretion and enhancing plant productivity (Barras et al. 2003, DeLaune et al. 2003, Day et al. 2011). Based on model projections, the Davis Pond and Caernarvon Freshwater Diversions are predicted to reduce land loss by 50% in their respective watersheds (Barras et al. 2003). However, Landsat imagery of wetlands adjacent to three existing diversions showed no significant increases in vegetation or total marsh area (Kearney et al. 2011). This outcome is not especially surprising given that these freshwater diversions were engineered to regulate salinity within the coastal bays rather than introduce large volumes of sediment into coastal marshes (Wheelock 2003, Snedden et al. 2007). Thus, the sediment requirements necessary to promote new plant establishment and sustain vigorous growth in a subsiding deltaic environment are likely not being met using the current “freshwater” diversion design (Kesel et al. 1992, Slocum et al. 2005, Stagg and Mendelssohn 2010, Day et al. 2013).

In addition to limited sediments in Mississippi River freshwater diversions, the water quality of the Mississippi River has changed dramatically over recent decades, thereby bringing into question the use of river diversions as a tool for wetland restoration (Kearney et al. 2011, Turner 2011). In particular, the Mississippi River nitrate load increased 2.5 fold between 1960 and 1998 (McIsaac et al. 2001, Turner and Rabalais 2003), while total N (i.e., nitrate + ammonium +

organic N) doubled during a similar timeframe (Goolsby and Battaglin 2001). Furthermore, total phosphorus (P) (inorganic and organic) concentration was negligible in the Lower Mississippi River prior to 1972 (i.e., < 2µg/L total P), but reached higher and variable concentrations (3-6 µg/L) between 1975- 1990, although no specific temporal trend was apparent (Turner and Rabalais 1991). Meanwhile, the long-term trend for SO₄ concentrations (20-50mg/L) increased between the early 1900s and 1990 (Hem 1993).

Although nutrient enrichment of wetlands is often considered beneficial in increasing plant function (Morris et al. 2013), many negative impacts have been documented. For example, common responses to nutrient enrichment are alterations in plant species composition and diversity (Aerts and Berendse 1988, Belanger et al. 1989, Doren et al. 1997, Bedford et al. 1999), as well as increased herbivore activity (Bertness et al. 2008). Although nutrient enrichment generally stimulates aboveground biomass and productivity (Buresh et al. 1980, Broome et al. 1983, Slocum and Mendelssohn 2008, Graham and Mendelssohn 2010), reduced belowground biomass (Valiela et al. 1976, Hines et al. 2006, Darby and Turner 2008, Davey et al. 2011, Ket et al. 2011, Deegan et al. 2012), accelerated soil organic matter decomposition via enhanced soil respiration (Morris and Bradley 1999, Wigand et al. 2009, Anisfeld and Hill 2012), and loss of soil strength (Turner et al. 2009, Turner 2011) can simultaneously occur. For instance, nine years of experimental nutrient enrichment of a New England salt marsh, although increasing aboveground biomass, decreased the belowground accumulation of roots and increased organic matter decomposition, which apparently contributed to creek bank collapse and conversion to mud flats (Deegan et al. 2012). However, belowground responses to nutrient enrichment are equivocal, as others have observed no net negative effect on ecosystem function (Langley et al.

2009, Anisfeld and Hill 2012, Day et al. 2013), and in a few cases, have even shown that nutrient enrichment can stimulate soil accretion (Morris et al. 2002, Fox et al. 2012).

The applicability of these fertilization-experimental results to diversions and their universality has been hotly debated, with many questions remaining unanswered. For example, assuming that elevated nutrients in Mississippi River diversion water are detrimental to wetlands, the primary culprit is presently unknown. Also, existing diversions were initially designed to deliver freshwater not sediment, and therefore may not be an appropriate analog for future diversions, which will be engineered to maximize sediment input, and promote wetland stability within the MRDC.

As part of an initial attempt to address these information gaps, a greenhouse mesocosm experiment was designed to answer three primary questions: 1) do nutrient enrichment and sediment addition, similar to that presently occurring at the outfall of the Davis Pond Diversion of Mississippi River, have the potential to Alter ecological functioning in adjacent wetlands? 2) Which of the major nutrients are responsible for any observed responses? And 3) does the presence of sediments in diversion waters modify nutrient effects? I addressed these questions in a 2.5-year greenhouse study in which I exposed natural oligohaline marsh sods to two treatments: 1) nutrients (NO_3 , NH_4 , PO_4 , SO_4 , $\text{NO}_3+\text{NH}_4+\text{PO}_4+\text{SO}_4$, and control) and 2) sediment (with and without). I measured ecological responses including above- and belowground biomass and production, organic matter decomposition, soil shear strength, soil physicochemical variables and how plant species richness and species composition are affected by nutrients and sediments.

METHODOLOGY

Experimental Design

To investigate nutrient and sediment effects on the wetland plant community structure and function, 60 marsh sods, each about 30 cm in diameter and 25 cm height, were collected in December 2010 from an oligohaline wetland with a Histosol soil and dominated by *Sagittaria lancifolia* (Alistmataceae) (Slocum and Mendelssohn 2008, Graham and Mendelssohn 2010). The donor wetland is located on the north shore of Lake Pontchartrain on the west bank of the Tchefuncte River (30° 23' 12.87" N, 90° 09' 34.38" W), approximately 62 km from New Orleans. The sods were transported to a greenhouse on the Louisiana State University campus and acclimated to greenhouse conditions for 6 months and then randomly assigned to two different treatments: Nutrient and Sediment, in a full factorial arrangement. The nutrient treatment consisted of 6 levels: ammonium in form of NH_4Cl ($100 \text{ g N m}^{-2} \text{ yr}^{-1}$), nitrate in form of $\text{Ca}(\text{NO}_3)_2 \cdot 4 \text{ H}_2\text{O}$ ($100 \text{ g N m}^{-2} \text{ yr}^{-1}$), phosphate in form of KH_2PO_4 ($5 \text{ g P m}^{-2} \text{ yr}^{-1}$), sulfate in form of $\text{MgSO}_4 \cdot 7 \text{ H}_2\text{O}$ ($900 \text{ g S m}^{-2} \text{ yr}^{-1}$), a combination of all four nutrients (combo), and a no nutrient addition (control). The sediment treatment consisted of 2 levels: with sediment ($10 \text{ kg m}^{-2} \text{ yr}^{-1}$) and without. Loading rates were based on those from the Davis Pond Diversion (average nutrient concentrations in the Mississippi River at St. Francisville, LA between 2006 and 2011 [except NH_4 loading was made equivalent to NO_3 loading], a discharge rate of $75 \text{ m}^3 \text{ s}^{-1}$ (2006-2009), and project area of 3,700 ha). Sediment and nutrients were added in slurry-form to the surface of the sods. The sediment came from the Mississippi River, and was provided by Richfield Riversilt, Baton Rouge. Sediment grain size percentages were as follows; 49.8% of silt, 17.2 % of clay, and 33% of sand. The sods were irrigated with tap water every other day to

create alternating flooded and drained soil-conditions. Flooding depth was 1 cm above the sod surface, and soil water drainage never exceeded a sod-depth of 15 cm.

Each treatment was applied monthly at 1/12 the yearly loading rate to each of five replicate sods for 25 months. During the experiment, a number of biotic and abiotic variables were measured, such as above- and belowground biomass, belowground biomass accumulation, total primary productivity, organic matter decomposition, species composition and dominance, as well as soil shear strength, soil bulk density, pore water nutrients, pH, Eh and conductivity . To avoid edge effects, all the core samples for analysis were taken from the middle of the sods (see details in Analytical methods).

Analytical methods

Aboveground biomass

At the end of the 2011 and 2012 growing seasons and at the termination of the experiment in July 2013, all vegetation was clipped at ground level, sorted by species and into live and dead components to attain plant aboveground biomass (AG) (g sod^{-1}). Clipped plants were weighed after being dried at 60° C in a forced air oven. Aboveground biomass for each sod was calculated by summing the weights of all species, both dead and live, for each growing season. Aboveground plant material that died during each year was periodically collected and weighed, and added to the dead biomass collected at the end of each growing season.

Belowground plant accumulation

Belowground accumulation was measured by in-growth bags (6 cm in diameter x 25 cm height), which were filled with root/rhizome-free organic matter (finely ground peat). During April 2012, ingrowth bags were inserted into each sod and were harvested after an 18-month

period. During this time, plant roots and rhizomes had an opportunity to grow into the bags. At harvest, the ingrowth bags were removed from the soil by cutting the soil around each in-growth bag and removing any soil and roots on the outside of the bags. In the lab, the ingrowth bags were cut length-wise to facilitate peat removal, and roots and rhizomes were collected after washing the peat material over a 0.2 mm sieve. Roots and rhizomes were separated according to whether they were alive or dead, and thereafter dried at 60°C in an oven before weighing (McKee and Seneca 1982). Live roots were generally lighter in color, turgid and not friable (Robertson et al. 1999).

Belowground biomass

In addition to belowground accumulation, we also measured belowground biomass in April 2012 and July 2013 with a 5 cm diameter by 20 cm long core. The core was sectioned into two equal depth segments: 0-10 and 10-20 cm. The cores were washed as for the ingrowth bags, and root and rhizome material sorted into live and dead components. Belowground material was dried at 60°C in an oven and weighted. The remaining holes were filled with ingrowth bags.

