ABSTRACT

Title of Thesis: ***MANAGEMENT OPTIONS FOR FARMERS FACING SALTWATER INTRUSION ON THE EASTERN SHORE OF THE CHESAPEAKE BAY***

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Rising sea levels, storms, and perigean spring tides push saltwater into coastal agricultural fields. This phenomenon, known as saltwater intrusion, alters nutrient cycling and damages crop yields. As sea levels continue to rise, saltwater intrusion will only worsen, with devastating consequences to agroecosystems along the coast of the Chesapeake Bay. Researchers and farmers alike are looking for solutions to adapt to and mitigate the effects of saltwater intrusion. Landowners may respond by altering their management practices. Farmers may 1) adapt by planting a salt-tolerant crop, 2) attempt to remediate soils with trap crops, 3) restore native marsh grasses, or 4) abandon fields altogether. My project investigates the survival of different crops and plant treatments under saltwater-intruded conditions and the potential for these plants to survive and to remove excess nutrients (e.g. sodium and phosphorus) from the soil, with the overall goal to benefit both the farming community and water quality in the Chesapeake Bay. Results from this study will help inform new management practices to

increase soil health and maintain crop yields. Finally, the goal of this work is to guide local best management practices and potential easement opportunities for landowners facing saltwater intrusion, and ultimately determine optimal strategies for climate resilience.

MANAGEMENT OPTIONS FOR FARMERS FACING SALTWATER INTRUSION ON THE EASTERN SHORE OF THE CHESAPEAKE BAY

by

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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). Landowners may respond to SWI by altering their management practices. Farmers may 1) *adapt* by planting a salt-tolerant crop (e.g., sorghum or soybean), 2) attempt to *remediate* soils with trap crops (e.g., switchgrass), 3) *restore* native marsh grasses (e.g., saltmarsh hay), or 4) *abandon* fields altogether (e.g., native switchgrass or crabgrass). My work will determine the ability of each of these management practices to remove phosphorus (P) from salt-intruded soils, with the overall goal to benefit both the farming community and water quality in the Chesapeake Bay. We show that remediation and restoration management practices are efficient at taking up soil and porewater P that may ultimately decrease P runoff into the Chesapeake Bay. Overall, patterns of P pools were influenced by biomass levels, and after three years of planting, the uptake of P in switchgrass (remediation species) and saltmarsh hay (restoration species) biomass was consistent with the decrease in available soil P pools (also known as P stocks), reducing the levels of P to potentially be released as soluble reactive P. In sum, planting both switchgrass and saltmarsh hay have economic and environmental benefits that will mitigate the effects of SWI. Results from this study will help inform new management practices to improve nutrient uptake, increase soil health, and

maintain crop yields. Finally, this work will help inform state-level coastal management policies and determine optimal strategies for climate resilience.

Introduction

Sea level rise (SLR) poses a threat to coastal regions. Globally, sea level is projected to rise by 0.8 to 1.6 feet (0.24 to 0.49 m) by 2050 and 1.2 to 3.0 feet (0.37 to 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 et al. 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. 2019; Guimond & Michael 2021). Saltwater intrusion is facilitated by the 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 decreased agricultural productivity through soil aggregate dispersion (e.g., sodium inputs from saltwater), eutrophication through the loss of nutrients (e.g., P outputs), and submerging of farmland, converting fields into tidal marshes (Tully et al. 2019). As sea levels continue to rise and our coastlines change, we need to implement new management strategies to reduce farm-level 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 stick on clay surfaces or the iron (Fe) and aluminum oxides and hydroxides present in soil, but with the influx of salts from SLR and SWI, legacy P can be released (Prasad & Chakraborty 2019). 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) and sulfate (SO_4) for Fe on soil mineral surfaces (Jordan et al. 2008), especially as soils become anoxic, reducing Fe minerals and releasing PO_4 from binding sites. In saltwater systems, SO_4 in the solution competes with PO_4 for binding sites as the soils dry, leaving PO_4 in solution and prone to loss (Upreti et al. 2015; Weissman et al. 2019). In some cases, land easement programs (e.g. Conservation Reserve Enhancement Program; CREP 2022, Environmental Quality Incentives Program; EQIP 2021) 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. 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 (Weissman et al. 2019). Thus, strategies are needed to draw down soil P levels before hydrology is restored.

Rising seas and elevated soil salinity levels have forced farmers to reconsider their management practices, and these decisions will impact downstream ecosystems.

Different frameworks are used to categorize human responses to climate change and natural hazards such as floods (Deshmukh et al. 2012; Canyon et al. 2015; van Wesenbeeck et al. 2017; Doberstein et al. 2019). For example, people may protect their land by implementing networks of dikes or installing tide gates (Doberstein et al. 2019). Accommodation strategies may include elevating fields to allow the continued use of the vulnerable land, or to reduce risk by utilizing salt-tolerant crops (Oppenheimer et al. 2019). Retreat approaches are those where landowners repurpose their fields by enrolling them in easement programs (e.g. Conservation Reserve Program; CRP 2022) (van Wesenbeeck et al. 2017). The final option is to avoid building or continuing the use of flood-prone areas (Hu et al. 2015; Doberstein et al. 2019). Similar to these other frameworks, farmer may implement management practices in the Eastern Shore of Maryland to accommodate SLR (and SWI) by planting salt-tolerant crops, or allowing the transition of fields into marshland. In our study, we focused on four primary responses to rising seas and salinity on coastal farms: adaptation to increasing salinity levels by farming with salt-tolerant crops, remediation of fields for a few years before returning to agriculture, restoration the land to native marshes, or abandonment of salt-damaged fields entirely (Baker et al. 2014; Tully et al. 2019).

Many landowners desire to continue farming and may seek new crop options (Kleinman et al. 2015; Tully et al. 2019). Across the globe, new crop varieties are being developed to tolerate increasing salinity, such as salt-tolerant potatoes (*Solanum tuberosum*), rice (*Oryza sativa*), quinoa (*Chenopodium quinoa*), and salt-tolerant soybean (Chawla et al. 2013; Blom-Zandstra et al. 2014; de la Reguera et al.

2020; Li et al. 2022). Other crops, such as sorghum (*Sorghum bicolor*) and barley (*Hordeum vulgare*) are naturally more resistant to salinity stress (Maas and Grattan 1999). On the Eastern Shore of Maryland, there is no market for quinoa or potatoes, so we focus on salt-tolerant soybean (*Glycine max*) and sorghum (*Sorghum bicolor*), which are often used as chicken feed, as representative species for the *adaptation* approach. In agricultural systems, the *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 & Singh 2005; Colomb et al. 2007; Kleinman et al. 2007; Dixit et al. 2015). In the study region, switchgrass (*Panicum virgatum*) is a fast-growing, perennial grass species that tolerates both water and nutrient limitations (Di Virgilio et al. 2007). It could be used for biofuel, poultry bedding, or otherwise harvested to remove P from the system (Lee et al. 1998; David & 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 2022). On the Eastern Shore of Maryland, planting saltmarsh hay (*Spartina 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 & Dukes 2009; Hu et al. 2015; Gedan et al. 2011). Preliminary data from the field sites in Somerset County, MD suggest that the planting of remediation and restoration species may reduce porewater P concentrations (Weissman unpublished). Finally, farmers may choose to *abandon* their fields, allowing weeds and native species to grow, and enroll their land in conservation programs (e.g. CRP 2022; MD DNR Coastal Resilience Easements). Land abandonment may be the best option for those who do not want to invest any more time or money battling SWI. A dominant species in abandoned fields is a native grass (*Panicum dichotomiflorum*) that could benefit the land by reducing soil erosion through their extensive root systems (Jordan & Vatovec 2004). After two years, the most common species in the abandonment treatment was large crabgrass (*Digitaria sanguinalis*), which can tolerate slightly higher soil salinity levels and may aid in the transition of agricultural land to marshland (Zhang et al. 2012; Gedan et al. 2019). We have yet to understand the influence of these management practices on biogeochemical cycling, specifically P dynamics. Improving soil health and plant productivity hinges on quantifying P concentrations and pools in the soil, plant tissues, and porewater to identify differences in management options under SWI conditions.

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. The objectives of this research are to: (1) quantify soil available P and total P, porewater P, and leaf tissue P concentrations and pools among adaptation, remediation, restoration, and abandonment systems and species; (2) determine if field management practices (e.g., adapt, remediate, restore, abandon) change soil P concentrations and pools along a SWI gradient; (3) determine if field management practices (e.g., adapt, remediate, restore, abandon) change porewater P concentrations along a SWI gradient; and (4) determine if biomass P pools in representative species (e.g., adapt, remediate, restore, abandon) change along a SWI gradient. 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 improve nutrient uptake, increase soil health, and maintain crop yields. Finally, this work will help inform state-level coastal management policies and determine optimal strategies for climate resilience.

Methods

Study sites

Our study took place 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 2018), and where agriculture is responsible for 27% of the phosphorus (P) entering the Bay (CAST 2019). The climate in the region is humid subtropical with a mean annual average temperature of 9.7ºC, and annual precipitation of 1194 mm (NRCC 2022). The Chesapeake Bay waters range from fresh to saline (0 to 30 ppt; Chesapeake Bay Program 2020). Low-lying coastal regions along the Eastern Shore are vulnerable to high tide flooding and sea-level rise (Sweet et al. 2018), as many homes and farms in Dorchester and Somerset counties have a mean elevation of ~2.7 m above sea level with little to no slope. Dorchester and Somerset counties combined have over 3,000 km of coastline and about 68% of the land area is in cropland (USDA 2017). These areas are severely threatened by sea-level rise, as this landscape is connected to the Bay through extensive ditch networks that facilitate the movement of saline water onto agricultural fields during high tide (Bhattachan et al. 2019). The region is predicted to experience sea-level rise of 0.8 to 1.6 feet (0.24 to 0.49 m) by 2050 (Figure 1.1; Boesch et al. 2018). In addition, recent work suggests that as much as 30% of the land area in Dorchester and Somerset counties is at risk of saltwater intrusion (Epanchin-Niell 2019). In fact, this area is already experiencing the impacts of saltwater intrusion, as roughly 2% of agricultural land has transitioned into marshland over the past eight years (2009-2017; Gedan et al. 2020).

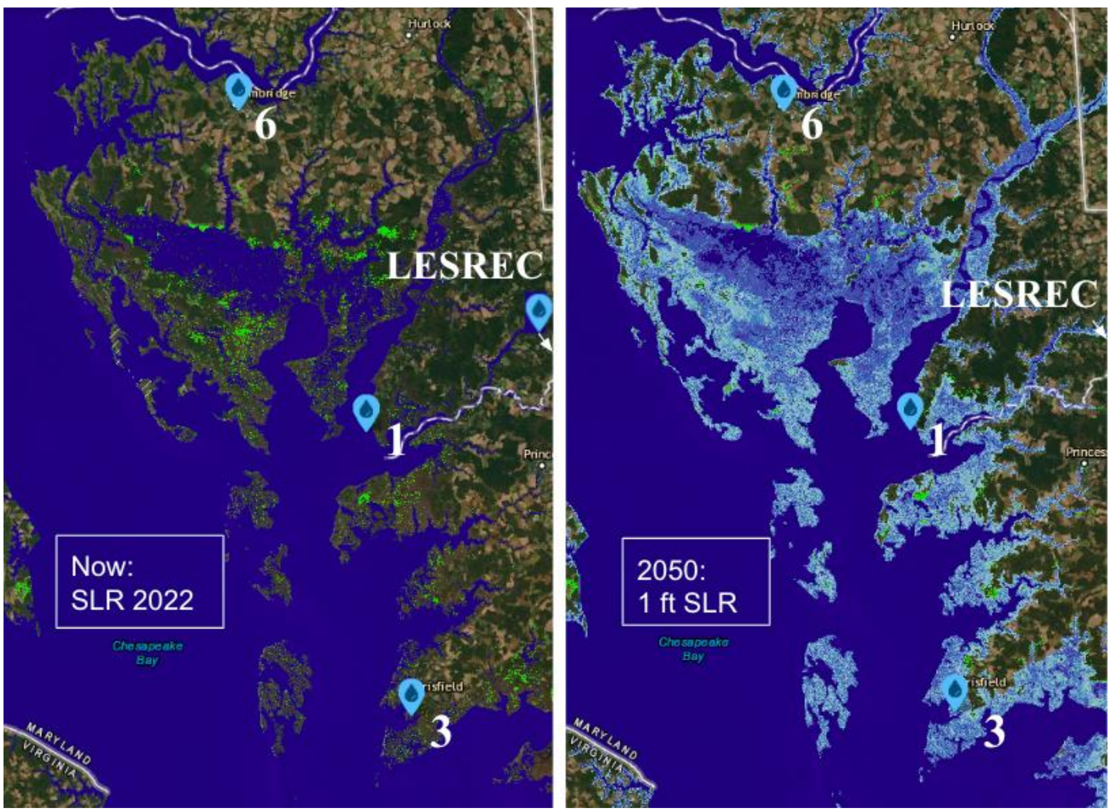


Figure 1. 1. Sea level rise impacts in 2022 compared to 2050 on coastal agricultural field sites located on the Lower Eastern Shore of Maryland. Sea level rise impacts (land coverage) in 2022 compared to projected 1.0 ft (0.305 m) of SLR by 2050 (NOAA 2022). Agricultural field sites indicated by blue points labeled 1, 3, 6, and LESREC (no salt control).

Experimental design

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 control in November; Figure 1.1). One field (Farm 6) was located near Cambridge, MD in Dorchester Co., MD (38.5632° N, -76.0785° W). The other two fields were located in Somerset Co., MD with one near Princess Anne, MD (Farm 1; 38.1608°N, -75.7907 °W) and the other near Crisfield, MD (Farm 3; 37.9834°N, -75.8545°W). A fourth site was located inland and did not experience saltwater intrusion (established as a no-salt control) 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 LESREC, the control site, soils were Mattapex silt loams (mesic Aquic Hapludults). Baseline soil properties are shown in Table 1.1.

Table 1. 1. Soil characteristics of field sites on the Lower Eastern Shore of Maryland collected from 0-10 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

Plots 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 control). 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) (Figure 1.2). There was a 0.5 m buffer between treatments and a 1 m buffer between each plot.

The no-salt control field site was established in November of 2018 and has a layout of 16 plots with the same treatments (1-3) and a salt-tolerant crop rotation (chloride-excluding soybean, sorghum, and barley) with each entry point present every year, and a remediation species (treatment 4; switchgrass). As the field was not affected by saltwater, we did not install the restoration species (treatment 5; saltmarsh hay). Additionally, we did not allow for natural recruitment of weeds (treatment 6; native grasses), as this was not recommended by the LESREC farm manager in order to maintain the integrity of the fields for future researchers. Plots in the no-salt control site were also 3 m x 20 m with a 0.5 m buffer between treatments and 1 m buffer between blocks.

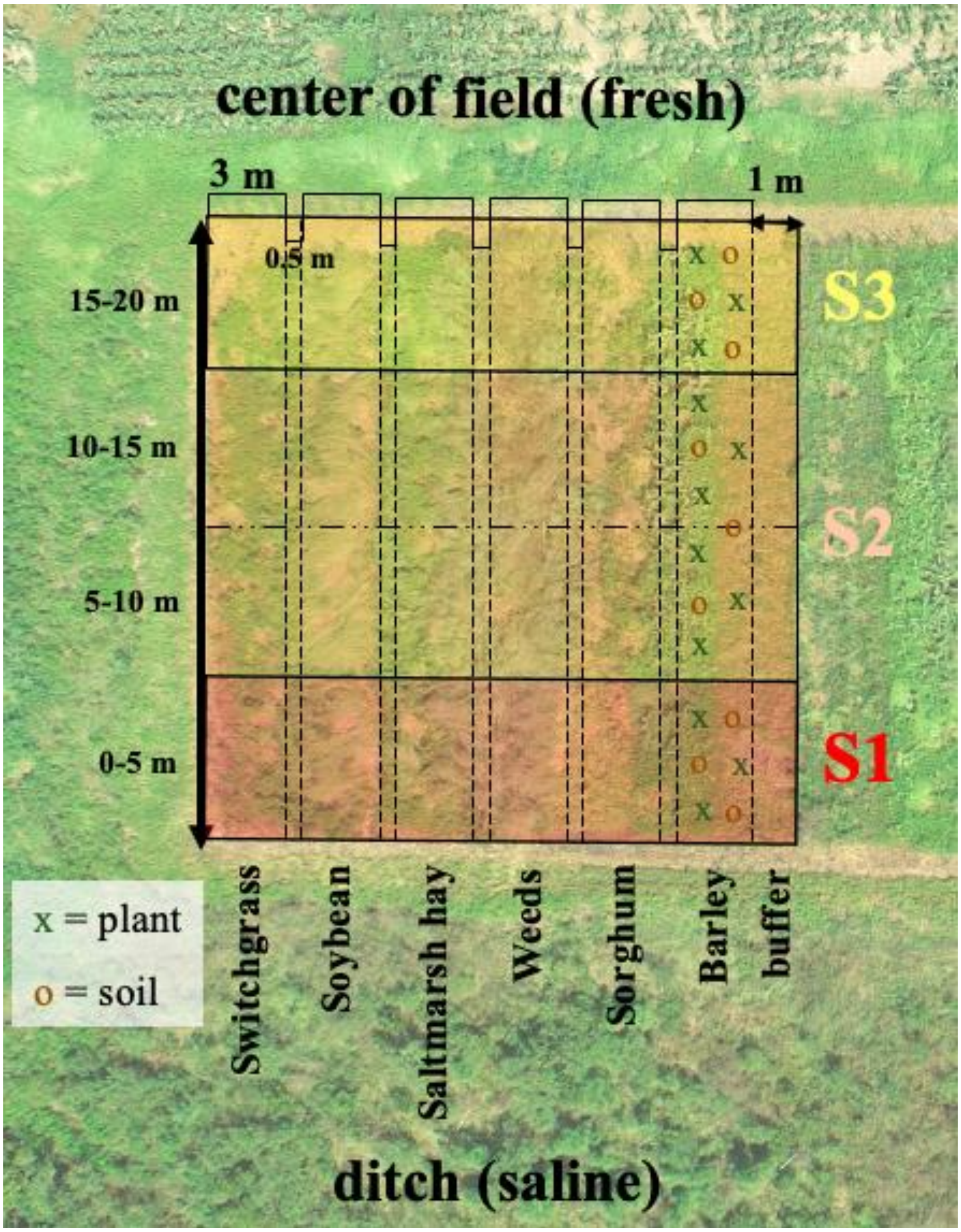


Figure 1. 2. Sampling design for soil cores and plant biomass in an example replicate block in a saltwater-intruded field site on the Lower Eastern Shore of Maryland. Soil cores (represented by brown o) were collected from each plot to 60 cm (0-10, 10-20, 20-30, 30-60 cm; three replicates for each depth) 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).

Field activities

The full crop rotation included sorghum, winter barley (*Hordeum vulgare*), double crop soybeans, sorghum, and full season soybeans. The crops were split into three plots differing in the starting point of the rotation (Figure 1.2). All sorghum and soybean plots were sprayed with glyphosate (~0.45 kg active ingredient per acre) in early May each year (2018-2021) to suppress weeds. Each year, sorghum and chloride-excluding soybean seeds were sown in the field in mid-May to early-June using a 1.52-m Tye drill. Sorghum and chloride-excluding soybean plots were planted in 38.1 cm rows with sorghum seeding rates at 80k acre^{-1} in 2018, then 120k acre^{-1} from 2019-2022 and soybean seeding rates at 180k acre^{-1} each year. Chloride-excluding soybean plots received potassium as potash (KCl; 39.2 kg ha^{-1}) and sorghum received N as urea (84 kg-N ha^{-1}) each year at planting. Sorghum was harvested from all sites in late-September to early-October each year. Soybeans were harvested in late October from all sites each year.

Switchgrass and saltmarsh hay were started from seed in plug trays and grown 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 switchgrass and saltmarsh hay at Farm 1 and only switchgrass at LESREC and gap-filled in April of 2020. This process was repeated in 2020 as necessary to ensure good stands of our perennial treatments.

Soil collection

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 bulk density cores were collected every 0-10, 10-20, 20-30, and 30-60 cm. Soil samples were collected for 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. In both cases, soils were returned to the University of Maryland - College Park and dried at 105ºC for 7 days. Core volume was used to calculate bulk density (g cm^{-3}). Soil bulk density at LESREC 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 LESREC in November of 2018 using the same method as above. Baseline soils were analyzed for total C and N using dry combustion (LECO TruMac N, St. Joseph, MI). Soil available P, Ca, K, Mg, Na, and S were determined on an ICP following Mehlich-3 extraction (Mehlich 1984). Texture was determined using the hydrometer method (Gee and Bauder 1979; Table 1.1).

Starting in November 2019, we collected 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 (Figure 1.2). Since the LESREC site did not have a salinity gradient, we sampled only at the plot-level. Because we found that soil P concentrations approached zero below a depth of 30 cm in 2018 and 2019, we only analyzed soils collected from 0-30 cm depths in 2021 and 2022.

Annual soil analysis

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 for two days prior to analysis (60 ºC). 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 from the 2021 collection 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 Loveand, CO) using the molybdate-blue method for PO_4-P (detection limit 0.01 mg PO_4-P/L; Murphy & Riley 1962). Bioavailable P was also analyzed using the molybdate-blue method as above. Upon initial inspection, available 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 for 2018 and 2021 collections.

To calculate soil nutrient pools, the nutrient concentrations from each individual depth increment (e.g., 0-10, 10-20, 20-30, and 30-60 cm) was multiplied by the depth of the segment (in cm) and the bulk density (g cm^{-3}).

Ground and dried soil samples were 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).

Porewater collection

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 (2018-2021). 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.

Porewater 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 soil solution was acidified using 10% hydrochloride (HCl) solution (150 μL 10% HCl in 15 mL soil 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 & Riley 1962).

Plant collection

We used agronomic methods of biomass estimation for crops (*adapt*), and ecological methods for all other treatments (*remediate*, *restore*, *abandon*). Biomass was collected from the weeds, saltmarsh hay, and switchgrass plots from the three saltwater-intruded sites (Farm 1, Farm 3, Farm 6) in August 2018, 2019, 2020, and 2021. 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 brackish water ditch that borders the study sites to capture the effects of salinity (Figure 1.2). Switchgrass was collected from LESREC at the plot level. Total biomass samples were clipped within 2 cm of the soil surface, weighed wet, allowed to air-dry, sorted to species, and then clipped again into sub-samples for processing. Sub-samples were oven-dried for at least 48 h at 60 ºC and weighed for conversion of total plant biomass to dry-weight-equivalent (DWE). 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 panic grass (*Panicum dichotomiflorum*), a native variety of switchgrass, was a species common to nearly all weed plots across all sites therefore, was retained for tissue analysis. In the following years (2019-2022), the most common species was *Digitaria sanguinalis*, so this species was retained for tissue analysis.

