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SALT MARSH MANAGEMENT: MONITORING THREATS AND IMPLEMENTING CLIMATE ADAPTATION INITIATIVES

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SALT MARSH MANAGEMENT: MONITORING THREATS
AND IMPLEMENTING CLIMATE ADAPTATION
INITIATIVES

BY
DANIELLE C. PERRY

A DISSERTATION SUBMITTED IN PARTIAL FULFILLMENT OF THE
REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY
IN
BIOLOGICAL AND ENVIRONMENTAL SCIENCES

UNIVERSITY OF RHODE ISLAND

2020

DOCTOR OF PHILOSOPHY IN BIOLOGICAL AND ENVIRONMENTAL
SCIENCES DISSERTATION

OF
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2020

ABSTRACT

Salt marshes serve as important coastal resources both economically and environmentally and are particularly susceptible to the negative implications of climate change. As climate change impacts become more prevalent, it is important to understand how salt marshes will respond. Two climate change impacts on salt marshes are macroalgae inundation and accelerated relative sea-level rise (hereafter referred to as SLR). Due to warming temperatures and sea level rise, salt marshes can be inundated by macroalgae through tidal movement, and the impacts of this excess macroalgal accumulation are poorly understood. Sea level rise also has negative implications on salt marsh by leading to vegetation die-back and ultimately salt marsh loss. Coastal managers within Rhode Island have successfully collaborated to implement climate change adaptation projects through strategic planning and communication. This dissertation highlights the important steps of salt marsh management: 1) Identifying the impacts of potential salt marsh threats (Chapter 1), 2) Monitoring salt marsh response to climate change adaptation projects (Chapter 2), and 3) Describing the important steps to plan and implement climate change adaptation projects (Chapter 3).

Chapter 1 focuses on the impacts of macroalgal densities within Rhode Island on salt marsh environments. In this study, we investigated how current, relatively low density ephemeral (*Ulva* spp.) and perennial (*Fucus* spp.) algal wrack coverage impacts *Spartina alterniflora* (low marsh plant) survival and associated greenhouse gas fluxes. We created mesocosms with *S. alterniflora*-vegetated soil cores using a 2 X 2 factorial design with *Fucus* and *Ulva* present or absent to test the effects of

macroalgal inundation. We found that *S. alterniflora* was resilient to these densities of macroalgal coverage, and this coverage did not significantly impact greenhouse gas fluxes.

In Chapter 2, we investigated the impacts of two climate change adaptation projects, hydrological restoration (dredged runnels) and sediment enhancement, that have been implemented in Rhode Island to combat the effects of sea level rise within salt marshes. Here, we investigated the impacts of these adaptation techniques on soil properties, vegetation composition, and greenhouse gas fluxes (methane and carbon dioxide). We found that plant density in certain runnel areas increased after runnels were installed and runnels assisted in maintaining *Spartina patens* (high marsh plant) coverage. Carbon dioxide uptake increased in a runnel treatment and control as vegetation increased over time. Sediment enhancement treatment did not lead to vegetation or belowground biomass recolonization and produced insignificant greenhouse gas fluxes.

Chapter 3 focuses on the incorporation of adaptive management into climate change adaptation implementation. This chapter describes the techniques and mechanisms used by the Rhode Island Coastal Resources Management Council to implement an adaptive management approach to restore a drowning salt marsh using the climate adaptation strategy, sediment enhancement. Through effective communication and active stakeholder involvement, this project successfully incorporated interdisciplinary partner and stakeholder collaboration and developed an iterative learning strategy that highlights the adaptive management method.

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PREFACE

This dissertation is being submitted in manuscript format. It is comprised of three chapters each of which have been submitted, or are in preparation of submission, for peer-reviewed publication. Chapter one, “Greenhouse gas response and *Spartina alterniflora* resilience to macroalgal exposure,” has been published in the journal, *Aquatic Botany*. Chapter two, “Monitoring salt marsh vegetation and soil response to climate change adaptation” is in preparation for submission to the journal, *Biological Conservation*. Chapter three, “Successful implementation of adaptive management into a climate change adaptation strategy,” has been accepted pending revisions to the *Journal of Environmental Management*.