Organic matter decomposition

Plant litter decomposition was assessed with the litter bag technique. Nylon litter bags with a 0.4 mm mesh were filled with oven-dried (60°C) aboveground or belowground *Sagittaria lancifolia* plant material and were placed separately in the soil to a depth of 5 cm from end of July 2012 till May 2013 (257days). Litter decomposition was calculated as percentage of plant material mass lost in the litter bags during this time.

Furthermore, cellulose degradation was measured with the cotton strip technique, providing an index of cellulolytic fungal and microbial activity (Maltby 1987). Cotton strips, 10 cm x 30

cm, were inserted vertically into the soil and allowed to degrade over a 10-day period. Measurements were made twice during the study in November 2012 and June 2013. After retrieval, the cotton strips were washed of soil, air dried, cut horizontally in 2-cm segments, and the tensile strength of each segment measured as described by Slocum et al. (2009). Cellulose degradation was expressed as percent loss of tensile strength per day (Maltby 1987, Slocum et al. 2009).

Soil shear strength

Because nutrients and sediments affect belowground biomass and decomposition, the strength of the soil may have also been affected by the treatments. Thus, soil shear strength was measured in all sods in July 2013 at 5, 10, and 15 cm-depths with the Geotechnics Geovane (Model H-4221, Geotechnics, Auckland, New Zealand). Measurements began at 5 cm depth and continued by pushing the blade of the Geovane to the 10 cm and 15 cm depths. Two blade sizes were used in the measurements. Where root and rhizome biomass was high, a 19 mm blade was used, while a 33 mm blade was used where belowground plant biomass was less and greater sensitivity needed. The torque gauge values were converted to shear strength (KPa) by multiplying by 0.2857 (large blade) or 1.48 (small blade).

Species richness

Species richness was calculated as the total number of species in each sod, recorded at the end of a growing season, namely, at the end of December 2011, 2012 and at the end of experiment in late June 2013. Moreover, the number of new species in years 2 and 3 were calculated.

Species dominance

Relative species dominance was determined as the percent of species-specific biomass per sod. Due to presence of many species at low frequency in each sod, only the relative dominances of the four most dominant species (*Alternanthera philoxeroides*, *Polypogon monspeliensis*, *Sagittaria lancifolia*, and *Schoenoplectus robustus*) were statistically analyzed (Graham and Mendelssohn 2010).

Bulk density

Bulk density was determined in July 2013 for both 5-cm and 10-cm soil depths with two different coring devices. The top 5 cm of soil was collected with a thin-walled aluminum core tube, 2 cm in diameter by 5 cm long, while the top 20 cm of soil was collected with a 5-cm diameter by 20-cm long core tube and divided into equal 10-cm segments. All cores were sealed in watertight plastic bags, and kept at 4°C (Carter 1993). Each core sample was weighed for its wet bulk density and then dried at 60° C in a forced-air oven to determine dry bulk density (g cm^{-3}).

pH and Conductivity

Interstitial water pH and conductivity were measured in July 2012 and May 2013. A 20 ml pore water sample was collected from an integrated depth of 10 to 15 cm below the soil surface with plastic syringe sippers, kept refrigerated, and analyzed within 24 hours (Koch and Mendelssohn 1989, McKee and Mendelssohn 1989). A Cole-Parmer conductivity meter was used for measuring conductivity and an Orion pH meter and combination pH electrode was used to determine the pH.

Eh

Soil Eh (redox potential) was measured in July 2012 and June 2013. In July 2012, 2 bright platinum-tip electrodes were inserted to a depth of 15 cm below the soil surface at least 24 hours before the reading was taken. Redox potential was measured with these 2 electrodes and a calomel reference electrode, which was attached to a Fisher Scientific Accumet portable mV meter. In June 2013, 3 bright platinum-tip electrodes were inserted into soil to a depth of 2 cm and mV measurements taken after 24 h. The electrodes were then pushed to a soil depth of 15 cm, and mV measurements were again taken after 24 hours (Gambrell and Patrick 1978, Faulkner et al. 1989). All electrode measurements were converted to Eh by adding +245, the potential of the calomel half-cell.

Pore water nutrient samples

In July 2012 and May 2013, interstitial water nutrient concentrations ($\text{NH}_4^+\text{-N}$, $\text{NO}_x^-\text{-N}$, and $\text{PO}_4^{3-}\text{-P}$) were measured by collecting 20 ml pore water samples at an integrated soil depth of 10-15 cm in each pot with plastic syringe sippers (Koch and Mendelssohn 1989, McKee and Mendelssohn 1989). Porewater samples were then filtered with 0.45 μm nylon syringe filters, and kept frozen before analysis. Additionally, to measure sulfide concentration a 3 ml pore water sample was mixed with a sulfide antioxidant buffer. Samples were kept in an ice chest during sampling and analyzed with an Orion sulfide electrode within 24 hours in the laboratory. Furthermore, a 20 ml filtered water sample for SO_4^{2-} determination was collected in May 2013, kept at 4°C and analyzed within a month with a Westco Smartchem 200 discrete analyzer.

Data Analysis

Data were analyzed with univariate mixed-model ANOVAs (PROC Mixed procedure) with repeated measurements, and the Tukey-Kramer multiple comparison adjustment in SAS (SAS

version 9.3, SAS Institute). Data were used to determine the overall and interactive effects of nutrients and sediments as independent variables to investigate the specific reaction of dependent variables such as biomass production) to the overall effect (Graham and Mendelssohn, 2010). The results were compared at a $p < 0.05$ significant level, unless otherwise stated. Shapiro-Wilk tests were used to test normality of the residuals while homogeneity of variance was evaluated with box plots. Because residuals were neither normally distributed nor homogenous, all data except for the number of new species in years 2 and 3 required logarithmic or square root transformations prior to analysis (Slocum and Mendelssohn 2008).

RESULTS

Aboveground biomass

After two years of sediment and nutrient additions to oligohaline marsh sods, both nutrients (Figure 1) and sediment (Figure 2) significantly affected aboveground biomass ($p < 0.001$ and $p = 0.06$, respectively). NH_4 , NO_3 and Combo significantly increased aboveground biomass compared to the control, while SO_4 decreased it and PO_4 had no significant effect. However, the stimulatory effect of NH_4 , NO_3 and Combo on aboveground biomass tended to increase in year 2 while that for SO_4 and PO_4 tended to decrease (significant nutrient x year interaction, $p = 0.001$, Figure 1 insert). The effect of sediment on aboveground biomass did not significantly vary by nutrient or year ($p > 0.05$).

In year 3, which consisted of 7 months of growth before harvest, the effects of nutrient addition on aboveground biomass were again significant ($p < 0.0001$, Figure 3). Nutrient trends were similar to those in years 1 and 2, except that the negative effect of SO_4 was more pronounced in year 3 (Figures 1 and 3). Sediment, however, did not affect aboveground biomass in year 3, nor was there a significant interaction with nutrients (data not shown).

Belowground biomass accumulation

Belowground biomass accumulation (live + dead), determined by the ingrowth method over an 18-month period, was significantly lower ($p = 0.0006$), compared to the control, with sulfate addition (Figure 4). Although belowground biomass was greater with sediment than without, 9.55 ± 1.8 g/sod (mean \pm standard error [SE]) and 6.2 ± 1 g/sod, respectively, this effect was not statistically significant ($p = 0.12$). Also, there was no significant interaction ($p = 0.2$) between the nutrient and sediment treatments.

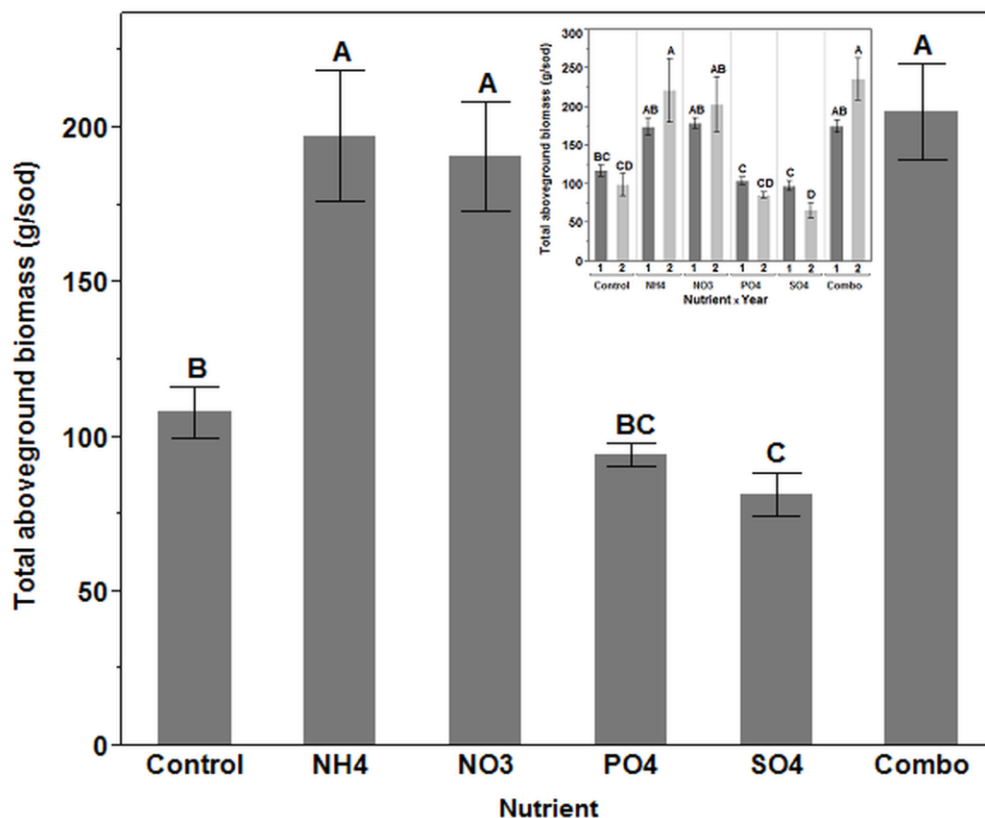


Figure 1. Effect of nutrients on total aboveground biomass in years 1 and 2 (main Graph). Interaction of nutrient and year (inset graph). Different letters indicate significant differences between different treatment-level (mean \pm SE) ($p < 0.05$).