Mature sorghum seed heads were collected from the sorghum plots in late September to early October and soybean plots in late October 2018, 2019, and 2020 along the saltwater intrusion gradient as described above (0-5, 5-10, 10-15, 15-20 m from the ditch). Soybean and sorghum yields were collected using a 0.25 m^2 quadrat as a guide and collected by row in each plot. Five rows were collected of soybeans and three rows were collected of sorghum. Both sorghum and soybean seeds were passed through a plot combine (Massey-Ferguson 8XP Clarington, Ontario), bagged, and weighed on a scale to estimate the total yield per subplot (in bu $acre^{-1}$). In the no-salt control field (LESREC), we collected biomass at the plot-level (as there was no salinity gradient) and processed as described above. Sorghum and soybean were collected in 2021 differently. Random individual sorghum and soybean plants were clipped within 2 cm of the soil surface from each subplot (three replicates) of their respective plots in October. The whole plant (including mature seed head or soybean seeds) was weighed wet, oven-dried for at least 48 h at 60 ºC, and weighed dry for conversion of total plant biomass to dry-weight-equivalent. The remaining crops were dried then passed through the plot combine, bagged, and weighed. Yields were adjusted to the correct moisture. Harvestable yields and biomass were converted to an area-basis (kg ha^{-1}).

Plant analysis

Oven-dry (60 ºC) plant sub-samples from each treatment 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 Loveand, CO) using the molybdate-blue method for PO_4-P (detection limit 0.01 mg PO_4-P/L;

Murphy & Riley 1962). Biomass P pools, also known as P stocks, were calculated by multiplying nutrient concentrations by the aboveground biomass (DWE) and yields (DWE; kg ha^{-1}) collected from the plots.

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., adapt, remediate, restore, abandon) on soil P concentrations and pools averaged at the plot-level. We examined each 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 year, site, section, and depth individually (e.g. soil bioavailable P in 2021 from Farm 3 in S1 at 0-10 cm depth). Soil P concentrations were measured in 2018 (to understand baseline P levels) and 2021 (after a full crop rotation). Tukey *post-hoc* comparisons were used to determine significant differences among management strategies. 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., adapt, remediate, restore, abandon) on porewater P concentrations averaged at the plot-level. We examined each 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 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., adapt, remediate, restore, abandon) on leaf tissue P concentrations and pools averaged at the plot-level. We examined each year and site individually with management strategy (treatment) and section (SWI) 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 individually. Tukey *post-hoc* comparisons were used to evaluate significant differences among management strategies using the *multcomp* and *stats* packages (Hothorn et al. 2022, R Core Team 2023). Finally, univariate regression was used to determine if there is a relationship between soil and plant tissue P concentrations. All statistics were run in the R environment for Mac (R Core Team 2021).

<u>*Results*</u>

Soil extractable P concentrations

In the soil, concentrations of Mehlich-3 extractable P ([M3P]) decreased by 30-50% in the topsoil (0-10 cm) at two sites after three years of planting (33.9% at Farm 3 and 52.8% at Farm 6; Figure 1.3). At Farm 6, [M3P] decreased from 109.3 mg kg^{-1} in 2018 to 51.6 mg kg^{-1} in 2021 in the top 10 cm of the soil. There was a significant effect of distance from the ditch at Farm 6 ($p<0.05$; Figure 1.3), where the section closest to the ditch (0-5 m) had higher [M3P] than the section furthest from the ditch (15-20 m). There was a significant effect of depth at all sites ([M3P] decreases with depth; $p<0.01$), but there was no effect of treatment.

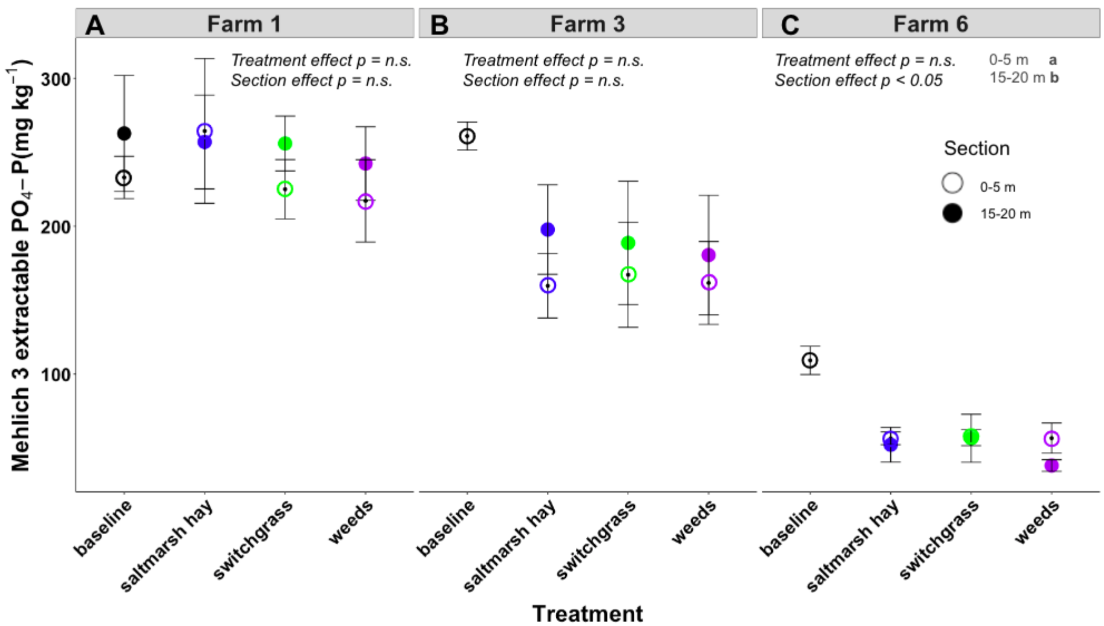


Figure 1. 3. Soil extractable P concentrations in topsoil (0-10 cm) at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland in 2018 (baseline) and 2021. The x-axis is treatment and the y-axis is Mehlich-3 extractable P concentrations in milligrams per kilogram. Treatments are represented by colors, where the saltmarsh hay treatment is blue, the switchgrass treatment is green, and the weeds treatment is purple and section is represented by open or closed circles, where open circles are 0-5 m away from the ditch and closed circles are 15-20 m away from the ditch.

Soil total P concentrations

Overall, soil total [P] decreased at all SWI sites after three years of planting, and concentrations decreased by 20% in the topsoil at Farm 1 and Farm 6 (19.6% and 18.3%, respectively; Figure 1.4). The total [P] in 2018 at Farm 6 was 507.4 mg kg^{-1} and decreased to 414.5 mg kg^{-1}. There was a significant effect of distance from the ditch at Farm 3 and Farm 6 ($p<0.05$; Figure 1.4), where the section closest to the ditch (0-5 m) had higher [M3P] than the section furthest from the ditch (15-20 m) at Farm 6 and the opposite was seen at Farm 3. Similar to the soil extractable [P], there was no significant effect of treatment, and there was a significant effect of depth (total [P] decreases with depth; $p<0.01$).

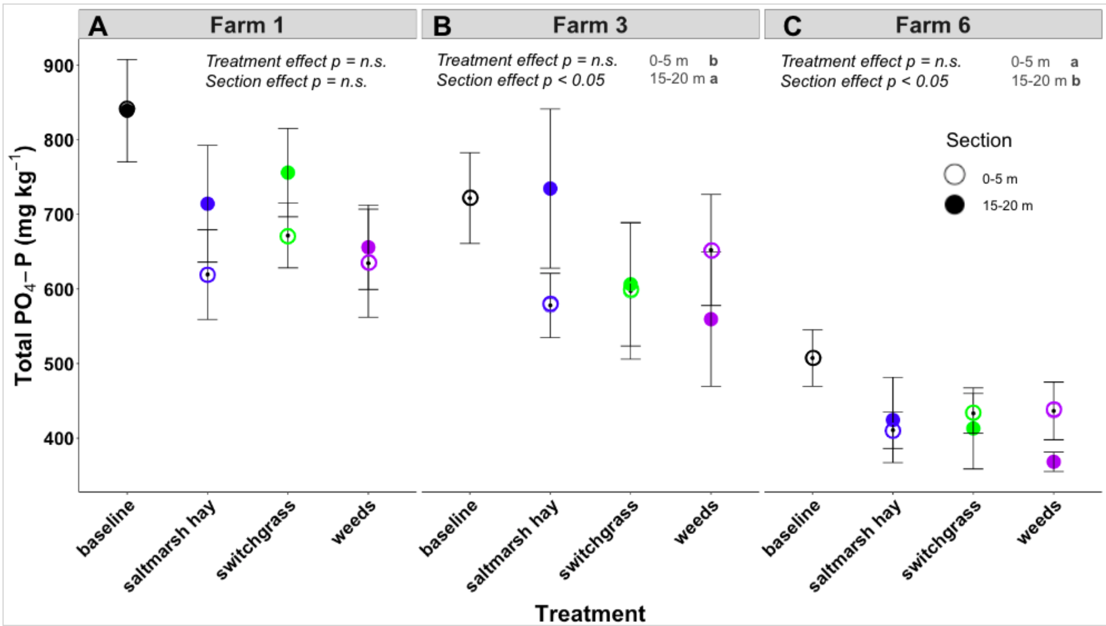


Figure 1. 4. Soil total P concentrations in topsoil (0-10 cm) at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland in 2018 (baseline) and 2021. The x-axis is treatment and the y-axis is total P concentrations in milligrams per kilogram. Treatments are represented by colors, where the saltmarsh hay treatment is blue, the switchgrass treatment is green, and the weeds treatment is purple and section is represented by open or closed circles, where open circles are 0-5 m away from the ditch and closed circles are 15-20 m away from the ditch.

Soil extractable P pools

Similar to the M3P concentrations, soil M3P pools decreased by 30-50% in the topsoil at two sites after three years of planting (33.9% at Farm 3, 52.9% at Farm 6; Figure 1.5). There was a significant effect of distance from the ditch at Farm 3 and Farm 6 ($p<0.05$; Figure 1.5; Table 1.2), 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 and the opposite was seen at Farm 3. There was a significant effect of depth ($p<0.01$; Table 1.2). In 2018, soil M3P pools in the 0-10 cm depth (176-352 kg ha^{-1}) had the highest M3P pools, the next 10 cm (10-20 cm) had intermediate levels of M3P pools

(48.02-267.6 kg ha^{-1}), and finally the deeper depth (20-30 cm) had the lowest M3P pools (8.47-77.4 kg ha^{-1}). In 2021, soil M3P pools in the topsoil (0-10 cm) were greater (83.0-301.0 kg ha^{-1}) than the 10-20 cm depth (19.9-150.0 kg ha^{-1}) and the 20-30 cm depth (7.44-51.49 kg ha^{-1}).

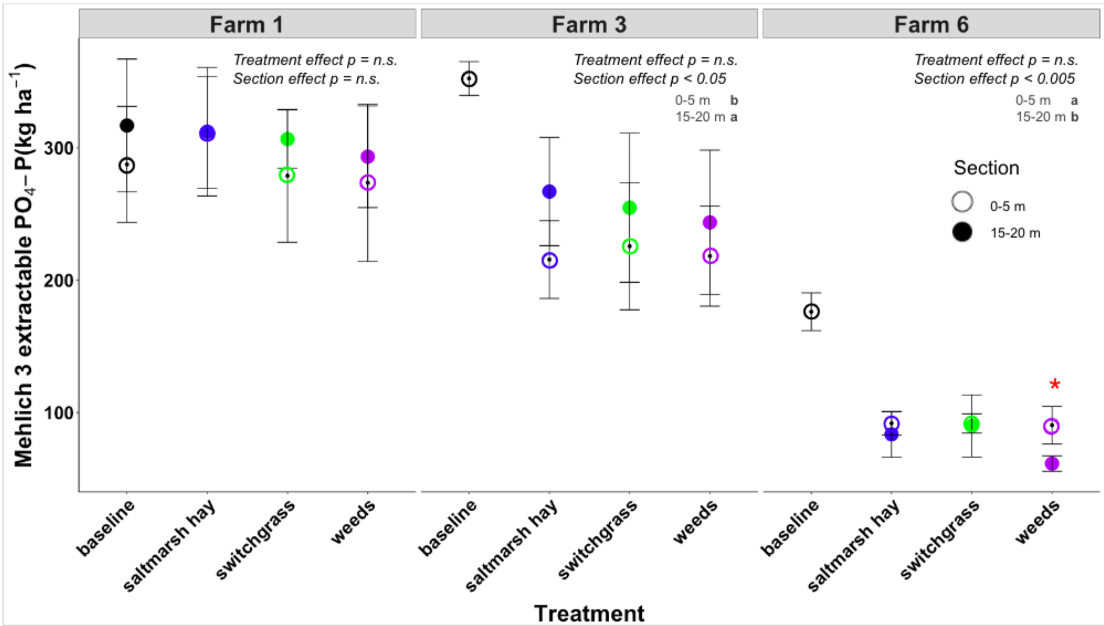


Figure 1. 5. Soil extractable P pools in topsoil (0-10 cm) at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland in 2018 (baseline) and 2021. The x-axis is treatment and the y-axis is Mehlich-3-extractable P pools in kilograms per hectare. Treatments are represented by colors, where the saltmarsh hay treatment is blue, the switchgrass treatment is green, and the weeds treatment is purple and section is represented by open or closed circles, where open circles are 0-5 m away from the ditch and closed circles are 15-20 m away from the ditch.

Table 1. 2. ANOVA results from LME models testing the effects of treatment and distance from the salinity source with the dependent variable as soil available P pools from each saltwater-intruded site on the Lower Eastern Shore of Maryland in year 1 and year 4, followed by Tukey HSD results indicating statistical significance between variables.

ANOVA

Year 1 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Section	1	365	365	0.35	0.6
Depth	2	41683	20841	33.12	< 0.001

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-19.24	17.74	0.5
20-30	0-10	-133.53	17.74	< 0.001

20-30	10-20	-114.29	17.74	< 0.001

ANOVA

Year 1 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Depth	2	8624.1	4312	91.14	< 0.001

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-21.92	4.86	< 0.001
20-30	0-10	-64.57	4.86	< 0.001
20-30	10-20	-42.65	4.86	< 0.001

ANOVA

Year 1 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Depth	2	61481	30740	66.82	< 0.001

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-128.15	15.17	< 0.001
20-30	0-10	-167.70	15.17	< 0.001
20-30	10-20	-39.56	15.17	0.03

ANOVA

Year 4 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	4	3209	802	1.47	0.2
Section	1	3991	3991	7.30	0.01
Depth	2	116261	58130	106.36	< 0.001

Tukey HSD

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
15-20	0-5	11.53	4.27	0.007

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-9.71	5.23	0.2
20-30	0-10	-70.35	5.23	< 0.001
20-30	10-20	-60.64	5.23	< 0.001

ANOVA

Year 4 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	4	26.01	6.50	1.30	0.3
Section	1	70.99	70.99	14.17	< 0.001
Depth	2	2256.42	1128.21	225.15	< 0.001

Tukey HSD

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
15-20	0-5	1.54	0.41	< 0.001

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-3.54	0.50	< 0.001
20-30	0-10	-10.44	0.50	< 0.001
20-30	10-20	-6.91	0.50	< 0.001

ANOVA

Year 4 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	4	1.48	0.37	4.97	0.005
Section	1	0.79	0.79	10.59	0.003
Depth	2	66.16	33.08	445.57	< 0.001

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	0.23	0.079	0.03
Weeds	Saltmarsh hay	-0.072	0.079	0.9
Weeds	Switchgrass	-0.30	0.079	0.001

Tukey HSD

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
15-20	0-5	-0.16	0.050	0.001

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-1.08	0.061	< 0.001
20-30	0-10	-1.81	0.061	< 0.001
20-30	10-20	-0.73	0.061	< 0.001

Soil total P pools

Following a similar pattern as the soil total P concentrations, soil P pools decreased, but by 20% in the topsoil at two sites after three years of planting (18% at Farm 1, 18.4% at Farm 6; Figure 1.6). There was a significant effect of distance from the ditch at all SWI sites ($p<0.05$; Figure 1.6; Table 1.3), where the section furthest from the ditch (15-20 m) had higher P levels than the section closest to the ditch (0-5 m) at Farm 1 and Farm 3 and the opposite was seen at Farm 6. There was a significant effect of depth ($p<0.01$; Table 1.3), where baseline total soil P pools in the 0-10 cm depth (819-993 kg ha^{-1}) had the highest total P pools, the next 10 cm (10-20 cm) had intermediate levels of P (441-889 kg ha^{-1}), and finally the deeper depth (20-30 cm) had the smallest total P pools (247-503 kg ha^{-1}). After three years of planting, the depth effect remained consistent but total soil P was reduced at all levels; total soil P pools in the topsoil (0-10 cm) were greater (668-834 kg ha^{-1}) than the 10-20 cm depth (326-743 kg ha^{-1}) and the 20-30 cm depth (223-432 kg ha^{-1}).

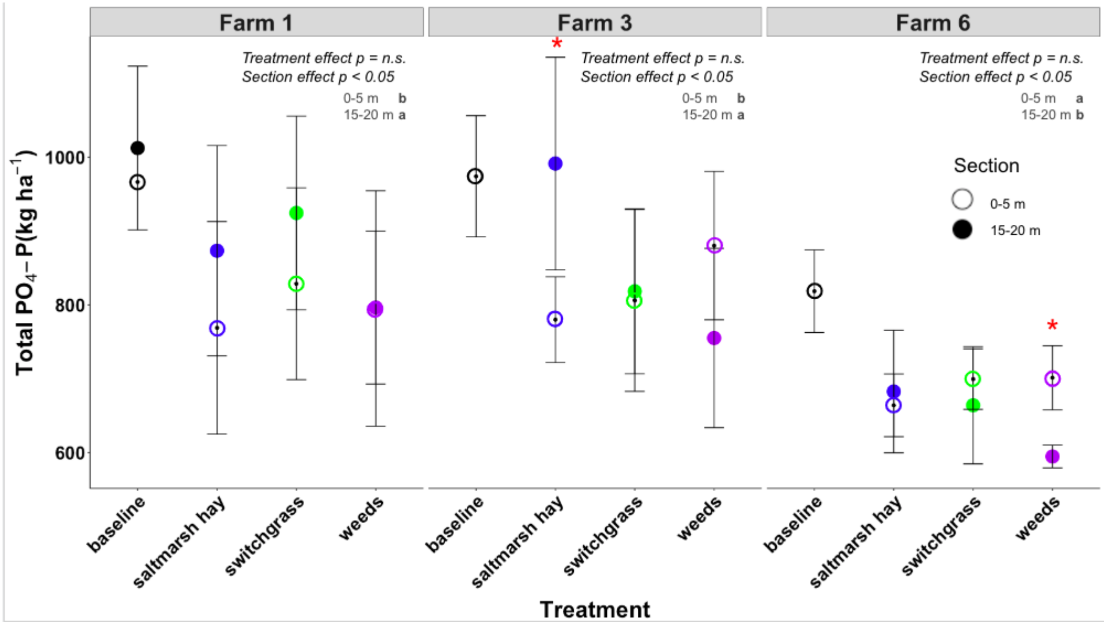


Figure 1. 6. Soil total P pools in topsoil (0-10 cm) at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland in 2018 (baseline) and 2021. The x-axis is treatment and the y-axis is total P pools in kilograms per hectare. Treatments are represented by colors, where the saltmarsh hay treatment is blue, the switchgrass treatment is green, and the weeds treatment is purple and section is represented by open or closed circles, where open circles are 0-5 m away from the ditch and closed circles are 15-20 m away from the ditch.

Table 1. 3. ANOVA results from LME models testing the effects of treatment and distance from the salinity source with the dependent variable as soil total P pools from each saltwater-intruded site on the Lower Eastern Shore of Maryland in year 1 and year 4, followed by Tukey HSD results indicating statistical significance between variables.

ANOVA

Year 1 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Section	1	3729	3729	1.00	0.4
Depth	2	281271	140635	37.87	< 0.001

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-57.81	31.76	0.2
20-30	0-10	-258.73	31.76	< 0.001
20-30	10-20	-200.92	30.47	< 0.001

ANOVA

Year 1 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Depth	2	64479	32240	28.97	< 0.001

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-58.55	23.59	0.04
20-30	0-10	-176.27	23.59	< 0.001
20-30	10-20	-117.72	23.59	< 0.001

ANOVA

Year 1 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Depth	2	0.25	0.13	42.96	< 0.001

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-0.19	0.038	< 0.001
20-30	0-10	-0.35	0.038	< 0.001
20-30	10-20	-0.17	0.038	< 0.001

ANOVA

Year 4 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	4	963.9	241.0	3.52	0.2
Section	1	642.3	642.3	9.37	0.005
Depth	2	18963.6	9481.8	138.32	< 0.001

Tukey HSD

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
15-20	0-5	4.63	1.51	0.002

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-5.17	1.85	0.01
20-30	0-10	-28.87	1.85	< 0.001
20-30	10-20	-23.71	1.85	< 0.001

ANOVA

Year 4 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	4	36.05	9.01	1.74	0.2
Section	1	44.80	44.80	8.66	0.007
Depth	2	2734.23	1367.12	264.37	< 0.001

Tukey HSD

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
15-20	0-5	1.22	0.42	0.003

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-4.44	0.51	< 0.001
20-30	0-10	-11.59	0.51	< 0.001
20-30	10-20	-7.15	0.51	< 0.001

ANOVA

Year 4 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	4	0.13	0.033	3.28	0.03
Section	1	0.099	0.099	9.95	0.004
Depth	2	7.67	3.84	385.77	< 0.001

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	0.043	0.029	0.6
Weeds	Saltmarsh hay	-0.057	0.029	0.3
Weeds	Switchgrass	-0.10	0.029	0.004

Tukey HSD

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
15-20	0-5	4.63	1.51	0.002

Tukey HSD

Depth 1	Depth 2	Mean Difference	Standard Error	*p-value*
10-20	0-10	-5.17	1.85	0.01
20-30	0-10	-28.87	1.85	< 0.001
20-30	10-20	-23.71	1.85	< 0.001

Topsoil EC and topsoil P concentration correlations

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 $r^2 > 0.2$ in year 4; $p<0.001$ and $p<0.01$, respectively).

