Title: Salt marsh climate adaptation: Using runnels to adapt to accelerating sea level rise within a drowning New England salt marsh

Running Head: Salt marsh climate adaptation

Danielle C. Perry[1,2*] Wenley Ferguson[3] and Carol S. Thornber[1]

[1] Department of Natural Resources Science, University of Rhode Island, Kingston, RI USA 02881

[2]The Massachusetts Audubon Society, Lincoln, MA, USA 02451

[3]Save The Bay, Providence, RI USA 02881

*Corresponding author email: dperry@massaudubon.org

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WF, DP, CT conceived and designed the research; DP, WF performed the experiments and collected data; DP analyzed the data; CT contributed materials/analysis tools; DP, CT, WF wrote and edited the manuscript.

Abstract

Sea level rise within New England is accelerating at a rate faster than the global average, leaving salt marshes particularly susceptible to degradation. Hydrological alteration is a type of climate change adaptation technique that has been implemented worldwide to combat the effects of sea level rise within salt marshes. Runnels (shallow channels) are a type of climate adaptation strategy used to enhance drainage in drowning marshes. In this study, we investigated the impacts of runnel installations, three-five years post-implementation, on soil properties, vegetation composition, and greenhouse gas fluxes. We studied two runnel treatments, Low Elevation Runnel and High Elevation Runnels, *Spartina alterniflora* stem density () significantly increased after three growing seasons after runnels were installed, and the high marsh plant, *Spartina patens*, persisted in the High Elevation Runnel areas. There was a significant difference in carbon dioxide uptake rates among treatments, with the unmanipulated (Reference) areas having the highest uptake rates and an increase in CO_2 uptake over time seen in certain runnel treatments. These findings highlight the potential use of a climate change adaptation strategy to combat sea level rise impacts and provide insights for future adaptation efforts.

Key Words: climate change adaptation; hydrological restoration; runnels; salt marsh; sea level rise

Implications for Practice

- Runnels are an innovative sea level rise mitigation technique that has shown initial potential to promote plant recolonization within degraded salt marshes, which warrants further study to investigate runnel long-term impacts.

- This study has shown runnels to have positive implications on carbon dioxide removal from the atmosphere through plant recolonization, which has potential for climate change mitigation purposes that can be explore in further studies.
- Studies that evaluate the effectiveness of runnels and other climate change adaptation techniques to combat sea level rise and preserve salt marsh habitat are imperative to influence regulatory policy (permitting) and to promote large-scale implementation to maximize impact.

Introduction

One of the major threats to coastal environments is accelerated relative sea level rise (hereafter referred to as SLR), which is a prevalent impact of climate change (Schuerch et al. 2018). Coastal environments within the Northeastern USA, in particular, are greatly impacted by SLR effects, with New England itself facing SLR rates three to four times the global average (Sallenger et al. 2012; Weston 2014; Carey et al. 2017). SLR can result in increased flooding, decreased resiliency to storms, damage to infrastructure in low-lying and coastal areas, and loss of coastal wetlands (Ashton et al. 2008; Wigand et al. 2017). Wetlands with efficient accretion and sedimentation rates as well as high wetland migration potential can combat SLR impacts (Delaune et al. 1983; Boyd & Sommerfield 2016; Borchert et al. 2018). However, areas with low sedimentation rates, and highly populated coastal areas, may not be able to migrate or accrete at a rate that can withstand SLR, resulting in loss of key wetland ecosystem functions (Weston 2014a).

Due to high productivity and slow decomposition rates, salt marshes serve as important carbon sinks (Reddy & DeLaune 2010). They are also a vital food source, breeding habitat, and nursery ground for birds (including the vulnerable Saltmarsh Sparrow, *Ammodramus* *caudacutus*) and aquaculture and fishery species including fish and shellfish (Raposa & Roman 2006; Hanson & Shriver 2006; Bayard & Elphick 2011). These environments also provide flood abatement and help prevent coastal erosion (Leonard & Luther 1995; Barbier et al. 2011). These functions are essential for regions that rely on aquaculture to support the economy and densely populated coastal areas that benefit from flood abatement and erosion control to prevent damage to infrastructure.

Due to reduced sediment supply caused by coastal development, reforestation, and dam construction, marshes within the Northeast USA, including those in Narragansett Bay, have lower accretion rates than current and projected SLR rates (Sallenger et al. 2012; Weston 2014; Watson et al. 2017). Organic matter supply, a major contributor to New England marsh growth, has also been reduced due to the negative impacts of coastal eutrophication on marsh belowground biomass and organic matter production (Allen 1990; Deegan et al. 2012). Coastal development within Narragansett Bay has also lowered the potential for marsh migration, which is another natural mechanism and response to SLR effects (Roman et al. 2000). Narragansett Bay's low elevation marshes add an additional challenge to combat SLR effects. As a result of these factors, Narragansett Bay marshes are ponding and not fully draining even during low tides, leading to waterlogged soils, dieback and bare areas, and vegetation loss (Alber et al. 2008; Carey et al. 2017; Raposa et al. 2017a). Rhode Island marshes have also experienced changes in vegetation composition, where low marsh plant species, such as *Spartina alterniflora*, are replacing high marsh plant species, such as *Spartina patens* (Raposa et al. 2017a).

