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# The Effect of Biochar on Marsh Phosphorus Retention: Implications for P Biogeochemical Ecosystem Services from Prescribed Burns

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## Abstract

Mid-Atlantic tidal salt marshes provide a number of ecosystem services including the storage of phosphorus (P) within their soils, which reduces nutrient export to estuaries. The storage of P in marsh soils is vegetation-specific, and management efforts targeting the removal of the invasive common reed, *Phragmites australis* may reduce the P ecosystem service. The use of prescribed burns to remove *P. australis* introduces biochar into marsh soils, which has been shown to sorb excess P in agricultural and wastewater systems. Biochar additions with burning may thus recoup P storage services lost via *P. australis* replacement. Organic and inorganic total P content and relative P mobility (determined by sequential extraction) were measured along with a suite of physical and biogeochemical parameters in field collected soil cores from a marsh that had been recently burned and two comparator marshes. Neither burn frequency nor quantities of black carbon, a biochar proxy, were positively associated with P content in any of the P pools indicating that burns are not enhancing P storage in these marshes. Biochar added to experimental plots similarly showed no P storage benefit over control plots. Instead, environmental (e.g., hydrological, vegetation, soil pH, metal content) factors dominated P concentration variations in this work. The lack of an added P storage observed in these data is indicative that biochar properties vary depending on their production conditions, and low temperature (<400 °C) prescribed burns of *P. australis* are unlikely to yield biochars with pH and cation exchange properties favorable for the enhanced P sorption.

**Keywords** Salt marsh · Phosphorus · Biochar · Nutrient retention · Phragmites australis

## Introduction

Salt marshes are highly productive coastal habitats that are valued for a variety of ecosystem services like providing faunal habitat, limiting erosion by flood abatement and wave attenuation, reducing pollution and sediment loads, and providing recreational opportunities (Barbier et al., 2011; Cahoon et al., 2010; Fischer et al., 2000; Leonard et al., 2002; Reboreda & Caçador, 2007; Teuchies et al., 2013). Salt marshes also provide biogeochemical benefits by storing carbon (C), nitrogen (N), and phosphorus (P) in soils, which reduces atmospheric carbon dioxide ($CO_2$) and nutrient pollution into estuaries (Giblin et al., 2013; Henton et al., 2013; Kutcher et al., 2018; Wigand et al., 2017; Yao et al., 2018). Because P, the focus of this study, is an important part of biochemical compounds (e.g. DNA, phospholipids), excess amounts can lead to water quality problems. Watersheds with anthropogenic activities like urbanization, deforestation, and agriculture, have excess P input which could lead to eutrophication upon delivery to coastal waterways where P can be seasonally limiting (Fisher et al., 1992; Schafer & Mack, 2018; Sharp et al., 2009; Volk et al., 2012). This can lead to negative downstream impacts such as hypoxia and harmful algal blooms (Paerl et al., 1998; Sharp et al., 2009). Further, nutrient enrichment in marshes has been associated with decreases in belowground biomass production and increases in microbial decomposition yielding destabilized peat and decreased accretion rates (Darby & Turner, 2008;

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Kutcher et al., 2018), both of which can contribute to marsh loss. It is estimated that salt marshes bury 0.8–1.3 g P $m^{-2}$ $yr^{-1}$ across the United States, and in the Mid-Atlantic region specifically, a New Jersey marsh was found to accumulate 0.13–2.6 g P $m^{-2}$ $yr^{-1}$ (Craft, 2007; Velinsky et al., 2017).

Disturbances like the invasion of *Phragmites australis* alter marshes' ability to provide ecosystem services like P storage. *P. australis* grows in tall, dense, monotypic stands that often outcompete native grasses, reduce habitat quality, and alter soil conditions (Ruggeri, 2014; Sun et al., 2006). Notably, *P. australis* provides some benefits compared to the native salt marsh grasses including greater sediment trapping to support higher accretion rates, and important to this study, *P. australis* uptakes and stores more $CO_2$ and nutrients (N, P) due to their large aboveground biomass, rapid growth, and extensive root systems (Kutcher et al., 2018; Meyerson et al., 2009; Ruggeri, 2014; Windham & Meyerson, 2003), which enhance the P storage capacity of marshes.

In Mid-Atlantic United States marshes, prescribed burning is employed as a management mechanism to remove *P. australis* and promote the growth of native marsh grasses that provide better habitat and more nutritious food sources for marsh fauna (Cahoon et al., 2010; Geatz et al., 2013; Henton et al., 2013; Leonard et al., 2010; Sun et al., 2006; Taillie & Moorman, 2019). This prescribed burning introduces biochar, which has been known to effectively sorb P (Bolton et al., 2019; Faridullah et al., 2012; Zhang et al., 2019) and may consequently enhance P storage in marsh soils thereby mitigating or negating the reduction of sequestration services expected to accompany the replacement of *P. australis* with native vegetation. Biochar, which is an artifact of the incomplete combustion of biomass such as *P. australis*, has a variable composition but is typically composed of condensed aromatic carbon compounds with high porosity and surface acidic functional groups that give it a high sorption potential (Zhang et al., 2019). Lab-created biochars created from a range of starting materials and produced over a range of temperatures have been shown to adsorb excess P in agricultural (poultry litter biochar, 600 °C, Faridullah et al., 2012), wastewater treatment (hemp biochar, 400–425 °C, Bolton et al., 2019), and marine sediment (peach tree wood biochar, unreported temperature; Zhang et al., 2019) experiments, and increased soil P has been observed after the controlled burning of agricultural fields (finger millet, ground nuts, cassava, unreported temperature; Strømgaard, 1992).

While biochar has been shown to sorb P in previous work, it is unclear whether biochar from prescribed burns and/or the influence of environmental factors within salt marshes will yield the same results. Biochar molecular composition and properties (e.g., pH, cation exchange capacity, surface area) are influenced by pyrolysis conditions like temperature, feedstock material, metal enrichment, and chemical activation (Dugdug et al., 2018; Ghodszad et al., 2021; Jassal et al., 2015; Keiluweit et al., 2010; Oliveira et al., 2017; Tesfaye et al., 2021; Wozniak et al., 2020; Zhang et al., 2017). Since temperature is unable to be controlled during prescribed burning, biochar formed by prescribed burning will have inherently different properties than lab-created biochar (which can include chemically activated biochar; Bolton et al., 2019; Faridullah et al., 2012; Zhang et al., 2019), that could influence the biochar's P sorption capacity. Because P sorption depends on soil properties like pH and cation content (Devau et al., 2009; Daly et al., 2015), the ability of a biochar to sorb P very likely also depends on its production conditions (temperature, feedstock material).

Salt marshes also experience environmental factors like tidal exchange as well as variations in salinity and soil pH that differentiate them from terrestrial systems where the interactions of biochar and P have been more extensively studied. P is often particle-bound, but greater porewater salinity reduces the affinity of P adsorption to soil, allowing it to instead be released to porewaters (Sundareswar & Morris, 1999; Tobias & Neubauer, 2019; Upreti et al., 2015; Zhang & Huang, 2011), due to the decoupling of iron (Fe)-P associations in favor of Fe-sulfur (S) associations (Upreti et al., 2015). Phosphorus and metal reactions can also be pH dependent; at low pH ($<6.5$), hydrogen sulfide is produced, and S outcompetes P to bind with soluble Fe(II), releasing or preventing P from precipitation in soils (Kim et al., 2016; Paludan & Morris, 1999; Reimold & Daiber, 1970; Rozan et al., 2002; Sundareshwar & Morris, 1999; Upreti et al., 2015). Once released from soil to the porewaters, this soluble reactive phosphate is available for biological uptake or export from the marsh. P can still be soluble at higher pH in the absence of cations ($>8$; Fox et al., 2014; Gao & DeLuca, 2020), but particulate soil P concentrations tend to be enhanced because precipitation reactions with aluminum (Al), calcium (Ca), magnesium (Mg), and/or Fe become favorable due to the absence of competition from sulfide (Bolton et al., 2019; Rozan et al., 2002; Shen et al., 2016).

The export of P, particularly mobile P, into estuarine systems is important to understand so the effects of alterations to biogeochemical processes can be predicted. Thus, understanding the role that biochar produced during prescribed burning events plays in P cycling is important for assessing the impacts of invasive vegetation management. The dominant P form is negatively charged orthophosphate which can bind with available metals or be incorporated in organic matter. While organic P (OP) is presumed to be unavailable for plant uptake, inorganic P (IP) includes a spectrum of mobility from soluble, plant-available P that can contribute to coastal algal blooms, to stable mineral and Ca-associated P (Tobias & Neubauer, 2019; Zhang et al., 2019). While specific P forms have varying environmental interactions, functionally separating them into groups based on their mobility allows us to evaluate whether the presence of biochar shifts the distribution of P to less mobile forms and could in turn effect the stability of soil P in the marsh environment.

To evaluate the effects of biochar from prescribed burning to remove *P. australis* on P storage in salt marshes, the current study uses three years of field collection data in marshes with and without histories of prescribed burning to evaluate the influences of biochar on soil P concentrations and relative mobility within the context of environmental interactions. The interaction between biochar and P is also isolated from environmental variation using sorption experiments including biochar amended field manipulation plots and a laboratory sorption experiment. Because the high porosity, positive surface charge, and acidic functional groups characteristic of biochar are thought to promote P sorption, it is hypothesized that sorption and retention would be enhanced in soils with (a) a history of prescribed burning and (b) biochar sediment additions. Further, we expect greater P concentration ([P]), especially less mobile forms of P, to be positively correlated with black C concentration, a proxy for biochar input. Analysis of bulk organic and inorganic P concentrations (hereafter bulk P fractions) as well as P mobility by sequential extraction (hereafter mobility fractions) will improve our understanding of P cycling between marshes and adjacent coastal waters. The results of this study aim to provide salt marsh managers with information that will aid decisions on the use of prescribed burns, particularly since post-burn comprehensive nutrient assessments are not frequently conducted.

## Methods

Two experimental strategies were used to evaluate the influence of environmental variation distinct to salt marsh environments on biochar influenced P sorption. Field samples were collected from marshes of varying burn history to capture environmental variation within marsh sites (and thus different elevation and hydrological influence) and between marsh sites (to include variations in salinity, vegetation, external watershed inputs). This gives context to the relative strength of environmental and biochar influence on P sorption in our analyses. A second approach was to conduct a biochar sorption experiment in the field. Variability in environmental parameters is removed to clearly examine any effect of biochar on soil P storage, but the biochar amended soil is still exposed to the marsh environment, which is novel to the existing studies. Finally, a simple isolated laboratory experiment was conducted to compare the P-sorption of the study biochar to the existing research.

