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Wetland Soils

Nutrient Availability and Soil Organic Matter Decomposition Response to Prescribed Burns in Mid-Atlantic Brackish Tidal Marshes

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Prescribed fire in tidal marshes has been shown to generate short-term increases in plant-available nutrients, but the long-term implications of fire on nutrient availability and organic matter decomposition have not been well established. Two manipulative experiments were conducted over 1 yr within long-term annual burn and no-burn blocks at the Blackwater National Wildlife Refuge in Cambridge, MD to study the effects of canopy removal and ash deposition on nutrient availability and organic matter decomposition rates following burning. Ash deposition had no significant effects on study variables. At no-burn sites, porewater NH_4^+ and PO_4^{3+} were significantly lower in July in sites with canopy removal (0.15 mg NH_4^+ L^{-1} and 0.04 mg PO_4^{3+} L^{-1}) compared to sites with a canopy (0.73 mg NH_4^+ L^{-1} and 0.08 mg PO_4^{3+} L^{-1}). Similar results were found through a canopy replacement treatment at annual burn sites. Decreased porewater NH_4^+ and PO_4^{3+} corresponded to increased total biomass nutrient content. No-burn sites with canopy removal treatments showed significantly lower organic matter decomposition rates than did treatments with a canopy in July (66.5 vs. 74.1% cotton tensile strength loss), corresponding to decreases in porewater NH_4^+ and PO_4^{3+}. Plant ash provided a fertilizer pulse of 0.22 g N m^{-2} and 0.16 g P m^{-2}, which were amounts of N and P likely too small to increase plant productivity when deposited during late winter/early spring. Prescribed fire appears to affect nutrient availability and organic matter decomposition in these marsh soils primarily through the mechanism of increased biomass production due to canopy removal.

Abbreviations: CTSL, cotton tensile strength loss.

Fire is a natural component of many wetland ecosystems (Lynch, 1941; Kirby et al., 1988; Nyman and Chabreck, 1995). Historically, prescribed fire has been used to remove vegetation to facilitate seasonal hunting and trapping in tidal marsh ecosystems (O'Neil, 1949), for example to facilitate the trapping of muskrat (*Ondatra zibethicus*) by making their lodges more visible to hunters (Lay, 1945). Prescribed fire has become an integral part of resource management in coastal wetlands and is widely used as a technique to promote the growth of favorable wetland vegetation on the U.S. East and Gulf Coasts (Lynch, 1941; Nyman and Chabreck, 1995; Mitchell et al., 2006; Owens et al., 2007) and to reduce the risk of unpredictable and/or uncontrollable wildfire, which pose serious risks to property owners and the public (Nyman and Chabreck, 1995; Mitchell et al., 2006; Flores et al., 2011). Prescribed fire often has a stimulatory effect on plant production; researchers have suggested that this effect may be the result of fertilization through ash deposition, canopy removal, or the ephemeral heat during a fire (Hackney and de la Cruz, 1983; Nyman and Chabreck, 1995; Johnson and

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Knapp, 1993; Gabrey and Afton, 2001; Cahoon et al., 2004). There is increasing concern that prescribed fire may contribute to marsh loss due to the removal of C and an associated lowering of peat development; however, the stimulatory effects of burning on plant production, particularly belowground, may offset these losses and may cause a net increase in accretion rates (Nyman and Chabreck, 1995; Cahoon et al., 2004). Prescribed fire may also indirectly affect peat decomposition rates through effects on nutrient availability.

Fire has both direct and indirect effects on nutrient cycling in tidal marsh ecosystems that may affect plant productivity and organic matter decomposition dynamics (Faulkner and de la Cruz, 1982; Wilbur and Christensen, 1983; Schmalzer and Hinkle, 1992; Wan et al., 2001). Nutrients contained in surface vegetation are redistributed through particulate (ash) and non-particulate (volatilization) pathways following fire (Raison et al., 1985). Particulate material can be redistributed from a burn site to adjacent areas through wind, rainfall, erosion, runoff, and leaching or can remain on-site and may have a significant impact on the soil nutrient status and water quality (Badia and Marti, 2003; Hauer and Spencer, 1998; Thomas et al., 1999; Townsend and Douglas, 2004). Increased plant-available nutrient concentrations in soils and nearby surface waters following prescribed burns have been observed in several studies; specifically higher levels of N, P, and base cations have been found (Schmalzer and Hinkle, 1992; Ilstedt et al., 2003; Murphy et al., 2006; Ubeda et al., 2005). Nutrients with relatively low volatilization temperatures, such as C, N, and S, may be removed from the burn site through volatilization, resulting in atmospheric pollution (Cachier et al., 1995; Liu et al., 2000; McNaughton et al., 1998; Wan et al., 2001). Fire has been found to increase the nutritional content of marsh vegetative regrowth to benefit marsh wildlife (Smith et al., 1984; Nyman and Chabreck, 1995). Removal of standing vegetation stock also affects site microclimate and may alter microbial nutrient cycling by affecting soil temperature (Schmalzer and Hinkle, 1992). Soil microclimate will likely be warmer due to increased solar radiation contacting the soil surface, potentially favoring increased mineralization of plant nutrients.

Organic matter dynamics following prescribed fire events in tidal brackish marshes have not been extensively studied. Fire removes plant biomass from the marsh system, with associated mineralization and volatilization of nutrients, but the net effects on plant productivity and marsh accretion rates are uncertain (Smith et al., 2001). Burning may stimulate plant biomass production so that the net effect is an increase in organic matter and associated increases in marsh elevation (Schmalzer and Hinkle, 1992; Nyman and Chabreck, 1995; Cahoon et al., 2004). Stimulation of belowground production has been observed following prescribed burning in tidal marshes and may play a key role in marsh accretion (Pendleton and Stevenson, 1983). One study found that burned marsh areas recovered elevation faster than unburned areas following a peat collapse, which was attributed to an increase in soil organic matter volume (Cahoon et al., 2004).

Tidal brackish marshes must accrete vertically to maintain elevation in the intertidal zone or they will be lost to open water (Cahoon et al., 1998; Cahoon et al., 2006). The repeated removal of plant litter through burning could lead to decreased vertical accretion (Gabrey and Afton, 2001). Although plant production may increase after burning, it is unknown whether burned marshes export or accumulate the same amount of organic matter as unburned marshes (Nyman and Chabreck, 1995). Fire may also have both positive and negative effects on soil organic matter decomposition, which may indirectly affect accretion rates. Further, organic matter decomposition may also release plant nutrients, thereby stimulating increased biomass production, leading to a net organic matter increase and associated elevation gains (Cahoon et al., 2004). Highly organic soils are almost exclusively dependent on organic matter accumulation from root production and litter deposition for accretion, yet it is unknown if burning is a sustainable practice in tidal marsh ecosystems with organic-rich soils such as those in our study.

The objective of our study was to elucidate the soil organic matter decomposition and soil nutrient availability responses of brackish tidal marsh ecosystems to two primary mechanisms of prescribed fire—ash deposition and canopy removal. To our knowledge, this is the first study to separate out the influence of these separate mechanisms operating after prescribed fire events on organic matter decomposition and nutrient availability in these systems. We addressed this objective with a 1-yr manipulative study embedded in no-burn and annual burn treatment sites at the Blackwater National Wildlife Refuge in Cambridge, MD. This study was conducted as a companion to the study conducted by Bickford et al. (2012), who found that canopy removal, and not ash deposition, was the dominant mechanism affecting plant production in these systems. Bickford et al. (2012) also analyzed the vegetative community response to fire in the context of disturbance ecology.

