

# “Monitoring and Ecosystem Functioning of the Gansevoort Peninsula Salt Marsh”

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## Abstract

Living shorelines are constructed to provide important benefits to coastal communities including creating habitat, mitigating wave energy, and removing excess nutrients. The timeframe in which these benefits are realized, however, is not documented as well as the environmental drivers that impact these processes. In this study, we measured *S. alterniflora* growth, hydrological conditions, sediment nutrient fluxes, sediment oxygen demand, and denitrification within a newly constructed living shoreline, the Gansevoort Peninsula salt marsh in Hudson River Park (New York City, NY). We found strong growth of marsh plants within the first year with plants closest to the landward side of the marsh measuring significantly shorter. Measurements of erosion dynamics and sediment deposition suggest these plants' growth may be stunted by wave energy. We hypothesized that the size of the plants would influence sediment nutrient and gas fluxes but did not observe differences due to plant size. The living shoreline, despite being less than one year old, was a net nitrogen sink with a mean denitrification rate of  $121 \mu\text{mol N m}^{-2} \text{h}^{-1}$  and net uptake of dissolved inorganic nitrogen. The results suggest that newly constructed living shorelines in urban eutrophic ecosystems can provide important ecosystem services, specifically nutrient mitigation, within the first year.

## Introduction

Coastal areas have long been attractive locations for the establishment of cities, given the resources available and the ease of access via local waterways. As a means of making these zones habitable, humans have historically eliminated salt marsh ecosystems through draining and filling areas with landfill, which has resulted in a loss of ~50% of global marshes over the last century (Kirwan & Megonigal, 2013). Coastal regions are vulnerable to shoreline erosion (Toimil et al., 2020) and must contend with climate-change related sea level rise (White et al. 2005), challenges that are exacerbated by the removal of wetlands. Prior to the 21<sup>st</sup> century, most coastal resilience efforts took the form of shoreline armoring (*i.e.*, bulkheads, rip rap, etc.), but such forms of protection do not serve as permanent solutions in the face of rising sea levels and persistent wave energy (Gittman et al., 2015) and have also been documented to alter surrounding ecological communities (Gittman et al., 2016). Wetlands are unique in that they offer similar protection to shoreline armoring (Smee, 2019) while also providing ecosystem

services such as biodiversity support, carbon sequestration, and the removal of pollutants (Zedler & Kercher, 2005).

In recent years, cities have opted to implement more nature-based solutions, such as the installation of living shorelines. Living shoreline projects incorporate “hard” infrastructure (*i.e.*, groins, rock sill, etc.) in conjunction with “soft” features (*i.e.*, marsh habitat, oyster reefs, etc.; Burke et al., 2005; Currin et al., 2010). Living shorelines can adapt to rising sea levels through sediment accretion, serve as critical habitat for young fish and migratory birds, and aid in nutrient cycling (Guthrie et al., 2022; Davis et al., 2015; Young et al., 2021). The creation of successful living shorelines tends to be a complex process, especially given knowledge gaps that exist concerning effective design and how environmental variability impacts each site (Morris et al., 2022). As such, beyond implementing nature-based solutions, there is a need for continued evaluation to identify key environmental conditions, potential hurdles and how the constructed environment is performing.

Wetlands are threatened by several anthropogenic factors (*e.g.*, nutrient loading, metal pollutants, etc...; Gilby et al., 2021; Feher et al., 2017; Ostrowski et al., 2021), resulting in heavy degradation that can lead to ecological collapse and a subsequent loss of associated ecosystem services. As a result, wetland restoration projects have been conducted to improve the structure and function of deteriorating ecosystems (Spieles, 2022). Salt marsh restoration is critical for returning ecological functions, with restored marshes seeing improvements in vegetation development, soil accretion, biodiversity conservation, and biogeochemical management (Alikhani et al., 2021). In urbanized areas, restored marshes experience significant stressors that may differ based on local conditions (Wigand et al., 2014), therefore, it is important to identify the challenges faced in individual restoration scenarios. Restoration design can play a significant

role in the success of wetland structure and function (Nadeau & Conway, 2015; Liu et al., 2021). As such, design plans for prospective restoration often employ nearby reference sites (both intact and restored) to determine optimal treatment methods.

Despite the potential for success, constructed and restored wetlands are understood to experience a delay in the recovery of structural and functional capacities compared to their natural counterparts. Past research has posited that, even in sites that have been restored for decades, biological structure and biogeochemical functioning tend to reach on average 75% equivalency to intact, natural sites with a tendency to plateau 20-30 years after restoration efforts (Moreno-Mateos et al., 2012). The slow recovery of organic matter (OM) content is consistent across restored and constructed wetlands (Craft et al., 2003; Tatariw et al., 2021) and is estimated to take ~90 years for carbon stocks to fully recover (Craft et al., 2002). Nitrogen (N) removal rates follow a slightly different trajectory, tending to reach equivalency ~30 years after construction, and may have the potential to outperform the denitrification rates of unrestored reference marshes (Ledford et al., 2021). Thus, if nutrient mitigation is a goal of marsh restoration, it is important to evaluate this service through its development so interventions may be performed if needed, such as the dispersal of a thin layer of sediment for accretion purposes (Mitchell & Bilkovic, 2019).

Nutrient cycling, particularly the capacity for N retention and removal, is a key ecosystem service performed by coastal wetlands. Both nitrification, the process of converting ammonium ( $\text{NH}_4^+$ ) to nitrite ( $\text{NO}_2^-$ ), and denitrification, the process wherein heterotrophic anaerobic bacteria convert nitrate ( $\text{NO}_3^-$ ) to N gases (Groffman et al., 2006), are critical and often complimentary reactions of N cycling. These processes are dependent on many factors, with oxygen and carbon (C) availability chief among them. The oxygen content of soils is heavily affected by inundation frequency (determined by both tidal cycles and elevation), with lower

concentrations creating better environs for anaerobic, denitrifying microbes (Wang et al., 2019; Wigand et al., 2016). These anaerobic bacteria also require a source of organic C that can be provided by decaying plant matter or water column organics that settle in the marsh. Organic C can be a limiting factor for denitrification in living shorelines that are built on sand substrate. Anaerobic sediments that favor denitrification, however, can create toxic conditions for marsh plants via sulfide accumulation (Wigand et al., 2014; Alldred et al., 2020). Marsh plants can introduce oxygen back to soils through the rhizosphere, which can aid in coupled nitrification-denitrification. Due to this complex cycle of processes, wholistic assessment of nitrogen cycling is critical to understanding shoreline functioning and health.

Urban marshes are subject to a number of anthropogenic stressors that negatively impact the structural integrity of these ecosystems, namely N loading from combined sewer overflows and wastewater treatment plants (Wigand et al., 2014; Alldred et al., 2020; Morris et al., 2023). Therefore, it is important to identify the site-specific challenges faced in individual restoration scenarios. Eutrophic conditions stemming from anthropogenic nutrient inputs are a major concern for coastal wetlands (Smith, 2003), with algal blooms facilitated by N loading leading to hypoxic conditions and further structural decline of wetlands (Deegan et al., 2012). As such, it is vital to evaluate the condition and functioning of restored marshes and living shorelines in urban, eutrophic ecosystems across various metrics.

