RESTORATION ECOLOGY
The Journal of the Society for Ecological Restoration

SER SOCIETY FOR ECOLOGICAL RESTORATION

RESEARCH ARTICLE

Differential growth dynamics of common salt marsh species *Spartina alterniflora* and *Juncus roemerianus* under varied sediment amendments

Caitlin Hemphill[1,2], Richard P. Hale[3], Erik S. Yando[1,2]

Abstract

Introduction: Many coastal wetlands are at high risk of degradation or loss due to sea-level rise. Restoration techniques for maintaining coastal marshes are paramount, with thin-layer placement (TLP) emerging as one feasible solution. Despite TLP's utilization, additional research is needed on species- and sediment-specific responses.

Objectives: Our research combines greenhouse and field experimentation to test species and sediment composition responses aimed at maximizing vegetation while maintaining increased elevation.

Methods: The greenhouse experiment used two common marsh plants, *Spartina alterniflora* and *Juncus roemarianus*. Five centimeters of sediment (mud, sand, and sand/mud mix) was added, and subsequent sediment and vegetation metrics were monitored. The field experiment focused on *S. alterniflora* and examined multiple sediments (mud and sand/mud mix) in coastal Virginia, United States.

Results: In the greenhouse, *Spartina* and *Juncus* had similar responses to sediment types with no differences to the control, but sand producing greater aboveground biomass compared to mud. *Spartina* outperformed *Juncus* overall and *Juncus* root biomass was negatively impacted by sand compared to the control. Compaction occurred across all sediments, with greater loss in the sand compared to the mud and control. In the field, mud and mix treatments had lower aboveground metrics compared to the control groups and saw no differences in elevation even after amendments.

Conclusions: These results showcase the importance of initial marsh plant response in a species- and sediment-specific manner. Further, they highlight differences between research settings, sediment additions, and metric specific outcomes.

Implications for Practice: Controlled greenhouse experiments paired with field experiments provide a valuable window into species-specific and short-term response of coastal marsh response to thin-layer placement practice. By examining multiple species, a more comprehensive view of responses is possible for targeted approaches and benchmarks. While long-term success is an ultimate goal, short-term impacts can have immediate and lasting implications on multiple ecosystem properties and processes. Understanding these short-term fine-scale responses can resolve recovery trajectories early on and provide critical information on implementation, which may allow enough time and planning to provide mitigation in subsequent seasons. Ideally, identifying potentially negative impacts of restoration activities in the short term to improve long-term outcomes.

Key words: beneficial use, marsh resilience, sea-level rise, thin-layer placement, tidal saline marsh

Introduction

Efforts to enhance ecosystem resilience are paramount to promote ecosystem health and functions under degradation. Tidal salt marshes are commonly the focus of such efforts due to their position at the land-sea interface, historic degradation/loss, and the critical ecosystem functions and services they provide (Barbier et al. 2011; Enwright et al. 2016; Friess et al. 2020). Despite their importance, salt marshes are at risk of drowning due to changes in inundation with ongoing and future sea-level rise (Saintilan et al. 2022). As global temperatures increase, melting land ice and thermal expansion are causing eustatic sea-level rise (Nicholls & Cazenave 2010). Additionally, some coastal areas are experiencing land subsidence because of both groundwater withdrawal (Sherpa et al. 2023) and post-glacial isostatic adjustments (Love et al. 2016), such as the mid-Atlantic and portions of the southeastern United States. Higher relative sea levels can impact salt marshes through prolonged inundation if accretion rates do not keep pace, leading to decreased productivity and marsh degradation (Osland et al. 2024). Ultimately, this reduces salt marsh

Author contributions: CH, RPH, ESY, conceptualized and designed the study; CH led the investigation, data curation, and validation; CH, RPH, ESY conducted the formal analysis and visualization, CH wrote the original draft of the paper; CH, RPH, ESY edited the manuscript; CH, RPH, ESY provided funding acquisition, resources, and project administration; ESY, RPH provided supervision.

[1]Department of Biological Sciences, Old Dominion University, Norfolk, VA, U.S.A.
[2]Address correspondence to E. S. Yando, email eyando@odu.edu and C. Hemphill, email cahemphill19@gmail.com
[3]Department of Ocean and Earth Sciences, Old Dominion University, Norfolk, VA, U.S.A.

© 2026 The Author(s). Restoration Ecology *published by Wiley Periodicals LLC on behalf of Society for Ecological Restoration.*
This is an open access article under the terms of the Creative Commons Attribution License, which permits use, distribution and reproduction in any medium, provided the original work is properly cited.
doi: 10.1111/rec.70462
Supporting information at:
http://onlinelibrary.wiley.com/doi/10.1111/rec.70462/suppinfo

resilience, increases the potential for ecosystem loss, and can result in subsequent state change (Beisner et al. 2003).

Salt marshes are highly productive tidal saline wetlands found on sheltered coastlines worldwide (Mcowen et al. 2017). Their relative elevation and hypsometry are strongly influenced by the collective effect of tidal, wave, and river forcings, adjacent ecosystems, species tolerances, and local landscape evolution (reviewed by Yando, Jones, & James et al. 2023), along with the ability to maintain relative position via accretion (Friedrichs & Perry 2001). In many marshes, tides and high-water events (e.g. caused by storms, river flooding, or wind) can deposit allochthonous sediment onto the surface of the marsh and receding waters can remove or redistribute sediments (Nolte et al. 2013). Further, vegetation debris and accumulated organic matter contributes to autochthonous sediment present in the system (Mudd et al. 2009). Sediment deposition and net accretion rates depend on suitable source concentrations and characteristics, distance from the source, inundation duration, erosion dynamics, and marsh vegetation characteristics (Stumpf 1983; Möller et al. 2014; Tempest et al. 2015). These sources paired with root zone expansion can allow for vertical gains (Cahoon et al. 2021) and when relative sea-level rise rates are modest this allows salt marshes to maintain their relative vertical position in the tidal frame (Kirwan & Megonigal 2013). Despite this ability to maintain their position, exceeding the threshold of inundation (sensu Hughes et al. 2019) in salt marshes directly impacts the establishment, survival, and resilience of dominant vegetation and overall marsh health (Kirwan & Murray 2007; Wang & Temmerman 2013).

Ecosystem resilience is the ability to maintain existing functions and structures while experiencing disturbance (Gunderson et al. 2012; Lamothe et al. 2019). Marsh resilience in response to rising sea levels relies on net accretion and root zone expansion dynamics to increase elevation, or landward retreat when inundation surpasses a plant community's tolerance (Mariotti & Carr 2014). Current sea-level rise rates in many locations are outpacing the ability for marshes to keep up vertically and resulting in decreased marsh resilience with the potential for eventual conversion to an alternative stable state, such as tidal flats or open water (Moffett et al. 2015). Further, limited landward accommodation space is a common issue with human-made barriers present in many coastal reaches or unsuitable hypsometry in adjacent uplands (Enwright et al. 2016). The trajectory of this state shift can be gradual or sudden depending on the specific ecological thresholds, resilience, and resistance of the system (Beisner et al. 2003; Walker & Meyers 2004; Lamothe et al. 2019). The loss of vegetation in salt marshes due to greater inundation fuels a negative feedback loop, with less sediment accreted and retained, causing a decrease in marsh elevation and subsequent increases in inundation periods (Stagg et al. 2020). These further lower biomass generation, degrading remaining marsh and lowering the resilience and resistance of the system further, eventually leading to state change (Beisner et al. 2003). This regime shift from marsh to tidal flat results in large decreases in ecosystem extent, functions, services, and biodiversity (Scheffer et al. 2001).

Restoration techniques for maintaining coastal marsh resilience continue to be developed and refined. One such technique is thin-layer placement, hereafter TLP. In TLP, sediment from a nearby source is placed onto a salt marsh with the expectation of vegetation growing through the applied sediment. Manually increasing marsh elevation increases marsh resilience, buffers against rising sea levels, and prevents the shift to an alternative state (e.g. tidal flat) and mimics allochthonous sediment contributions. Originally developed in coastal Georgia in the late 1970s (Reimold et al. 1978), projects of different spatial scales have been implemented and studied primarily in the Gulf of Mexico and East Coast of the United States. Existing literature has been focused on soil physical and chemical properties (Koning 2004; Slocum et al. 2005; Stagg & Mendelssohn 2011), plant community response (Ford et al. 1999; Schrift et al. 2008; Raposa et al. 2022), and marsh elevation change or accretion in response to TLP (Cornu & Sadro 2002; La Peyre et al. 2009; Davis et al. 2022) in both experimental and project-based studies occurring over both single and multi-seasons. Few papers, however, look at species-specific responses (Allison 1996) or use paired greenhouse and field studies, but it is important to understand implications when considering TLP research and project success.

The aim of this research was to evaluate how the composition of sediment used in TLP affects the sustained plant growth and sediment accumulation of the marsh, and how different salt marsh species respond to this practice. We addressed the following questions: (1) Which sediment composition in TLP allows for the greatest vegetation biomass production while still maintaining an increased soil elevation? and (2) Does response differ by species? To answer our primary question regarding sediment composition, we established both a greenhouse-based mesocosm study and a field study to evaluate the effects of TLP sediment type; further, we examined species-specific responses in a greenhouse setting to control for edaphic conditions between two common marsh species, *Spartina alterniflora* and *Juncus roemerianus*. We hypothesized that a combination of sand and mud would promote the highest biomass regardless of species or setting as it will allow the marsh to maintain its elevation compared to other treatments due to the combination of the mud's nutrient load and sand's increased bulk density. Furthermore, we hypothesized that *S. alterniflora's* rapid colonization and expansion strategy would make it more likely to take advantage of additional sediments compared to *J. roemerianus*.

