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Management strategies for reducing phosphorus levels in saltwater-intruded agricultural fields

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ABSTRACT

As sea levels continue to rise and high tide flooding events increase in frequency, researchers and farmers alike are looking for solutions to adapt to and mitigate the effects of saltwater intrusion (SWI). Some landowners on the Lower Eastern Shore of Maryland respond to SWI by taking land out of agriculture. For example, they (1) attempt to *remediate* salt-damaged soils (e.g., planting switchgrass, *Panicum virgatum*), (2) *restore* native marsh grasses (e.g., planting saltmarsh hay, *Spartina patens*), or (3) *abandon* fields altogether (e.g., allow for natural recruitment). This work examines the ability of each of these land management practices to reduce phosphorus (P) levels in soils and porewater, with the overall goal to benefit both the farming community and water quality in the Chesapeake Bay. We show that remediation and restoration practices are efficient at taking up soil P and reducing porewater P concentrations through biomass P uptake. After three years of growth, we observed an increase in P uptake in biomass of *Panicum virgatum* (remediation species; 11–30 kg ha^{-1}) and *Spartina patens* (restoration species; 4–18 kg ha^{-1}) and a decline in available soil P pools (M3P; 30–50 % kg M3P ha^{-1}). At all farms, under all three management strategies, the P fertility index value (FIV) in the topsoil was 33–50 % lower than baseline conditions, likely reducing the potential release of P to nearby waterways. Results from this work will help inform state-level coastal management policies and determine optimal strategies for climate resilience.

1. Introduction

Sea level rise (SLR) poses a threat to coastal regions. Globally, sea level is projected to rise by 0.8–1.6 feet (0.24–0.49 m) by 2050 and 1.2–3.0 feet (0.37–0.91 m) by 2100, along with the increased frequency of storm events and periods of drought (Boesch et al., 2018; Sweet et al., 2018). On the Eastern Seaboard of the U.S., saltwater intrusion (SWI) accompanies SLR, damaging agricultural systems, and negatively impacting soil health and plant productivity (Bhattachan et al., 2018; Gedan and Fernández-Pascual, 2019). Saltwater intrusion is the movement of sea salts into freshwater aquifers and shallow groundwater tables. As sea levels continue to rise, SWI will further encroach on the land, salinizing a greater area of agroecosystems along the coast of the Chesapeake Bay (Tully et al., 2019a; Guimond and Michael, 2020). Saltwater intrusion in the Chesapeake Bay region is facilitated by an extensive ditch network, which connects coastal agricultural lands to saline water bodies (Bhattachan et al., 2018). As SWI advances across the landscape, it will alter soil salinity and phosphorus (P) concentrations, thus altering biogeochemical cycling, leading to (i) decreased agricultural productivity through soil aggregate dispersion (e.g., sodium inputs from saltwater), (ii) eutrophication through nutrient export (e.g., dissolved P), and (iii) loss of farmland as fields convert into tidal marshes (Tully et al., 2019a; Gedan et al., 2020; Mondal et al., 2023). As sea levels continue to rise and our coastlines change, we need to implement new land management strategies to reduce P losses to nearby water bodies.

Phosphorus has accumulated in agricultural soils in the Delmarva Peninsula of the U.S. due to decades of nitrogen-based, phosphorus-rich fertilizer application with poultry manure. This accumulation of soil P is known as "legacy P" (Kleinman et al., 2007; Kleinman et al., 2015). Phosphorus is able to sorb on clay surfaces, or the iron (Fe) and aluminum oxides and hydroxides present in soil, but with the influx of

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salts from SLR and SWI, legacy P can be released (Nair et al., 2010; Baker et al., 2017; Chakraborty et al., 2022). Studies have shown that legacy P can be mobilized decades after application, with negative consequences for downstream water quality (Carpenter, 2008; Kleinman et al., 2011). Saltwater intrusion may facilitate the release of legacy P due to competition between phosphate (PO_4^{3-}) and sulfate (SO_4^{2-}) for Fe on soil mineral surfaces (Jordan et al., 2008), especially as soils become anoxic, reducing Fe minerals and releasing PO_4^{3-} from binding sites (Patrick and Khalid, 1974; Tully et al., 2019b). In saltwater systems, SO_4^{2-} in the solution competes with PO_4^{3-} for binding sites as the soils dry, leaving PO_4^{3-} in solution and prone to loss (Upreti et al., 2015; Tully et al., 2019b). In some cases, land easement programs (e.g., Conservation Reserve Enhancement Program, Environmental Quality Incentives Program) may prioritize flooding fields (e.g., by plugging drainage ditches) in an effort to return the hydrology to its "natural" state rather than restoring native vegetation (CREP, 2022 and EQIP, 2021). If adopted at a large scale, such practices may have negative consequences for water quality as large amounts of legacy P could be mobilized when fields are allowed to flood with saltwater (Tully et al., 2019b). Thus, strategies are needed to draw down soil P levels before hydrology is restored. This study will investigate how different management strategies will affect P in soils, plants, and porewater in saltwater-intruded agricultural fields.

Rising seas and elevated soil salinity levels have forced farmers to reconsider their management practices, and these decisions will impact downstream ecosystems (Sudol et al., 2023). Different frameworks are used to categorize human responses to climate change (Deshmukh and Hastak, 2012; Canyon et al., 2015; van Wesenbeeck et al., 2017; Doberstein et al., 2019). For example, people may protect their land by installing tide gates (Doberstein et al., 2019), accommodate by elevating fields (Oppenheimer et al., 2019), retreat and enroll fields in easement programs (van Wesenbeeck et al., 2017), or avoid using the land altogether (Hu et al., 2015).On the Eastern Shore of Maryland, where groundwater levels are too close to the soil surface to accommodate SLR (and SWI) by elevating fields (Oppenheimer et al., 2019; Sudol et al., 2023), farmers may instead allow the transition of fields into marshland. In our study, we focused on three primary responses to rising seas and salinity on coastal farms: *remediation* of fields for a few years before returning to agriculture, *restoration* of the land to native marshes, or *abandonment* of salt-damaged fields entirely (Baker et al., 2014; Tully et al., 2019a).

In agricultural systems, a *remediation* approach utilizes low-input crops that produce large quantities of biomass in order to draw down soil contaminants (e.g., legacy P, salts, heavy metals) and either be harvested or cultivated for a few years before returning to agriculture (Shen et al., 2002; Ghosh and Singh, 2005; Colomb et al., 2007; Kleinman et al., 2007; Dixit et al., 2015). In the study region, switchgrass (*Panicum virgatum* [*P. virgatum*]) is a fast-growing, perennial grass species that is more tolerant of saline and dry conditions due to its deep root system and strong associations with mycorrhizal fungi, so it is able to take up more water and nutrients than most agronomic crops (Di Virgilio et al., 2007; Schmer et al., 2011; Meyer et al., 2014; USDA NRCS, 2020). Additionally, the extensive root system can be as deep as three meters and as wide as six meters, allowing *P. virgatum* to act as a buffer to take up soil N and P and prevent nutrient loss to nearby waterbodies (Blanco-Canqui et al., 2004; Lemus et al., 2008; Kibet et al., 2016; Kumar et al., 2019). It could be used for biofuel, poultry bedding, or otherwise harvested to remove P from the system (Lee et al., 1998; David and Ragauskas, 2010; Purswell et al., 2020).

Restoring farm fields involves planting native marsh species, which encourages the return of wildlife and may support new recreational activities like waterfowl hunting (Baker et al., 2014). There are also state- and federal-level conservation programs that can support restoration practices (e.g., Environmental Quality Incentives Program; EQIP, 2021). On the Eastern Shore of Maryland, planting saltmarsh hay (*Spartina patens* [*S. patens*]) may facilitate the transition to tidal marshes that can protect coastlines from flooding and nutrient loss (e.g., P) to the Chesapeake Bay (Williams et al., 1992; Charles and Dukes, 2009; Hu et al., 2015; Gedan et al., 2011).

Once crops fail entirely and it is too challenging to operate equipment on saturated fields, some farmers choose to abandon a field. In this case, they may enroll their land in a conservation program and allow for natural plant recruitment to occur (e.g., WREP, 2022; MD DNR, 2022). Land abandonment may be the simplest option for those who do not want to invest any more time or money battling SWI. We have yet to understand the influence of remediating, restoring or abandoning fields on biogeochemical cycling, specifically P dynamics in coastal agricultural lands.