Although climate change mitigation research (e.g. efforts to reduce greenhouse gas emissions) serves an important purpose, climate change adaption research is equally needed, as it can focus on preparing for, coping with, and responding to the impacts of current and future system changes (Stein et al. 2013; Wigand et al. 2017). Runnels are a climate change adaptation technique to offset SLR impacts, where shallow excavated channels (runnels) are excavated to drain impounded water (standing water) off the marsh platform that has formed due to SLR impacts and former agricultural features (i.e. ditches). The drainage facilitated by the runnels is expected to alleviate flooding stress and promote vegetation recolonization within die-off areas. Historically, deep ditches have been used for mosquito control and agriculture purposes (Breitfuss & Connolly 2004; Dale & Knight 2006; Dale 2008). However, runnels vary from ditches due to their shallow depths and help to preserve marsh ecological functions in light of climate change. Runnels constructed in this study are shallow (0.15–0.3 m wide by 0.2-0.3 m depth) and strategically placed to drain standing water on the marsh platform. This standing water is a result of accelerated SLR that leads to vegetation die off and displacement of high marsh plants (Raposa et al. 2017). Since runnels are shallower than ditches, it helps to mitigate some of the negative effects (peat oxidation, marsh subsidence) of ditches of greater depths (Dale et al. 1993).

The purpose of this study was to assess the response of salt marsh habitats to excavated runnels; we assessed the vegetation, soil (salinity, belowground biomass, moisture), and greenhouse gas fluxes of treated (runnel) vs. reference (untreated) areas. We expected that runnels would promote plant recolonization in previous areas of shallow impounded water, including increases in belowground biomass, and allow for the persistence of salt marsh plants adjacent to previous areas of standing water. We also hypothesized that the runnels would enhance greenhouse gas removal as the vegetation recolonizes and more CO_2 is removed from the atmosphere through photosynthesis. Our findings suggest a potential climate change adaptation strategy to mitigate SLR impacts.

Methodology

Our field sites were located along fringing salt marshes along the Narrow River in Narragansett, Rhode Island USA, where excavated runnels were created in 2015 were dug by hand using shovels and a low ground pressure excavator by the Rhode Island Department of Environmental Management's Mosquito Abatement Program, U.S. Fish and Wildlife Service and Save The Bay. Save The Bay (STB) conducted pre-restoration monitoring in 2014, including vegetation surveys, before the runnels were installed and a year after the runnels were installed in 2015. As those methods were different from those used in our study to track recovery over time, we used STB results as an estimated comparison but do not include them in statistical analyses.

For our post-manipulation study, we had three treatments: Reference (unaltered), Low Elevation Runnel, and High Elevation Runnel. To measure effectiveness of the runnel treatment, the Reference was in areas not impacted by impounded water to compare the restoration action to healthier marsh conditions. We separated runnels into two different treatments based on their field characteristics and elevation at the start of this study in 2017; the Low Elevation Runnel treatment represented runnel areas that were originally bare, surrounded by unconsolidated sediment, and at a lower elevation, while the High Elevation Runnels treatment represented areas that were at a higher elevation than the Low Elevation Runnel treatment, surrounded by vegetation and more stable sediment. The High Elevation Runnel treatment was at a significantly higher elevation than the Reference and the low elevation runnel treatments ($F_{2,25}$=8.62, p<0.01). Nearby areas that represented healthy marsh were limited, which led to the differences in elevation between the Reference and High Elevation Runnel treatment. Average elevations for treatments collected in 2019 were as follows: 1) Reference: 30.9cm $\pm$ 5.1 (1.0ft NAVD88); 2)

Low Elevation Runnel: 24.2cm ± 0.8 (0.8ft NAVD88); and 3) High Elevation Runnel: 39.6 ± 10.1 (1.3ft NAVD88).

We collected post-manipulation data using two salt marsh sites for each treatment, with one linear transect (12m) per site. Along each transect, we established six circular plots (26cm in diameter), spaced 2m apart. Plot measurements were taken in October of 2017 and August and October of 2018 and 2019; these times represented the peak (August) and end (October) of each salt marsh growing season in New England. We were unable to sample during June or July due to closure of our field sites for Saltmarsh Sparrow nesting. Within each plot, the *S. alterniflora* stems were counted, and the percent cover of each additional plant species was measured. The stem height of 5 haphazardly chosen *S. alterniflora* stems were measured from each plot. The salinity of each plot was measured as well using a porewater sampler and refractometer. Core samples (5cm height, 5 cm diameter) were collected 0.5m outside of the plot to avoid disturbing the plot, and later processed to measure belowground biomass and percent moisture. The core samples were weighed for wet mass, dried at 30^{o}C for 3 days, weighed for dry mass, and then sieved (2mm sieve). The belowground biomass was then determined. Percent moisture was calculated using the wet mass and dry mass of each soil sample.

Greenhouse gas (GHG) fluxes (carbon dioxide and methane) of each plot were measured monthly from August to October of 2017-2019. Measurements were taken between 9:00 AM and 3:00 PM on clear days to optimize on light availability. Carbon dioxide (CO_2) and methane (CH_4) fluxes of each plot were measured using a cavity-ring down spectrometer (CRDS) (Model G2508, Picarro Inc., Santa Clara, California, USA). CRDS analyzer sampling methods were based on methodology described in Martin and Moseman-Valtierra (2015). Before the GHG flux measurements took place, a polyether foam ring was secured over the plot and vegetation (without disturbing the plants). Each time prior to measuring, a transparent polycarbonate chamber (41 cm tall x 27 cm diameter) was placed over the foam ring to create a gas tight seal (as described in Martin and Moseman-Valtierra 2015). The chamber was connected to the CRDS via a vacuum pump and tubing (0.8mm in diameter). Two battery powered fans were installed within the chamber to homogenize the air. Gas measurements were conducted for 4-8min per plot, based on observed periods of linear rates of change (Perry et al. 2020; Yang et al. 2020;). A temperature logger (Hobo, Bourne, MA) was mounted within the chamber, recording the temperature every 10 seconds during this period.