Methods

Greenhouse—Experimental Design

Spartina alterniflora Loisel (hereafter *Spartina*) and *Juncus roemerianus* Scheele (hereafter *Juncus*) sods were harvested from a coastal salt marsh along the Chesapeake Bay in southeastern Virginia and transported to a nearby greenhouse. Sods of approximately 12 stems were placed in 3-gal buckets (11.35-L; height = 24 cm; diam. top = 26.5 cm; diam. bottom = 23 cm) to reduce pot effects on plants (reviewed by Poorter et al. 2012). Pots were pre-filled with 6 cm of pea gravel

for drainage and supplemented with mud from a nearby tidal flat to fill any voids. Each pot contained one species and all pots were allowed to acclimate for 6 weeks prior to the start of the experiment (Fig. 1A). At the start of the experiment, units received either 5 cm of mud (mud), sand (sand), 1:1 sand/mud mixture (mix), or no additional sediment (control) (Fig. 1B) with a total of 10 replicates of each species–treatment combination and a total of 80 pots (hereafter experimental units). Five centimeters was chosen as it was the median value of thin-layer experiments in the literature as determined by a literature analysis. Granulometry and organic content measurements were conducted on all applied sediments (see Textural and Organic Matter Analysis section for methods). Sediments included sand from nearby shorelines and mud collected from the aforementioned tidal flat. The sand/mud mixture was created by adding equal parts by volume of mud and sand in a large plastic tub and homogenized by hand. Experimental units were randomly placed within five mesocosm tanks (block) using a randomized design within each mesocosm with two replicates of each type per mesocosm.

Rectangular mesocosm tanks (hereafter mesocosms) measured 1.22 m × 1.85 m × 0.95 m (internal height × length × width; 2.14 m^3 volume) and were each connected to a paired reservoir tank of the same dimensions to simulate daily semi-diurnal tidal fluctuations using paired pond pumps (Fig. 1B) and kept at 20 ppt salinity for the duration of the experiment using Instant Ocean Sea Salt to mimic local marsh conditions. Experimental units within the mesocosms were uniformly elevated on crates to allow for a greater range of water levels. Semi-diurnal 0.73 m tides were established based on the average tidal levels for southeastern Virginia in the past 5 years during the growing season (May–October) (NOAA Tidal Gauge 8638610 Sewells Point Norfolk, Virginia 2023). High tide was set to reach a height of 0.48 m above soil surface, and low tides were set to −0.25 m below the soil surface of experimental units using locally collected marsh elevations via GNSS RTK-GPS (TopCon Hiper VR). Pumps filled and emptied mesocosms over 5.5 hours and maintained water levels for an hour to mimic a natural semi-diurnal tidal cycle. Experimental units were allowed to grow for 3 months after sediment application (July 1 to September 30, 2024) to avoid plants becoming root bound (reviewed by Poorter et al. 2012) with pumps running continuously during this time.

Greenhouse—Data Collection

Initial measurements (stem count, stem height, percent cover, and soil bulk density) for each experimental unit were collected prior to sediment addition. Stem count, average stem height, stem diameter, and percent cover were also collected every other

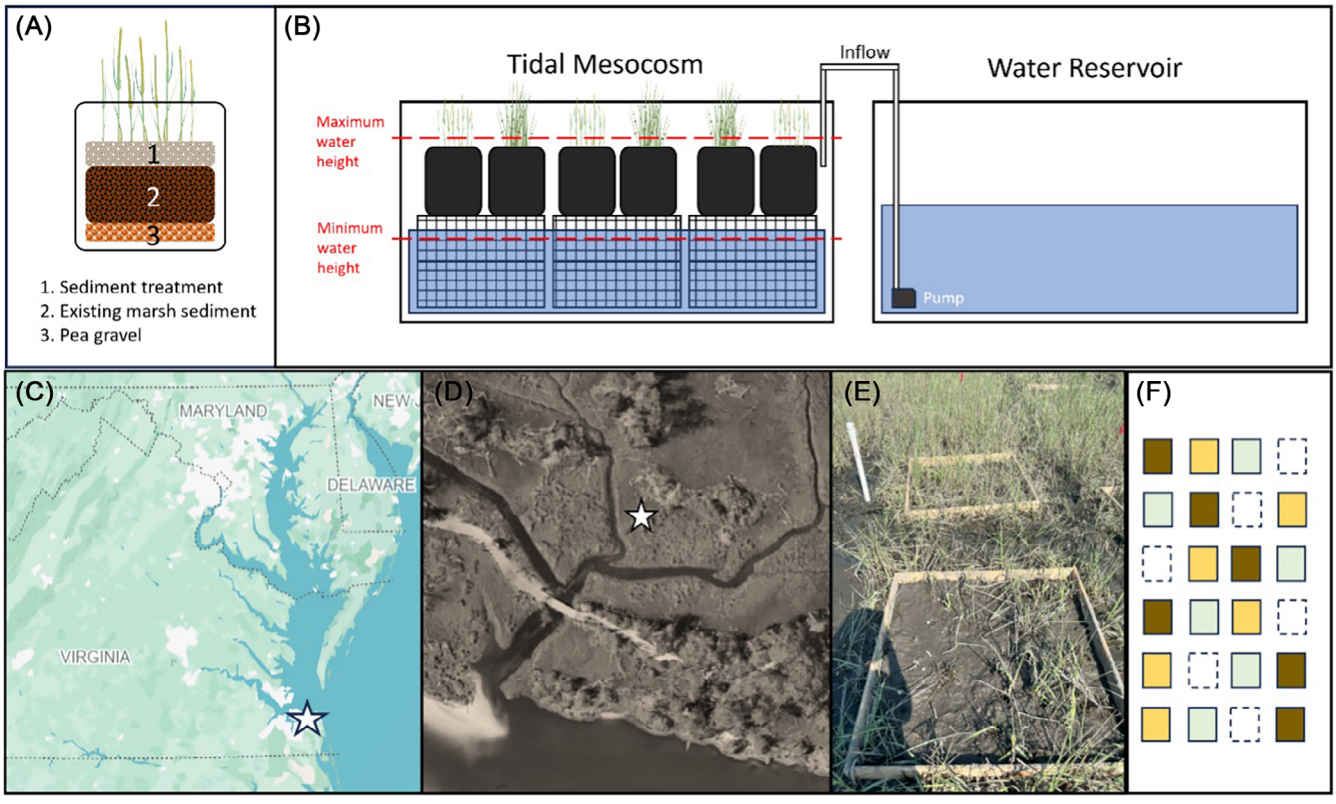


Figure 1. Experimental setup for greenhouse study and field study. (A) Diagram of experimental units in greenhouse study. (B) Mesocosm tank and corresponding water reservoir tank set up. (C) Location of site for field study in Virginia Beach, Virginia, United States. (D) Location of site within First Landing State Park. (E) Field study plots with wooden frames in *Spartina alterniflora*. (F) Diagram of block design for field study plots at site. Treatment type is as follows: brown = mud, yellow = sand/mud mix, light green = control, white with dotted outline = reference.

week throughout the duration of the experiment. Stem count included all stems present in each experimental unit. Percent cover was determined via visual estimation by the same person for every measuring event. Three stem diameters were randomly collected from different locations within each pot to determine average stem diameter. Bulk density was calculated by taking a 5-cm diameter soil core using a custom-made piano hinge corer. Soil cores were weighed for wet weight, dried in an oven at 60°C until constant mass was achieved (approximately 14 days), then weighed for dry weight. Bulk density was then calculated using the soil dry weight and volume of corer (g/cm^3) (Blake & Hartge 1986). Sediment height was measured monthly to determine compaction and expansion dynamics via a "mini" surface elevation table (mini-SET) throughout the experiment (as described in Stagg et al. 2022). Two perpendicular measurements were made across the pot's surface with the arm of the mini-SET, and lengths were collected for 12 sediment surface elevation data points in each experimental unit. Containers were monitored for being root bound throughout the growing season, and the experiment was ended when evidence of root restriction first arose (Day & Megonigal 1993; F. P. Day, 2024, ODU, personal communication). At the completion of the greenhouse experiment, aboveground biomass, belowground biomass, and bulk density samples were collected along with final bimonthly measurements and final sediment height as above. Ten stems from each pot were randomly collected for an estimation of aboveground biomass measurements, dried in an oven at 60°C until constant mass was achieved, and then weighed for dry weight (Madsen 1993) and used to calculate an average biomass per stem which was then extrapolated to the experimental unit based on the total stem count. A final soil core was also taken, as described above, for end of experiment bulk density. The remaining material in the pot was rinsed of soil using a stacked 2-mm sieve and 1-mm sieve to extract all roots, which were subsequently dried in an oven for 14 days at 60°C until constant mass was achieved and weighed for dry weight for total belowground biomass (Madsen 1993).

Field Study—Study Site and Experimental Design

This portion of our study was completed in a coastal salt marsh dominated by *S. alterniflora* at First Landing State Park, off Linkhorn Bay-Lynnhaven River, in Virginia Beach, Virginia, United States (36°53′13″N; 76°00′27″W; hereafter "Site"; Fig. 1C). The site was located in a marsh complex with an average elevation of 0.2 m (NAVD88-Geoid 18) in a humid subtropical climate (Beck et al. 2023) receiving 131 cm of precipitation annually (National Oceanic and Atmospheric Administration 2023). It is a microtidal system, with an average tidal range of approximately 50 cm (National Oceanic and Atmospheric Administration 2024). The marsh complex is a tidal estuary and has adjacent mudflats in the subtidal portions and low-lying coastal forest in the supratidal zone.