As sea levels continue to rise and high tide flooding events increase in frequency, researchers and farmers alike are looking for solutions to adapt to and mitigate the effects of SWI, specifically how we can locally manage land from agriculture to healthy marshes that can supply an extensive list of ecosystem services. The objectives of this research are to quantify P concentrations and pools across different management practices (e.g., remediate, restore, abandon) along a SWI gradient (1) in soil (by depth), (2) porewater, and (3) plant biomass. This research will benefit both the farming community and water quality in the Chesapeake Bay. For example, P uptake by plant tissues and removal of their biomass will decrease P concentrations in the soil and reduce farm-level P losses that contribute to eutrophication. Results from this study will help inform new management practices to help inform state-level coastal management policies and determine optimal strategies for coastal agricultural lands.

2. Methods

2.1. Study sites

Our study was located on the Lower Eastern Shore of the Chesapeake Bay in Maryland (MD), where sea levels are rising at twice the average global rate (3.4 mm compared to 1.7 mm per year; NOAA, 2021), and where agriculture is responsible for 27 % of the phosphorus (P) entering the Chesapeake Bay (CAST, 2019). The climate in the region is humid subtropical with a mean annual average temperature of 9.7ºC (mean annual minimum temperature of 8.3ºC; mean annual maximum temperature of 19.7ºC), and annual precipitation of 1194 mm (NRCC, 2022). The Chesapeake Bay waters range from fresh to saline (0–30 ppt; Chesapeake Bay Program 2020), where surface salinity ranges from 8.8 to 15.8 ppt near our experimental farms (NOAA, 2021). This study focuses on two low-lying, coastal counties along the Eastern Shore of Maryland, which are vulnerable to high tide flooding and sea-level rise (Sweet et al., 2018), as many homes and farms have a mean elevation of ~2.7 m above sea level with little to no slope. Dorchester and Somerset counties combined have over 3000 km of coastline and about 68 % of the land area is in cropland (USDA, 2019). These areas are severely threatened by sea-level rise, as this landscape is connected to the Chesapeake Bay through extensive ditch networks that facilitate the movement of saline water onto agricultural fields during high tide (Bhattachan et al., 2018). The region is predicted to experience sea-level rise of 0.8–1.6 feet (0.24–0.49 m) by 2050 (Fig. 1; Boesch et al., 2018). In addition, data suggests that as much as 30 % of the land area in Dorchester and Somerset counties is at risk of saltwater intrusion (Epanchin-Niell et al., 2023). In fact, this area is already experiencing the impacts of saltwater intrusion, as roughly 1 % of agricultural land has transitioned into marshland over the past eight years (2009–2017; Gedan et al., 2020).

A field experiment was established on the Lower Eastern Shore of Maryland in 2018 on three saltwater-intruded agricultural fields (Farm 6 and Farm 3 in May and Farm 1 in November) and one field not affected by saltwater intrusion (no-salt site in November; Fig. 1). One field (Farm 6) was located near Cambridge, MD in Dorchester Co., MD (38.5633° N, −76.0786° W). The other two fields were located in Somerset Co., MD with one near Princess Anne, MD (Farm 1; 38.2029°N, −75.6924 °W)

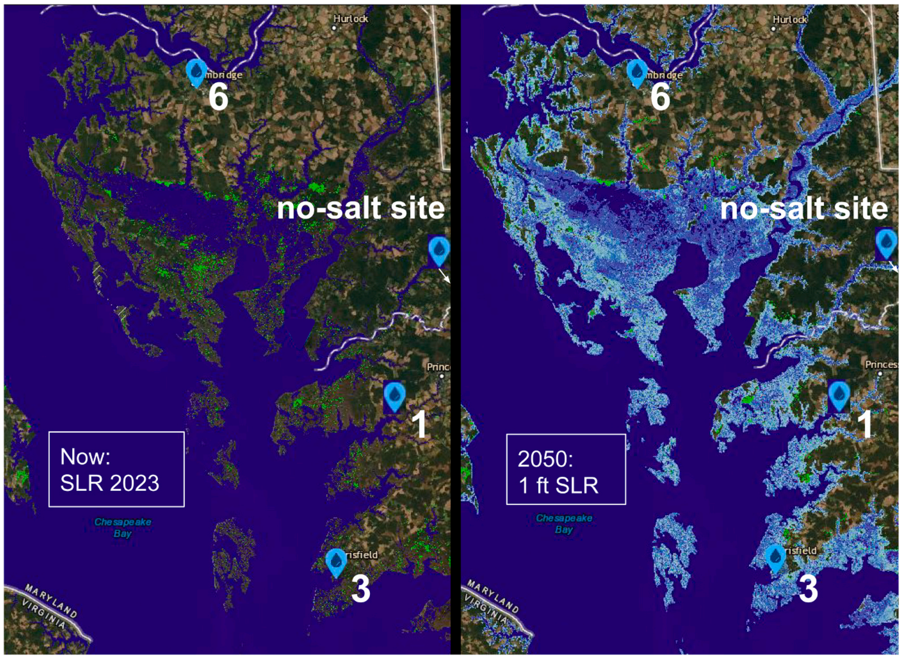

Fig. 1. Sea level rise impacts in 2023 compared to 2050 on coastal agricultural field sites located on the Lower Eastern Shore of Maryland. Sea level rise impacts (land coverage) in 2023 compared to projected 1.0 ft (0.305 m) of SLR by 2050 (NOAA, 2021). Agricultural field sites indicated by blue points labeled 1, 3, 6, and no-salt site.

and the other near Crisfield, MD (Farm 3; 37.9835°N, −75.8538°W). A fourth site was located inland and did not experience saltwater intrusion (established as a no-salt site) at the University of Maryland Lower Eastern Shore Research and Education Center (LESREC) near Quantico, MD in Wicomico Co. (38.3549° N, −75.7768°W). Soil at Farm 1 consisted primarily of Queponco silt loam (mesic Typic Hapludults). At Farm 6, soils consisted mainly of Elkton silt loam (mesic Typic Endoaquults), while the soils at Farm 3 were a primarily a mixture of Fallsington sandy loam (mesic Typic Endoaquults) and Othello silt loam (mesic Typic Endoaquults). At the no-salt site, soils were Mattapex silt loams (mesic Aquic Hapludults). Baseline soil properties are shown in Table 1.

2.2. Land use history

The study region was heavily forested from the mid 1600 s to the early 1700 s, with only 5 % of the land in low-intensity agriculture (Benitez and Fisher, 2004). By the 1800 s, the landscape changed dramatically and about 80 % of the land was cultivated in tobacco and wheat and very little primary forest remained (Benitez and Fisher, 2004). Currently, the typical crop rotation in the region is corn (*Zea mays*)-soybean (*Glycine max*)-wheat (*Triticum aestivum*), and no-till techniques have been utilized for the last 40 yrs (Huggins and Reganold, 2008).

The sites we selected for our study have different histories of land use management, which reflects the variability in how farmers respond to SWI. Farm 1 has been in agriculture since the 1950 s, and in a corn-soybean-winter wheat cover crop rotation since 2015. The study area has had salt damage since 2000, and when the field was planted in corn in 2018, the yield suffered substantially. While the field did not receive gypsum, the farmer applied dry lime every 2–3 years (~1121 kg ha^{-1}) at a variable rate.

Farm 3 was first cultivated in the 1950 s as a vegetable farm (e.g. tomatoes, string beans). From 1970–2000, soybeans were primarily planted in the field. The farmer first noticed signs of salt damage in the mid 1990 s along the field edge. Around the year 2000, the farmer

Table 1
Soil characteristics of field sites on the Lower Eastern Shore of Maryland collected from 0 to 10 cm in 2018.

Site	Location	Texture	Baseline SOM 0–10 cm (%)	Baseline bioavailable P 0–10 cm (mg/kg)	Clay (%)	Silt (%)	Sand (%)	Taxonomic Classification
Farm 1	Somerset Co., MD *Nearest town:* Princess Anne	Queponco silt loam	2.1 ± 0.4	284 ± 73	16.74	38.31	44.95	mesic Typic Hapludults
Farm 3	Somerset Co., MD *Nearest town:* Crisfield	Othello-Fallsington complex sandy loam	2.7 ± 0.5	275 ± 16	8.50	21.83	69.83	mesic Typic Endoaquults
Farm 6	Dorchester Co., MD *Nearest town:* Cambridge	Elkton silt loam	2.3 ± 0.3	130 ± 25	23.57	71.59	4.84	mesic Typic Endoaquults
LESREC (Control)	Wicomico Co., MD *Nearest town:* Quantico	Mattapex silt loams	2.0 ± 0.2	32 ± 8	22.93	64.75	12.32	mesic Aquic Hapludults

switched to planting sorghum (*Sorghum bicolor*). The field was last planted in sorghum in 2016.