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CHAPTER 1

GREENHOUSE GAS RESPONSE AND *SPARTINA ALTERNIFLORA* RESILIENCE TO MACROALGAL EXPOSURE

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Abstract

Climate change can alter salt marsh plant communities and their associated carbon flux dynamics via several mechanisms. Due to warming waters and sea level rise, macroalgal wrack accumulation rates in salt marshes are expected to increase. The smothering and shading effects of macroalgal wrack can have adverse effects on salt marsh vegetation. Most studies have focused on the impacts of high density accumulation or future accumulation projections, but the impacts of current accumulations of macroalgae on *Spartina alterniflora* (*S. alterniflora*) are unclear. We investigated how current, relatively low density ephemeral (*Ulva* spp.) and perennial (*Fucus* spp.) algal wrack coverage impacts *S. alterniflora* survival and associated greenhouse gas fluxes. We created mesocosms with *S. alterniflora*-vegetated soil cores using a 2 X 2 factorial design with *Fucus* and *Ulva* present or absent. After one month, *S. alterniflora* cores with *Ulva* (but no *Fucus*) had a significantly lower stem density, but they later recovered, attesting to *S. alterniflora*'s resilience to low density macroalgal coverage. Our results also suggest that *Fucus* can alter the interaction between *Ulva* and *S. alterniflora* by potentially mitigating *Ulva*'s impacts. Macroalgal presence did not significantly influence greenhouse gas fluxes, which suggests the low density exposure is not altering the salt marsh carbon flux dynamics.

Introduction

Salt marsh habitats serve as carbon sinks due to slow soil decomposition processes and carbon dioxide uptake by vegetation (Reddy and DeLaune 2008). However, they are increasingly threatened due to anthropogenic factors including climate change (causing rapid sea level rise, increased water temperatures, changing precipitation patterns, and other factors), eutrophication, and coastal development (McLeod et al., 2011; Pennings et al., 2002; Watson et al., 2016, 2014). Marsh loss can reduce storm protection, nutrient sequestration and filtration, water quality maintenance, as well as habitat for fish, shellfish, and wildlife (van de Koppel et al., 2005; Valiela and Fox 2008; McLeod et al., 2011; Wigand et al., 2017).

Macroalgae have an under-recognized but significant presence in salt marshes worldwide. Brown macroalgae, such as the fucoids *Fucus* and *Ascophyllum*, are perennial, can originate in salt marshes, and are often layered on the sediment surface or attached to hard substrate. They are slow-growing, branched algae that can survive for several years (Tyrrell et al., 2015). Overall, perennial fucoids play a beneficial role in salt marsh environments by increasing sedimentation and accretion rates, although they can inhibit *Spartina alterniflora* (*S. alterniflora*) seedling colonization (Tyrrell et al. 2015).

In contrast, bloom-forming algae are ephemeral, and commonly occur (mostly in the summer months) in salt marsh environments, with shallow, calm waters and high nutrient inputs (Newton and Thornber 2012; Lyons et al. 2014). They also serve as a food source and shelter for salt marsh invertebrate species, and release nutrients upon decomposition into the salt marsh environment. Blooms typically consist of

‘opportunistic,’ ephemeral macroalgal genera with high nutrient uptake rates and rapid growth and decomposition rates, such as *Ulva*, *Ceramium*, and *Gracilaria* (Peckol and Rivers, 1995; Scanlan et al., 2007). These algae take advantage of seasonally warm temperatures and light that allow for rapid growth (Luo et al. 2012). As a result, bloom-forming algae are prolific during the summer months, which is enhanced by coastal eutrophication. Large accumulations of bloom-forming algae often inundate salt marshes (via tidal action) creating wrack disturbance and shading on salt marsh vegetation (Wasson et al., 2017). Due to their differences in life span and morphology, we predicted that the environmental impacts of bloom-forming algae and fucoids on salt marsh plants will vary.