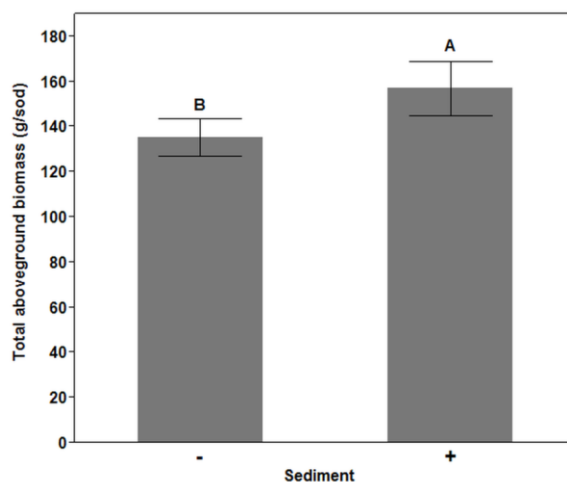


Figure 2. Sediment effect on the total (live + dead) aboveground biomass for years 1 and 2 combined (no significant year \times sediment effect). Different letters indicate significant differences between different treatment (mean \pm SE) ($p = 0.06$).

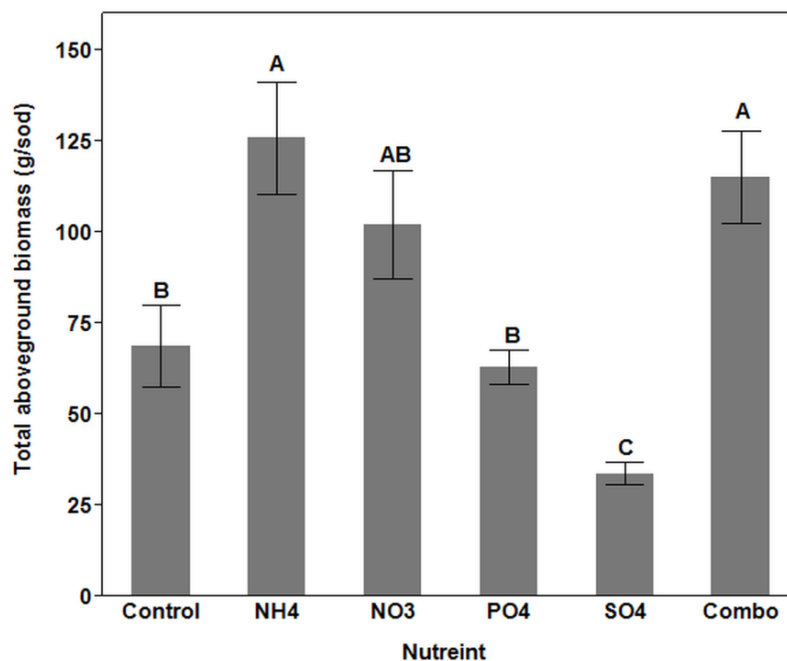


Figure 3. Effect of nutrients on the aboveground biomass in year 3 (July 2013) after 25-months of nutrient addition. Different letters indicate significant differences between different treatment-level (mean \pm SE) ($p < 0.05$).

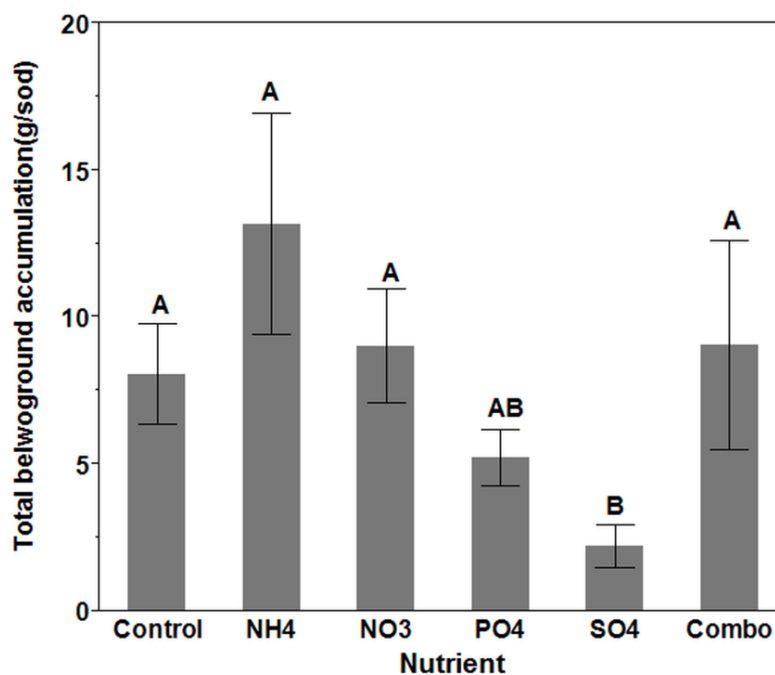


Figure 4. Effect of nutrients on belowground biomass accumulation. Different letters indicate significant differences between different treatment (mean \pm SE) ($p < 0.05$).

Belowground Standing Crop

In addition to belowground accumulation, belowground standing crop (live, dead and total) was determined. However, neither nutrient ($p = 0.13$), sediment application ($p = 0.41$), nor their interaction ($p = 0.74$) had a significant effect on any of the belowground standing crop components. Means and standard errors for the main effect of nutrients on total belowground standing crop are as follows: (control: 71.7 ± 5.9 g/sod; NH_4 : 94.78 ± 9.9 g/sod; NO_3 : 76.8 ± 5.3 g/sod; PO_4 : 70.5 ± 5.9 g/sod; SO_4 : 79.5 ± 7.1 g/sod; Combo: 88 ± 6.9 g/sod).

Total Production

Total biomass production was calculated for year 2 by adding total aboveground production in year 2 to belowground accumulation during the same period (year 2 was the only year that total production could be determined). The nutrient treatment had a significant effect on total production ($p < 0.0001$). Relative to control, NH_4 , NO_3 and Combo increased total production while PO_4 and SO_4 had no significant effect, although sulfate generated the lowest total production of any of the nutrient treatment-levels (Figure 5). The sediment treatment tended ($p = 0.1$) to increase total production (with sediment: 176.5 ± 25.6 g/sod; without sediment: 141.3 ± 15.5 g/sod); however its interaction with the nutrient treatment was not significant ($p = 0.7$).

Litter Decomposition

Because nutrients and sediments can affect organic matter decomposition and because it is such an important biogeochemical process, both aboveground and belowground plant litter decomposition were assessed with the litter bag technique. For aboveground tissue, SO_4 and Combo both significantly lowered decomposition rates compared to the control ($p < 0.0001$) while the other nutrients had no significant effect (Figure 6). For belowground tissue, the same effect was observed ($p < 0.0001$) with SO_4 and Combo reducing decomposition relative to the control

(Figure 7). Sediment only had a significant effect on belowground tissue decomposition in the presence of added PO_4 (significant nutrient x sediment interaction, $p=0.002$), decreasing decomposition by 32 % (PO_4 treatment-level: without sediment = $72.6 \pm 3.5\%$ and with sediment = $48.9 \pm 4.5\%$).

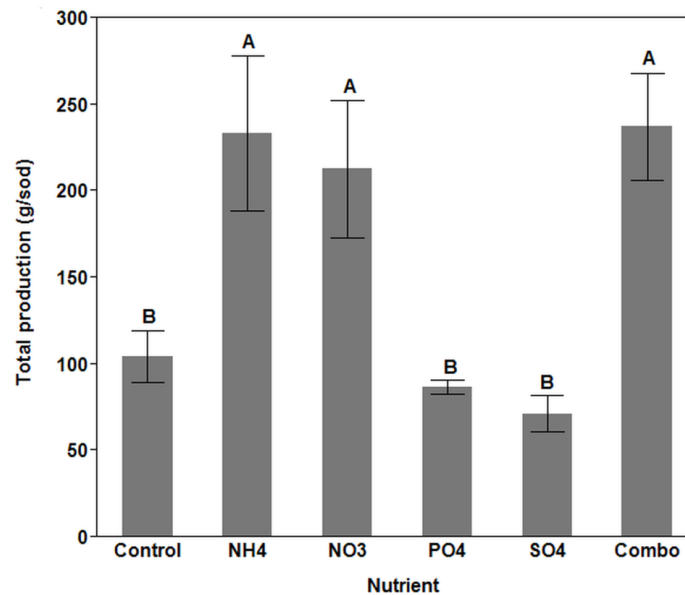


Figure 5. Effect of nutrients on total production (above and belowground) in year 2. Different letters indicate significant differences between different treatment (mean \pm SE) ($p < 0.05$).

