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Effective use of thin layer sediment application in *Spartina alterniflora* marshes is guided by elevation-biomass relationship

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ABSTRACT

Thin layer application (TLA) of dredged sediments has emerged as an adaptive management option for marshes that are not increasing in elevation fast enough to keep pace with local relative sea level rise. While there have been multiple previous demonstrations of TLA, there is limited pre-and post-application monitoring data available from past projects to develop guidance concerning optimal siting, design and implementation of projects. To address this gap, we conducted a controlled TLA experiment in a *Spartina alterniflora* marsh adjacent to the Atlantic Intracoastal Waterway in central North Carolina. The project was informed by the results of a long-term marsh monitoring and research program which had identified the project area as particularly vulnerable to sea level rise. The experimental design included triplicate meso-scale ($24m^2$) treatment and control cells and quarterly (during the first year) to annual monitoring efforts to document trends in marsh surface elevation and vegetative biomass. In addition, sampling was conducted to characterize porewater chemistry and sediment characteristics. Thin layer application of dredged sediments resulted in an average elevation gain of 6 cm in treatment plots. Monitoring results indicate that the gains in elevation achieved by TLA have been maintained over five growing seasons. The response of vegetative biomass to the added elevation was well-predicted by the elevation-biomass relationship of the surrounding marsh platform. Treatment cells exhibited continued increases in surface elevation after placement, while control cell elevation remained constant. These data illustrate the potential of TLA as a tool for increasing the resilience of low-lying marshes to sea level rise and illustrate the value of the surface elevation-marsh biomass relationship for providing guidance for its use in *S. alterniflora*-dominated marshes.

1. Introduction

Coastal salt marshes exist at the boundary between land and sea; their distribution is limited by inundation and erosion at lower elevations and vegetative competition at the landward boundary (Mendelssohn et al., 1981; Pennings and Moore, 2001). As a result, salt marshes are particularly vulnerable to impacts of sea level rise (SLR). Marshes have persisted for millennia, despite rising sea levels, by building elevation in place, migrating landward, or a combination of the two. Landward migration occurs most readily in coastal areas with gently sloping topography where there are no manmade barriers to migration like roads or other built infrastructure (Kirwan et al., 2016; Fagherazzi et al., 2019). Increases in marsh surface elevation are achieved through trapping of sediments that are carried into the marsh by flooding tidal waters (Li and Yang, 2009; Mudd et al., 2010; Moskalski and Sommerfield, 2012) and by production of roots and rhizomes which contribute to soil volume (Cherry et al., 2009; Blum et al., 2020). In cases where marshes are not able to migrate inland and sea level rise outpaces the rate of surface elevation increase (Cahoon and Guntenspergen, 2010), marshes will convert to unvegetated mudflats and eventually, to open water if no mitigative action is taken.

Sea levels are increasing in most parts of the US and are predicted to continue to do so (Sweet et al., 2017); thus many marshes are experiencing rising waters. At the same time, shoreline hardening (Ganju, 2019) and river damming and diversions (Blum and Roberts, 2009; Weston, 2014) are negatively influencing sediment supply and, consequently, the ability of marshes to build elevation. The situation is further exacerbated by dredging which, though necessary for maintenance of navigable waterways, represents a net loss of sediment to the local estuarine system as dredged sediments are often deposited in upland containment facilities or in open ocean disposal areas (Zentar et al., 2009). The plight of coastal marshes, combined with the increasing

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challenge and cost of managing dredged sediments has fueled interest in the beneficial use of dredged sediments to increase surface elevation in marshes that are highly vulnerable to sea level rise (Ray, 2007).

A given marsh's resilience to sea level rise is predicted by its current elevation capital (the relative elevation of the marsh platform within the intertidal zone; Cahoon and Guntenspergen, 2010). Coastal marshes generally occupy elevations between mean sea level and mean high water (McKee and Patrick, 1988) with optimum growth occurring somewhere near the middle of the range (Morris et al., 2002; Morris et al., 2013). All other factors being equal, marshes that are lower in the intertidal zone (low elevation capital) are in more imminent danger of drowning and converting to open water than those that are higher. Marshes that are below the optimum elevation for growth tend to have lower biomass and consequently, are less efficient at building elevation and more vulnerable to future SLR; thus, position within the tidal frame is a reasonably robust determinant of marsh vulnerability (Wasson et al., 2019).

Application of clean dredged sediments provides a mechanism for increasing elevation capital and potentially increasing the amount of SLR that a marsh can withstand. Perhaps even more importantly, increasing position within the tidal frame to an elevation that is optimal for plant growth may increase a marsh's capacity to build elevation, rendering it more resilient to future sea level rise. There have been several previous demonstrations of this method that range in scale from small plots to tens of acres (Wigand et al., 2016; McAtee et al., 2020). While results of these previous efforts indicate that sediment addition can be a viable management strategy in some settings, there is currently no broadly applicable guidance for its use. Uncertainty about which sites are likely to benefit from sediment addition, optimal project design and siting criteria, and implications for future marsh function are among the obstacles to widespread use of thin layer application as a management strategy.

To address these information gaps, we conducted an experimental TLA project in a microtidal *Spartina alterniflora* marsh that did not build elevation at a rate commensurate with SLR over the past decade. The experimental design involved replicated, meso-scale treatment and control plots. The elevation of treatment plots was increased 6 cm, on average, with dredged sediments. Over the course of five years, we tracked changes in vegetative abundance and morphology, sediment surface elevation, and inundation to investigate whether sediment-treated plots performed differently than controls and to test the hypothesis that addition of elevation through application of dredged sediments would lead to increased vegetative biomass.

2. Methods

2.1. Study site

The Freeman Creek marsh complex is adjacent to the Atlantic Intracoastal Waterway (AIWW) as it traverses Marine Corps Base Camp Lejeune (MCBCL) on the central coast of North Carolina (Fig. 1). The *S. alterniflora*-dominated marsh platform on the mainland side of the AIWW lies low in the tidal frame (Cunningham et al., 2013). Previous efforts to document the relationship between *S. alterniflora* biomass and elevation within this system showed that biomass exhibits a parabolic distribution across the marsh elevation gradient with the greatest biomass occurring at an elevation of 0.02 m North American Vertical Datum of 1988 (NAVD88; Davis et al., 2017). LiDAR data provided by MCBCL indicate that large expanses of the Freeman Creek marsh system are below this optimal elevation (Fig. 1).

Surface elevation tables (SETs) installed in the Freeman Creek marsh at an average elevation of 0.02 m NAVD88 (Fig. 1) measured an average rate of elevation gain of 8.4 ($\pm$ 1.09) mm yr^{-1} ($n = 3$) from 2008 to 2016 (Cunningham et al., 2018). Comparison of water level trends measured at Freeman Creek with a pressure sensor (ONSET Computer Corporation, Bourne, MA) from June of 2016 through January of 2017 to that of the control station at Beaufort NC (National Water Level Observation Network [NWLON] #8656483) confirmed applicability of Beaufort water level data to the Freeman Creek system (Hilting et al., 2021). Linear regression analysis of de-trended monthly mean water level data at Beaufort over the same time period of the SET record demonstrates a local rate of relative sea level rise (RSLR) of 12.3 mm yr^{-1}(+/− 5.8 mm 95% confidence interval). Thus, mean sea level rise outpaced marsh elevation growth by roughly 4 mm annually over the 8 year period.