As macroalgal blooms have become more abundant during the summer months, macroalgal wrack accumulation has also become a prevalent problem within salt marsh environments (Newton and Thornber 2012; Wasson et al. 2017). Previous studies have shown that plant wrack debris, formed from species such as *S. alterniflora*, *Phragmites australis*, and seagrasses, can decrease *S. alterniflora* coverage and stem height and increase bare areas within salt marshes (Byer et al., 2004; Hartman et al., 1983; Macreadie et al., 2013). Wrack deposition has led to the formation of salt marsh pannes, with less dense vegetation coverage and plants with stunted growth (Hartman et al. 1983). Wasson et al. (2017) showed that coverage by dense *Ulva* mats lead to the decline in *Salicornia* stem density and formation of bare salt marsh areas, erosion, and the landward retreat of salt marsh edge boundaries. Within New England salt marshes, macroalgal mats can hinder the growth of *S. alterniflora* (Newton and Thornber 2012; Ober and Martin 2018). High *Ulva*

abundance has also led to reductions in *S. alterniflora* above and belowground biomass and higher porewater sulfide concentrations and toxic ammonium levels in salt marsh soils (Watson et al., 2015). These accumulations often occur closest to creek banks and the water's edge, but as sea level rise occurs, macroalgae are expected to be transported further into the marsh (Wasson et al. 2017; Ober and Martin 2018). Most studies have investigated the impacts of high-density accumulation or future macroalgal projections, but the impacts of current macroalgal conditions on salt marsh vegetation are unclear.

In this study, we examined the impacts of current marsh densities of perennial brown algae, *Fucus* spp., and the ephemeral bloom-forming algae, *Ulva* spp. (Newton and Thornber 2012), on the growth, stem density, and nutrient content of vegetated cores of *S. alterniflora*, the dominant low marsh plant on the Atlantic coast. We also measured the carbon dioxide and methane fluxes of the vegetated soil cores to determine the impacts of macroalgal addition on greenhouse gas fluxes. We expect that *Ulva*, due to blade-morphology, would negatively impact stem density via smothering, but cause elevated *S. alterniflora* stem nitrogen content due to its rapid decomposition. By contrast, we did not expect significant impacts of *Fucus* on stem density or stem nitrogen content, due to its branched morphology, slow growth rate, and slow decomposition rate. By focusing on current macroalgal accumulation densities, we hope to understand the threshold of *S. alterniflora* resilience to macroalgal coverage. We interpret our results in light of a changing climate and discuss potential roles of changes in algal abundances in salt marsh decline.

Methods

MESOCOSOM DESIGN

In June of 2016, we transplanted twenty-five *S. alterniflora*-vegetated soil cores (20cm diameter, 20cm height) from Fox Hill Salt Marsh in Jamestown, Rhode Island to the outdoor Marine Science Research Facility at the University of Rhode Island (URI)'s Narragansett Bay Campus in Narragansett, RI. A mesocosm consisted of an individual core that was placed in an individual 125 liter translucent rectangular bin (Fig. 1a, b). The mesocosms were transplanted to an open outdoor area, with complete exposure to natural outdoor conditions (e.g. ambient temperature, irradiance). During the experiment, the mean air temperature was 23° C (daily minimum range 6° C to 17° C, maximum range 25° C to 36° C).

To simulate natural tidal cycles, raw, unfiltered ambient temperature seawater (nutrient concentrations listed in Table 1) was pumped into the mesocosms via a manifold (5cm in diameter PVC pipes). Salinity of the seawater was measured weekly and ranged from 30.8 to 32.1psu. Water flow was regulated by a solenoid valve (WIC 3/4" anti-corrosion salt water solenoid valve NC) and timer (GE 24-hr timer), allowing for alternating four-hour periods of inundation and draining. Six-hour tidal cycles, the semidiurnal pattern in the Northeast USA, were unattainable due to mechanical limitations in our system. To simulation natural tide conditions, the *S. alterniflora* stems were completely submerged at the peak of the four-hour tidal cycle (high tide) and completely drained at the end of the four-hour tidal cycle (low tide). This was regulated by the high-tide and low-tide outflow valves created for each mesocosm.