A novel living shoreline was constructed in 2023 on the north side of Hudson River Park's Gansevoort Peninsula, along the Hudson River shoreline of lower Manhattan (New York City, NY, USA). In an effort to create a baseline for salt marsh features, this study evaluated *Spartina alterniflora* structure and functioning at the end of its first growing season. We observed different patterns of plant growth through the marsh area and predicted that the marsh

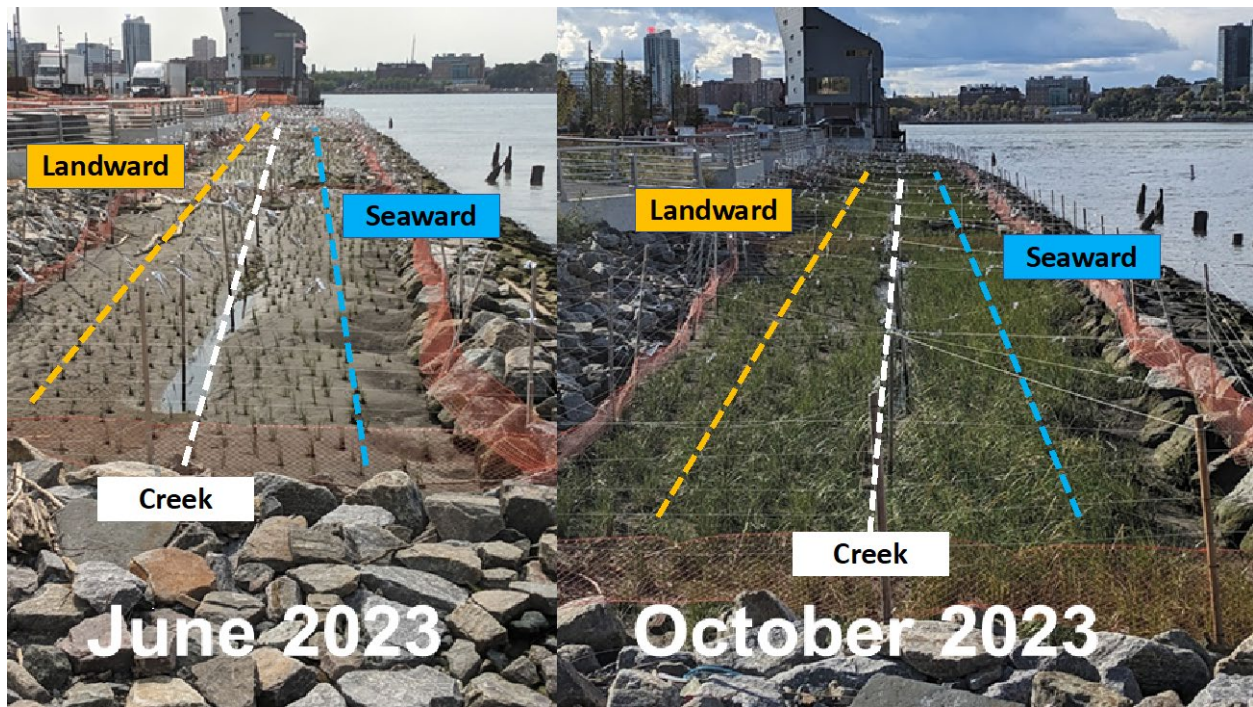
area with the largest plants would have the highest rates of denitrification due to larger plants trapping more carbon in the sediment (i.e., promoting settling) and having greater belowground biomass that would enhance nitrification and thus supply  $\text{NO}_3^-$ . Additionally, we predicted that the shorter plants would have intermediate levels of denitrification and the unvegetated sediment would have the lowest rates of denitrification.

## **Methods**

### *Study Site*

Gansevoort Peninsula is a 5.5-acre park constructed along the Hudson River within Hudson River Park. Built on solid land, the adjacent area was previously comprised of piers that served the Meatpacking District and other commercial operations for many decades. The land was acquired by the NYC Department of Sanitation and operated as an incinerator through the 1990s, and later saw the construction of a road-salt storage shed. The area was in disuse until 2016, when Hudson River Park Trust began re-development of Gansevoort Peninsula, designed by James Corner Field Operations with extensive community input. The space was made available to the public in October of 2023, featuring Manhattan's first public beachfront and many living shoreline features.

The living shoreline component was completed by June of 2023, featuring a robust low marsh zone, *Spartina alterniflora*, approximately 100 m in length and 10-12 m in width. Built over landfill, the marsh has a drainage layer of gravel underneath the surface layer of sandy sediment. The marsh is fully surrounded by rock revetment and features two inlets and a tidal creek running down the marsh's center (Fig. 1).



**Fig. 1.** Images of the Gansevoort salt marsh in June 2023 after planting (left) and in October 2023 at the end of the growing season. Note the creek in the center of the marsh and rock revetment surrounding the marsh area. Photos by C. Zarnoch

### *Vegetation Characteristics*

To examine the marsh at peak biomass (indicated by the presence of seed heads which is the last *S. alterniflora* growth stage before annual senescence [Zheng et al., 2016]), sampling was conducted in October 2023, three line transects were established parallel to the marsh edge with ten sampling points 8-10 m apart (n=30) Each transect was ~2 m in distance to the next transect and were designated “Seaward,” “Creek,” and “Landward” (See Fig. 1). Elevation was determined at each point using a real time kinetic GPS (Trimble R8s GNSS; Westminster, CO). Vegetation coverage (point intercept) was determined within a 1 m<sup>2</sup> quadrat, as was maximum stem height (n=12). A 0.25 m<sup>2</sup> quadrat was placed at each sampling point to measure stem density.

### *Inundation and Water Motion*

Inundation rate ( $\text{h day}^{-1}$ ) for the site was calculated by deploying a HOBO water level logger (Onset, Cape Cod, MA, US) in the center of the marsh, situated within the tidal creek. The logger was deployed from August 2023 to October 2023, recording differences in pressure at 15-minute intervals. A reference logger was kept near the marsh to collect baseline atmospheric pressure.

Cylindrical blocks made of plaster of Paris were deployed to indirectly measure water motion and potential erosion differences experienced by landward and seaward transects through weight loss experienced by the plaster cylinders (Cusson & Bourget, 1997; Potouroglou et al., 2017). All units were created using a 2:1 ratio of water to plaster of Paris mix poured into 100 ml cups. Before setting, a looped cable tie was inserted into the wet mixture to serve as an attachment point. Pre-deployment, the plaster cylinders were sanded down to  $133.5 \pm 1.5$  g and were attached to 20 cm metal stakes at the site. 20 units were deployed along the seaward transect and 20 were deployed along the landward transect ( $n=40$ ), while alternating positioning to avoid interference between the different placements. Units were deployed for six days. Upon retrieval, the cylinders were returned to the lab and dried at  $70^\circ\text{C}$  for 48 hours to eliminate remaining water. Given the brittle nature of the desiccated plasters, a linear regression was constructed to derive the initial dry weight of the units to then calculate the dissolution of the cylinders (weight loss per day) between pre- and post-deployment.

Sediment deposition was studied as an additional indicator of water motion (Fischman et al., 2023). Sediment collectors were made with 10 cm pre-weighed metal pans fitted with 9 cm pre-weighed filter paper (Whatman) and deployed at each point along the landward and seaward transect lines ( $n=20$ ) using 20 cm metal stakes. Collectors were set out for four days before retrieval, then dried at  $70^\circ\text{C}$  for 48 hours. Dried filter paper was removed from the collector and



rinsed with deionized water before being moved to a new pan and dried for another 48 h. Any sediment that remained in the collector was rinsed and dried for 48h. Filter paper and the pan containing the sediment were weighed and sediment deposition rate was calculated.

#### *Sediment core collection and incubations*

Five sediment cores (30 cm x 7.6 cm) were collected for each treatment which included the landward and seaward transects as well as unvegetated areas in the marsh for a total of 15 cores (*i.e.*, 5 treatment<sup>-1</sup>). Each of the cores for the landward and seaward treatments included *S. alterniflora* while the unvegetated contained no plant matter. Seawater (salinity = 15 ppt) from the Hudson River was collected in multiple 20 L carboys from the dock at Pier 40. Cores were kept in a dark cooler and immediately transported with the seawater back to the laboratory. *S. alterniflora* stems were cut and plugged with silicone to minimize gas exchange through the stem. All cores were slowly filled with ~230 ml of seawater and then fitted with a gas tight lid with an inflow and outflow line. Continuous-flow sediment core incubations (Gardner and McCarthy 2009) were performed by pumping site water through the cores with a peristaltic pump at a rate of 1.25 ml min<sup>-1</sup>. The cores were held in a dark environmental chamber at 21°C during the incubation, which was the same temperature as the river at time of collection. After 24 h of continuous-flow through the cores, sample collection was performed. Inflow and outflow samples for nutrient analysis were collected in triplicate by filtering the seawater through a 0.2 µm nylon syringe filter (Thermo Scientific, Rockwood, Tennessee, USA) into 20 ml scintillation vials. Vials were stored at -20°C until analyzed. Triplicate outflow samples for gas analysis were collected in 12 ml exetainer sample vials (Labco Ltd., Lampeter, United Kingdom) by placing the outflow line direct into the vial and letting it overflow for several volumes. Inflow samples were transferred with the syringe and tubing into the bottom of the exetainer vials with overflow.

All gas samples were given 200  $\mu\text{l}$  of zinc chloride, capped with no air bubbles, and stored underwater at 4°C until analyzed.