### Field Collections from Marshes of Different Burn Histories

Soil samples were collected from three marshes with varying histories of burning (hereafter referred to as the 'field collected samples'). A total of 62 soil cores were collected from three marshes over three sampling seasons (January 2022, November 2022, November 2023; Fig. 1). Roberts Farm (RF; Townsend, DE; $n=23$) has a recent burn history with several burns between 2017 and 2021. Rocks Tract (RT; Townsend, DE; $n=20$) has not been burned in at least 20 years (Eric Ludwig, DNREC Delaware Fish and Wildlife, personal communication), but it is located adjacent to Roberts Farm and may have hydrological or atmospheric

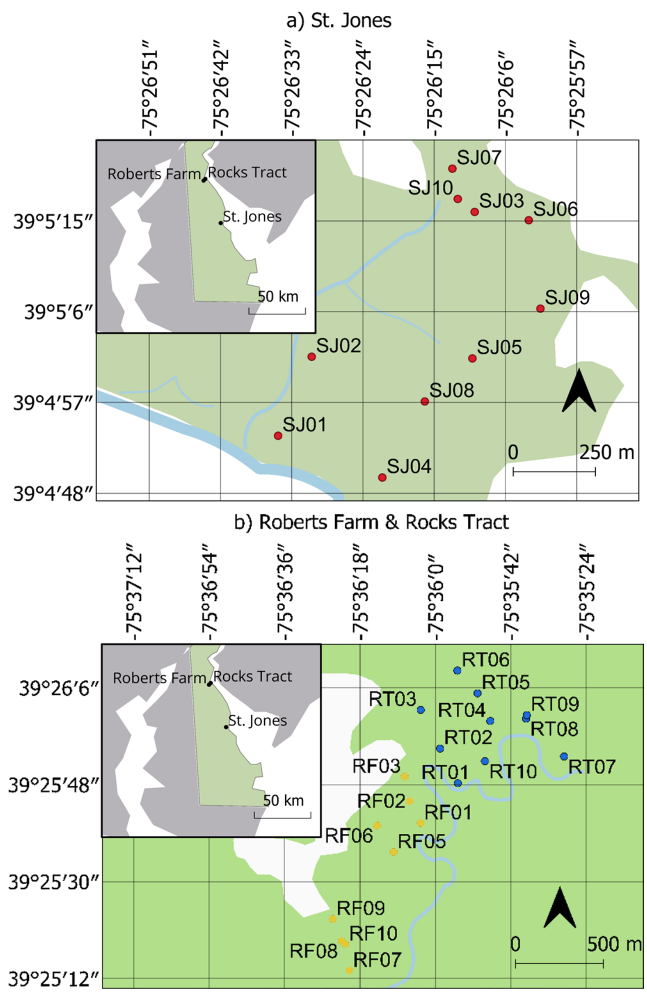


**Fig. 1** Locations of core collection sites in the (**a**) St. Jones (SJ) and (**b**) Roberts Farm (RF) and Rocks Tract (RT) study marshes. Inset shows the marsh locations along the western coast of Delaware Bay. Made with QGIS

transport/deposition of biochar from the more recent prescribed burns in that marsh. The marsh at St. Jones Reserve, part of the Delaware National Estuarine Research Reserve (SJ; Dover, DE; $n=19$) is directly adjacent to Dover Air Force Base and has never been burned due to its use as a research site as well as potential effects on power lines over the marsh. SJ is about 25 mi south of RF and RT and should be unaffected by prescribed burn deposition from RF and RT. SJ has an annual average salinity of $10.8 \pm 7.0$ ppt while the average salinity in both RF and RT is $2.0 \pm 2.1$ ppt (Kelly, 2023 using https://cdmo.baruch.sc.edu//). *Spartina alterniflora* and *P. australis* dominate SJ, while RF and RT contain *S. alterniflora*, *Spartina patens*, *Spartina cynosuroides*, *P. australis*, and some *Typha spp.* SJ is thus a useful comparison to evaluate the importance of site-specific conditions and give context to any influence of biochar on soil P storage relative to the influence of environmental factors.

Soil samples ($n=10$ per site) were collected and processed similarly to Dunne et al. (2010) using 10.5 cm diameter push cores. Three transects running perpendicularly to the main tidal creek toward the upland-marsh ecotone were established to capture a range of creek influences and soil elevations. Three cores were collected along each transect, and a 10th core was collected within a *P. australis* stand at each site. Whole sealed cores were frozen upon returning to the lab, then were later thawed and sectioned to 5 cm increments in a glove bag of $N_2$ gas to preserve redox-sensitive analytes for analyses not reported in this study. These sections were re-frozen, freeze dried, then sieved at 500 µm to remove large biomass particles. Sieved, homogenized samples were placed in glass jars, topped with $N_2$ gas, and frozen until analysis.

A variety of environmental factors could lead to spatial and temporal variations in P fractions within and between sites including tidal inundation, salinity, pH, Fe and S interactions, and the microbial and plant cycling of P (Rozan et al., 2002; Sundareswar & Morris, 1999; Tobias & Neubauer, 2019; Upreti et al., 2015). We monitored several physical and biogeochemical marsh soil properties to account for some of these potential environmental influences in the field study. Organic carbon (OC) and total nitrogen (TN) were measured using an adaptation of Menzel and Vaccaro (1964) and Ehrhardt and Koeve (1999) by CHNS elemental analysis using a COSTECH ECS 4010. Since OC concentrations are positively correlated with plant biomass (Zhang et al., 2015), OC and TN concentrations were used as biomass proxies and were tested for correlation with [P]. Black C (hereafter referred to BCCTO to identify the operational measurement used to quantify black carbon) is a quantifiable proxy for biochar input and was determined by Kelly (2023) using the CTO-375 method in which acidified, dry samples are combusted in the presence of oxygen at 375 °C for 24 hours to remove non-pyrogenic carbon before elemental analysis (Gustafsson et al., 2001).

Elevation and position of sampling times within diurnal and spring tidal cycles were used to describe tidal inundation. Logistical challenges (e.g., hunting/trapping periods, the need to sample during daylight hours, availability of personnel, number of sites and cores) precluded the sampling of marsh soils at consistent positions within diurnal and neap-spring tide cycles. The position of sampling time within tide cycles was assessed as the number of days between sampling and the last spring tide (spring tides) and the number of hours between sampling and the last high tide (diurnal tides). There was no correlation between sampling time within tide cycles and [IP] or [OP] (Kendall's tau coefficient test: $\tau$ = -0.07–0.08, $p=0.13$–0.78; Fig S5) Elevation data from LiDAR were provided by Daniel Warner (Delaware Geological Survey; McKenna et al., 2018) and used as a $3 \times 3$ m average plotted in QGIS. Dry soil pH was measured in a 1:5 soil: water mixture (5 g:25 mL) using a handheld pH meter (Hanna Instruments, I991003; Faria et al., 2023; Faridullah et al., 2012).

## Biochar Sorption Experiments

Field and lab biochar sorption experiments were conducted to reduce the impact of environmental variability on the interaction of biochar and P and to highlight differences in P storage between native and invasive marsh grass species. For the field amendment experiment, five $1 \times 1$ m experimental plots were installed at St. Jones Reserve in June 2022. Untreated wood frames were placed in native *S. alterniflora* and invasive *P. australis* stands, and vegetation was trimmed to the soil base (<5 cm height) to help simulate burn conditions and allow for even distribution of sediment applications. There were two plots (one *S. alterniflora*, one *P. australis*) amended with estuarine sediment (~8 cm thick) and two plots (one *S. alterniflora*, one *P. australis*) amended with a sediment and biochar mixture (9 sediment: 1 biochar by volume, ~8 cm thick; Raposa et al., 2023). The biochar used for these experiments is a custom biochar from *P. australis* biomass made by Sustainable Material Solutions using a custom-built pyrolyzer. The detailed biochar properties are listed in Table 1. The applied estuarine sediment was collected from a nearby tidal creek and homogenized in a small, manual cement mixer before application. The initial creek sediment and 10% biochar amended mixture were analyzed for bulk and sequential P concentrations to determine if P sorption occurred during the experiment.

Three cores were collected from each plot in July 2023 using a similar procedure as described above, except cores were sectioned in the field to separate the applied sediment from the original sediment, which is now in the root zone.

**Table 1** Physicochemical properties of the *P. australis* biochar as analyzed by Control Laboratories in Watsonville, CA according to the International BioChar Initiative Laboratory Tests for Certification Program

| Property | Value | Method |
|---|---|---|
| H: C | 0.46 | Dry Combustion-ASTM D 4373 |
| pH | 8.43 | 4.11USCC: dil. Rajkovich |
| % Ash | 24.6% | ASTM D-1762-84 |
| Surface Area | 233 $m^2$ $g^{-1}$ | Butane activity surface area correlation |
| [Fe] | 3442 mg $kg^{-1}$ | EPA 3050B/EPA 6010 |
| [P] | 857 mg $kg^{-1}$ | EPA 3050B/EPA 6010 |

The applied sediment remained visibly different in color and texture than the original. Cores were similarly frozen, freeze-dried, sieved, and preserved. An oxidation reduction potential (ORP) meter (Hanna Instruments, I991003) was used to collect in situ porewater pH measurements at each core location in triplicate.

A small-scale laboratory experiment was also conducted to determine if *P. australis* biochar effectively sorbs P without environmental interaction. 0.5 g of dried, homogenized *P. australis* biochar was added to 0, 0.2, 0.5, 1, and 2 mg P $L^{-1}$ solutions of dihydrogen phosphate and shaken for 24 h. Adsorption rates (%) were calculated as $\left(\frac{[P]_{initial}-[P]_{final}}{[P]_{initial}}\right) * 100$ (Dugdug et al., 2018).

## Phosphorus Analysis

To quantify total organic and inorganic P, referred to as bulk OP and IP respectively, homogenized soils were first dried at 105 °C to a constant mass. An aliquot of each sample was ignited at 550 °C for 2 h to oxidize organic material for total P analysis (Aspila et al., 1976). All unignited and ignited aliquots were extracted 16–18 h in 1 M hydrochloric acid on a shaker table at 200 RPM (1:100 *w: v*). Samples were centrifuged for 20 min at ~2,500 RPM to separate the extracts. This technique was conducted on all field collected samples and amendment experiment samples.

Sequential extractions were conducted following Hedley et al. (1982) as adapted by Faridullah et al. (2012) and Gu et al. (2020). Sequential extraction protocols operationally define P fractions based on their mobility (Anton et al., 2021), in this case from immediately available to recalcitrant P with IP and OP quantified in each fraction. Homogenized, dry samples were first extracted by a solution of MilliQ water and artificial seawater (Instant Ocean) to match the salinity of the sampled marsh during sampling time (5 ppt for RF and RT; 10 ppt for SJ). The water extraction removed immediately plant available P which is presumably available for plant uptake. After separation by centrifugation, the soil residue was rinsed with the previous extract solution (MilliQ water for step 1) by shaking for 1 h to reverse any secondary adsorption (Song & Liu, 2015). After centrifuging the rinse, the residue was dried as described above. Next, the soil residue was similarly extracted in sequence using 0.5 M sodium bicarbonate (to isolate the easily exchangeable P), followed by 0.1 M sodium hydroxide (to isolate the intermediately available P), then by 1 M hydrochloric acid (to isolate the recalcitrant P). The extract to soil ratio was increased from 60 (used for the subset of field collected samples) to 100 for the field manipulation plot samples to improve the yield of OP based on trials. Organic P was determined for each sequentially separated P pool using wet chemical oxidation by adding 0.18 M potassium persulfate to the extracts and autoclaving at 121 °C for two hours, the duration needed to maximize P release (Ridal & Moore, 1990). This procedure was conducted for all amendment experiment samples as well as a subset of 27 field collected samples from all three sites and depth increments which were selected to represent a range in $[BC_{CTO}]$.