Gas fluxes were calculated from linear rates of change in gas emission concentrations (ppm) over time using the Ideal Gas Law (as described in Martin and Moseman-Valtierra 2015). Positive fluxes were defined as those in which gas concentrations increased over time within the chamber, representing emission from the plot surface to the atmosphere. Negative fluxes are defined as those in which gas concentrations decreased over time, representing net uptake from the atmosphere by the salt marsh plot (Moseman-Valtierra et al. 2016).

We used two-way and repeated measures ANOVAs using JMP v.12 (www.jmp.com) to test for differences among treatments (Reference, Low Elevation Runnel, and High Elevation Runnel) over time (time factor used was month/year combination i.e. October 2018 and October 2019 were compared separately). We used two-way ANOVAs and Tukey HSD to analyze salinity and percent moisture and repeated measures ANOVAs to analyze *S. alterniflora* stem density, species percent cover, belowground biomass, and gas fluxes. Data were tested for normality and homogeneity of variance and were transformed when appropriate (Underwood 1997).

Results

Pre/Post treatment community composition

The data collected in 2014 represents cumulative data of the runnel area prior to restoration (both Low Elevation and High Elevation Runnel areas; pre-restoration Figure 1) including control sites located within areas of impounded water (control area in 2014 differed from Reference site studied from 2017-2019), while the data described from 2018 and 2019 represents a subset of the runnel area surveyed in 2014 specific to the Low Elevation and High Elevation Runnels conditions described in the Methodology section. Save the Bay data showed that open water within the runnel areas decreased from 27% (prior) to 0% after the runnels were implemented (2015), while the open water in the control area remained consistent (Table 1). The *S. alterniflora* percent cover of the Low Elevation Runnel area was approximately ~37% higher in August 2018 and 2019 than the runnel area in 2014 (Figure 1). The *S. patens* percent cover in the High Elevation Runnel area was 49% and 54% higher in August 2018 and 2019, respectively, than the runnel area in 2014 (Figure 2).

Post treatment vegetation

There was a trend of decrease in bare areas over time for the Low Elevation Runnel area, while the percent of bare area in the Reference remained consistent (Figure 3). However, there was not a significant treatment effect (overall model $F_{46,71} = 4.05$, $p<0.0001$; treatment: $F_{2,71} = 3.02$, $p=0.057$; time: $F_{4,71}= 10.95$, $p<0.0001$; treatment*time: $F_{8,71} = 1.78$, $p=0.096$).

There was a mean increase in *S. alterniflora* stem density in the Low Elevation Runnel and Reference areas, 4x and 1.9x in October 2018 and 2.2x and 1.3x in October 2019, respectively, compared to October 2017 (Figure 4). By contrast, *S. alterniflora* stem density was consistent over time in the High Elevation Runnel area (overall model $F_{47,104} = 6.01$, $p<0.0001$; treatment: $F_{2,104} = 27.84$, $p<0.0001$; time: $F_{4,104}= 18.02$, $p<0.0001$; treatment*time: $F_{8,104} = 4.16$,

$p<0.01$). There was no significant difference in *S. alterniflora* stem height among treatments, but there was an interaction effect (treatment: $F_{2,100} = 0.19$, $p = 0.83$; time $F_{4,100} = 4.36$, $p<0.01$; treatment*time: $F_{8,100} = 7.70$, $p<0.01$).

In the Low Elevation Runnel area, *S. alterniflora* percent cover was close to zero in October 2017 then reached 40% cover (~237x higher) in October 2019 (Figure 1). The *S. alterniflora* percent cover in the Reference area was ~2x greater in October 2019 than October 2017, while the High Elevation Runnel did not show a distinct change between October 2017 and October 2019 (overall model $F_{47,97} = 6.85$, $p<0.0001$; treatment: $F_{2,97} = 28.72$, $p<0.0001$; time $F_{4,97} = 38.02$, $p<0.0001$; treatment*time: $F_{8,97} = 4.36$, $p<0.01$).

S. patens was not present in the Low Elevation Runnel areas from 2017-2019, while it declined in the Reference from 8% to 0% from 2017-2019 (Figure 2). By contrast, the percent cover of *S. patens* was over eleven times higher in the High Elevation Runnel than in the other treatments (overall model $F_{47,98} = 6.41$, $p<0.0001$; treatment: $F_{2,98} = 19.45$, $p<0.0001$; time: $F_{4,98} = 0.34$, $p = 0.85$; treatment*time: $F_{8,98} = 1.09$, $p=0.38$).

Belowground Biomass (BGB)

Belowground biomass was two times greater in the Reference than the Low Elevation Runnel and High Elevation Runnel areas (Figure 5; overall model $F_{38,78}= 4.03$, $p<0.0001$; treatment: $F_{2,78} = 55.73$, $p<0.0001$; time: $F_{3,78} = 6.97$, $p<0.01$; treatment*time: $F_{6,78} = 1.15$, $p=0.34$).