After seawater sample collection was completed, triplicate 25  $\text{cm}^3$  sediment samples were collected from the sediment surface (0-2.5 cm) of the core. The remaining sediment was then washed through a sieve with 1 mm mesh openings to separate the belowground plant material. The aboveground plant material that was clipped at the start of the incubations was washed to remove sand and seawater. Sediment and above- and belowground biomass were dried at 60°C and weighed. Bulk density ( $\text{g cm}^{-3}$ ), aboveground biomass ( $\text{g m}^{-2}$ ) and belowground biomass were then calculated. Subsamples of dried sediment were placed in a muffle furnace and ashed at 500°C for four hours to determine the sediment organic matter content (%OM).

#### *Nutrient and gas analysis*

Nutrient analysis of core incubation samples (i.e., inflows and outflows) included soluble reactive phosphorous (SRP), ammonium ( $\text{NH}_4^+$ ), nitrate + nitrite ( $\text{NO}_x^-$ ), and nitrite ( $\text{NO}_2^-$ ). Measurements were performed on a SealAQ300 discrete analyzer (Seal Analytical Inc., Mequon, Wisconsin, USA). The phenol-hypochlorite assay (Solorzano 1969) was used to measure  $\text{NH}_4^+$ , and SRP was measured with the antimonyl tartrate method (Murphy and Riley 1962).  $\text{NO}_x^-$  was measured using the sulfanilamide method with cadmium reduction while  $\text{NO}_2^-$  was measured with the sulfanilamide method without cadmium reduction (APHA 1998).

Core incubation samples collected for gas analysis were measured with membrane inlet mass spectrometry (MIMS; Bay Instruments, Easton, MD, USA; Kana et al. 1994) which quantifies the concentrations of  $^{28}\text{N}_2$ ,  $^{32}\text{O}_2$ , and  $^{40}\text{Ar}$  that can be used to calculate net  $\text{N}_2$  fluxes and sediment oxygen demand. A standard, made with artificial seawater of the same salinity as

the samples, was held in a water bath at 21°C and equilibrated to atmospheric gases by low-speed stirring. The standard was measured alongside all samples to correct for instrument drift.

To calculate the gas and nutrient fluxes ( $\mu\text{mol element m}^{-2} \text{ h}^{-1}$ ), the outflow concentrations were subtracted from the inflow concentrations and this value was then multiplied by the flow rate ( $1.25 \text{ ml min}^{-1}$ ) and divided by the core surface area ( $0.003957 \text{ m}^2$ ). Net sediment uptake of the nutrient or gas was indicated by negative fluxes while a release of the nutrient or gas into the water column was indicated by a positive flux. The calculated  $\text{N}_2$  fluxes are considered a net flux since denitrification and nitrogen fixation were likely co-occurring (Fulweiler et al. 2013). A positive net  $\text{N}_2$  flux would indicate that denitrification was the dominant process while a negative net  $\text{N}_2$  flux would indicate that nitrogen fixation was the dominant process.

### *Statistical analysis*

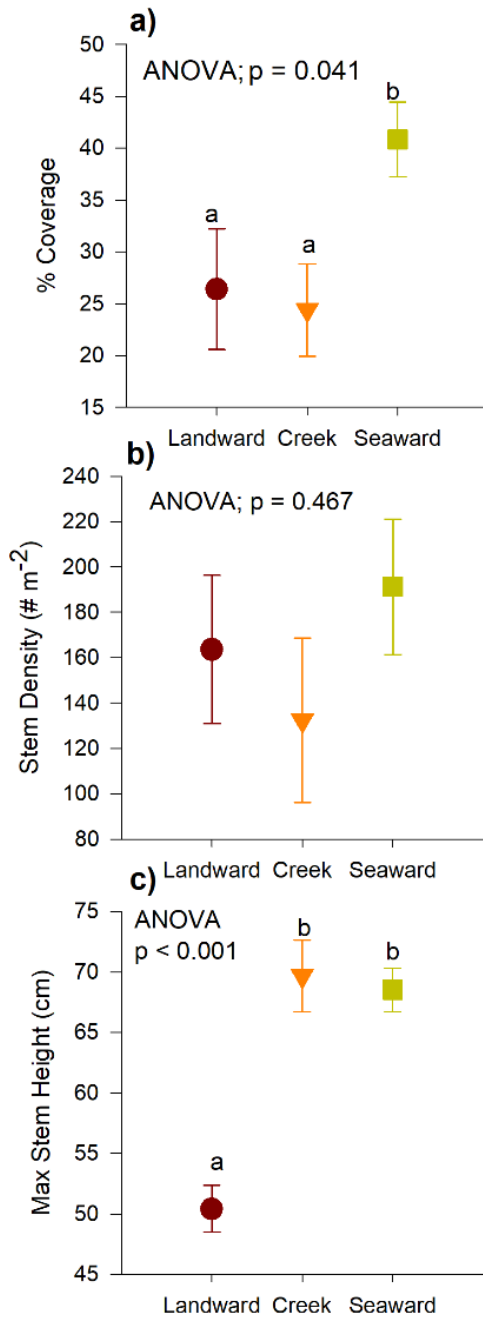
Differences among treatments or transects for vegetation characteristics, sediment organic matter, sediment bulk density, elevation, nutrient fluxes, and gas fluxes were assessed using a one-way analysis of variance. When a significant difference was found among groups, a Tukey test was performed to determine which groups differed from each other. The SRP flux data and the mass loss rates from the plaster of Paris cylinders did not pass the normality tests (Shapiro-Wilk) and could not be appropriately transformed. Instead, the SRP fluxes were analyzed with the Kruskal-Wallis one-way analysis of variance on ranks and the plaster of Paris mass loss rates were analyzed with the Mann-Whitney Rank Sum Test. *S. alterniflora* aboveground biomass, belowground biomass, and sediment deposition, collected from the landward and seaward cores, were compared with *t*-tests. Lastly, linear regressions were used to

explore potential drivers of denitrification. One core from the unvegetated treatment was removed from the  $\text{NO}_3^-$  flux analysis and linear regression due to its flux being an outlier ( $710 \mu\text{mol N m}^{-2} \text{ h}^{-1}$ ); thus this treatment had four replicates of  $\text{NO}_3^-$  fluxes rather than five. Means are presented with the standard error. All statistical analyses were performed with SigmaPlot 11.0 (Systat Software Inc., San Jose, CA).

## Results

### *Vegetation characteristics and elevation*

Although the width of the constructed wetland was relatively narrow ( $\sim 12\text{m}$ ), there were significant differences in vegetative characteristics among the transects. *S. alterniflora* coverage differed among transects (Fig. 2a; ANOVA;  $F = 3.62$ ;  $p = 0.04$ ) with the seaward transect having a mean of  $40.8\% \pm 3.6$  coverage while coverage was similar between the landward ( $26.3\% \pm 5.8$ ) and creek ( $24.4\% \pm 4.5$ ) transects. The overall mean for % coverage was  $30.1\% \pm 3.0$ . Stem density was similar across transects (Fig. 2b; ANOVA;  $F = 0.79$ ;  $p = 0.47$ ) with an overall mean of  $160.4 \text{ stems m}^{-2} \text{ C}$ . Mean maximum stem heights differed among the transects (Fig. 2c; ANOVA;  $F = 22.04$ ;  $P < 0.001$ ) with the landward transect having significantly shorter plants ( $50.4 \text{ cm} \pm 1.9$ ) than the other transects. The overall mean stem height for the constructed marsh was  $62.3 \text{ cm} \pm 3.5$ .



**Fig. 2.** Mean of *S. alterniflora* percent coverage (a), stem density (b), and maximum stem height (c) for the three transects sampled in Gansevoort marsh. Different letters indicate statistically different groups (Tukey).

Analysis of elevation measurements showed a significant difference among the transects (S1; ANOVA;  $F = 26.1$ ;  $p < 0.001$ ) with the creek having a lower elevation ( $0.27\text{m} \pm 0.03$  NAVD88) than the landward ( $0.48\text{m} \pm 0.02$  NAVD88) and seaward ( $0.43\text{m} \pm 0.02$  NAVD88) transects which were similar to each other (Tukey).

#### *Inundation, water motion, and sediment deposition*

The water level logger deployed in the middle of the creek transect showed a general pattern of inundation of approximately  $6\text{-}10$  h  $\text{day}^{-1}$  (S2) depending on moon phase. There was, however, a period of longer inundation due to the effects of Tropical Storm Ophelia in late September 2023 where inundation peaked at  $\sim 15$  h  $\text{day}^{-1}$ . Thus, the total mean inundation rate during the sampling period was  $10$  h  $\text{day}^{-1}$ .