To quantify phosphorus from each of the experimental isolates, the molybdate blue method was adapted (Aspila et al., 1976; Murphy & Riley, 1962). Calibration standards of 0–20 µM P were created using a potassium dihydrogen phosphate stock. Bulk P samples were diluted by a factor of 100 and sequential P samples were diluted by a factor of 1–2 for water extracts and 30 for the other extracts to ensure that concentrations were in the linear range of the standard curve trend between absorbance values and P concentration (Aspila et al., 1976). A mixed reagent of sulfuric acid, ammonium heptamolybdate tetrahydrate, and potassium antimony tartrate was made daily. The mixed reagent and an ascorbic acid solution were then added to the standards and the samples. The color was developed and ready for analysis on a Horiba Aqualog using a quartz cuvette at a wavelength of 880 nm after 10 min. The absorbance data were converted to mg P $kg^{-1}$ soil. Inorganic P data were derived from measurements made on the unignited/unoxidized samples with organic P calculated as P from the unignited/unoxidized samples subtracted from that of the ignited/oxidized samples. An external sediment P standard was used to verify the P concentrations (CRM BCR-684; Fig. S1). Attempts to measure the raw biochar [P] by the molybdate blue method resulted in falsely high values likely due to high silica content in the biochar (Aspila et al., 1976). Biochar was found to have greater [silica] than its rice husk feedstock (15–20% vs. 1–5%; Karam et al., 2022; Nguyen, 2021), and *P. australis* has been found to accumulate silica in its biomass (Struyf et al., 2005). The silica artifact found for P measurements of the biochar is not thought to have impacted P measurements in the soils. When the raw biochars were ignited, they changed color from black to white suggestive of this silica interference; the same color change was not found upon igniting the soil samples.

### Statistical Analyses

Most data distributions were found to be not normal by Shapiro-Wilk tests; examinations of box, q-q, and histogram plots; and sample sizes are generally small, so non-parametric statistical tests were chosen (Table S2). Kendall's τ correlation coefficients tests give p-values that indicate if linear relationships between studied variables exist. Mann-Whitney tests for two independent samples were used to investigate differences between sampled groups. Kruskal-Wallis and Dunn's post hoc tests evaluated differences between the three sampled marshes. A correlation matrix showing Kendall's τ correlation and p values was created to examine the effects of multicollinearity (Table S3). While p values indicated that correlations exist between some variables, all correlation values are less than 0.5 indicating that these correlations should have a weak impact on model predictions. Linear mixed-effects models describing the influence of the measured environmental parameters on IP and OP were created using the R package *lmerTest* to determine the influence of environmental factors on P concentrations in the field collected samples of all three marshes combined. The fixed effects were environmental variables depth, elevation, [$BC_{CTO}$], [organic C], [N], time since diurnal high and spring tides, and distance to the center of the nearest primary or secondary tidal creek (The primary creek is defined as the St. Jones River at the SJ site and Blackbird Creek at the RF and RT sites. Secondary creeks are defined as an initial branch from the primary creek). The marsh site (SJ, RF, and RT) was a random effect. All combinations of modeled fixed effects were compared by determining Akaike information criterion scores corrected for small sample sizes (AICc) using the dredge function in the *MuMin* package (Barton, 2020). The AICc of the base model (includes all variables), the lowest AICc scores, and the null model are shown in Table S4. If the model with the lowest AICc included environmental variables that were not significant predictors of [P], those variables were removed from the linear mixed-effect model. Values are reported as mean ± SE unless stated otherwise. There was no change in OP or IP concentrations between collection years one and two (January vs. November 2022 collections) for each sampled marsh and all marshes grouped by Wilcoxon Signed-Rank Test for Paired Samples (All marshes: IP: z = 0.541, $p$ = 0.591 and OP: z = 0.110, $p$ = 0.269; Individual marshes: z = 1.51–0.228, $p$ = 0.822 − 0.132). Thus, year one and two data are pooled together when applicable. To account for potential differences in mineral to organic content of marsh sediments between sites, the analysis was repeated with P concentrations normalized to OC ($[P_{(mg/kg)}]/[OC_{(mg/kg)}]$). This normalization did not affect the results (Fig. S2) and is not discussed further.

## Results

### Field Collected Samples from Marshes of Different Burn Histories

Inorganic and organic P concentrations did not vary between sites with an overall mean of 628.3 ± 28.09 mg IP kg-1 and 566.8 ± 23.43 mg OP kg-1 ($n$ = 162; Table S1.) By site, the mean concentrations were 601.9 ± 48.48 mg IP kg-1 and 695.4 ± 40.20 mg OP kg-1 at SJ ($n$ = 57), 745.1 ± 53.57 mg IP kg-1 and 506.3 ± 48.09 mg OP kg-1 at RT ($n$ = 54), and 534.2 ± 37.78 mg IP kg-1 and 487.0 ± 20.57 mg OP kg-1 at RF ($n$ = 51). Comparison of P concentration between the marshes binned by depth by Kruskal-Wallis shows the only differences in P concentration with burn history to be [OP] at the 2.5 cm and 7.5 cm depth (H2, 56 = 15.0, 8.35; $p$ = 0.0005, 0.015; respectively). Dunn's post hoc found greater [OP] at the 2.5 cm depth in SJ than RF and RT ($p$ = 0.01 and $p$ = 0.0001, respectively) and greater [OP] at the 7.5 cm depth at SJ than RT ($p$ = 0.005; Fig. 2)

Correlations between P concentrations and biochar input were evaluated using $BC_{CTO}$ as a proxy. There is no relationship between [OP] and [$BC_{CTO}$] (Kendall's tau coefficient test: τ = 0.014, z = 0.198, $p$ = 0.84), but there is a weak (τ = -0.20) negative relationship between [IP] and [$BC_{CTO}$] (Kendall's tau coefficient test: z= -2.82, $p$ = 0.005). $BC_{CTO}$ concentrations were higher at SJ than RF and RT (Kruskal-Wallis: $H_{2, 161}$ = 18.06, $p$ = 0.0001; Dunn's post hoc SJ > RF, SJ > RT: z = 2.42, 4.23; $p$ = 0.016, 0.00002; Table 2). The high [$BC_{CTO}$] at SJ indicates an abundant, unforeseen, non-prescribed burn source of pyrogenic C to this marsh (e.g., from the more urbanized St. Jones watershed and/or atmospheric deposition from regional or local sources like the nearby Dover Air Force Base). [P] in RF and RT were compared directly and revealed no enhanced P storage in the recently burned RF soils. RF and RT have similar OP distributions (Mann-Whitney Test: $z_{103}$ = 1.62, $p$ = 0.11), but RT has more IP than RF (Mann-Whitney Test: $z_{103}$ = 2.64, $p$ = 0.009).

Correlations between soil $BC_{CTO}$ concentrations and IP and OP mobility fraction concentrations were tested to evaluate the influence of biochar concentration (and, therefore, the influence of burning) on P mobility (Fig. 3). From most to least mobile, mean [IP] and [OP] were 2.01 ± 0.57 mg IP $kg^{-1}$ and 10.53 ± 1.76 mg OP $kg^{-1}$ for immediately available, 121.4 ± 11.66 mg IP $kg^{-1}$ and 82.55 ± 5.95 mg OP $kg^{-1}$ for easily exchangeable, 251.0 ± 26.42 mg IP $kg^{-1}$

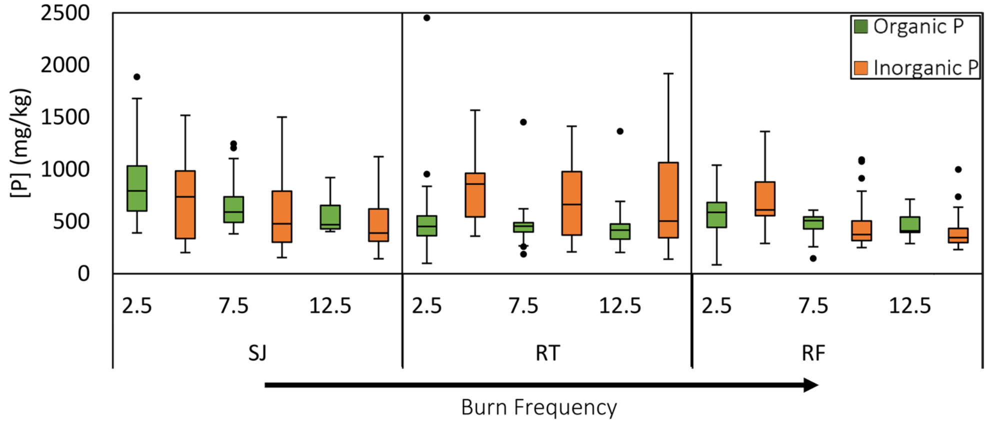


**Fig. 2** Box and whisker plots of organic (green) and inorganic (orange) P concentrations (mg kg$^{-1}$) in the field collected samples from the 3 marshes binned by increments of 5 cm depth below the soil surface. Box borders represent the lower and upper quartiles of concentrations, and horizontal lines within boxes represent the median of all cores from years 1 and 2 ($n$=17–19). Whiskers indicate minimum and maximum [P] excluding outliers, which are represented by black dots

**Table 2** Soil black C concentration (mg $BC_{CTO}$ g$^{-1}$) summary statistics at all marsh sites

| Site Name | Mean | Standard Error | Minimum | Maximum | *n* |
|---|---|---|---|---|---|
| SJ[a] | 1.87 | 0.14 | 0.28 | 4.54 | 57 |
| RT | 1.22 | 0.08 | 0.38 | 3.45 | 54 |
| RF | 1.31 | 0.06 | 0.43 | 2.28 | 51 |
| Total | 1.48 | 0.06 | 0.28 | 4.54 | 162 |

[a][$BC_{CTO}$] is greater at SJ than RF and RT (Kruskal-Wallis: H(2, 161)=18.06, $p$=0.00012; Dunn's post hoc SJ>RF, SJ>RT: z=2.42, 4.23; $p$=0.016, 0.00002)

and 128.0±8.29 mg OP kg$^{-1}$ for intermediately available, and 337.4±23.95 mg IP kg$^{-1}$ and 35.02±4.41 mg OP kg$^{-1}$ for recalcitrant P pools (Table 3). While the relationships between [$BC_{CTO}$] and less mobile [P] appear to be negative, Kendall's correlation coefficients indicate that there is no relationship between $BC_{CTO}$ and any of the P pools (τ = -0.18–0.21, $p$=0.13–0.93).