Abiotic Soil Factors: Salinity and percent moisture

The Reference had significantly higher salinity, 29% and 25%, than Low Elevation Runnel and High Elevation Runnel areas, respectively (Table 2; overall model $F_{17,159} = 15.41$, $p<0.0001$; treatment: $F_{2,159} = 68.57$, $p<0.0001$; time: $F_{5,159} = 14.44$, $p<0.0001$; treatment*time:

$F_{10,159}$ = 3.53, p <0.01). By contrast, the Low Elevation Runnel had significantly higher moisture than all other treatments while the Reference had significantly higher moisture than the High Elevation Runnel (Table 3; overall model $F_{11,105}$ = 4.58, p<0.0001; treatment: $F_{2,105}$ = 22.06, p<0.0001; time: $F_{3,105}$= 0.29, p=0.83; treatment*time: $F_{6,105}$ = 0.87, p = 0.52).

Greenhouse gas fluxes

Uptake of carbon dioxide was twice as high in the Reference than the other treatments (Figure 6A; overall model $F_{63,146}$= 10.32, p<0.0001; treatment: $F_{1,146}$ = 14.03, p < 0.01; time: $F_{7,146}$= 43.35, p<0.0001; treatment*time: $F_{15,146}$ = 9.72, p <0.0001). The Low Elevation Runnel showed more CO_2 uptake in 2018 and 2019 than in 2017 (Figure 6A). There was a significant difference in methane flux among treatments (Figure 6B; overall model: $F_{64,159}$ = 1.50, p =0.02; treatment: $F_{1,159}$= 4.03, p=0.046; time: $F_{7,159}$= 2.86, p<0.01; treatment*time: $F_{15,159}$ = 0.60, p=0.87).

Discussion

Initial runnel impacts were apparent, as the standing water drained after the runnels were implemented. The Low Elevation Runnel and High Elevation Runnel were expected to differ from each other in vegetation response, due to the initial vegetation composition recorded in 2017 as well as the difference in elevation. Since the Low Elevation Runnel was predominantly bare in 2017, we expected that *S. alterniflora* would be an initial colonizer, which was depicted in the results. The *S. alterniflora* percent cover increased in the Reference and the Low Elevation Runnel treatments from October 2017 to October 2018, but the rate of increase was ~17x greater in the Low Elevation Runnel than the Reference, suggesting a drainage impact. Further supporting the positive impact of the runnels, the Low Elevation Runnel had twice the stem density increase than the Reference in October 2018 and 2019, compared to October 2017. Plant recolonization occurred after 3 growing seasons after the runnels were installed, but this result may vary per project based on site conditions and initial levels of degradation.

A prevalent issue within New England salt marshes is that low marsh species are displacing high marsh plants due to accelerated sea level rise impacts (Donnelly & Bertness 2001; Watson et al. 2016; Raposa et al. 2017). This pattern is seen in the Reference, as *S. patens* gradually decreased from 2017-2019, as well as a decrease from initial conditions in *S. patens* coverage in 2014. Raposa et al. (2015) reports the displacement of *S. patens* in the high marsh by *S. alterniflora* throughout Rhode Island, with shifts seen within a 4-year time period, due to SLR and explains that during the current period of rapidly accelerating sea level rise this effect is exacerbated. However, our High Elevation Runnel results showed the persistence of *S. patens* over the 3-year period without evidence of *S. alterniflora* increase. These results are suggesting runnel effects, in combination with higher elevations, can potentially help prevent *S. patens* displacement by *S. alterniflor*a. If this pattern persists, this runnel impact could have positive implications on the vulnerable bird species, *Ammodramus caudacutus* or saltmarsh sparrow, that nests within *S. patens* habitat (Bayard & Elphick 2011). Without intervention, the impacts of accelerated SLR and excess tidal inundation may result in the continued decline in high marsh plants, suggested in Raposa et al. (2015), which will be detrimental to the saltmarsh sparrow and other bird and invertebrate species that rely on high marsh habitat for survival (Bayard & Elphick 2011; Zajac et al. 2017).

In the past, hydrological manipulations (berm creation, culverts, and dikes) within salt marshes have been used to promote *S. patens* growth, mostly for agricultural purposes (Britton 1912; Smith & Bridges 1982; Sebold 1998). However, some of these methods have limited sedimentation, promoted the spread of the invasive species *Phragmites australis* within New

England marshes, and altered biogeochemical processes including changes in soil chemistry often cause by increased oxygen presence, tidal restrictions, and salt marsh drainage (Crain et al. 2009; Tonjes 2013). These causes can lead to salt marsh subsidence due to more rapid soil decomposition rates with higher soil oxygen content (Vincent et al. 2013). Therefore, other methods are needed to preserve high marsh habitat that does not lead to adverse effects. In this study, hydrological manipulation through the use of runnels demonstrated the potential to combat SLR impacts and preserve high marsh habitat, but the permanence of these impacts, considering accelerating SLR rates, needs further study.

The runnels successfully drained standing water, which led to an initial increase in bare areas that were higher than the Reference that represented areas of standing water in 2015. We saw a trend of decrease in bare areas in the Low Elevation Runnel treatment, but further monitoring is warranted to determine if this trend becomes significant. The vegetation growth response of the Low Elevation Runnel treatment shows positive implications on recolonization, which suggests a continued decrease in bare areas over time.