MATERIALS AND METHODS

Site Description

The study was conducted at the Blackwater National Wildlife Refuge (38°27′0″ N lat; 76°7′12″ W long) located on the Eastern Shore of Maryland in Dorchester County (Fig. 1). The refuge is located 19 km south of Cambridge, MD and is managed by the U.S. Fish and Wildlife Service Chesapeake Marshlands Complex. It is over 109 km^2 in size, with approximately one-third of the refuge in fresh and brackish tidal wetlands.

The study sites were dominated primarily by the salt marsh species *Schoenoplectus americanus* (Pers.) Volkart ex Schinz & R. Keller (threesquare bulrush), *Spartina patens* (Aiton) Muhl. (salt hay), and *Distichlis spicata* (L.) Greene (salt grass), and were near communities of *Spartina alterniflora* Loisel. (smooth cordgrass) and *Juncus roemerianus* Scheele (black needlerush) in some cases (The PLANTS Database, http://plants.usda.gov, accessed 23 Aug. 2011). The dominant soil series in the study area were Bestpitch (clayey, mixed, euic, mesic Terric Sulfihemist) and Transquaking (euic, mesic Typic Sulfihemist), which are both characterized by thick organic surface deposits, often at depths greater than a meter, overlying loamy or fine mineral sediments (Table 1). Two of the sites were located on areas where the marsh was fringing on upland forest, with Honga (loamy, mixed, euic, mesic Terric Sulfihemist) and Sunken (fine-silty, mixed, active, mesic Typic Endoaqualf) soils (Table 1). The horizons in these soils typically display a sequence of fibric soil material near the surface grading into hemic organic horizons above the mineral layer at depth. The tidal cycle at Blackwater is diurnal but strongly influenced by wind. Typically, water levels are maintained in the range of 10 cm above and below the marsh surface, although storm surges and wind events can increase this range to 30 cm above and below the soil surface for days at a time (Sipple, 1979). Salinity measurements throughout the experiment ranged from 5 to 15 ppt, but more typical values were in the 8 to 12 ppt range, placing Blackwater in the mesohaline salinity range (Cowardin et al., 1979).

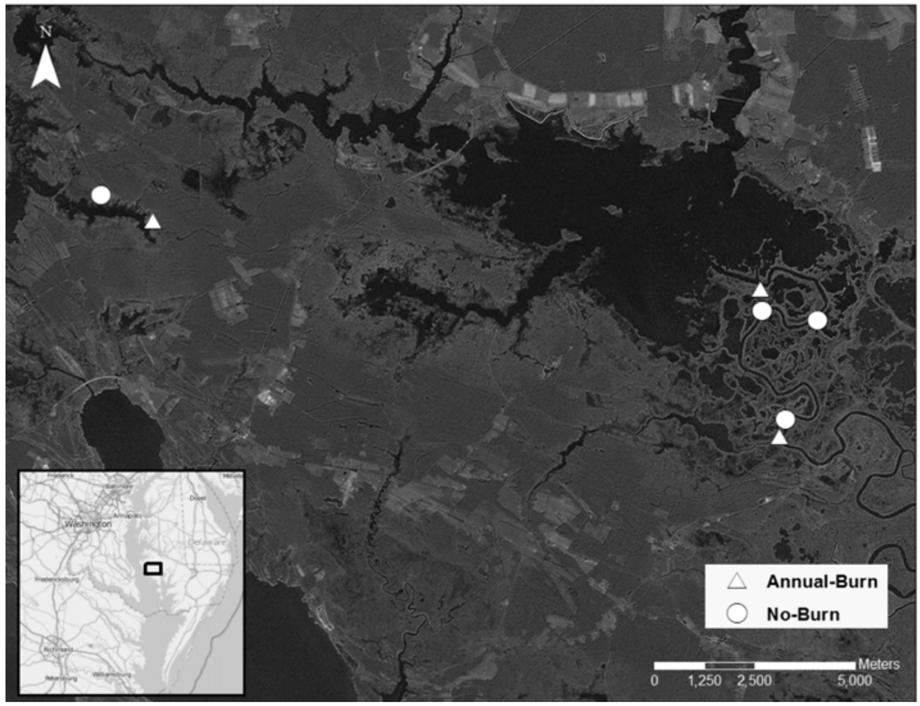

Fig. 1. Map of the Blackwater National Wildlife Refuge on the Eastern Shore of Maryland; inset provides map location within mid-Atlantic coast.

The study period was 1 yr from January 2009 through January 2010. The Blackwater National Wildlife Refuge has been conducting annual burning of most marsh areas throughout the refuge from 1970 through 1995, when management blocks of different fire frequencies were established. Thus, the no-burn sites had been released from burning practices for 14 yr (U.S. Department of the Interior, 2006).

Site Selection

Two separate experiments were established in January of 2009 within blocks of the long-term fire management study being conducted at the refuge. Both experiments had a completely randomized block design. The first experiment took place at four sites in annual burn blocks; the second experiment took place at three sites on no-burn blocks (one site could not be burned due to environmental conditions and was removed) (Table 1). Plot establishment and data collection were identical in each experiment. Selection of plots within each area was conducted with the goal of matching the vegetation and topographic characteristics of adjacent USGS surface elevation table (SET) sites (Cahoon et al., 2010). Ponds, holes, muskrat dens, and wildlife trails were avoided in site selection. Plots measured 3 by 4 m.

No-Burn Experiment

Four treatments were applied to each of the four no-burn sites: canopy removal, ash deposition, canopy removal + ash deposition, and a control, which was left undisturbed within these no-burn blocks. Treatments were randomly applied and replicated three times per site ($n = 48$). Canopy removal was applied on 17 Mar. 2009 and involved clipping of senesced vegetation at ground level using hedge trimmers. Approximately 3 cm of plant shoot stubble was left protruding from the marsh surface following this treatment, reproducing aboveground plant conditions similar to those following a prescribed burn. The entire clipped aboveground biomass was removed from the plot and used to generate an ash/biochar mixture (referred to as “ash”) for the ash deposition treatment. This biomass was dried at 40°C for 24 h and ignited in galvanized steel bins, during which temperature was taken by using

Table 1. Site characteristics in the no-burn and annual burn experiments at the Blackwater National Wildlife Refuge in Dorchester County, MD. Salinity and pH values are the mean across the duration of the study.

Experiment	Site ID	pH	Salinity	Dominant soil series	% cover *S. americanus* (sedge)	% cover *S. patens/D. spicata* (grasses)
			ppt			
No-Burn	1D	6.6	8.9	Sunken	5 ± 2	47 ± 5
	2D	6.5	7.6	Bestpitch & Transquaking	43 ± 2	3 ± 1
	3D	6.6	8.8	Bestpitch & Transquaking	14 ± 4	31 ± 8
	7D	6.5	8.7	Bestpitch & Transquaking	28 ± 2	1 ± 0
Annual Burn	1A	6.4	12.7	Honga	30 ± 3	11 ± 2
	2A	6.4	10.2	Bestpitch & Transquaking	8 ± 1	38 ± 5
	3A	6.4	9.7	Bestpitch & Transquaking	10 ± 2	35 ± 3

an infrared temperature gun. After combustion, the ash was homogenized and spread evenly over the ash deposition plots using a 1-mm mesh sifter. This ash was deposited on the soil surface of the ash deposition and canopy removal + ash deposition plots in April 2009. Water table levels were below the marsh surface during application, which ensured good soil contact.