Using a randomized block design, we established twenty-four 1-m^2 plots and tested the impacts of two different TLP sediment types—mud and a sand/mud mix (Fig. 1D). These two types represent the most likely sediment applications the park may use in the future. Eighteen of the twenty-four plots had a 1-m^2 wooden frame made of 2.5 cm by 10.1 cm untreated pine boards, anchored into the ground with wood stakes (Fig. 1E). Six of the framed plots had 5 cm of mud added and six of the framed plots had 5 cm of sand/mud mix added, while the remaining plots served as control plots (frame, but no sediment) or reference plots (no frame or sediment). All sediments were collected in nearby portions of the marsh or the nearby beach, and the sand/mud mix was completed as described in the greenhouse study portion. In plots receiving sediment, a ceramic sediment plate was set on top of the original marsh surface before sediments were applied to serve as a marker horizon to track sediment accretion/erosion/compaction relative to the surface of the plate (Nolte et al. 2013). The mud and the sand/mud mix sediment treatments were added to the plots in the last week of May and first week of June 2024 and the plots were monitored till the end of the growing season in late September to capture short-term TLP impacts, while ensuring integrity of manipulative devices (frames) over the course of the experiment and reducing long-term impacts of researchers (Rinehart et al. 2024).

Field—Data Collection

Initial measurements (stem count, stem height, vegetation percent cover, and soil bulk density) for each experimental unit were collected at the time of sediment addition. Stem count, number of flowering stems, average stem height, and percent cover were collected biweekly throughout the remainder of the growing season until late September 2024. Stem count and number of flowering stems were collected by counting all stems within a 0.25 m^2 quadrat (0.5 × 0.5 m) of the plot. Average stem height was collected randomly by measuring three live stems from throughout the plot. Percent cover was determined via visual estimation. Bulk density was collected to 30 cm using the aforementioned corer. The core was divided into three portions 0–5, 5–15, and 15–30 cm, with the middle 5-cm of each subsection being used for belowground analysis. Samples were brought back to the lab and analyzed as described in the greenhouse study. At the completion of the field experiment in late September, aboveground biomass, belowground biomass, and bulk density samples were collected along with soil redox potential and final biweekly and monthly measurements. Ten live stems from each plot were haphazardly collected from throughout the plot to estimate aboveground biomass and were dried in an oven for 14 days at 60°C until constant mass was achieved, then weighed for dry weight (Madsen 1993). Two 30 cm × 5 cm cores were taken from each plot to determine belowground biomass and soil bulk density. Bulk density samples were treated as above, and the belowground biomass soil sample was broken down into three 10-cm subsections (0–10, 10–20, and 20–30 cm) and rinsed of soil to extract total roots using a stacked 2 mm sieve and 1 mm sieve, which were then dried in an oven for 14 days at 60°C and then weighed for dry weight (Bledsoe et al. 1999). Soil redox potential (Eh) was collected at the end of the season at 10 cm below the surface in the center of each plot using a Luktron YK-23RP ORP Meter and Paleo Terra redox probe (Niedermeier & Robinson 2007). We

did not conduct an Eh correction; the Pt electrodes and reference electrode were calibrated using a standard redox solution prior to use.

Marsh elevation was collected prior to sediment addition using a GNSS RTK-GPS (TopCon HiPer VR) and automated laser level (Spectra Precision LL300N) (RTK) and then measured again at the end of the growing season to determine overall change in marsh elevation. The height of the applied sediment was also collected by measuring the thickness of the sediment on top of the previously described ceramic sediment plate and was measured monthly until the end of the growing season to determine expansion via root zone expansion, compaction, or erosion of the added sediments.

Textural and Organic Matter Analysis

Representative sediment samples for both greenhouse and field sediment additions (sand, mud, and/or mix) were analyzed for their sand: mud ratio to establish sediment texture and allow for comparison between experiments. Approximately 10 g of sediment was soaked in a solution of deionized (DI) water and sodium hexametaphosphate for greater than 24 hours to facilitate disaggregation of fine-grained particles. Samples were then sonicated for 2 minutes prior to application on a stainless-steel sieve with 63-μm mesh. The sieve was rinsed with DI water until all fine-grained particles had passed through, with the sand and mud portions captured in pre-weighed vessels and dried to constant weight (Folk 1974). Additionally, approximately 2 g subsamples of aforementioned sediments were dried at 60°C until constant weight was achieved, ground to 2 mm by hand, and combusted in a box furnace at 500°C for 5 hours to determine organic matter content via loss on ignition (similar to Wang et al. 2011).

Statistical Analysis

Independent variables included sediment composition (both studies), plant species (greenhouse), mesocosm block (greenhouse), block (field), and time (both studies). The dependent variables included aboveground biomass, belowground biomass, bulk density, redox, stem count, stem height, stem diameter (greenhouse only), percent cover, and number of flowering stems. We tested for treatment effects and block effects on aboveground biomass, belowground biomass, and elevation change by conducting analysis of variance (ANOVA) analyses in R Studio (version 2023.12.1; R Core Team 2023) (Table 1) with subsequent Tukey HSD (honestly significant difference) analysis. For all other metrics, we used mixed effect models to examine metrics over time or by depth using the "lme4" package (version 1.1-35.1; Bates et al. 2023) using both linear and nonlinear (exponential) functions (Table 2). For all mixed effect models, ANOVA with Satterthwaite's method and suitable pairwise comparisons via a marginal means in the "emmeans" package (version 1.11.2-8; Lenth 2025) were also completed when suitable (Table S1).

Results

Greenhouse Study—Vegetation

Sediment application treatments did not significantly affect aboveground biomass relative to the control ($p > 0.05$; Tables 1 & S2; Fig. 2A), although sand application yielded significantly more aboveground biomass compared to the mud treatment ($p < 0.03$). Further, *Spartina* exhibited significantly greater overall aboveground biomass at the end of the season compared to *Juncus* ($p < 0.0001$) regardless of treatment (Tables 1 & S2; Fig. 2A). End of season belowground biomass was significantly less in the *Juncus* sand treatment compared to the *Juncus* control treatment (species × treatment: $p < 0.05$; Tables 1 & S2; Fig. 2) with no other differences occurring between any other treatments, species, or combination. Live cover was significantly greater in *Spartina* compared to *Juncus* and saw greater increases over time ($p < 0.0001$; Tables 2, S1, & S2; Fig. 3A), regardless of treatment. Stem counts saw a greater rate of increase in *Spartina* over time compared to *Juncus* ($p < 0.0001$). Further, *Spartina* stem counts in both the sand and mix treatments had a slower rate of increase over the experiment relative to the control treatments ($p < 0.0001$), but the mud treatment did not see this pattern (Tables 2, S1, & S2; Fig. 3B). Stem heights were significantly greater in *Juncus* compared to *Spartina* with *Juncus* stem heights in mix and sand treatments showing greater stem heights compared to the control; however, these treatment differences were largely eliminated by the end of the season ($p < 0.0001$; Tables S1 & 2; Fig. S1). Stem diameter also saw an increase over the course of the experiment, but did not differ by species, treatment, or any interactions ($p < 0.0001$; Tables S1 & S2; Fig. S2). Finally, the number of flowering stems followed an exponential curve over time in *Spartina*, but did not differ by treatment compared to the control or any other treatment (Tables 2, S1, & S2; Fig. S2B). Note, flowering was not monitored in *Juncus*

Table 1. Analysis of variance (ANOVA) *F*-values, degrees of freedom, and *p* values for treatment, species, and treatment × species for greenhouse results and field results. **p* less than 0.05; ***p* less than 0.01; ****p* less than 0.001. NS, not significant.

Greenhouse	*Treatment*	*Species*	*Treatment × species*
Aboveground biomass	$F_{[3,75]} = 4.364$**	$F_{[1,75]} = 21.83$***	*NS*
Belowground biomass	*NS*	*NS*	$F_{[3,72]} = 3.133$*
Bulk density	$F_{[3,3]} = 3.818$*	*NS*	*NS*
Elevation	$F_{[3,75]} = 4.725$**	*NS*	NS

Field	*Treatment*	*Depth*	*Treatment × depth*
Aboveground biomass	$F_{[3,20]} = 5.212$**		
Redox potential	*NS*		
Elevation change	*NS*		
Sediment loss	*NS*		

Table 2. Metric and model type, variables, estimates, SE, degrees of freedom (*df*), *t*-values, and *p* values for greenhouse and field data. *p less than 0.05; **p less than 0.01; ***p less than 0.001. SPAL = *Spartina alterniflora*.

Greenhouse	*Independent variable*	*Estimate*	*SE*	df	t-*Value* (p *value*)
Live cover (LME)	Treatment	NS			
	Species	SPAL = 20.039	SPAL = 4.101	SPAL = 113.332	SPAL = 4.886***
	Week	NS			
	Treatment × species	NS			
	Treatment × week	NS			
	Species × week	SPAL × week = 3.751	SPAL × week = 0.387	SPAL × week = 391.036	SPAL × week = 9.690***
	Treatment × species × week	NS			
Stem height (LME)	Treatment	Mix = 8.170; Sand = 12.820	Mix = 3.861; Sand = 3.861	Mix = 133.568; Sand = 133.568	Mix = 2.116*; Sand = 3.320**
	Species	SPAL = −16.482	SPAL = 3.861	SPAL = 133.568	SPAL = -4.268 ***
	Week	NS			
	Treatment × species	NS			
	Treatment × week	Mud × week = −0.955; Sand × week = −1.180	Mud × week = 0.419; Sand × week = 0.419	Mud × week = 392.001; Sand × week = 392.001	Mud × week = −2.279*; Sand × week = −2.815**
	Species × week	NS			
	Treatment × species × week	NS			
Stem count (LME)	Treatment	NS			
	Species	NS			
	Week	NS			
	Treatment × species	NS			
	Treatment × week	NS			
	Species × week	SPAL × week = 1.633	SPAL × week = 0.186	SPAL × week = 389.174	SPAL × week = 8.792***
	Treatment × species × week	Mix × SPAL × week = −0.708; Sand × SPAL × week = −0.823	Mix × SPAL × week = 0.261 Sand × SPAL × week = 0.2613	Mix × SPAL × week = 389.177 Sand × SPAL × week = 389.177	Mix × SPAL × week = −2.710** Sand × SPAL × week = −3.151**
Stem diameter (LME)	Treatment	NS			
	Species	NS			
	Week	0.36128	0.0335	320.000	10.780***
	Treatment × species	NS			
	Treatment × week	NS			
	Species × week	NS			
	Treatment × species × week	NS			