The rate of mass loss ( $\text{g day}^{-1}$ ) was significantly higher (Mann-Whitney Rank Sum;  $U = 124$ ;  $p = 0.041$ ) in the plaster of Paris cylinders deployed in the landward transect than the seaward transect (S3).

Sediment deposition was significantly greater (S4; *t*-test;  $t = 2.32$ ;  $p = 0.037$ ) in the landward transect ( $440.7 \pm 99.7 \text{ g m}^{-2} \text{ d}^{-1}$ ) than the seaward transect ( $177.9 \pm 62.8 \text{ g m}^{-2} \text{ d}^{-1}$ ).

### *Nutrient and gas fluxes*

Nutrient fluxes varied across treatments. The  $\text{NH}_4^+$  fluxes were negative for all treatments which is indicative of sediment uptake of  $\text{NH}_4^+$ . The unvegetated treatment had the greatest uptake, although differences among treatments were not statistically significant (Table 1; ANOVA;  $F = 1.22$ ;  $p = 0.33$ ). Similarly, the  $\text{NO}_2^-$  and SRP fluxes had net uptake for all treatments (Table 1). The uptake of SRP was similar across treatments (ANOVA;  $F = 1.1$ ;  $p = 0.35$ ). However,  $\text{NO}_2^-$  uptake differed significantly among treatments (Kruskal-Wallis;  $H = 7.2$ ;  $p = 0.027$ ). The unvegetated cores had significantly more net  $\text{NO}_2^-$  uptake than the seaward cores (Tukey). The  $\text{NO}_3^-$  fluxes were not statistically different among treatments (Table 1; ANOVA;  $F = 2.97$ ;  $p = 0.35$ ), due to variable fluxes within treatments. However, the landward and seaward cores, both with *S. alterniflora*, showed net uptake of  $\text{NO}_3^-$  while the unvegetated cores had a net release of  $\text{NO}_3^-$  to the water column. Ambient water collected from the Hudson River for use in the continuous-flow sediment core incubations had  $11.1 \text{ }\mu\text{M}$  of  $\text{NH}_4^+$ ,  $4.2 \text{ }\mu\text{M}$  of  $\text{NO}_2^-$ ,  $34.6 \text{ }\mu\text{M}$  of  $\text{NO}_3^-$ , and  $2.6 \text{ }\mu\text{M}$  of SRP.

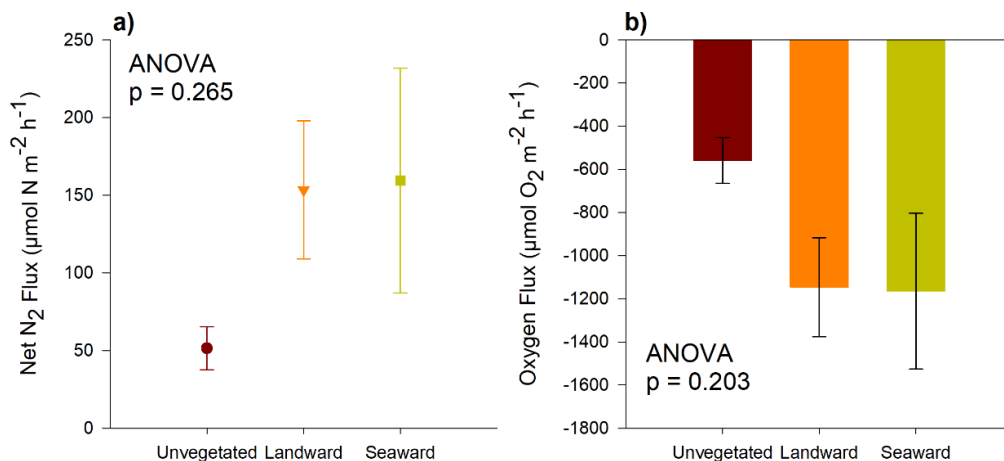
**Table 1.** Mean ( $\pm$ SE) fluxes of nutrients from treatments used in the continuous-flow sediment core incubations. Statistical comparisons were made across rows and the p-value is indicated in the ANOVA column. Significant p-values are bolded. Different superscript letters among treatments indicate a statistical difference (Tukey).

<b>Nutrient Flux</b>	<b>Unvegetated</b>	<b>Landward</b>	<b>Seaward</b>	<b>ANOVA</b>
$\text{NH}_4^+$ ( $\mu\text{mol N m}^{-2} \text{ h}^{-1}$ )	-99.856 (36.826)	-35.070 (31.780)	-20.286 (45.294)	$p = 0.33$
$\text{NO}_3^-$ ( $\mu\text{mol N m}^{-2} \text{ h}^{-1}$ )	28.052 (31.473)	-102.924 (33.852)	-107.831 (54.538)	$p = 0.093$
$\text{NO}_2^-$ ( $\mu\text{mol N m}^{-2} \text{ h}^{-1}$ )	-19.885 <sup>a</sup> (6.0)	-8.849 <sup>a,b</sup> (2.175)	-3.184 <sup>b</sup> (2.748)	<b><math>p = 0.027</math></b>
SRP ( $\mu\text{mol P m}^{-2} \text{ h}^{-1}$ )	-4.553 (0.610)	-7.858 (1.912)	-7.888 (2.366)	$p = 0.351$

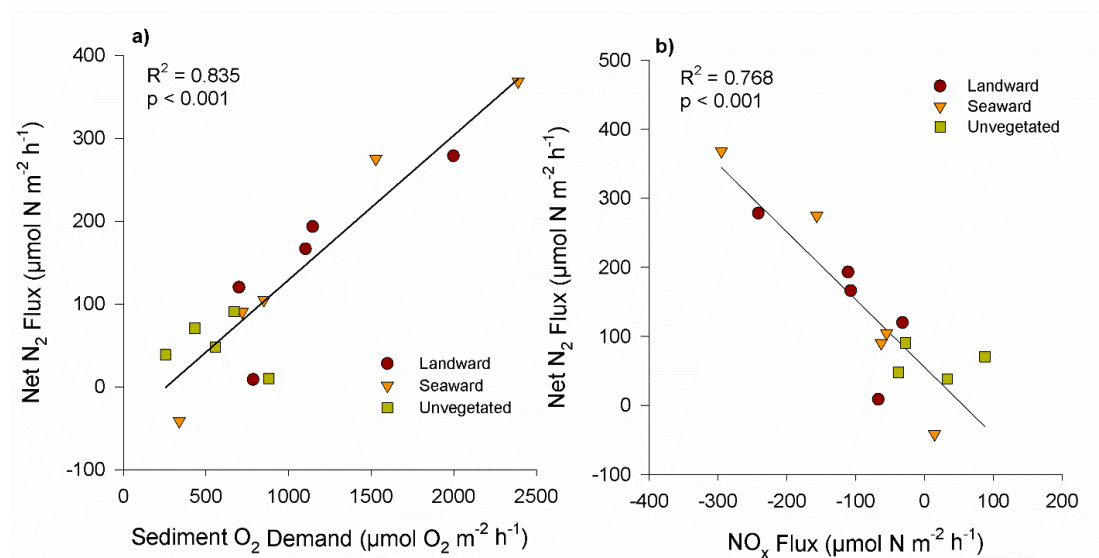
The net N<sub>2</sub> fluxes across all treatments were positive, which was indicative of net denitrification. Although there were no statistically significant differences among treatments (ANOVA;  $F = 1.5$ ;  $p = 0.265$ ), the landward and seaward cores, both with *S. alterniflora*, had net N<sub>2</sub> fluxes that were ~3 times greater than the unvegetated cores (Fig. 3a). The O<sub>2</sub> fluxes were negative, indicating sediment uptake of O<sub>2</sub> for all treatments. No statistical difference was found among treatments (ANOVA;  $F = 1.8$ ;  $p = 0.203$ ) but the landward and seaward cores had nearly double the O<sub>2</sub> demand of the unvegetated cores (Fig. 3a). Regression analyses indicate that the net N<sub>2</sub> fluxes were positively related to sediment oxygen demand (Fig. 4a;  $F = 65.75$ ;  $p < 0.001$ ) and NO<sub>x</sub><sup>-</sup> (NO<sub>3</sub><sup>-</sup> + NO<sub>2</sub><sup>-</sup>) uptake (Fig. 4b;  $F = 39.76$ ;  $p < 0.001$ ).