Cellulolytic activity

As observed for plant litter decomposition, cellulolytic activity, determined with the cotton strip technique, was significantly lower in sods receiving SO_4 and SO_4 in combination with the other nutrients (Combo) compared to control ($p < 0.0001$) (Figure 8). However, cellulolytic activity also decreased with NH_4 addition (Figure 8), which was not observed for the decomposition of plant litter. Also, in contrast to the negative effect of these nutrients on cellulolytic degradation, sediment had a positive effect ($p=0.01$) (Figure 9). No significant interaction between nutrients and sediments on cellulose degradation was identified ($p > 0.12$).

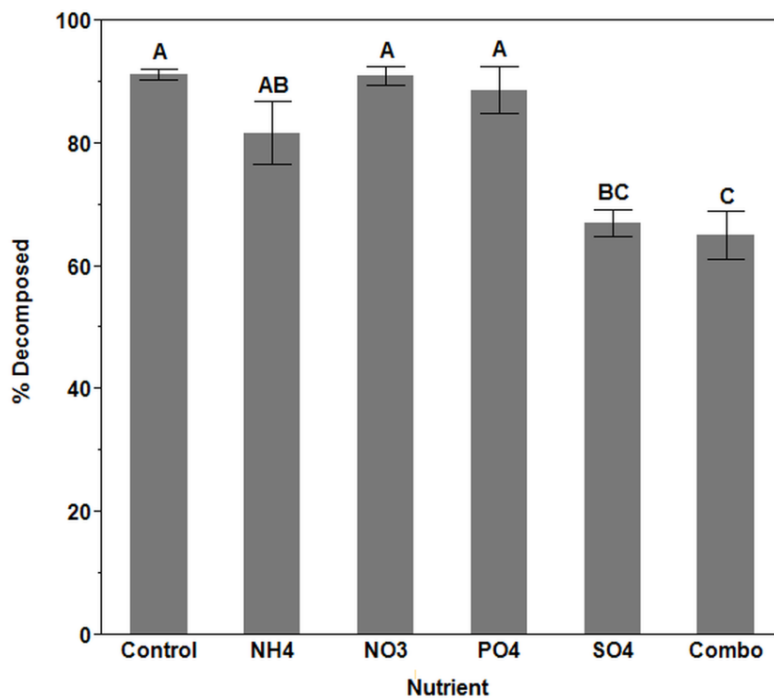


Figure 6. Effect of nutrients on aboveground tissue decomposition. Different letters indicate significant differences between different treatment (mean \pm SE) ($p < 0.05$).

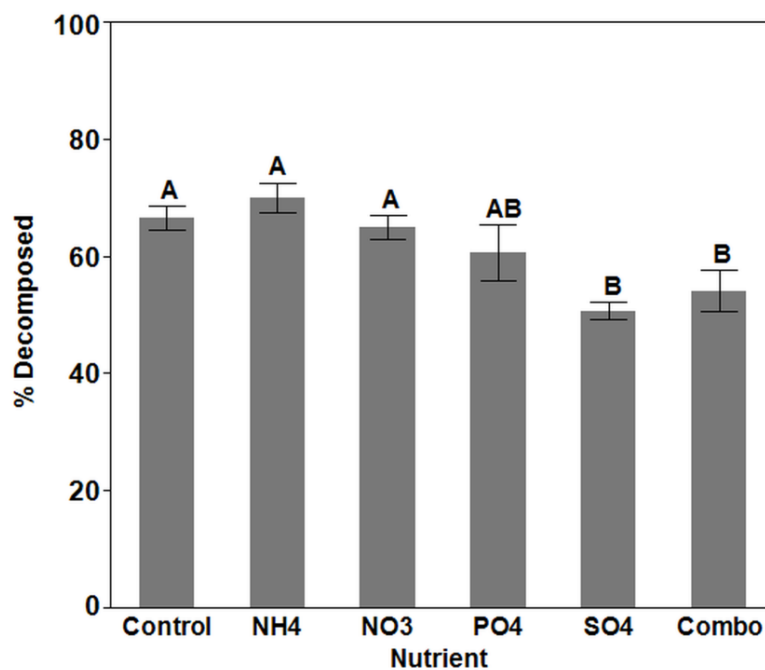


Figure 7. Effect of nutrients on belowground tissue decomposition. Different letters indicate significant differences between different treatment (mean \pm SE) ($p < 0.05$).

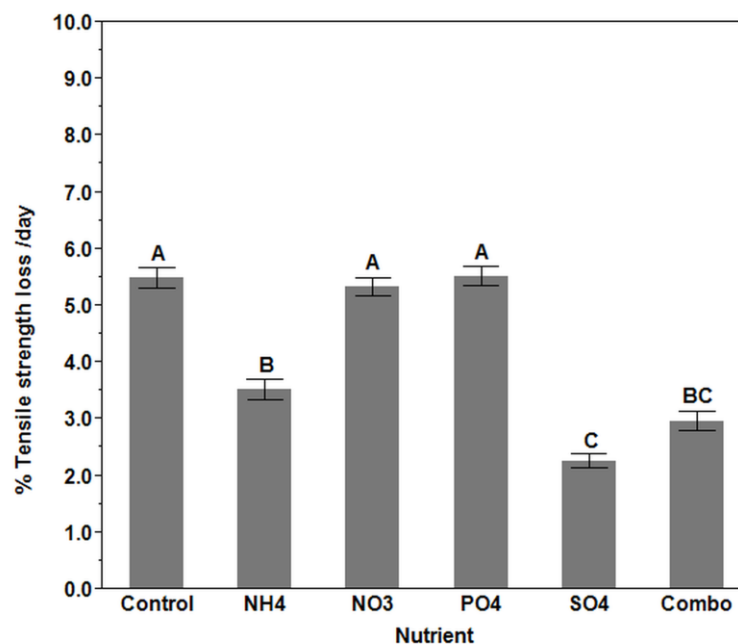


Figure 8. Effect of nutrients on cellulolytic activity in the soil determined with the cotton strip technique. Different letters indicate significant differences between different treatment (mean \pm SE) ($p < 0.05$).

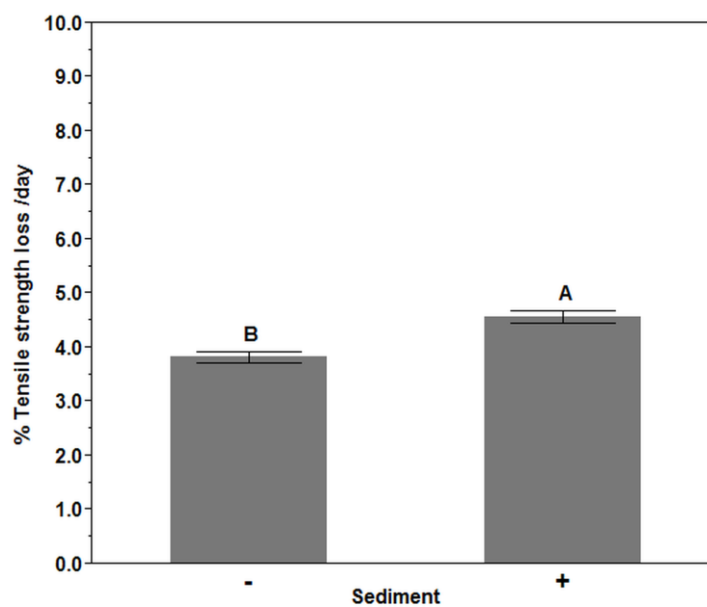


Figure 9. Effect of sediment on cellulolytic activity in the soil determined with the cotton strip technique. Different letters indicate significant differences between different treatment (mean \pm SE) ($p < 0.05$).

Shear strength

Soil shear strength at 5, 10, and 15 cm depths in the sods were measured. Relative to the control, only NH_4 affected soil shear strength ($p=0.01$), showing an increase when averaged over depth (Figure 10). None of the other nutrients statistically affected soil shear strength (Figure 10). Sediments also did not have a significant effect ($p=0.18$), and there was no interaction between sediments and nutrients ($p = 0.8$). Soil shear stress decreased with depth (highest at 5 cm: 34.71 ± 4.04 KPa, lowest at 15 cm: 13.98 ± 1.7 KPa, and generally intermediate at 10 cm: 19.91 ± 1.9 KPa; ($p<0.0001$).