Field	*Independent variable*	*Estimate*	*SE*	df	t-*Value* (p *value*)
Stem height (LME)	Treatment	NS			
	Time	NS			
	Treatment × time	NS			
Stem count (LME)	Treatment	Mix = −11.628; Mud = −15.879	Mix = 4.725; Mud = 4.725	Mix = 35.448; Mud = 35.448	Mix = −2.461*; Mud = −3.361**

Table 2. Continued

Field	*Independent variable*	*Estimate*	*SE*	df	t-*Value* (p *value*)
	Time	NS			
	Treatment × time	Mix × time = −0.766	Mix × time = 0.293	Mix × time = 164.000	Mix × time = −2.614**
Live cover (LME)	Treatment	Mix = −22.1508	Mix = 9.9374	Mix = 23.5318	Mix = −2.229*
	Time	NS			
	Treatment × time	Mix × time = −0.966; Mud × time = −0.838	Mix × time = 0.343; Mud × time = 0.343	Mix × time = 164.000; Mud × time = 164.000	Mix × time = −2.819**; Mud × time = −2.445*
Flowers (NLME)	Treatment	NS			
	Time	0.852	0.140	184	6.083***
	Treatment × time	Mix = −0.501	Mix = 0.198	Mix = 184	Mix = −2.531
Belowground biomass (LME)	Treatment	Mud = −4.092	Mud = 2.042	Mud = 59.377	Mud = −2.004*
	Depth	20–30 = −5.327	20–30 = 1.967	20–30 = 40.000	20–30 = −2.708**
	Treatment × depth	Mud × 10–20 = 6.585	Mud × 10–20 = 2.781	Mud × 10–20 = 40.000	Mud × 10–20 = 2.367*
		Mud × 20–30 = 6.138	Mud × 20–30 = 2.781	Mud × 20–30 = 40.000	Mud × 20–30 = 2.207*
Bulk density (LME)	Treatment	Mix = 0.643	Mix = 0.059	Mix = 59.572	Mix = −10.835***
	Depth	NS			
	Treatment × depth	Mix × 5–15 = −0.606	Mix × 5–15 = 0.082	Mix × 5–15 = 40.000	Mix × 5–15 = −7.384***
		Mix × 15–30 = −0.626	Mix × 15–30 = 0.082	Mix × 15–30 = 40.000	Mix × 15–30 = −7.618***

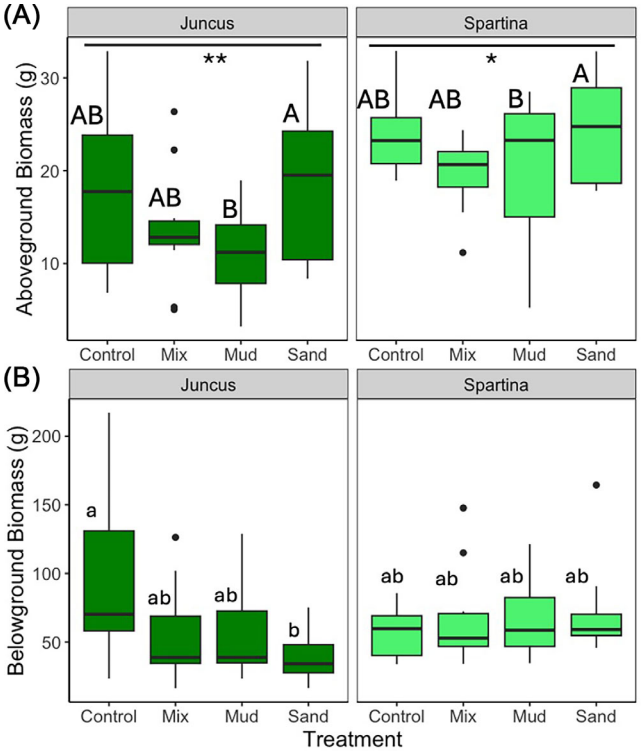


Figure 2. Box plots displaying (A) aboveground and (B) belowground biomass per experimental unit for each species and each treatment type in the greenhouse study (*Juncus* = left panel in dark green, *Spartina* = right panel in light green). Asterisks under line indicate significant differences between species response. Capital letters within graphs denote significant differences between treatment types and lowercase letters within graphs denote significant differences in the species × treatment interaction. The thick horizontal line in the middle of each box represents the median, the outside of the boxes represents the range between the first quartile and third quartile of data, and the whiskers represent the minimum and maximum values. Outliers are represented by single points.

in the greenhouse study as flowering had already largely finished at the start of the study in June 2024.

Greenhouse Study—Sediment

For the greenhouse, the applied sand treatment sediment contained 100% sand, the mix contained 77% sand, and the mud contained 53% sand; organic matter ranged from 0% in sand to 7% in the mud (Table S2). The sand treatment saw significantly greater loss of elevation by the end of the season compared to the control ($p < 0.01$; Tables 1 & S2) as well as the mud treatment ($p < 0.05$; Table 1; Figs. 3C & S3) in both *Juncus* and *Spartina*. Bulk density was not significantly different in any of the treatments compared to the control; however, it was significantly greater in the mud treatment compared to the mix treatment regardless of species ($p < 0.05$; Tables 1 & S2).

Field Study—Vegetation

End of season aboveground biomass was significantly less in the mix treatment compared to both the reference and the control

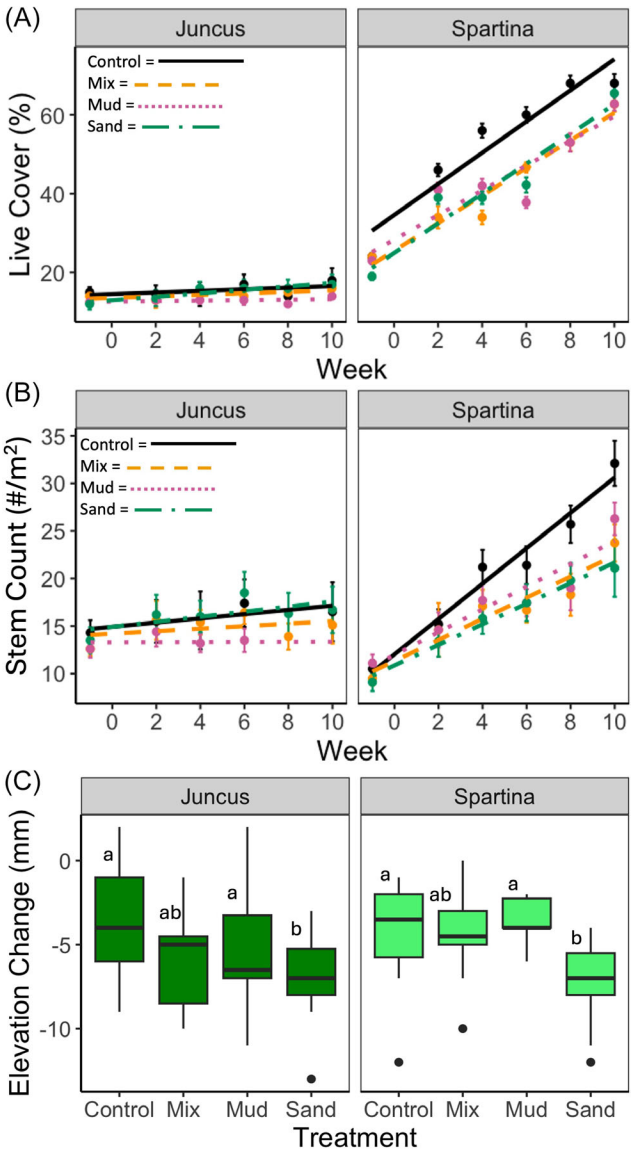


Figure 3. Live cover (A) and stem counts (B) over time by treatment. Box plots displaying elevation change over the course of the experiment (C). Values for *Juncus* are on the left panel and values for *Spartina* are on the right panel. The thick horizontal line in the middle of each box represents the median; the outside of the boxes represents the range between the first quartile and third quartile of data, and the whiskers represent the minimum and maximum values. Outliers are represented by single points. Lowercase letters within graphs denote significant differences in the species × treatment interaction.

treatments ($p < 0.04$; Table 1; Fig. 4A), but the mud treatment did not differ. Belowground biomass was overall less in the mud treatment compared to reference (Tables 2, S1, & S3; Fig. 4B), but this difference was only present in the top 10 cm (Tables 2 & S3). Additionally, the lowest depth interval (20–30 cm) had significantly less biomass compared to the upper depths (Tables 2, S1, & S3; Fig. 4B). Live cover saw a greater rate of decrease over time in the mix and mud treatments compared to the reference and control plots (treatment × time = $p < 0.01$, $p < 0.001$, respectively; Tables 2, S1, & S3; Fig. 4C) with the mix treatment also having lower overall cover ($p < 0.001$; Tables 2 & S3). End of season cover was approximately 20–30% lower than reference and control (Fig. 4C). Stem counts were significantly lower in mud and mix treatments compared to reference ($p < 0.01$, $p < 0.001$, respectively; Tables 2, S1, & S3; Fig. 4D) with the mix treatment additionally seeing a negative interaction over time ($p < 0.001$). Stem heights did not differ by treatment, over time, or the treatment × time interaction (Tables 2, S1, & S3). Finally, flowering saw an exponential increase over time and a negative interaction in the mix treatment over time with fewer flowers being produced compared to the reference (Tables 2, S1, & S3; Fig. 4E).