Since the field collected samples indicate that burning or biochar are not strong influences on P fractions, a suite of additional environmental variables was examined to determine if other factors could be overshadowing an interaction between P and biochar. Results from linear mixed-effects analysis demonstrate that soil depth, [$BC_{CTO}$], [OC], and elevation uniquely contribute to explaining the variation in soil [IP] with a conditional and marginal $R^2$ of 0.35 and 0.30, respectively. Depth, [$BC_{CTO}$], [OC], time since the most recent diurnal high tide (hereafter referred to as high tide), [N], and distance to the nearest creek uniquely contribute to

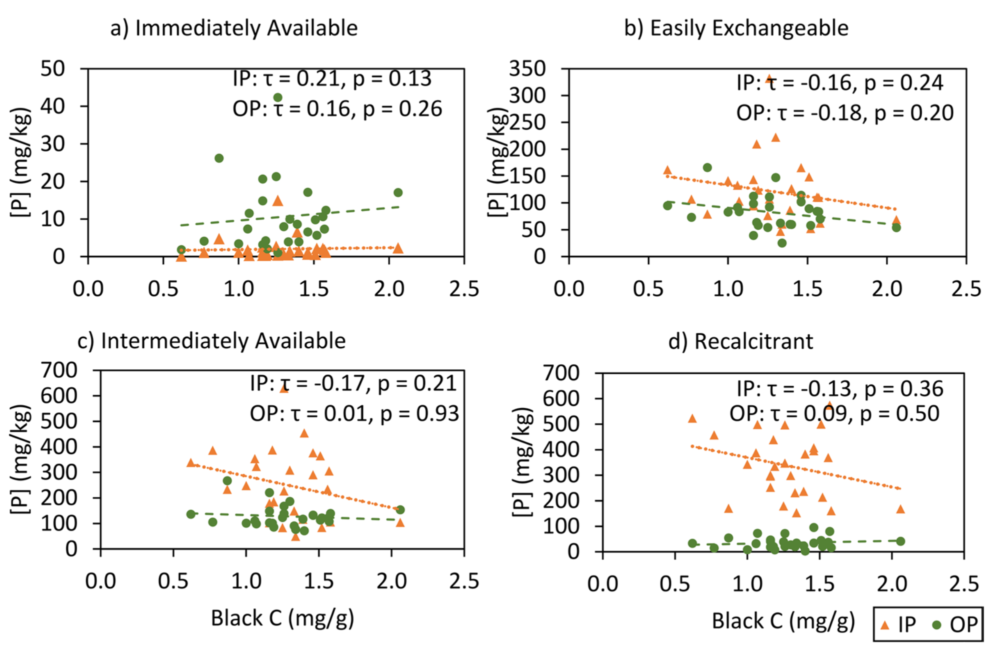


**Fig. 3** Concentrations (mg P kg-1 soil) of organic phosphorus (OP, green) and inorganic phosphorus (IP, orange) in each sequentially extracted P pool plotted versus black CCTO concentration (mg C g-1 soil) for a subset of the year 1 core samples (n=27). P availability and mobility decreases from **a**) Immediately Available to **b**) Easily Exchangeable to **c**) Intermediately Available to **d**) Recalcitrant

**Table 3** Summary statistics for IP and OP concentrations (mg P $kg^{-1}$) in each sequentially extracted soil P fractions at a subset of the marsh sites ($n=27$)

| | | Mean | Standard Error | Minimum | Maximum |
|---|---|---|---|---|---|
| IP (mg/kg) | Immediately Available | 2.01 | 0.57 | 0.15 | 14.94 |
| | Easily Exchangeable | 121.4 | 11.66 | 46.93 | 331.9 |
| | Intermediately Available | 251.0 | 26.42 | 48.93 | 628.9 |
| | Recalcitrant | 337.4 | 23.95 | 152.6 | 574.2 |
| OP (mg/kg) | Immediately Available | 10.53 | 1.76 | 1.02 | 42.34 |
| | Easily Exchangeable | 82.55 | 5.95 | 25.15 | 165.6 |
| | Intermediately Available | 128.0 | 8.29 | 70.76 | 267.2 |
| | Recalcitrant | 35.02 | 4.41 | 2.77 | 94.84 |

[OP] variation with a conditional and marginal $R^2$ of 0.71 and 0.71, respectively. Estimated effect values shown in Table 4 describe the strength of the relationship between the specified variable and its influence on P variability. In sum, the measured environmental variables explain soil [OP] variability well, but [IP] variability is influenced by variables not measured here.

Another important environmental factor to consider is the influence of native vs. invasive grass species on soil P storage since a goal of these prescribed burns is to reduce the invasive vegetation in favor of the native. Mean OP and IP concentrations in native *Spartina spp.* stands were similar to those in invasive *P. australis* (Mann-Whitney Test: $z=0.27$, $p=0.78$; $z=0.67$, $p=0.095$, respectively), with [OP] of 570.2 ± 24.08 mg OP $kg^{-1}$ in *Spartina spp* and 548.8 ± 47.08 mg OP $kg^{-1}$ in *P. australis* and [IP] of 627.2 ± 28.52 mg IP $kg^{-1}$ in *Spartina spp.* and 741.2 ± 84.31 mg IP $kg^{-1}$ in *P. australis.*

## Biochar Sorption Experiments

Samples collected during the amendment experiment setup were used to determine that bulk [IP] and [OP] in the unamended creek sediment measured 456.2 and 410.7 mg P $kg^{-1}$, respectively. There were greater [P] in the creek sediment amended with 10% biochar (v/v; 636.4 mg IP $kg^{-1}$ and 585.7 mg OP $kg^{-1}$) likely due to the biochar which contained 857 mg TP $kg^{-1}$ (Table S5). After 13 months of incubation, [IP] in the amended sediment (top section of the biochar-amended plots) decreased by 250.7 mg IP $kg^{-1}$ $soil_{dry\ weight}$ while [OP] increased by 221.1 mg OP $kg^{-1}$ $soil_{dry\ weight}$. The unamended (top section of creek sediment only plots) soil [IP] decreased by 36.05 mg IP $kg^{-1}$ $soil_{dry\ weight}$ while [OP] increased by 287.1 mg OP $kg^{-1}$ $soil_{dry\ weight}$ (Fig. S4).

At the end of the experiment, Mann-Whitney tests show there was no relationship between bulk P concentrations at unamended vs. amended plots ($z=0.08$–1.46, $p=0.15$–0.93). Average [P] from the plots with biochar amendment was 385.7 ± 32.66 mg IP $kg^{-1}$ and 806.8 ± 56.85 mg OP $kg^{-1}$ in the top of the cores (applied sediment) and 341.1 ± 48.14 mg IP $kg^{-1}$ and 1092.4 ± 98.13 mg OP $kg^{-1}$ at the bottom of the cores (original sediment). Average [P] from the plots without biochar amendment was 420.1 ± 27.74 mg IP $kg^{-1}$ and 697.9 ± 54.55 mg OP $kg^{-1}$ in the top of the cores (applied sediment) and 320.8 ± 21.81 mg IP $kg^{-1}$ and 1249 ± 163.8 mg OP $kg^{-1}$ at the bottom of the cores (original sediment; Table S5). There was also no difference between P mobility fractions in the amended vs. unamended plots except for less immediately available OP in the root zone section of the biochar-amended plots (Mann-Whitney Test: $z=0.21$–1.46, $p=0.15$–0.84 vs. $z=2.80$, $p=0.002$), a short-lived and very small portion of TP (2.6%; Fig. 4). Mean [P] of each fraction is shown in Table S6.

The laboratory biochar sorption experiment revealed the biochar to release P as evidenced by the negative adsorption rates for all [P] (-520 to -78%; Fig. S3) suggesting

**Table 4** Summary of linear regression analysis of the contribution of the measured environmental factors on IP and OP variability. The models best suited to the data are reported with additional models in Table S4

| | Conditional $R^2$ | Marginal $R^2$ | Significant Variables | Estimated Effect | SE | p |
|---|---|---|---|---|---|---|
| IP | 0.348 | 0.302 | Intercept | 840.39 | 114.69 | $3.78 \times 10^{-08}$ |
| | | | $[BC]_{CTO}$ | -116.82 | 39.44 | 0.004 |
| | | | Depth | -15.81 | 6.09 | 0.010 |
| | | | [OC] | -3.06 | 1.03 | 0.004 |
| | | | Elevation | 340.89 | 93.12 | $3.47 \times 10^{-04}$ |
| OP | 0.707 | 0.707 | Intercept | 199.84 | 47.93 | $7.72 \times 10^{-05}$ |
| | | | $[BC]_{CTO}$ | -58.28 | 21.98 | 0.009 |
| | | | Depth | -8.51 | 3.40 | 0.013 |
| | | | [OC] | 7.05 | 0.67 | $2.00 \times 10^{-16}$ |
| | | | High Tide | -10.78 | 4.33 | 0.014 |
| | | | [N] | 18.21 | 5.52 | 0.001 |
| | | | Creek Distance | 0.35 | 0.12 | 0.004 |

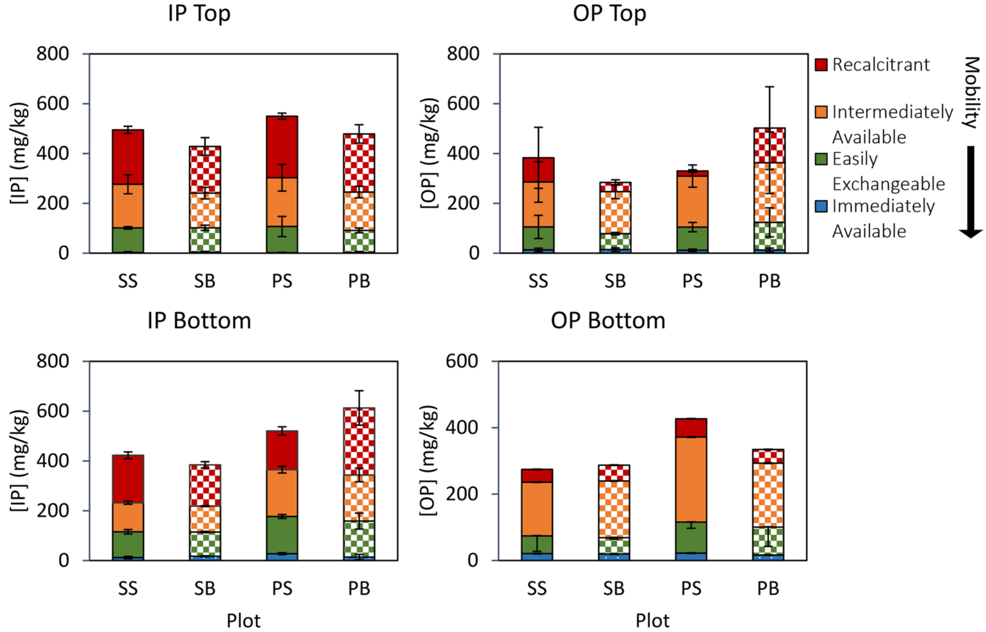


**Fig. 4** Mean concentrations of IP (**a**, **c**) and OP (**b**, **d**) in each sequentially extracted P fraction (indicated by color) in the top (**a**, **b**) and bottom (**c**, **d**) core sections from the amendment experiment plots. Plots are SS (*S. alterniflora* with sediment addition), SB (*S. alterniflora* with biochar-amended sediment addition), PS (*P. australis* with sediment addition), and PB (*P. australis* with biochar-amended sediment addition). Solid bars have creek sediment applied and textured bars have sediment with biochar applied. Error bars represent standard deviation ($n=3$)

much of the P in the biochar to be readily dissolvable into water. There is no evidence that the study biochar, which was designed to mimic a low pyrolysis temperature natural biochar resulting from prescribed burning, is effective at P sorption on the timescale of these experiments. Instead, it may act as a source of mobile and bioavailable P.