Table 1. 4. Topsoil EC (0-10 cm) and topsoil total and available P concentration correlations in year 1 and year 4 for the section closest to the ditch (0-5 m) and the section furthest from the ditch (15-20 m) with each saltwater-intruded site on the Lower Eastern Shore of Maryland averaged. Values are r^2 and symbols are p-values. Colors represent the strength of the correlation (-0.00*** = strong negative; -0.00** = moderate negative; -0.00* = negative; -0.00^ = weak negative; 0.00^ = weak positive; 0.00* = positive; 0.00** = moderate positive; 0.00*** = strong positive). ^ = $p<0.05$, * = $p<0.01$, ** = $p<0.005$, *** = $p<0.001$

	Baseline		Year 4	
	0-5 m	15-20 m	0-5 m	15-20 m
EC~soil P	0.51***	0.57***	0.51***	0.51***
EC~soil M3P	0.68***	0.54***	0.31**	0.27*

Porewater P concentrations

In the first study year, porewater soluble reactive P ([SRP]) was highest in the summer (0.047-0.057 mg L^{-1}) and decreased after planting (0.026-0.036 mg L^{-1}; Figure 1.7). In 2019, porewater [SRP] decreased slightly (0.019-0.022 mg L^{-1}), but was still high in the summer months (0.034-0.058 mg L^{-1}). In the summer of 2018 and 2019, porewater [SRP] was highest under the saltmarsh hay plots (0.057-0.058 mg L^{-1}) and lowest under the switchgrass plots (0.034-0.047 mg L^{-1}). The last lysimeter collection occurred two years after planting (in 2020), and porewater [SRP] decreased significantly under each treatment in both the summer and autumn ([SRP] < 0.01 mg L^{-1}; $p < 0.05$; Figure 1.7), although there were no significant differences between treatments or between distance from the ditch at any of the SWI sites.

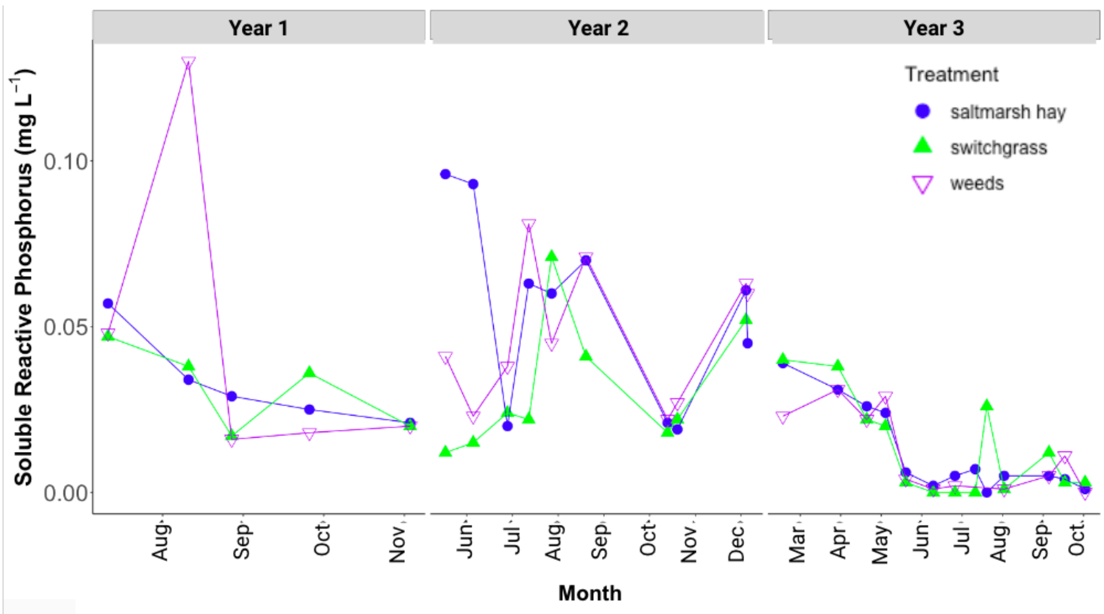


Figure 1. 7. Porewater P concentrations from year 1 to year 3 (2018-2020) with all three saltwater-intruded sites on the Lower Eastern Shore of Maryland grouped together. The x-axis is month and the y-axis is porewater soluble reactive P in milligrams per liter. Treatments are represented by colors, where the saltmarsh hay treatment is a blue circle, the switchgrass treatment is a green triangle, and the weeds treatment is a purple upside-down triangle. Distance from the salinity source was not included.

Plant leaf tissue P concentrations

We found similar patterns in plant leaf tissue P concentrations ([P]) among the three saltwater-intruded sites across all study years where the highest concentrations (3.8-4.7 g P kg^{-1}) were always found in weed tissue and the lowest [P] were found in saltmarsh hay tissue (1.9-2.2 g P kg^{-1}) with intermediate levels in switchgrass tissues

(2.8-3.9 g P kg^{-1}; $p<0.03$ in all cases; Figure 1.8). There was no clear relationship between plant leaf tissue [P] and distance from the salinity source although in some cases, we observed a significant effect of distance from the ditch. At the no-salt control site, we only planted switchgrass, which showed concentrations lower than the saltwater-intruded sites (2.7 g P kg^{-1}).

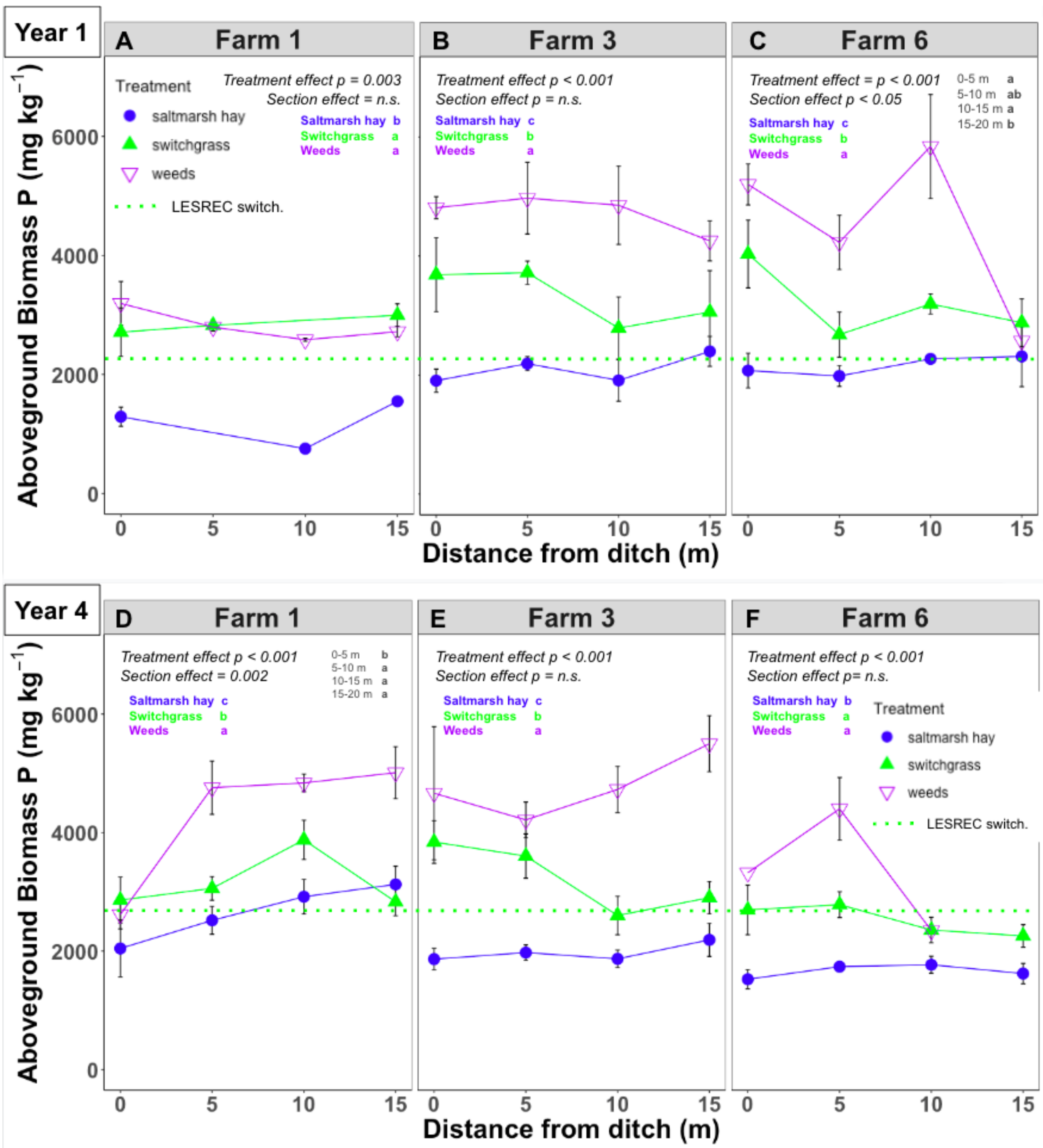


Figure 1. 8. Plant aboveground biomass P concentrations along the salinity gradient at three saltwater-intruded farms on the Lower Eastern Shore of Maryland from year 1 (A-C) and year 4 (D-F). The x-axis is distance from the ditch in meters and the y-axis is aboveground biomass P concentrations in milligrams per kilogram. Treatments are represented by colors and symbols, where the saltmarsh hay treatment is a blue circle, the switchgrass treatment is a green triangle, and the weeds treatment is a purple upside-down triangle. The green dotted line is LESREC switchgrass levels (because there was no saline ditch).

Species biomass

Across all years, Farm 6 had the same patterns in biomass, where switchgrass accumulated the highest biomass levels (0.98-14.5 Mg ha^{-1}), weeds produced the lowest biomass levels (0.11-3.36 Mg ha^{-1}), and saltmarsh hay biomass had intermediate biomass levels (0.77-10.9 Mg ha^{-1}; $p<0.01$ in all cases; Figure 1.9). The SWI gradient had an effect here in 2018 and 2019, where biomass increased with distance from the ditch ($p<0.01$ in both cases; Figure 1.9). In year 1 (2019) and year 2 (2020) at Farm 1, the weeds maintained high biomass levels (0.63-2.45 Mg ha^{-1}), but there was no significant effect of distance from the ditch. We found similar patterns in 2021 at both Farm 1 and Farm 6, where switchgrass accumulated the highest biomass levels (4.77-11.3 Mg ha^{-1}), weeds produced the lowest biomass levels (0.11-0.75 Mg ha^{-1}), and saltmarsh hay biomass varied (1.72-8.55 Mg ha^{-1}; $p<0.01$ in all cases; Figure 1.9). In contrast, saltmarsh hay at Farm 3 had the highest biomass in year 3 (2020) and year 4 (2021). At the no-salt control site, we only planted switchgrass, which had lower biomass than the SWI-impacted sites (0.18-6.78 Mg ha^{-1}).

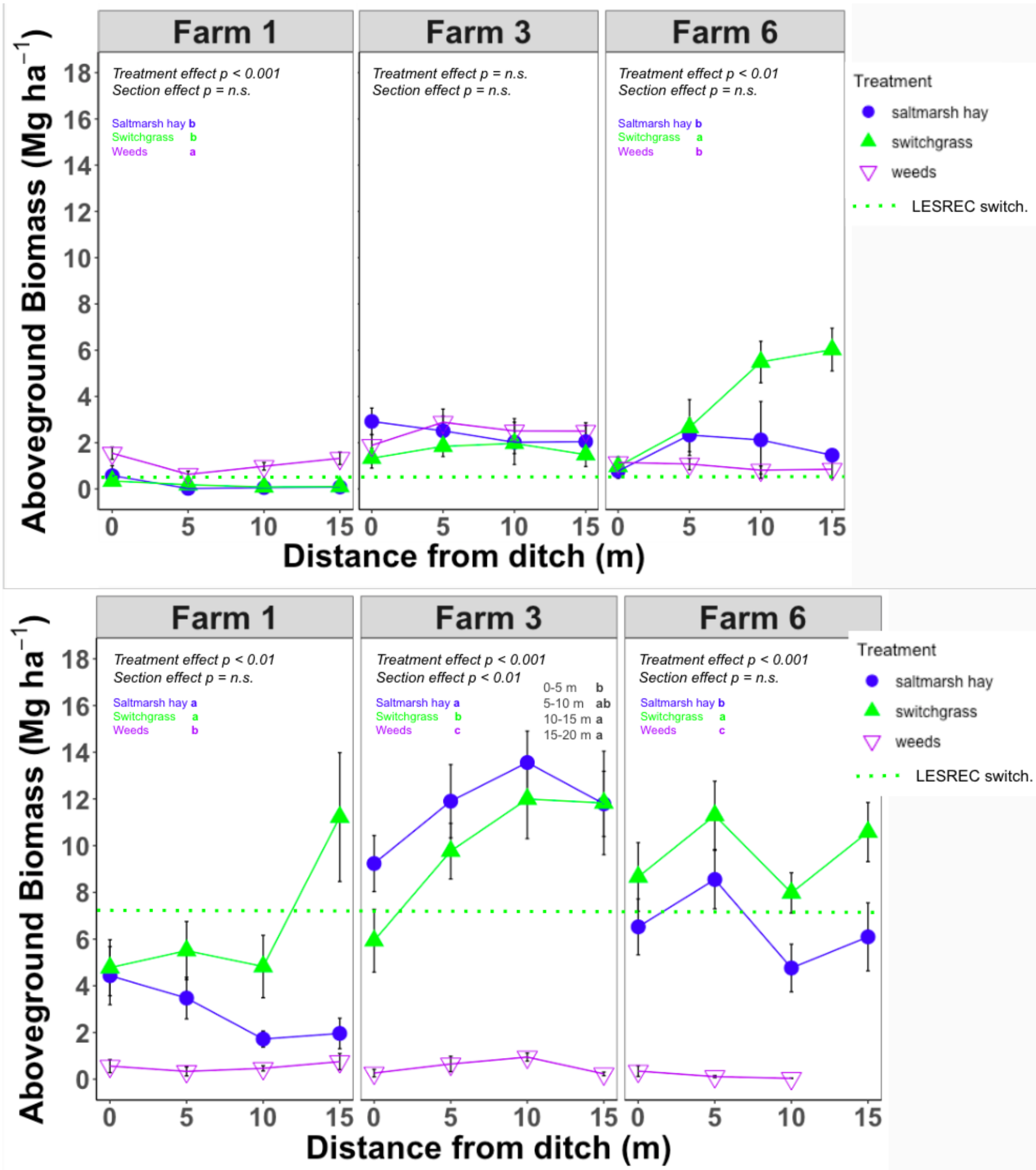


Figure 1. 9. Species biomass along the salinity gradient at three saltwater-intruded farms on the Lower Eastern Shore of Maryland from year 1 (top) and year 4 (bottom). The x-axis is distance from the ditch in meters and the y-axis is aboveground biomass in megagrams per hectare. Treatments are represented by colors and symbols, where the saltmarsh hay treatment is a blue circle, the switchgrass treatment is a green triangle, and the weeds treatment is a purple upside-down triangle. The green dotted line is LESREC switchgrass levels (because there was no saline ditch).

Plant leaf tissue P pools

In the first study year, weeds at all saltwater-intruded sites had the largest aboveground (AG) biomass P pools ($p<0.003$ in all cases; Figure 1.10; Table 1.5). In years 2, 3, and 4, at Farm 3 and Farm 6 we observed the highest AG biomass P pools in the switchgrass plots ($p<0.01$ in all cases; Figure 1.10; Table 1.5). However, in year 2 at Farm 1 weeds remained the highest AG biomass P pools ($p<0.001$). It was not until year 3 that switchgrass dominated, and we observed the highest AG biomass pools in switchgrass plots ($p=0.01$). P pools in the no-salt control showed a similar pattern to Farm 1 where switchgrass P pools were low in years 1 and 2, but increased in year 3.

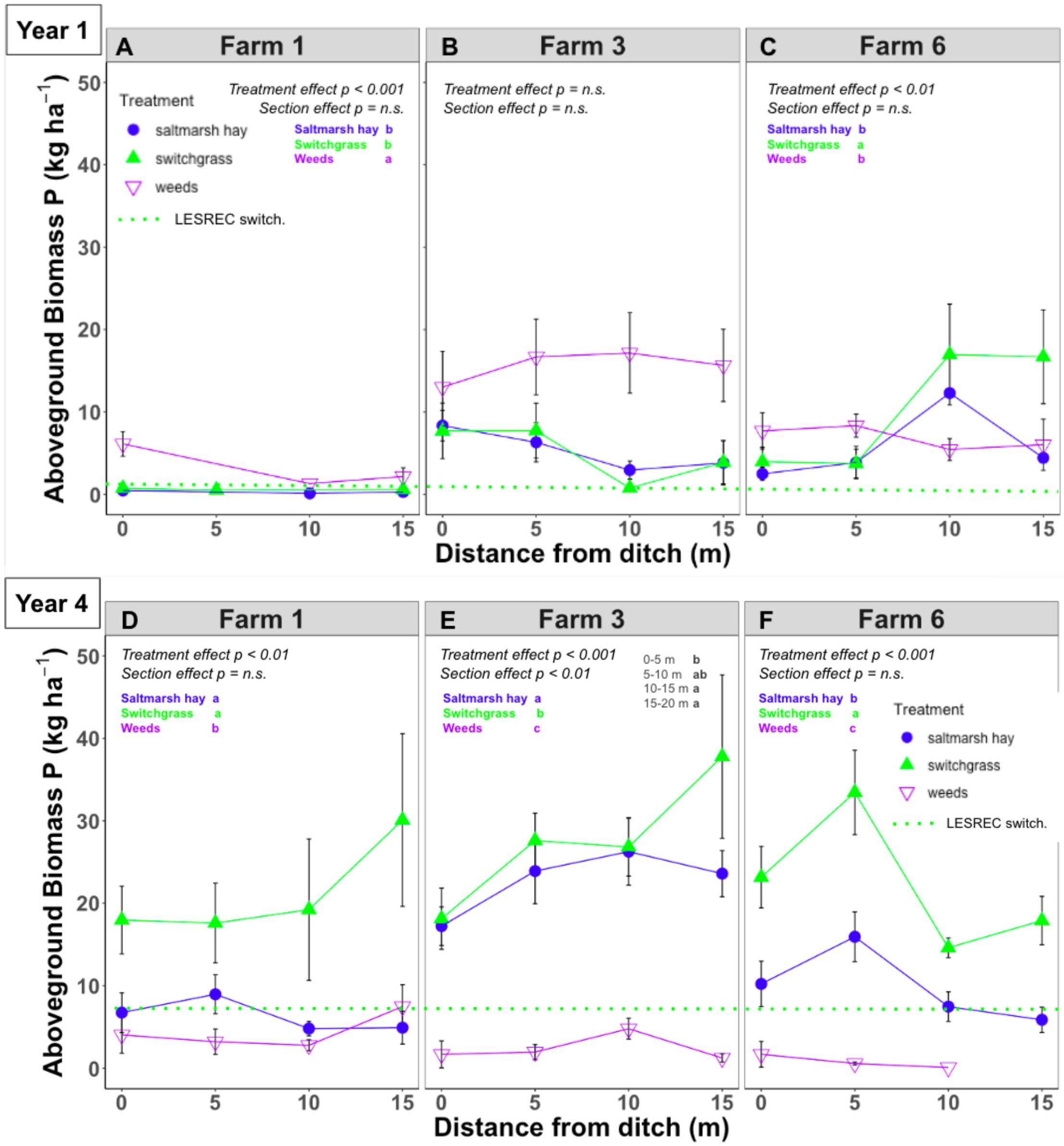


Figure 1. 10. Plant aboveground biomass P pools along the salinity gradient at three saltwater-intruded farms on the Lower Eastern Shore of Maryland from year 1 (A-C) and year 4 (D-F). The x-axis is distance from the saline ditch in meters and the y-axis is aboveground biomass P pools in kilograms per hectare. Treatments are represented by colors and symbols, where the saltmarsh hay treatment is a blue circle, the switchgrass treatment is a green triangle, and the weeds treatment is a purple upside-down triangle. The green dotted line is LESREC switchgrass levels (because there was no saline ditch).

Table 1. 5. ANOVA results from LME models testing the effects of treatment and distance from the salinity source with the dependent variable as plant biomass P pools from each saltwater-intruded site on the Lower Eastern Shore of Maryland from years 1 to 4, followed by Tukey HSD results indicating statistical significance between variables.