Farm 6 was cleared for agriculture in 1987 for corn and soybean production. Around 1989, the farmer started growing sorghum every 3–4 years, which he found to be more salt-tolerant than corn. The field flooded during Superstorm Sandy in 2012, and the field never recovered from the salt damage. The farmer attempted to grow soybeans and sorghum, but eventually, parts of the field were abandoned although still mowed regularly to control weeds.

The no-salt site was established in 1969. Before its establishment, the fields were under grain or vegetable cultivation since 1955. Prior to 1955, the land was likely cultivated, but the land history is unavailable. The field where our research plots were located was under conventional management in a tilled corn soybean-wheat rotation until from 1988 to 2011 when it was converted to no-till corn-wheat-soybean-barley rotation with sorghum occasionally planted instead of corn.

2.3. Experimental design

Plots in the salt-damaged fields were arranged in a randomized complete block design planted with four replicate blocks of six treatments: (1−3) salt-tolerant crop rotation (chloride-excluding soybean, *Glycine max L. Merr*; sorghum, *Sorghum bicolor L.*; and barley, *Hordeum vulgare L.*), (4) perennial remediation species (switchgrass, *Panicum virgatum*), (5) perennial restoration species (saltmarsh hay, *Spartina patens*), and (6) abandoned (natural recruitment weeds). Each plot was 3 m wide and 20 m long to capture a natural salinity gradient from the edge of the field (visible saltwater intrusion; 0–5 m from the agricultural ditch) to the center of the field (no visible saltwater intrusion; 15–20 m) (Fig. 2). There was a 0.5 m buffer between plots and a 1 m buffer between each block. This work focuses on data from the remediation, restoration, and abandonment treatments (4, 5, and 6 above).

The no-salt field site had a layout of 16 plots with the same treatments (1−3) and a salt-tolerant crop rotation (chloride-excluding soybean, sorghum, and barley), and a remediation species (treatment 4; *P. virgatum*). As the field was not affected by saltwater, we did not install the restoration species (treatment 5; *S. patens*). Additionally, we did not allow for natural recruitment of weeds (treatment 6; native grasses), in order to maintain the integrity of the fields for future researchers. It is important to note, the no-salt site was examined as a research site in itself. That is, we examined the changes in soil P pools (and concentrations) with depth in the remediation strategy over time. We only examined how plant P pools changed across study years. The no-salt site allowed us to measure remediation biomass production potential in the absence of salt.

2.4. Field activities

Both *P. virgatum* and *S. patens* plants were grown from seed in plug trays in the University of Maryland greenhouse until they reached a size of about 20 cm. The seedlings were then planted by hand in six rows with 0.5 m between rows and 0.5 m between plants for a total of 240 plants per plot. Farm 3 and Farm 6 were hand-planted in June of 2018 and were gap-filled with new seedlings in the following year in June where plants did not survive. Also, in June of 2019, we hand planted *P. virgatum* and *S. patens* at Farm 1 and only *P. virgatum* at the no-salt site and gap-filled in April of 2020. This process was repeated in 2020 as necessary to ensure good stands of our perennial treatments.

2.5. Soil collection and analysis

In 2018, soil bulk density was collected from each block at Farm 6 and Farm 3 using a 5-cm diameter x 15-cm deep core and an AMS compact slide hammer (Core Sampler Complete, AMS, American Falls, ID, USA). Soil cores were collected for bulk density at 0–10, 10–20, 20–30, and 30–60 cm depth intervals. Soil samples were collected for

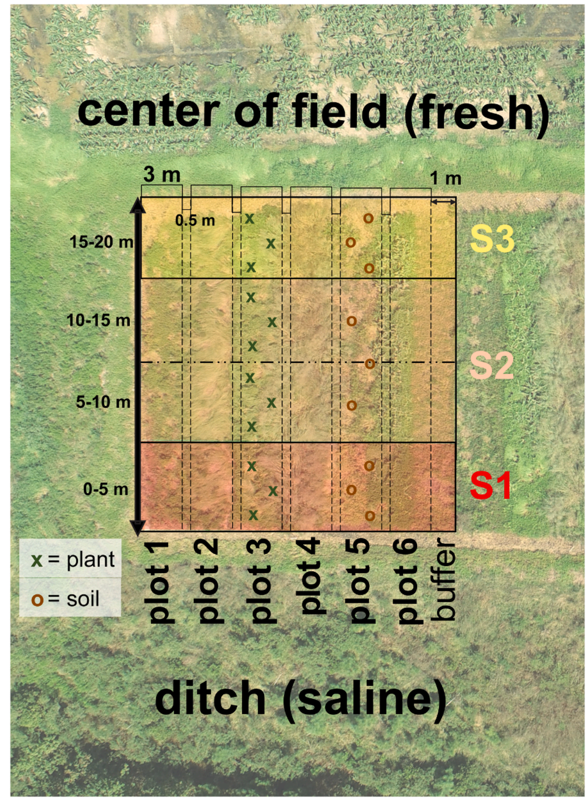

Fig. 2. Sampling design for soil and plant biomass in an example replicate block of one saltwater-intruded field site on the Lower Eastern Shore of Maryland. Soil cores (represented by brown o) were collected from each plot to 60 cm (0–10, 10–20, 20–30, 30–60 cm; three replicates for each depth) at 5, 10, and 15 m from the edge of the field (S1–3). Plant biomass samples (represented by green x) were collected from three replicate 0.25 m^2 quadrats in four subplots: 0–5, 5–10, 10–15, and 15–20 m from the brackish water ditch that borders the study sites to capture the effects of salinity (S2 split into two sections; 5–10 and 10–15 m, respectively).

bulk density in 2019 from each block at Farm 1 by digging pits to 55 cm and a 5-cm aluminum core was pushed horizontally into the side of the pit at 0–10, 10–20, 20–30, and 30–60 cm. Three cores were taken at each depth increment. In both cases, soils were returned to the University of Maryland - College Park and dried at 105ºC for 7 days. Sample dry weight and core volume were used to calculate bulk density (g cm^{-3}). Soil bulk density at the no-salt site was measured using the same process in September of 2021.

Baseline soils were collected from Farm 6 and Farm 3 in March of 2018 by compositing three cores taken at each depth (0–10 cm, 10–20 cm, 20–30 cm, and 30–60 cm) in each subplot using a 22-mm diameter push probe (AMS, Idaho Falls, ID, USA). Baseline soils were collected from Farm 1 and the no-salt site in November of 2018 using the same method as above. Texture was determined using the hydrometer method (Gee and Bauder, 1979; Table 1).

Starting in November 2019, we collected annual soil samples at the plot-level using the same depths as above (0–10, 10–20, 20–30, 30–60 cm; three replicates for each depth) at 5, 10, and 15 m from the edge of the field (Fig. 2). Since the no-salt site did not have a salinity gradient, we sampled only at the plot-level.

Annual soil samples were air-dried for at least three days before homogenization by grinding through a 2-mm mesh sieve. Ground soils were oven-dried (at 60 ºC) for two days prior to analysis. Each depth was analyzed for texture (hydrometer method; Gee and Bauder, 1979), total P (modified Kjeldahl digestion; Bradstreet, 1954), and bioavailable P (Mehlich-3 extraction; Mehlich, 1984). Soils were digested at 160 ºC with sulfuric/salicylic acid solution to break the carbon bonds between organic P compounds, which converts organic elements to a measurable inorganic form (e.g., total P; Bradstreet, 1954). The digestate solution was analyzed for total P on a LACHAT QuikChem (LACHAT Instruments Loveland, CO) using the molybdate-blue method for PO_4-P (detection limit 0.01 mg PO_4-P L^{-1}; Murphy and Riley, 1962). Bioavailable P was also analyzed using the molybdate-blue method as above. Upon initial inspection, bioavailable P levels in the 30–60 cm depth were low (less than 25 mg kg^{-1}) and similar across site, treatment, and section, so we focused on soil P in the 0–30 cm depths. Levels of P in the soil between years were also similar, but decreasing over time, so we focused on soil P in the first year (Baseline; 2018) and the fourth year of the study (2021).

To calculate soil nutrient pools, the nutrient concentrations (mg kg^{-1}) from each individual depth increment (e.g., 0–10, 10–20, and 20–30 cm) were multiplied by the depth of the segment (in cm) and the bulk density (g cm^{-3}). Ground and dried soil samples were also analyzed for electrical conductivity (EC) in 2018, 2019, 2020, and 2021 in a 1:5 soil to water slurry. Electrical conductivity was measured using a Thermo Scientific Orion Versa Star Pro (Thermo Fisher Scientific, Hampton, NH).