An additional finding is that while biochar did not influence P concentrations in the field experimental plots, Mann-Whitney Tests show that grass species did in the bottom section of the cores (root zone/original marsh soil), where there was more bulk IP in the *P. australis* stands than in the native *S. alterniflora* stands ($z=2.80$, $p=0.002$). There was $398.1\pm29.75$ mg IP $kg^{-1}$ and 1357 mg OP $kg^{-1}$ in the *P. australis* plots and $263.8\pm12.04$ mg IP $kg^{-1}$ and $984.5\pm59.12$ mg OP $kg^{-1}$ in the *S. alterniflora* plots (Table S5). More specifically, there was more Easily Exchangeable and Intermediately Available IP and OP in the bottom section from the *P. australis* stands than from *S. alterniflora* stands ($z=2.48$–$2.80$, $p=0.002$–$0.009$; Table S6, Fig. 4).

## Discussion

This study revealed no evidence for enhanced biochar-facilitated P storage in field collected samples from marshes with more recent burning. The expectations of greater [IP], [OP], and portion of recalcitrant P at RF were not observed. Further, positive correlations between [$BC_{CTO}$] and more recalcitrant P pools did not exist (Fig. 4). [$BC_{CTO}$] was negatively correlated with [IP] and [OP] using linear mixed-effect models (Table 4), and Kendall's tau agreed that [$BC_{CTO}$] and [IP] are weakly negatively correlated ($p=0.005$, $\tau=-0.20$) in the field collected samples, but [$BC_{CTO}$] and [OP] were not ($p=0.84$). Relationships between $BC_{CTO}$ and P were hypothesized to be positive given the expectation that biochar contains $BC_{CTO}$ with surface chemical properties allowing it to sorb P and transform it into lower mobility forms. Similarly, the amendment experiment provided no evidence for enhanced P storage in the biochar amended plots as well as no difference in [P] or P mobility in the biochar-amended vs. unamended plots. The OP pool did increase after 13 months of application in the amended sediments suggesting a potential immobilization of IP into the organic pool; however, the same observation was made in the unamended plots suggesting no added P storage benefit from the biochar amendment. Similarly, the lab experiment revealed no benefit in P storage as added biochar resulted in the release of P into solution. This indicates that the biochar in the plots serves as a source of bioavailable and mobile P that could be taken up in biomass or exported from the marsh into the estuary. As described below, the reasons for a lack of a positive P sorption in these experiments likely results from a combination of environmental influences on soil P and biochar properties that differ from those used in other work (Stromgaard et al., 1992; Faridullah et al., 2012; Zhang et al., 2019).

While this study found no evidence for biochar facilitated P storage, mean P concentrations of nearly 1,200 mg P $kg^{-1}$ soil ($628.3\pm28.09$ mg IP $kg^{-1}$ and $566.8\pm23.43$ mg OP $kg^{-1}$) found in the field collected soil samples are higher than values reported in other east coast marshes but within the range of values reported in a global survey of marsh soils (Li et al., 2023). TP concentrations averaged 588–635 mg P $kg^{-1}$ $soil_{dry\ weight}$ in three brackish marshes in Georgia (Loomis & Craft, 2010), 697 mg TP $kg^{-1}$ $soil_{dry\ weight}$ to 921 mg TP $kg^{-1}$ $soil_{dry\ weight}$ in a South Carolina marsh (Sundareshwar & Morris, 1999), and an average of 710 mg TP $kg^{-1}$ $soil_{dry\ weight}$ (up to 3,900 mg TP $kg^{-1}$ $soil_{dry\ weight}$) in a tidal New Jersey marsh affected by urban and agricultural runoff (Velinsky et al., 2017). OP represented about 42–55% of TP in the current study, which is within the wide expected range of 30–90% OP for wetland soils (DeLaune et al., 2013).

Sequentially extracted P fraction reports are rare in wetland literature. Thus, this study provides important data on P mobility in marshes and offers insights into OP contributions to the sequentially extracted pools. While 2–9% of the TP in the field collected soil samples was found to be immediately available, previous work has found this fraction to be 10–20% (Álvarez-Rogel et al., 2007; White et al., 2008). The low values in the current study are similar to the data from Fox et al. (2014) who reported plant available P to be < 1% of TP in agricultural soils. 21% of soil P in the sampled marshes is found to be easily exchangeable, similar to reports for South Carolina marshes (up to 24%; Paludan & Morris, 1999; Tobias & Neubauer, 2019). Recalcitrant P was reported to comprise 13–38% of TP in brackish South Carolina marshes, and our observations fall at the high end of that range (35% recalcitrant P in SJ, 39% recalcitrant P in RF and RT; Paludan & Morris, 1999). Notably, most studies of sequentially extracted P fractions do not measure the OP portion within each pool. OP is typically only measured at the end of the extraction sequence after igniting or digesting the sample residue (Álvarez-Rogel et al., 2007; Ruttenberg, 1992; Zhang et al., 2019). This study detected OP in all sequentially separated P fractions, which suggests that the traditional quantification of OP at the end of the extraction sequence is likely missing more mobile OP. This finding is consistent with the growing recognition of the molecular complexity of soil organic matter with implications for a range of influences and fates (e.g., microbial lability, mineral associations; Kögel-Knabner & Rumpel, 2018; Bahureska et al., 2021). Additionally, the sum of OP fractions was not equal to the bulk [OP] (Fig. S4), indicating that the wet chemical oxidation method of OP determination may not as fully convert OP as the ignition method.

## Importance of Environmental Factors on Phosphorus Fractions

Results of the linear mixed-effect analysis suggest that biomass ([N] and [OC]) and hydrological (time since high tide, distance to creek, elevation) factors exert a greater control on P variability than any effects of biochar and/or may have impacted the interaction between P and biochar (Table 4). $BC_{CTO}$ is identified as an independently significant variable for the IP and OP models, but in each case, $BC_{CTO}$ is negatively correlated with the P parameter suggesting it may negatively impact P retention in marshes.

Relationships between P and [OC] and [N] indicate an influence of vegetation on P fractions (Table 4). [IP] is negatively correlated with [OC] likely due to IP uptake increasing vegetative growth and associated soil [OC]. Similarly, [OP] is positively correlated with [OC] and [N], a somewhat intuitive result given that each are important biomass components that are released during decomposition. Field observations of the study marshes found vegetation near the SJ core sites to be strongly dominated by dense monocultures of *S. alterniflora* with some *P. australis* stands while vegetation at RF and RT was more variable (mixed *S. alterniflora*, *S. cynosuroides*, *S. patens*, and *P. australis)*. The high stem density of *S. alterniflora* (Coleman et al., 2023; Leonard et al., 2002) in SJ could be contributing to the larger concentrations of OM at that site and the associated influence on P concentrations.

[IP] is positively correlated with elevation signifying that a land source like stormwater runoff could deliver P to the study marshes. However, [OP] is negatively correlated with time since diurnal high tides, which may suggest OP delivery or enhanced OP production with the high tide intrusions and/or that OP is decomposed after tides recede. [IP] and [OP] are both negatively correlated with depth, which was also observed by Álvarez-Rogel et al. (2007) and Tobias and Neubauer (2019).

There are additional factors (e.g., watershed P loading, salinity, pH, metals content) not evaluated in the linear mixed-effect models that could be influencing soil P variability in this study, especially [IP] variability given the low $R^2$ values in that model. Differences in allochthonous P sources in the watersheds could influence soil P concentrations between sites. Water quality stations near our sites measured average P concentrations of 0.19 mg/L in St. Jones River that are > 25% higher than those at Blackbird Creek (0.15 mg/L) in 2022 (https://cema.udel.edu/applications/waterquality/). The St. Jones Watershed has greater urban land use compared to the Blackbird Creek watershed that influences the RT and RF sites (33% vs. 3%). That urban land use delivers higher amounts of nutrient pollution due to more impervious landcover (DNREC, 2012, 2022). Additionally, while the Blackbird Creek Watershed currently has more agricultural use than St. Jones (44% vs. 37%), earlier land use reports define 53% of the St. Jones watershed as agricultural, so historical fertilization application could be enhancing soil P and N (DNREC, 2005; Dunne et al., 2010; Kennedy et al., 2017) at SJ compared to RF and RT. Though no differences were observed between sites, enhanced watershed P loading in SJ may balance lesser influences of other factors. Loomis and Craft (2010) and Sundareshwar and Morris (1999) suggest that a reduction in anions in lower salinity marshes can limit competition with P for sediment sorption yielding higher P concentrations in more oligohaline marshes. Salinity differences are also likely not a controlling factor in P fractions between our sites since the lower salinity RF and RT do not have greater P concentrations than SJ (Fig. 2; Table S1).

Dry soil pH was measured for a subset of field collected samples ($n=35$) and yielded no correlation with any P fractions (Kendall's τ: τ = -0.20–0.20, $p=0.16$–0.88) suggesting dry soil pH to not be an important factor describing P variability in these soils. However, the accuracy of field pH measurements diminishes over time (Bartlett & James, 1980; Blake et al., 2000). The expected association of higher pH and less porewater P, and presumably more soil P, was observed in the in situ amendment experiment measurements. However, these plots were adjacent to each other and had a very small range in pH (6.36–6.54). Measurements of in situ soil pH and metal concentrations that interact with P (Fe, Al, Ca, and/or Mg) could elucidate the influence of precipitation reactions on P variability within and between the sites.

## Biochar Properties Influence Potential Phosphorus Sorption

The properties of the biochar used in this study (Table 1) likely help explain why no evidence for enhanced P storage was observed in our burned sites. Biochar properties that influence P sorption capacity like surface area and porosity, ion exchange capacity, surface acidic functional groups, and pH are largely determined by the feedstock material and pyrolysis conditions (Dugdug et al., 2018; Ghodszad et al., 2021; Jassal et al., 2015; Oliveira et al., 2017; Tesfaye et al., 2021; Zhang et al., 2017). Biochars derived from prescribed burning in marshes are produced at relatively low temperatures (<400 °C) due to the high moisture content in tidally inundated marsh soils (Geatz et al., 2013). This study's *P. australis* biochar was intentionally produced at similarly low temperature to mimic biochar produced from prescribed burns. These conditions contrast with previous studies that used higher temperatures (400–600 °C) to yield biochars with increased pH, cations, and surface area that were more effective at P sorption (Eduah et al., 2019; Jassal et al., 2015; Ngatia et al., 2017; Oliveira et al., 2017; Tesfaye et al., 2021; Zhang et al., 2017). For example, Ngatia et al. (2017) found that low temperature biochar (200–500 °C) with a similar pH (8.43) to the study biochar are associated with minimal P sorption or the release of P relative to higher pH and higher temperature biochars (550–750 °C). Prescribed burns and wildfires also tend to have more variable temperature and oxygen availability than laboratory-controlled biochars (Myers-Pigg et al., 2024). The higher oxygen availability during pyrolysis in these open-air burns has been suggested to yield oxygenated biochars with more soluble organic matter components (including C and N; Myers-Pigg et al., 2024) and may be less effective at P sorption.