EXPERIMENTAL DESIGN

The experiment took place from June –September of 2016 during the active salt marsh growing season. The macroalgal treatment densities within this experiment represented mean low-density accumulation (recorded here as wet mass) within Rhode Island salt marshes, which refers to accumulations that occur outside of the peak growing season of July and August; high density accumulations (occurring within July and August) were 420 g/m² (Newton and Thornber 2012; Newton and Thornber 2013). Mesocosms (n= 5 for each treatment) were set up in a 2 X 2 factorial design: 0 or 210 g/m² of fucoids added and 0 or 210 g/m² of *Ulva*. An additional five mesocosms consisting of 210 g/m² of both fucoids and *Ulva* were added to investigate the interaction effect of these algal types on *S. alterniflora*. To test for the effects of total biomass vs. species composition, an additional five mesocosms were set up consisting of 105 g/m² of both fucoids and *Ulva*.

Since the species composition of *Fucus* and *Ulva* wrack is variable within salt marsh environments (*Fucus distichus*, *Fucus spiralis*, *Fucus vesiculosus*, *Ulva australis*, *Ulva compressa*, *Ulva laetevirens*, and *Ulva rigida*), we used a mix of fucoid and/or *Ulva* species to best represent typical wrack cover in New England marshes (Newton and Thornber 2012). In this context, we define wrack as macroalgal deposits transported by tidal inundation (Newton and Thornber 2012; Wasson et al. 2017). In relation to our cores, the 210 g/m² corresponded to 6g of added macroalgae per 20cm diameter core and 105 g/m² corresponded to 3g of macroalgal addition. The treatment with no algal addition will hereafter be referred as the control, 210 g/m² of *Fucus* addition as the *Fucus* 6 treatment, 210 g/m² of *Ulva* addition as the *Ulva* 6 treatment, 210 g/m² *Fucus* plus 210 g/m² *Ulva* addition as the Mix 6 treatment, and

105 g/m² *Fucus* plus 105 g/m² *Ulva* addition as the Mix 3 treatment. The treatments were placed in a row in the repetitive order of control, *Fucus* 6 treatment, *Ulva* 6 treatment, Mix 3 treatment, and Mix 6 treatment. All measurements were taken in this order.

Every two weeks for the duration of the experiment, to account for rapid *Ulva* decomposition and to simulate macroalgal *Ulva* spp. bloom deposition, *Ulva* (105 or 210 g/m² depending upon the treatment) was applied to the mesocosms (Conover et al., 2016). For all fucoid treatments, mesocosms containing fucoids were checked every two weeks to ensure the allotted amount remained on the cores, and additional fucoids were added as needed to maintain treatment densities. The macroalgae were spread at the base and top of the *S. alterniflora* stems to simulate natural conditions in the field, since natural tidal deposition is not uniform. Although the consistency of algal coverage can be variable in a marsh environment, Wasson et al (2017) found that macroalgal coverage persisted over multiple months. Thus, we placed a mesh (netting mesh ~4mm) around each core, including controls, to prevent algae from leaving the mesocosm during water drainage.

STEM DENSITY AND GROWTH MEASUREMENTS

Each month, we measured the mean shoot height (taken from 3 random placements within each mesocosm) and stem density (total number of shoots per mesocosm). We documented the percent change from initial height measurements to represent stem growth as well as the percent change in initial stem density to represent stem density changes over time. We used the following formula to calculate percent change from initial measure:

$\{ \text{Monthly stem measure (height or density)} / \text{Initial stem measure (height or density)} - 1 \} * 100$

At the end of the experiment, the belowground biomass was extracted from each core. The belowground biomass was washed of sediment, dried at 30°C for 3 days, and then weighed.