In a drainage enhancement study, Raposa et al. (2019) used studied creeks that were wider (~1.3-3.3m in width and ~0.5m in depth) than the runnels used in our experiment and found no significant changes in pre-existing vegetation in control or the creek areas at the marsh-wide scale. However, on a smaller-scale, areas that were initially bare completely recolonized after 3 years, which is similar to the results of the Low Elevation Runnel in our study. These similar results highlight the effectiveness of drainage enhancement on a smaller-marsh scale, but runnels rather than creeks can be a more effective method since the creeks lead to elevation decline. Raposa et al. (2019) found that elevation declined in areas of creeks of high widths and depths, potentially due to the introduction of oxygen in marsh peat enhancing decomposition and marsh subsidence, and although drainage was enhanced the loss of elevation capitol prevented changes in pre-existing low marsh vegetation or colonization of high marsh plants (Watson et al. 2016; Burdick et al. 2020). The use of shallow runnels provides the benefits of enhanced drainage without the risk of elevation decline seen in creeks or ditches of greater depths and promotes positive impacts on vegetation recolonization. There are limited studies on the runnel technique. However, Dale (2008) in Australia found positive impacts on vegetation of a runnel study over a 20 year period, the density and size of dominant salt marsh plant species were higher in runnel areas than the control, and that previously dominant species had disappeared in the control and replaced by more flood-tolerant species.

Our study found an increase of belowground biomass in the Low Elevation Runnel and decline in the Reference when comparing August 2018 to August 2019, but a longer period of study is needed to determine if this trend continues and to determine the overall effects of runnels on soil carbon stocks. The Reference represented higher overall belowground biomass than the other treatments as expected since this treatment was chosen to represent healthy marsh conditions to monitor the effectiveness of the restoration treatment to improve habitat quality. The Low Elevation Runnel showed an increase in belowground biomass over 3 years after restoration, but is not yet representing marsh conditions as seen in the Reference. Although the Low Elevation Runnel is showing improvements over time, the results suggest that more time is needed before reaching fully restored conditions. There was a non-significant trend in the High Elevation Runnel belowground biomass, which may be due to the decrease in *S. alterniflora* stems and prominence of *S. patens*, as *S. patens* has finer roots than *S. alterniflora* (Moseman-Valtierra et al. 2016). This could also explain the significantly lower belowground biomass in the High Elevation Runnel compared to the Reference, due to the difference in dominant vegetation.

One of the goals of climate change adaptation is to preserve coastal wetlands ecosystem functions for environmental benefits (Barbier et al. 2011; Crosby et al. 2016; Wigand et al. 2017). Greenhouse gas fluxes can assess the capability of these adaptation strategies to maintain an important salt marsh ecosystem function of carbon sequestration (McLeod et al. 2011; Moseman-Valtierra 2013; Martin & Moseman-Valtierra 2015). The Low Elevation Runnel had higher carbon dioxide uptake within the growing season from August-October in 2018 and 2019 than in 2017, which suggests an improvement in this ecological function in coordination with vegetation recovery. The Reference showed the highest rates of CO_2 uptake in 2018 likely due to the higher *S. alterniflora* stem count than the other treatments, which was expected as the Reference was used to represent healthier marsh conditions. Methane was higher in the Reference likely due to the higher belowground biomass, indicating organic material present to support methanogenesis (Reddy et al. 2000). Although the Low Elevation Runnel conditions have not reached the higher quality conditions of the Reference, this treatment has shown improvements over time seen in the increase in *S. alterniflora* stem density and percent cover as well as the increase in CO_2 uptake, which shows the positive impacts of drainage provided by the runnels. Further monitoring is warranted to see if conditions continue to improve and reach standards of the Reference.

Due to the higher elevation of the High Elevation Runnel compared to the Reference and Low Elevation Runnel, we expected to see the differences in vegetation and CO_2 uptake. Moseman-Valtierra et al. (2016) showed that *S. alterniflora* dominated areas removed CO_2 from the atmosphere at a faster rate than *S. patens* dominated areas. Due to this, it was not unexpected that the CO_2 fluxes in the High Elevation Runnel are less than the other treatments since the High Elevation Runnel had pre-dominantly *S. patens* while Low Elevation and Reference was predominantly *S. alterniflora*. Carbon dioxide uptake was lower in all treatments in 2019 than 2018, which suggests that abiotic factors rather than treatment were having the greatest impact on CO_2 fluxes during that period of measurements (Portnoy & Valiela 1997; Wilson et al. 2015; Martin & Moseman-Valtierra 2017).

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Table 1. Mean percent cover ±1 SE of salt marsh habitat types before (2014) and one year after (2015) runnels were installed. This represents Save The Bay data.

Habitat Composition	Control areas (2014)	Runnel areas (2014)	Control Area (2015)	Runnel Area (2015)
Open water	13.56 ± 0.14	27.0 ± 9.11	13.6 ± 3.61	0 ± 0
Spartina alterniflora	53.3 ± 2.91	39.6 ± 0.71	52.5 ± 0.17	43 ± 4.16
Spartina patens	26.0 ± 1.95	21.0 ± 0.61	26.2 ± 0.47	20.8 ± 3.15
Other Vegetation	12.89 ± 5.99	8.6 ± 0.58	10.87 ± 13.29	13.56 ± 8.15
Bare	2.0 ± 1	12.08 ± 1.81	3.43 ± 0.22	21.9 ± 8.49

Table 2. Mean salinity (psu) per treatment ± 1 standard error.

Reference	Low Elevation Runnel	High Elevation Runnel	Month and Year
35.5 ± 2.8	22.8 ± 4.0	26.6 ± 3.3	August 2018
29.1 ± 3.9	21.5 ± 5.5	23.6 ± 3.9	October 2018
31.9 ± 0.9	27.3 ± 1.3	25.7 ± 0.9	August 2019
24.8 ± 2.2	22.4 ± 3.1	21.7 ± 2.2	October 2019

Table 3. Mean percent moisture per treatment ± 1 standard error.