Field Study—Sediment and Edaphic Conditions

For the field study, the mix sediment application contained 65% sand and 3% organic matter and the mud treatment contained 24% sand and 26% organic matter (Table S3). Despite relatively uniform elevations prior to the start of the experiment, overall marsh elevation at the end of the season did not differ with either sediment addition (mix or mud) compared to both the control and reference, even with the addition of 5 cm of sediment (Fig. 5A). Thickness of the sediment addition treatment layer, as measured from the sediment plates in the mud and mix treatment plots, decreased over time ($p < 0.001$), but did not differ between the mud and mix treatments with 3–3.5 cm being lost in both sediment types by the end of the experiment from the original 5.0 cm (Table S3; Fig. 5B). Soil bulk density was greater in the mix treatment overall ($p < 0.001$; Tables 2, S1, & S3) compared to the reference, but this was only seen in the top 5 cm and the lower depth intervals in the mix treatment returned to reference bulk density for their respective depths (Fig. S4, Tables 2 & S3). Redox potential exhibited no change between any of the treatment types (Tables 1 & S3).

Discussion

Understanding how salt marsh species respond to differing sediment types in TLP projects will aid in understanding marsh resilience, ecosystem development trajectories, and project planning, and provides critical information on the interaction between soil and vegetation in coastal salt marshes. Our study focuses on sediment composition and species responses to identify sediment effects on plant growth metrics and elevation in a controlled mesocosm and focuses on a natural marsh response in a field setting in the short-term period after TLP application. Understanding short-term responses of TLP application can allow mitigating immediate detrimental impacts, rapid restoration of properties, functions, and services, and providing a detailed assessment of ecosystem responses to allow for adaptive management to meet long-term goals (Berkowitz et al. 2021; Reilly et al. 2026). Several species-specific responses were identified in the greenhouse study, with *Juncus* and *Spartina* having some differing responses to treatments; *Spartina* outperformed *Juncus* in aboveground biomass production regardless of treatment and *Juncus* belowground biomass

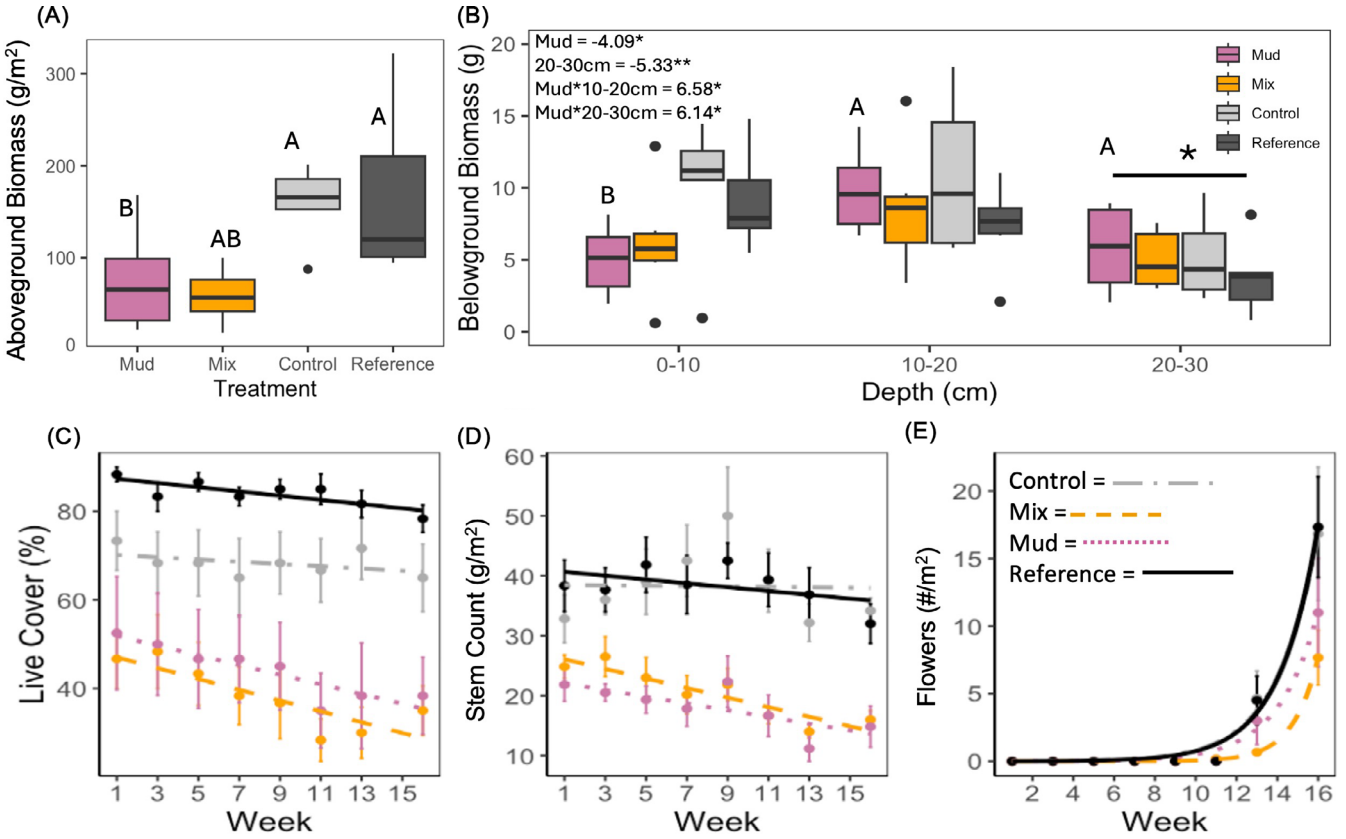


Figure 4. Aboveground biomass (A) and belowground biomass by depth (B) per treatment type. Live cover per m^2 (C), stem density per m^2 (D), and number of flowers per m^2 (E) by treatment type over time. Asterisks and capital letters denote significant differences between treatments, depths, or their interactions. For box plots, the thick horizontal line in the middle of each box represents the median, the outside of the boxes represents the range between the first quartile and third quartile of data, and the whiskers represent the minimum and maximum values with outliers are represented by single points. All points represent mean values with error bars showing SE. Legend for (A) found in panel (B). Legend for (C) and (D) found in panel (E). Panel (B) showcases estimates and significant differences (*$p < 0.05$, **$p < 0.01$) from mixed effects model output. Full outputs can be found in Table 2.

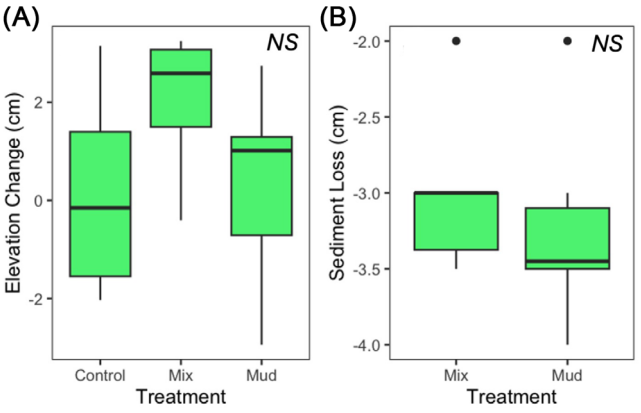


Figure 5. (A) Elevation change from pre-season to post-season of control, mix, and mud treatment plots relative to reference marsh plots and (B) sediment loss from initial sediment placement to end of experiment based on sediment plate measurements in mix and mud treatments shown in box plots. The thick horizontal line in the middle of each box represents the median, the outside of the boxes represents the range between the first quartile and third quartile of data, and the whiskers represent the minimum and maximum values. Outliers are represented by single points.

was more negatively affected by the sand treatment. In the field, it was found that overall marsh elevation, as measured by the RTK-GPS, did not differ significantly between treatment types over the course of the growing season despite sediment additions; additionally, multiple growth differences were observed at the conclusion of the growing season with plots receiving experimental additions having muted or delayed growth.

In the greenhouse, *Spartina* overall grew more than *Juncus*, as demonstrated by greater end of season biomass, greater values and increases in percent cover, and greater stem counts over time. The inherent morphological differences between *Juncus* and *Spartina*, as with many rushes and grasses, likely contributed to some of the differences (Eleuterius 1976; Xu & Chang 2017; Nelson & Moore 2020). Because of these structural differences, we expected to see greater percent cover from *Spartina* due to its larger, more outspread leaves, greater stem counts, and vigorous growth form (Hartman 1988; Proffitt et al. 2003). *Spartina* had average percent covers ranging from 46 to 60% across all treatments and *Juncus* had average percent covers ranging from 13 to 16%. Conversely, *Juncus* had average heights of approximately 62 cm across all treatments and *Spartina* had average heights of approximately 45 cm. Both species

had greater aboveground biomass in sand treatments compared to mud treatment, which aligns with prior studies showing higher productivity with greater sand content, potentially by improved aeration/drainage (Slocum et al. 2005; La Peyre et al. 2009). However, *Juncus* also showed a negative belowground biomass response to the sand treatment, possibly due to decreased nutrient retention. The mud treatment bulk density did not differ from the control but did have greater bulk density than the mix treatment regardless of species. While most TLP studies commonly report increased bulk density in sandier sediments (Slocum et al. 2005; La Peyre et al. 2009), this may be due to the mechanical mixing of the two sediments at the start of the experiment. Pots with the sand treatment exhibited the most elevation loss, potentially due to lesser root zone expansion compared to other treatments as well as increased surface drainage due to sand content. While we did not measure roots by depth in the greenhouse, *Juncus* saw lower total belowground biomass, potentially impacting root zone expansion impacts on elevation. It is important to note that the control also lost some elevation due to compaction of sediments in the sods and the supplemental mud to fill in gaps and edges. The mud treatment maintained its elevation the most compared to the other treatments and was similar to the control potentially due to its already compacted nature, making it more structurally stable and limiting additional compaction throughout the experiment (Burland 1990).