*P. australis* is not a well-documented biochar feedstock in the literature, but its morphology suggests its biochar properties would be more similar to those of maize straw and switchgrass biochars relative to woody biochars. Studies have demonstrated those non-woody biochars to have lower surface area (4.05 vs. 97.20 $m^2\ g^{-1}$ BET surface area) but higher pH (10.4 vs. 5.82) and ion exchange to facilitate sorption of dissolved P to surface functional groups compared to woody biochars (Chintala et al., 2014; Zhao et al., 2017). Elevated concentrations of P-binding metals like Ca, Mg, and Fe in the feedstock (and corresponding biochar) can also increase P sorption capacity by providing P- binding reactants on biochar surfaces (Dugdug et al., 2018; Ghodszad et al., 2021). The data show that [Fe] in the *P. australis* biochar used in the amendment experiment (3,442 mg Fe $kg^{-1}$) was closer to pine wood char (4,470 mg Fe $kg^{-1}$; Oustriere et al., 2017) than a corn husk biochar (620 mg Fe $kg^{-1}$; Eduah et al., 2019), but the observed [Fe] may have been immobilized by sulfide in the reducing marsh soils as was discussed to occur in low pH salt marsh environments. Calcium (Ca) and Magnesium (Mg) were not measured in this work, but Dugdug et al. (2018) found wheat straw biochar (likely of similar morphology to *P. australis)* to have lower Ca and Mg concentrations and sorb less P than a willow wood derived biochar (1.4 vs. 19.6 mg Ca $g^{-1}$; 0.18 vs. 2.14 mg Mg $g^{-1}$).

We thus suggest that non-woody feedstocks, low temperature, and heterogeneous open-air conditions of the prescribed burns result in biochars of very different properties than lab-created biochars from carefully selected feedstocks produced at constant temperatures under low oxygen for long durations. The key differences in pH, cation exchange capacity, and surface area likely render marsh prescribed burn-derived biochars ineffective for sorbing P.

## Implications for Understanding the Impacts of Prescribed Burns

The results of this work suggest that biochars produced under prescribed burn-like conditions provide no added P storage ecosystem services to tidal marshes. Potential losses of P storage due to *P. australis* removal and replacement with native vegetation thus remain a lost ecosystem service regardless of management technique. Work with biochars designed specifically to maximize P sorption in marshes is needed to understand whether and how biochar production conditions can be optimized for use in living shoreline or thin layer placement interventions as described in Raposa et al. (2023). Such studies should employ lab-created biochars produced at high temperatures as well as using or amending the feedstock with high concentrations of P-binding metals like Fe, Mg, and Ca to maximize P sorption. Activation by pyrolysis with steam and $CO_2$ can also enhance P sorption by increasing pH and P-binding metal concentrations (Zhang et al., 2017).

While our work suggests no P ecosystem services, the biogeochemical mechanisms for potential carbon and nitrogen ecosystem services differ from those of P. Whereas P reduction relies on sorption and uptake, N may be reduced via denitrification, and C is stored by being refractory and resistant to degradation. Further work is needed to assess whether prescribed burns provide C and N biogeochemical services.

Future studies monitoring salt marsh P fractions should measure in situ soil and porewater pH, redox state, as well as Fe, Al, Ca, and/or Mg concentrations to determine how pH dependent P precipitation interactions are influencing P compositions. Ratios of Al, Fe, and P can indicate P mobility (Joshi et al., 2015; Kleeberg et al., 2010). Quantifying vegetation density and/or above and belowground biomass to examine relationships to soil P concentrations would also further elucidate the impacts of vegetation and grass species on marsh P retention. Alternatively, mesocosm experiments could be used to isolate the influence of these environmental factors on P processes on marshes. Finally, it should be noted that biochars designed specifically for P sorption may provide added P sorption/storage ecosystem benefits. More work is needed to understand how the conditions of prescribed burning and wildfires impact the properties of a resulting biochar and/or to optimize laboratory-designed chars for P sorption. The use of biochars in thin layer placements or living shorelines, for example, may offer P ecosystem services with chars designed specifically for this purpose.

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**Author contributions** Andrew Wozniak and Kari St. Laurent, secured funding for, conceived, and prepared the study. Data collection and analysis were performed by Pamela Edris under the supervision of Andrew Wozniak. The first draft of the manuscript was written by Pamela Edris and all authors commented on previous versions of the manuscript. All authors read and approved of the final manuscript.

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**Data Availability** Data is available at Wozniak, A. S., & Edris, P. (2024). Wozniak NERRS Science Collaborative BiMRAC. Open Science Framework. osf.io/kq3hw.

## Declarations

**Conflict of interest** The authors have no relevant financial or non-financial interests to disclose.

**Disclaimer** Statements, findings, conclusions, and recommendations are those of the authors and do not necessarily reflect the views of NOAA.



## References

Anton, J. M., Ervin, E., & Keeler, C. (2021). Identification of phosphorus sources in the East Creek using isotope signatures of the residual phosphorus pool. *Dissertations & Theses @ University of Delaware.* https://udspace.udel.edu/handle/19716/30446

Aspila, K. I., Agemian, H., & Chau, A. S. Y. (1976). A semi-automated method for the determination of inorganic, organic and total phosphate in sediments. *The Analyst*, *101*(1200), 187–197. https://doi.org/10.1039/AN9760100187

Bahureksa, W., Tfaily, M. M., Boiteau, R. M., Young, R. B., Logan, M. N., McKenna, A. M., & Borch, T. (2021). Soil Organic Matter Characterization by Fourier Transform Ion Cyclotron Resonance Mass Spectrometry (FTICR MS): A Critical Review of Sample Preparation, Analysis, and Data Interpretation. *Environmental Science & Technology*, *55*(14), 9637–9656. https://doi.org/10.1021/ACS.EST.1C01135

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., & Silliman, B. R. (2011). The Value of Estuarine and Coastal Ecosystem Services. *Ecological Monographs*, *81*(2), 169–193. https://doi.org/10.1890/10-1510.1

Bartlett, R., & James, B. (1980). Studying Dried, Stored Soil Samples — Some Pitfalls. *Soil Science Society of America Journal*, *44*(4), 721–724. https://doi.org/10.2136/SSSAJ1980.03615995004400040011X

Barton, K. (2020). MuMin: Multi-model inference. R package version 1.43.17.

Blake, L., Goulding, K. W. T., Mott, C. J. B., & Poulton, P. R. (2000). Temporal changes in chemical properties of air-dried stored soils and their interpretation for long-term experiments. *European Journal of Soil Science*, *51*(2), 345–353. https://doi.org/10.1046/J.1365-2389.2000.00307.X

Bolton, L., Joseph, S., Greenway, M., Donne, S., Munroe, P., & Marjo, C. E. (2019). Phosphorus adsorption onto an enriched biochar substrate in constructed wetlands treating wastewater. *Ecological*

*Engineering*, *142*, 100005. https://doi.org/10.1016/J.ECOENA.2019.100005

Cahoon, D. R., Guntenspergen, G., & Baird, S. (2010). Do annual prescribed fires enhance or slow the loss of coastal marsh habitat at Blackwater National Wildlife Refuge? *JFSP Research Project Reports*, *117*. http://digitalcommons.unl.edu/jfspresearch

Chintala, R., Schumacher, T. E., Mcdonald, L. M., Clay, D. E., Malo, D. D., Papiernik, S. K., Clay, S. A., & Julson, J. L. (2014). Phosphorus sorption and availability from biochars and soil/biochar mixtures. *CLEAN – Soil Air Water*, *42*(5), 626–634. https://doi.org/10.1002/CLEN.201300089

Coleman, D. J., Cassalho, F., Miesse, T. W., & Ferreira, C. M. (2023). The role of invasive Phragmites australis in wave attenuation in the eastern United States. *Estuaries and Coasts*, *46*(2), 404–416. https://doi.org/10.1007/s12237-022-01138-x

Craft, C. (2007). Freshwater input structures soil properties, vertical accretion, and nutrient accumulation of Georgia and U.S. tidal marshes. *Limnology and Oceanography*, *52*(3), 1220–1230. https://doi.org/10.4319/LO.2007.52.3.1220

Daly, K., Styles, D., Lalor, S., & Wall, D. P. (2015). Phosphorus sorption, supply potential and availability in soils with contrasting parent material and soil chemical properties. *European journal of soil science*, *66*(4), 792–801. https://doi.org/10.1111/ejss.12260

Darby, F. A., & Turner, R. E. (2008). Below- and aboveground biomass of Spartina alterniflora: Response to nutrient addition in a Louisiana salt marsh. *Estuaries and Coasts*, *31*(2), 326–334. https://www.jstor.org/stable/40663432

DeLaune, R. D., Reddy, K. R., Richardson, C. J., & Megonigal, J. P. (2013). *Methods in Biogeochemistry of Wetlands*. American Society of Agronomy and Soil Science Society of America. https://doi.org/10.2136/SSSABOOKSER10

Devau, N., Le Cadre, E., Hinsinger, P., Jaillard, B., & Gérard, F. (2009). Soil pH controls the environmental availability of phosphorus: experimental and mechanistic modelling approaches. *Applied Geochemistry*, *24*(11), 2163–2174. https://doi.org/10.1016/j.apgeochem.2009.09.020

DNREC (2005). Delaware Bay and Estuary Assessment Report. *Delaware Bay and Estuary Basin Whole Basin Management Team.* Doc. No. 40-01-01/05/02/01. https://documents.dnrec.delaware.gov/Watershed/Assessment/Whole-Basin/Assessments/Delaware-Bay.pdf

DNREC (2022). Delaware National Estuarine Research Reserve 2022–2027 Management Plan. https://documents.dnrec.delaware.gov/coastal/DNERR/2022-DNERR-Management-Plan.pdf

DNREC Watershed Assessment and Management Section (2012). *St. Jones River Watershed Pollution Control Strategy: A watershed-based strategy to implement total maximum daily loads in Delaware*. https://documents.dnrec.delaware.gov/swc/wa/Documents/WatershedPlans/St%20Jones%202012%20PCS%20final.pdf

Dugdug, A. A., Chang, S. X., Ok, Y. S., Rajapaksha, A. U., & Anyia, A. (2018). Phosphorus sorption capacity of biochars varies with biochar type and salinity level. *Environmental Science and Pollution Research*, *25*(26), 25799–25812. https://doi.org/10.1007/s11356-018-1368-9

Dunne, E. J., Clark, M. W., Mitchell, J., Jawitz, J. W., & Reddy, K. R. (2010). Soil phosphorus flux from emergent marsh wetlands and surrounding grazed pasture uplands. *Ecological Engineering*, *36*(10), 1392–1400. https://doi.org/10.1016/J.ECOLENG.2010.06.018

Eduah, J. O., Nartey, E. K., Abekoe, M. K., Breuning-Madsen, H., & Andersen, M. N. (2019). Phosphorus retention and availability in three contrasting soils amended with rice husk and corn cob biochar at varying pyrolysis temperatures. *Geoderma*, *341*, 10–17. https://doi.org/10.1016/J.GEODERMA.2019.01.016

Ehrhardt, M., & Koeve, W. (1999). Determination of particulate organic carbon and nitrogen. In K Grasshoff, K. Kremling, & M. Ehrhardt (Eds.), *Methods of Seawater Analysis* (3rd ed., pp. 437–444). Wiley. https://doi.org/10.1002/9783527613984.CH17