Annual Burn Experiment

Two treatments were applied to each of the three annual burn sites: canopy replacement, which consisted of construction of an artificial canopy placed on the burned marsh surface, and a control, which was therefore burned within these annual burn blocks. Treatments were randomly applied and replicated three times per site (n = 18). Following prescribed burning of each of the sites in February and April 2009, artificial canopies were constructed and put over plots receiving the canopy replacement treatment. Wood frames of 3 by 4 m were covered with 6.35-mm mesh hardware cloth. Wooden legs allowed the frame to be hammered securely into the marsh surface until the hardware cloth was 15 cm off of the ground. Senesced plant shoots were then inserted into the hardware cloth using plant material clipped from nearby unburned marsh areas (Fig. 2). Notes on species richness, percent cover, and percent light penetration taken preburn in January and February were used to transplant a mix and density of species that mimicked natural conditions. Light availability measurements were taken using a line quantum sensor that was 1 m long by 12.7 mm wide (Model LI-191SA; LI-COR, Lincoln, NE) that measured photosynthetically active radiation (PAR) as wavelengths of 400 to 700 nm. Light penetration through the artificial canopies was slightly lower than the natural canopies, likely due to a slightly denser placement of vegetation in the mesh cloth compared with the natural system.

Data Collection and Laboratory Analyses

Decomposition of cellulose present in cotton fabric was used as a relative index of labile soil organic matter decomposition (Latter and Harrison, 1988; Harrison et al., 1988). The cotton strip technique for measuring organic matter decomposition is based on the loss of tensile strength of a buried cotton fabric composed of cellulose fibers. Cellulose comprises about 70% of the organic C found in plant tissue, and therefore, the rate of its decay is a key factor in plant matter decomposition (Mendelssohn et al., 1999; Mendelssohn and Slocum, 2004). Cotton tensile strength loss gives a quantitative measurement of the decomposition rate of a standard material such that comparisons across treatments reflect differences in the decomposition environment and not differences in C quality (French, 1988; Harrison et al., 1988; Mendelssohn et al., 1999; Mendelssohn and Slocum, 2004).

At each plot, a strip of 10 by 35 cm of a 200-thread-count unbleached muslin fabric was inserted into the marsh soil by first making a pilot hole with the aid of a sharpshooter shovel, and then placing the fabric against the back of the shovel and inserting it into the hole. The saturated environment of the marsh soil provided enough suction to gently pull the cotton strip away from the shovel and into contact with the soil. The soil surface level was marked with a lateral cut into the fabric for future reference during analysis. The first set of cotton strips were placed into the center of the plots on 18 and 19 May and were retrieved from the ground on 2 and 3 June. The second set of cotton strips were deployed on 30 June through 2 July and collected on 15–17 July. These dates were selected to represent periods of rapid growth in marsh systems, during which plants are most likely to be nutrient limited. The 15-d deployment period was similar to that used by Mendelssohn et al. (1999) in a *Phragmites australis* (Cav.) Trin. ex Steud. (common reed) marsh in Denmark. Three reference cotton strips were inserted into the marsh and immediately removed at each deployment date. These strips were used to quantify the tensile strength of the nondecomposed cotton fabric, taking into account insertion and removal strength loss issues. Retrieval involved a mixture of light pulling on the exposed fabric and excavation around the strip. Immediately following retrieval, the strips were washed with freshwater to remove any attached soil particles or debris and then were placed in bags for transport. Back in the laboratory, these strips were washed more thoroughly with deionized water and allowed to air dry for 72 h. The strips were then cut laterally into 4-cm horizontal segments from 0 to 20 cm, giving strip segments representing 0–4, 4–8, 8–12, 12–16, and 16–20 cm. Hand fraying was then used to reduce the 4-cm strip segments down to the middle 2.5 cm of the strips, to ensure close to equal widths of all the strips and because this was the largest size the tensile strength testing machine would accept. Tensile strength was measured with an Instron 4201 tensile strength machine with 2.5 cm by 2.5 cm clamp teeth spaced 5 cm apart. Measurements were made at lab room temperature (~24°C) at 100% humidity attained by saturating the strips with deionized water before analysis. Individual strip segment tensile strength loss in kilograms-force was calculated

Fig. 2. Photos of canopy replacement treatment applied to annually burned sites at the Blackwater National Wildlife Refuge in Dorchester County, MD. Plant shoot placement in mesh hardware cloth shown on left. Artificial canopies sat ∼15 cm off of the soil surface (shown on right).

relative to the mean of the reference cotton strips and expressed as a percent.

Porewater was sampled using "sipper" wells (Teflon tubing wells constructed as described in Marsh et al., 2005) installed in each plot via a pilot hole. The Teflon tubing (9 mm outer diameter by 6 mm inner diameter) had perforations extending 2.5 cm above and below the target sampling depths and were sealed at the bottom with silicone caulk. The top of the wells were capped with a Tygon tube that terminated in a three-way stopcock. A plastic syringe was used to draw porewater twice during the growing season, once on 21 and 22 May and again on 9 and 10 July, coinciding with the midpoint of the two cotton strip deployment set dates. At each date, one 30-mL syringe sample of porewater was filtered in the field through a 0.45 μm uniflow in-line filter and acidified with hydrochloric acid. The clear filtered samples were then used to measure NH_4^+, PO_4^{3-}, and H_2S concentration in the marsh porewater. Samples for porewater NH_4^+ and PO_4^{3-} (Kuo, 1996) were frozen until analysis using an autoanalyzer. Sulfide was measured the same day the sample was taken using an ion specific electrode. Ammonium levels were of particular interest in this study, as this is the main form of N in tidal marshes and the limiting nutrient in these ecosystems (Valiela and Teal, 1974).

Aboveground biomass was sampled during the peak of the growing season in late July 2009, when plants were in the very early stages of senescence. Biomass was collected from two 0.25-m^2 quadrats in each plot by clipping at the stem base. Biomass samples were dried at 60°C and ground using a no. 20 sieve.

Plant biomass and ash samples were analyzed for C and N content using a LECO CHN2000 Analyzer (LECO Laboratory Equipment Corp., St. Joseph, MI) and for total P, K, Ca, Mg, and S by digestion and ICP (by the Penn State Agricultural Analytical Services Lab). The biomass collected from plots with an artificial canopy in the annual burn study was green growth from the 2009 growing season. The senesced vegetation, which was implanted into the artificial canopy, was not collected for analyses. Automated temperature loggers were also installed at depths of 5 and 15 cm (data presented in Bickford, 2011).

Nutrient Loss and Burn Temperature Modeling

We extended our understanding of ash-deposited nutrient additions broadly by using a model developed by Qian et al. (2009) for the estimation of postfire nutrient loss from cattail (*Typha angustifolia* L.) and sawgrass (*Cladium mariscus* (L.) Pohl) vegetation in the Everglades. This model estimates postburn ash N contents using the equation $y = 0.01(x^{1.46})$ where y = % N loss and x = burn temperature (°C) ($r^2 = 0.83$). Qian et al. (2009) also developed a model for ash total P/leaf total P vs. burn temperature of $y = 5.96 + 3.25e^{0.004x}$ where y = ashTP/leafTP and x = burn temperature (°C) ($r^2 = 0.73$).