Reference	Low Elevation Runnel	High Elevation Runnel	Month and Year
85.0 ± 0.7	87.8 ± 1.0	84.2 ± 0.8	August 2018
85.3 ± 0.6	88.4 ± 0.9	84.4 ± 0.6	October 2018
85.8 ± 0.9	88.1 ± 1.2	84.6 ± 0.9	August 2019
86.3 ± 0.5	86.6 ± 0.7	84.8 ± 0.5	October 2019

Figure 1. Mean *Spartina alterniflora* percent cover + 1 SE (per 20cm diameter plot). Pre-Restoration represents the entire runnel area before the runnels were constructed.

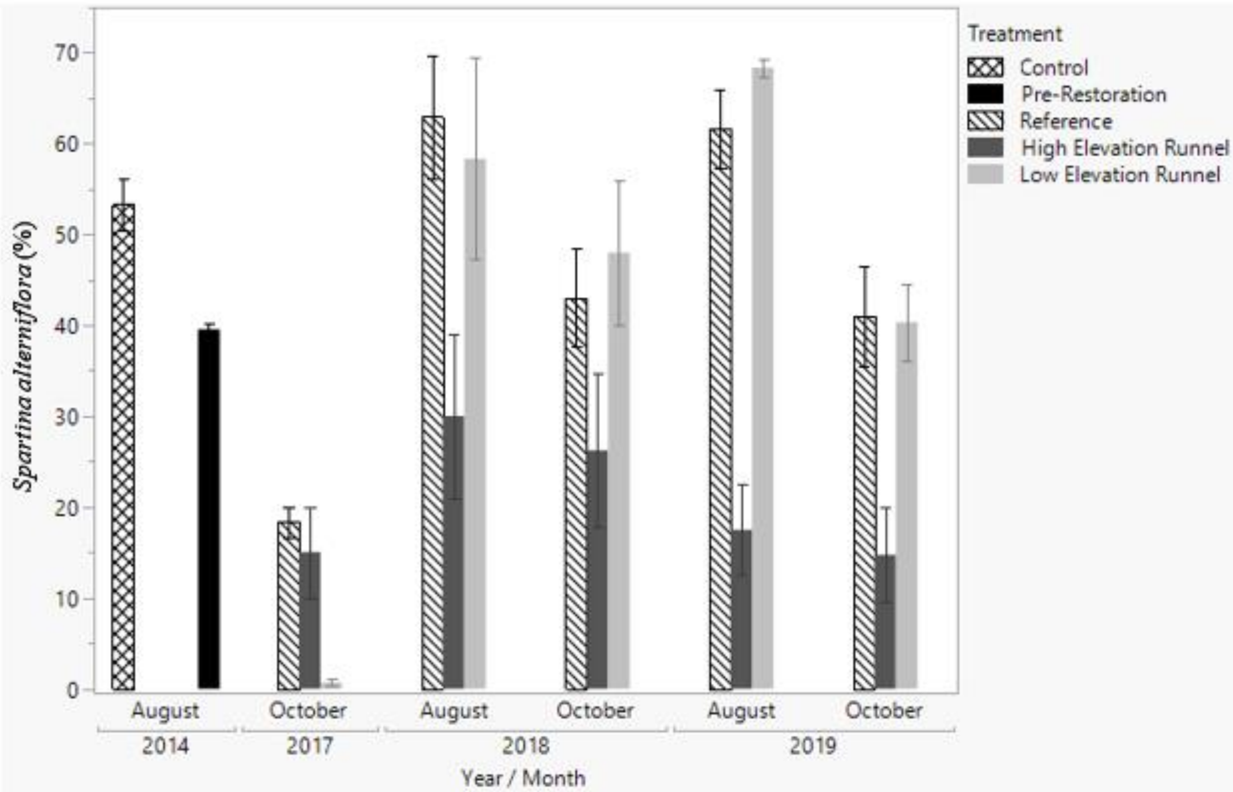

Figure 2. Mean *Spartina patens* percent cover + 1 SE (per 26cm diameter plot). Pre-Restoration represents the entire runnel area before the runnels were constructed.