NUTRIENT ANALYSIS: STEM TISSUE AND POREWATER

Each month, we measured % carbon, hydrogen, and nitrogen (CHN) content of *S. alterniflora* shoot tissues. For CHN analysis, a stem tissue sample (a piece of a plant stem from three haphazardly selected *S. alterniflora* stems) was removed per mesocosm (~8mg dry weight per treatment), dried at 30°C for 3 days, ground with a mortar and pestle, and then analyzed via mass spectrometry (Elemental Combustion System CHNS-O, ESC 4010 Model NC2100). The stem pieces used for the CHN analysis were removed of algae before processing.

A 20mL porewater sample was collected from each core using Rhizon samplers (<https://www.rhizosphere.com/rhizons>) and filtered with 0.2 µm PES filters (EMD Millipore™ Millex™ Nonsterile Syringe Filters with PE Housing-PES, EMD Millipore™ SLGP033NS). After filtration, the samples were placed into a freezer before nutrient analysis (completed by the URI Marine Science Research Facility). Each sample was analyzed for nitrite, nitrate, nitrite+nitrate, phosphate, and ammonium concentrations.

GREENHOUSE GAS FLUX MEASUREMENTS

Greenhouse gas (GHG) fluxes (carbon dioxide and methane) of each mesocosm were measured every two weeks from June to September. Measurements were taken

between 9:00 AM and 3:00 PM on clear days to optimize on light availability. The average light intensity ranged from 15,560 to 30,111 lum/m². Carbon dioxide (CO₂) and methane (CH₄) fluxes of the mesocosm 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 GHG flux measurements took place, each core was transferred intact into a 19 liter bucket. A polyether foam ring was secured over the vegetation (without disturbing the plants) and positioned to seal the top of the bucket for chamber placement. Due to plant height, the vegetation was above the bucket opening, which ensured that light availability was optimized for the plants. A transparent polycarbonate chamber (41 cm tall x 27 cm diameter) was then 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. Chamber deployments were maintained for each mesocosm for 4 minutes. A temperature logger (Hobo, Bourne, MA) was mounted within the chamber, recording the temperature and luminosity 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 vegetated soil core surface to atmosphere. Negative fluxes are defined as those in

which gas concentrations decreased over time, representing net uptake from the atmosphere by the vegetated soil core (Moseman-Valtierra et al., 2016).

STATISTICAL ANALYSIS

Two-Way ANOVAs were used to assess differences among treatments and sampling dates for *S. alterniflora* shoot density, height, stem nutrient content, and porewater nutrient. A One-Way ANOVA was used to assess differences in belowground biomass among treatments. Repeated measures ANOVAs using JMP v.12 (www.jmp.com) were used to assess differences in CO₂ and CH₄ fluxes among algal treatments. When statistical models found significant effects, Tukey's HSD tests were performed. Data were tested for normality and homogeneity of variance and were transformed when appropriate (Underwood 1997).

Results

STEM DENSITY AND GROWTH MEASUREMENTS

There was not a significant difference in core stem density at the start of the experiment (prior to algal additions). After the first four weeks, *S. alterniflora* plants under the *Ulva* 6 treatment had significantly lower stem density compared to all other treatments ($F_{4,60} = 4.37$, $p = 0.0036$), which represented an initial 45% decrease (Fig. 2). Stem density in the *Ulva* 6 treatment increased in the subsequent weeks and reached similar measurements by the end of the experiment, but the stem increase was due to the growth of new stems rather than the recovery of original stems. There were no other significant responses of stem density across the other treatments. There was also no significant difference in stem growth among treatments ($F_{4,60} = 0.6935$, $p = 0.5994$; Fig. 3). There no significant difference in belowground biomass as well

($F_{4,20} = 0.9545$, $p=0.4537$).