Statistical Methods

The no-burn and annual burn studies represented two separate randomized complete block experiments. Different sites at the refuge represented blocks. The no-burn experiment was a two-way factorial arrangement (with ash deposition and canopy removal as the two factors) consisting of four blocks with three replications per block. The annual burn experiment was a one-way design consisting of three blocks with three replications per block. An initial baseline dataset was not available to estimate variability, thus type 2 errors could potentially have been made. Differences in environmental variables were tested using ANOVA. Cotton strips were analyzed by depth using repeated measures ANOVA. Statistical analyses were performed using the "Proc Mixed" function in SAS (SAS Institute, Cary, NC). Variations in marsh site conditions were analyzed statistically as the block variable.

Significant treatment × site interactions at no-burn sites were further analyzed based on dominant species composition (Table 1). Species composition of sites 2D and 7D was dominated by a high percentage cover of the sedge *S. americanus*, which made up 83% of the total vegetative cover. Sites 1D and 3D were dominated by a high percentage cover of the grasses *S. patens* and *D. spicata*, making up 88% of the vegetative cover. Therefore, sites 2D and 7D were categorized as sedge-dominated and sites 1D and 3D were categorized as grass-dominated.

RESULTS

Cotton Strip Decomposition

No-Burn Experiment

No significant main effects or interactions involving the ash treatments were observed in the no-burn experiment; therefore, results are presented for canopy removal effects only. In May, cotton strip decomposition rates did not significantly differ between canopy removal treatments in the no-burn experiment for the 0- to 20-cm depth (Fig. 3). In July, there was a significant canopy removal main effect ($P = 0.04$), with 7.6% lower rates of decomposition

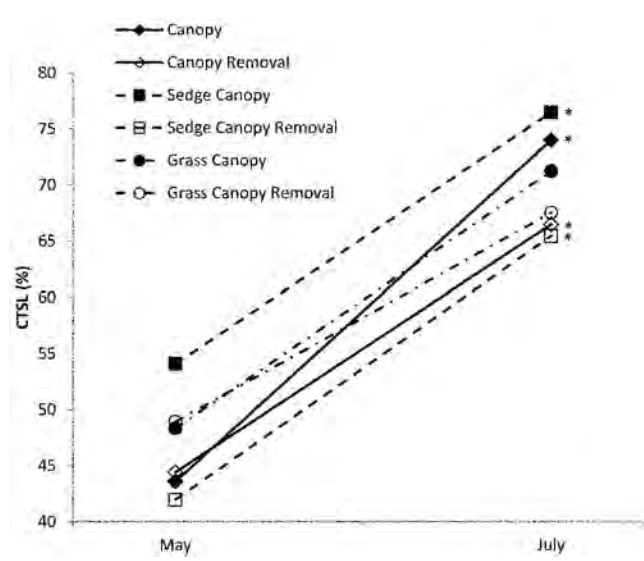

Fig. 3. Mean cotton tensile strength loss (CTSL) for no-burn sites from 0- to 20-cm depths for May and July sets shown over the entire no-burn experiment and sedge- and grass-dominated blocks ($n = 48$) at the Blackwater National Wildlife Refuge in Dorchester County, MD. Asterisks indicate that a pair of measurements for a given month within a given section of the experiment (pairs with same line type) were significantly different ($\alpha = 0.05$).

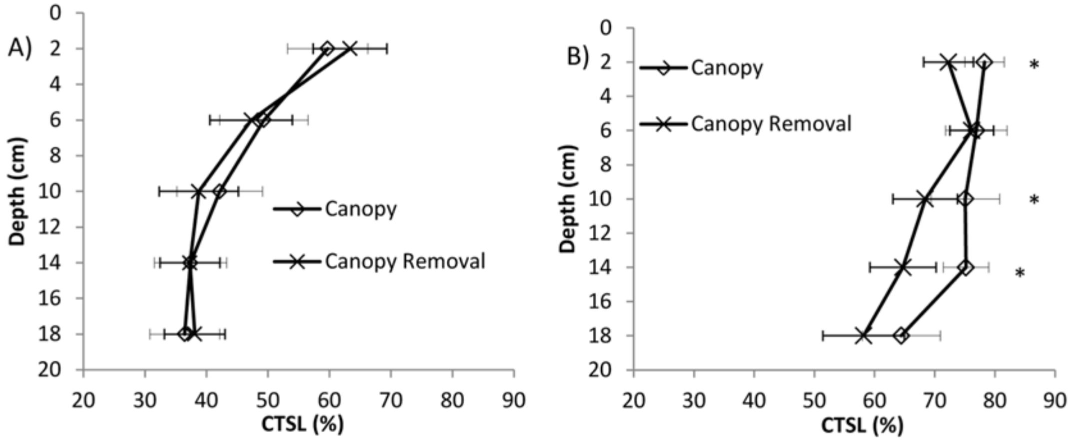

Fig. 4. Cotton tensile strength loss (CTSL) by depth for no-burn sites from 0- to 20-cm depths for (a) May and (b) July sets, with bars representing standard errors (*n* = 48) at the Blackwater National Wildlife Refuge in Dorchester County, MD. Asterisks indicate that a pair of measurements for a given depth increment in a given month were significantly different (α = 0.05).

with canopy removal. Overall, decomposition rates were greater in July than in May (presumably due to warmer temperatures).

When analyzed by dominant species, no significant effects of canopy removal were observed in grass-dominated sites; significant effects in sedge-dominated sites were observed for decomposition from 0 to 20 cm in July only (Fig. 3). For the July sedge-dominated sites, canopy removal treatments had 11% lower decomposition rates than sites with canopy removal (P = 0.01). A site × canopy removal interaction was observed in July, with site 2D having a 14% decrease in mean decomposition rate with canopy removal, while the decrease at site 1D was 2.8%. A nonsignificant canopy removal effect trend was observed in the May sedge-dominated sites (P = 0.07), with canopy removal sites trending towards lower rates of decomposition.

The highest rates of cotton strip decomposition rates were near the soil surface, with decreasing decomposition rates at depth. The May cotton strip set showed no significant main effects or interactions by depth (Fig. 4a). However, the July data (Fig. 4b) showed a significant canopy removal effect in which canopy removal decreased decomposition for the 0- to 4-cm depth (P = 0.02), 8- to 12-cm depth (P = 0.02), and 12- to 16-cm depth (P = 0.03). Additionally, the 0- to 4-cm depth had a significant site × canopy removal interaction (P = 0.01) due to a relatively small difference in decomposition rates between canopy and canopy removal treatments at site 1D [2.1% cotton tensile strength loss (CTSL)], while there was at least a 14% CTSL difference between treatments at the other three sites.

The May sedge-dominated sites showed no treatment or interaction effects at any depth (Fig. 5b). The July sedge-dominated sites (Fig. 6b) showed a significant canopy effect at the 0- to 4-cm depth (P = 0.01), as canopy removal sites exhibited 19% lower decomposition rates than plots with a canopy. A significant canopy removal effect was also observed at the 8- to 12-cm depth (P = 0.02), as canopy removal plots exhibited 16% lower decomposition rates than canopy sites.

Grass-dominated sites in May (Fig. 5a) showed a significant site × canopy removal interaction (P = 0.04), but no significant canopy removal main effect from 0 to 4 cm. Plots with canopy removal tended to have lower decomposition rates than sites with a canopy, especially at site 1D, in which the canopy removal plots showed 11% lower decomposition than canopy plots. No site or treatment effects were observed for the grass-dominated sites in July (Fig. 6a).