The state of Maryland uses Fertility Index Values (FIV) to provide farmers with a semi-quantitative method of determining if a field has an excess, optimum, or deficient level of a specific nutrient (Coale et al., 2021). The FIV is commonly used throughout the Maryland Cooperative Extension system as farmers are required to submit soils for testing every 3–5 years with their nutrient management plan. As each lab uses a slightly different method for measuring soil P concentrations, the FIV calculation allows for comparison among labs and soils across the state. For example, in the case of soil test P, the FIV helps Extension agents determine if a field is at a low, medium, or high risk of leading to P loss and eutrophication of water bodies (Coale et al., 2021; MDA, 2021). That is, low risk refers to soil that is inadequate for optimum growth of most crops and fertilizer addition would likely be completely taken up by crops, where high risk refers to soil that is more than adequate for optimum growth, but addition of nutrients could runoff into nearby waterbodies (MDA, 2021).

Each lab calculated specific factors (X and Y) used for converting regional soil-testing laboratory report data to Maryland FIV scale that sets the highest P concentration within the "optimum" range (< 100–150) to an FIV of 100 to make the values comparable (Eq. (1)). To calculate FIV in the topsoil (0–10 cm; sections were averaged together), Mehlich-3 extractable P concentrations (in mg kg^{-1}) were multiplied by 1.20 and added by 3 (Brookside Laboratories conversion factors; Eq. (2)) found in the Soil Fertility Management Table 2 (Coale et al., 2021).

$$FIV\ 100 = \left(optimum\ P\ concentration\ \left[mg\ kg^{-1}\right] \times X\right) + Y \tag{1}$$

$$FIV = \left(P\ concentration\ \left[mg\ kg^{-1}\right] \times 1.20\right) + 3 \tag{2}$$

2.6. Porewater collection and analysis

Porous cup lysimeters (22 mm diameter; Soil Solution Access Tubes, Irrometer Riverside, California, USA) were installed in March 2018 (Farm 3, Farm 6) and October 2018 (Farm 1) to 60 cm depth at 5 m (visible saltwater intrusion) and 15 m from the edge of the plot (no visible saltwater intrusion). Lysimeters were installed by removing soil to 60 cm with a 22-mm diameter soil probe (AMS, Idaho Falls, ID, USA). Lysimeters were sealed at the soil surface with a bentonite/clay mixture (Benseal, Halliburton, TX, USA). Pilot studies confirmed that soil solution collection was only possible following rain events that were greater than or equal to 6 mm, thus soil solution was only collected following rain events of this level. Samples were collected at roughly 2-week intervals from March through November of each year (2018–2020; n = 2610 samples). The day before sampling, lysimeters were purged of any water, and an internal pressure of −60 to −70 kPa was applied. Soil solution samples were collected, filtered (1 µm glass fiber filters), and stored in a freezer at −20 °C until further analysis.

Electrical conductivity (EC) was measured using a Thermo Scientific Orion Versa Star Pro (Thermo Fisher Scientific, Hampton, New Hampshire, USA). A subsample of the filtered solution was acidified using 10 % hydrochloric acid (HCl) solution (150 µL 10 % HCl in 15 mL solution). This sample was analyzed colorimetrically on a LACHAT QuikChem (LACHAT Instruments, Loveland, CO, USA) using the molybdate-blue method for PO_4-P (detection limit 0.01 mg PO_4-P/L; Murphy and Riley, 1962). Porewater P concentrations were similar across each site and distance from the ditch, so we focused on change over time under each of the treatments.

2.7. Plant collection and analysis

Biomass was collected from the weeds, *S. patens*, and *P. virgatum* plots from the three saltwater-intruded sites (Farm 1, Farm 3, Farm 6) in August 2018, 2019, 2020, and 2021. It is important to note that there were other species growing in the plots in addition to our target species. All aboveground biomass was collected from three replicate 0.25 m^2 quadrats in four subplots: 0–5, 5–10, 10–15, and 15–20 m from the tidal ditch at the field edges of all sites (Fig. 2). *P. virgatum* was collected from the no-salt site at the plot level. Biomass samples were clipped within 2 cm of the soil surface, sorted by species, and then oven-dried for at least 48 h at 60 °C and weighed. Biomass of each species was summed at the quadrat-level and scaled to kg ha^{-1} to measure treatment effects on plant biomass. Each year, we determined the dominant weed species after processing. In 2018, fall panicgrass (*Panicum dichotomiflorum*), a native, annual switchgrass, was a species common to nearly all weed plots across all sites and, therefore, was retained for tissue analysis. In the following years (2019–2022), the most common species was hairy crabgrass (*Digitaria sanguinalis*), so this species was retained for tissue analysis.

Oven-dry (60 °C) plant sub-samples from each treatment (target species) and year were homogenized by grinding through a 2-mm mesh screen (Wiley Mill, Swedesboro, New Jersey) and analyzed for total P. Plant tissue samples were digested as above (modified Kjeldahl; Bradstreet, 1954). The digestate solution was analyzed for total P using colorimetry on a LACHAT QuikChem (LACHAT Instruments Loveland, CO) using the molybdate-blue method for PO_4-P (detection limit 0.01 mg PO_4-P L^{-1}; Murphy and Riley, 1962). Biomass P pools, also known as aboveground P stocks, were calculated by multiplying nutrient concentrations (mg kg^{-1}) by the aboveground biomass (dry-weight equivalent [DWE]; g m^2) collected from the plots.

Results are displayed in terms of study year rather than a calendar year. For example, the first year of growth is denoted study year 1 and includes 2018 data from Farm 3 and 6. Year 1 included 2019 data for Farm 1 and the no-salt site as they were established later.

2.8. Statistical approach

A linear mixed-effects (LME) model (*lme4* package for R; Bates et al., 2013) was used to examine the effect of management strategy (e.g., remediate, restore, abandon) on soil P concentrations and pools averaged at the plot-level. We examined each study year and site individually with management strategy (treatment), section (distance from the ditch), and depth (0–10, 10–20, and 20–30 cm) as the main effects and block as the random effect. Once we determined there was a depth effect (and the interaction between the main effects were not significant), we examined each study year, site, section, and depth individually (e.g. soil bioavailable P in 2021 from Farm 3 in S1 at 0–10 cm depth). The no-salt site was also examined separately, but since there was no section (because there was no salty ditch) and for this study, we only examined the remediation strategy, we conducted one LME per study year, examining how soil P pools changed with depth. Soil P concentrations were measured in 2018 (to understand baseline P levels) and 2021. Tukey *post-hoc* comparisons were used to determine significant differences of the main effects using the *multcomp* and *stats* packages (Hothorn et al., 2016, R Core Team, 2023). Finally, univariate regression was used to determine if there is a relationship between topsoil EC (0–10 cm) and topsoil P concentrations.

A linear mixed-effects (LME) model (*lme4* package for R; Bates et al., 2013) was used to examine the effect of management strategy (e.g., remediate, restore, abandon) on porewater P concentrations averaged at the plot-level. We examined each study year and site individually with management strategy (treatment) and section (saltwater intrusion) as the main effects and block as the random effect, however, there were no significant differences among saltwater-intruded sites. We determined that there was no significant interaction between treatment and site, so we averaged the saltwater-intruded sites together. If we found that there was no significant interaction between treatment and distance from the ditch, we determined the significance of treatment and section on porewater P concentrations individually. Tukey *post-hoc* comparisons were used to evaluate significant differences among management strategies.

A linear mixed-effects (LME) model (*lme4* package for R; Bates et al., 2013 was used to examine the effect of management strategy (e.g., remediate, restore, abandon) on leaf tissue P concentrations and pools averaged at the plot-level. We examined each study year and site individually with management strategy (treatment) and section (saltwater intrusion) as the main effects and block as the random effect. If we found that there was no significant interaction between treatment and distance from the ditch, we analyzed the significance of treatment and section on leaf tissue P concentrations and pools individually. The no-salt site was also examined separately, but since there was no section (because there was no salty ditch) and for this study, we only examined the remediation strategy, we only compared plant P pools across the study years. Tukey *post-hoc* comparisons were used to evaluate significant differences of the main effects using the *multcomp* and *stats* packages (Hothorn et al., 2016, R Core Team, 2023). Finally, univariate regression was used to determine if there is a relationship between soil and plant tissue P concentrations and pools. All statistics were run in the R environment for Mac (R Core Team, 2023).

3. Results

3.1. Soil available P pools

Soil Mehlich-3 extractable P (M3P) pools decreased by 30–50 % in the topsoil at two sites after three years of growth (33.9 % at Farm 3, 52.9 % at Farm 6; Fig. 3; Table S1). There was a significant effect of distance from the saline ditch at Farm 3 and Farm 6, where the section closest to the ditch (0–5 m) had higher M3P levels than the section furthest from the ditch (15–20 m) at Farm 6 ($p < 0.05$; Table S2). In Farm 3, we observed the opposite, with higher M3P levels in the section closest to the saline ditch (0–5 m; $p < 0.01$; Fig. 3; Table S2).