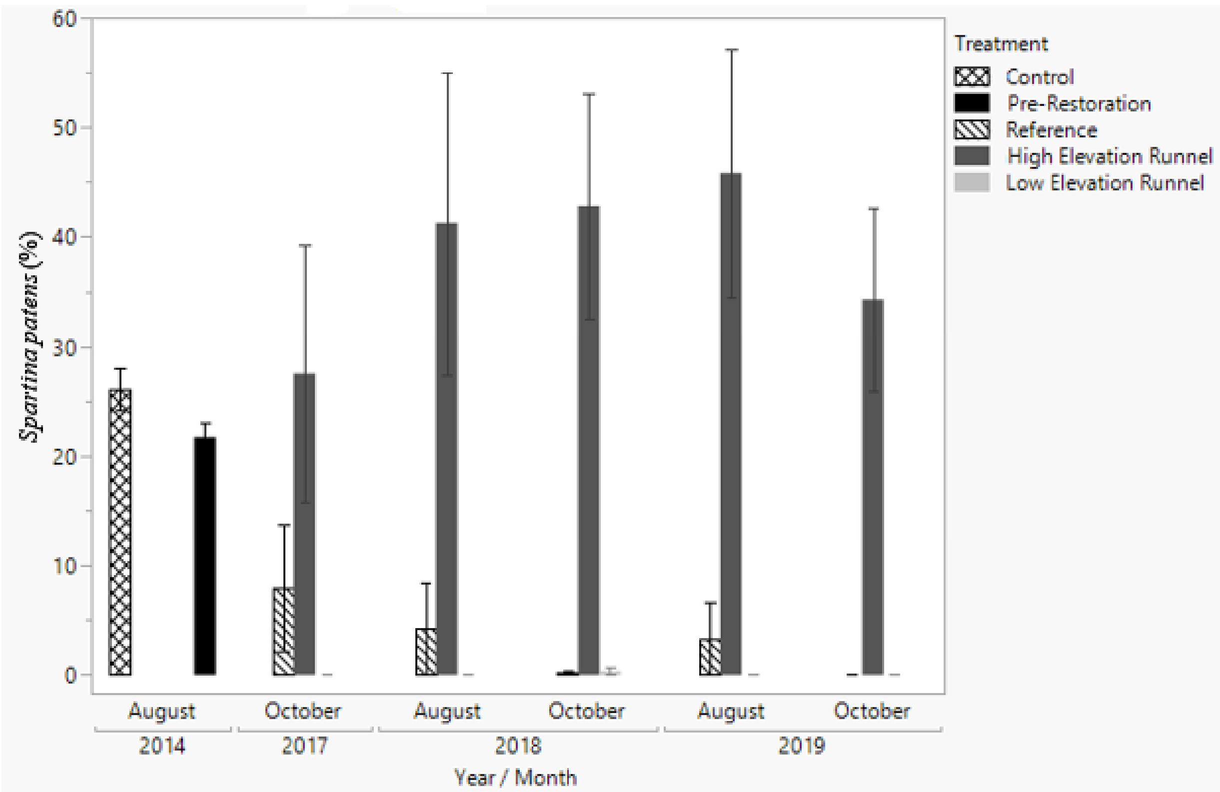

Figure 3. Mean bare percent cover + 1 SE (per 20cm diameter plot).

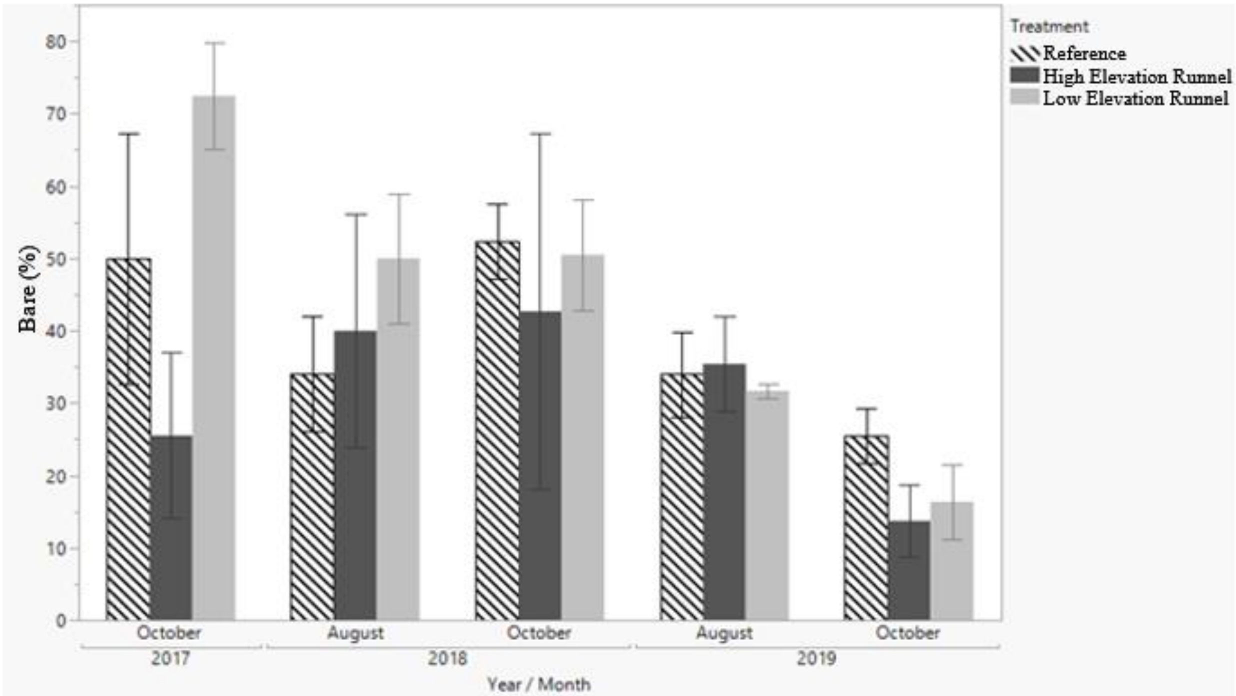

Figure 4. Mean *Spartina alterniflora* stem density ± 1 SE.