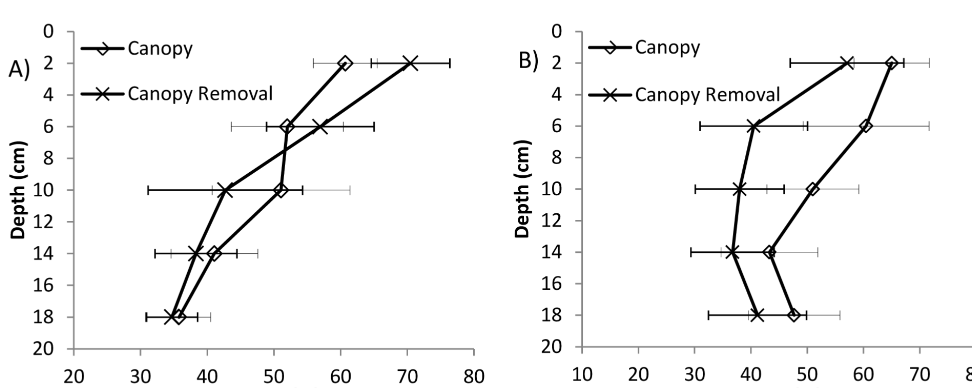

Fig. 5. Cotton tensile strength loss (CTSL) by depth for no-burn sites at the Blackwater National Wildlife Refuge in Dorchester County, MD from 0- to 20-cm depths for May sets separated by (a) grass-dominated and (b) sedge-dominated sites, with bars representing standard errors (*n* = 48). No pair of measurements at any depth were significantly different from each other (α = 0.05).

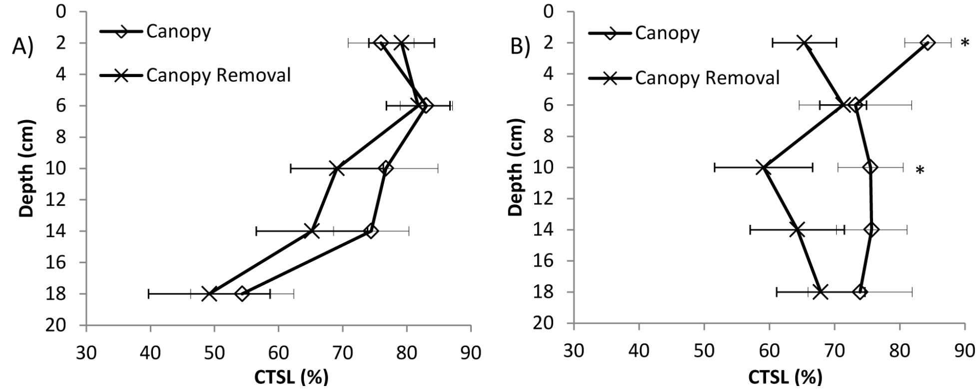

Fig. 6. Cotton tensile strength loss (CTSL) by depth for no-burn sites at the Blackwater National Wildlife Refuge in Dorchester County, MD from 0- to 20-cm depths for July sets separated by (a) grass dominated and (b) sedge-dominated sites, with bars representing standard errors (*n* = 48). Asterisks indicate that a pair of measurements for a given depth increment in a given set of sites were significantly different (α = 0.05).

Annual Burn Experiment

The annual burn experiment did not have any significant canopy replacement effects for the rate of decomposition in either May or July when analyzed over the 0- to 20-cm depth or when analyzed by depth increments (Fig. 7). The July decomposition rates did show a significant ($P = 0.03$) site × canopy replacement effect from 0 to 4 cm, with site 3A having lower rates of decomposition with canopy replacement treatment, while the other two sites did not show differences between treatments.

Porewater Nutrients

No-Burn Experiment

As was found for the cotton strip decomposition data, no significant main effects or interactions involving the ash treatment were observed; results are presented only for canopy removal effects.

For no-burn experiment sites, only the July porewater NH_4^+ ($P = 0.004$) and PO_4^{3-} concentrations ($P = 0.019$) had a significant canopy removal main effect (Table 2). Sites with canopy removal had 80% lower porewater NH_4^+ than sites with a canopy. Additionally in July, porewater NH_4^+ showed a significant site × canopy removal interaction ($P = 0.04$) due to a relatively small difference in porewater NH_4^+ between canopy and canopy removal treatments at site 1D (0.02 mg NH_4^+ L^{-1}, respectively), while there was at least a 0.29 mg NH_4^+ L^{-1} difference between treatments at the other three sites. Porewater P was lower in the canopy removal treatments (0.04 ± 0.01 mg PO_4^{3+} L^{-1}) than in sites with a canopy (0.08 ± 0.02 mg PO_4^{3+} L^{-1}). A significant site × canopy removal effect ($P = 0.04$) was observed for May porewater NH_4^+, with sites with the canopy removed having lower porewater NH_4^+ for all sites except site 7D, where the canopy removal treatments were higher than plots with the canopy intact by 0.28 mg NH_4^+ L^{-1}.

When analyzed by dominant species, the only significant effect was for NH_4^+ ($P = 0.006$) for the July sedge-dominated sites (Table 3); sites with canopy removal had lower concentrations of porewater NH_4^+ than sites with a canopy. There was also a nonsignificant trend ($P = 0.058$) of lower porewater PO_4^{3-} concentrations at the sites with canopy removal.

No significant differences existed between treatments for porewater nutrients for either grass-dominated or sedge-dominated sites during May (Table 4). A significant canopy removal effect ($P = 0.048$) existed for porewater sulfide only at the sedge-dominated sites in May. For these sites, canopy removal treatments had higher porewater sulfide concentrations (14.77 ± 6.29 mg H_2S L^{-1}) than sites with a canopy present (6.51 ± 3.91 mg H_2S L^{-1}).

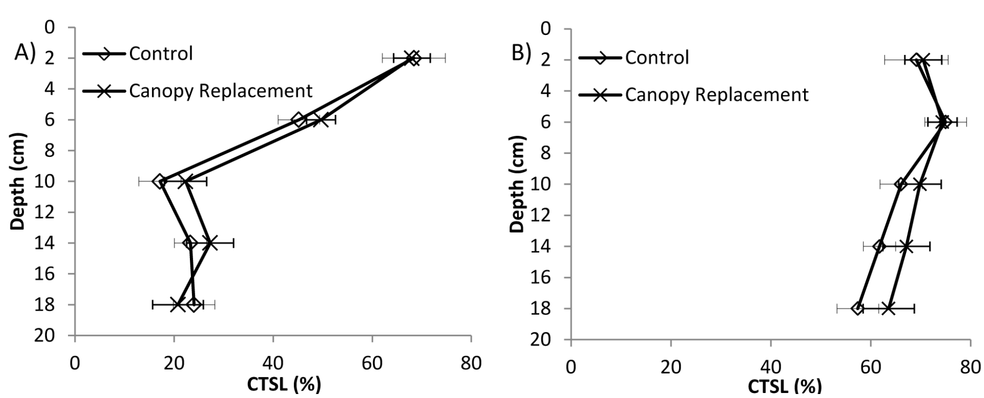

Fig. 7. Cotton tensile strength loss (CTSL) by depth for annual burn sites at the Blackwater National Wildlife Refuge in Dorchester County, MD from 0- to 20-cm depths for (a) May and (b) July sets, with bars representing standard errors (*n* = 48). Results of a preplanned contrast between treatments at the same depth measurements indicated that no pair of treatments were significantly different from each other (α = 0.05).

Table 2. May and July no-burn site porewater nutrients with treatments (n = 48) at the Blackwater National Wildlife Refuge in Dorchester County, MD. Letters indicate the results of an ANOVA between measurements; means with different letters were significantly different from each other (α = 0.05).