ANOVA

Year 1 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	17.81	8.90	10.23	0.001
Section	3	1.90	0.63	0.73	0.55

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	0.89	0.69	0.4
Weeds	Saltmarsh hay	2.39	0.57	< 0.001
Weeds	Switchgrass	1.50	0.58	0.03

ANOVA

Year 1 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	51.22	25.61	11.71	< 0.001
Section	3	10.23	3.41	1.56	0.23
Treatment:Section	6	15.72	2.62	1.20	0.32

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	-0.23	0.57	0.92
Weeds	Saltmarsh hay	2.22	0.55	< 0.001
Weeds	Switchgrass	2.45	0.57	< 0.001

ANOVA

Year 1 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	8.84	4.42	1.95	0.17
Section	3	7.53	2.51	1.11	0.37
Treatment:Section	6	23.73	3.96	1.75	0.17

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	1.08	0.74	0.31
Weeds	Saltmarsh hay	0.46	0.72	0.80
Weeds	Switchgrass	-0.62	0.66	0.61

ANOVA

Year 2 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	55.43	27.71	25.13	< 0.001
Section	3	2.67	0.89	0.81	0.5
Treatment:Section	6	6.87	1.15	1.04	0.4

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	1.90	0.42	< 0.001
Weeds	Saltmarsh hay	3.05	0.45	< 0.001
Weeds	Switchgrass	1.15	0.42	0.02

ANOVA

Year 2 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	99.68	49.84	27.62	< 0.001
Section	3	20.45	6.82	3.78	0.03

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	0.84	0.53	0.25
Weeds	Saltmarsh hay	-3.00	0.56	< 0.001
Weeds	Switchgrass	-3.84	0.55	< 0.001

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
5-10	0-5	0.55	0.63	0.81
10-15	0-5	1.82	0.65	0.03
15-20	0-5	-0.23	0.56	0.98
10-15	5-10	1.30	0.70	0.26
15-20	5-10	-0.79	0.62	0.59
15-20	10-15	-2.06	0.65	< 0.01

ANOVA

Year 2 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	57.53	28.76	5.71	0.01
Section	3	48.80	16.26	3.23	0.047
Treatment:Section	6	31.14	5.19	1.03	0.38

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	2.69	0.94	0.01
Weeds	Saltmarsh hay	-0.16	0.89	0.98
Weeds	Switchgrass	-2.86	0.90	0.0043

ANOVA

Year 3 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	19.89	9.95	4.98	0.02
Section	3	12.54	4.18	2.09	0.1
Treatment:Section	6	13.76	2.29	1.15	0.3

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	1.67	0.74	0.06
Weeds	Saltmarsh hay	-0.68	0.73	0.6
Weeds	Switchgrass	-2.36	0.71	0.003

ANOVA

Year 3 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	161.28	80.64	27.11	< 0.001
Section	3	12.69	4.23	1.42	0.3

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	1.07	0.63	0.21
Weeds	Saltmarsh hay	-3.66	0.70	< 0.001
Weeds	Switchgrass	-4.73	0.68	< 0.001

ANOVA

Year 3 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	189.32	94.66	29.72	< 0.001
Section	3	1.87	0.62	0.20	0.90

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	2.61	0.63	< 0.001
Weeds	Saltmarsh hay	-2.95	0.73	< 0.001
Weeds	Switchgrass	-5.56	0.73	< 0.001

ANOVA

Year 4 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	19.89	9.95	4.98	0.02
Section	3	12.54	4.18	2.09	0.1
Treatment:Section	6	13.76	2.29	1.15	0.3

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	1.67	0.74	0.06
Weeds	Saltmarsh hay	-0.68	0.73	0.6
Weeds	Switchgrass	-2.36	0.71	0.003

ANOVA

Year 4 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	393.78	196.89	100.08	< 0.001
Section	3	23.41	7.81	3.97	0.025
Treatment:Section	6	18.78	3.13	1.59	0.2

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	1.49	0.55	0.02
Weeds	Saltmarsh hay	-6.90	0.61	< 0.001
Weeds	Switchgrass	-8.39	0.64	< 0.001

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
5-10	0-5	1.29	0.68	0.2
10-15	0-5	1.97	0.67	0.02
15-20	0-5	1.77	0.65	0.03
10-15	5-10	0.68	0.68	0.8
15-20	5-10	0.48	0.67	0.9
15-20	10-15	-0.2	0.65	1.0

ANOVA

Year 4 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	2	176.49	88.24	58.61	< 0.001
Section	3	18.46	6.16	4.09	0.02

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Switchgrass	Saltmarsh hay	2.94	0.47	< 0.001
Weeds	Saltmarsh hay	-4.5	0.67	< 0.001
Weeds	Switchgrass	-7.46	0.68	< 0.001

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
5-10	0-5	0.85	0.57	0.4
10-15	0-5	-0.99	0.62	0.4
15-20	0-5	-1.01	0.62	0.4
10-15	5-10	-1.85	0.63	0.02
15-20	5-10	-1.87	0.63	0.02
15-20	10-15	-0.02	0.67	1.0

Grain P concentrations

We found similar patterns in grain [P] among all saltwater-intruded sites across all study years with significantly higher [P] in soybean grain (4.5-9.0 g kg^{-1}) than sorghum grains (5.1-7.0 g kg^{-1}; $p=0.01$ in all cases; Figure 1.11). Soybean grain P concentrations were highest at Farm 1 in 2021 (6.1-8.0 g kg^{-1}; $p<0.001$). There was no significant effect of distance from the salinity source at any of the sites each year (2019-2021). Finally, soybean P concentrations were lower (4.5-6.1 g kg^{-1}) at the no-salt control site (LESREC) than the SWI-impacted sites in 2021.

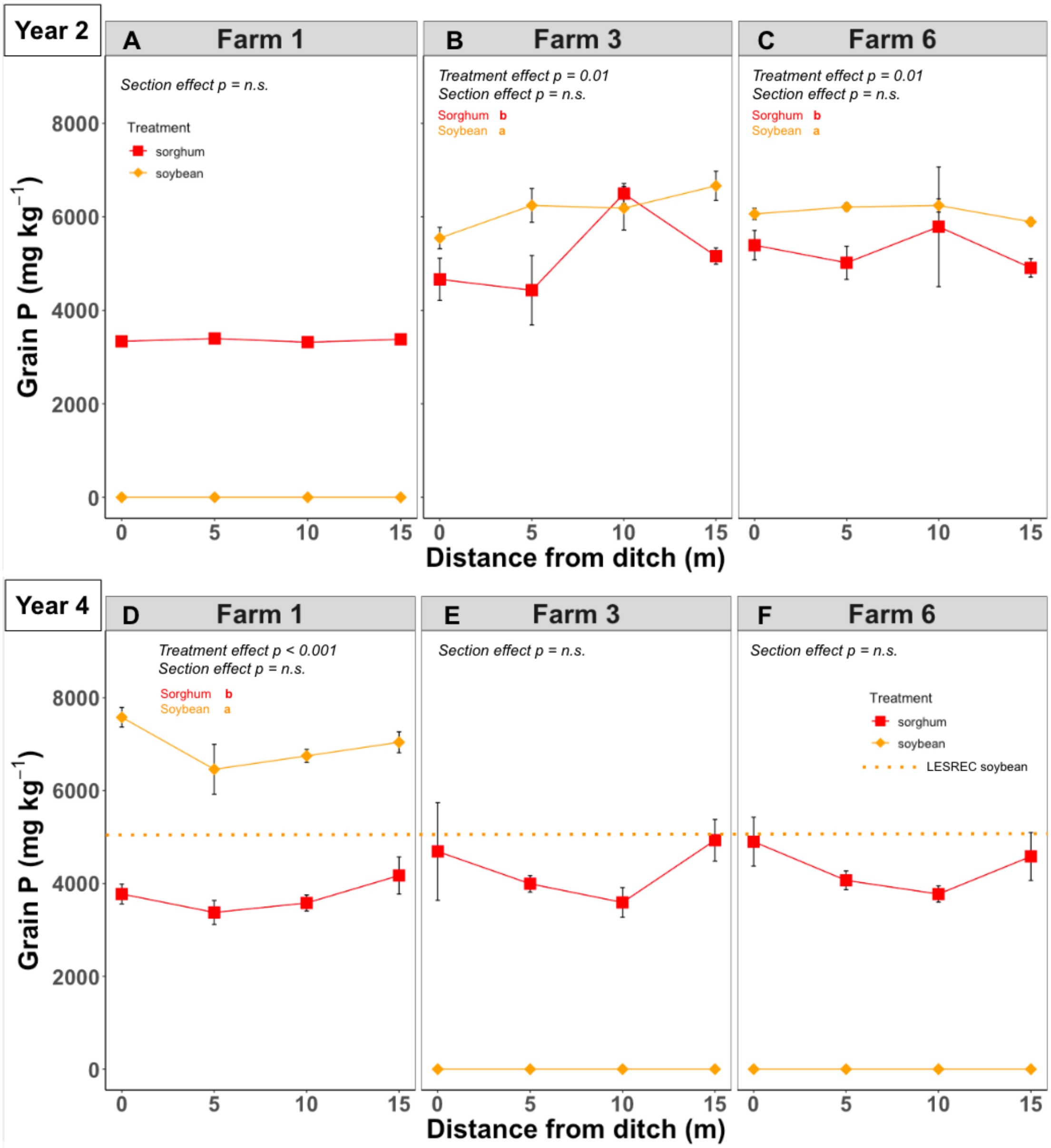


Figure 1. 11. Grain P concentrations along the salinity gradient at three saltwater-intruded farms on the Lower Eastern Shore of Maryland from year 2 to year 4. The x-axis is distance from the ditch in meters and the y-axis is grain P concentration in milligrams per kilogram. Treatments are represented by colors and symbols, where the sorghum treatment is a red square, and the soybean treatment is an orange diamond. The orange dotted line is LESREC soybean levels (because there was no saline ditch).

Grain yield

The grain yield at Farm 1 in 2019 showed sorghum having higher yield (79.5-123.9 bu acre^{-1}) than the county and no-salt control site averages (76 and 66 bu acre^{-1}, respectively), but in the following two years, sorghum yield decreased to below average (0.78-23.5 bu acre^{-1}; Table 1.6. Soybean yield did not meet county and LESREC averages (55 and 58 bu acre^{-1}, respectively) in any years at the saltwater-intruded sites (1.6-34.5 bu acre^{-1}; Table 1.6). Soybean grain was not collected at Farm 1 and Farm 3 in 2020 likely because of birds or deer eating the crops, so the values were zero. In 2021, the weeds at Farm 3 took over the field, and soybeans were unable to grow, and deer likely ate the grain at Farm 6.

Table 1. 6. Sorghum and soybean grain yields in bushels per acre from each research site on the Lower Eastern Shore of Maryland from 2019 to 2021 compared to Dorchester and Somerset County (MD) averages calculated in 2021.

Year	Farm	Sorghum yield (bu acre^{-1})	Soybean yield (bu acre^{-1})
2019	1	79.5-123.9	0
2019	3	19.0-46.1	12.2-28.5
2019	6	31.1-43.5	17.2-27.5
2019	LESREC	57.5	58.4
2020	1	0.8-4.8	0
2020	3	0.8-3.4	0
2020	6	1.3-10.6	2.1-7.8
2020	LESREC	69.1	47.2
2021	1	12.0-23.5	1.6-34.5
2021	3	0.2-2.1	0
2021	6	2.2-8.8	0
2021	LESREC	72.2	69.0
MD average		76.0	55.0

Grain P pools

Sorghum and soybean grain P pools showed different patterns than grain yield and grain [P] at all SWI sites for all years. In year 2, sorghum grain P was slightly higher (2.71-9.81 kg ha^{-1}) than soybean grain (3.46-5.95 kg ha^{-1}; $p=0.01$; Figure 1.12; Table 1.7), but there was no significant effect of distance from the ditch. There was no significant effect of treatment or distance from the ditch at Farm 3. Since there was only one species collected at Farm 1 and Farm 3 in year 3, there was no significant difference between species, but there was no significant effect of distance from the ditch either. Similarly, at Farm 6, there was no significance of distance from the ditch or treatment on grain P pools. The following year, there was only a significant difference between species at Farm 1 (*p=0.01*; Figure 1.12). There was a significant effect of distance from the ditch at Farm 3 (*p=0.01*; Figure 1.12) on grain P pools, but not at Farm 1 and Farm 6. Finally, the no-salt control (LESREC) soybean grain P pools were higher than pools at the SWI sites (22.5 kg grain-P ha^{-1}; Figure 1.12).

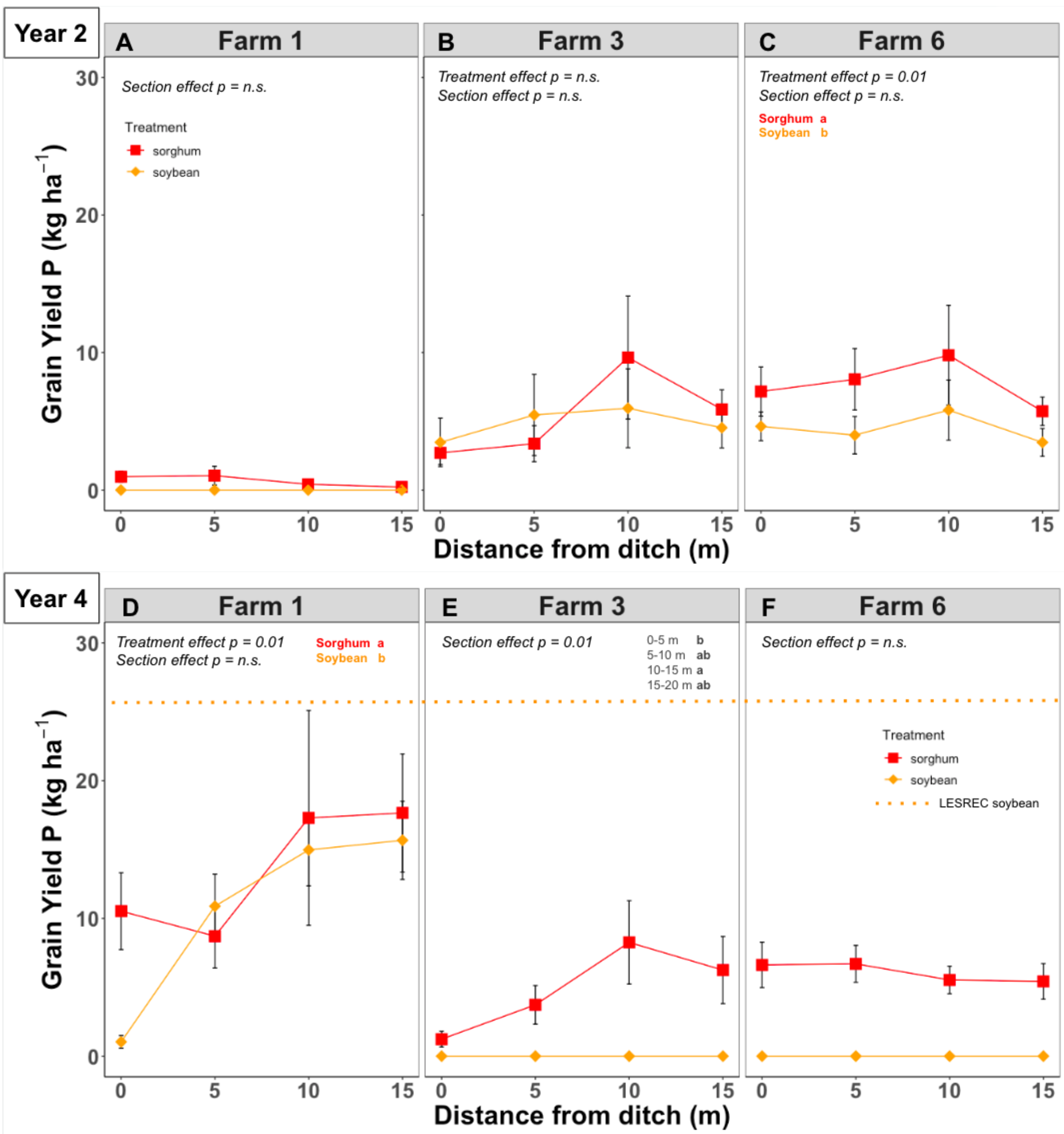


Figure 1. 12. Grain yield P pools along the salinity gradient at three saltwater-intruded farms on the Lower Eastern Shore of Maryland from year 2 (A-C) and year 4 (D-F). The x-axis is distance from the ditch in meters and the y-axis is grain yield P pools in kilograms per hectare. Treatments are represented by colors and symbols, where the sorghum treatment is a red square, and the soybean treatment is an orange diamond. The orange dotted line is LESREC soybean levels (because there was no saline ditch).

Table 1. 7. ANOVA results from LME models testing the effects of treatment and distance from the salinity source with the dependent variable as grain yield P pools from each site and each year followed by Tukey HSD results indicating statistical significance between variables.

ANOVA

Year 2 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Section	3	1.75	0.58	1.97	0.2

ANOVA

Year 2 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	1	0.91	0.91	1.09	0.32
Section	3	5.19	1.73	2.07	0.17

ANOVA

Year 2 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	1	9.93	9.93	10.46	0.01
Section	3	5.60	1.87	1.97	0.2

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Soybean	Sorghum	-1.11	0.34	0.001

ANOVA

Year 3 - Farm 1	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Treatment	1	7.52	7.52	8.35	0.02
Section	3	5.71	1.90	2.12	0.2

Tukey HSD

Treatment 1	Treatment 2	Mean Difference	Standard Error	*p-value*
Soybean	Sorghum	1.03	0.43	0.02

ANOVA

Year 4 - Farm 3	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Section	3	4.82	1.61	17.28	< 0.001

Tukey HSD

Section 1	Section 2	Mean Difference	Standard Error	*p-value*
5-10	0-5	2.14	0.31	< 0.001
10-15	0-5	1.10	0.31	0.002
15-20	0-5	1.49	0.31	< 0.001
10-15	5-10	-1.04	0.31	0.004
15-20	5-10	-0.65	0.31	0.2
15-20	10-15	0.39	0.31	0.6

ANOVA

Year 4 - Farm 6	*df*	Sum of Squares	Mean Square	*F*	*p-value*
Section	3	2.55	0.85	1.49	0.3

Plant and soil P concentration correlations

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 1.8). There were no clear relationships between plant leaf tissue P and topsoil available and total P, however, there was a strong positive correlation in the saltmarsh hay treatment only in the section furthest from the ditch ($r^2 > 0.3$; $p<0.005$ in both cases; Table 1.8).

Table 1. 8. Plant leaf tissue P and topsoil total and available P concentration correlations in year 4 for saltmarsh hay, switchgrass, and weeds in both the section closest to the ditch (0-5 m) and the section furthest from the ditch (15-20 m) with each saltwater-intruded site on the Lower Eastern Shore of Maryland averaged. Values are r^2 and symbols are p-values. Colors represent the strength of the correlation (-0.00*** = strong negative; -0.00** = moderate negative; -0.00* = negative; -0.00^ = weak negative; 0.00^ = weak positive; 0.00* = positive; 0.00** = moderate positive; 0.00*** = strong positive). *^ = p<0.05, * = p<0.01, ** = p<0.005, *** = p<0.001*

	Saltmarsh hay		Switchgrass		Weeds	
	0-5 m	15-20 m	0-5 m	15-20 m	0-5 m	15-20 m
plt P~soil P	0.10	0.35**	0.045	0.14	-0.082	-0.26
plt P~soil M3P	0.09	0.43***	0.11	0.17	-0.20	-0.21
soil P~soil M3P	0.87***	0.92***	0.96***	0.91***	0.91***	0.95***

Discussion

Our results have implications for how planting remediation and restoration species can draw down legacy soil P in coastal agricultural soils experiencing SWI.

After analyzing the aboveground biomass and soil beneath five proxy species for management practices (adapt, remediate, restore, abandon) on three farms experiencing SWI, we were able to determine if soil and biomass P pools changed along a SWI gradient. 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. Overall, we saw significant differences in soil and plant P pools among each of the SWI sites (Figs. 1.5, 1.6, 1.7), which indicates that even with differences among the saltwater-intruded farm fields (degree of SWI, precipitation, soil type, etc.), the uptake of P in biomass was consistent with the decrease in soil P concentrations and pools after three years of planting. On the Lower Eastern Shore, soils are high in legacy P due to historic application of poultry litter (Kleinman et al. 2013). Soils have high levels of both total and available P in the soil, which could be mobilized and released when fields are exposed to saltwater (Weissman et al. 2019), therefore it is important to determine best management practices that will draw down soil P. In this study, the adaptation species yield was too low, so this management practice would not be able to draw down P. However, after three years of planting both remediation and restoration species (switchgrass and saltmarsh hay), there was an increase in aboveground biomass P pools as biomass accumulated over time, and this was associated with a decline in soil total and available P pools.

Switchgrass is efficient at taking up soil and porewater P

We found switchgrass biomass had higher P pools than both saltmarsh hay and weeds from saltwater-intruded fields, suggesting that switchgrass could be used as a P removal tool. Previous research has shown switchgrass can accumulate large quantities of biomass after the establishment year and can last over 5-10 years (McLaughlin & Kszos 2005; Wullschleger et al. 2010; Richner et al. 2014). We found that high levels of switchgrass biomass were associated with large aboveground P pools, and soil total and available P pools decreased after three years of planting (Figs. 1.5, 1.6, 1.7), which indicates that species with high levels of biomass and deep root systems can uptake large amounts of soil P (Schmer et al. 2011; Basyal & Emery 2020). Porewater P was also lowest under the switchgrass plots after the second year and continued to decrease in the third year, and this was likely because of the decrease in soil P pools, so there is less P to be released as SRP (Meyerson et al. 1999). Therefore, planting switchgrass on coastal agricultural fields could be a mitigation 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).

Switchgrass is a practical option for planting in the mid-Atlantic. First, switchgrass 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 switchgrass 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). Switchgrass is a perennial that does not need to be replanted each year with relatively low seed cost and no need for fertilizer amendments (UMD Extension 2010; AGMRC 2018), incurring lower annual input costs than other crops. Finally, switchgrass can be used as a cellulose biofuel or poultry bedding, so landowners could continue to make a profit from planting, albeit at a reduced price per acre than typical corn and soybean crops ($265 per acre compared to $749 and $402 per acre, for corn and soybean respectively; Moyle et al. 2016; Purswell et al. 2020; UMD Extension 2022). In sum, this study showed switchgrass would provide a great tool to buffer against P loss, maintain low input costs, and be used as a biofuel or poultry bedding.

Saltmarsh hay is efficient at taking up P in soil with high salinity

We found saltmarsh hay biomass had intermediate biomass P pools compared to other treatments from saltwater-intruded fields, suggesting the potential to be used as a P removal tool. Previous research indicated that fields experiencing SWI will naturally transition to host more native marsh species such as saltmarsh hay (Gedan et al. 2019). Our work shows that the increase in saltmarsh hay biomass and aboveground P pools provide an opportunity to harvest this biomass to remove both sodium and P in the soil. At Farm 3 where salinity was the highest, saltmarsh hay had the highest biomass levels because of the higher tolerance to salt concentrations and was able to uptake similar levels of P as switchgrass (Figure 1.5). The uptake of P in saltmarsh hay biomass was associated with decreased levels of soil P pools and porewater [SRP] (Figs. 1.5, 1.6; Figure S1.7). Similar to switchgrass, our work indicates that allowing the natural conversion of fields to saltmarsh hay, or deliberately planting this species can serve as a tool for reducing P losses to nearby waterbodies by utilizing their high root porosity to maintain nutrient uptake (Burdick & Mendelssohn 1990; DeLaune et al. 2005).

Saltmarsh hay is a practical option for planting in high salinity soils on the Eastern Shore. First, saltmarsh hay 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 2022, Wetland Reserve Easements; WRE 2022). Finally, planting saltmarsh hay (*Spartina patens*) may facilitate the transition to tidal marshes that can protect coastlines from flooding by reducing strong winds and storm surges 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 the Delaware, Maryland, Virginia (DMV) area 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 (Smith & Katz 2013; Narayan et al. 2017; USGCRP 2017; Li et al. 2020). Salt marsh presence reduces average annual flood losses by 18-70% (Narayan et al. 2017). In sum, restoring farm fields has benefits for nearby waterbodies by taking up soil P, landowners by maintaining revenue, and communities by protecting coastlines.