Suspended sediment supply, along with the rate of RSLR, is a key variable in determining the resiliency of a salt marsh system to SLR (Kirwan et al., 2010). Suspended sediment concentrations in the Freeman Creek system average 20 mg/L, and appear to be primarily of oceanic origin rather than from the local watershed (Ensign et al., 2017). Frequent shoaling in the area suggests that much of the material that comes in through nearby Brown's Inlet gets trapped in the AIWW rather than deposited on the marsh surface. The combination of low-lying

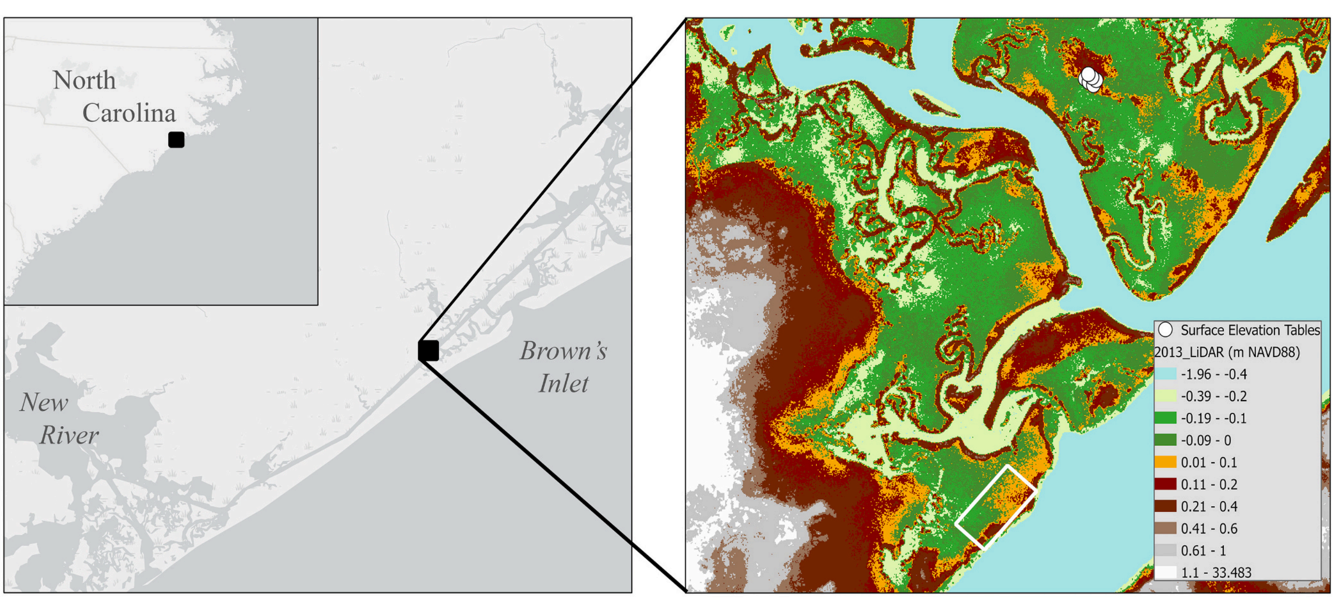

Fig. 1. Left – Project area within eastern North Carolina. Right – LiDAR map of project area within Freeman Creek marsh system, classified by elevation interval. White circles represent location of surface elevation tables (SETs), white rectangle indicates thin layer placement project area.

marsh that is not keeping pace with RSLR and a ready source of sediments that negatively impacts navigation along the AIWW made this an ideal place to test TLA as a means of increasing marsh elevation capital.

2.2. Experimental design

The experimental design included 3 treatment cells and 3 control cells (Fig. 2). Each of the six experimental cells were 8 m $\times$ 3 m (24 m^2), with 1 m between cells. The experimental cells were intentionally designed to be large enough to avoid "container effects" that might be expected with smaller applications (marsh organs, buckets and 1 m^2 wooden frames have been used in previous efforts) but still small enough to allow for experimental replication. Treatment cells were delineated with coir fiber logs, (12 in. diameter) placed directly on the existing marsh surface and held in place with wooden stakes. The experimental cells were established perpendicular to the shoreline and parallel to each other. A system of fiberglass and wooden boardwalks was installed around the experimental cells to minimize disturbance of the adjacent marsh surface during sediment application and monitoring. An elevation benchmark consisting of a threaded stainless steel rod driven into the marsh to the point of refusal (6.1 m) and encased in concrete was installed on site. The orthometric elevation of the top of the installed rod (0.182 m NAVD88 +/− 0.02) was determined through duplicate static GPS collections with survey grade Global Navigation Satellite System (GNSS) receivers (Trimble® R6–3) and post processing with the National Geodetic Survey (NGS) Online Positioning User Service Program (OPUS).

Long-term monitoring plots were established by dividing each cell into a grid of 24, 1 m^2 plots and using a random number generator to select five of the plots as fixed locations for repeated measures of elevation and vegetative characteristics. The same five relative locations were used in all six cells. Additional data (sediment cores, and porewater samples) were collected from the same relative position within each cell using the sampling grid as a guide (Fig. 3). Three additional monitoring plots were established in the undisturbed marsh just outside of the footprint of the treatment and control plots to serve as reference plots.

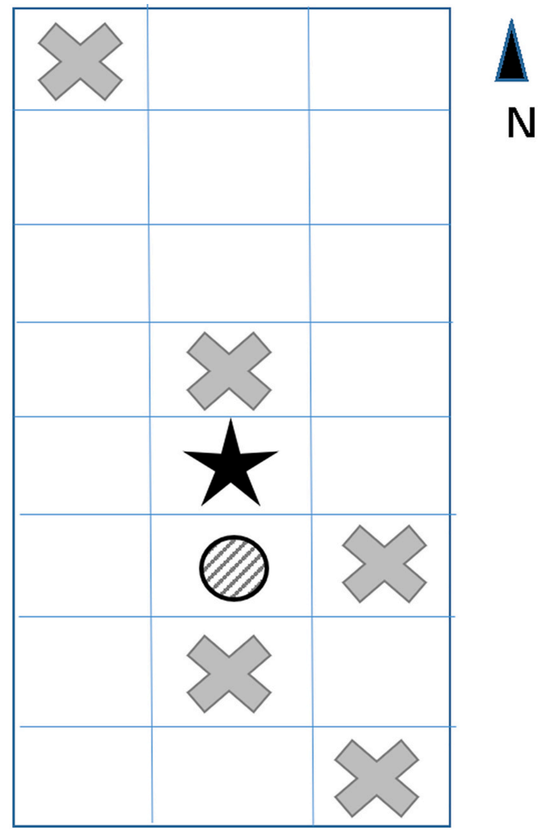

Fig. 3. Within cell sampling scheme. Each monitoring plot represents one square meter. Grey x's represent relative locations of vegetation and elevation monitoring plots within each treatment and control cell. Filled circle indicates location of pre-application sediment core collection and black star indicates location of porewater and post-sediment application core collection.