There was a significant effect of soil depth on soil M3P pools at all farms ($p < 0.01$; Table S2). In 2018, soil M3P pools in the 0–10 cm depth had the highest M3P pools, the next 10 cm (10–20 cm) had intermediate levels of M3P pools, and finally the deeper depth (20–30 cm) had the lowest M3P pools ($p < 0.01$ in all cases; Table S2). In 2021, soil M3P pools in the topsoil (0–10 cm) were greater than the 10–20 cm depth and the 20–30 cm depth ($p < 0.01$ in all cases; Table S2). We observed a decrease in soil M3P after three years of growth in all soil depths with a few exceptions in the 20–30 cm depth under all treatments at Farm 1 and 3 ($p < 0.05$; Table S2).

Farm 3 had the highest M3P levels across the saltwater-intruded farms ($p < 0.01$; medium risk group; FIV = 425), and Farm 6 had the lowest M3P levels of all farms ($p < 0.01$; low risk group; FIV = 214). Farm 1 had similar M3P levels to Farm 3 in the topsoil in year 1 ($p < 0.01$; medium risk group; FIV = 365). The FIV decreased at Farm 1 by 33 %, at Farm 3 by 37 %, and at Farm 6 by 54 % under all treatments (Fig. 7), however, there were no significant differences among the treatments in the soil (Table S2). Under remediation, restoration, and abandonment treatments at Farm 6, FIV levels in the soil were reduced to reach optimum levels (< 100–150).

3.2. Soil total P pools

Soil P pools decreased by approximately 20 % in the topsoil at two sites over the three study years (Farm 1 and Farm 6; Fig. 4). There was a significant effect of distance from the ditch at all SWI sites, where the

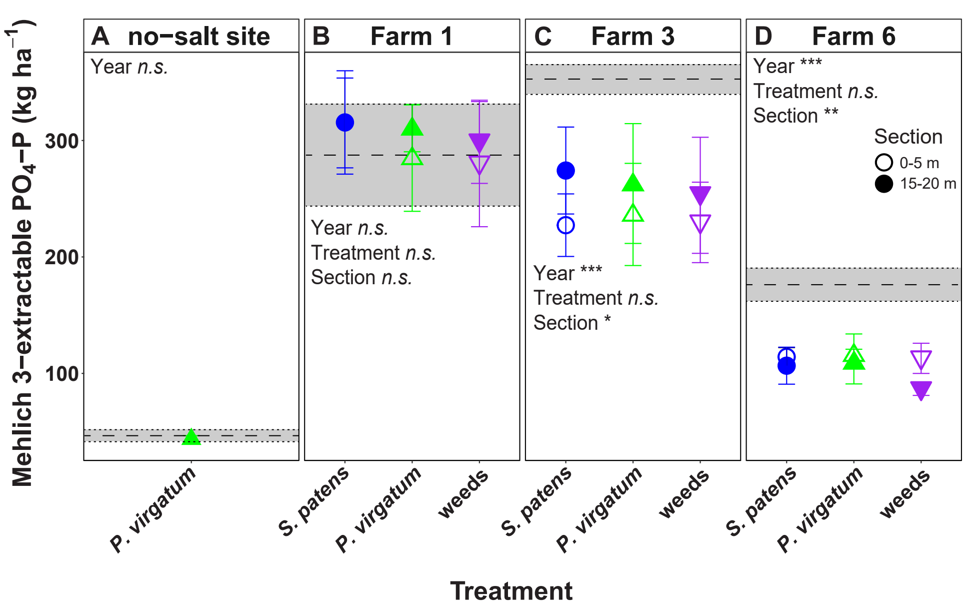

Fig. 3. Mehlich 3-extractable soil P pools (kg ha^{-1}) in topsoil (0–10 cm) at three saltwater-intruded field sites and one no-salt site on the Lower Eastern Shore of Maryland in 2018 (baseline) and 2021. Data from the saltwater-intruded sites are presented by treatment (colored symbols) and section (open symbols indicate samples close to the ditch and closed symbols far from the ditch). Baseline data are represented by the gray box, where the black dashed line is the mean and the black dotted lines are the standard error of the mean. Error bars are standard error of the mean. Statistical significance (*p-value*) is indicated by symbols * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.005$.

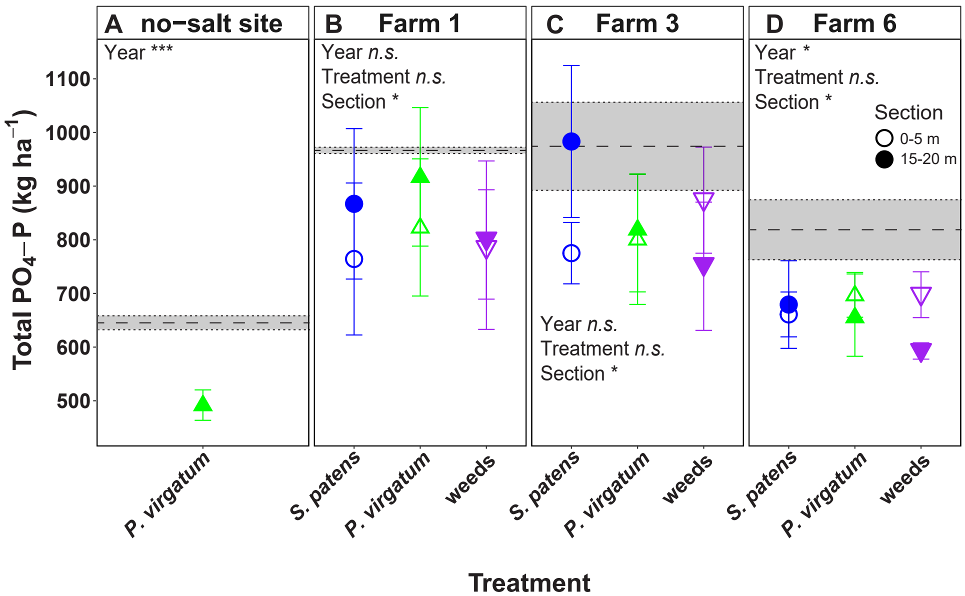

Fig. 4. Soil total P pools (kg ha^{-1}) in topsoil (0–10 cm) at three saltwater-intruded field sites and one no-salt site on the Lower Eastern Shore of Maryland in 2018 (baseline) and 2021. Data from the saltwater-intruded sites are presented by treatment (colored symbols) and section (open symbols indicate samples close to the ditch and closed symbols far from the ditch). Baseline data are represented by the gray box, where the black dashed line is the mean and the black dotted lines are the standard error of the mean. Error bars are standard error of the mean. Statistical significance (*p-value*) is indicated by symbols * = $p<0.05$, ** = $p<0.01$, *** = $p<0.005$.

section furthest from the ditch (15–20 m) had higher P pools than the section closest to the ditch (0–5 m) at Farm 1 and Farm 3 ($p<0.05$; Fig. 4); the opposite was observed at Farm 6 ($p<0.05$; Fig. 4; Table S3). There was a significant effect of soil depth on total P pools in baseline soils, where total soil P pools were highest in the 0–10 cm depth intermediate at 10–20 cm, and smallest at 20–30 cm depth ($p<0.01$ in all cases; Table S3). After three years of growth, there was still a significant effect of depth on total soil P pools, although the size of the pools was smaller across the soil profile (Table S3).

We found similar patterns in soil available and total P concentrations among the three saltwater-intruded sites across all study years where the both the soil closest to the ditch and the soil furthest from the ditch were strongly positively correlated with topsoil EC (0–10 cm; $r^2 > 0.5$ in year 1 and r2 > 0.2 in year 4; $p<0.001$ and $p<0.01$, respectively; Table S4).