Results from our field study provided a paired real-world comparison for *Spartina* in the first season after sediment application, a critical time for both project success and for impacts on ecosystem functioning. Aboveground and belowground biomass was significantly lower at the end of the season for the mud treatment compared to the control and reference. Additionally, both the mud and mix treatments saw losses in the live cover and stem counts over the course of the experiment. These differences may have been caused by the physical stress of the sediment application or by the physiochemical properties of the additional sediment, such as nutrient content and availability or changes in drainage/inundation (Schrift et al. 2008; Davis et al. 2022). Porewater redox potential showed no difference between treatments, as predicted by prior TLP research showing redox potential changing significantly required more than one season, and is unlikely to be the driver of these differences (Reimold et al. 1978; Mendelssohn & Kuhn 2003). Initial dieback is common in the first season after sediment application in other TLP work due to these stresses, thus explaining the multiple lower plant metrics for the treated plots compared to control and reference plots. All plots, regardless of treatment, exhibited a dip in average stem height at week 9 that correlated with an increase in number of stems at the same time, indicating the emergence of new shoots. With this occurring in the late summer, these new shoots are likely the fall stems that will provide overwintering support for the following year's growth (Wijte & Gallagher 1991); however, previous experiments have also seen new shoot emergence site wide after the disturbance of the sediment application and perhaps a shortcoming of plot based TLP studies (Turner 1988). Overall, the dieback seen in our study can cause a myriad of short-term losses in ecosystem functions and service, particularly for those relying on aboveground metrics delivered in the immediate time scale (e.g. wave attenuation and habitat), even if more equal growth is expected in subsequent seasons (Croft et al. 2006; Van Zomeren et al. 2018; Moore et al. 2021).

In the field, overall marsh elevation did not differ significantly between treatment types at the end of the experiment, even with the additional 5 cm of sediment in the mud and mix treatments. The mud treatment and control plots were almost identical in elevation despite the additional sediment, and the mix treatment maintained only 2 cm greater elevation on average, although this was not significantly different. Despite these minimal total elevation changes, both the mix and mud treatments still had approximately 2 cm of sediment on top of the sediment plate at the end of the season indicating potential compaction of applied sediments or deeper horizons (e.g. Graham & Mendelssohn 2013). Finally, the mix treatment had a higher bulk density in the top horizon (0–5 cm) likely due to the introduction of greater amounts of sands in the sediment used, but lower horizons were the same as reference plots at those depths. This is inconsistent with the greenhouse, but may be a product of total versus surface bulk density. Overall, these changes indicate that sediments added either eroded away or compacted after application (Morris et al. 2016). It should also be noted that error within the GNSS RTK-GPS equipment (approximately 1–2 cm) and tidal/seasonal inundation influences on marsh elevation may also play a role in our findings (Paquette et al. 2004). Regardless, this deserves greater attention in future TLP work with the need for more detailed elevation monitoring and accounting for sediment compaction.

Based on these overall results, sediment composition of the thin layer of applied material does determine the response of *Spartina alterniflora* with some metrics expected to recover in future seasons and sediments having a more muted response in *Juncus romerianus*. Treatment effects were seen in some metrics of the greenhouse and field experiments for *Spartina*, previous studies have demonstrated that dieback in the first growing season is expected and is likely to rebound in subsequent growing seasons (Ford et al. 1999; Raposa et al. 2023). While *Juncus* had limited responses to different treatments in the greenhouse, the observed reductions in belowground biomass with sand addition may cause a delayed response in aboveground growth. Species responses may equalize over time, but initial marsh response to TLP, vegetation variability may affect project results in the first growing season and in the short term, with *Spartina* performing in a more robust manner than *Juncus*. The presence of sand, in either the pure sand treatment or the mix treatment, played a strong role in both vegetation and sediment response. The mix treatment, being a 1:1 ratio of pure sand mixed with marsh mud, which already contained 53% sand in the greenhouse and 24% in the field, caused the mix treatments to have 77 and 65% sand, respectively, which is coarser than the mud matrix upon which it was introduced. We also recognize that these differences in sand content between the greenhouse and field are substantial and will certainly impact results. While the presence of sand in most studies allowed the marsh to better maintain its increased elevation post-application, we did not

see any differences in the field experiment (mix vs. mud) and greater elevation loss in the greenhouse. Further, some vegetation metrics in the sand treatments exhibited decreased growth over the initial growing season. Although nutrient analyses were not possible for this study, coarse-textured sediments have been shown to retain nutrients and moisture less efficiently than fine-textured sediments and may not promote as much biomass generation (Broome et al. 1986; Boyer & Zedler 1998). However, sediment studies have been conducted confirming sand presence improves soil strength in salt marshes (Nittrouer et al. 2012; Stoorvogel et al. 2024). If *Spartina* does rebound in future seasons, even if it is equal to other treatments' vegetation, a mix of sand and mud would be the preferred sediment composition for TLP projects in this region. The mix treatment has been demonstrated in both a greenhouse and field setting to maintain its elevation better than other compositions under conditions present in the southern Chesapeake Bay. Dredge material from areas composed of this combination of sediment would be well suited for *Spartina* marshes. However, this is contingent on vegetation recovery something that is not always observed or observed to the degree in which it is expected after multiple seasons (Raposa et al. 2023). Restoration and resilience practitioners should consider implications of sediment composition on elevation change when setting elevation goals for projects, with composition affecting the rate of initial sediment compaction to meet desired elevations. Project elevation goals should be determined by the preferred elevation of the dominant species within the intertidal zone, incorporating the initial height of the marsh, potential elevation loss of the applied sediment, and rate of sea-level rise for the project site, for optimal productivity over the preferred lifespan of the project and if there will be repeated sediment additions (Slocum et al. 2005; Stagg & Mendelssohn 2010; Reilly et al. 2026). Finally, despite the relatively short duration of our experiments, our findings provide information on ecosystem response with potentially major implications for short-term processes and properties (e.g. faunal habitat and wave attenuation). Efforts should be made to minimize short-term impacts to improve long-term outcomes and ensure continuity in ecosystem functioning (Berkowitz et al. 2021).

Future research on TLP should continue to consider species-specific responses by incorporating additional marsh species to evaluate vegetation rebound and community reestablishment and development. Our current understanding of *Spartina* growth is that it responds quickly in response to changing conditions, including TLP, but less is known in general about *Juncus* growth patterns (Blum 1993). Despite being a common species in many marshes throughout the southeastern United States, *Juncus* has limited data on relevant components of its life history, such as belowground biomass production and favorable elevations within the intertidal zone. Research has also been minimal concerning its role in the marsh ecosystem compared to other graminoid species in eastern North American. Further, long-term monitoring of established projects needs to be established to identify whether the short-term trajectory of species-level responses continues (Slocum et al. 2005; Raposa et al. 2022, 2023). Our results from a single growing season provide information on short-term responses, which represent a critical time period for restoration success, and remain logistically feasible for greenhouse experimentation. There remains, however, a pressing need to understand multi-year and decadal responses (Berkowitz et al. 2021; Reilly et al. 2026). Further, sediment dynamics should also be monitored at a finer scale, ideally with the installation of a surface elevation table (SET) and shallow rods to evaluate root zone expansion (Cahoon 2024), to establish baseline sedimentation rates and establish whether additional adaptive management is necessary. Finally, future work could test the species-level responses to TLP-layer thickness while holding the sediment type constant. This work's multi-species and multi-sediment comparison as well as its comparison of greenhouse and field-based experiment provide a multi-level comparison of TLP efficacy during the critical time immediately after application that links scientific inquiry to real-world practice.

Acknowledgments

We are grateful for the support from members of the Coastal Plant and Ecotone Ecology and Wetland Plant and Restoration Ecology labs at Old Dominion University as well as the many volunteers who assisted with this work (particularly J. Hemphill, L. Hemphill, and O. Dubovyk). We thank Drs. F. Day and C. Stagg for their advice and support of the research and review of this document. A special thanks to the Knotts Creek Nursery, Virginia DCR, and First Landing State Park for land access and logistics support, specifically M. and A. Hilton, E. Moleen, and B. Widender. This work was supported through the generous support of the Paul W. Kirk, Jr. Wetlands Biology Research Award, the Navy SEAL Foundation Scholarship, the Virginia Association of Wetland Professionals Student Research Scholarship, and the Hampton Roads Sanitation District Environmental Opportunity Grant.

LITERATURE CITED

Allison SK (1996) Recruitment and establishment of salt marsh plants following disturbance by flooding. The American Midland Naturalist 136:232–247. https://doi.org/10.2307/2426728

Barbier EB, Hacker SD, Kennedy C, Koch EW, Stier AC, Silliman BR (2011) The value of estuarine and coastal ecosystem services. Ecological Monographs 81:169–193. https://doi.org/10.1890/10-1510.1

Bates D, Maechler M, Bolker B, Walker S (2023) lme4: linear mixed-effects models using 'Eigen' and S4 (version 1.1-35.1) [computer software]. R Foundation for Statistical Computing. https://cran.r-project.org/package=lme4 (accessed 3 September 2024)

Beck HE, McVicar TR, Vergopolan N, Berg A, Lutsko NJ, Dufour A, Zeng Z, Jiang X, van Dijk AIJM, Miralles DG (2023) High-resolution (1 km) Köppen-Geiger maps for 1901–2099 based on constrained CMIP6 projections. Scientific Data 10:724. https://doi.org/10.1038/s41597-023-02549-6

Beisner BE, Haydon DT, Cuddington K (2003) Alternative stable states in ecology. Frontiers in Ecology and the Environment 1:376–382.