#### *Sediment characteristics and plant biomass in cores*

The *S. alterniflora* collected in the landward and seaward had significantly different (*t*-test;  $t = -5.024$ ;  $p = 0.001$ ) aboveground biomass with the landward having  $75.36 \pm 18.23$  g m<sup>-2</sup> and the seaward with  $779.68 \pm 139.01$  g m<sup>-2</sup>. Belowground biomass, however, did not differ (*t*-test;  $t = -0.401$ ;  $p = 0.7$ ) between these treatments and the total mean was  $438.51 \pm 101.19$  g m<sup>-2</sup>. There was also no significant difference in sediment organic matter (ANOVA;  $F = 0.88$ ;  $p = 0.44$ ) and bulk density (ANOVA;  $F = 0.302$ ;  $p = 0.745$ ) among the three treatments. The mean for organic matter (OM) samples was  $1.31\% \pm 0.05$  and the mean for bulk density was  $1.66 \pm 0.03$  g cm<sup>-3</sup>.



**Fig. 3.** Mean net N<sub>2</sub> (a) and O<sub>2</sub> fluxes (b) from sediment cores collected from three locations (unvegetated, landward, and seaward) within Gansevoort marsh.



**Fig. 4.** Regressions of sediment oxygen demand and net N<sub>2</sub> fluxes (a) as well as NO<sub>x</sub><sup>-</sup> (NO<sub>3</sub><sup>-</sup> + NO<sub>2</sub><sup>-</sup>) fluxes with net N<sub>2</sub> fluxes (b).

## Discussion

Based on observed differences in *S. alterniflora* growth throughout the Gansevoort Peninsula living shoreline, we hypothesized that seaward sediments (containing the tallest plants) would show the highest denitrification rates due to greater belowground biomass trapping more



C for microbial activity and fostering greater coupled nitrification-denitrification than sediments with shorter plants. Conversely, we predicted that unvegetated sediments would have relatively low denitrification compared to vegetated areas. However, patterns of aboveground plant biomass (higher seaward biomass vs. lower landward biomass) did not change sediment OM and belowground biomass was not different between treatments despite differences in aboveground growth. Similarly, there was no statistically significant treatment effect on denitrification rates, nutrient uptake (except  $\text{NO}_2^-$ ), or sediment oxygen demand. This was surprising given that denitrification was three times higher in vegetated cores than unvegetated cores. However, this was likely related to high variability in flux rates associated with the heterogeneity of a newly constructed marsh. Landward and seaward areas differed in terms of their erosion and sediment deposition dynamics, which may not have resulted in significant biogeochemical differences after the first year of planting but may create disparities in ecosystem service provision between these marsh areas in future years. It is likely that denitrification will become a more significant source of nitrogen (N) removal in future years as the marsh accumulates C and continues to experience high availability of water column  $\text{NO}_3^-$ . Overall, the Gansevoort Peninsula vegetated marsh acted as a net N sink within its first year of construction and should become a greater sink with continued development of the marsh area, emphasizing its value for urban water quality mitigation.

### *Vegetation Characteristics Promote Marsh Stability*

Few studies illuminate growth patterns of living shorelines after the first year of planting, but *S. alterniflora* biomass already differed within marsh zones during the first growing season. Plants in the seaward transect had the most vegetative coverage (Fig. 2a) of the three transects studied. Plant characteristics in this section of the marsh are comparable to restored marsh areas

of greater age elsewhere in the Hudson River Estuary system (Morris et al., 2023). Mean seaward aboveground biomass on ( $779.68 \pm 139.01 \text{ g m}^{-2}$ ; Table 1) was higher than that of 3-, 5-, 9-, and 12-year post-restoration marshes in Jamaica Bay, and mean *S. alterniflora* stem density ( $160.4 \text{ stems m}^{-2}$ ; Table 1) was similar to the stem density of a 5-year-old restored marsh known as Elders West (Morris et al., 2023). It is probable that robust establishment in seaward vegetation will continue in future years, given previously described growth patterns of *S. alterniflora* in other living shoreline restoration studies and conditions specific to the site. For example, rapid growth within the first year of planting has been observed in other planted living shorelines relative to areas left to colonize naturally, likely due to lack of competition with plants starting from seed (Howard et al., 2019). Future growth is especially likely in the second year after planting. *S. alterniflora* experiences more robust growth two years after restoration through a combination of seed and rhizomatic propagation (VanZomeren et al., 2018). Similarly, *S. alterniflora* coverage in planted living shoreline areas has been described as equivalent to natural sites after two years (Howard et al., 2019).

High N loads in the estuary may further contribute to aboveground biomass growth (Crosby et al., 2021). Of available dissolved inorganic N species,  $\text{NH}_4^+$  best facilitates *S. alterniflora* growth, but these plants will also incorporate  $\text{NO}_3^-$  when it is abundant (Bowen et al., 2020). Mean water column  $\text{NH}_4^+$  ( $11.1 \text{ }\mu\text{M}$ ) at the time of our core sampling was more than twice as high as previously described average  $\text{NH}_4^+$  concentrations for the Hudson River Estuary (Lampman et al., 1999). Similarly, a mean  $\text{NO}_3^-$  concentration of  $34.6 \text{ }\mu\text{M}$  in our study is comparable with mean concentrations estimated at  $\sim 40 \text{ }\mu\text{M}$  in the Hudson River, which has been described to sustain high primary productivity (Lampman et al., 1999).

### *Sediment Transport Creates Growth Disparity*

Landward *S. alterniflora* had consistently lower biomass than seaward conspecifics (Figs. 2a-c) despite the similarity in elevation among transects (S1). Thus, nutrient loads may have facilitated aboveground growth overall, but they likely did not differentially impact growth based on elevation/inundation regimes. Instead, these differences in biomass may be explained by differences in sediment and water movement. Landward sediment deposition was significantly greater ( $440.7 \pm 99.7 \text{ g m}^{-2} \text{ d}^{-1}$ ) than deposition in the seaward transect ( $177.9 \pm 62.8 \text{ g m}^{-2} \text{ d}^{-1}$ ) (S4), and landward plaster erosion cylinders lost significantly ( $p = 0.041$ ; S3) more mass than those deployed on the seaward side. While seemingly paradoxical, greater sediment deposition and erosion in the landward section might suggest a feedback of sorts: high erosion may lead to shorter, less dense plants that cannot keep pace with high-intensity sediment turnover, causing further erosion. It is also possible that the landward side of the marsh experiences impacts from large floating debris trapped beneath the park's overhanging bulkhead, but this has not been quantified. However, after the first year of planting, differences in sediment transport dynamics or aboveground biomass do not appear to affect biogeochemical processes.

### *Newly Planted Shorelines Foster Denitrification*

Both vegetated and unvegetated sediments showed net positive fluxes of  $\text{N}_2$  (Fig. 4a), indicating that the living shoreline overall is denitrification-dominated. These results were similar to those of another study on living shorelines (aged 0-20 years) that showed marsh sediments to be denitrification dominated across seasons and for all ages (Onorevole et al., 2018). In contrast, newly restored marshes have been found to have higher rates of nitrogen fixation (Currin et al., 1995). Denitrification may be limited by porewater nutrients (Thompson et al., 1995) and may take many years to provide equivalent nitrogen removal in constructed as

compared to natural marshes (Broom and Craft, 2000). In urban eutrophic systems, however, constructed marshes were found to have similar rates of denitrification as natural marshes (Govinda et al., 2023). Denitrification-related microbial activity was also found to be high in 3- and 5-year post-restoration marshes within eutrophic Jamaica Bay (NY; Morris et al., 2023). Vegetated sediments in the landward and seaward transects showed similar rates of net nutrient uptake (Table 1), and denitrification rates in vegetated sediments were around three times as high as unvegetated treatments (Fig. 5a), although variability in core flux rates likely precluded statistical significance. This high degree of variation may have been a product of the relative youth of the marsh: recently restored (~3 years post-restoration) marshes have higher microbial diversity than older restorations (Morris et al., 2023), and therefore may demonstrate more heterogeneous denitrification responses. Across treatment types, N<sub>2</sub> production appeared to comprise of a combination of coupled nitrification-denitrification (shown by a positive relationship between N<sub>2</sub> production and sediment oxygen demand; Fig. 6a [Seitzinger et al., 2006; Piehler and Smyth, 2011]) and direct denitrification (evidenced by a significant relationship between N<sub>2</sub> production and NO<sub>3</sub><sup>-</sup> uptake; Fig. 6b [Vieillard and Fulweiler, 2012]). Direct and coupled denitrification have been typically found to co-occur in natural marshes (Hamersley and Howes 2005; Koop-Jakobsen and Giblin, 2010), and the mean rates of combined denitrification types found in the Gansevoort Peninsula marsh fit into the ranges of N<sub>2</sub> production of these marshes (between 71.3 and 186.4 μmol N m<sup>-2</sup>h<sup>-1</sup> [Koop-Jakobsen and Giblin, 2010]).