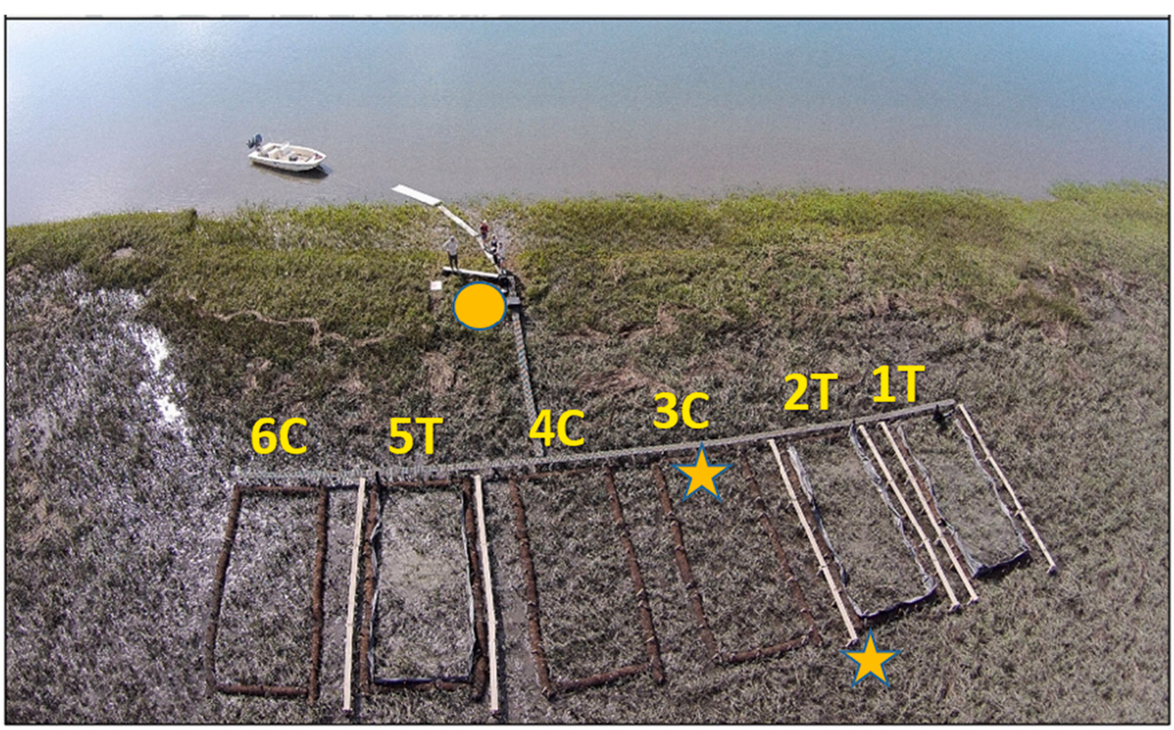

Fig. 2. Plots 1, 2 and 5 were treated with dredged sediment, plots 3, 4 and 6 were controls. Orange stars indicate water level logger locations and orange circle indicates position of the elevation benchmark. Image was taken half way through the sediment application.

2.3. Sediment application

Sediments were dredged from a nearshore shelf of the adjacent Atlantic Intracoastal Waterway (AIWW), approximately 5 m waterward of the lowest edge of marsh vegetation beginning April 10, 2017. Silt fencing (1 m tall) was installed inside the coir logs of all treatment plots before sediment placement began. Dredging was conducted with 5 hp. diaphragm pumps. Two pumps were run simultaneously from small boats anchored within the dredge area and the effluent was pumped roughly 30 m to the treatment cells. Dredging was restricted to times when the marsh was flooded so as to minimize scouring of the existing marsh surface and to help distribute the dredged sediments more uniformly. The discharge pipe was moved manually within the treatment cells at regular intervals during pumping to achieve uniform application depths. The sediment slurry concentrations achieved were on the order of 1–2% solids by volume. Pumping continued over the course of nine working days for approximately four hours per day until May 4, 2017. Total cumulative pump time (considering both pumps) for all cells was approximately 70 h.

2.4. Sediment characterization

Prior to sediment application, triplicate cores (7 cm diameter by 15 cm deep) were collected from the area to be dredged for grain size analysis. Duplicate cores (5.5 cm diameter by 10 cm deep) were collected from the approximate middle of each experimental cell (Fig. 3) for analysis of grain size, bulk density, and percent organic matter content (% OM) of the existing marsh sediments. In all cases, cores were collected by gently hammering clear polycarbonate tubes into the ground with a rubber mallet. There was no visual evidence of compaction during core collection. Grain size analysis was conducted by separating the entire contents of each core into Macro Organic Matter (MOM) (> 2 mm; visually identifiable plant biomass), sand (> 67 mm) and silt +clay (< 67 mm) fractions by rinsing the entire contents of each core over successive wet sieves. MOM was the only material present on the 2 mm screen and no gravel was detected in any of the samples. One core from each experimental cell was used for grain size analysis, and the remaining core was dried (60 °C for 72 h) and then weighed for determination of bulk density. The entire content of each dried core was then ground by hand with a mortar and pestle and its organic matter content was estimated from weight loss on ignition (450 °C for 4 h).

In 2019, a single core (7 cm diameter by 20 cm deep) was collected from the same relative position within each treatment and control cell (Fig. 3) and from the center of each natural marsh plot. Cores were extruded in 4 cm depth segments. While still wet, the full content of each segment was homogenized by gentle hand mixing with a rubber spatula. From each, approximately 50 mg (roughly a small handful) of material was removed for particle size analysis as described above. An additional 10 mg (purposefully avoiding any MOM) was dried (60 °C for 72 h), then ground to a fine powder with a mechanical ball mill. Each powdered sample was analyzed in triplicate for organic carbon content with a Costech® ECS 4010 Elemental Analyzer after acidification by dropwise addition of 1 N HCl to detect and remove carbonates. Acid addition did not result in bubbling (the visual evidence of carbonate conversion to CO_2) in any of the samples. All remaining material was rinsed over a 2 mm sieve for collection of MOM and the MOM from the 50 mg subsample was added to this fraction to provide a total accounting of MOM in each depth segment.

2.5. Vegetation and elevation

Vegetation and elevation data were collected from the five long-term monitoring plots within each cell. Data were collected prior to sediment application (March 2017), immediately after placement was completed (May 2017), and then again in July and November of 2017, March and July of 2018, and May and August of 2019 and July of 2021. Vegetation and elevation data were also collected in natural plots during peak biomass each year (July 2017 and 2018, August 2019).

Vegetative percent cover was categorized according to the Carolina Vegetation Survey (CVS) method (Peet et al., 1998) which classifies the percent cover of each species in a 1 m^2 plot according to ten pre-defined categories. In all sampling plots, the maximum green leaf height was documented for ten *S. alterniflora* stems randomly chosen as the first ten stems that intersected a string running across the centerline of the plot. Stem density was measured as the total number of *S. alterniflora* stems in one quarter of the quadrat (0.25 m^2). Average plant height (average of the ten measured stems) and stem density were used to estimate total standing *S. alterniflora* biomass based on previously identified allometric relationships (Davis et al., 2017).

Marsh surface elevations were measured relative to the local benchmark by leveling (optical or laser) to a position in the center of each fixed monitoring plot. Surface elevation data were collected in March 2017 before placement; then again in May, July and November of 2017; March, July and December of 2018; May and August of 2019; and July of 2021. All 2017 elevation data were collected with a self-leveling CST/Berger Lasermark© LM800 laser level (accuracy +/− 1.5 mm at 30 m). In July 2018, elevation data were collected with a Leica Geosystems Sprinter 250 M digital level (accuracy +/− 1.0 mm at 30 m,). All other elevation data were collected with a Dewalt® DW096 Auto Optical Level (26×, accuracy +/− 1.5 mm at 30 m). All levels were calibrated and collimation-checked according to the manufacturer's prescribed techniques. On all sample dates, the level was centrally located within the project area so that all measured points were < 20 m from the level.