Grain crops will not maintain high yield with increasing SWI

The alternative grain crops tested in our field trials demonstrated efficient grain yield P uptake rates when the crops were able to survive. Sorghum is a salt-tolerant crop (relative to corn and soybeans) that maintains a good yield under increasing saline conditions (Grieve et al. 2012). Our data suggest that sorghum is efficient at taking up soil P pools, and we predict that soil P levels may decrease if the total aboveground biomass of the grain crop was removed at harvest. Utilizing this mitigation strategy would allow continued farming operations on the land because there is a market for sorghum as feed on the Lower Eastern Shore of MD (J. Miller, personal communication). Unfortunately, the value for the rest of the plant beyond the grain and the cost for specialized equipment to harvest the whole plant does not outweigh the option to grow alternative species (e.g., switchgrass, saltmarsh hay). Additionally, we planted Cl-excluding soybeans, which had lower yield than expected even under increasing saline conditions. Both sorghum and soybean total aboveground biomass is usually returned to the field as chaff during the harvesting process, so significant removal of P from the system is not likely with the adaptation management strategy. Finally, our research shows that these alternative grains, although efficient in taking up P, will not maintain high yield with increasing SWI.

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. This work on agricultural fields experiencing SWI aims to examine the survival of different plant species and better understand nutrient dynamics in plant biomass, soils, and porewater (Tully et al. 2019; Weissman & Tully 2020). We show that remediation and restoration management practices are efficient in taking up soil and porewater P that may ultimately decrease P runoff into the Chesapeake Bay. Overall, patterns of P pools were influenced by biomass levels, and after three years of planting, switchgrass and saltmarsh hay produced more biomass than weeds, sorghum, or salt-tolerant soybeans, and their production was influenced by the degree of SWI at each site. In switchgrass and saltmarsh hay, the uptake of P was consistent with the decrease in soil P and M3P concentrations and pools, reducing the levels of P to be released as SRP by about 20%. In sum, planting either switchgrass or saltmarsh hay would have economic and environmental benefits in mitigating the adverse effects of SWI. 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 if farmers' goals are to continue crop production while avoiding addition of excess nutrients to the Bay. The findings of this research will help inform farmer management decisions given economic and environmental objectives and tradeoffs.

Chapter 2: Cation Chapter: The effects of saltwater on plant and soil cation dynamics

Abstract

Rising sea levels, storms, and perigean spring tides, push saltwater into coastal agricultural fields. This phenomenon, known as saltwater intrusion (SWI), alters nutrient cycling and damages crop yields. Researchers and farmers alike are looking for solutions to adapt to and mitigate the effects of SWI. Landowners may respond by altering their management practices. Farmers may 1) *adapt* by planting a salt-tolerant crop, 2) attempt to *remediate* soils with trap crops, 3) *restore* native marsh grasses, or 4) *abandon* fields altogether. This study focused on the survival of different crops and plant treatments under saltwater-intruded conditions and the potential for these plants to survive and to remove excess nutrients (e.g. calcium, magnesium, potassium, sodium) from the soil. We observed patterns of regulation in nutrient uptake in all species (saltmarsh hay, switchgrass, and weeds). Overall, we show restoration management practices are efficient at taking up soil Na and may decrease Na concentrations and pools in the soil over time, which may benefit both soil and plant health on saltwater-intruded fields. In sum, results from this study will help inform new management practices to increase soil health and maintain crop yields. The goal of this work is to guide local best management practices and potential easement opportunities for landowners facing SWI.

Introduction

Low-lying coastal regions in the Mid-Atlantic are vulnerable to high tide flooding and sea level rise (SLR; Sweet et al. 2018). In the Chesapeake Bay, sea levels are rising at twice the average global rate (3.2 compared to 1.7 mm per year; NOAA 2016). Saltwater intrusion (SWI) along the Eastern Shore of Maryland will only worsen as a result of SLR, the frequency of storm events, and drought (Neubauer et al. 2013; Sweet et al. 2018; Tully et al. 2019). Ditches, built to drain farm fields, can facilitate SWI during major storm events and high tides (Bhattachan et al. 2019). Recent work suggests that as much as 30% of the land area in Dorchester and Somerset counties are at risk of SWI (Epanchin-Niell 2019). Saltwater is a complex mixture of chloride (Cl), sodium (Na), sulfate, and other ions; it alters soil chemistry when introduced to farm fields. Chemical shifts caused by SWI can suppress agricultural productivity by impeding water infiltration through soil aggregate dispersion, salt imbalances (e.g. calcium [Ca], magnesium [Mg], potassium [K]) in crop tissues, and overall loss of farmland through the flooding of fields and conversion into tidal marshes (Warrence et al. 2002; Gedan et al. 2019; Tully et. al 2019; Tanu 2022). Indeed, about 2% of agricultural land in Somerset Co., MD converted to marshland over just eight years (5.7 km^2; 2009-2017; Gedan et al. 2020). As sea levels continue to rise and our coastlines change, we need to implement new management strategies to reduce salt damage in agricultural soils and to better understand the influence of SWI on salt dynamics (Ca, K, Mg, Na) in affected soils and plant leaf tissue.

Globally, Na toxicity threatens 25-33% of all agricultural land, contributing to $12 to $27.3 billion in annual economic losses based mainly on reduced crop yields (Ghassemi et al. 1995; Rengasamy 2010; Kronzucker & Britto 2011; Qadir et al. 2014). In arid regions, Na is introduced to the system when crops are irrigated with water pumped from mineral-rich aquifers (Wichelns & Qadir 2015; Machado & Serralheiro 2017). In saltwater-intruded systems, Na is introduced via seawater. The introduction of Na can replace Ca and Mg on the soil exchange complex, causing the dispersion of soil colloid particles, and creating unstable microaggregates that facilitate soil compaction and decrease water infiltration (Machado & Serralheiro 2017; Weil & Brady 2017). Additionally, as fields salinize, the increase in dissolved salt concentrations can impede the movement of water and nutrients (e.g., nitrogen) across root membranes in plants (osmotic stress), which can inhibit water transport, photosynthesis, and ultimately, plant growth (Greenway & Munns 1980; Hasegawa et al. 2000; Netondo et al. 2004). The presence of high levels of Na also decreases plant growth by causing cation deficiency, or inhibiting the uptake and accumulation of Ca, Mg, and K by roots (Netondo et al. 2004; Paranychianakis & Chartzoulakis 2005). Plants employ a variety of strategies to deal with this salt stress, which will affect coupled soil-plant cation dynamics in fields undergoing SWI. In response to a decrease in crop yield due to increased Na concentrations, scientists are developing salt-tolerant varieties of many agronomic crops such as potatoes (*Solanum tuberosum*), soybeans, and rice (Deinlein et al. 2014, Londhe 2016; Table 2.1).

Some plants (e.g., rice [*Oryza sativa*], barley [*Hordeum vulgare*]) can trigger osmotic adjustment and accumulate solutes to change the osmotic pressure allowing water to enter the roots (Chen & Jiang 2010; Horie et al. 2012; Munns et al. 2020). Lost energy used to make osmotic adjustments can result in reduced growth. Other plants (e.g., saltmarsh hay [*Spartina patens*], switchgrass [*Panicum virgatum*], quinoa [*Chenopodium quinoa*]) are able to excrete excess Na through specific salt glands or salt bladders onto their leaf surface (Kim et al. 2012; Shabala et al. 2014). Species such as barley and sorghum [*Sorghum bicolor*] are able to sequester and concentrate Na in vacuoles in the stems and roots (Matsushita & Matoh 1991). These same plants might be able to uptake high levels of Na to remove them from the soil, decreasing soil dispersion, and increasing overall plant productivity (Tobias et al. 2010; Fageria et al. 2011). Dominant agronomic crops on Maryland's Eastern Shore (e.g., corn [*Zea mays*] and soybean [*Glycine max*]) do not have any of these salt stress strategies (i.e., salinity threshold of 1-3 ppt; Tanji & Kielen 2003) and therefore yields are suppressed under SWI (Table 2.1). Elevated salinity levels have forced farmers to reconsider their management practices. Landowners in coastal regions facing SWI have several choices: <u>adapt</u> to increasing salinity levels by farming with salt-tolerant crops, <u>remediate</u> fields for a few years before returning to agriculture, <u>restore</u> the land to native marshes, or <u>abandon</u> salt-damaged fields entirely (Baker et al. 2014; Tully et al. 2019). This study will help us understand the influence of these management practices on biogeochemical cycling, specifically Ca, K, Mg, and Na in soils and plants.

Table 2. 1. Common crops and species uses, salinity and electrical conductivity thresholds, and stress management strategies (how each species responds to high salinity).

Species	Uses	Salinity Threshold (ppt)	Electrical Conductivity Threshold (dS/m)	Stress Management
Corn (*Zea mays*)	Chicken feed	0.9	1.8	none
Soybean (*Glycine max*)	Chicken feed	3.0	5.0	none
Wheat (*Triticum spp.*)	Crop/food crop Animal feed	3.3	6.1	salt-excluder (Munns et al. 2020)
Barley (*Hordeum vulgare*)	Crop Beer brewing	4.4	8.0	salt-excluder (Greenway 1965) salt-includer (Mian et al. 2011)
Quinoa (*Chenopodium quinoa*)	Potential food crop	18.9	30.5	salt-excluder (Yuan et al. 2016)
Rice (*Oryza sativa*)	Potential food crop	0.6	1.2	salt-excluder (Garcia et al. 1997)
Salt-tolerant potatoes (*Solanum tuberosum*)	Potential food crop	5.7	10.1	salt-includer
Cl-excluding soybean (*L-Glycine max*)	Chicken feed	6.0	10.6	salt-excluder (Cao et al. 2018)
Sorghum (*Sorghum bicolor*)	Chicken feed Fuel	3.7	6.8	salt-includer (Bafeel 2014) salt-excluder (Sun et al. 2018)

Switchgrass (*Panicum virgatum*)	Biofuel Poultry bedding	17.5	28.4	salt-excluder (Kim et al. 2012)
Saltmarsh hay (*Spartina patens*)	Poultry bedding	63.0	88.6	salt-excluder (Bradley and Morris 1991)

We established a study to investigate the concentrations and pools of salts (Ca, K, Mg, and Na) in the soil and plant leaf tissue of four different management strategies represented by five proxy species: adapt (e.g., Cl-excluding soybeans [*Glycine max*], sorghum [*Sorghum bicolor*]), remediate (e.g., switchgrass [*Panicum virgatum*]), restore (e.g., saltmarsh hay [*Spartina patens*]), and abandon (e.g., weeds [*Panicum dichotomiflorum*]) on fields experiencing SWI. We expected to see that increased concentrations of Na in the soil would prevent the uptake of Ca (Lazoff & Bernstein 1999) and K into the plant (Munns & Tester 2008). We also predicted a decrease in biomass due to excess Na due to the osmotic and toxic effects of Na on plant growth and cell function (Flowers et al. 1991). The objectives of this research were to: (1) quantify soil, leaf tissue and grain salt (Ca, K, Mg, and Na) concentrations and pools among adaptation, remediation, restoration, and abandonment systems and species; (2) determine if field management practices (e.g., adapt, remediate, restore, abandon) change soil salt concentrations and pools along a SWI gradient; and (3) determine if plant biomass salt pools change along a SWI gradient. We also examined the relationship between soil and plant salt concentrations, as well as each of the salt concentrations to each other (e.g., soil Na vs. plant Na and Ca vs. Na). This research will help us better understand salt dynamics in the soil and plant leaf tissue on coastal agricultural fields experiencing SWI. Results from this study will help inform new management practices to improve nutrient uptake, increase soil health, and maintain crop yields.

Methods

Experimental design

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 (LESREC in November; Figure 2.1). One field (Farm 6) was located near Cambridge, MD in Dorchester Co., MD (38.5632° N, -76.0785° W). The other two fields (Farm 1 and Farm 3) were located in Somerset Co., MD, with one near Princess Anne, MD (38.1608°N, -75.7907 °W) and the other near Crisfield, MD (37.9834°N, -75.8545°W). A fourth site was located inland and did not experience saltwater intrusion (established as a no-salt control) 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 LESREC, the control site, soils were Mattapex silt loams (mesic Aquic Hapludults). Baseline soil properties are shown in Table 2.2.

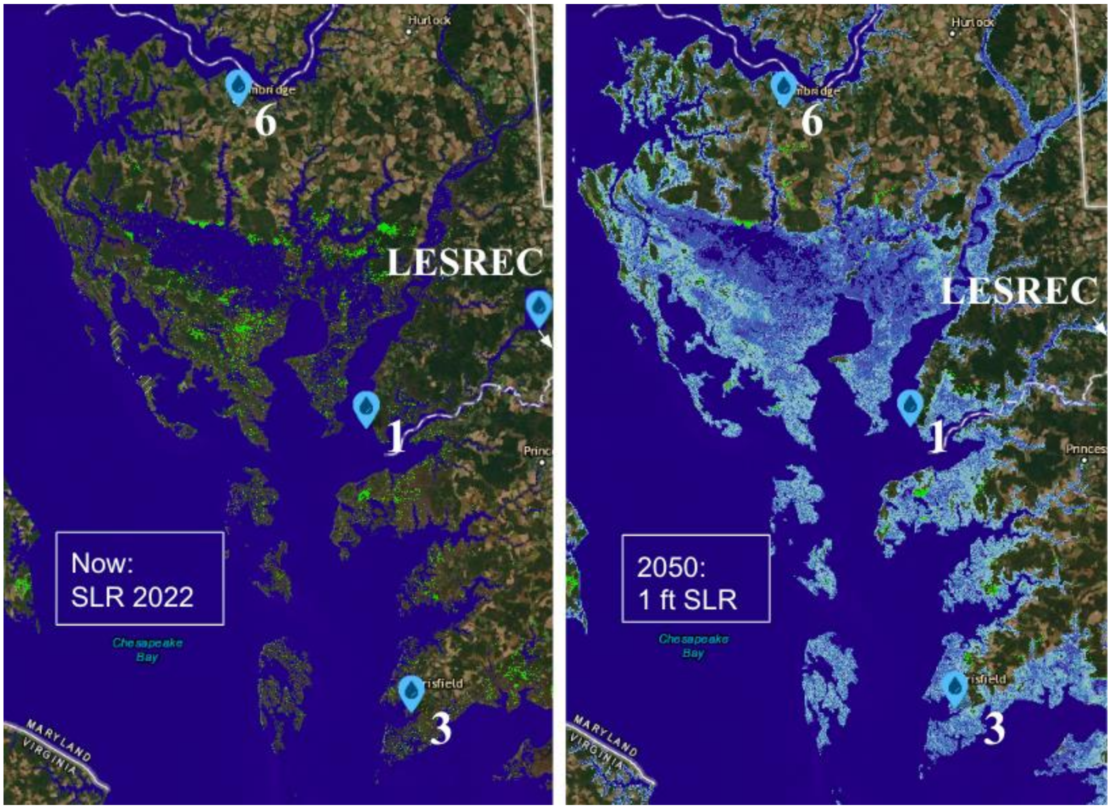


Figure 2. 1. Sea level rise impacts in 2022 compared to 2050 on coastal agricultural field sites located on the Lower Eastern Shore of Maryland. Sea level rise impacts (land coverage) in 2022 compared to projected 1.0 ft (0.305 m) of SLR by 2050 (NOAA 2022). Agricultural field sites indicated by blue points labeled 1, 3, 6, and LESREC (no salt control).

Plots 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 control). 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) (Figure 2.2). There was a 0.5 m buffer between treatments and a 1 m buffer between each plot.

The no-salt control field site was established in November of 2018 and has a layout of 16 plots with the same treatments (1-3) and a salt-tolerant crop rotation (chloride-excluding soybean, sorghum, and barley) with each entry point present every year, and a remediation species (treatment 4; switchgrass). As the field was not affected by saltwater, we did not install the restoration species (treatment 5; saltmarsh hay). Additionally, we did not allow for natural recruitment of weeds (treatment 6; native grasses), as this was not recommended by the LESREC farm manager in order to maintain the integrity of the fields for future researchers. Plots in the no-salt control site were also 3 m x 20 m with a 0.5 m buffer between treatments and 1 m buffer between blocks.

Table 2. 2. Soil characteristics of field sites on the Lower Eastern Shore of Maryland collected from 0-10 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

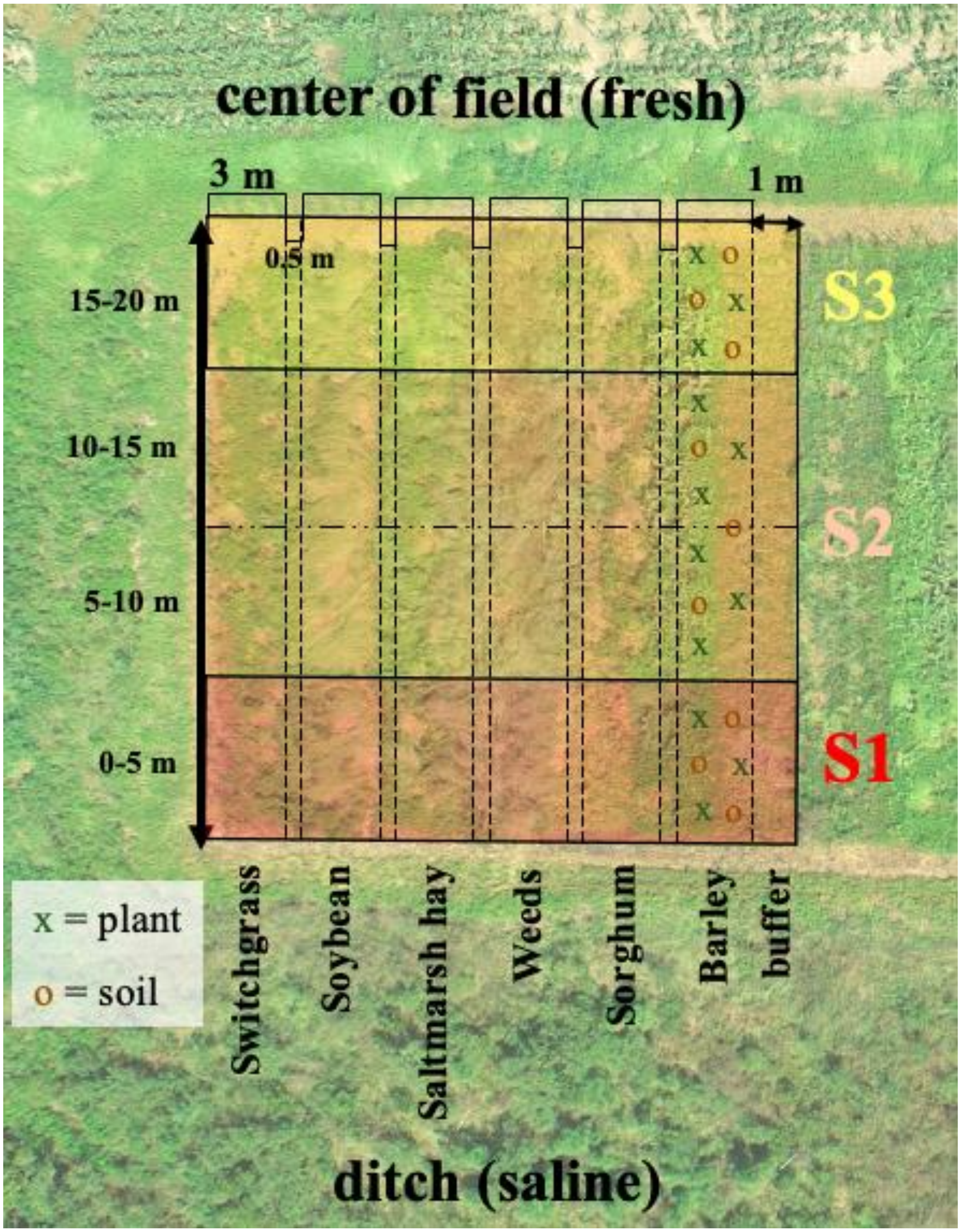


Figure 2. 2. Sampling design for soil cores and plant biomass in an example replicate block in a saltwater-intruded field site on the Lower Eastern Shore of Maryland. Soil cores (represented by brown o) were collected from each plot to 60 cm (0-10, 10-20, 20-30, 30-60 cm; three replicates for each depth) 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).

Field activities

The full crop rotation included sorghum, winter barley (*Hordeum vulgare*), double crop soybeans, sorghum, and full season soybeans. The crops were split into three plots differing in the starting point of the rotation (Figure 2.2). All sorghum and soybean plots were sprayed with glyphosate (~0.45 kg active ingredient per acre) in early May each year (2018-2021) to suppress weeds. Each year, sorghum and chloride-excluding soybean seeds were sown in the field in mid-May to early-June using a 1.52-m Tye drill. Sorghum and chloride-excluding soybean plots were planted in 38.1 cm rows with sorghum seeding rates at 80K acre^{-1} in 2018, then 120k acre^{-1} from 2019-2022 and soybean seeding rates at 195K acre^{-1} each year. Chloride-excluding soybean plots received potassium as potash (KCl; 39.2 kg ha^{-1}) and sorghum received N as urea (84 kg ha^{-1}) each year at planting. Sorghum was harvested from all sites in late-September to early-October each year. Soybeans were harvested in late October from all sites each year.

Switchgrass and saltmarsh hay were started from seed in plug trays and grown in the 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 switchgrass and saltmarsh hay at Farm 1 and only switchgrass at LESREC and gap-filled in April of 2020. This process was repeated in 2020 as necessary to ensure good stands of our perennial treatments.

Soil collection

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 bulk density cores were collected at 0-10, 10-20, 20-30, and 30-60 cm. Soil samples were collected for bulk density in 2019 from each block at Farm 1 and LESREC 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. In both cases, soils were returned to the University of Maryland College Park and dried at 105ºC for 7 days. Core volume was used to calculate bulk density (g cm^{-3}). Soil bulk density at LESREC 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 LESREC in November of 2018 using the same method as above. Baseline soils were analyzed for total C and N using dry combustion (LECO TruMac N, St. Joseph, MI). Soil available P, Ca, K, Mg, Na, and S were determined on an ICP following Mehlich-3 extraction (Mehlich 1984). Texture was determined using the hydrometer method (Gee and Bauder 1979; Table 2.2).