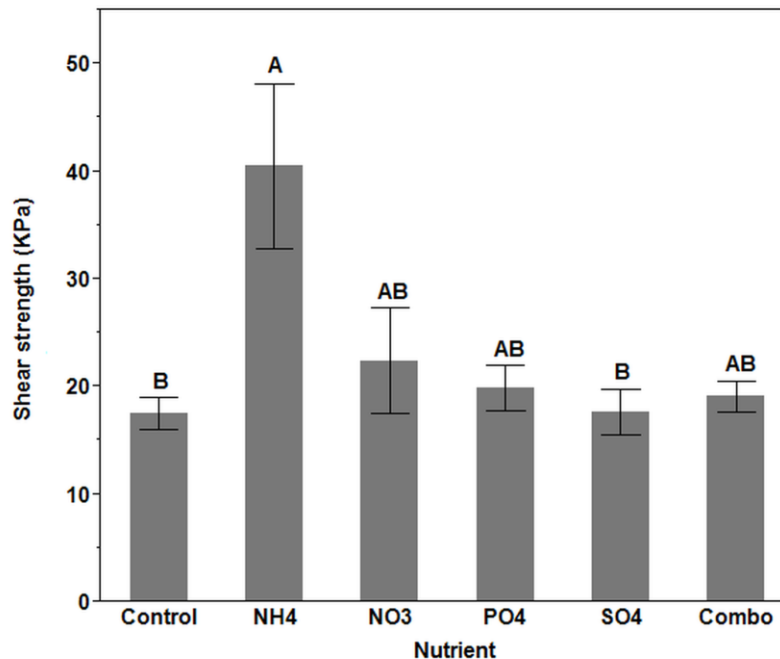


Figure 10. Effect of nutrients on soil shear strength measured with the Geovane shear strength vane. Different letters indicate significant differences between different treatment (mean \pm SE) ($p<0.05$).

Species richness

Twenty-two species from nine plant families were identified in the mesocosms during the experiment (Table 1). After two and a half years neither nutrient ($p=0.2$) nor sediment ($p=0.5$)

nor their interaction ($p=0.2$) had significant effects on species richness. In addition, the number of new species that were encountered in year 2 was not significantly affected by nutrient ($p=0.2$) or sediment ($p=0.6$) enrichment nor their interaction ($p=0.08$). Again, in year 3, neither nutrient ($p=0.6$) nor sediment ($p=0.1$) significantly increased the number of new species; however, their interaction was significant ($p=0.005$, Table 2). Different treatments were not statistically different from control. Sediment loading increased the number of new species for control, NO_3 and PO_4 treatment-levels, while decreasing it with NH_4 , SO_4 and combo additions, although treatment-means were not statistically different from controls (data not presented). Total species number in year 2 was not significantly affected by nutrients ($p=0.3$) or, sediment ($p=0.7$), but their interaction was significant ($p=0.05$), i.e., sediment increased species richness only for the NH_4 treatment-level (data not presented). By year 3, total species number again was significantly affected by nutrients ($p=0.03$), however the nutrient effect was dependent on sediment addition (significant sediment by nutrient interaction, $p=0.03$, Table 2) Sediment tended to increase species richness in the control and NH_4 treatment-levels, but tended to decrease species richness with the addition of PO_4 , NO_3 , SO_4 , and combo (data not presented). Sediment ($p=0.7$) by itself had no effect on total species number, while its interaction with nutrients was significant ($p=0.03$, Table 2); however the treatments were not statistically different from the control.

Table 1. Plant species collected in the sods during the experiment.

Species	Family	Species	Family
<i>Alternanthera philoxeroides</i>	Amaranthaceae	<i>Polygonum punctatum</i>	Polygonaceae
<i>Amaranthus australis</i>	Amaranthaceae	<i>Polypogon monspeliensis</i>	Poaceae
<i>Ammannia latifolia</i>	Lythraceae	<i>Ptilimnium capillaceum</i>	Apiaceae
<i>Baccharis halmifolia</i>	Asteraceae	<i>Rumex crispus</i>	Polygonaceae
<i>Cyperus odoratus</i>	Cyperaceae	<i>Schoenoplectus robustus</i>	Cyperaceae

Table 1 (continued)

Species	Family	Species	Family
<i>Echinochloa crus-gali</i>	Poaceae	<i>Schoenoplectus tabernaemontani</i>	Cyperaceae
<i>Eleocharis fallax</i>	Cyperaceae	<i>Sagittaria lancifolia</i>	Alismataceae
<i>Ipomoea sagittata</i>	Convolvulaceae	<i>Solidago sempervirens</i>	Asteraceae
<i>Iva frutescens</i>	Asteraceae	<i>Spartina patens</i>	Poaceae
<i>Lythrum lineare</i>	Lythraceae	<i>Symphotrichum subulatum</i>	Asteraceae
<i>Panicum repens</i>	Poaceae	<i>Polygonum punctatum</i>	Polygonaceae
<i>Paspalum vaginatum</i>	Poaceae		

Relative dominance

Relative dominance of *Alternanthera philoxeroides*, *Polypogon monspeliensis*, *Sagittaria lancifolia*, and *Schoenoplectus robustus*, the four most common species occurring in the experimental sods, was statistically analyzed (Graham and Mendelssohn 2010). *Sagittaria lancifolia* was the dominant species in the first year, while *S. robustus* became co-dominant with *S. lancifolia* in the second year (Figure 11). *Schoenoplectus robustus* became dominant in the third year and replaced *S. lancifolia* ($p < 0.0001$, Figure 11). By the end of the experiment, the nutrient treatment had no significant effect on species dominance ($p = 0.7$), sediment ($p = 0.6$) and its interaction with nutrients also did not change relative dominance ($p = 0.2$). However, the interaction of nutrients with species was significant ($p < 0.000$, Figure 12).

Table 2. Results of univariate Analysis of Variance for species richness. Significant p values are in bold ($p < 0.05$).

Effect	No. of species in years 1, 2, and 3	New species in year2	New species in year3	Total species in 2 years	Total species in 3 years
	P-value	P-value	P-value	P-value	P-value
Nutrient	0.1664	0.2023	0.5999	0.323	0.034

Table 2 (continued).

Effect	No. of species in years 1, 2, and 3	New species in year2	New species in year3	Total species in 2 years	Total species in 3 years
	P-value	P-value	P-value	P-value	P-value
Sediment	0.4669	0.6506	0.1421	0.7112	0.9471
Nutrient*Sediment	0.1727	0.0845	0.0051	0.0518	0.0362
Year	<0.0001	—	—	—	—
Nutrient*Year	0.0238	—	—	—	—
Sediment*Year	0.0738	—	—	—	—
Nutrient*Sediment*Year	0.0012	—	—	—	—

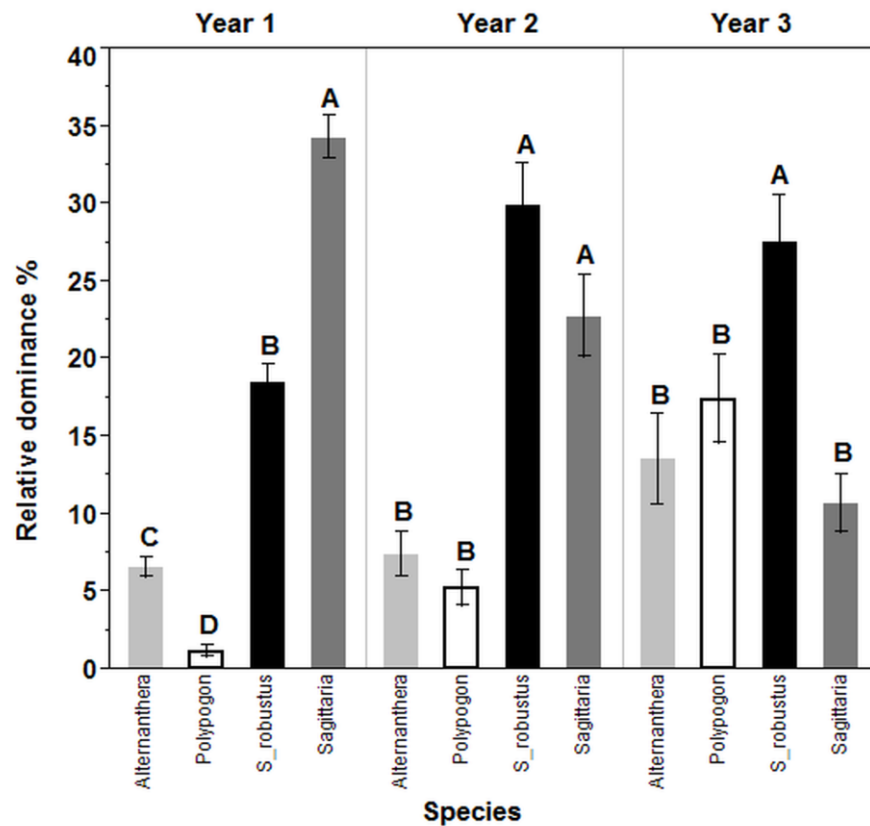


Figure 11. Relative dominance of four common species in each of the three study-years. Different letters indicate significant differences between different treatment (mean \pm SE) ($p < 0.001$).