3.3. Porewater P concentrations

After two years of growth, there was an overall decrease in porewater soluble reactive P (SRP) concentrations under all treatments. There was not a significant interactive effect of treatment by site among the saltwater-intruded sites, therefore we were able to average the concentrations from each treatment from all of the sites together (Fig. 5). There was also no effect of distance from the agricultural ditch on SRP concentrations at any farm in any year of the study. Overall, there was not a strong treatment effect on porewater SRP concentrations. However, in study year 2, we observed higher concentrations of SRP in the

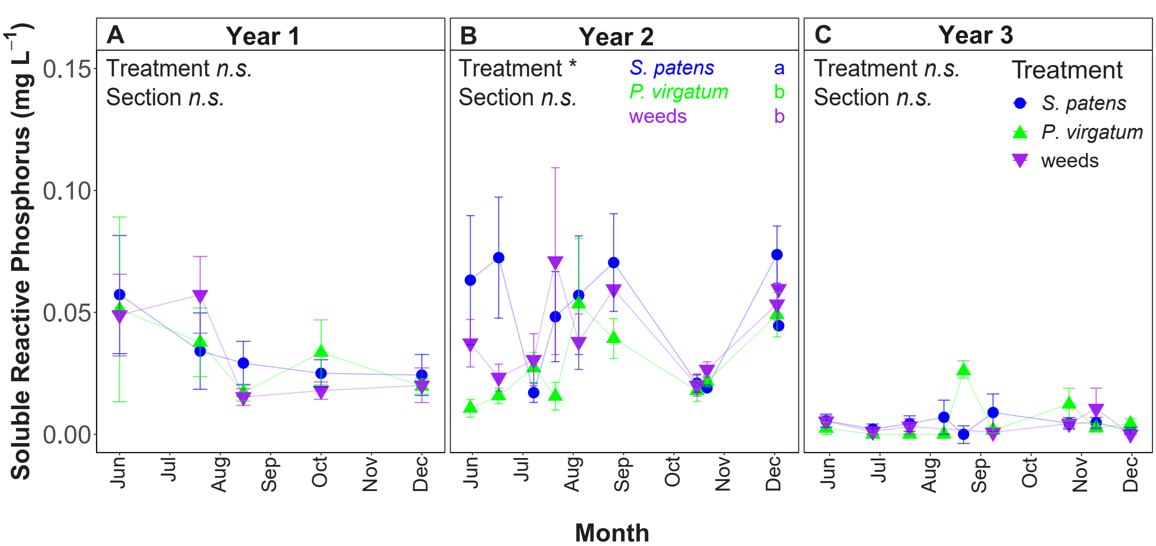

Fig. 5. Soluble reactive phosphorus concentrations (mg L^{-1}) in soil porewater in study years 1-3 by treatment (colored symbols). The saltwater-intruded sites pooled means were plotted because the interaction between porewater P and site was not significant. Error bars are standard error of the mean. Statistical significance (*p-value*) is indicated by symbols * = $p<0.05$, ** = $p<0.01$, *** = $p<0.005$.

S. patens treatment compared to the weeds and *P. virgatum* treatments ($p < 0.05$; Fig. 5). Porewater SRP concentrations differed significantly through time ($p < 0.01$). Overall, porewater SRP concentrations ranged from 0.01 to 0.10 mg L^{-1} and decreased significantly from study year 1 to year 3 (< 0.01 mg L^{-1}; $p < 0.01$; Fig. 5). There was a slight increase in SRP concentrations under all treatments from the end of study year 1 (November) to the start of year 2 (June, July, August), which decreased in October/November of year 2 and increased again in December. In May of study year 3, SRP decreased to below 0.01 mg L^{-1}.

3.4. Trends in plant P concentrations and pools

We found similar patterns in plant leaf tissue P concentrations among the three saltwater-intruded sites across all study years where the highest P concentrations were always found in weed tissue and the lowest P concentrations were found in *S. patens* tissue with intermediate concentrations in *P. virgatum* tissues ($p < 0.03$ in all cases; Fig. S3; Table S5). There was no clear relationship between plant leaf tissue P concentrations and distance from the saline ditch ($p < 0.01$; Fig. S3; Table S4).

Plant biomass drove patterns in plant pools of P across sites, the SWI gradient, and study years. In the first study year, at Farm 1 and Farm 3, where weeds accumulated the largest biomass, weed P pools were the highest compared to *P. virgatum* and *S. patens* ($p < 0.003$; Fig. 6). This trend continued into study year 2 at Farm 1, however at Farm 3 and Farm 6, *P. virgatum* biomass levels were highest, therefore *P. virgatum* P pools were highest compared to *S. patens* and weeds ($p < 0.01$; Fig. 6; Fig. S4). In the third and fourth years of growth, *P. virgatum* P pools were the highest at all saltwater-intruded field sites ($p < 0.001$; Fig. 6). At Farm 1 and Farm 6, *P. virgatum* biomass levels were highest compared to *S. patens* and weeds in year 3 and 4 ($p < 0.01$; Fig. S4), however at Farm 3, *S. patens* biomass was highest in study years 3 and 4. Despite *S. patens* having higher biomass levels than *P. virgatum* at our saltiest field site, *P. virgatum* P pools still had higher P pools than *S. patens* (although they were both higher than weeds; $p < 0.01$; Fig. 6; Fig. S4).

In study years 1 and 2 at Farm 6, biomass increased with distance from the ditch ($p < 0.01$ in both cases; Fig. S4). However, at Farm 1 and Farm 3, there was no significant effect of distance from the ditch on aboveground plant biomass for any of the study years.

At the no-salt site, we only planted *P. virgatum*, which had similar plant tissue P concentrations and biomass levels as the saltwater-intruded sites (Fig. S3; Fig. S4; Table S5). Aboveground biomass P pools in the no-salt site showed a similar pattern to Farm 1, where *P. virgatum* P pools were low in study years 1 and 2 but increased in study year 3 (Fig. 6; Table S5).

We found similar patterns in soil available and total P concentrations among the three saltwater-intruded sites across all study years where both the soil closest to the ditch and the soil furthest from the ditch were strongly positively correlated with each other (0–10 cm; $r^2 > 0.8$; $p < 0.001$ in all cases; Table S6). There were no clear relationships between plant leaf tissue P and topsoil available and total soil P pools (Table S6).

4. Discussion

4.1. Panicum virgatum is efficient at taking up P

We found *P. virgatum* biomass (remediation strategy) had higher P pools than both *S. patens* (restoration strategy) and weeds (abandon strategy) in saltwater-intruded fields, suggesting that *P. virgatum* could be used as a P removal tool. Previous research has shown *P. virgatum* can accumulate large quantities of biomass after the establishment year, and once mature, can produce consistent biomass for many years (McLaughlin and Adams Kszos, 2005; Fike et al., 2006; Wullschleger et al., 2010; Richner et al., 2014). We found that high levels of *P. virgatum* biomass were associated with large aboveground P pools, and soil total and available P pools decreased after three years of growth (Figs. 5,6,7), which indicates that species with high levels of biomass can uptake large amounts of soil P (Schmer et al., 2011; Basyal and Emery, 2021). Although not specific to *P. virgatum,* we observed declines in porewater P concentrations under all management strategies declined throughout the course of the study, likely because plant P uptake reduced soil P pools, leaving less available P to be released as SRP (Meyerson et al., 1999). Our data suggest that planting *P. virgatum* on coastal agricultural fields could be a remediation strategy to remove soil P, if harvested, and may ultimately decrease the amount of P available to runoff into the Chesapeake Bay (Schmer et al., 2011; Ashworth et al., 2017; Rivera-Chacon et al., 2022).

There are additional benefits to implementing a remediation strategy like planting *P. virgatum* on salt-intruded fields. *P. virgatum* is a perennial grass that does not need to be replanted each year with relatively low seed cost and no need for fertilizer amendments (Agricultural Marketing Resource Center AGMRC, 2018; MDA, 2021), incurring lower annual input costs than annual crops like corn and soy. *P. virgatum* can be used as a cellulose biofuel or poultry bedding. Thus, alongside P removal, landowners could continue to make a profit from planting, although at a reduced price per acre than typical corn and soybean crops in the current market ($265 per acre compared to $749 and $402 per acre, for corn and soybean respectively; Moyle et al., 2016; Purswell et al., 2020; University of Maryland UMD Extension, 2022). As a source of bioenergy, planting *P. virgatum* could have both environmental and economic benefits including improved soil and water quality by removal of excess P and increased net economic returns in areas where other crops are not as easily grown due to high salinity in the soil (Romm et al., 1998; McLaughlin and Adams Kszos, 2005). Future research will focus on *P. virgatum* as co-feedstocks for anaerobic digestion of poultry litter to track greenhouse gasses, soil carbon, nutrients, and energy flows from the field to the final digestion products.