2.6. Porewater collection and analysis

In 2019, porewater samples were collected from a depth profile in each cell with the sampling depths (4, 7, 10 and 16 cm) designed to correspond to the sediment core depth segments (described above). At each sediment coring location, before core collection, a push point sampler attached to a peristaltic pump with gas tight tubing was inserted to successively deeper depths and porewater was collected by pumping into a syringe. All samples were filtered immediately upon collection (0.45 μm Polyether Sulfone). Samples for ammonium (NH_4^+) and orthophosphate (PO_4^{3-}) were stored frozen until analysis by standard spectrophotometric methods (Parsons et al., 1984). Hydrogen sulfide (H_2S) samples were complexed with zinc acetate in the field and stored refrigerated until analysis by Cline's method (Cline, 1969).

2.7. Inundation

Two water level loggers (HOBO® U20; Onset Computer Corporation, Bourne, MA) were installed on site from June 25, 2018 through September 10, 2018 to record inundation time and depth within a control cell and within the natural marsh to determine whether the coir logs surrounding the experimental cells influenced inundation time by slowing drainage. One logger was installed in plot 3 C, the other in the natural marsh at the end of the boardwalk between plot 2 T and 3 C (Fig. 1). Water logger installation involved driving a two meter length of five cm diameter PVC pipe into the ground with a mallet. The pipe was vented to allow water to flow through freely and a large metal bolt was installed across the diameter of the pipe just above ground level. A smaller PVC pipe with the water level logger attached was lowered into the larger pipe such that the bottom of the smaller pipe rested on the metal bolt. Precise elevations of the loggers were established by leveling to the bolt on each vented pipe and accounting for the distance between the sensor and the bottom of the smaller pipe to which the sensors were attached.

2.8. Statistics

All statistical analyses were conducted in JMP (v 12, SAS Institute, Cary NC) at an alpha value of 0.05. Tukey's test was used for all post hoc comparisons.

3. Results

3.1. Elevation change

Mean starting elevation of the fixed monitoring plots, prior to sediment addition, ranged from −0.012 to −0.11 m NAVD88 and was not significantly different by cell or plot (ie. relative position within cell). Total elevation change as a result of dredged sediment deposition (measured as the difference in elevation at each of the 30 fixed monitoring plots between March and May of 2017) ranged from 2.0 to 12.6 cm among all treatment plots and averaged 6.3, 4.9 and 7.7 cm in cells 1 T, 2 T and 5 T respectively (Fig. 4). Control plot elevations did not change significantly between March and May of 2017. A two-way ANOVA of post placement plot elevation by cell and sample time indicated significant differences by cell only ($F_{53,216} = 9.72$, $p < 0.001$). Post hoc comparisons indicated significant differences by treatment with control cells consistently lower in elevation than treatment cells. Linear regression of mean cell elevation versus time indicated significant elevation trends in treatment cells (1 T slope = 7 mm yr^{-1}, $r^2 = 0.53$, $p < 0.05$; 2 T slope = 4.3 mm yr^{-1}, $r^2 = 0.47$, $p < 0.05$, and; 5 T slope = 5.8 mm yr^{-1}, $r^2 = 0.59$, $p < 0.01$); there were no significant trends between mean cell elevation and time in any of the control cells (Fig. 5).

3.2. Sediment characterization

Source sediments from the dredge area were > 90% sand by weight and were devoid of MOM. In contrast, average composition of the marsh sediments by weight was 54% sand, 41% silt and 5% MOM (Fig. 6). Pre-application bulk density of the existing marsh sediments averaged across all experimental cells was 0.79 ± 0.11 g cm^{-3}, bulk density of the source sediments was not determined. A two-way ANOVA demonstrated significant differences in post-application sand content by depth and treatment type with a significant interaction between the two ($F_{14,23} = 6.54$, $p = 0.001$). As of August 2019, the upper 12 cm (0–12 cm depth) in treatment plots exhibited greater percent sand by weight than those of natural and control plots (80–90% vs 40–60%, respectively; Fig. 7). At depths greater than 12 cm, sediment grain size did not differ significantly among treatment types. Sediment carbon density was more than 2× greater in surface sediments (< 8 cm) of natural and control plots than treatment plots but not significantly different among treatment types at depths greater than 12 cm ($F_{14,24} = 4.35$, $p = 0.0008$). Macro organic matter content varied by treatment type and depth with lower concentrations in treatments than controls ($F_{9,18} = 5.23$, $p < 0.0014$) and increasing concentrations with depth in both.

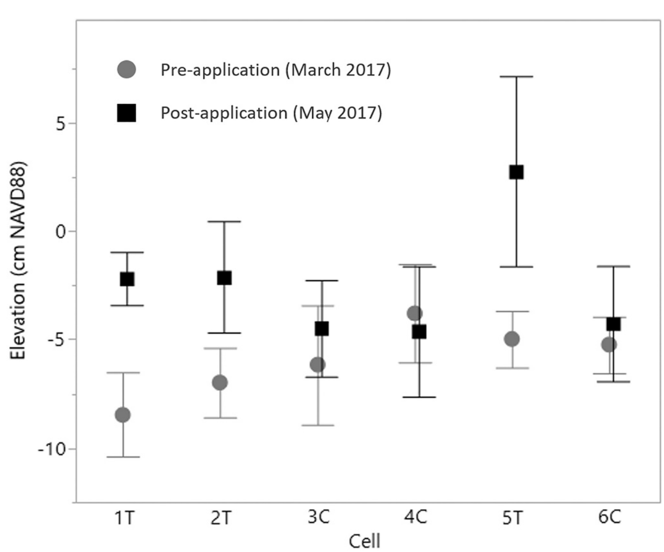

Fig. 4. Average pre- vs post-placement elevation. Data represent the average and standard deviation from the five fixed monitoring plots within each experimental cell. Mean elevation change as a result of sediment placement averaged 6.3, 4.9 and 7.7 cm in cells 1 T, 2 T and 5 T respectively. The maximum change measured at any individual monitoring plot was 12.6 cm.

3.3. Vegetation

Prior to sediment application there were no significant differences in stem density or stem height among treatment and control cells. Values of both parameters remained similar among treatment types until November of 2017 when both stem density ($F_{17,254} = 14.3$; $p < 0.0001$) and stem height ($F_{17,230} = 39.2$, $p < 0.0001$) in treatment cells exceeded those of controls. Greater stem density and height were subsequently detected in treatment sites compared to control sites for all sample periods from November 2017 until the end of the study (Table 1). Increases in both of these metrics translated into increases in standing live biomass which was greater in treatment cells ($F_{17,230} = 25.1$, $p < 0.0001$) from November 2017 through the remainder of the study (Table 1). The number and locations of natural plots were not consistent across all sample times. These data are shown in Table 1 for comparison, but were not included in the statistical analyses due to the uneven sample sizes.

3.4. Porewater constituents

Porewater orthophosphate concentrations ranged from below the limit of detection (0.3 μM) to 1.5 μM and did not vary significantly by depth or treatment type ($F_{11,17} = 2.4$, $p < 0.054$). Notably, phosphate concentrations were low in the shallowest depth segment (0–4 cm) of all treatment cells (Fig. 8). Porewater ammonium concentrations ranged from below detection (0.5 μM) to 24 μM and did not differ significantly by depth or treatment type ($F_{11,18} = 1.26$, $p = 0.32$; Fig. 8). Porewater hydrogen sulfide was highly variable, ranging from below detection in the majority of samples to values as high as 860 μM (data not shown). There were no detectable differences in hydrogen sulfide concentration by depth or treatment ($F_{11,18} = 1.4$, $p = 0.27$).

3.5. Inundation

Visual observations of water levels (WL) during falling tides confirmed that patterns of inundation within the experimental cells were not influenced by the presence of the coir logs surrounding each cell. Similarly, comparison of water levels between loggers in the natural and control plots showed strong agreement (Natural WL = −0.018 + 1.04*Control WL; $r^2 = 0.999$, $p < 0.001$). Inundation times calculated based on the mean elevation within each cell indicated that on average, treatment plots were inundated for 1 h less than control plots each day (30 min/tidal cycle).