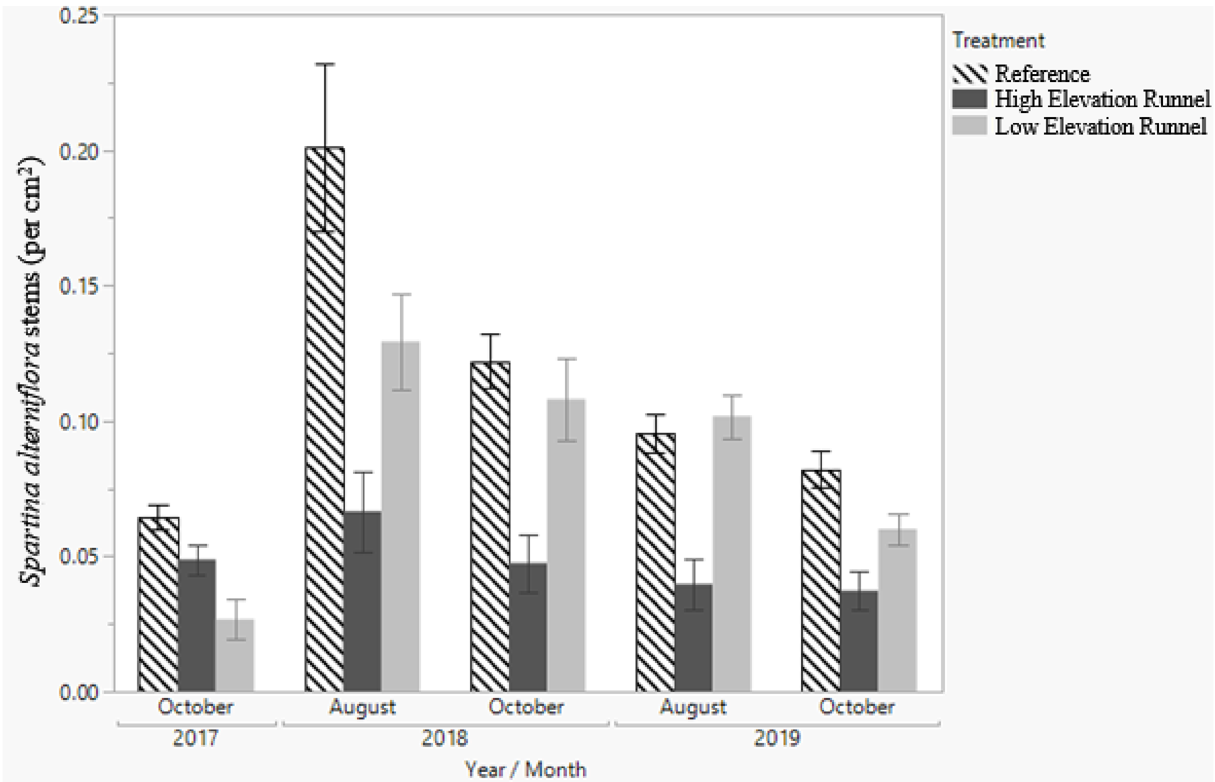

Figure 5. Mean belowground biomass ± 1 SE.

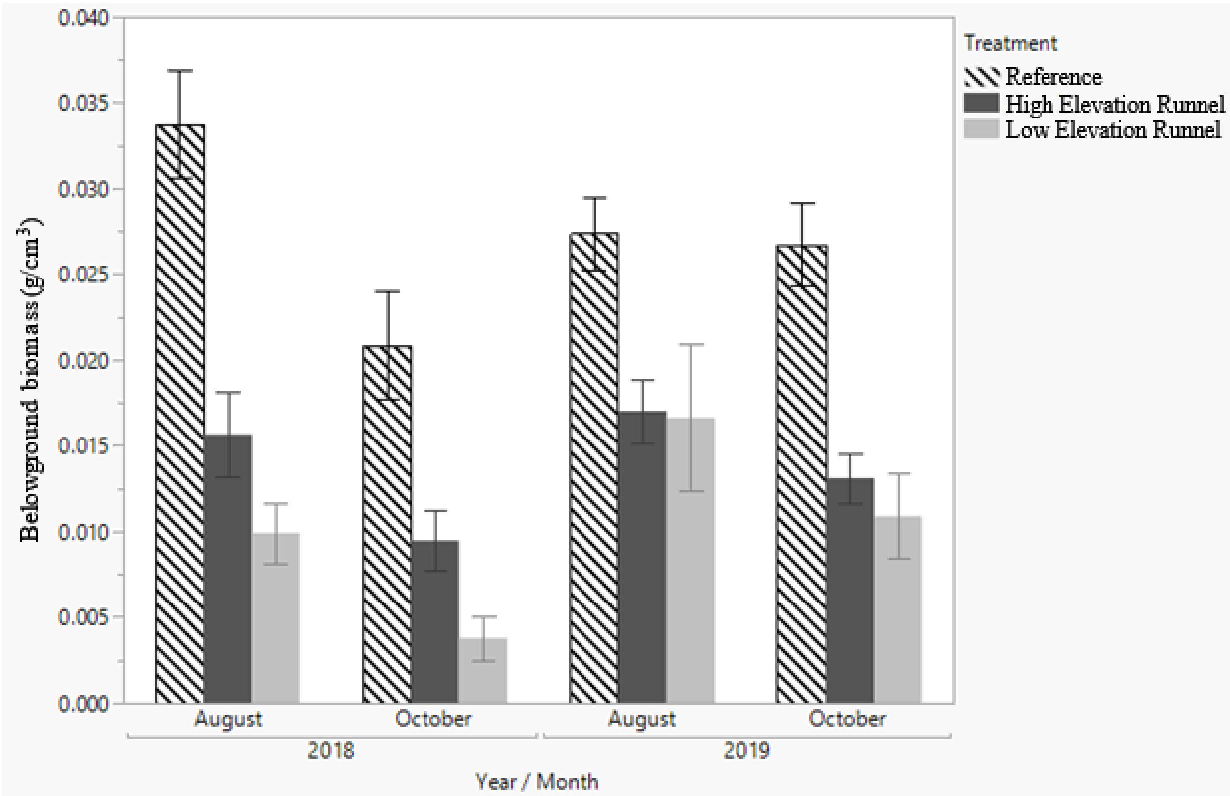

Figure 6. Mean carbon dioxide (A) and methane (B) flux ± 1 SE.