Despite comparable rates of denitrification with natural marsh sites, sediment characteristics in this study more closely matched those in recently restored marshes. Overall, OM content was low compared to a natural marsh as expected (Craft et al., 2003; Morris et al.,

2023), at  $1.31\% \pm 0.05$ , but was higher than sediments sampled from 3- and 5-year post-restoration marshes in Jamaica Bay ( $0.85\% \pm 0.67$  and  $0.37\% \pm 0.10$  respectively) (Morris et al., 2023). The relationship between sediment OM and denitrification is unclear at this site, but it is likely that, as observed in other restored marshes, sediment C is limiting (Craft et al., 2003). Continued development of the marsh should lead to greater accumulation of C in the sediments. Given that direct denitrification and coupled nitrification-denitrification appear to be co-occurring, it would be interesting to see if denitrification increases with future increases in sediment C or if other factors end up limiting denitrification.

#### *Biogeochemical Differences in Marsh Zones May Appear with Increasing Age*

$N_2$  production and nutrient uptake rates between the two vegetated sites may have been comparable due to the similarity in belowground biomass and sediment OM after the marsh's first year. However, given the differences in sediment deposition/erosion dynamics between the landward and seaward sections of the shoreline, future differences in the biogeochemistry of these zones may become more evident over time. As *S. alterniflora* is likely to continue to proliferate in the seaward zone (possibly from less active erosion), soils may be slower to wash away, which, over the next decade or so, may provide greater accretion of sediment C (Drexler et al., 2019). Additionally, if seaward *S. alterniflora* continues to establish in predicted patterns, belowground biomass is also likely to increase (Howard et al., 2019). This may further enhance the potential for coupled nitrification-denitrification in high-biomass areas (Allred et al., 2020); less densely vegetated landward areas are more likely instead to be dominated by direct denitrification if over time belowground biomass does not significantly accrue. As the marsh is still less than a year old, however, seasonal, and annual monitoring will help determine whether predicted patterns manifest.

### *Conclusion: Living Shoreline Provides Ecosystem Service Benefits Overall*

Although the results of the core incubation study contradict our original hypothesis, it is evident that vegetated areas in this newly constructed salt marsh currently offer ecosystem services of denitrification and sediment deposition, both of which are valuable in highly urbanized systems such as the Hudson River. Furthermore, both vegetated and unvegetated areas were net denitrification-dominated, indicating the value of “soft” shorelines with marsh grasses in terms of mitigating eutrophication. For example, if we were to apply the mean rate of denitrification to the entire area of the marsh and assume this rate is consistent over the entire day (Koop-Jakobsen and Giblin, 2010), the marsh would remove  $\sim 49 \text{ kg N day}^{-1}$  during the period of sampling. Future measurements across seasons could produce an annual rate of removal. Longer-term, it is probable that the living shoreline at the Gansevoort Peninsula will continue to provide N removal services, and eventually C accretion in more densely vegetated areas, as *Spartina* growth is established. Future studies will need to identify changes in plant biomass and C accumulation over time as well as other indicators of living shoreline stability (e.g., impacts of marine debris). Lastly, additional research could document more ecosystem service benefits from the living shoreline, including habitat creation and social/aesthetic value particularly given the highly urban, populated location of the Gansevoort living shoreline.

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## References

- Alikhani, S., Nummi, P., & Ojala, A. (2021). Urban wetlands: A review on ecological and cultural values. *Water*, 13(22), 3301.
- Alldred, M., Borrelli, J. J., Hoellein, T., Bruesewitz, D., & Zarnoch, C. (2020). Marsh Plants Enhance Coastal Marsh Resilience by Changing Sediment Oxygen and Sulfide Concentrations in an Urban, Eutrophic Estuary. *Estuaries and Coasts* 43, 801–13. <https://doi.org/10.1007/s12237-020-00700-9>.
- American Public Health Association (APHA) (1998) Standard methods for the examination of water and wastewater. Baltimore
- Bowen, J. L., Giblin, A. E., Murphy, A. E, Bulseco, A. N., Deegan, L. A., Johnson, D. S., Nelson, J. A., Mozdzer, T. J., and Sullivan, H. L. (2020). Not All Nitrogen Is Created Equal: Differential Effects of Nitrate and Ammonium Enrichment in Coastal Wetlands. *BioScience* 70, 1108–19. <https://doi.org/10.1093/biosci/biaa140>.
- Broome, S. W., & Craft, C. B. (2000). Tidal salt marsh restoration, creation, and mitigation. *Reclamation of drastically disturbed lands*, 41, 939-959.
- Burke, D. G., Koch, E. W., & J. Court Stevenson. (2005). Assessment of Hybrid Type Shore Erosion Control Projects in Maryland's Chesapeake Bay: Phases I & II. Maryland Department of Natural Resources.
- Craft, C., Broome, S., & Campbell, C. (2002). Fifteen years of vegetation and soil development after brackish-water marsh creation. *Restoration Ecology*, 10(2), 248-258.
- Craft, C., Megonigal, P., Broome, S., Stevenson, J., Freese, R., Cornell, J., Zheng, L. and Sacco, J., 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. *Ecological applications*, 13(5), pp.1417-1432. <https://doi.org/10.1890/02-5086>
- Chambers, R. M., Gorsky, A. L., Isdell, R. E., Mitchell, M. M., & Bilkovic, D. M. (2021). Comparison of nutrient accrual in constructed living shoreline and natural fringing marshes. *Ocean & Coastal Management*, 199, 105401.
- Crosby, S. C., Spiller, N. C., Healy, D. S., Brideau, L., Stewart, L. M., Vaudrey, J. M. P., Tietz, K. E., & Fraboni, P. J. (2021). Assessing the Resiliency of Salt Marshes Under Increasing Nitrogen Loading. *Estuaries and Coasts* 44, 1658–70. <https://doi.org/10.1007/s12237-021-00899-1>.
- Currin, C. A., Joye, S. B., & Paerl, H. W. (1996). Diel Rates of N<sub>2</sub>-fixation and Denitrification in a Transplanted *Spartina alterniflora* Marsh: Implications for N-flux Dynamics. *Estuarine, Coastal and Shelf Science*, 42(5), 597-616.
- Currin, C. A., Chappell, W. S., & Deaton, A. (2010). Developing alternative shoreline armoring strategies: the living shoreline approach in North Carolina.
- Cusson, M., & Bourget, E. (1997). Influence of topographic heterogeneity and spatial scales on the structure of the neighbouring intertidal endobenthic macrofaunal community. *Marine Ecology Progress Series*, 150, 181-193.
- Davis, J. L., Currin, C. A., O'Brien, C., Raffenburg, C., & Davis, A. (2015). Living shorelines: coastal resilience with a blue carbon benefit. *PloS one*, 10(11), e0142595.
- Davidson, N. C. (2014). How much wetland has the world lost? Long-term and recent trends in global wetland area. *Marine and Freshwater Research*, 65(10), 934-941.
- Deegan, L. A., Johnson, D. S., Warren, R. S., Peterson, B. J., Fleeger, J. W., Fagherazzi, S., & Wollheim, W. M. (2012). Coastal eutrophication as a driver of salt marsh loss. *Nature*, 490(7420), 388-392.