4. Discussion

Thin layer application of dredged sediments has been proposed as a viable solution for marshes that are not able to build elevation at a pace that allows them to keep pace with sea level rise (Raposa et al., 2020). Surface elevation tables installed in the Freeman Creek marsh system measured elevation change rates on the order of 8 mm yr^{-1}. Thus, the 6 cm of average elevation gain achieved by application of dredged sediments was equivalent to approximately 7.5 years' worth of natural sediment accumulation. The artificially added elevation resulted in a significant increase in standing above ground biomass that was evident by the end of the first growing season and that persisted for four years post-placement and was accompanied by an increased capacity to build further elevation. These data demonstrate that TLA can be used successfully to build elevation capital, and increase standing live biomass

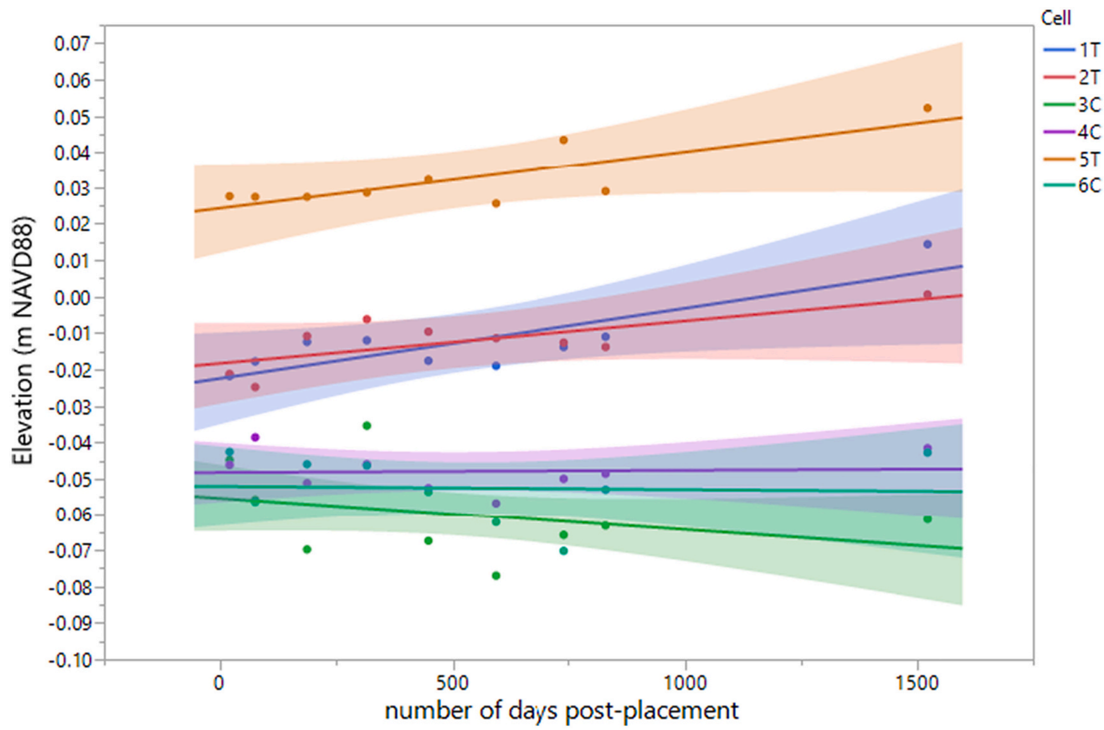

Fig. 5. Mean cell elevations over time. Post-placement elevation trends in treatment cells were significantly different from zero. Slopes of the regressions indicate elevation change rates of 7, 4.3 and 5.8 mm yr^{-1} in cells 1 T, 2 T, and 5 T respectively. Regression of mean elevation over time in control cells were not significantly different than zero.

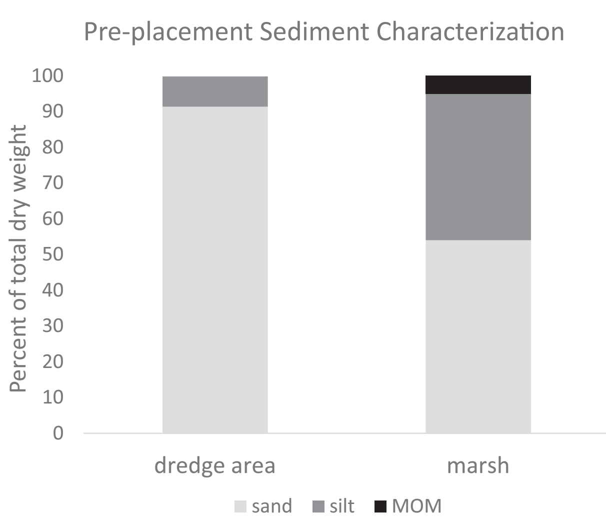

Fig. 6. Pre-placement sediment characterization. Dredge area = mean of 3 samples and represents the material that was deposited during the application process. Marsh samples represent average of 6 cores, 1 from each experimental cell.

and resilience to future SLR in *S. alterniflora* marshes with sandy sediments. Widespread acceptance and use of thin layer application as a management strategy will require advances in application technology and further investigations of its use with a wide range of vegetation and sediment types.

4.1. Application methods and sediment characteristics

Thin layer application methods play a significant role in the consistency of application depths across a project site, and in vegetative community recovery time. To date, one of the more commonly used thin layer application methods is high pressure spraying in which the sediments are applied by spraying aerially over the marsh surface (Ray, 2007). The rain of sediments produced by this approach can land with high enough impact to knock existing vegetation over and partially bury it, even with total application depths on the order of a few centimeters. The magnitude and longevity of impacts to the existing marsh vegetation will increase with application depth. Plants knocked over by spray application have been shown to recover fully from applications depths of 2–3 cm (Ford et al., 1999) while applications of 10–15 cm can result in permanent smothering of existing plants (Cahoon and Cowan Jr., 1988; McAtee et al., 2020). VanZomeren et al. (2018) documented significant revegetation within 6 months of application, and the ability of *S. alterniflora* to grow upward through the applied sediments (5–20 cm) in Avalon, New Jersey where application depths ranged from 5 to 20 cm. When there is a need for rapid establishment of vegetation (eg. to stabilize the placed sediments against erosive losses) it may be necessary to plant or seed the area to speed recovery.