Berkowitz JF, Beane NR, Philley KD, Hurst NR, Jung JF (2021) An assessment of long-term, multipurpose ecosystem functions and engineering benefits derived from historical dredged sediment beneficial use projects. In: Dredging operations environmental research and engineering with nature programs. ERDC/EL TR-21-4. US Army Corps of Engineers, Washington, D.C.

Blake GR, Hartge KH (1986) Bulk density. Pages 363–375. In: Klute A (ed) Methods of soil analysis. American Society of Agronomy, Inc. & Soil Science Society of America, Inc., Madison, Wisconsin. https://doi.org/10.2136/sssabookser5.1.2ed.c13

Bledsoe CS, Fahey TJ, Day FP, Ruess RW (1999) Measurement of static root parameters. Pages 413–436. In: Robertson GP, Coleman DC, Bledsoe CS, Sollins P (eds) Standard soil methods for long-term ecological research. Oxford University Press, New York. https://doi.org/10.1093/oso/9780195120837.003.0019

Blum LK (1993) *Spartina alterniflora* root dynamics in a Virginia marsh. Marine Ecology Progress Series 102:169–178. https://doi.org/10.3354/meps102169

Boyer KE, Zedler JB (1998) Effects of nitrogen additions on the vertical structure of a constructed cordgrass marsh. Ecological Applications 8:692–705.

Broome SW, Seneca ED, Woodhouse WW (1986) Long-term growth and development of transplants of the salt-marsh grass *Spartina alterniflora*. Estuaries 9:63–74. https://doi.org/10.2307/1352194

Burland JB (1990) On the compressibility and shear strength of natural clays. Géotechnique 40:329–378. https://doi.org/10.1680/geot.1990.40.3.329

Cahoon DR (2024) Measuring and interpreting the surface and shallow subsurface process influences on coastal wetland elevation: a review. Estuaries and Coasts 47:1708–1734. https://doi.org/10.1007/s12237-024-01332-z

Cahoon DR, McKee KL, Morris JT (2021) How plants influence resilience of salt marsh and mangrove wetlands to sea-level rise. Estuaries and Coasts 44:883–898. https://doi.org/10.1007/s12237-020-00834-w

Cornu CE, Sadro S (2002) Physical and functional responses to experimental marsh surface elevation manipulation in Coos Bay's south slough. Restoration Ecology 10:474–486. https://doi.org/10.1046/j.1526-100X.2002.01035.x

Croft AL, Leonard LA, Alphin TD, Cahoon LB, Posey MH (2006) The effects of thin layer sand renourishment on tidal marsh processes: Masonboro Island, North Carolina. Estuaries and Coasts 29:737–750. https://doi.org/10.1007/BF02786525

Davis J, Currin C, Mushegian N (2022) Effective use of thin layer sediment application in *Spartina alterniflora* marshes is guided by elevation-biomass relationship. Ecological Engineering 177:106566. https://doi.org/10.1016/j.ecoleng.2022.106566

Day FP, Megonigal JP (1993) The relationship between variable hydroperiod, production allocation, and belowground organic turnover in forested wetlands. Wetlands 13:115–121. https://doi.org/10.1007/BF03160871

Eleuterius LN (1976) The distribution of *Juncus roemerianus* in the salt marshes of North America. Chesapeake Science 17:289–292. https://doi.org/10.2307/1350516

Enwright NM, Griffith KT, Osland MJ (2016) Barriers to and opportunities for landward migration of coastal wetlands with sea-level rise. Frontiers in Ecology and the Environment 14:307–316. https://doi.org/10.1002/fee.1282

Folk RL (1974) Petrology of sedimentary rocks. Hemphill Publishing Co., Austin, Texas

Ford MA, Cahoon DR, Lynch JC (1999) Restoring marsh elevation in a rapidly subsiding salt marsh by thin-layer deposition of dredged material. Ecological Engineering 12:189–205. https://doi.org/10.1016/S0925-8574(98)00061-5

Friedrichs CT, Perry JE (2001) Tidal salt marsh morphodynamics: a synthesis. Journal of Coastal Research 17:7–37. https://www.jstor.org/stable/25736162

Friess DA, Yando ES, Alemu JB, Wong LW, Soto SD, Bhatia N (2020) Ecosystem services and disservices of mangrove forests and salt marshes. Pages 107–138. In: Oceanography and marine biology. Vol 58. CRC Press, Boca Raton, Florida. https://doi.org/10.1201/9780429351495-3

Graham SA, Mendelssohn IA (2013) Functional assessment of differential sediment slurry applications in a deteriorating brackish marsh. Ecological Engineering 51:264–274. https://doi.org/10.1016/j.ecoleng.2012.12.031

Gunderson LH, Allen CR, Holling CS (2012) Foundations of ecological resilience. Island Press, Chicago, Illinois

Hartman JM (1988) Recolonization of small disturbance patches in a New England salt marsh. American Journal of Botany 75:1625–1631. https://doi.org/10.1002/j.1537-2197.1988.tb11239.x

Hughes MG, Rogers K, Wen L (2019) Saline wetland extents and tidal inundation regimes on a micro-tidal coast, New South Wales, Australia. Estuarine, Coastal and Shelf Science 227:106297. https://doi.org/10.1016/j.ecss.2019.106297

Kirwan ML, Megonigal JP (2013) Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504:53–60. https://doi.org/10.1038/nature12856

Kirwan ML, Murray AB (2007) A coupled geomorphic and ecological model of tidal marsh evolution. Proceedings of the National Academy of Sciences U.S.A. 104:6118–6122. https://doi.org/10.1073/pnas.0700958104

Koning CO (2004) Impacts of small amounts of sandy sediment on wetland soils and vegetation: results from field and greenhouse studies. Wetlands 24:295–308.

La Peyre MK, Gossman B, Piazza BP (2009) Short- and long-term response of deteriorating brackish marshes and open-water ponds to sediment enhancement by thin-layer dredge disposal. Estuaries and Coasts 32:390–402. https://doi.org/10.1007/s12237-008-9126-8

Lamothe KA, Somers KM, Jackson DA (2019) Linking the ball-and-cup analogy and ordination trajectories to describe ecosystem stability, resistance, and resilience. Ecosphere 10:e02629. https://doi.org/10.1002/ecs2.2629

Lenth R (2025) emmeans: estimated marginal means, aka least-squares means. R package version 1.11.2-8. https://CRAN.R-project.org/package=emmeans (accessed 15 August 2025)

Love R, Milne GA, Tarasov L, Engelhart SE, Hijma MP, Latychev K, Horton BP, Törnqvist TE (2016) The contribution of glacial isostatic adjustment to projections of sea-level change along the Atlantic and Gulf coasts of North America. Earth's Future 4:440–464. https://doi.org/10.1002/2016EF000363

Madsen JD (1993) Biomass techniques for monitoring and assessing control of aquatic vegetation. Lake and Reservoir Management 7:141–154. https://doi.org/10.1080/07438149309354266

Mariotti G, Carr J (2014) Dual role of salt marsh retreat: long-term loss and short-term resilience. Water Resources Research 50:2963–2974. https://doi.org/10.1002/2013WR014676

Mcowen CJ, Weatherdon LV, Van Bochove JW, Sullivan E, Blyth S, Zockler C, et al. (2017) A global map of saltmarshes. Biodiversity Data Journal 5:e11764. https://doi.org/10.3897/BDJ.5.e11764

Mendelssohn IA, Kuhn NL (2003) Sediment subsidy: effects on soil–plant responses in a rapidly submerging coastal salt marsh. Ecological Engineering 21:115–128. https://doi.org/10.1016/j.ecoleng.2003.09.006

Moffett KB, Nardin W, Silvestri S, Wang C, Temmerman S (2015) Multiple stable states and catastrophic shifts in coastal wetlands: progress, challenges, and opportunities in validating theory using remote sensing and other methods. Remote Sensing 7:10184–10226. https://doi.org/10.3390/rs70810184

Möller I, Kudella M, Rupprecht F, Spencer T, Paul M, van Wesenbeeck BK, et al. (2014) Wave attenuation over coastal salt marshes under storm surge conditions. Nature Geoscience 7:727–731. https://doi.org/10.1038/ngeo2251

Moore GE, Burdick DM, Routhier MR, Novak AB, Payne AR (2021) Effects of a large-scale, natural sediment deposition event on plant cover in a Massachusetts salt marsh. PLoS One 16:e0245564. https://doi.org/10.1371/journal.pone.0245564

Morris JT, Barber DC, Callaway JC, Chambers R, Hagen SC, Hopkinson CS, et al. (2016) Contributions of organic and inorganic matter to sediment volume and accretion in tidal wetlands at steady state. Earth's Future 4:110–121. https://doi.org/10.1002/2015EF000334

Mudd SM, Howell SM, Morris JT (2009) Impact of dynamic feedbacks between sedimentation, sea-level rise, and biomass production on near-surface

marsh stratigraphy and carbon accumulation. Estuarine, Coastal and Shelf Science 82:377–389. https://doi.org/10.1016/j.ecss.2009.01.028

National Oceanic and Atmospheric Administration (2023) High and low tide data for Sewells point, VA (station ID: 8638610), May 1–October 31, 2023 [Dataset]. NOAA Tides & Currents. https://tidesandcurrents.noaa.gov/reports.html?type=maxmin&bdate=20230427&edate=20231027&units=metric&datum=NAVD&id=8638610&retrieve=Retrieve (accessed Apr 2024)

National Oceanic and Atmospheric Administration (2024) Tide predictions for Broad Bay Canal East, Virginia (station ID: 8638923) [Dataset]. NOAA Tides & Currents. https://tidesandcurrents.noaa.gov/noaatideannual.html?id=8638923 (accessed Nov 2024)

Nelson CJ, Moore KJ (2020) Grass morphology. Pages 23–49. In: Moore KJ, Collins M, Nelson CJ, Redfearn DD (eds) Forages: the science of grassland agriculture. 7th edition. Vol 2. John Wiley & Sons, West Sussex, United Kingdom