	Treatment			
	May		July	
Nutrient	Canopy	Canopy removal	Canopy	Canopy removal
	mg L^{-1}			
NH_4^+	0.52 ± 0.22	0.34 ± 0.13	0.73a ± 0.28	0.15b ± 0.06
PO_4^{3-}	0.06 ± 0.02	0.04 ± 0.01	0.08a ± 0.02	0.04b ± 0.01
S^{2-}	6.77 ± 3.18	8.00 ± 3.66	6.89 ± 1.39	7.32 ± 1.39

Annual Burn Experiment

A significant treatment (P = 0.014) effect existed for porewater NH_4^+ concentration in July, as control plots had less porewater NH_4^+ than plots with canopy replacement (Table 5). A significant site × treatment interaction (P = 0.01) existed for July porewater NH_4^+. Site 3A contained more NH_4^+ (1.32 mg NH_4^+ L^{-1} in canopy replacement treatment) than Sites 1A and 2A (0.36 mg NH_4^+ L^{-1} mean of canopy replacement treatments), and therefore, the magnitude of difference between treatments at Site 3A was greater than at the other two sites.

Plant Elemental Content

Sites with canopy removal had higher nutrient stocks due to greater biomass production (P < 0.05) (Table 6). Note that these greater nutrient stocks were observed despite generally lower plant nutrient concentrations (Geatz, 2012). Aboveground biomass in the no-burn sites was 40% higher in plots with canopy removal (447 ± 16 g m^{-2}) than in plots without (320 ± 23 g m^{-2}) (Bickford et al., 2012). In the annual burn sites, aboveground biomass was 68% higher in control plots (525 ± 33 g m^{-2}) than in plots with canopy replacement (312 ± 33 g m^{-2}) (Bickford et al., 2012). Belowground production showed similar trends, with a generally greater magnitude of effects (Bickford et al., 2012). We found that significant (P < 0.05) canopy removal main effects were observed for total plant tissue elemental stocks in the no-burn experiment for C, N, P, K, Mg, and S. Additionally, significant canopy removal and site × canopy removal effects (P < 0.05) were observed for these nutrients as the stimulation by canopy removal of increasing plant biomass varied in magnitude across sites, specifically with respect to whether or not the site was dominated by sedge or grass species. Sites dominated more heavily by the sedge had a greater biomass response to treatments, and therefore had higher nutrient stock differences than grass-dominated sites. Mean N/P ratios for grass vegetation were 9.4 (±0.4 SE); sedge vegetation had a mean ratio of 10.9 (±0.4 SE).

A significant treatment effect (P < 0.05) was also observed for all element concentrations (C, N, P, K, Mg, and S) examined in the annual burn experiment, with canopy replacement decreasing total plant tissue nutrient stocks (Table 7). A significant site × treatment effect (P < 0.05) was observed for C and N. Sites with a canopy replacement had lower stocks of nutrients than sites with no canopy due to burning. N/P ratios were similar to one another in both grass and sedge species. Mean N/P ratios for grass vegetation were 11.7 (±0.6 SE), while sedge vegetation N/P ratios were 11.2 (±0.6 SE).

Table 3. July no-burn site pore-water nutrients with treatments for grass- and sedge-dominated sites (n = 24) at the Blackwater National Wildlife Refuge in Dorchester County, MD. Letters indicate the results of an ANOVA between measurements; means with different letters were significantly different from each other (α = 0.05).

	Treatment			
	Grass-dominated sites		Sedge-dominated sites	
Nutrient	Canopy	Canopy removal	Canopy	Canopy removal
	mg L^{-1}			
NH_4^+	0.75 ± 0.52	0.27 ± 0.10	0.70a ± 0.42	0.13b ± 0.05
PO_4^{3-}	0.08 ± 0.03	0.04 ± 0.01	0.07 ± 0.02	0.04 ± 0.01
S^{2-}	5.52 ± 2.14	6.25 ± 2.25	8.36 ± 1.67	8.38 ± 1.72

Table 4. May no-burn site porewater nutrients with treatments for grass- and sedge-dominated sites (n = 24) at the Blackwater National Wildlife Refuge in Dorchester County, MD. Letters indicate the results of an ANOVA between measurements; means with different letters were significantly different from each other (α = 0.05).

	Treatment			
	Grass-dominated sites		Sedge-dominated sites	
Nutrient	Canopy	Canopy removal	Canopy	Canopy removal
	mg L^{-1}			
NH_4^+	0.68 ± 0.25	0.19 ± 0.11	0.36 ± 0.21	0.49 ± 0.21
PO_4^{3-}	0.06 ± 0.03	0.04 ± 0.01	0.05 ± 0.02	0.05 ± 0.02
S^{2-}	4.58 ± 2.00	1.23 ± 0.87	6.51a ± 3.91	14.77b ± 6.29

Ash Characteristics

Following controlled combustion of senesced vegetation for the no-burn experiment, 6.67 ± 0.45% of plant tissue dry-weight biomass was recovered as ash (Table 8). Both C and N were lost in amounts greater than 90% compared to original plant biomass stocks. Over 80% of the original stocks of S was volatilized and lost to the atmosphere during combustion. Approximately half of the original P was lost due to particulate losses, while Ca, K, and Mg losses were negligible.

Table 5. May and July annual burn site pore-water nutrients with treatments (n = 18) at the Blackwater National Wildlife Refuge in Dorchester County, MD. Letters indicate the results of an ANOVA between measurements; means with different letters were significantly different from each other (α = 0.05).

	Treatment			
	May		July	
Nutrient	Control	Canopy replacement	Control	Canopy replacement
	mg L^{-1}			
NH_4^+	0.27 ± 0.14	0.43 ± 0.15	0.13a ± 0.04	0.79b ± 0.20
PO_4^{3-}	0.05 ± 0.02	0.05 ± 0.02	0.04 ± 0.01	0.08 ± 0.03
S^{2-}	2.82 ± 1.54	6.38 ± 2.86	6.32 ± 1.83	5.44 ± 1.85

DISCUSSION

Nutrient and Organic Matter Decomposition Rate Dynamics

The strongest treatments effects we observed were the decrease of porewater NH_4^+ and PO_4^{3-} concentrations and cotton strip decomposition rates in July due to canopy removal (no-burn experiment) and the corresponding increase of these values due to canopy replacement (burn experiment). These effects appear to have been driven by increased nutrient uptake associated with increased above and belowground biomass production, which was presented in a companion study (Bickford et al., 2012). Aboveground plant nutrient stocks were substantially greater in these treatments, a response driven by greater biomass. Nutrient uptake explains why nutrient effects were not observed early in the growing season (May) as plant uptake rates were likely greater later in the growing season (July). An alternative explanation for these porewater nutrient effects is that denitrification rates may have been increased by the higher temperatures in sites without a canopy. Higher soil temperatures were observed at sites without a canopy; however, these temperature differences were strongest from mid-April through mid-June and were below 0.6°C later in the growing season, once the plant canopy had been reestablished (Bickford, 2011). If this denitrification mechanism were responsible for the nutrient changes, we should have seen the lower NH_4^+ concentrations in May, when temperature gradients between the sites with and without canopies were greater, rather than in July, when the temperature gradients were less pronounced. Also, enhanced microbial activity would not explain the higher P values observed in July plots with a canopy.