The alternative to high pressure spraying is low pressure hydraulic dredging in which the sediments are pumped through a pipe that extends from the dredge vessel to the placement location. In this method, the sediment slurry is deposited onto the marsh surface at a discreet location and distributed by passive sheet flow across the marsh surface. While the impact of this slurry can cause scouring of the marsh surface and burial of plants near the discharge site, it does not tend to flatten vegetation that is not in the direct vicinity of the discharge. The downside is that this approach can result in sorting of the deposited sediments such that coarser materials drop out of suspension and form mounds near the discharge end of the pipe, while smaller particles are carried farther away (Whitbeck et al., 2019). Mounding of sediments can be minimized by moving the pipe at regular intervals but the time required to move the pipe and costs associated with lost hours of dredging can be prohibitive. The cost of TLA approaches varies widely among projects as a function of mobilization costs (higher for more remote sites), distance over which the material is transported, and

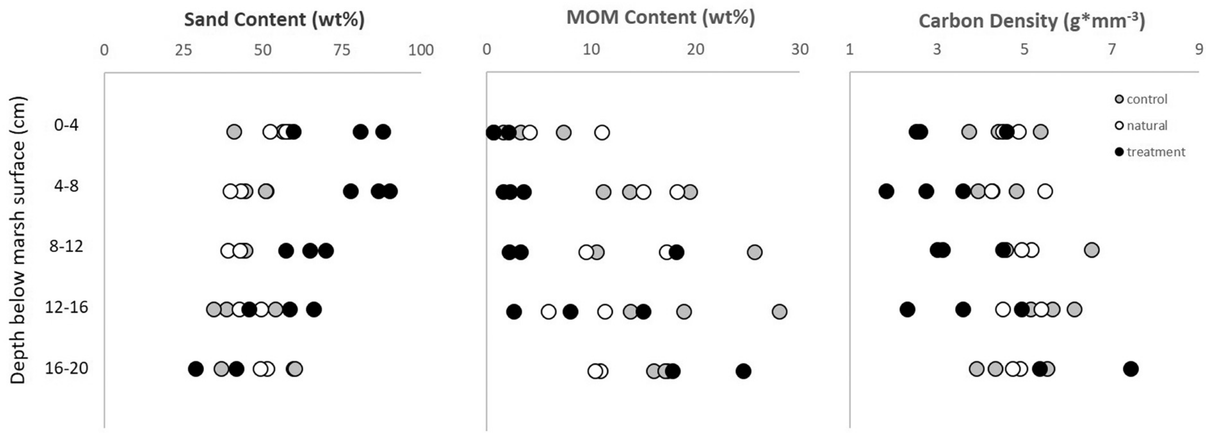

Fig. 7. Sediment depth profiles two years after thin layer placement.

Table 1
Mean values of vegetative parameters from repeated samplings. Report values are mean and standard error from all treatment and control plots ($n = 15$) and natural plots ($n = 3$). Asterisks indicate statistically significant differences between treatment and control plots at the indicated time point.

	Stem Density (m^{-2})			Stem Height (cm)			Biomass (g*m^{-2})		
Sample Time	treatment	control	natural	treatment	control	natural	treatment	control	natural
March 2017	61 (5.3)	70 (6.3)		40 (2.6)	37 (3.0)		52 (22)	58 (23)	
May 2017	58 (4.5)	48 (3.9)		60 (2.6)	61 (1.9)		141 (17)	121 (15)	
July 2017	77 (5.4)	69 (6.1)	59 (11.5)	57 (2.6)	55 (1.5)	65 (6.5)	270 (26)	210 (27)	339 (86)
November 2017	101 (11.3)	65 (9.3)*		49 (3.1)	33 (2.2)*		256 (48)	75 (12)*	
March 2018	141 (9.9)	82 (8.6)*		31 (1.3)	23 (1.0)		94 (8)	42 (5)	
July 2018	156 (13.6)	98 (17.8)*	99 (21)	66 (3.5)	56 (2.9)	82 (6.3)	702 (91)	319 (57)*	791(217)
May 2019	126 (5.0)	74 (6.3)*		73 (1.9)*	59 (2.0)*		474 (34)	165 (22)*	
August 2019	132 (10.5)	78 (8.9)*	56 (24)	67 (3.5)	59 (3.6)	58 (3.9)	670 (49)	291 (35)*	216 (59)
July 2021	6.2 (4.4)	50 (3.8)		81 (1.7)	75 (2.7)		320 (29)	233 (22)	

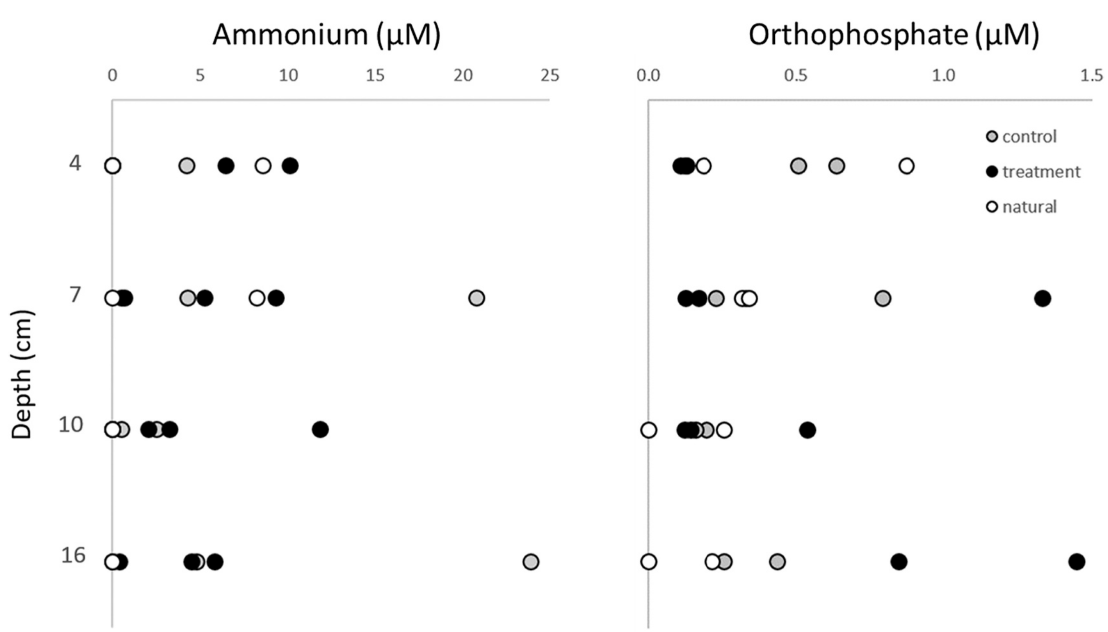

Fig. 8. Porewater ammonium and phosphate concentrations by depth and treatment.

extent to which the material must be graded to reach desired elevations after application, among other factors.

Sediment application in the current study at Freeman Creek involved a modified version of the low pressure pumping approach. The combination of small treatment cells, and small pipe diameter (15 cm) allowed the flexibility to move the pipe manually without significant downtime. The discharge end of the pipe was suspended approximately 15 cm above the marsh surface thus, the slurry was delivered within the marsh canopy rather than above it. The 'low and slow' application approach used here resulted in relatively consistent application depths without any noticeable impacts to the existing vegetation, but this approach is challenging to replicate on a large scale. Advances in placement technology that allow for deposition of relatively uniform amounts of sediment across a large area will be required for thin layer application to become operational at a scale that is meaningful for managing large areas of marsh.

Sediment grain size and OM content of dredged sediments and the marsh platform to which they are applied can influence the longevity of initial elevation gains. When sediments are added to highly organic marshes, the combined impacts of compression of the underlying marsh and consolidation of the applied sediment layer can lead to a loss of elevation over time (VanZomeren and Piercy, 2020). The impacts of compression tend to increase with the depth of applied sediments. As a result of compression, thin-layer elevation gains of up to 23 cm were largely erased within 2.5 years in a brackish marsh in Louisiana (Graham and Mendelssohn, 2013). In that study, the native marsh platform was characterized by low bulk density sediments (0.09 g*cm^{-3}) and sediment addition led to substantial increases in bulk density of surface sediments (0.23 to 0.44 g*cm^{-3}). Sediments with high sand content have higher bulk density and are less susceptible to compression (Mudd et al., 2009) than those with lower bulk density. In the Freeman Creek system, high sand content in both the native marsh sediments (30–50% by weight) and the applied sediments (80% by weight) minimized compaction or compression, and resulted in stable elevations over time. Characterizations of the existing marsh sediments and dredged sediments are necessary precursors to any thin layer project.