Starting in November 2019, we collected 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 (Figure 1.2). Since the LESREC site did not have a salinity gradient, we sampled only at the plot-level. Because we found that soil cation concentrations approached zero below a depth of 30 cm in 2018 and 2019, we only analyzed soils collected from 0-30 cm depths in 2021 and 2022.

Annual soil analysis

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 two days prior to analysis (60 ºC). Each depth was analyzed for texture (hydrometer method; Gee and Bauder 1979) and exchangeable Ca, K, Mg, and Na (Mehlich-3 extraction; Mehlich 1984). Exchangeable Ca, K, Mg, and Na in the 2021 soil samples were analyzed using air-acetylene flame Atomic Absorption Spectrometry (AAS; PerkinElmer Waltham, MA) and Inductively Coupled Plasma Mass Spectrometry (ICP-MS; Agilent Technologies Santa Clara, CA). Upon initial inspection, available Ca, K, Mg, and Na levels in the 30-60 cm depth were low and similar across site, treatment, and section, so we focused on soil cation concentrations in the 0-30 cm depths for 2018 and 2021 collections. We also found that there was not a significant difference between section across site, treatment, and depth, so we analyzed soil cation concentrations from the 0-5 m and 15-20 m distance from the ditch for the 2021 collection.

To calculate soil cation pools, the cation concentrations from each individual depth increment (e.g., 0-10, 10-20, and 20-30 cm) was multiplied by the depth of the segment (in cm) and the bulk density (g cm^{-3}).

Ground and dried soil samples were 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).

Plant collection

We used agronomic methods of biomass estimation for crops (*adapt*), and ecological methods for all other treatments (*remediate*, *restore*, *abandon*). Biomass was collected from the weeds, saltmarsh hay, and switchgrass plots from the saltwater-intruded sites (Farm 1, Farm 3, Farm 6) in August 2018, 2019, 2020, and 2021. 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 brackish water ditch that borders the study sites to capture the effects of salinity (Figure 2.2). Switchgrass was collected from LESREC at the plot-level. Total biomass samples were clipped within 2 cm of the soil surface, weighed wet, allowed to air-dry, sorted to species, and then clipped again into sub-samples for processing. Sub-samples were oven-dried for at least 48 h at 60 ºC and weighed for conversion of total plant biomass to dry-weight-equivalent (DWE). 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 sample processing. In 2018, fall panic grass (*Panicum dichotomiflorum*), a native variety of switchgrass, was a species common to nearly all weed plots across all sites therefore, was retained for tissue analysis. In the following years (2019-2022), the most common species was crabgrass (*Digitaria sanguinalis*), so this species was retained for tissue analysis.

Plant analysis

Oven-dry (60 ºC) plant sub-samples from each treatment and year were homogenized by grinding through a 2-mm mesh screen (Wiley Mill, Swedesboro, NJ). Only plant samples from the section closest to the ditch (0-5 m) and the section furthest from the ditch (15-20 m) were analyzed. Plant tissue was pre-digested in concentrated nitric acid then digested in a heat block for 90 min after 30% hydrogen peroxide was added (Luh & Schulte 1985). National Institute of Standards and Technology (NIST) Standard Reference Material (SRM 1547 peach leaves) were included in every digestion to assure the effectiveness (95% confidence interval) of the nitric acid digestion. Digestion solutions were analyzed for Ca, K, Mg, and Na on an ICP Emission Spectroscopy (Agilent Technologies Santa Clara, CA). Biomass Ca, K, Mg, and Na pools were calculated by multiplying nutrient concentrations by aboveground biomass (DWE) and yields (DWE; kg ha^{-1}) collected from the plots.

Statistical analysis

A linear mixed-effects (LME) model (*lme4* package for R; Bates et al. 2013) was used to examine the effect of species (e.g., switchgrass, saltmarsh hay, weeds) on soil exchangeable Ca, K, Mg, and Na concentrations and pools averaged at the plot-level. We examined each year and site individually with species (treatment), distance from 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 year, site, distance from ditch, and depth individually (e.g. soil bioavailable Na in 2021 from Farm 3 in S1 at 0-10 cm). Soil Ca, K, Mg, and Na concentrations were measured in 2018 (to understand baseline cation levels) and 2021 (after a full crop rotation). Tukey *post-*

hoc comparisons were used to determine significant differences among treatments. Finally, univariate regression was used to determine if there is a relationship between topsoil EC (0-10 cm) and topsoil cation concentrations.

A linear mixed-effects (LME) model (*lme4* package for R; Bates et al. 2013) was used to examine the effect of species (e.g., switchgrass, saltmarsh hay, weeds) on leaf tissue Ca, K, Mg, and Na concentrations and pools averaged at the plot-level. We examined each year and site individually with species (treatment) and distance from ditch as the main effects and block as the random effect. Tukey *post-hoc* comparisons were used to determine significant differences among species. Finally, univariate regression was used to determine if there is a relationship between topsoil EC (0-10 cm) and plant leaf tissue cation concentrations. All statistics were run in the R environment for Mac (R Core Team 2021).

Correlation tests (*PerformanceAnalytics* package for R; Peterson et al. 2020) were performed to determine the relationship between different variables. We grouped all saltwater-intruded sites together and ran the test considering distance from ditch (section) as a variable for the plant leaf tissue and depth in addition to distance from ditch as variables for the soil from year 1 to year 4. The first test was between soil available cation concentrations (e.g., does the concentration of Na in the soil change the concentration of K in the soil). The second test was between plant leaf tissue cation concentrations (e.g., is saltmarsh hay Na levels influencing how much Ca is in their leaf tissue). The final test was between soil available cations and plant leaf tissue cations (e.g., will increasing concentrations of Na in the soil effect the concentration of Ca in the leaf tissue).

Results

Soil available cation concentrations

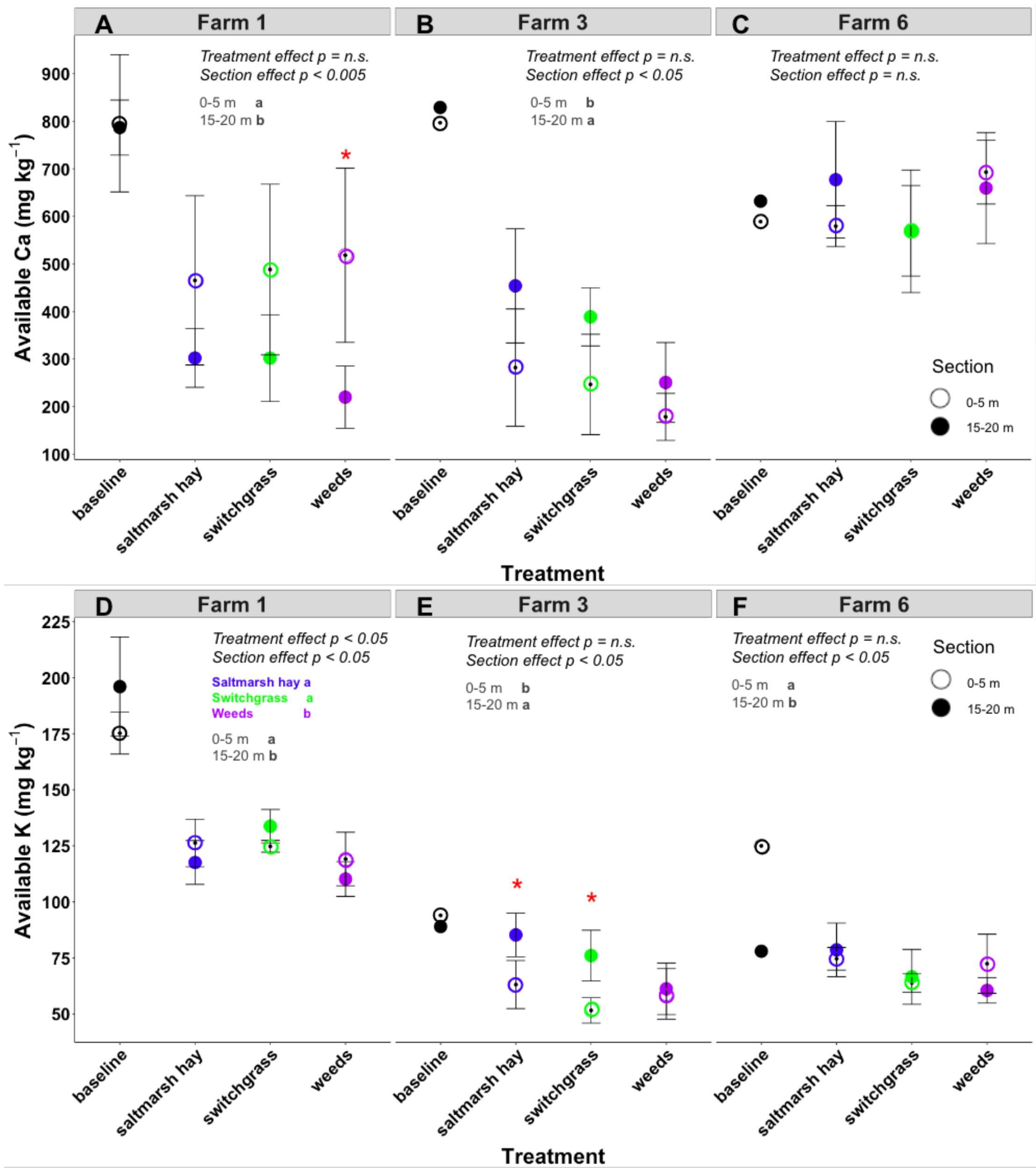
In the soil, concentrations of Mehlich-3 extractable Ca ([M3Ca]) decreased by 50-60% in the 0-10 cm depth at two of the sites after three years of planting (51.6% at Farm 1 and 63.1% at Farm 3; Figure 2.3). There was no change in Mehlich-3 extractable Mg ([M3Mg]) over time at any of the saltwater-intruded sites, however, Farm 6 there was an increase in soil [M3Mg] in the 0-10 depth after three years of planting (149 to 275 mg Mg kg^{-1}; $p<0.01$; Figure 2.3). There was no clear relationship between [M3Ca] and [M3Mg] and treatment across all saltwater-intruded sites. At Farm 1, distance from the ditch affected how much [M3Ca] and [M3Mg] was in the soil, where the section closest to the ditch had higher concentrations of M3Ca and M3Mg than the section further from the ditch ($p<0.005$ in both cases). At Farm 3, distance from the ditch and depth were significant ($p<0.05$ in both cases) where the soil closer to the ditch had higher [M3Ca] than the section further from the ditch and the 0-10 cm depth had higher [M3Ca] than the deeper depths (10-30 cm). There was also a depth effect at Farm 3, where [M3Mg] decreases with depth ($p=0.01$). At Farm 6, there was no change in [M3Ca] over time, however, there was a significant effect of depth where [M3Ca] increased with depth ($p<0.01$; Figure 2.3), but there was no effect of SWI gradient. Finally, Farm 6 was the only site that had more [M3Mg] than the no-salt control.

Concentrations of Mehlich-3 extractable K ([M3K]) decreased by 25-30% at all saltwater-intruded sites after three years of planting (34.3% at Farm 1, 27.8% at Farm 3, 31.6% at Farm 6; Figure 2.3). Treatment was only significant at Farm 1 where saltmarsh hay and switchgrass had similar [M3K] in all soil depths (71.8-126.3

mg K kg^{-1} and 85.5-133.8 mg K kg^{-1}, respectively) compared to the weeds (63.7-119.1 mg K kg^{-1}; $p<0.05$; Figure 2.3).

Across all years among the SWI sites, soil depth was significant, where the 0-10 cm had the highest [M3K] (72.4-137.9 mg K kg^{-1}) compared to the deeper soil depths (10-20 and 20-30 cm; 37.9-102.3 mg K kg^{-1} and 35.0-88.9 mg K kg^{-1}; $p<0.005$ in all cases). In 2021, distance from the ditch was significant at all saltwater-intruded sites, where Farm 1 and Farm 6 showed high [M3K] in the section closest to the ditch, but the opposite was seen at Farm 3, where [M3K] was actually higher further from the ditch.

Contrastingly, concentrations of Mehlich-3 extractable Na ([M3Na]) increased by 35-60% in the section closest to the ditch (43.8% at Farm 1, 59.3% at Farm 3, 36.8% at Farm 6) and by 60-90% furthest away from the ditch (88.3% at Farm 1, 90.1% at Farm 3, 59.7% at Farm 6; Figure 2.3) at all saltwater-intruded sites from year 1 to year 4. Distance from the ditch was significant at all SWI sites in 2021, where the section closest to the ditch had the highest [M3Na] in the soil (589.9-936.5 mg Na kg^{-1}; $p<0.05$ in all cases). Treatment was only significant at Farm 6 where saltmarsh hay and weeds had similar [M3Na] in all soil depths (468.4-689.7 mg Na kg^{-1} and 424-764.7 mg Na kg^{-1}, respectively) compared to switchgrass (385.3-589.9 mg Na kg^{-1}; $p<0.05$; Figure 2.3). Across all years, depth was significant at all SWI sites where the deepest depth (20-30 cm) had the highest [M3Na] in 2018, and the shallow depth (0-10 cm) had the highest [M3Na] in 2021 ($p<0.05$ in all cases). Finally, the saltwater-intruded sites had significantly more [Na] in the soil than the no-salt control (450 mg Na kg^{-1} and 31 mg Na kg^{-1}, respectively; $p<0.001$).


A
Farm 1
B
Farm 3
C
Farm 6
Treatment effect p = n.s.
Section effect p < 0.005
0-5 m a
15-20 m b
Treatment effect p = n.s.
Section effect p < 0.05
0-5 m b
15-20 m a
Treatment effect p = n.s.
Section effect p = n.s.
Available Ca (mg kg-1)
Section
0-5 m
15-20 m
baseline
saltmarsh hay
switchgrass
weeds
Treatment
D
Farm 1
E
Farm 3
F
Farm 6
Treatment effect p < 0.05
Section effect p < 0.05
Saltmarsh hay a
Switchgrass a
Weeds b
0-5 m a
15-20 m b
Treatment effect p = n.s.
Section effect p < 0.05
0-5 m b
15-20 m a
Treatment effect p = n.s.
Section effect p < 0.05
0-5 m a
15-20 m b
Available K (mg kg-1)
Treatment

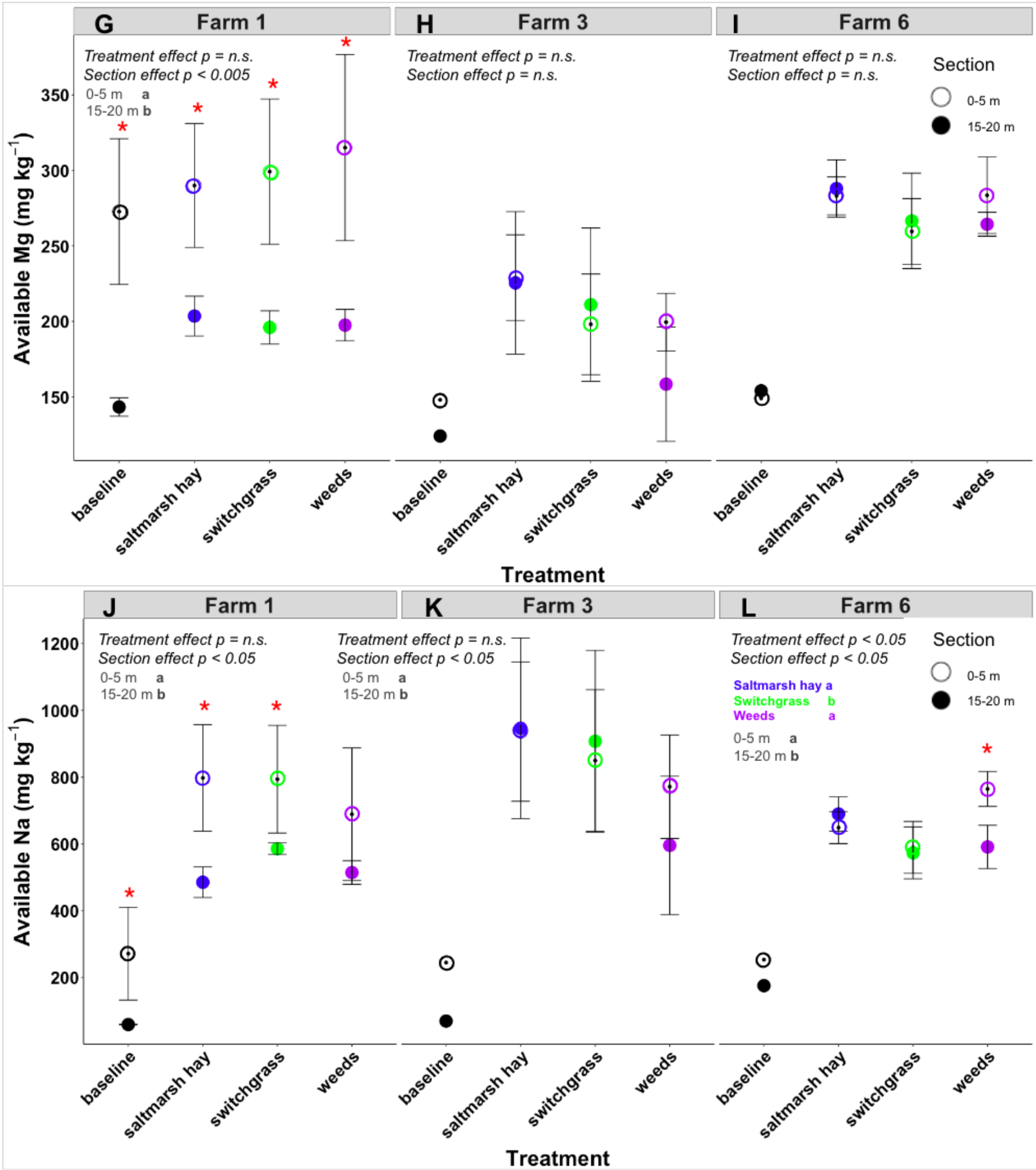


Figure 2. 3. Topsoil (0-10 cm) available cation concentrations at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland from year 1 (baseline) to year 4. The x-axis is treatment and the y-axis is Mehlich-3-extractable Ca (A-C); K (D-F); Mg (G-I); and Na (J-L) in milligrams per kilogram. Treatments are represented by colors, where the saltmarsh hay treatment is blue, the switchgrass treatment is green, and the weeds treatment is purple and section is represented by open or closed circles, where open circles are 0-5 m away from the ditch and closed circles are 15-20 m away from the ditch.

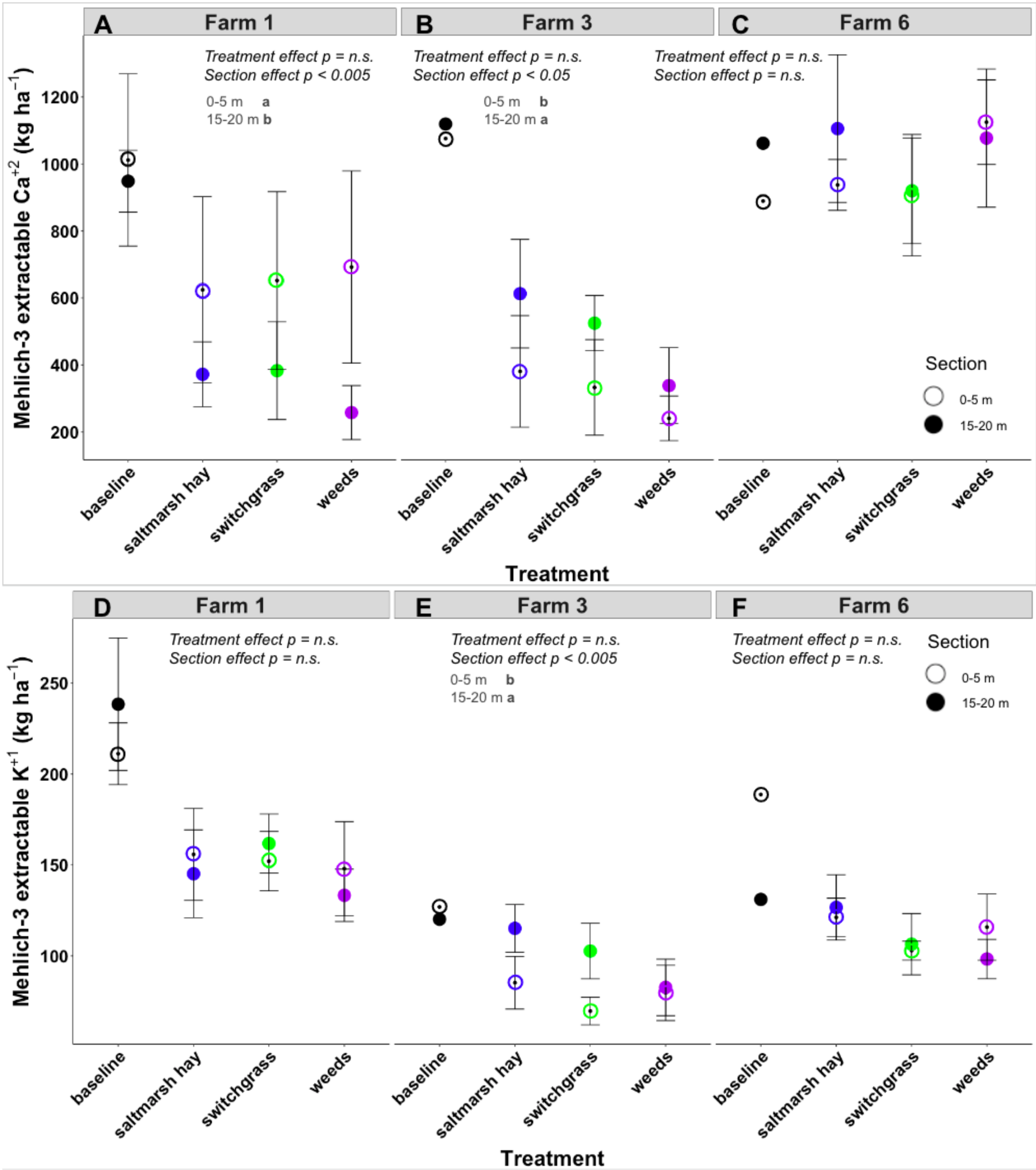
Soil available cation pools

In the soil, Mehlich-3 extractable Ca pools decreased by 50-60% in the 0-10 cm depth at two of the sites after three years of planting (49.8% at Farm 1 and 63.6% at Farm 3; Figure 2.4). There was an increase in Mehlich-3 extractable Mg pools over time at all of the saltwater-intruded sites, especially at Farm 6 where there was an increase in soil M3Mg pools in the 0-10 depth after three years of planting (225 to 444 kg Mg ha^{-1}; $p<0.01$; Figure 2.4). There was no clear relationship between M3Ca and M3Mg pools and treatment across all saltwater-intruded sites. At Farm 1, distance from the ditch affected M3Ca and M3Mg pools, where the section closest to the ditch had higher pools of M3Ca and M3Mg than the section further from the ditch ($p<0.005$ in both cases). At Farm 3, distance from the ditch and depth were significant ($p<0.05$ in both cases) where the soil further from the ditch had larger M3Ca pools than the section closer to the ditch and the 0-10 cm depth had larger M3Ca pools than the deeper depths (10-30 cm). There was also a depth effect at Farm 3, where M3Mg pools decrease with depth ($p=0.01$). At Farm 6, there was no change in M3Ca pools over time, however, there was a significant effect of depth where M3Ca pools increased with depth ($p<0.01$; Figure 2.4), but there was no effect of distance from the ditch. Finally, Farm 6 was the only site that had greater M3Mg pools than the no-salt control.