Faria, M., Bertocco, T., Barroso, A., Carvalho, M., Fonseca, F., Delerue Matos, C., Figueiredo, T., Sequeira Braga, A., Valente, T., & Jiménez-Ballesta, R. (2023). A Comparison of Analytical Methods for the Determination of Soil pH: Case Study on Burned Soils in Northern Portugal. *Fire*, *6*(6), 227. https://doi.org/10.3390/FIRE6060227

Faridullah, U., Waseem, A., Irshad, M., Mahmood, Q., Sabir, M. A., & Alam, A. (2012). Leachability of phosphorus in soils after application of burned and unburned poultry litter. *International Journal of the Physical Sciences*, *7*(5), 834–840. https://doi.org/10.5897/IJPS11.1719

Fischer, J. M., Reed-Andersen, T., Klug, J. L., & Chalmers, A. G. (2000). Spatial Pattern of Localized Disturbance along a Southeastern Salt Marsh Tidal Creek. *Estuaries*, *23*(4), 565–571. https://doi.org/10.2307/1353146

Fisher, T., Peele, E., Ammerman, J., & HardingJr, L. (1992). Nutrient limitation of phytoplankton in Chesapeake Bay. *Marine Ecology Progress Series*, *82*(1), 51–63. https://www.jstor.org/stable/24827423

Fox, A., Kwapinski, W., Griffiths, B. S., & Schmalenberger, A. (2014). The role of sulfur- and phosphorus-mobilizing bacteria in biochar-induced growth promotion of Lolium perenne. *FEMS Microbiology Ecology*, *90*(1), 78–91. https://doi.org/10.1111/1574-6941.12374

Gao, S., & DeLuca, T. H. (2020). Biochar alters nitrogen and phosphorus dynamics in a western rangeland ecosystem. *Soil Biology and Biochemistry*, *148*, 107868. https://doi.org/10.1016/J.SOILBIO.2020.107868

Geatz, G. W., Needelman, B. A., Weil, R. R., & Megonigal, J. P. (2013). Nutrient availability and soil organic matter decomposition response to prescribed burns in Mid-Atlantic brackish tidal marshes. *Soil Science Society of America Journal*, *77*(5), 1852–1864. https://doi.org/10.2136/sssaj2012.0272

Ghodszad, L., Reyhanitabar, A., Maghsoodi, M. R., Lajayer, A., B., & Chang, S. X. (2021). Biochar affects the fate of phosphorus in soil and water: A critical review. *Chemosphere*, *283*, 131176. https://doi.org/10.1016/J.CHEMOSPHERE.2021.131176

Giblin, A. E., Tobias, C. R., Song, B., Weston, N., Banta, G. T., & Rivera-Monroy, V. H. (2013). The importance of dissimilatory nitrate reduction to ammonium (DNRA) in the nitrogen cycle of coastal ecosystems. *Oceanography*, *26*(3), 124–131. https://doi.org/10.5670/OCEANOG.2013.54

Gu, C., Dam, T., Hart, S. C., Turner, B. L., Chadwick, O. A., Berhe, A., Hu, Y., & Zhu, M. (2020). Quantifying Uncertainties in Sequential Chemical Extraction of Soil Phosphorus Using XANES Spectroscopy. *Environmental Science & Technology*, *54*(4), 2257–2267. https://doi.org/10.1021/acs.est.9b05278

Gustafsson, Ö., Bucheli, T. D., Kukulska, Z., Andersson, M., Largeau, C., Rouzaud, J., Reddy, C. M., & Eglinton, T. I. (2001). Evaluation of a Protocol for the Quantification of Black Carbon in Sediments. *Global Biogeochemical Cycles*, *15*(4), 881–890. https://doi.org/10.1029/2000GB001380

Hedley, M. J., Stewart, J. W. B., & Chauhan, B. S. (1982). Changes in inorganic and organic soil phosphorus fractions induced by cultivation practices and by laboratory incubations. *Soil Science Society of America Journal*, *46*(5), 970–976. https://doi.org/10.2136/SSSAJ1982.03615995004600050017X

Henton, C. M., Reese, C. A., Andrew, J., Fleming, S., & Heitmuller, F. T. (2013). Impact of prescribed burns on marsh surface elevation: Big Branch Marsh, Louisiana. *Southeastern Geographer*, *53*(1), 61–78. https://doi.org/10.1353/sgo.2013.0008

Jassal, R. S., Johnson, M. S., Molodovskaya, M., Black, T. A., Jollymore, A., & Sveinson, K. (2015). Nitrogen enrichment potential of biochar in relation to pyrolysis temperature and feedstock quality. *Journal of Environmental Management*, *152*, 140–144. https://doi.org/10.1016/J.JENVMAN.2015.01.021

Joshi, S. R., Kukkadapu, R. K., Burdige, D. J., Bowden, M. E., Sparks, D. L., & Jaisi, D. P. (2015). Organic matter remineralization predominates phosphorus cycling in the mid-bay sediments in the Chesapeake Bay. *Environmental Science and Technology*, *49*(10), 5887–5896. https://doi.org/10.1021/es5059617

Karam, D. S., Nagabovanalli, P., Sundara Rajoo, K., Fauziah Ishak, C., Abdu, A., Rosli, Z., Melissa Muharam, F., & Zulperi, D. (2022). An overview on the preparation of rice husk biochar, factors affecting its properties, and its agriculture application. *Journal of the Saudi Society of Agricultural Sciences*, *21*(3), 149–159. https://doi.org/10.1016/J.JSSAS.2021.07.005

Keiluweit, M., Nico, P. S., Johnson, M. G., & Kleber, M. (2010). Dynamic molecular structure of plant biomass-derived black carbon (biochar). *Environmental science & technology*, *44*(4), 1247–1253. https://doi.org/10.1021/es9031419

Kelly, C. (2023). Do prescribed burns yield enhanced carbon storage in salt marshes? A comparative assessment in Delaware salt marshes with varied burn histories. *Dissertations & Theses @ University of Delaware; ProQuest Dissertations & Theses Global. (2867942883).* https://www.proquest.com/docview/2867942883

Kennedy, D., Kowalski, J., & Amand, S. (2017). S. Blackbird Creek Preservation and Action Charter (BCPAC). https://www.wrc.udel.edu/wp-content/uploads/2017/07/BCPAC-Blackbird-Cr.-DE-Report-2017.pdf

Kögel-Knabner, I., & Rumpel, C. (2018). Advances in Molecular Approaches for Understanding Soil Organic Matter Composition, Origin, and Turnover: A Historical Overview. In D. L. Spark (Ed.) *Advances in Agronomy* (Vol. 149, pp. 1–48). Academic Press. https://doi.org/10.1016/BS.AGRON.2018.01.003

Kim, B., Gautier, M., Simidoff, A., Sanglar, C., Chatain, V., Michel, P., & Gourdon, R. (2016). pH and Eh effects on phosphorus fate in constructed wetland's sludge surface deposit. *Journal of Environmental Management*, *183*, 175–181. https://doi.org/10.1016/J.JENVMAN.2016.08.064

Kleeberg, A., Herzog, C., Jordan, S., & Hupfer, M. (2010). What drives the evolution of the sedimentary phosphorus cycle? *Limnologica*, *40*(2), 102–113. https://doi.org/10.1016/J.LIMNO.2009.11.001

Kutcher, T. E., Chaffee, C., & Raposa, K. B. (2018). *Rhode Island Coastal Wetland Restoration Strategy*. http://www.crmc.ri.gov/habitatrestoration/RICWRestorationStrategy.pdf

Leonard, C. A., Ahn, C., & Birch, D. (2010). Above- and below-ground vegetative responses to prescribed fire regimes in a Chesapeake Bay tidal brackish marsh. *Journal of Ecology and Environment*, *33*(4), 351–361. https://doi.org/10.5141/JEFB.2010.33.4.351

Leonard, L. A., Wren, P. A., & Beavers, R. L. (2002). Flow dynamics and sedimentation in *Spartina alterniflora* and *Phragmites australis* marshes of the Chesapeake Bay. *Wetlands*, *22*, 415–424. https://doi.org/10.1672/0277-5212(2002)022[0415:FDASIS]2.0.CO;2.

Li, Y., Fu, C., Hu, J., Zeng, L., Tu, C., & Luo, Y. (2023). Soil Carbon, Nitrogen, and Phosphorus Stoichiometry and Fractions in Blue Carbon Ecosystems: Implications for Carbon Accumulation in Allochthonous-Dominated Habitats. *Environmental Science and Technology*, *57*(14), 5913–5923. https://doi.org/10.1021/acs.est.3c00012

Loomis, M. J., & Craft, C. B. (2010). Carbon sequestration and nutrient (Nitrogen, Phosphorus) accumulation in river-dominated tidal marshes, Georgia, USA. *Soil Science Society of America Journal*, *74*(3), 1028–1036. https://doi.org/10.2136/SSSAJ2009.0171

Álvarez-Rogel, J., Jiménez-Cárceles, F. J., & Egea-Nicolás, C. (2007). Phosphorus retention in a coastal salt marsh in SE Spain. *Science of The Total Environment*, *378*(1–2), 71–74. https://doi.org/10.1016/J.SCITOTENV.2007.01.016

McKenna, T. E., Callahan, J. A., Medlock, C. L., & Bates, N. S. (2018). Creation of improved accuracy LiDAR-based digital elevation models for the St. Jones River and Blackbird Creek Watersheds. Technical Report. 1–30.

Menzel, D. W., & Vaccaro, R. F. (1964). The measurement of dissolved organic and particulate carbon in seawater. *Limnology and Oceanography*, *9*(1), 138–142. https://doi.org/10.4319/LO.1964.9.1.0138

Meyerson, L. A., Saltonstall, K., & Chambers, R. M. (2009). Phragmites australis in eastern North America: a historical and ecological perspective. In B. R. Silliman, M. D. Bertness, & D. Strong (Eds.), *Salt Marshes under Global Siege* (pp. 57–82). University of California Press.