In the current study, the clear pattern of sand enrichment in surface sediments of treatment cells relative to those of controls allowed for distinction between the applied sediment layer and the native marsh beneath it (Fig. 7). Elevated sand content in treatment plots was apparent to depths of 12 cm, despite an overall increase in elevation of less than 7 cm, indicating mixing of applied sediments with native marsh sediments. It is unclear whether this mixing occurred during the deposition process or afterward due to bioturbation, or a combination of both. While we did not sample for benthic infauna, we did note the presence of a large number of fiddler crab (*Uca* spp.) burrows in treatment plots by the summer of 2018, one year after placement. Crab burrowing activities will likely result in further mixing between applied and native sediment layers over time (Natálio et al., 2017). Treatment plot sediments were depleted in MOM in the upper 12 cm compared to control plots. In control plots, the MOM pool included both living root and rhizome material and detrital material in varying stages of decomposition. In treatment plots, the MOM fraction in the upper 12 cm was dominated by live root and rhizome material as there had not been time for accumulation of a significant detrital fraction.

4.2. Vegetative response

The goal of adding dredged sediments to the Freeman Creek marsh was to raise its relative position within the tidal frame to an optimal elevation for vegetative growth. We previously demonstrated that *S. alterniflora* biomass in this system exhibits a parabolic distribution with respect to elevation (Davis et al., 2017). The peak of the parabola, which represents the elevation of maximum biomass (ie. optimal growth), occurs at approximately 0.1 m NAVD88. Much of the existing marsh platform in the project area falls below this "ideal" elevation for growth and spends approximately 50% of the time inundated (Davis et al., 2017). The low relative elevation of this marsh was further evidenced by the existence of small clumps of live oysters in the control plots. The 6 cm average elevation gain achieved in this experiment brought the surface elevation of the treatment plots closer to the optimal growth elevation and reduced inundation by 1 h per day on average. The result was a doubling of standing live *S. alterniflora* biomass which was accurately predicted by the previously determined biomass elevation relationship (Fig. 9). The *S. alterniflora* elevation biomass relationship provided a clear justification for the use of thin layer application and identified a target elevation to be achieved. In cases where biomass varies across the marsh elevation gradient, this relationship is invaluable for project design.

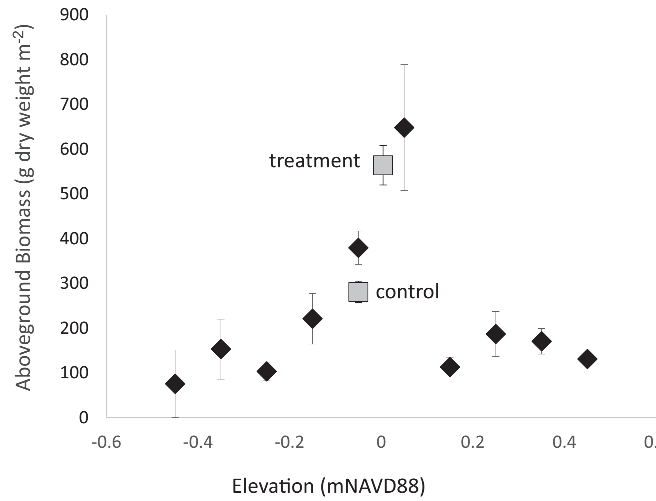

Fig. 9. Elevation-Biomass relationship. Black diamonds indicate previously determined relationship between elevation and biomass in Freeman Creek marsh system (Davis et al., 2017). Grey squares represent mean of all biomass and elevation measures in treatment ($n = 45$) and control ($n = 45$) plots during summer/peak biomass sampling events in 2018, 2019 and 2021.

Several previous works have documented increased vegetative growth in response to sediment addition (Ford et al., 1999; Croft et al., 2006; Schrift et al., 2008; Stagg and Mendelssohn, 2010; Graham and Mendelssohn, 2013). The magnitude and duration of these responses varied by site, application depth, and extent to which the elevation gain persisted over time. In contrast, Wilber (1992) noted decreased vegetative biomass in treated marshes relative to adjacent controls sites 10 years after thin layer application in Gull Rock, NC. The marshes in the study by Wilbur were oligohaline and dominated by *Juncus romerianus*. Decreases in biomass at treated sites were driven by losses in *J. romerianus*; total standing biomass of *S. alterniflora* and *Distichlis spicata* was similar in control and treatment plots. LaSalle (1992) noted a similar outcome at the Dog Lake placement site in Louisiana where 6 years after addition of 10–20 cm of sediment, *Juncus* sp. was abundant in reference plots but not in the treatment area. These findings suggest that the response of vegetative biomass to thin layer application is not consistent among species and there is still much to learn about how vegetative communities other than those dominated by *S. alterniflora* or *Juncus* sp. will respond.

The magnitude of vegetative response to sediment addition has frequently been evaluated in reference to depth of added sediments (Mendelssohn and Kuhn, 2003; Croft et al., 2006). This complicates among-site comparisons as the precise amount of elevation required to raise a marsh to optimal growth elevations varies as a function of tidal range and the starting marsh elevation (Cahoon et al., 2018; VanZomeren and Piercy, 2020). We suggest that in tidal *S. alterniflora* dominated marshes, understanding site-specific tidal range and the local marsh biomass-elevation relationship, as was done for this study, is essential to generating consistent and predictable vegetative responses to sediment application.

Total standing biomass is directly related to the efficiency with which marshes are able to build elevation through belowground biomass accumulation (Slocum et al., 2005) and sediment trapping (Morris et al., 2002). Czapla et al. (2020a) previously demonstrated the importance of vegetative biomass to sediment trapping in the Freeman Creek system with a nutrient fertilization experiment where fertilization resulted in a 3× increase in *S. alterniflora* biomass and a concomitant 2× increase in sediment deposition measured over feldspar marker horizons (Czapla et al., 2020a; Czapla et al., 2020b). Cahoon et al. (2018) provide a long-term demonstration of this same phenomenon in Jamaica Bay, NY. At the NY site, thin layer application via high pressure spraying was used to add 45 cm of elevation to a fragmented *S. alterniflora* marsh. After 14 years, the treatment site exhibited 25% higher plant cover and increased elevation gain relative to a nearby untreated control. These previous results demonstrate the relationship between plant biomass and the capacity to build elevation and consequently, the value of adaptive management actions like thin layer application that can lead to increased biomass. At Freeman Creek, we observed a post-placement increase in marsh surface elevation on the order of 6 mm yr^{-1} in treatment plots, while control plots failed to build elevation over the same time period. The long-term elevation trend detected by the nearby SETs was slightly greater than that of the treatment plots (8.4 mm yr^{-1} at SETs). The mean marsh surface elevation at the SETs is 0.02 m NAVD88, while the mean post-application surface elevation of treatment plots was −0.01 m NAVD88 and surface elevation of control plots averaged −0.045 m NAVD88. Thus both the standing biomass and elevation change rates vary demonstrably with elevation. We note that the method used to measure post-placement elevation change (leveling) is not capable of detecting the mm-scale change that SET's can detect. Thus, the control plots may be building elevation at a rate that is low enough to preclude detection. Still, the distinct difference between treatment and control elevation trends indicates that sediment application resulted in an increased capacity of treatment plots to build further elevation. This supports the positive relationship between increased plant biomass and marsh surface elevation change, and points to the inability of low-lying *S. alterniflora* marshes to keep up with sea level rise.