Pools of Mehlich-3 extractable K decreased by 25-30% at all saltwater-intruded sites after three years of planting (34.3% at Farm 1, 27.8% at Farm 3, 31.6% at Farm 6; Figure 2.4). Treatment was only significant at Farm 1 where saltmarsh hay and switchgrass had similar M3K pools in the soil below (85.8-155.8 kg K ha^{-1} and 110.3-161.8 kg K ha^{-1}, respectively) compared to the weeds (81.8-147.9 kg K ha^{-1};

$p<0.05$; Figure 2.4). Across all years among the SWI sites, depth was significant, where the 0-10 cm had the highest M3K pools (95.2-168.2 kg K ha^{-1}) compared to the deeper soil depths (10-20 and 20-30 cm; 54.4-129.8 kg K ha^{-1} and 50.6-114.9 kg K ha^{-1}; *$p<0.005$* in all cases). In 2021, distance from the ditch was significant only at Farm 3, where high M3K pools were found in the section furthest from the ditch.

Contrastingly, Mehlich-3 extractable Na pools increased by >100% in the section closest and furthest away from the ditch (Figure 2.4) at all saltwater-intruded sites from year 1 to year 4. Distance from the ditch was significant at all SWI sites in 2021, where the section closest to the ditch had the largest M3Na pools in the soil (436.0-999.7 kg Na ha^{-1} at Farm 1, 401.9-1264.3 kg Na ha^{-1} at Farm 3, and 701.9-1235.3 kg Na ha^{-1} at Farm 6; *$p<0.05$* in all cases). Treatment was only significant at Farm 6 where saltmarsh hay and weeds had similar M3Na pools in the soil below (714.4-1112.2 kg Na ha^{-1} and 649.1-1235.3 kg Na ha^{-1}, respectively) compared to switchgrass (588.5-955.5 kg Na ha^{-1}; *$p<0.05$*; Figure 2.4). Across all years, depth was significant at all SWI sites where the deepest depth (20-30 cm) had high M3Na pools in 2018, and the shallow depth (0-10 cm) had the largest M3Na pools in 2021 (*$p<0.05$* in all cases). Finally, the saltwater-intruded sites had significantly higher Na pools in the soil than the no-salt control (646 kg Na ha^{-1} and 47.7 kg Na ha^{-1}, respectively; *$p<0.001$*).


A
Farm 1
B
Farm 3
C
Farm 6
Treatment effect p = n.s.
Section effect p < 0.005
0-5 m a
15-20 m b
Treatment effect p = n.s.
Section effect p < 0.05
0-5 m b
15-20 m a
Treatment effect p = n.s.
Section effect p = n.s.
Mehlich-3 extractable Ca+2 (kg ha-1)
Section
0-5 m
15-20 m
baseline
saltmarsh hay
switchgrass
weeds
Treatment
D
Farm 1
E
Farm 3
F
Farm 6
Treatment effect p = n.s.
Section effect p = n.s.
Treatment effect p = n.s.
Section effect p < 0.005
0-5 m b
15-20 m a
Treatment effect p = n.s.
Section effect p = n.s.
Mehlich-3 extractable K+1 (kg ha-1)
Treatment

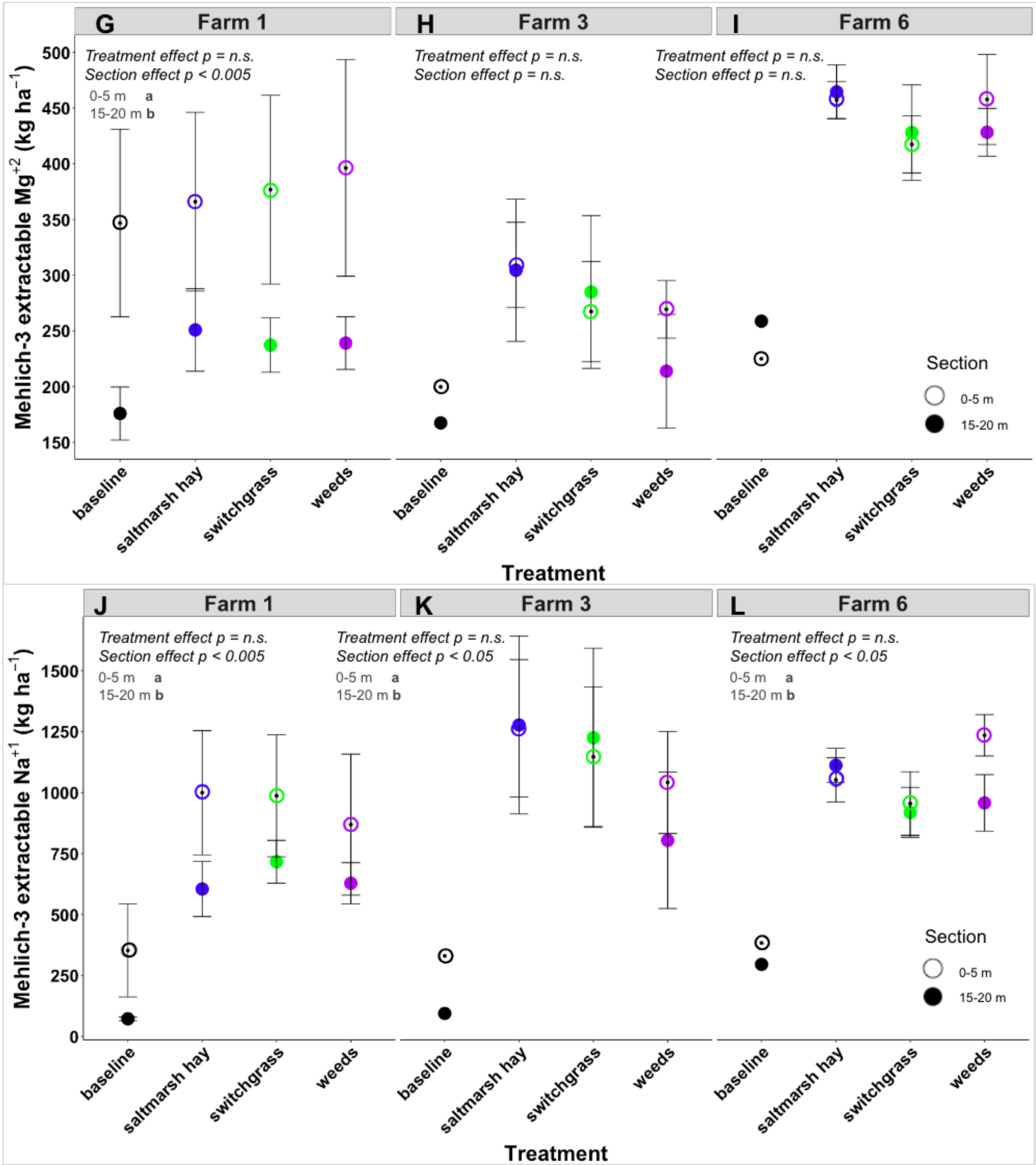


Figure 2. 4. Soil extractable cation pools in topsoil (0-10 cm) at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland in 2018 (baseline) and 2021. The x-axis is treatment and the y-axis is Mehlich-3 extractable Ca (A-C); K (D-F); Mg (G-I); and Na (J-L) pools in kilograms per hectare. Treatments are represented by colors, where the saltmarsh hay treatment is blue, the switchgrass treatment is green, and the weeds treatment is purple and section is represented by open or closed circles, where open circles are 0-5 m away from the ditch and closed circles are 15-20 m away from the ditch.

Topsoil electrical conductivity

In 2018 (baseline), topsoil (0-10 cm) electrical conductivity (EC) was low at all saltwater-intruded sites, where distance from the ditch was significant (0-5 m from ditch > 15-20 m from ditch; $p<0.01$; Figure 2.5). After three years, EC increased under all plant treatments, but there was no significant difference between them. At Farm 3, baseline soil EC farthest from the ditch was closer to the 2021 soil EC under all treatments. At Farm 6, in the soil under the weeds, there was a significant difference between the distance closest to and farthest from the ditch, where section closer to the ditch had higher EC than section further from the ditch ($p<0.01$; Figure 2.5). Soil EC was higher at all SWI sites compared to the no-salt control where EC in 2018 was approximately 0.076 dS m^{-1} and increased to 0.094 dS m^{-1} in 2021.

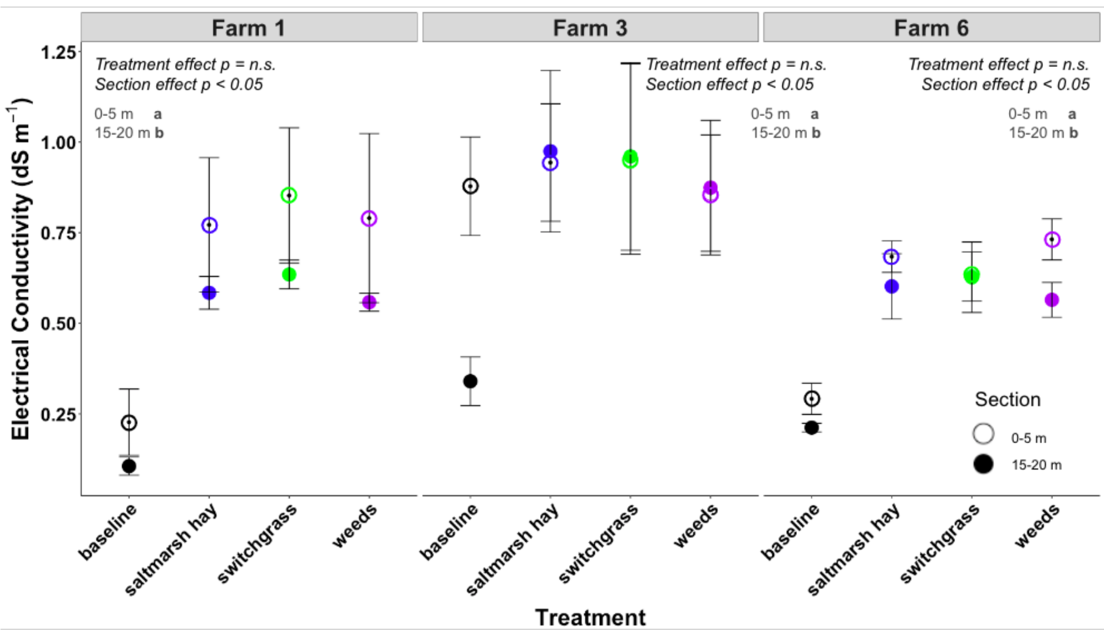


Figure 2. 5. Soil EC in topsoil (0-10 cm) at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland in 2018 (baseline) and 2021. The x-axis is treatment and the y-axis is electrical conductivity in decisiemens per meter. Treatments are represented by colors, where the saltmarsh hay treatment is blue, the switchgrass treatment is green, and the weeds treatment is purple and section is represented by open or closed circles, where open circles are 0-5 m away from the ditch and closed circles are 15-20 m away from the ditch.

Plant leaf tissue cation concentrations

Across all saltwater-intruded sites across all years, there was no clear relationship between plant leaf tissue cation (Ca^{2+}, K^{+}, Mg^{2+}, and Na^{+}) concentrations and distance from the salinity source. At the no-salt control site, we only planted switchgrass, which showed [Ca^{2+}], [K^{+}], and [Mg^{2+}] that were similar to the saltwater-intruded sites (2.1-4.0 g Ca kg^{-1}; 2.1-2.8 g Mg kg^{-1}; 10.3-11.5 g K kg^{-1}) in both 2019 and 2021. Across all study years and all sites, plant leaf tissue Na concentrations ([Na^{+}]) were highest in saltmarsh hay (2.1-3.8 g Na kg^{-1}) compared to weeds (0.2-2.0 g Na kg^{-1}) and switchgrass (0.07-0.3 g Na kg^{-1}; $p<0.001$).

In 2019 at Farm 1, saltmarsh hay had the highest [Ca^{2+}] (2.5 g Ca kg^{-1}; $p<0.01$; Figure 2.6). However, two years later, Ca concentrations in leaf tissue increased in the switchgrass and weed treatments, such that there were no longer differences among the treatments. At Farm 3 and Farm 6, switchgrass had the highest concentrations (2.3-2.6 g Ca kg^{-1}), saltmarsh hay had intermediate concentrations (2.15-2.18 g Ca kg^{-1}), and weeds had the lowest concentrations (2.0-2.1 g Ca kg^{-1}). In 2021, switchgrass tissues had the highest [Ca^{2+}] (4.2-5.0 g Ca kg^{-1}; $p<0.01$; Figure 2.6) in Farms 3 and 6.

We found similar patterns among all saltwater-intruded sites across all study years where the highest concentrations of Mg were always found in the weed tissue (2.1-2.9 g Mg kg^{-1}) and the lowest [Mg^{2+}] were found in saltmarsh hay tissue (1.2-1.7 g Mg kg^{-1}) with intermediate levels in switchgrass tissues (1.7-3.1 g Mg kg^{-1}; $p<0.01$ in all cases; Figure 2.6).

We found similar patterns among all saltwater-intruded sites across all study years where the highest concentrations of K were always found in the weed tissue

(10.0-21.2 g K kg^{-1}) and the lowest [K^+] were found in saltmarsh hay tissue (5.0-9.0 g K kg^{-1}) with intermediate levels in switchgrass tissues (8.1-12.0 g K kg^{-1}; *p<0.01* in all cases; Figure 2.6).

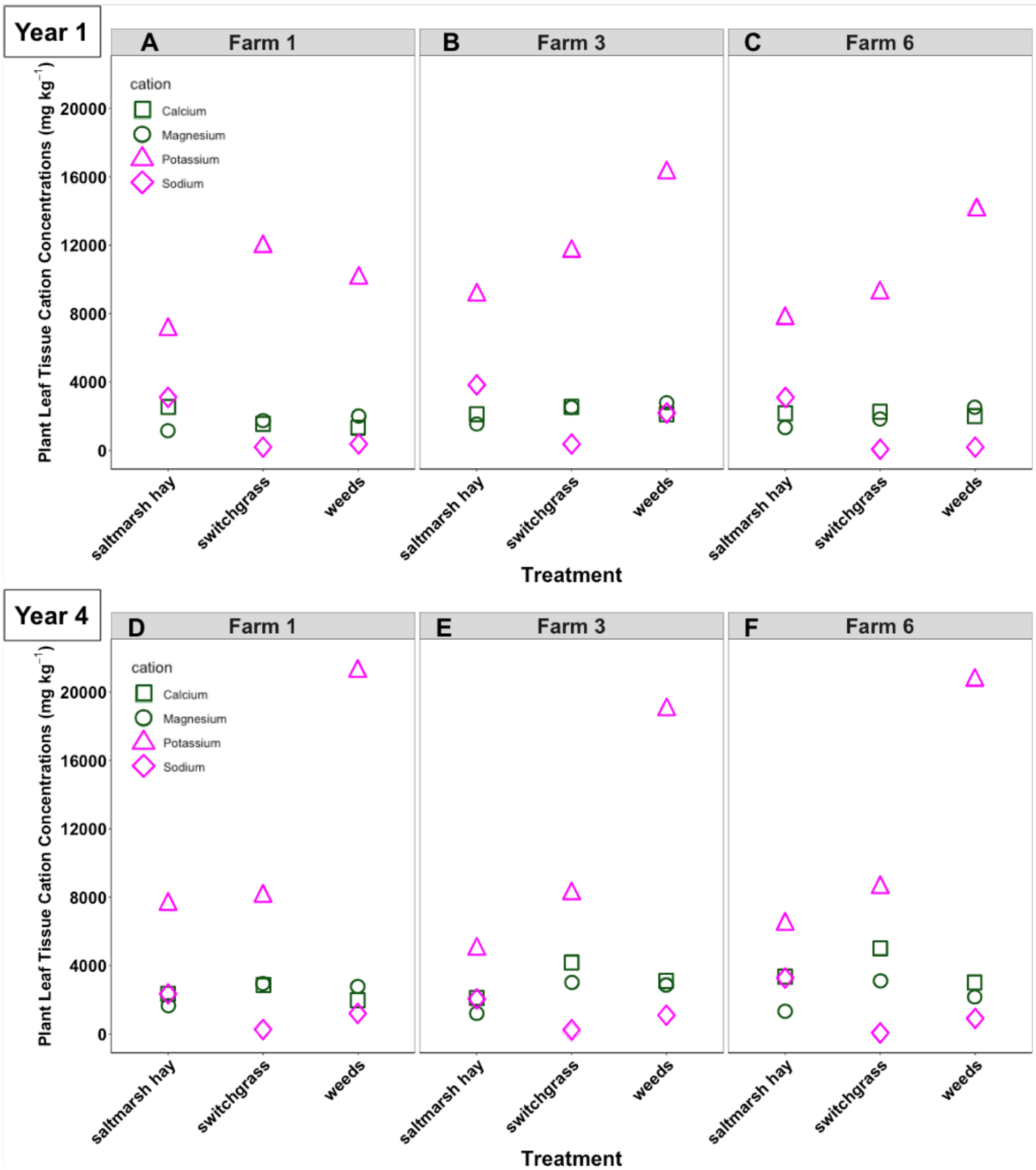


Figure 2. 6. Plant leaf tissue cation concentrations along the salinity gradient at three saltwater-intruded farms on the Lower Eastern Shore of Maryland in year 1 (A-C) and year 4 (D-F). The x-axis is treatment at each of the three farm sites, and the y-axis is cation concentrations in plant leaf tissue in milligrams per kilogram. Alkaline earth metals (Mg [circles] and Ca [squares]) are represented by the color green and alkali metals (Na [diamonds] and K [triangles]) are represented by the color pink.

Plant biomass cation pools

Across all saltwater-intruded sites across all years, there was no clear relationship between plant biomass cation (Ca, K, Mg, and Na) pools and distance from the salinity source. At the no-salt control site, we only planted switchgrass, which showed Ca, K, and Mg pools that were similar to the saltwater-intruded sites in both 2019 and 2021. Across all study years and all sites, plant biomass Na pools were highest in saltmarsh hay (1.3-26.8 kg Na ha^{-1}) compared to weeds (0.09-3.6 kg Na ha^{-1}) and switchgrass pools (0.009-2.5 kg Na ha^{-1}; $p<0.001$; Figure 2.7).

In 2019 at Farm 1, weeds had the highest Ca, K, and Mg pools (1.6-2.5 kg Ca ha^{-1}; 10.3-19.7 kg K ha^{-1}; 2.1-4.7 kg Mg ha^{-1}; $p<0.01$; Figure 2.7). However, two years later, Ca, K, and Mg in plant biomass increased in the switchgrass, so the pools were highest (8.3-34.2 kg Ca ha^{-1}; 27.9-98.2 kg K ha^{-1}; 9.2-36.4 kg Mg ha^{-1}), weeds had the lowest pools (1.3-5.0 kg Ca ha^{-1}; 16.6-64.6 kg K ha^{-1}; 1.9-7.3 kg Mg ha^{-1}), and saltmarsh hay had intermediate pools (5.8-8.7 kg Ca ha^{-1}; 18.4-26.1 kg K ha^{-1}; 3.8-4.2 kg Mg ha^{-1}; $p<0.001$; Figure 2.7).

Across all years at Farm 3 and Farm 6, switchgrass had the highest Ca, K, and Mg pools (14.2-64.5 kg Ca ha^{-1}; 53.9-165.9 kg K ha^{-1}; 13.8-49.8 kg Mg ha^{-1}), saltmarsh hay had intermediate pools (6.4-22.1 kg Ca ha^{-1}; 25.2-57.9 kg K ha^{-1}; 4.2-14.2 kg Mg ha^{-1}), and weeds had the lowest pools (0.6-7.6 kg Ca ha^{-1}; 14.0-48.4 kg K ha^{-1}; 0.4-9.9 kg Mg ha^{-1}; $p<0.001$; Figure 2.7).