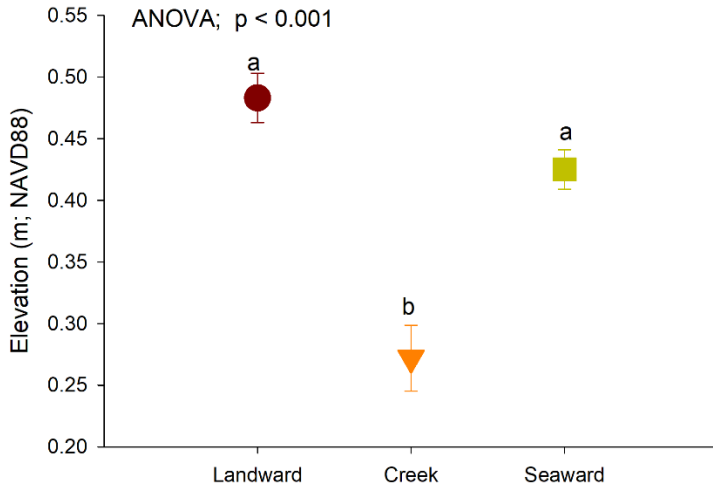
- Feher, L. C., Osland, M. J., Griffith, K. T., Grace, J. B., Howard, R. J., Stagg, C. L., ... & Rogers, K. (2017). Linear and nonlinear effects of temperature and precipitation on ecosystem properties in tidal saline wetlands. *Ecosphere*, 8(10), e01956.
- Fischman, H. S., Smyth, A. R., & Angelini, C. (2023). Invasive consumers provoke ecosystem-wide disruption of salt marsh functions by dismantling a keystone mutualism. *Biological Invasions*, 1-17.
- Fulweiler, R. W., Brown, S. M., Nixon, S. W., & Jenkins, B. D. (2013). Evidence and a conceptual model for the co-occurrence of nitrogen fixation and denitrification in heterotrophic marine sediments. *Marine Ecology Progress Series*, 482, 57-68.
- Gardner, W. S., & McCarthy, M. J. (2009). Nitrogen dynamics at the sediment–water interface in shallow, sub-tropical Florida Bay: why denitrification efficiency may decrease with increased eutrophication. *Biogeochemistry*, 95, 185-198.
- Gilby, B. L., Weinstein, M. P., Baker, R., Cebrian, J., Alford, S. B., Chelsky, A., Colombano, D., Connolly, R.M., Currin, C.A., Feller, I.C. and Frank, A.,... & Ziegler, S. L. (2021). Human actions alter tidal marsh seascapes and the provision of ecosystem services. *Estuaries and Coasts*, 44, 1628-1636.
- Gittman, R.K., Fodrie, F.J., Popowich, A.M., Keller, D.A., Bruno, J.F., Currin, C.A., Peterson, C.H. and Piehler, M.F. (2015). Engineering away our natural defenses: an analysis of shoreline hardening in the US. *Frontiers in Ecology and the Environment*, 13(6), 301-307.
- Gittman, R. K., Scyphers, S. B., Smith, C. S., Neylan, I. P., & Grabowski, J. H. (2016). Ecological consequences of shoreline hardening: a meta-analysis. *BioScience*, 66(9), 763-773.
- Govinda, N., Groffman, P. M., Durand, S. E., Zarnoch, C. B., & Elkins, W. (2023). Denitrification Potential of Surface Soils of Constructed Wetlands in Newtown Creek, an Urban Superfund Site. *J. of Environmental Quality*  
DOI: 10.1002/jeq2.20495
- Groffman, P. M., Altabet, M. A., Böhlke, H., Butterbach-Bahl, K., David, M. B., Firestone, M. K., ... & Voytek, M. A. (2006). Methods for measuring denitrification: diverse approaches to a difficult problem. *Ecological applications*, 16(6), 2091-2122.
- Guthrie, A. G., Bilkovic, D. M., Mitchell, M., Chambers, R., Thompson, J. S., & Isdell, R. E. (2022). Ecological equivalency of living shorelines and natural marshes for fish and crustacean communities. *Ecological Engineering*, 176, 106511.
- Hamersley, M. R., & Howes, B. L. (2005). Coupled Nitrification–Denitrification Measured in Situ in a *Spartina Alterniflora* Marsh with a  $15\text{NH}_4^+$  Tracer. *Marine Ecology Progress Series* 299: 123–35.  
<https://doi.org/10.3354/meps299123>.
- Howard, R. J., Rafferty, P. S., & Johnson, D. J. (2020). Plant community establishment in a coastal marsh restored using sediment additions. *Wetlands*, 40(4), 877-892.
- Isdell, R. E., Bilkovic, D. M., Guthrie, A. G., Mitchell, M. M., Chambers, R. M., Leu, M., & Hershner, C. (2021). Living shorelines achieve functional equivalence to natural fringe marshes across multiple ecological metrics. *PeerJ*, 9, e11815.
- Kana, T. M., Darkangelo, C., Hunt, M. D., Oldham, J. B., Bennett, G. E., & Cornwell, J. C. (1994). Membrane inlet mass spectrometer for rapid high-precision determination of  $\text{N}_2$ ,  $\text{O}_2$ , and Ar in environmental water samples. *Analytical chemistry*, 66(23), 4166-4170.
- Koop-Jakobsen, K. & Giblin, A. E. (2010). The Effect of Increased Nitrate Loading on Nitrate Reduction via Denitrification and DNRA in Salt Marsh Sediments. *Limnology and Oceanography* 55789–802.  
<https://doi.org/10.4319/lo.2010.55.2.0789>.



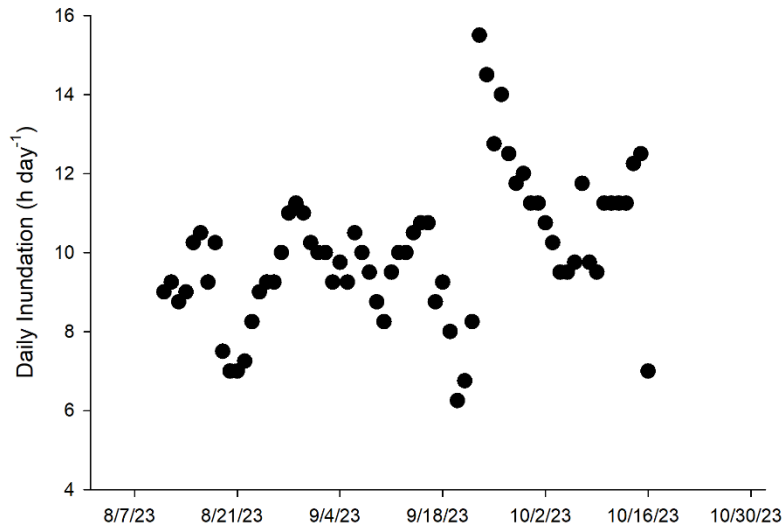
- Lampman, G. G., Caraco, N. F., & Cole, J. J. (1999). Spatial and Temporal Patterns of Nutrient Concentration and Export in the Tidal Hudson River. *Estuaries*, 22(2). <https://doi.org/10.2307/1352984>
- Ledford, T. C., Mortazavi, B., Tatariw, C., Starr, S. F., Smyth, E., Wood, A. G., ... & Cherry, J. A. (2021). Ecosystem carbon exchange and nitrogen removal rates in two 33-year-old constructed salt marshes are similar to those in a nearby natural marsh. *Restoration Ecology*, 29(7), e13439.
- Liu, Z., Fagherazzi, S., & Cui, B. (2021). Success of coastal wetlands restoration is driven by sediment availability. *Communications Earth & Environment*, 2(1), 44.
- Mitchell, M., Bilkovic, D. M., & Pinto, R. (2019). Embracing dynamic design for climate-resilient living shorelines. *The Journal of Applied Ecology*, 56(5), 1099–1105. <https://doi.org/10.1111/1365-2664.13371>
- Moreno-Mateos, D., Power, M. E., Comín, F. A., & Yockteng, R. (2012). Structural and functional loss in restored wetland ecosystems. *PLoS biology*, 10(1), e1001247.
- Morris, R. L., Bilkovic, D. M., Walles, B., & Strain, E. M. (2022). Nature-based coastal defence: Developing the knowledge needed for wider implementation of living shorelines. *Ecological Engineering*, 185, 106798.
- Morris, N., Alldred, M., Zarnoch, C., & Alter, S. E. (2023). Estuarine Sediment Microbiomes from a Chronosequence of Restored Urban Salt Marshes. *Microbial Ecology* 85, 916–30. <https://doi.org/10.1007/s00248-023-02193-y>.
- Murphy, J. A. M. E. S., & Riley, J. P. (1962). A modified single solution method for the determination of phosphate in natural waters. *Analytica chimica acta*, 27, 31-36.
- Nadeau, C. P., & Conway, C. J. (2015). Optimizing water depth for wetland-dependent wildlife could increase wetland restoration success, water efficiency, and water security. *Restoration Ecology*, 23(3), 292-300.
- Onorevole, K. M., Thompson, S. P., & Piehler, M. F. (2018). Living shorelines enhance nitrogen removal capacity over time. *Ecological engineering*, 120, 238-248.
- Ostrowski, A., Connolly, R. M., & Sievers, M. (2021). Evaluating multiple stressor research in coastal wetlands: a systematic review. *Marine Environmental Research*, 164, 105239.
- Piehler, M. F., & Smyth, A. R. (2011). Habitat-specific Distinctions in Estuarine Denitrification Affect Both Ecosystem Function and Services. *Ecosphere* 2, 1–17. <https://doi.org/10.1890/ES10-00082.1>.
- Potouroglou, M., Bull, J. C., Krauss, K. W., Kennedy, H. A., Fusi, M., Daffonchio, D., ... & Huxham, M. (2017). Measuring the role of seagrasses in regulating sediment surface elevation. *Scientific reports*, 7(1), 11917.
- Seitzinger, S., Harrison, J. A., Böhlke, J. K., Bouwman, A. F., Lowrance, R., Peterson, B., Tobias, C., & Van Drecht, G. (2006). Denitrification Across Landscapes and Waterscapes: A Synthesis. *Ecological Applications* 16, 2064–90. [https://doi.org/10.1890/1051-0761\(2006\)016\[2064:DALAWA\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2006)016[2064:DALAWA]2.0.CO;2).
- Smee, D. L. (2019). Coastal ecology: living shorelines reduce coastal erosion. *Current Biology*, 29(11), R411-R413.
- Solorzano, L. (1969). Determination of ammonia in natural waters by the phenol hypochlorite method. *Limnology and oceanography*, 14(5), 799-801.
- Smith, V. H. (2003). Eutrophication of freshwater and coastal marine ecosystems a global problem. *Environmental Science and Pollution Research*, 10, 126-139.
- Spieles, D. J. (2022). Wetland construction, restoration, and integration: A comparative review. *Land*, 11(4), 554.
- Tatariw, C., Mason, O. U., & Mortazavi, B. (2021). Ditching nutrients: Roadside drainage networks are hotspots for microbial nitrogen removal. *Journal of Geophysical Research: Biogeosciences*, 126(7), e2020JG006115.