Moreover, eutrophication caused a shift in species dominance. For example, the species dominance of *S. lancifolia* was affected by nutrients ($p<0.0001$), sediment ($p=0.01$) and their interaction ($p=0.001$). In sods, which received combo and SO_4 , *S. lancifolia* dominance decreased relative to the control and it was totally eliminated in sods with SO_4 treatment while *S. robustus* and *P. monspeliensis* were co-dominant in all sods after all treatment enrichment (Figure 12). Meanwhile *A. philoxeroides* dominance increased with NH_4 and combo treatment-levels relative to control (Figure 12). Sediment did not significantly affect *S. lancifolia* dominance. NH_4 in combination with sediment and PO_4 with and without sediment had positive effects on *S. lancifolia* dominance relative to the control. In contrast, combo and SO_4 regardless of the addition of sediment negatively affected *S. lancifolia* dominance relative to control (Figure 12). Other treatment enrichments were not significantly different from the control (Table 3).

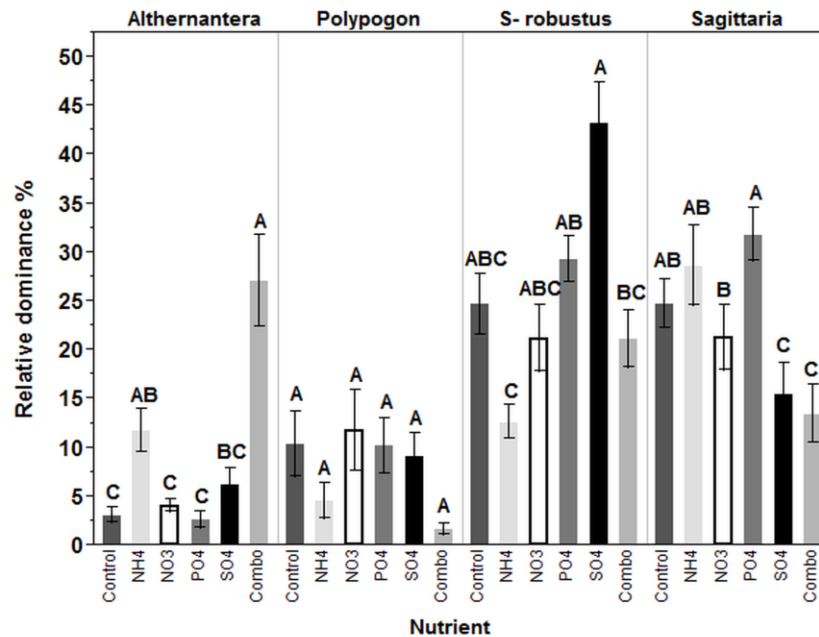


Figure 12. Nutrient effects on relative dominance of the four most common species over a three-year period. Different letters indicate significant differences between different treatment (mean \pm SE).

Table 3. Results of univariate Analysis of Variance for relative dominance of the four most dominant species. Significant p values are denoted in bold at ($p < 0.05$).

	<i>A. philoxeroides</i>	<i>P.monspelienis</i>	<i>S. robustus</i>	<i>S.lancifolia</i>
Effect	P- value	P- value	P- value	P- value
Nutrient	<0.0001	0.0611	0.0018	<0.0001
Sediment	0.1742	0.2533	0.7906	0.0184
Nutrient*Sediment	0.6462	0.5131	0.3966	0.0019
Year	0.0010	<0.0001	0.0448	<0.0001
Nutrient*Year	<0.0001	0.0002	<0.0001	<0.0001
Sediment*Year	0.1466	0.3274	0.3248	0.1565
Nutrient*Sediment*Year	0.5529	0.4014	0.3220	0.0276

Bulk density

Bulk density was determined for the 0-5 cm soil depth. Sediment addition significantly increased bulk density ($p < 0.0001$, Figure 13). Nutrients, however, had no significant effect on bulk density, nor was there a significant nutrient x sediment interaction ($p = 0.39$).

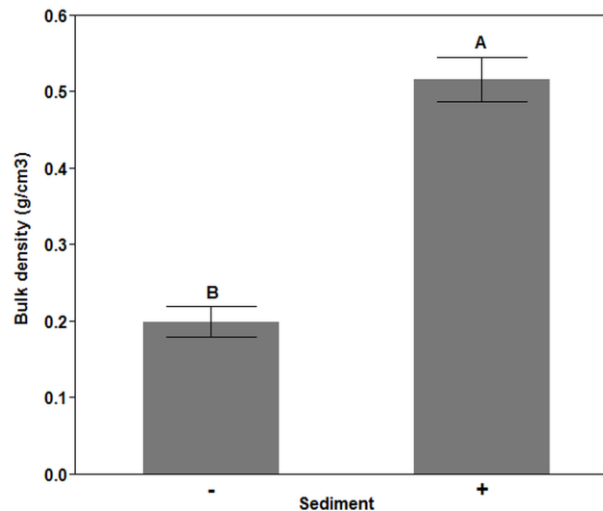


Figure 13. Effect of sediment on bulk density in the top 5 cm of the soil. Different letters indicate significant differences between different treatment (mean \pm SE) ($p < 0.05$).

pH and Conductivity

Although mean pH in the sod pore waters only varied from 6.2 to 7.3 pH units, significant differences occurred among the nutrients. Pore waters from sods that received NO_3 and PO_4 had mean pH's around 7, similar to the control, while NH_4 , SO_4 , and Combo generated more acidic pore water environments ($p < 0.0001$, Figure 14-A). Electrical conductivity followed a similar but inverse trend ($p < 0.0001$) with SO_4 and Combo approximately doubling the pore water conductivity from these nutrients (Figure 14-B). Although sediment ($p=0.91$) and its interaction with nutrients ($p=0.34$) did not significantly affect conductivity, sediment addition increased pH from 6.8 to 7, but only in the presence of added NH_4 (significant sediment x nutrient interaction, $p=0.02$).

Redox potential

Redox potential was significantly lower in sods receiving SO_4 and Combo and significantly higher in the NH_4 treatment ($p < 0.0001$), all relative to the control. Sediment alone ($p=0.6$) or in combination with nutrients did not affect soil Eh ($p=0.4$) (Figure 14-C).

Pore water nutrients

Pore water $\text{NH}_4^+\text{-N}$ concentration was higher in sods that received SO_4 and Combo relative to the control and lower in sods that received added NH_4 and NO_3 ($p < 0.0001$, Figure 14-D). Sod with added PO_4 also tended to have lower NH_4 than the control, but this difference was not significant at $p=0.05$. Sediment overall did not affect $\text{NH}_4^+\text{-N}$ concentration in the pore water ($p=0.9$), but sediment in combination with the SO_4 and Combo treatment-levels yielded higher $\text{NH}_4^+\text{-N}$ than without sediment (significant sediment x nutrient interaction; $p=0.01$). $\text{NO}_3^-\text{-N}$ concentration in pore water was similar to the control in the PO_4 and NO_3 treated sods, but lower with Combo, NH_4 , and SO_4 ($p < 0.0001$, Figure 14-E). Sediment ($p=0.5$) and its interaction with

nutrients did not affect NO_3 concentrations ($p=0.6$). , PO_4^{3-} -P concentrations in pore waters were similar to those in control sods with added NO_3 and combo, but increased in sods enriched with PO_4 and SO_4 ($p<.0001$, Figure 14-F). Sediment ($p=0.5$) and its interaction with nutrients had no significant effect ($p=0.6$). SO_4^{3-} concentrations in the pore water were very high in sods that received SO_4 and Combo, but not different from the control for the other nutrient amendments ($p<.0001$, Figure 14-G). SO_4^{3-} concentration was neither significantly affected by sediment ($p=0.4$) nor its interaction with nutrients ($p=0.2$). The nutrient treatment significantly affected pore water sulfide ($p<.0001$, Figure 14-H), the product of microbial sulfate reduction. Both SO_4 and Combo increased pore water sulfide compared to the control and other nutrient amendments (Figure 14-H). Sediment ($p=0.1$) and its interaction with nutrients ($p=0.4$) did not significantly affect sulfide concentrations.