NUTRIENT ANALYSIS: STEM TISSUE AND POREWATER

There was a significant increase in nitrogen content of the *Ulva* 6 treatment from the initial measurement (before macroalgal addition) to 4 weeks (Fig. 4; Table 3). After 4 weeks of macroalgal coverage, the percent nitrogen of *S. alterniflora* in the *Ulva* 6 treatment was significantly higher than the control and Mix 3 treatment (overall model: treatment: $F_{4,77} = 16.50$, $p<0.0001$; time: $F_{3,77} = 64.10$, $p<0.0001$; treatment x time $F_{12,77} = 44.86$, $p<0.0001$; Fig. 4). Also at 4 weeks, the percent nitrogen content of *S. alterniflora* stems in the Mix 3 treatment was significantly higher than the control, *Fucus*, and Mix 6 treatments (Tukey post-hoc, $p<0.05$). However, after 8 weeks, the *S. alterniflora* percent nitrogen content in the *Ulva* 6 treatment was not significantly different from the control or the Mix 3 treatments, and was significantly lower than *Fucus* 6 and Mix 6 treatments (Tukey post-hoc, $p<0.05$). After 12 weeks, percent nitrogen content for *S. alterniflora* tissues in the *Ulva* 6 and control treatments was significantly lower than the Mix 3 and Mix 6 treatments (Tukey post-hoc, $p<0.05$). At 8 and 12 weeks, the control had significantly lower *S. alterniflora* stem nutrient content than the *Fucus* treatment.

There were not significant differences among treatments in porewater nutrient content, but July porewater nitrite concentrations were significantly lower than August and September (Table 2; Tukey post-hoc, $p<0.05$).

GREENHOUSE GAS FLUX MEASUREMENTS

There was a trend of net CO₂ emission within the *Ulva* 6, Mix 3 and Mix 6 treatment and net CO₂ uptake within the control and *Fucus* 6 treatments (Fig. 5),

although the differences were not statistically significant ($F_{4,20} = 0.75$, $p = 0.57$). The greatest CO₂ fluxes were seen in August at the peak of the growing season ($F_{4,17} = 7.65$, $p = 0.001$) with no significant interaction ($F_{4,20} = 1.70$, $p = 0.19$).

Methane fluxes were not significantly different among treatments ($F_{4,20} = 1.62$, $p = 0.21$) (Fig. 6). The treatments containing *Ulva* showed a trend of higher CH₄ fluxes than the treatments without *Ulva* (*Fucus* 6 and control) (Fig. 6). There was a significant difference among months, where the August measurements (peak growing season) were significantly higher than June measurements ($F_{1,7,34.3} = 5.87$, $p = 0.0088$) with no significant interaction ($F_{6,8,34.3} = 0.71$, $p = 0.66$).

Discussion

STEM DENSITY AND GROWTH

Under the relatively low-density macroalgal accumulation studied in our experiment, we found that *S. alterniflora* stem density in our *Ulva* 6 treatment was initially negatively impacted, but later recovered (via growth of new stems), while stem density in the other treatments was unaffected. The shading from algal mats, similar to plant wrack, can limit light penetration and smother vegetation, negatively impacting aboveground biomass, which was initially witnessed in our study by *Ulva* coverage (Newton and Thornber, 2013; Ström et al., 2003; van Hulzen et al., 2006). However, *S. alterniflora* was able to recover through the production of new stems. *S. alterniflora* demonstrated an initial stress response to coverage but showed resiliency that persisted over time. Watson et al. (2015) demonstrated that high-density algal exposure resulted in lasting negative implications on *S. alterniflora* stem density. Furthermore, *S. alterniflora* stem density has shown resilience to current macroalgal

density coverage, but are not tolerant of the future projections described in Watson et al. (2015).

However, the Mix 6 treatment did not have a significant impact on *S. alterniflora* stem density, despite containing the same density of *Ulva* as the *Ulva* 6 treatment. These results suggest that there is a potential interaction between *Fucus* and *Ulva*, where *Fucus* may be ameliorating the negative impacts of *Ulva* coverage (Yates and Peckol 1993). We hypothesize that shading from *Fucus* is causing a faster decomposition of *Ulva* thus minimizing the smothering impact (Higgins et al., 2008; Chen et al., 2009). However, this relationship needs further study.