4.3. Porewater chemistry

Post-placement increases in plant biomass may be partly fueled by nutrients that are inadvertently deposited with the dredged sediments. Delaune et al. (1990) measured nutrient content of *S. alterniflora* in sediment-treated marshes and found significantly greater tissue phosphorous, iron and manganese content in plants from plots treated with 4–10 cm of dredged sediments. In that study, concentrations of soil extractable nutrients were greater in plots with greater amounts of added dredged material. Other investigators have found significantly lower concentrations of extractible and potentially mineralizable nutrients in pore waters and sediments of applied sediment layers relative to underlying native marsh sediments (Mendelssohn and Kuhn, 2003; Stagg and Mendelssohn, 2010; VanZomeren et al., 2018). The likelihood of a fertilization effect is dependent on the character of the applied sediments. In cases where sediment application results in increased nutrient availability, it may be challenging to quantify the relative contributions of increased elevation versus nutrient fertilization to increases in marsh biomass. The distinction is important because if the biomass response is primarily driven by fertilization, it will likely only last until the supply of excess nutrient is exhausted (Slocum et al., 2005). Growth of *S. alterniflora* in the Freeman Creek system has been shown to be nutrient (N + P) limited (Davis et al., 2017; Czapla et al., 2020b). In the current study, there were no significant differences in porewater concentrations of ammonium or phosphate between control and sediment treated plots. Further, the high sand content and low clay and organic matter contents of the applied sediments suggests that they were not likely a significant source of bioavailable nutrients.

Coarse sandy sediments are characterized by increased drainage and aeration relative to fine sediments with high organic content (Tamborski et al., 2017). Wetland soils with poor drainage tend to experience greater build-up of metabolites like hydrogen sulfide, which has been shown to be toxic to *S. alterniflora* at high concentrations (Lamers et al., 2013) and to limit plant growth by interfering with nitrogen uptake (Bradley and Morris, 1990). One of the purported potential benefits of thin layer application is amelioration of sulfide stress due to shorter inundation times and enhanced drainage associated with applied sediment layers (VanZomeren and Piercy, 2020). Previous characterizations of porewater in the Freeman Creek system have found hydrogen sulfide concentrations to exhibit a great deal of spatial variability but to be low relative to other nearby marshes (Czapla et al., 2020b). In the current study, porewater sulfide concentrations were similarly variable (0–800 uM) with 15 of the 28 total samples having no detectable hydrogen sulfide; those 15 samples were distributed evenly among treatment, natural and control plots and sample depths. The lack of significant difference in sulfide concentrations between treatment and controls suggest that reduced sulfide stress was not the driver of elevated biomass in treatment plots. Similarly, low porewater concentrations of bioavailable N and P in treatment plots cast doubt on the likelihood of a fertilization response. We hypothesize that the measured increases in biomass in sediment treated cells are the result of greater soil aeration and its direct impacts on root metabolism (Mendelssohn and McKee, 1988). We predict that a biomass differential between treatment and control plots at Freeman Creek will be maintained for as long as the differences in sediment character and elevation persist.

4.4. Impact of thin layer application on marsh carbon burial

Wetlands are highly efficient at carbon sequestration (aka blue carbon burial) due to: 1) anoxic conditions in their sediments which serve to slow decomposition rates and consequently the remineralization of buried organic carbon, and; 2) the capacity of these sediments to increase in absolute volume over time in response to increasing sea levels (McTigue et al., 2019). The addition of sediment volume via thin layer application mimics the natural process by which marshes grow in elevation and creates additional “space” for carbon accumulation. While ingrowth of belowground biomass represents an immediate contribution to soil organic matter content in newly placed sediments, it is the slow and incomplete turnover of this biomass that ultimately contributes to long term carbon storage in marsh sediments. If we assume that carbon density in the applied sediment layers will eventually reach values comparable to those of the surrounding native marsh, the maximum blue carbon potential of the placed sediments can be estimated from their total volume and the measured average soil carbon density (SCD) in the surrounding marsh. In the Freeman Creek system, the additional 6 cm of elevation over 81 m^2 (the cumulative area of the three treatment plots) and SCD averaged over control and natural plots (4.9 Kg/m^3) equates to a blue carbon potential of 23 Kg carbon. This estimated value will take decades to be realized (Craft et al., 2003; Davis et al., 2015), and is reliant on the habitat persisting as an *S. alterniflora* marsh. It is worth noting that carbon density in the Freeman Creek marsh system is well below the average for tidal wetlands (27 Kg/m^3; Holmquist et al., 2017) likely due to the minerogenic nature of this marsh. The blue carbon potential of thin layer applications in other systems will be proportional to the average carbon content of the surrounding marsh. While this back of the envelope calculation provides a first order estimate of the value of thin layer application to a site's future carbon burial potential, the more significant impact of thin layer addition on marsh blue carbon storage will likely be a function of preservation of the existing marsh and the carbon that is already buried in its sediments (McTigue et al., 2019; Pendleton et al., 2012).

4.5. Planning for future sea level rise

A common challenge among TLA projects is how to best design for future SLR. One approach is to increase elevations to levels that will be optimal for marsh growth in future decades as sea level increases. Another option involves designing for current conditions (ie. raising the marsh surface to the present optimal elevation for growth) recognizing that repeat application may be necessary to maintain the marsh at a fixed position within the tidal frame. The former approach is more cost effective in terms of sediment movement. However, this approach results in the short-term conversion of tidal marsh to supratidal habitat which will likely increase the challenge of getting a project permitted due to Clean Water Act (33 U.S·C. 1344) restrictions on filling wetlands. The latter approach (repeated application of dredged sediments), may be feasible for marshes that are proximal to frequently dredged channels.

A strategy that combines both approaches, raising part of the marsh to or just above the upper threshold of marsh growth while raising the rest to optimal, or just above optimal growth elevations, would ensure current and future provision of marsh habitat. This approach is more feasible for projects with large footprints and those in systems with a large tidal range where the vegetative communities are distributed over a wider range of elevations and thus there is more "wiggle room" in the final target elevations. Ultimately, decisions about how to best consider sea level rise in TLA project design will require discussions among project planners and regulators about the extent to which habitat tradeoffs (conversion of one habitat type to another) are acceptable.

5. Future directions

Beneficial use of dredged sediments to elevate marshes that are not keeping pace with sea level rise can increase marsh resilience while simultaneously providing a placement option for dredged sediments. In many cases, these sediments would otherwise be placed in upland containment facilities or in offshore disposal areas; both of these alternatives result in a net loss of sediments from the estuarine system where they are critical to supporting vegetated intertidal habitats. Mounting experimental evidence suggests that thin layer application can lead to enhanced vegetative growth and increased long-term resilience to sea level rise in *S. alterniflora* dominated marshes. This study demonstrated that the *S. alterniflora* growth response to increased surface elevation achieved with sediment application matched that predicted by the elevation-biomass relationship found in nearby natural marshes. There remains a need for advances in application technology to provide for the addition of thin (several cm) layers of sediment over large spatial scales. There is also still much to learn about how other wetland species (eg. *Juncus romerianus, Spartina patens*) will respond to sediment application before thin layer application can be used in these plant communities with confidence. The data presented here indicate that with appropriate control of application depths, TLA can be used with predictable results in *S. alterniflora* dominated marshes.

CRediT authorship contribution statement

Jenny Davis: Funding acquisition, Project administration, Conceptualization, Methodology, Writing – original draft, Formal analysis, Visualization. **Carolyn Currin:** Funding acquisition, Project administration, Conceptualization, Methodology, Writing – review & editing. **Natalia Mushegian:** Methodology, Formal analysis, Visualization, Writing – review & editing.

Declaration of Competing Interest

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

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