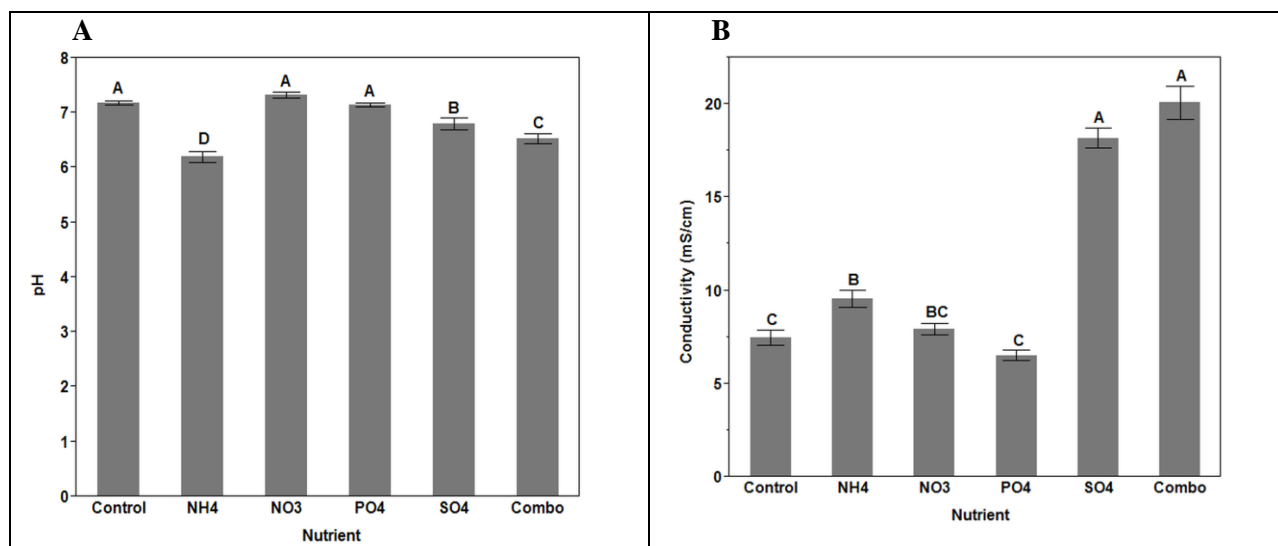


Figure 14. Effect of nutrients on pH (A), conductivity (B), soil redox potential (C), and NH_4 (D), NO_3 (E), PO_4 (F), SO_4 (G), and S-2 (H) concentrations. Different letters indicate significant differences between different treatment means ($p<0.05$).

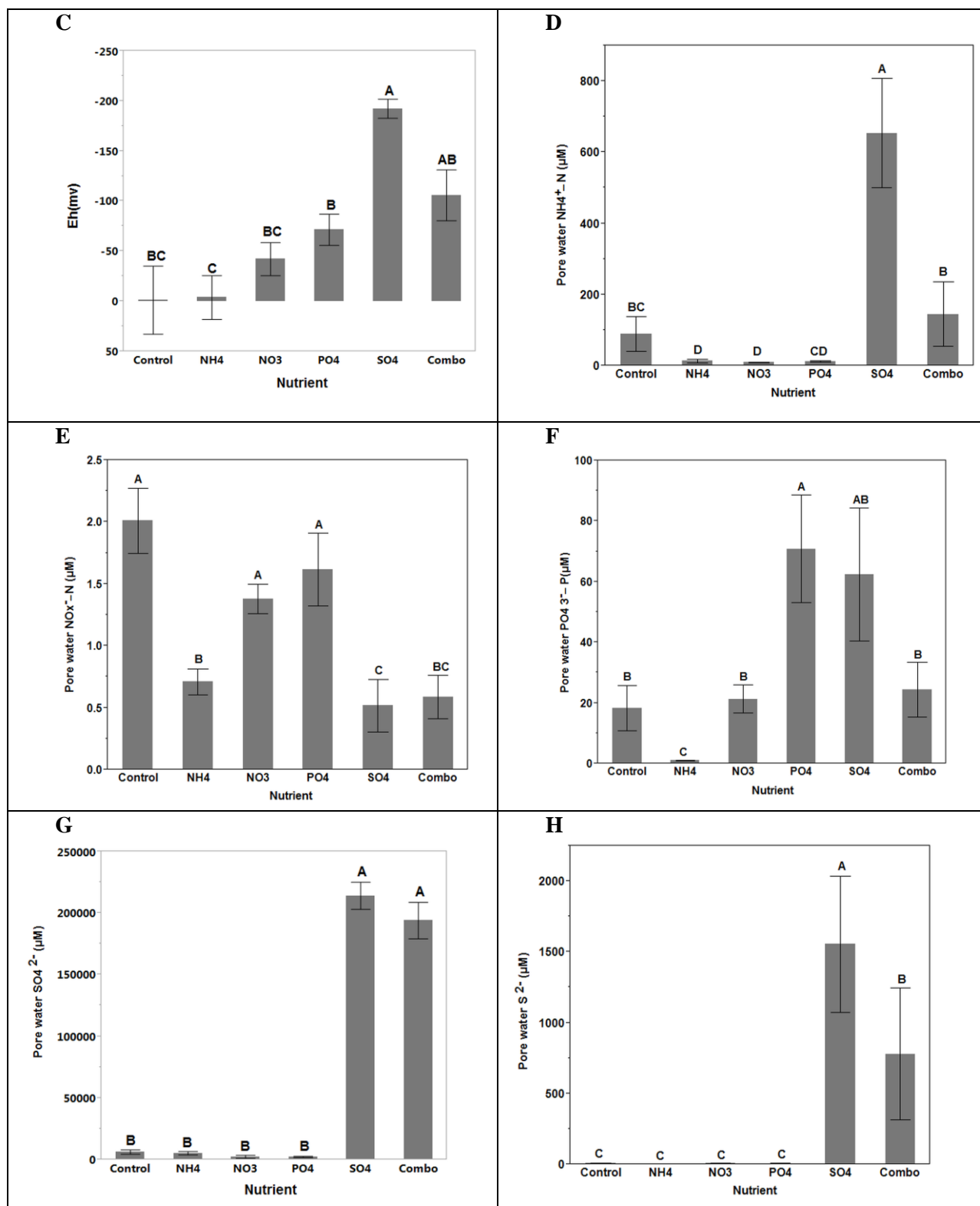


Figure 14 (continued).

DISCUSSION

Although a large body of literature exists that describes the responses of marsh vegetation and marsh processes to nutrient addition (Buresh et al. 1980, Broome et al. 1983, Chiang et al. 2000, DeLaune et al. 2005, Darby and Turner 2008, Graham and Mendelssohn 2010, Morris et al. 2013), to my knowledge this is the first investigation that simultaneously compares the effects of these nutrients in the same experiment. I identified significant nutrient effects on marsh species composition and several important marsh processes, whereas sediment had a lesser, albeit still important effect. Overall, plant production was limited by nitrogen availability and by sulfide-stress resulting from sulfate reduction. However, the concurrent application of nitrogen and phosphorus ameliorated the negative impact of sulfate addition on plant growth.

Nutrient-induced Effects

Nitrogen fertilization in the form of NH_4 or NO_3 increased aboveground standing crop significantly relative to controls. This research agrees with other studies identifying N as the primary nutrient limiting plant growth in coastal wetlands (Sullivan and Daiber 1974, Mendelssohn 1979, Morris 1982, Cargill and Jefferies 1984, Boyer et al. 2001, Wigand et al. 2004, DeLaune et al. 2005) and in an oligohaline marsh adjacent to the donor site used in my study (Graham and Mendelssohn 2010). However, the growth-stimulating effect of the two nitrogen forms did not significantly differ, suggesting that nitrate is also assimilated by marsh vegetation when it is available (Stewart et al. 1973, Mendelssohn 1979, Morris 1982). Neither NH_4 nor NO_3 had any negative effects on the processes evaluated in this research.

The lack of an aboveground productivity response to PO_4 is consistent with findings for other salt marshes in, for example, subarctic Canada (Cargill and Jefferies 1984); California (Boyer et al. 2001); Delaware (Sullivan and Daiber 1974), and an oligohaline marsh adjacent to the donor

marsh used in this research (Graham and Mendelssohn 2010) . In contrast, Baldwin (2013) reported N+P co-limitation in an oligohaline marsh in Maryland, supporting the findings of Crain (2007) in an oligohaline marsh in Maine. However, my results suggest that the oligohaline marsh plant community used in my study is N limited rather than N+P co-limited. Contradictory findings regarding nutrient limitation in different marshes can be attributed to tidal flushing, flooding depth and duration, eutrophic condition, and nutrient source and input (Bricker and Stevenson 1996).

For belowground biomass, I found no significant response to any of the nutrient additions. These findings are in agreement with those from a salt marsh in Connecticut, which received N, P and N+P fertilization (Anisfeld and Hill 2012). However, Valiela et al. (1976) reported the positive effect of N fertilization on roots while no significant effect was found on rhizomes in a Massachusetts salt marsh. Meanwhile Darby and Turner (2008) reported no significant effect of N fertilization for a saltmarsh in Louisiana, similar to my results for root biomass, but a negative effect of P fertilization on belowground biomass. The belowground biomass in my sods appeared to be sufficient for adequate nutrient uptake, given the absence of any change in belowground biomass due to nutrient addition after 25 months of nutrient application. I also measured belowground plant accumulation (production into ingrowth bags) and again found that neither N nor P significantly affected belowground growth compared to control, although on average PO_4 addition reduced belowground biomass accumulation to ca. 35%, albeit not significantly. The response-trends of belowground biomass accumulation to nutrient addition were similar to those identified for aboveground biomass, although there were fewer significant differences for belowground accumulation (Figures 1 and 4).