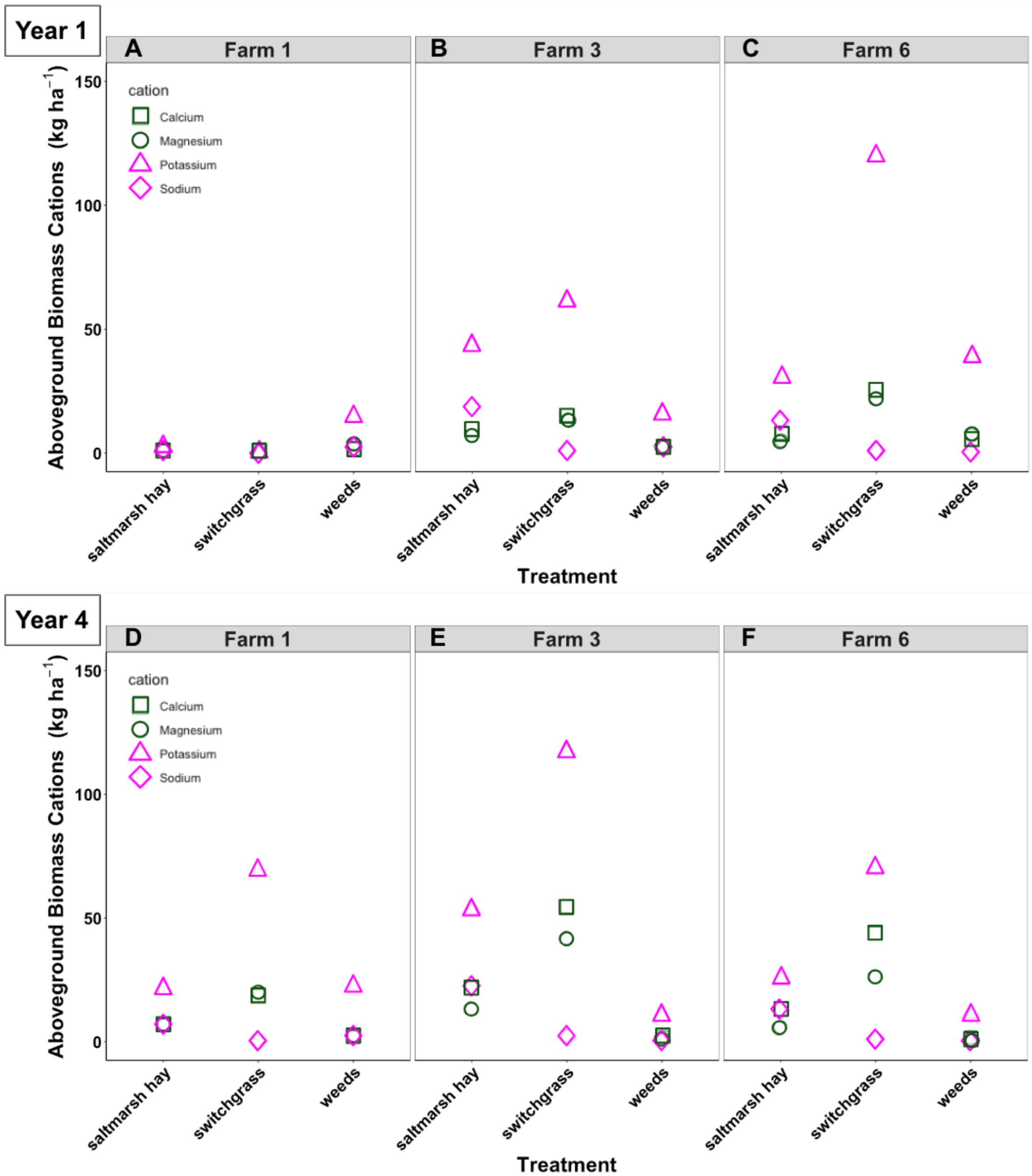


Figure 2. 7. Plant aboveground biomass cation pools at three saltwater-intruded field sites on the Lower Eastern Shore of Maryland in year 1 (A-C) and year 4 (D-F). The x-axis is treatment at each of the three farm sites, and the y-axis is cation pools in aboveground biomass in kilograms per hectare. Alkaline earth metals (Mg [circles] and Ca [squares]) are represented by the color green and alkali metals (Na [diamonds] and K [triangles]) are represented by the color pink.

Correlations among cations in soils

In 2018, across all sites, Ca and K were highest in the topsoil (0-20 cm; r^2 = -0.41 and r^2 = -0.56; $p<0.05$ and $p<0.01$, respectively; Table 2.3) where Mg and Na concentrations were highest in the deepest depth of the soil (20-30 cm; $r^2 = 0.2$; *p=0.1* in both cases). Ca and Mg and Ca and K concentrations in the soil were positively correlated ($r^2 = 0.4$; *p<0.01* in both cases). Mg and Na were also positively correlated ($r^2 = 0.5$; *p<0.001*). In 2021, across all sites, K, Mg, and Na were highest in the topsoil (0-20 cm; $r^2 < -0.2$; *p<0.05* in all cases; Table 2.3) and Mg and Na were highest closer to the ditch ($r^2 = -0.3$; *p<0.001* in both cases). Ca and Mg and Ca and Na concentrations in the soil were positively correlated ($r^2 = 0.6$ and $r^2 = 0.3$, respectively; *p<0.001* in both cases). Mg and Na ($r^2 = 0.7$; *p<0.001*) and Mg and K ($r^2 = 0.5$; *p<0.001*) were also positively correlated.

Table 2. 3. Soil cation concentration correlations in year 1 (baseline) and saltmarsh hay, switchgrass, and weeds in year 4 in both the section closest to the ditch (0-5 m) and the section furthest from the ditch (15-20 m) on three-saltwater-intruded fields on the Lower Eastern Shore of Maryland. Values are r^2 and symbols are p-values. Colors represent the strength of the correlation (-0.00*** = strong negative; -0.00** = moderate negative; -0.00* = negative; -0.00^ = weak negative; 0.00^ = weak positive; 0.00* = positive; 0.00** = moderate positive; 0.00*** = strong positive). *^ = p<0.05, * = p<0.01, ** = p<0.005, *** = p<0.001*

	Baseline		Saltmarsh hay		Switchgrass		Weeds	
	0-5 m	15-20 m	0-5 m	15-20 m	0-5 m	15-20 m	0-5 m	15-20 m
[M3Ca]~[M3K]	0.43^	0.21	0.32^	-0.22	0.31^	-0.15	0.36*	-0.22
[M3Ca]~[M3Mg]	0.63**	0.35	0.66***	0.59***	0.52**	0.65***	0.62***	0.62***
[M3Ca]~[M3Na]	0.0041	-0.30	0.05	0.40*	-0.018	0.29^	0.32^	0.30^
[M3K]~[M3Mg]	0.54*	-0.28	0.69***	0.30^	0.75***	0.29^	0.74***	0.26
[M3K]~[M3Na]	0.20	-0.57*	0.40*	0.27	0.37*	0.31^	0.37*	0.28^
[M3Mg]~[M3Na]	0.48*	0.46^	0.52**	0.84***	0.66***	0.74***	0.63***	0.74***

Topsoil EC vs. soil cation concentrations

In 2018, EC was negatively correlated with [M3Ca] and [M3K] (r^2 = -0.27, *p<0.05*; r^2 = -0.32, *p<0.01*; Table 2.4). Contrastingly, EC was strongly positively correlated with [M3Mg] and [M3Na] (r^2 = 0.54, *p<0.001*; r^2 = 0.75, *p<0.001*; Table 2.4). In 2021, EC was positively correlated with [M3K], [M3Mg], and [M3Na] (r^2 = 0.30, *p<0.05*; r^2 = 0.58, *p<0.001*; r^2 = 0.94, *p<0.001*; Table 2.4).

Table 2. 4. Soil EC and soil cation concentration correlations in year 1 (baseline) and year 4 in both the section closest to the ditch (0-5 m) and the section furthest from the ditch (15-20 m) on three-saltwater-intruded fields on the Lower Eastern Shore of Maryland. Values are r^2 and symbols are p-values. Colors represent the strength of the correlation (-0.00*** = strong negative; -0.00** = moderate negative; -0.00* = negative; -0.00^ = weak negative; 0.00^ = weak positive; 0.00* = positive; 0.00** = moderate positive; 0.00*** = strong positive). *^ = p<0.05, * = p<0.01, ** = p<0.005, *** = p<0.001*

	2018		2021	
	0-5 m	15-20 m	0-5 m	15-20 m
EC~[M3Ca]	-0.27^	-0.20	0.016	-0.0058
EC~[M3K]	-0.32*	-0.27^	0.30^	0.14
EC~[M3Mg]	-0.023	0.52***	0.58***	0.52**
EC~[M3Na]	0.54***	0.75***	0.94***	0.94***

Correlations among cations in plant leaf tissue

In 2019, across all saltwater-intruded sites, K and Mg concentrations in plant leaf tissue were the highest in weeds, Na was the highest in saltmarsh hay, and Ca was higher in the saltmarsh hay and switchgrass than weeds. In saltmarsh hay tissue, all cations were positively correlated, where K, Ca, Mg, and Na were strongly positively correlated with each other (r^2 > 0.5; *p<0.05* in all cases). In switchgrass plant leaf tissue, all cations were positively correlated (r^2 = 0.68; *p=0.001*, r^2 = 0.65;

p=0.002), except K~Ca which was weakly negatively correlated (r^2 = -0.1; *p=n.s.*). In the weed tissue, all cations were positively correlated with each other (r^2 > 0.4; *p<0.05* in all cases; Table 2.5).

In 2021, salt concentrations in plant tissues largely followed the same patterns within species: across all saltwater-intruded sites, again K and Mg concentrations in leaf tissue were highest in the weeds, again Na was highest in the saltmarsh hay, but in 2021, Ca was highest in the switchgrass rather than switchgrass and saltmarsh hay. In saltmarsh hay tissue, all cations were positively correlated with each other (r^2 > 0.5; *p<0.05* in all cases). In switchgrass plant leaf tissue, K was negatively correlated with Ca and Mg, and Na was negatively correlated with Ca and Mg. In the weeds, all cations were positively correlated with each other (r^2 = 0.75; *p<0.001*; Table 2.5).

Table 2. 5. Plant leaf tissue cation concentration correlations for saltmarsh hay, switchgrass, and weeds from three-saltwater-intruded fields on the Lower Eastern Shore of Maryland in year 2 and year 4. Values are r^2 and symbols are p-values. Colors represent the strength of the correlation (-0.00*** = strong negative; -0.00** = moderate negative; -0.00* = negative; -0.00^ = weak negative; 0.00^ = weak positive; 0.00* = positive; 0.00** = moderate positive; 0.00*** = strong positive). *^ = p<0.05, * = p<0.01, ** = p<0.005, *** = p<0.001*

	Saltmarsh hay		Switchgrass		Weeds	
	Year 2	Year 4	Year 2	Year 4	Year 2	Year 4
[Ca]~[K]	0.12	0.60**	-0.10	-0.29	0.56**	-0.019
[Ca]~[Mg]	0.59**	0.62**	0.65**	0.29	0.27	0.082
[Ca]~[Na]	0.54*	0.58**	0.046	-0.26	0.45*	0.28
[K]~[Mg]	0.56*	0.50*	0.17	-0.42^	0.34^	0.75***
[K]~[Na]	0.55*	0.35	0.68**	0.32	0.29	0.028
[Mg]~[Na]	0.67**	0.063	0.16	-0.40^	0.54**	0.10

Topsoil EC vs. plant cation concentrations

Correlations between soil EC and plant leaf tissue cation concentrations were mostly insignificant except for in the section closest to the ditch in switchgrass and weeds where EC was positively correlated with [Na] ($r^2 = 0.60$, $p<0.005$; $r^2 = 0.89$, $p<0.005$; Table 2.6). There was also a weak positive correlation between soil EC and [Mg] ($r^2 = 0.43$, $p<0.05$).

Table 2. 6. Soil EC and plant leaf tissue cation concentration correlations in year 4 for saltmarsh hay, switchgrass, and weeds in both the section closest to the ditch (0-5 m) and the section furthest from the ditch (15-20 m) from three-saltwater-intruded fields on the Lower Eastern Shore of Maryland. Values are r^2 and symbols are p-values. Colors represent the strength of the correlation (-0.00*** = strong negative; -0.00** = moderate negative; -0.00* = negative; -0.00^ = weak negative; 0.00^ = weak positive; 0.00* = positive; 0.00** = moderate positive; 0.00*** = strong positive). ^ = $p<0.05$, * = $p<0.01$, ** = $p<0.005$, *** = $p<0.001$

	Saltmarsh hay		Switchgrass		Weeds	
	0-5 m	15-20 m	0-5 m	15-20 m	0-5 m	15-20 m
EC~[Ca]	-0.16	-0.30	-0.059	0.0025	0.44	0.27
EC~[K]	-0.058	-0.095	0.33	0.32	-0.062	-0.055
EC~[Mg]	-0.07	-0.077	0.27	0.43^	0.28	-0.0066
EC~[Na]	0.16	-0.062	0.60**	0.16	0.89**	0.67

Soil and plant cation concentration relationships (topsoil 0-10 in 2021)

In 2021, topsoil [M3Na] and plant leaf tissue [Na] were strongly positively correlated among switchgrass and weeds in both the area closest to the ditch and furthest from the ditch ($r^2 > 0.5$, $p<0.05$ in all cases; Table 2.7). There was no correlation between any of the cations in the topsoil and saltmarsh hay plant leaf tissue in either section of the field. Switchgrass [Ca] in tissue closest to the ditch was negatively correlated with [M3K] in topsoils, while [Na] in tissues were positively correlated with [M3Mg] in topsoils (r^2 = -0.44, $p<0.01$; r^2 = 0.37, $p<0.05$). Switchgrass [K] and [Mg] in tissue furthest from the ditch was positively correlated with [M3Na] in topsoils (r^2 =0.46, $p<0.01$; r^2 = 0.39, $p<0.05$). The concentrations of Na in weed tissues in the 0-5 m section was strongly positively correlated with [M3K], [M3Mg], [M3Na] in soils (r^2 = 0.75, $p<0.005$; r^2 = 0.77, $p<0.005$). Weed tissue [K], [Mg], and [Na] concentrations furthest from the ditch was positively correlated with [M3Ca] (r^2 = 0.84, $p<0.005$; r^2 = 0.80, $p<0.01$; r^2 = 0.64, $p<0.05$).

Table 2. 7. Plant leaf tissue and topsoil cation concentration correlations in year 4 for saltmarsh hay, switchgrass, and weeds in both the section closest to the ditch (0-5 m) and the section furthest from the ditch (15-20 m) from three-saltwater-intruded fields on the Lower Eastern Shore of Maryland. Values are r^2 and symbols are p-values. Colors represent the strength of the correlation (-0.00*** = strong negative; -0.00** = moderate negative; -0.00* = negative; -0.00^ = weak negative; 0.00^ = weak positive; 0.00* = positive; 0.00** = moderate positive; 0.00*** = strong positive). ^ = *p<0.05*, * = *p<0.01*, ** = *p<0.005*, *** = *p<0.001*

	Saltmarsh hay		Switchgrass		Weeds	
	0-5 m	15-20 m	0-5 m	15-20 m	0-5 m	15-20 m
[M3Ca]~[Ca]	0.14	0.077	0.055	0.22	0.19	0.58
[M3Ca]~[K]	0.0072	-0.064	-0.33	0.068	0.41	0.84**
[M3Ca]~[Mg]	-0.01	0.014	-0.037	-0.10	-0.17	0.80*
[M3Ca]~[Na]	-0.0025	0.065	-0.18	0.24	0.25	0.64^
[M3K]~[Ca]	-0.27	-0.001	-0.44*	-0.14	-0.33	-0.39
[M3K]~[K]	-0.23	0.19	0.11	0.13	-0.012	0.43
[M3K]~[Mg]	-0.14	0.067	-0.34	0.35	-0.11	0.22
[M3K]~[Na]	-0.19	-0.007	0.16	0.0011	0.75*	0.17
[M3Mg]~[Ca]	-0.034	-0.078	-0.16	0.21	0.0034	-0.17
[M3Mg]~[K]	-0.053	-0.15	0.053	0.20	0.23	0.068

[M3Mg]~[Mg]	-0.21	-0.024	-0.031	0.22	0.061	-0.12
[M3Mg]~[Na]	0.0033	0.066	0.37^	0.23	0.77*	0.40
[M3Na]~[Ca]	-0.35	-0.30	-0.13	-0.0024	0.49	-0.0011
[M3Na]~[K]	-0.0036	-0.13	0.34	0.46*	0.026	-0.0014
[M3Na]~[Mg]	-0.24	-0.067	0.23	0.39^	0.14	-0.062
[M3Na]~[Na]	0.14	-0.016	0.61**	0.25	0.82**	0.52

Discussion

Our results from this study have implications for improved understanding of cation dynamics in plant leaf tissues and soils in saltwater intruded fields. After analyzing the aboveground biomass and soil under four management practices (adapt, remediate, restore, abandon) on three farms experiencing SWI, we determined that saltmarsh hay is efficient at taking up soil Na. We showed the relationships between plant leaf tissue and soil available cations are complex and differ by our variables (e.g., distance from the ditch, year) under saltwater-intruded conditions. Not surprisingly, concentrations of available K, Mg and Na in soils were higher close to the ditch than near the center of the agricultural field as the result of SWI and SLR (Arslan & Demir 2013; Figure 2.6). After three years of planting, available Ca and K decreased in topsoils and available Mg and Na increased in topsoils at all saltwater intruded sites (Figure 2.6). The significant decrease in soil available Ca in the topsoil (0-10 cm) over 2 years suggests that Ca was leached below 10 cm or that it was assimilated into plant tissues. While Ca is normally bound to soil colloids, with an excess of Na from saltwater, there is more opportunity for Na to exchange with Ca, causing Ca to release (Chidambaram et al. 2015). Soil available K decreased in the topsoil after 2 years likely due to plant uptake as K does not readily leach from soils (Mayland & Wilkinson 1989; Lehmann & Schroth 2002). On the other hand, soil available Mg and Na increased over 2 years which was expected as these cations are the main constituents that make up seawater and can be increased by SWI (Kester et al. 1967; Figure 2.6; Table 2.4). Overall, restoration (e.g. saltmarsh hay) management practices showed no increase in aboveground biomass Na concentrations as soil available Na accumulated over time, and this was indicative of the species' ability to co-regulate cations in their leaf tissue.

Plant tissue cation concentrations varied among the different species

We expected that as Na concentrations in the soil increased, it would prevent the uptake of Ca into plant tissues (Lazoff & Bernstein 1999). While there were no strong correlations between soil Na and leaf tissue Ca in our data (Table 2.7), we observed an increase in Ca concentrations in switchgrass and weeds after two years ($p<0.01$; Figure 2.6A-C). Increased soil Na has also been known to prevent the uptake of K into plant tissues (Munns & Tester 2008), which could have caused leaching from surface flooding, but this would cause an increase or no change in soil available K concentrations, which is not what we saw (observed decrease in soil available K; Figure 2.6D-F). However, we found similar patterns among all saltwater-intruded sites across all study years where the highest concentrations of K were always found in the weed tissue and intermediate levels in switchgrass tissues, and there was an increase in K concentrations for both species in 2021 (Figure 2.6D-F). Similarly, we found the highest concentrations of Mg were in the weed tissue, the lowest [Mg^{2+}] were found in saltmarsh hay tissue, and intermediate levels were found in switchgrass (Figure 2.6G-I). Previous literature has shown excess uptake of K in grasses lowers Mg content (Robinson et al. 1989). The differing results from our expectations could be due to how different species are able to regulate salt uptake and tolerate different levels of salinity (Grieve et al. 2012; Table 2.1). Switchgrass is known to have a higher tolerance to limiting water and nutrient conditions and may be able to uptake more soil nutrients due to their deep root system and relationship with mycorrhizal fungi (Di Virgilio et al. 2007; Xu et al. 2017). The weed species in 2019 was a native switchgrass species (*Panicum dichotomiflorum*) with extensive roots similar to our planted switchgrass. In 2021, the dominant weed was a marsh species, crabgrass (*Digitaria sanguinalis*), that is similar to saltmarsh hay that is able to regulate nutrient uptake under saline conditions and maintain biomass regardless of Ca and Mg additions (Grieve et al. 2012; Pierce et al. 2017).

Saltmarsh hay is able to regulate uptake of Na

We predicted that plant biomass would be suppressed by excess Na from SWI due to the osmotic and toxic effects on plant growth and cell function (Flowers et al. 1991). Topsoil [M3Na] and plant leaf tissue [Na] were strongly positively correlated in the switchgrass and weed treatments in both the area closest to the ditch and furthest from the ditch (Table 2.7) meaning both took up more Na when there was more Na in the soil (Kim et al. 2012; Rashidi et al. 2020). Because switchgrass and weeds were taking up more Na in their leaf tissue, this may have caused the inhibition of uptake of important nutrients that contribute to their biomass (Greenway & Munns 1980; Hasegawa et al. 2000; Netondo et al. 2004). There were no relationships between saltmarsh hay tissue and soil cations due to the salt-tolerant species' ability to regulate salt intake (Grieve et al. 2012), specifically taking in high concentrations of Na. A change in soil cation concentrations did not affect leaf tissue cation concentrations (Table 2.7), but interestingly, saltmarsh hay plant leaf tissue cation concentrations were very strongly positively correlated with each other (Table 2.5).

Across all study years and all sites, plant leaf tissue Na concentrations [Na^+] and biomass Na pools were highest in saltmarsh hay ($p<0.001$). Saltmarsh hay is efficient at taking up soil Na and, if planted along the saltier edges of the field, could be a mitigation strategy to decrease soil Na that is toxic to most plants in the area. Planting saltmarsh hay could facilitate the transition to native marsh which might be a better option if the field experiences more flooding (Gedan et al. 2011; Hu et al. 2015; Leonardi et al. 2018). Previous research indicated that fields experiencing SWI will naturally transition to host more native marsh species such as saltmarsh hay (Gedan et al. 2019). Federal reserve programs could provide landowners with financial assistance, or there are opportunities to use the land for other activities such as waterfowl hunting that could be an alternative source of revenue (e.g., Conservation Reserve Enhancement Program; CREP 2022, Environmental Quality Incentives Program; EQIP 2022, Wetland Reserve Easements; WRE 2022). Overall, restoring farm fields has benefits for the soil by taking up Na, landowners by maintaining revenue, and communities by protecting coastlines.

Conclusion

Rising seas and elevated soil salinity levels have forced farmers to reconsider their management practices, and these decisions will impact downstream ecosystems. Different frameworks are used to categorize human responses to climate change and natural hazards such as floods (Canyon et al. 2015; van Wesenbeeck et al. 2017; Doberstein et al. 2019). For example, retreat approaches are those where landowners repurpose their fields by enrolling them in easement programs (e.g. Conservation Reserve Program; CRP 2022) (van Wesenbeeck et al. 2017). Similar to these other frameworks, management practices for the specific region also attempt to accommodate SLR (and SWI) by planting native marsh species or retreating and allowing the transition of fields into marshland (Tully et al. 2019). We observed patterns of regulation in nutrient uptake in all species (saltmarsh hay, switchgrass, and weeds) despite the imbalance of normal nutrient uptake due to saltwater Mg. In sum, results from this study show restoration management practices are efficient at taking up soil Na, and may keep up with high Na concentrations and pools in the soil over time, which may benefit both soil and plant health on saltwater-intruded fields. Finally, this work will help inform state-level coastal management policies and determine optimal strategies for climate resilience.

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