The lowered NH_4^+ and PO_4^{3-} concentrations in plots without a canopy may explain the correlated reduction in cotton strip decomposition rates. Plant litter decomposition in nutrient-poor wetland ecosystems is generally slower than in nutrient-rich wetlands (Webster and Benfield, 1986), and higher availability of inorganic nutrients such as NH_4^+ and PO_4^{3-} to microbial decomposers can increase wetland plant biomass decomposition (Brinson et al., 1981). Meyer and Johnson (1983) found a similar effect in that leaf litter decomposed 2.8 times more rapidly in a N-enriched stream with NH_4^+ levels of 0.006 mg NH_4^+–N L^{-1} compared to an undisturbed stream with 0.002 mg NH_4^+–N L^{-1}. The authors attributed this to higher NO_3^- levels as well, which contributed to higher microbial biomass in the N-enriched streams. However, these NH_4^+ levels are substantially lower than those observed in our study.

The mechanism of nutrient depletion by plants suppressing decomposition is further supported when the data were analyzed separately for sedge-dominated and grass-dominated sites in the no-burn experiment. Sedge-dominated sites with canopy removal had significantly lower decomposition rates ($P = 0.01$) from 0 to 20 cm in July and also had lower NH_4^+ ($P = 0.006$) and lower PO_4^{3-} ($P = 0.058$) concentrations in the porewater; while no nutrient or decomposition effects were observed in the grass-dominated sites. These trends match the biomass response of these sites, with sedge-dominated sites showing substantially greater above and belowground biomass in canopy removal sites and grass-dominated sites showing no biomass response (Bickford et al., 2012). Depthwise analysis of the sedge-dominated sites showed that the decreased decomposition rates from 0 to 20 cm was due in large part from the surficial 0- to 4-cm zone, where environmental variables are more subject to dynamic change such as wetting/drying cycles and daily temperature fluctuations.

One potential mechanism that was considered was decreased rhizosphere O inputs due to the removal of aboveground plant tissue. By removing this active O input by vegetation shoots into the root zone, lower aerobic activity associated with

Table 7. Annual burn sites total plant tissue elemental content with treatments and standard errors ($n = 36$) at the Blackwater National Wildlife Refuge in Dorchester County, MD. ANOVA indicated each pair of plant tissue elemental content measurements was significantly different ($\alpha = 0.05$).

Element	Treatment: Control	Treatment: Canopy replacement
	g m^{-2}	
C	229.49 ± 21.80	146.13 ± 24.66
N	4.46 ± 0.36	2.92 ± 0.35
P	0.44 ± 0.05	0.29 ± 0.06
K	6.04 ± 0.42	3.53 ± 0.64
Ca	0.80 ± 0.07	0.53 ± 0.09
Mg	1.35 ± 0.12	0.87 ± 0.17
S	4.48 ± 0.63	3.19 ± 0.60

Table 6. No-burn sites total plant tissue elemental content with treatments ($n = 96$) at the Blackwater National Wildlife Refuge in Dorchester County, MD. The canopy removal main effect was statistically significant for all nutrients except Ca ($\alpha = 0.05$).

Element	Treatment: Ash deposition	Canopy removal	Canopy removal + ash	Control
	g m^{-2}			
C	148.72 ± 15.01	202.89 ± 28.58	184.69 ± 16.95	143.73 ± 33.96
N	3.96 ± 0.44	4.49 ± 0.54	4.66 ± 0.49	3.59 ± 0.82
P	0.40 ± 0.03	0.45 ± 0.05	0.48 ± 0.04	0.40 ± 0.09
K	4.86 ± 0.53	5.29 ± 0.75	5.34 ± 0.83	4.08 ± 1.18
Ca	0.82 ± 0.09	1.06 ± 0.12	0.99 ± 0.10	0.78 ± 0.17
Mg	1.29 ± 0.15	1.59 ± 0.14	1.49 ± 0.16	1.37 ± 0.28
S	3.63 ± 0.63	4.30 ± 0.77	3.79 ± 0.66	2.96 ± 0.82

Table 8. Elemental standing stocks of senesced vegetation, applied ash constituents, and percent volatilization for no-burn experiment sites with standard errors ($n = 96$) at the Blackwater National Wildlife Refuge in Dorchester County, MD.

Element	Pre-burn biomass constituents	Percent volatilized	Ash constituents
	g m^{-2}		g m^{-2}
C	137.05 ± 11.73	93.87 ± 1.43	6.12 ± 0.67
N	3.46 ± 0.37	90.57 ± 2.14	0.22 ± 0.02
P	0.34 ± 0.03	50.55 ± 6.70	0.16 ± 0.02
K	5.13 ± 0.81	6.35 ± 4.51	4.96 ± 0.05
Ca	0.75 ± 0.08	4.67 ± 3.86	0.71 ± 0.06
Mg	1.20 ± 0.12	6.80 ± 7.36	1.10 ± 0.06
S	4.06 ± 0.60	84.82 ± 4.10	0.31 ± 0.03

Table 9. Nutrient additions and resulting plant responses reported in the literature for tidal marshes.

Study	N added	P added	Increments	Yearly N addition	Yearly P addition	Findings
	—— $g\ m^{-2}$ ——			—— $g\ m^{-2}\ yr^{-1}$ ——		
Crain, 2007	54.3	36.7	Monthly, May–July 2001	162.9	110.1	*S. patens* 50% biomass increase (N only), 75% biomass increase (N + P)
DeLaune et al., 2005	10	8.7	Single, 3 mo greenhouse experiment	10	8.7	*S. patens* doubled in biomass production
Gusewell et al., 2003	0.95	0.06	Weekly, 15 wk in 1998	14.25	0.9	Small increase in aboveground biomass of all 15 plant species examined
Alberti et al., 2010	8.7	1.5	Monthly, Jan. 2006–June 2007	52.2	9.0	*S. densiflora* biomass 4.5 times increase (marsh edge), 6.5 times increase (low marsh)
Daleo et al., 2008	4.35	0.75	Biweekly, Dec.–Apr. 1993, 1994	34.8	6.0	*S. alterniflora* aboveground biomass increase of 5.3 times, doubling of stem density, stem height increase of 1.65 times
Levine et al., 1998	4.35	0.45	Biweekly, Apr.–Sept. 1993, 1994	43.5	4.50	Doubling of *S. alterniflora*, *S. patens*, and *D. spicata* biomass
Emery et al., 2001	4.35	0.45	Biweekly, mid-May–end Aug. 1997, 1998	30.45	3.15	Doubling of *S. alterniflora*, *S. patens*, and *D. spicata* biomass
Gratton and Denno, 2003	60	N/A	Biweekly, 4 wk, mid-May start	240	N/A	3-fold increase in *S. alterniflora* biomass, 1.5–1.8% increase in plant tissue N content
Gratton and Denno, 2003	120	N/A	Biweekly, 8 wk, mid-May start	960	N/A	2.5-fold increase in *S. alterniflora* biomass, 1.4% increase in plant tissue N content
Van Wijnen and Bakker, 1999	5 (low) and 25 (high)	2 (low) and 10 (high)	Early May 1994, 1995, 1996	5 (low) |and 25 (high)	2 (low) and 10 (high)	Low N application increased plant growth at some sites while high N increased plant growth at all sites. P application appeared negligible.
Mendelssohn, 1979	56	N/A	Twice, June and July 1976	112	N/A	Doubling of plant biomass
Buresh et al. 1980	20	20	Single, May 1976	20	20	N additions significantly increased total aboveground plant biomass and plant height. No effect of P additions.

the oxidized rhizosphere could decrease NO_3^- reduction and soluble reactive P production. This explanation may have some validity if we observed our results directly following burning. However, by the time we observed decreases in decomposition rates in sites with the canopy removed, it was July, and there was actually significantly more biomass and shoot growth in the plots with canopy treatments; therefore, there was likely an increase in O diffusion. This mechanism should have resulted in greater decomposition in sites with a canopy removal treatment as they would have had more biomass and shoot growth for O transport; however, we observed the opposite trend.