4.2. Spartina patens is efficient at taking up P in soil with high salinity

We found that the *S. patens* (restoration strategy) had intermediate biomass P pools compared to other strategies from saltwater-intruded fields, but still showing potential to be used as a buffer to prevent P loss. Previous research indicated that fields experiencing SWI will naturally transition to host more native marsh species such as *S. patens* (Gedan and Fernández-Pascual, 2019). Our work showed that the increase in *S. patens* biomass and aboveground P pools provided an opportunity to reduce P pools in the soil. At Farm 3 where salinity was the highest, *S. patens* had the highest biomass levels because of the higher salt tolerance (Fig. S4). The uptake of P in *S. patens* biomass was associated with decreased levels of soil P pools and porewater SRP concentrations (Figs. 5,6,7). Similar to *P. virgatum*, we showed that restoring (or facilitating the conversion to) natural wetlands can serve as a tool for reducing P losses to nearby waterbodies by utilizing their high root porosity to maintain nutrient uptake (Burdick and Mendelssohn, 1990; DeLaune et al., 2005; Barbier et al., 2011; Crosby et al., 2017).

S. patens is a practical option for planting or allowing to recruit naturally in high salinity soils on Maryland's Eastern Shore. First, *S. patens* is a halophytic C4 grass that can create a new wildlife habitat for waterfowl hunting and aid in restoring the land to native marsh. In addition to maintaining a profit by allowing hunting activity, landowners can receive financial assistance and guidance from state- and federal-level conservation programs that can support restoration practices (e.g., Conservation Reserve Enhancement Program; CREP, 2022, Environmental Quality Incentives Program; EQIP, 2021, Wetland Reserve Easements; WRE, 2022). Finally, planting *S. patens* may facilitate the transition to tidal marshes that can protect coastlines from flooding by reducing strong winds and storm surges (Guimond and Michael, 2020) with their strong stems and other physical properties (Hu et al., 2015; Leonardi et al., 2018). The restoration approach has been shown to save landowners in Delaware, Maryland, Virginia

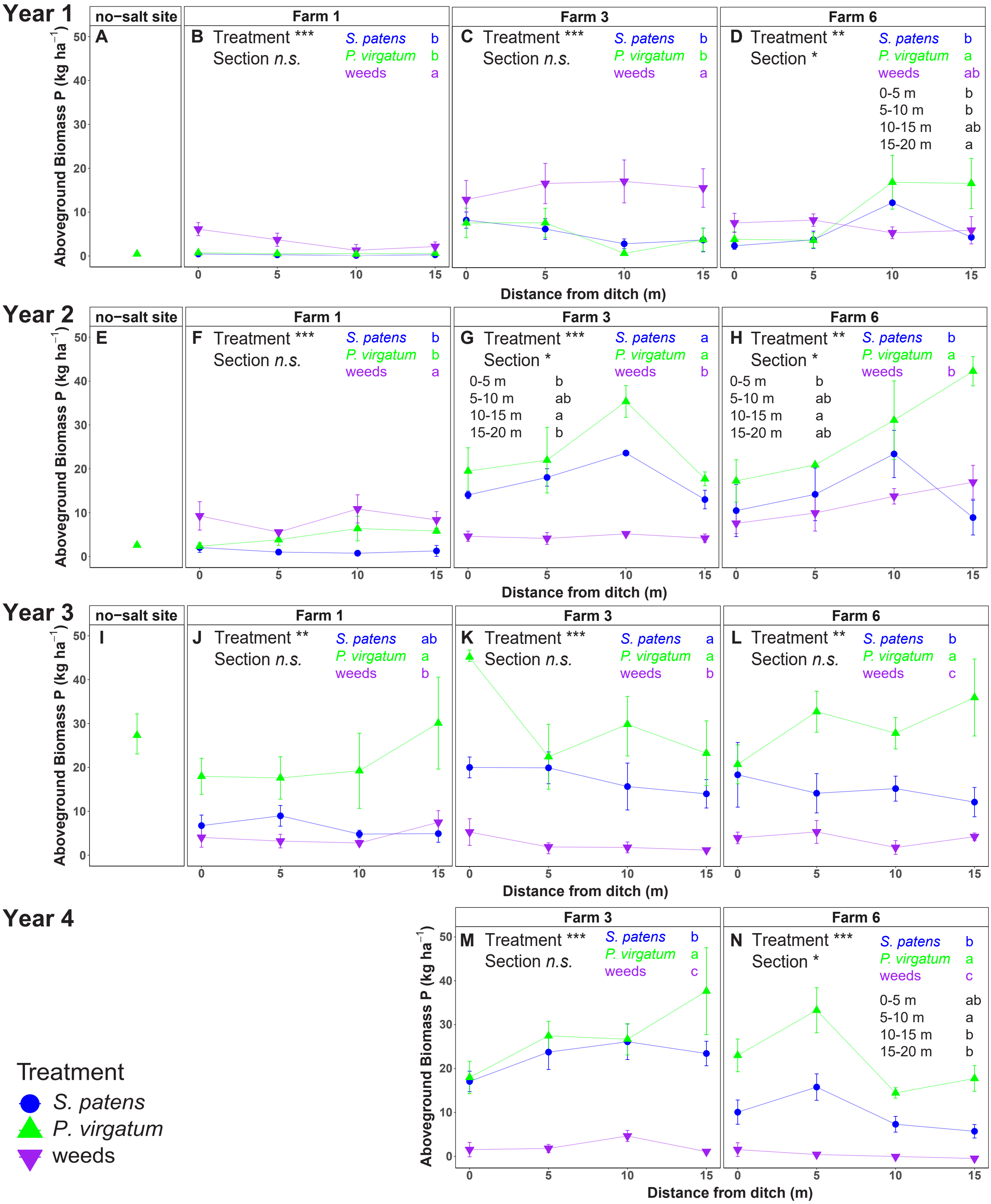

Fig. 6. Plant aboveground biomass P pools (kg ha^{-1}) by distance from salty ditch at three saltwater-intruded farms and one no-salt site on the Lower Eastern Shore of Maryland in study year 1 (a-d), year 2 (e-h), year 3 (i-l), and year 4 (m-n) by treatment (colored symbols). Farm 3 and 6 were established in 2018. Farm 1 and the no salt site were established in 2019. We present panels by study years as it is easier to compare how biomass accumulated P with each year since planting. Error bars are the standard error of the mean. Statistical significance (*p-value*) is indicated by symbols $* = p < 0.05$, $** = p < 0.01$, $*** = p < 0.005$.

between $4.5–23.8 million annually. For example, in Maryland, coastal wetlands reduced property damages from $20 million to $15.5 million from severe weather and flooding (Li et al., 2020; Narayan et al., 2017; Smith and Katz, 2013; United States Global Change Research Program USGCRP, 2017). The presence of salt marshes reduces average annual flood losses by 18–70 % (Narayan et al., 2017). In sum, restoring farm fields to native marshes could benefit (1) nearby waterbodies by reducing soil P levels, (2) landowners by maintaining revenue, and (3) communities by protecting against coastal flooding.

4.3. Allowing weeds to grow at low-risk farms can reduce FIV to optimum levels

Topsoil M3P levels declined under all strategies, however the soils under the abandonment strategy (weeds) had the largest decrease in M3P over the study period at all three SWI sites (Fig. 7). There was a decline in weed biomass, likely because the dominant species changed over time from a high-biomass fall panicgrass, *Panicum dichotomiflorum*, to a more salt-tolerant, smaller stature plant, *Digitaria sanguinalis*, which is short-statured (<1 m in height) (Uva et al., 1997; Hilty, 2012; Minnesota Environment and Natural Resources, 2019). It is important to note that we only measured the P concentration of the dominant species; however, there were other weed species present in the plot. All strategies (remediate, restore, abandon) reduced FIV levels (topsoil M3P) in saltwater-intruded farms. At Farm 1 and Farm 3, soil M3P was so high, we were unable to reduce the FIV to optimum levels (< 100–150), although FIV levels decreased a significant amount after three years of growth. As seen at Farm 6, at low-risk farms (FIV = 150–299), implementing any of the strategies (remediate, restore, abandon) can reduce FIV levels to optimum (FIV < 100–150) after at least three years, as lower initial soil P levels tend to drawdown P more quickly over time (Fiorellino et al., 2017). Allowing plants time to establish and not applying more fertilizer, biomass will draw down existing soil P and reduce excess P loss (Kibet et al., 2016, Kumar et al., 2019).

Some landowners may not want to invest the time or money in salt-damaged lands. Therefore, land abandonment may be the most pragmatic strategy and one that may still confer ecosystem benefits. For example, *Digitaria* species are tolerant of saline and drought conditions due to their extensive root system that produce stoloniferous roots from nodes of the stems close to the ground, allowing them to spread quickly both above- and below-ground (NC State Extension, 2020). *Digitaria sanguinalis* is a C4 annual salt-includer that may decrease the concentrations of salts in the soil (Hilty, 2012; Gotcher et al., 2014; NC State Extension, 2020). Allowing weeds to grow may facilitate the transition of the land back to native marsh (Zhang et al., 2012; Kirwan and Gedan, 2019), however it is important to manage invasive species, such as *Phragmites australis,* during the transition period, so landowners could enroll their land in wetland easement programs (Kirwan and Gedan, 2019; University of Maryland UMD Extension, 2022; WREP, 2022). Overall, we found total soil M3P decreased significantly under all strategies, especially under the abandonment strategy, all management practices (remediate, restore, abandon) can reduce FIV levels (topsoil M3P), and low-risk farms can reduce FIV levels to optimum using any of the strategies after at least three years.