The addition of sulfate to the marsh sods had detrimental effects to both aboveground biomass and belowground biomass accumulation and changed plant species dominance, responses similarly observed in freshwater sod-mesocosm experiments in the Netherlands (Lamers et al. 1998, Geurts et al. 2009). Sulfate can be reduced in the flooded anaerobic soils to sulfide and act as a phytotoxin (Smolders and Roelofs 1995, Lamers et al. 2001, Swarzenski et al. 2008). Sulfide toxicity, as a result of sulfate reduction, may have reduced plant uptake of NH_4 , PO_4 , or K (Koch and Mendelssohn 1989, Bradley and Morris 1990, Lamers et al. 1998, Morris et al. 2013) due to disturbances of cell metabolism and energy transfer (Lamers et al. 2013). A number of studies have reported negative effects of dissolved sulfide on the productivity of *Spartina alterniflora* in salt marshes (King et al. 1982, Koch and Mendelssohn 1989, DeLaune et al. 2003). Since plant growth in my mesocosms was N limited, changes in NH_4 uptake kinetics from sulfide toxicity might have caused the observed changes in growth response (Bradley and Morris 1990, Koch et al. 1990). Furthermore, production of hydrogen sulfide in the biochemically reduced conditions of the sods could have facilitated precipitation of trace elements such as ferrous iron, zinc and copper that are necessary for plant growth (Belzile and Morris 1995). Sulfate addition also increased porewater conductivity and sulfate, both of which may have played a role, along with porewater sulfide, in controlling the observed responses in plant growth.

In addition to changes in plant biomass production, plant community structure changed due to eutrophication. Species composition in this research was similar to that found in other studies investigating the oligohaline marshes adjacent to our donor marsh (Baldwin et al. 1996, Baldwin and Mendelssohn 1998, Slocum and Mendelssohn 2008). However, by the end of experiment, plant species did not respond equally to nutrient enrichment, which lead to a modification in

plant community composition. Slocum and Mendelssohn (2008) reported that nutrient enrichment in an oligohaline marsh adjacent to my donor marsh increased aboveground biomass by 41% and modified species dominance. However, unlike my findings, Slocum and Mendelssohn (2008) also found a 20% reduction in species richness. Other studies have also reported significant reductions in species richness due to nutrient enrichment (Aerts and Berendse 1988, Drexler and Bedford 2002). My findings, however, are supported by Graham and Mendelssohn (2010) who reported that an oligohaline marsh adjacent to my donor marsh could assimilate as much as $200 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ without any change in species richness, although a shift in species dominance occurred (Graham and Mendelssohn 2010) as I observed in present study. The absence of a shift in species richness may be a result of the relatively short period of my study (2.5 years) or the lack of wind and water transported propagules in a greenhouse mesocosm study. With respect to species dominance, changes have been reported in other marshes as a response to eutrophication (Steward and Ornes 1975, Aerts and Berendse 1988, Doren et al. 1997, Rickey and Anderson 2004, Li et al. 2010). Plant species dominance changed from *S. lancifolia* in year 1 to co-dominance with *S. robustus* in year 2. In the third year, *S. robustus* became dominant over *S. lancifolia*. This shift in species dominance may have occurred gradually due to *S. lancifolia* loss in all sods that received SO_4 and the differential negative effects of generated sulfide on plant nutrient uptake by these species. Also, as for biomass, higher conductivity and sulfate may have contributed to the changes in species dominance.

Nutrient enrichment may also influence soil processes, such as organic matter decomposition. Nutrient addition may accelerate plant litter decomposition by stimulating microbial activity, as reported in several studies (Fell et al. 1984, Carreiro et al. 2000, Allen and Schlesinger 2004). Laursen (2004) found a positive effect of P ($100 \text{ kg P ha}^{-1} \text{ yr}^{-1}$) together with a

high N loading ($1200 \text{ kg N ha}^{-1}\text{yr}^{-1}$) from a marsh adjacent to my donor oligohaline marsh. Feller et al. (2003) also reported the positive effect of P, but not N, on decomposition rate in a mangrove forest in Belize. Anisfeld and Hill (2012) did not find a significant effect of N or P fertilization on litter decomposition rate in a Connecticut salt marsh, similar to my findings for plant litter. However, I did find that sulfate negatively affected litter decomposition as well as cellulolytic activity in the sods. Slower litter decomposition with SO_4 could be a result of the somewhat more acidic environment in the SO_4 -amended sods (Doran and Jones 1996) to the negative effects of hydrogen sulfide, resulting from sulfate-reduction, on microbial activity, and/or to the elevated conductivity and sulfate levels in the sod porewater.

Changes in root biomass production and/or decomposition can influence soil shear strength. Plants with a greater mass of roots and rhizomes were associated with greater soil strength (Turner 2011). Turner (2011) reported that N+P did not affect the soil shear strength relative to the control in a salt marsh at the depth of 20 to 50 cm, although a decrease in soil shear strength was noted between 60 and 100 cm. In addition, (Kiehn 2013) reported no changes in soil shear strength after fertilization with $20 \text{ g N m}^{-2} \text{ y}^{-1}$ ($200 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) and $3 \text{ g P m}^{-2} \text{ y}^{-1}$ ($30 \text{ kg P ha}^{-1} \text{ yr}^{-1}$) in an oligohaline marsh in Louisiana. In this study, NH_4 addition did not affect belowground biomass significantly ($p > 0.05$); however, NH_4 on average tended to increase belowground accumulation by 140 % compared to controls and belowground biomass by 40 %, which may explain the positive effect of NH_4 enrichment on soil shear strength.

Sediment additions

Sediment accretion in the Mississippi River Delta is vital to wetland stability (Blum and Roberts 2009). However, Mississippi River levees and upstream dams prevent sediment accumulation and reduce loading rates to nearby marshes by 50% (Blum and Roberts 2009).

Sediment accretion, in association with organic matter accumulation, increases marsh elevation to keep pace with subsidence and sea level rise and to provide a more favorable condition for plant growth and marsh sustainability (Lane et al. 2006). Sediment addition ameliorates soil fertility and soil aeration, flooding, and interstitial sulfide stresses (Nyman et al. 1990, Mendelssohn and Kuhn 2003, Slocum and Mendelssohn 2008, Stagg and Mendelssohn 2010). Moreover, sediment enrichment accelerates the restoration of degraded salt marshes, as reported in Louisiana after extensive marsh dieback (Schrift et al. 2008).

In the present study, the bulk density of the soil increased with the addition of sediment, and aboveground biomass also concurrently increased. However, no significant increase in belowground biomass or belowground biomass accumulation following sediment loading was detected. An increase in aboveground biomass has been reported with large-scale sediment additions to degrading salt marshes in Louisiana (Mendelssohn and Kuhn 2003). However, freshwater diversion waters carry much less sediment. Hence, other researchers investigating plant response to freshwater diversions (Carpenter et al. 2007) have seen little effect on vegetation biomass. The absence of response to sediment addition may be related to an insufficient amount of sediment released from freshwater diversions or to a relatively short study period that was not long enough to observe significant changes in plant biomass production. However, the long-term effects of the sediment on marsh condition depends on a number of factors including sediment volume and texture, and its resistance to compaction (McKee and Cherry 2009).

Implication for Diversions and their effects on marsh stability

The original goals of Mississippi River freshwater diversions were to support a bountiful oyster crop by reducing salinity each year, and also to provide important nursery and habitat for

crab, shrimp and fish (Lane et al. 1999). In the interim, several studies confirmed that a substantial reduction in sediment transport to the lower Mississippi that occurred in the middle and at the end of the twentieth century, as well as other processes, may have accelerated the loss of wetlands (Boesch et al. 1994). Wetland loss in the Louisiana coastal zone suggested the potential role of freshwater diversions in providing sediment for wetlands restoration, despite the fact that these diversions were not designed for sediment delivery to cope with subsidence. Kearney et al. (2011) reported that freshwater diversions fail to conserve mature brackish and tidal freshwater marshes. DeLaune et al. (2003) suggested that if it was not possible to increase sediment delivery in freshwater diversions, freshwater discharge would at least reduce salinity levels, which consequently could reduce sediment requirements for marsh maintenance. However, my research has demonstrated that plant responses to nutrient and sediment enrichment based on simulated diversion water loading rates can have important effects on wetland structure and function, depending on the nutrient. Nitrogen addition stimulated biomass production, mainly aboveground. Sulfate, in contrast, had extensive negative impacts to above- and belowground biomass production. Sediment addition did not offset the negative effects of sulfate in this study, although sediment generally increased aboveground biomass and soil bulk density.

SUMMARY

The results of this study indicate the sulfate ion, which is elevated in Mississippi River diversion water, is a potentially problematic nutrient. No negative effects from any of the other nutrients were found. Sediment had a positive effect on aboveground biomass and soil bulk density, but it did not modify the negative impacts of sulfate on plant response. Although sediment enrichment has repeatedly been found to be a benefit for coastal wetlands in providing essential elements for plant growth and in facilitating hydrogen sulfide precipitation in the soil (Nyman et al. 1990, Mendelssohn and Kuhn 2003, Slocum and Mendelssohn 2008), I did not observe an offsetting effect, either positive or negative, of sediment addition on nutrient-induced plant response. However, these results must be interpreted cautiously because of the limitations imposed by greenhouse mesocosm experiments. Nonetheless, my results suggest future pathways for continued research, especially for field experiments. These results are informative for wetland restoration activity and suggest that sulfate may be an ion of concern in diversion water, although its impact may vary as a function of other environmental factors. Also, by increasing soil bulk density and promoting aboveground biomass, sediment loading can be beneficial to coastal wetlands, especially where subsidence rates are high and plant flooding excessive as found in the Mississippi River Delta ecosystem. Thus, the optimization of diversions to carry sediments, along with continued surveillance of potential nutrient effects, would be a wise management approach.

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