Nicholls RJ, Cazenave A (2010) Sea-level rise and its impact on coastal zones. Science 328:1517–1520. https://doi.org/10.1126/science.1185782

Niedermeier A, Robinson JS (2007) Hydrological controls on soil redox dynamics in a peat-based, restored wetland. Geoderma 137:318–326. https://doi.org/10.1016/j.geoderma.2006.08.027

Nittrouer JA, Best JL, Brantley C, Cash RW, Czapiga M, Kumar P, Parker G (2012) Mitigating land loss in coastal Louisiana by controlled diversion of Mississippi River sand. Nature Geoscience 5:534–537. https://doi.org/10.1038/ngeo1525

Nolte S, Koppenaal EC, Esselink P, Dijkema KS, Schuerch M, De Groot AV, Bakker JP, Temmerman S (2013) Measuring sedimentation in tidal marshes: a review on methods and their applicability in biogeomorphological studies. Journal of Coastal Conservation 17:301–325. https://doi.org/10.1007/s11852-013-0238-3

Osland MJ, Chivoiu B, Grace JB, Enwright NM, Guntenspergen GR, Buffington KJ, Thorne KM, Carr JA, Sweet WV, Couvillion BR (2024) Rising seas could cross thresholds for initiating coastal wetland drowning within decades across much of the United States. Communications Earth & Environment 5:1–8. https://doi.org/10.1038/s43247-024-01537-x

Paquette CH, Sundberg KL, Boumans RMJ, Chmura GL (2004) Changes in salt-marsh surface elevation due to variability in evapotranspiration and tidal flooding. Estuaries 27:82–89. https://doi.org/10.1007/BF02803562

Poorter H, Bühler J, van Dusschoten D, Climent J, Postma JA (2012) Pot size matters: a meta-analysis of the effects of rooting volume on plant growth. Functional Plant Biology 39:839–850. https://doi.org/10.1071/FP12049

Proffitt CE, Travis SE, Edwards KR (2003) Genotype and elevation influence *Spartina alterniflora* colonization and growth in a created salt marsh. Ecological Applications 13:180–192.

R Core Team (2023) R: a language and environment for statistical computing. R Foundation for Statistical Computing. https://www.R-project.org/ (accessed 1 May 2024)

Raposa KB, Bradley M, Chaffee C, Ernst N, Ferguson W, Kutcher TE, et al. (2022) Laying it on thick: ecosystem effects of sediment placement on a microtidal Rhode Island salt marsh. Frontiers in Environmental Science 10:939870. https://doi.org/10.3389/fenvs.2022.939870

Raposa KB, Woolfolk A, Endris CA, Fountain MC, Moore G, Tyrrell M, et al. (2023) Evaluating thin-layer sediment placement as a tool for enhancing tidal marsh resilience: a coordinated experiment across eight US National Estuarine Research Reserves. Estuaries and Coasts 46:595–615. https://doi.org/10.1007/s12237-022-01161-y

Reilly E, Haaf L, Moody J (2026) Improved reporting needed for comprehensive analysis of thin layer placement of dredge materials in salt marshes: a review. Estuaries and Coasts 49:43. https://doi.org/10.1007/s12237-025-01658-2

Reimold RJ, Hardisky MA, Adams PC (1978) The effects of smothering a *Spartina alterniflora* salt marsh with dredged material. Technical Report D-78-38. United States Army Corp of Engineers, Washington, D.C.

Rinehart SA, Dybiec JM, Richardson P, Walker JB, Peabody JD, Cherry JA (2024) Researcher effects on the biological structure and edaphic conditions of field sites and implications for management. Ecosphere 15: e4750. https://doi.org/10.1002/ecs2.4750

Saintilan N, Kovalenko KE, Guntenspergen G, Rogers K, Lynch JC, Cahoon DR, et al. (2022) Constraints on the adjustment of tidal marshes to accelerating sea level rise. Science 377:523–527. https://doi.org/10.1126/science.abo7872

Scheffer M, Carpenter S, Foley JA, Folke C, Walker B (2001) Catastrophic shifts in ecosystems. Nature 413:591–596. https://doi.org/10.1038/35098000

Schrift AM, Mendelssohn IA, Materne MD (2008) Salt marsh restoration with sediment-slurry amendments following a drought-induced large-scale disturbance. Wetlands 28:1071–1085. https://doi.org/10.1672/07-78.1

Sherpa SF, Shirzaei M, Ojha C (2023) Disruptive role of vertical land motion in future assessments of climate change-driven sea-level rise and coastal flooding hazards in the Chesapeake Bay. Journal of Geophysical Research: Solid Earth 128:e2022JB025993. https://doi.org/10.1029/2022JB025993

Slocum MG, Mendelssohn IA, Kuhn NL (2005) Effects of sediment slurry enrichment on salt marsh rehabilitation: plant and soil responses over seven years. Estuaries 28:519–528. https://doi.org/10.1007/BF02696063

Stagg CL, Laurenzano C, Vervaeke WC, Krauss KW, McKee KL (2022) Presence of the herbaceous marsh species *Schoenoplectus americanus* enhances surface elevation gain in transitional coastal wetland communities exposed to elevated CO_2 and sediment deposition events. Plants 11: 1259. https://doi.org/10.3390/plants11091259

Stagg CL, Mendelssohn IA (2010) Restoring ecological function to a submerged salt marsh. Restoration Ecology 18:10–17. https://doi.org/10.1111/j.1526-100X.2010.00718.x

Stagg CL, Mendelssohn IA (2011) Controls on resilience and stability in a sediment-subsidized salt marsh. Ecological Applications 21:1731–1744. https://doi.org/10.1890/09-2128.1

Stagg CL, Osland MJ, Moon JA, Hall CT, Feher LC, Jones WR, Couvillion BR, Hartley SB, Vervaeke WC (2020) Quantifying hydrologic controls on local- and landscape-scale indicators of coastal wetland loss. Annals of Botany 125:365–376. https://doi.org/10.1093/aob/mcz144

Stoorvogel MM, van Belzen J, Temmerman S, Wiesebron LE, Fivash GS, van IJzerloo L, van de Koppel J, Bouma TJ (2024) Salt marshes for nature-based flood defense: sediment type, drainage, and vegetation drive the development of strong sediment beds. Ecological Engineering 207: 107335. https://doi.org/10.1016/j.ecoleng.2024.107335

Stumpf RP (1983) The process of sedimentation on the surface of a salt marsh. Estuarine, Coastal and Shelf Science 17:495–508. https://doi.org/10.1016/0272-7714(83)90002-1

Tempest JA, Möller I, Spencer T (2015) A review of plant-flow interactions on salt marshes: the importance of vegetation structure and plant mechanical characteristics. WIREs Water 2:669–681. https://doi.org/10.1002/wat2.1103

Turner MG (1988) Multiple disturbances in a *Spartina alterniflora* salt marsh: are they additive? Bulletin of the Torrey Botanical Club 115:196. https://doi.org/10.2307/2995955

Van Zomeren CM, Berkowitz JF, Piercy CD, White JR (2018) Restoring a degraded marsh using thin layer sediment placement: short term effects on soil physical and biogeochemical properties. Ecological Engineering 120:61–67. https://doi.org/10.1016/j.ecoleng.2018.05.012

Walker B, Meyers JA (2004) Thresholds in ecological and social–ecological systems: a developing database. Ecology and Society 9:3. https://doi.org/10.5751/ES-00664-090203

Wang Q, Li Y, Wang Y (2011) Optimizing the weight loss-on-ignition methodology to quantify organic and carbonate carbon of sediments from diverse sources. Environmental Monitoring and Assessment 174:241–257. https://doi.org/10.1007/s10661-010-1454-z

Wang C, Temmerman S (2013) Does biogeomorphic feedback lead to abrupt shifts between alternative landscape states?: an empirical study on intertidal flats and marshes. Journal of Geophysical Research: Earth Surface 118: 229–240. https://doi.org/10.1029/2012JF002474

Wijte AHBM, Gallagher JL (1991) The importance of dead and young live shoots of *Spartina alterniflora* (Poaceae) in a mid-latitude salt marsh for overwintering and recoverability of underground reserves. Botanical Gazette 152: 509–513. https://doi.org/10.1086/337913

Xu Z, Chang L (2017) Juncaceae. Pages 821–834. In: Xu Z, Chang L (eds) Identification and control of common weeds. Vol 3. Springer Singapore, Singapore. https://doi.org/10.1007/978-981-10-5403-7_29

Yando ES, Jones SF, James WR, Colombano DD, Montemayor DI, Nolte S, et al. (2023) An integrative salt marsh conceptual framework for global comparisons. Limnology and Oceanography Letters 8:830–849. https://doi.org/10.1002/lol2.10346

Supporting Information

The following information may be found in the online version of this article:

Table S1. *F* values, degrees of freedom, and *p* values for analysis of variance on mixed effect models using Satterthwaite's method in greenhouse and field data.
Table S2. Greenhouse mean values ± SE for each species and treatment for end of season, pre-season sediment, and weekly measurements.
Table S3. Field mean values ± SE for each treatment for end of season and weekly measurements.
Figure S1. Stem heights at end of season (A) and over time (B) for *Juncus* (left) and *Spartina* (right).
Figure S2. Stem diameter over time for *Juncus* (left) and *Spartina* (right) (A) and flowering stems per m^2 over time in *Spartina* (B).
Figure S3. Mean change in pin height (cm) in mini-SET for each treatment type.
Figure S4. Bulk density of soil by depth per treatment type for the field experiment.

Coordinating Editor: Siobhan Fennessy

Received: 29 October, 2025; First decision: 20 January, 2026; Revised: 3 April, 2026; Accepted: 27 May, 2026