4.4. Study limitations and considerations

It is important to note that this study was not an attempt to construct a complete P budget. We acknowledge that the reduction of the soil P pool does not equal the increase in biomass P. This may be due to the fact that other species were present in the plots accumulating P. Further, it is possible that P was leached, however, most of this leaching would have occurred in the first year (Tully et al., 2019b; Weissman et al., 2021), and we did not see leaching down the soil profile (Table S1; Figs. S1-S2). Finally, P could have been lost horizontally across the field (Tully et al., 2019b; Weissman et al., 2021). There is a possibility of P erosion, especially during storm events, or even a high tide event. Perigean tides have been known to flood fields and have been known to drag P off the fields (Sims et al., 1998; Van der Molen et al., 1998; Kleinman et al., 2007; Kristensen et al., 2021), however, this was beyond the scope of this study. In fact, our research group is exploring this very phenomenon under a new project. Nevertheless, the study does show that P can be taken up and stored in focal plant species, suggesting remediation and restoration practices are great management strategies when climate change prevents growth of typical crops.

5. Conclusion

With rising sea levels increasing saltwater intrusion along the Eastern Shore of the Chesapeake Bay, landowners and farmers are in need of new strategies to mitigate the effects of climate change. We show that remediation and restoration management practices are efficient in taking up soil and porewater P that may ultimately decrease P losses to the Chesapeake Bay. After three years of growth, we observed a decrease in available soil P pools and FIV in the topsoil under all three management strategies, aligning with an increase in P uptake within the biomass of *P. virgatum* (remediation) and *S. patens* (restoration). In this study, we showed that remediation, restoration, and abandonment management strategies were able to reduce FIV levels in the soil after three years of growth to medium risk levels when soil P was high initially, and to optimum levels when soil P was medium originally. In sum, planting either *P. virgatum* or *S. patens* would have economic and environmental benefits in mitigating the adverse effects of SWI. Ultimately, land management decisions depend on the landowners' goals. If the goal is to maintain plant production or produce a profit, landowners could plant *P. virgatum*, removing the aboveground biomass after at least three years of growth. This practice will also reduce P on their previously farmed field and reduce the amount of P that may be lost to adjacent waterbodies. If soils are already salinized and regularly experience high tide flooding, landowners may transition their fields using *S. patens*, with the additional benefit of protecting coastlines from flooding and nutrient loss. Finally, allowing weeds to grow may facilitate the transition of the land back to native marsh, so landowners could enroll their land in easement programs, decrease the concentration of P in the soil, and reduce farm-level P losses that contribute to eutrophication.

By determining the efficacy of best management practices, we aim to help farmers keep their land profitable and remove excess P from the soil to promote better water quality. Improved economic and environmental sustainability in turn supports social sustainability by maintaining the viability of rural livelihoods and landscapes. Thus, it is important to understand the trajectories of different land management practices for farmers experiencing SLR and SWI.

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CRediT authorship contribution statement

Alison N. Schulenburg: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Jarrod O. Miller:** Methodology, Data curation, Writing – review & editing. **Keryn B. Gedan:** Conceptualization, Data curation, Methodology, Writing – review & editing. **Danielle Weissman:** Data curation, Formal analysis, Investigation. **Katherine L. Tully:** Writing – review & editing, Validation, Conceptualization, Funding acquisition, Methodology, Project administration, Supervision.

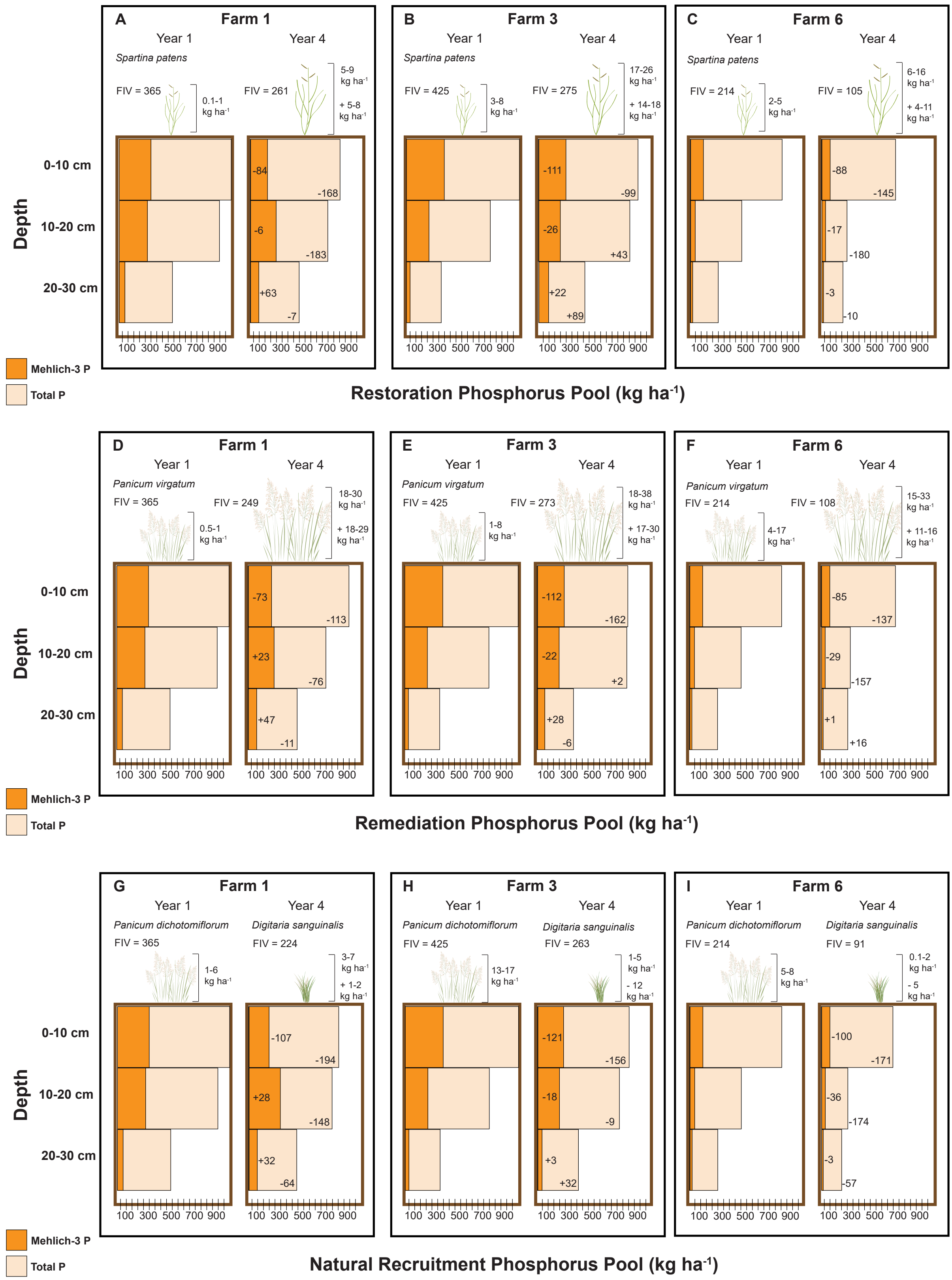

Fig. 7. Conceptual diagram of restoration (*Spartina patens*; A-C), remediation (*Panicum virgatum*; D-F), and natural recruitment (*Panicum dichotomiflorum and Digitaria sanguinalis*; G-I) aboveground biomass P and respective belowground total and available P pools from three depths (0–10, 10–20, and 20–30 cm) in year 1 (baseline) compared to year 4 at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland. The x-axis is P pools in kilograms per hectare and the y-axis is depth in centimeters. Mehlich-3 available P pools are the darker orange bars and total P pools are the lighter bars. Change in Mehlich-3 P from year 1 to year 4 are the values located in the middle of the bars, and the changes in total P from year 1 to year 4 are the values outside the bars on the bottom right ; positive numbers (+) indicate an increase in P pools and negative numbers (-) indicate a decrease in P pools over time. Means were pooled across the salinity gradient and visualized.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data Availability

Data will be made available on request.

Acknowledgements

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2024.109034.

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