Murphy, J., & Riley, J. P. (1962). A modified single solution method for the determination of phosphate in natural waters. *Analytica Chimica Acta*, *27*, 31–36. https://doi.org/10.1016/S0003-2670(00)88444-5

Myers-Pigg, A. N., Grieger, S., Roebuck, J. A., Barnes, M. E., Bladon, K. D., Bailey, J. D., Barton, R., Chu, R. K., Graham, E. B., Homolka, K. K., Kew, W., Lipton, A. S., Scheibe, T., Toyoda, J. G., & Wagner, S. (2024). Experimental Open Air Burning of Vegetation Enhances Organic Matter Chemical Heterogeneity Compared to Laboratory Burns. *Environmental Science and Technology*, *58*(22), 9679–9688. https://doi.org/10.1021/ACS.EST.3C10826

Ngatia, L. W., Hsieh, Y. P., Nemours, D., Fu, R., & Taylor, R. W. (2017). Potential phosphorus eutrophication mitigation strategy: Biochar carbon composition, thermal stability and pH influence phosphorus sorption. *Chemosphere*, *180*, 201–211. https://doi.org/10.1016/J.CHEMOSPHERE.2017.04.012

Nguyen, M. N. (2021). Potential use of silica-rich biochar for the formulation of adaptively controlled release fertilizers: A mini review. *Journal of Cleaner Production*, *307*, 127188. https://doi.org/10.1016/J.JCLEPRO.2021.127188

Oliveira, F. R., Patel, A. K., Jaisi, D. P., Adhikari, S., Lu, H., & Khanal, S. K. (2017). Environmental application of biochar: Current status and perspectives. *Bioresource Technology*, *246*, 110–122. https://doi.org/10.1016/J.BIORTECH.2017.08.122

Oustriere, N., Marchand, L., Rosette, G., Friesl-Hanl, W., & Mench, M. (2017). Wood-derived-biochar combined with compost or iron grit for in situ stabilization of Cd, Pb, and Zn in a contaminated soil. *Environmental Science and Pollution Research*, *24*(8), 7468–7481. https://doi.org/10.1007/s11356-017-8361-6

Paerl, H. W., Pinckney, J. L., Fear, J. M., & Peierls, B. L. (1998). Ecosystem responses to internal and watershed organic matter loading: consequences for hypoxia in the eutrophying Neuse River Estuary, North Carolina, USA. *Marine Ecology Progress Series*, *166*, 17–25. https://doi.org/10.3354/MEPS166017

Paludan, C., & Morris, J. T. (1999). Distribution and speciation of phosphorus along a salinity gradient in intertidal marsh sediments. *Biogeochemistry*, *45*(2), 197–221. https://doi.org/10.1007/BF01106781

Raposa, K. B., Woolfolk, A., Endris, C. A., Fountain, M. C., Moore, G., Tyrrell, M., Swerida, R., Lerberg, S., Puckett, B. J., Ferner, M. C., Hollister, J., Burdick, D. M., Champlin, L., Krause, J. R., Haines, D., Gray, A. B., Watson, E. B., & Wasson, K. (2023). Evaluating thin-layer sediment placement as a tool for enhancing tidal marsh resilience: A coordinated experiment across eight US National Estuarine Research Reserves. *Estuaries and Coasts*, *46*(3), 595–615. https://doi.org/10.1007/s12237-022-01161-y

Reboreda, R., & Caçador, I. (2007). Halophyte vegetation influences in salt marsh retention capacity for heavy metals. *Environmental Pollution*, *146*(1), 147–154. https://doi.org/10.1016/j.envpol.2006.05.035

Reimold, R. J., & Daiber, F. C. (1970). Dissolved phosphorus concentrations in a natural salt-marsh of Delaware. *Hydrobiologia*, *36*(3), 361–371. https://doi.org/10.1007/BF00039795

Ridal, J. J., & Moore, R. M. (1990). A re-examination of the measurement of dissolved organic phosphorus in seawater. *Marine*

*Chemistry*, *29*, 19–31. https://doi.org/10.1016/0304-4203(90)90003-U
Rozan, T. F., Taillefert, M., Trouwborst, R. E., Glazer, B. T., Ma, S., Herszage, J., Valdes, L. M., Price, K. S., & Luther, G. W. (2002). Iron-sulfur-phosphorus cycling in the sediments of a shallow coastal bay: Implications for sediment nutrient release and benthic macroalgal blooms. *Limnology and Oceanography*, *47*(5), 1346–1354. https://doi.org/10.4319/LO.2002.47.5.1346
Ruggeri, L. R. (2014). Analysis of herbicide treatment effectiveness on common reed (Phragmites australis) of Delaware Bay salt marshes. *Theses, Dissertations and Culminating Projects*. 598. https://digitalcommons.montclair.edu/etd/598
Ruttenberg, K. C. (1992). Development of a sequential extraction method for different forms of phosphorus in marine sediments. *Limnology and Oceanography*, *37*(7), 1460–1482. https://doi.org/10.4319/LO.1992.37.7.1460
Schafer, J. L., & Mack, M. C. (2018). Nutrient limitation of plant productivity in scrubby flatwoods: does fire shift nitrogen versus phosphorus limitation? *Plant Ecology*, *9*, 1063–1079. https://doi.org/10.1007/S11258-018-0859-6
Sharp, J. H., Yoshiyama, K., Parker, A. E., Schwartz, M. C., Curless, S. E., Beauregard, A. Y., Ossolinski, J. E., & Davis, A. R. (2009). A biogeochemical view of estuarine eutrophication: Seasonal and spatial trends and correlations in the Delaware Estuary. *Estuaries and Coasts*, *32*, 1023–1043. https://doi.org/10.1007/s12237-009-9210-8
Shen, Q., Hedley, M., Camps Arbestain, M., & Kirschbaum, M. U. F. (2016). Can biochar increase the bioavailability of phosphorus. *Journal of Soil Science and Plant Nutrition*, *16*(2), 268–286. https://doi.org/10.4067/S0718-95162016005000022
Song, G., & Liu, S. (2015). Yellow Sea and East China Sea and potential impacts on ecosystem. *Acta Oceanologica Sinica*, *34*(4), 84–91. https://doi.org/10.1007/s13131-015-0653-4
Strømgaard, P. (1992). Immediate and long-term effects of fire and ash-fertilization on a Zambian miombo woodland soil. *Agriculture Ecosystems & Environment*, *41*(1), 19–37. https://doi.org/10.1016/0167-8809(92)90177-D
Struyf, E., van Damme, S., Gribsholt, B., Middelburg, J. J., & Meire, P. (2005). Biogenic silica in tidal freshwater marsh sediments and vegetation (Schelde estuary, Belgium). *Marine Ecology Progress Series*, *303*, 51–60. https://doi.org/10.3354/MEPS303051
Sundareshwar, P., & Morris, J. T. (1999). Phosphorus sorption characteristics of intertidal marsh sediments along an estuarine salinity gradient. *Limnology and Oceanography*, *44*(7), 1693–1701. https://doi.org/10.4319/LO.1999.44.7.1693
Sun, H., Brown, A., Coppen, J., & Steblein, P. (2006). Response of Phragmites to environmental parameters associated with treatments. *Wetlands Ecology and Management*, *15*(1), 63–79. https://doi.org/10.1007/S11273-006-9013-7
Taillie, P. J., & Moorman, C. E. (2019). Marsh bird occupancy along the shoreline-to-forest gradient as marshes migrate from rising sea level. *Ecosphere*, *10*(1), e02555. https://doi.org/10.1002/ECS2.2555
Tesfaye, F., Liu, X., Zheng, J., Cheng, K., Bian, R., Zhang, X., Li, L., Drosos, M., Joseph, S., & Pan, G. (2021). Could biochar amendment be a tool to improve soil availability and plant uptake of phosphorus? A meta-analysis of published experiments. *Environmental Science and Pollution Research*, *28*(26), 34108–34120. https://doi.org/10.1007/S11356-021-14119-7
Teuchies, J., Vandenbruwaene, W., Carpentier, R., Bervoets, L., Temmerman, S., Wang, C., Maris, T., Cox, T. J., Van Braeckel, A., & Meire, P. (2013). Estuaries as filters: the role of tidal marshes in trace metal removal. *PloS one*, *8*(8), e70381. https://doi.org/10.1371/journal.pone.0070381
Tobias, C., & Neubauer, S. C. (2019). Salt Marsh Biogeochemistry—An Overview. In G. M. E. Perillo, E. Wolanski, D. R. Cahoon, & C. S. Hopkinson (Eds.), *Coastal Wetlands: An Integrated Ecosystem Approach*, (2nd ed., pp. 539–596). Elsevier. https://doi.org/10.1016/B978-0-444-63893-9.00016-2
Upreti, K., Joshi, S. R., McGrath, J., & Jaisi, D. P. (2015). Factors controlling phosphorus mobilization in a coastal plain tributary to the Chesapeake Bay. *Soil Science Society of America Journal*, *79*(3), 826–837. https://doi.org/10.2136/SSSAJ2015.03.0117
Velinsky, D. J., Paudel, B., Belton, T. J., & Sommerfield, C. K. (2017). Tidal marsh record of nutrient loadings in Barnegat Bay, New Jersey. *Journal of Coastal Research*, *78*(sp1), 79–88. https://doi.org/10.2112/SI78-008.1
Volk, J. A., Scudlark, J. R., Savidge, K. B., Andres, A. S., Stenger, R. J., & Ullman, W. J. (2012). Intra- and inter-annual trends in phosphorus loads and comparison with nitrogen loads to Rehoboth Bay, Delaware (USA). *Estuarine Coastal and Shelf Science*, *96*(1), 139–150. https://doi.org/10.1016/J.ECSS.2011.10.023
White, J. R., Gardner, L. M., Sees, M., & Corstanje, R. (2008). The short-term effects of prescribed burning on biomass removal and the release of nitrogen and phosphorus in a treatment wetland. *Journal of Environmental Quality*, *37*(6), 2386–2391. https://doi.org/10.2134/JEQ2008.0019
Wigand, C., Ardito, T., Chaffee, C., Ferguson, W., Paton, S., Raposa, K., Vandemoer, C., & Watson, E. (2017). A climate change adaptation strategy for management of coastal marsh systems. *Estuaries and Coasts*, *40*(3), 682–693. https://doi.org/10.1007/s12237-015-0003-y
Windham, L., & Meyerson, L. A. (2003). Effects of common reed (Phragmites australis) expansions on nitrogen dynamics of tidal marshes of the Northeastern U. *S Estuaries*, *26*(2), 452–464. https://www.jstor.org/stable/1353360
Wozniak, A. S., Goranov, A. I., Mitra, S., Bostick, K. W., Zimmerman, A. R., Schlesinger, D. R., Myneni, S., & Hatcher, P. G. (2020). Molecular heterogeneity in pyrogenic dissolved organic matter from a thermal series of oak and grass chars. *Organic Geochemistry*, *148*, 104065. https://doi.org/10.1016/j.orggeochem.2020.104065
Yao, S. Q., Groffman, P. M., Alewell, C., & Ballantine, K. (2018). Soil amendments promote denitrification in restored wetlands. *Restoration Ecology*, *26*(2), 294–302. https://doi.org/10.1111/rec.12573
Zhang, H. H., Cao, X. Y., Wang, H., Ma, Z., Li, J., Zhou, L. M., & Yang, G. P. (2019). Effect of black carbon on sorption and desorption of phosphorus onto sediments. *Marine Pollution Bulletin*, *146*, 435–441. https://doi.org/10.1016/J.MARPOLBUL.2019.06.059
Zhang, H., Voroney, R. P., & Price, G. W. (2017). Effects of Temperature and Activation on Biochar Chemical Properties and Their Impact on Ammonium, Nitrate, and Phosphate Sorption. *Journal of Environmental Quality*, *46*(4), 889–896. https://doi.org/10.2134/JEQ2017.02.0043
Zhang, J. Z., & Huang, X. L. (2011). Effect of temperature and salinity on phosphate sorption on marine sediments. *Environmental Science and Technology*, *45*(16), 6831–6837. https://doi.org/10.1021/es200867p
Zhang, W. L., Zeng, C. S., Tong, C., Zhai, S. J., Lin, X., & Gao, D. Z. (2015). Spatial distribution of phosphorus speciation in marsh sediments along a hydrologic gradient in a subtropical estuarine wetland, China. *Estuarine Coastal and Shelf Science*, *154*, 30–38. https://doi.org/10.1016/J.ECSS.2014.12.023
Zhao, S., Wang, B., Gao, Q., Gao, Y., & Liu, S. (2017). Adsorption of phosphorus by different biochars. *Spectroscopy Letters*, *50*(2), 73–80. https://doi.org/10.1080/00387010.2017.1287091