- Thompson, S. P., Paerl, H. W., & Go, M. C. (1995). Seasonal patterns of nitrification and denitrification in a natural and a restored salt marsh. *Estuaries*, *18*, 399-408.
- Toimil, A., Losada, I. J., Nicholls, R. J., Dalrymple, R. A., & Stive, M. J. (2020). Addressing the challenges of climate change risks and adaptation in coastal areas: A review. *Coastal Engineering*, *156*, 103611.
- VanZomerem, C. M., Berkowitz, J. F., Piercy, C. D., & White, J. R. (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>.
- Vieillard, A. M., Fulweiler, R. W. (2012). Impacts of Long-Term Fertilization on Salt Marsh Tidal Creek Benthic Nutrient and N<sub>2</sub> Gas Fluxes. *Marine Ecology Progress Series* *471*: 11–22. <https://doi.org/10.3354/meps10013>.
- Wang, F., Eagle, M., Kroeger, K. D., Spivak, A. C., & Tang, J. (2021). Plant biomass and rates of carbon dioxide uptake are enhanced by successful restoration of tidal connectivity in salt marshes. *Science of the Total Environment*, *750*, 141566.
- Wigand, C., Sundberg, K., Hanson, A., Davey, E., Johnson, R., Watson, E., & Morris, J. (2016). Varying inundation regimes differentially affect natural and sand-amended marsh sediments. *PLoS One*, *11*(10), e0164956.
- Wigand, C., Roman, C. T., Davey, E., Stolt, M., Johnson, R., Hanson, A., ... & Rafferty, P. (2014). Below the disappearing marshes of an urban estuary: historic nitrogen trends and soil structure. *Ecological Applications*, *24*(4), 633-649.
- White, N. J., Church, J. A., & Gregory, J. M. (2005). Coastal and global averaged sea level rise for 1950 to 2000. *Geophysical Research Letters*, *32*(1).
- Young, S. S., Rao, S., & Dorey, K. (2021). Monitoring the Erosion and Accretion of a Human-Built Living Shoreline with Drone Technology. *Environmental Challenges* *5*: 100383. <https://doi.org/10.1016/j.envc.2021.100383>.
- Zedler, J. B., & Kercher, S. (2005). Wetland resources: status, trends, ecosystem services, and restorability. *Annu. Rev. Environ. Resour.*, *30*, 39-74.
- Zheng, S., Shao, D., Asaeda, T., Sun, T., Luo, S. & Cheng, M. (2016) Modeling the Growth Dynamics of *Spartina Alterniflora* and the Effects of Its Control Measures. *Ecological Engineering* *97*: 144–56. <https://doi.org/10.1016/j.ecoleng.2016.09.006>.

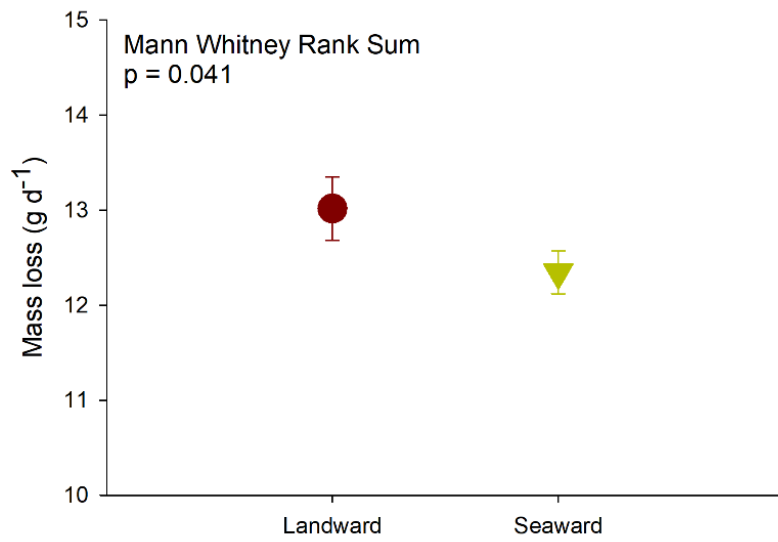
## Supplemental Appendices



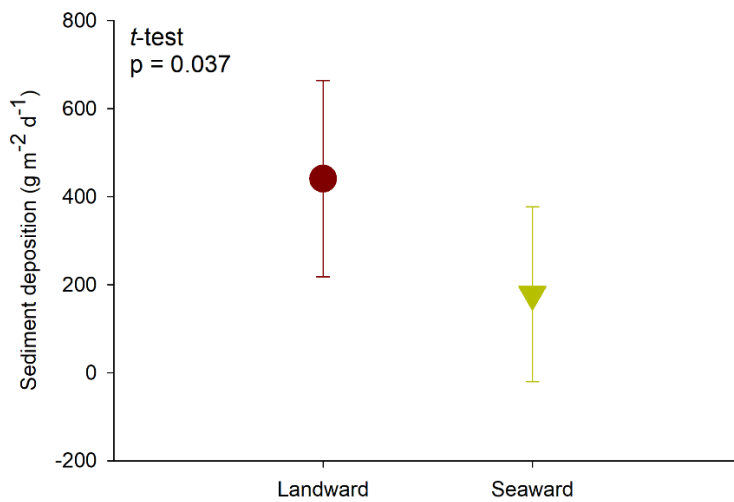
**S1.** Mean elevation (m;NAVD88) for the three transects sampled in Gansevoort marsh. Different letters indicate a statistical difference (Tukey).



**S2.** Mean daily inundation rate (h day<sup>-1</sup>) during the sampling period of peak *S. alterniflora* biomass. The water level logger was deployed in the creek in the center of the Gansevoort marsh.



**S3.** Mean mass loss (g day<sup>-1</sup>) of Plaster of Paris cylinders deployed in the landward and seaward transects of Gansevoort marsh.



**S4.** Sediment deposition (g m<sup>-2</sup> d<sup>-1</sup>) in sediment collectors deployed in the landward and seaward transects of Gansevoort marsh.