Lack of Ash Deposition Fertilization Effect

Nutrient additions through ash deposition have been widely suggested as a cause for the increase in plant productivity that is frequently observed in marshes following prescribed burns (Faulkner and de la Cruz, 1982; Wilbur and Christensen, 1983; Nyman and Chabreck, 1995; Cahoon et al., 2004). Several researchers have found that nutrient levels in marsh soils increase soon after a burn (Faulkner and de la Cruz, 1982; Schmalzer and Hinkle, 1992; Smith et al., 2001), and fertilization studies have shown that tidal marsh plant production is often limited by nutrient supply (Kiehl et al., 1997). However, until now, plant responses to nutrient additions through ash deposition had not formally been separated from the simultaneous effects of canopy removal. The evidence presented here suggests that the positive plant responses observed following fire were not due to a nutrient subsidy.

We conducted a review of the marsh fertilization literature to provide context for our results. We focused on N and P because salt marshes are generally N-limited ecosystems (Valiela and Teal, 1974), but may exhibit secondary P limitation (Crain, 2007). Our sites were likely N-limited because the N/P ratios of both the grass and sedge species analyzed across both of these studies were all less than 12, which is lower than the threshold of 14 that has been shown to indicate N limitation in wetland systems (Koerselman and Meuleman, 1996; Verhoeven et al., 1996).

Our literature review showed that while nutrient additions frequently generate increased plant production, the minimum rate that has been applied experimentally is 5 g N m^{-2} yr^{-1} (Table 9). At this 5 g N m^{-2} yr^{-1} rate, plants did not consistently show a response, especially in older marsh areas that were high in natural N reserves and organic matter (Van Wijnen and Bakker, 1999). In our study, 0.22 g N m^{-2} and 0.16 g P m^{-2} were added in the ash to our ash deposition treatment. Field experimentation is necessary to determine if such low doses can generate a plant stimulation response, particularly when applied during the late winter or early spring, as is common for prescribed burn management in the mid-Atlantic region.

We observed a 91% loss of N in our study, which would equate to a burn temperature of 513°C using the Qian et al. (2009) model. Our field measurements of burn temperatures exceeded the ~320°C maximum of our IR temperature gun. Our method of outdoor combustion in small bins likely produced higher temperatures than natural ash production during fires due to a concentration of the fuel load. We applied the Qian et al. (2009) model using our initial biomass N contents, data from a Crain (2007) study in *S. patens* marshes in Maine, and a Gratton and Denno (2003) study in *S. alterniflora* marshes in New Jersey

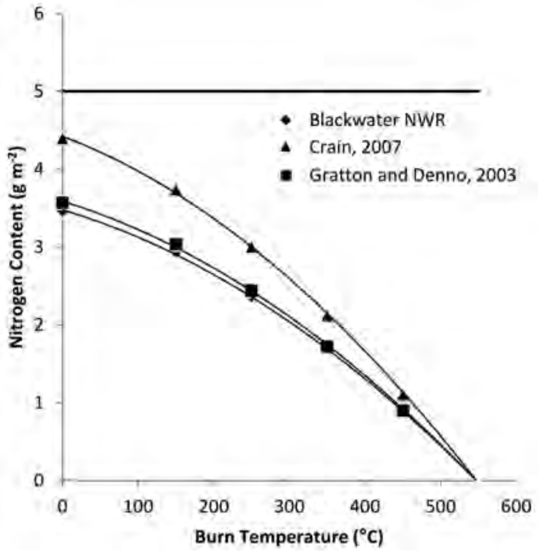

Fig. 8. Estimated N content of ash a function of burn temperature and initial plant N content (values from our study and from two published studies) using the model $y = 0.01(x^{1.46})$ where y = % N loss and x = burn temperature (°C) ($r^2 = 0.83$) developed by Qian et al. 2009. The horizontal line represents the minimum N application rate that has been used in the literature to estimate plant response to fertilization (Table 9).

(Fig. 8). Based on this analysis, standing plant nutrient stocks do not retain enough N following prescribed burns to provide N additions above the 5 g N m^{-2} level that is the minimum rate that has been used in field fertilization studies (Table 9), even at moderate burn temperatures.

Our observed ratio of ashTP/leafTP of 0.47 represented an unreasonably low burn temperature of approximately 171°C, which would have been detected by our temperature gun and, therefore, did not agree well with this model. However Qian et al. (2009) noted that the relationship between ashTP/leafTP may hold true for only laboratory settings, as P is lost primarily through the fine particulate pathway, with the volatilization pathway as a relatively minor contribution (Raison et al., 1985). Nitrogen is lost primarily through the volatilization pathway and, therefore, particulate loss did not present an additional pathway to take into account for our ash samples. The likely explanation for our low ashTP/leafTP ratio is that during the process of making ash in steel bins, convective currents were present and likely removed a significant amount of fine particulate P from our collected ash samples.

Further evidence for the absence of a significant ash effect is the timing of burning. Prescribed burning in this region is conducted during the winter months and is typically completed before the beginning of the growing season. During this time, deposited nutrients can be removed through tidal flushing (de la Cruz, 1973) and biogeochemical processes such as denitrification.

Implications

Prescribed fire does not likely provide a fertilization effect for vegetation in these tidal marsh systems. The depressed NH_4^+ and PO_4^{3-} concentrations late in the growing season we observed were likely caused by increased plant biomass uptake in response to canopy removal and probably caused the depression of decomposition rates in the rooting zone from 0 to 20 cm by decreasing available nutrients for microbial growth and activity. This decrease in decomposition from 0 to 20 cm due to lower available nutrients for microbial activity may have positive effects on marsh elevation trajectories if the results of labile organic matter decomposition from the cotton strips are representative of longer-term decomposition rates. Cahoon et al. (2010) found that annually burned marshes at The Blackwater National Wildlife Refuge had lower root zone subsidence than unburned areas (-0.4 ± 1.2 mm yr^{-1} vs. -6.2 ± 1.0 mm yr^{-1}). This difference is probably associated primarily with increases in belowground root and rhizome production, but our data suggest it may also be influenced by nutrient-related decreases in organic matter decomposition (Mendelssohn et al., 1999).

CONCLUSIONS

We found in a manipulative study that prescribed fire in tidal marshes did not provide a fertilization effect for vegetation through ash deposition, likely because the N content of the ash following combustion was low and the ash was deposited during the winter, which was months before the peak plant nutrient demand. Modeling the biomass nutrient stocks in other marshes with similar vegetation types appears to show that this lack of a fertilization effect may exist across marsh types, particularly for late-winter/early-spring ash depositions. Through the mechanism of canopy removal, organic matter decomposition rates in marsh areas tended to decrease in July but not in May. This decrease in decomposition rates corresponded to a decrease in porewater NH_4^+ and PO_4^{3-}, which were taken up in higher quantities with increased biomass production. Lower available nutrients later in the growing season likely caused resource stress for microbial decomposers, thereby lowering decomposition rates. These effects tended to be stronger and more consistent in areas dominated by the sedge *S. americanus*, which showed more of a biomass response to canopy removal. Future research efforts to understand the fertilization effects of burning should use lower rates of nutrient additions than is common in the literature to better understand the effects of low rates of fertilization.

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