Although other studies have linked nutrient additions with increased plant growth rates (Pennings et al., 2005; McFarlin et al., 2008), there were not significant differences in *S. alterniflora* growth rates among treatments. By contrast, high macroalgal additions have been shown to negatively impact mean stem height (Watson et al., 2015; Wasson et al., 2017). Since our low density treatments showed similar growth patterns by the end of the experiment, it suggests *S. alterniflora* growth is not significantly influenced under these macroalgal densities.

STEM AND POREWATER NUTRIENT CONTENT

Numerous studies have found positive correlations between nutrient additions and *S. alterniflora* growth rate, biomass, stem density, and photosynthetic rate (Valiela and Teal 1974; Levine et al., 1998; Pennings et al., 2005; McFarlin et al., 2008). Boyer and Fong (2005) traced nitrogen from macroalgae to plant stems, which demonstrates the influence of macroalgal exposure to plant nitrogen content. In our study, *S. alterniflora* in the *Ulva* 6 treatment had significantly higher stem percent

nitrogen than the control after four weeks, potentially due to nitrogen released from rapid *Ulva* decomposition (Castaldelli et al., 2003; Viaroli et al., 1992). Other nutrient addition studies have demonstrated similar patterns (Newton and Thornber 2013); McFarlin et al. (2008) found an increase in *S. alterniflora* nitrogen content with the addition of nutrients as well as an increase in cover, height, and biomass. *S. alterniflora* may also allocate nutrients to roots and rhizomes rather than to the stems, which could explain the decrease in stem nitrogen content of the *Ulva* 6 treatment during the subsequent weeks (Hopkinson and Schubauer, 1984; Newton and Thornber, 2013). At 8 and 12 weeks, the control had significantly lower *S. alterniflora* stem nutrient content than the *Fucus* 6 treatment, which suggests *S. alterniflora* is absorbing nutrients released from *Fucus* (Hunter, 1976; Nielsen et al., 2007). However, since there was not a significant difference in stem growth it suggests that the supplied nutrients from macroalgal decomposition at these current densities are not sufficient to enhance growth.

In our experiment, the impact of *Ulva* coverage and nutrient addition was clear due to the significant increase in *S. alterniflora* nitrogen content seen in the *Ulva* 6 treatment from the initial measurement (before *Ulva* addition) to 4 weeks. The results also show higher percent nitrogen seen in the *Ulva* 6 treatment occurring during the same time frame as the stem density decline, which suggests that higher stem nitrogen content may be due to less competition for nutrients among *S. alterniflora* stems. In the subsequent weeks, stem nitrogen content decreased when the stem density increased within the *Ulva* 6 treatment, which further supports this argument and suggests that nutrients were potentially allocated for new stem growth (Davis et al.,

2017). The results and relationship between the Mix 3, Mix 6 treatment, and the control also suggest that there is an interaction between *Fucus* and *Ulva* that is impacting *S. alterniflora* stem nutrient content, but this relationship needs further investigation (Fig. 4, Table 3).

However, at four weeks we did not find similar high values in *S. alterniflora* stem nitrogen content as the *Ulva* 6 treatment in our other algal addition treatments including the Mix 6 treatment (twice as much algae as the *Ulva* 6 treatment). Also at four weeks, the *S. alterniflora* of the Mix 6 treatment (contained double the amount of *Fucus* as Mix 3 treatment) had significantly lower stem nitrogen content than the Mix 3. This suggests that *Fucus* may be impacting the effects of *Ulva* coverage, creating a barrier between *Ulva* and *S. alterniflora* and potentially absorbing the nutrients released in *Ulva* decomposition leading to less consumption of nutrients by *S. alterniflora* (Yates and Peckol 1993). It is also possible that some nutrients are being flushed from the system as the water is drained during low tide.

The porewater analysis did not show a significant impact of algae on porewater nutrient concentration, which suggests that these nutrients may be exiting the system via drainage or absorbed and utilized by *S. alterniflora* (Howes and Goehring, 1994; Johnson et al., 2016). The relatively low algal densities used in this experiment may not be sufficient to instigate changes in porewater nutrient content as well. Watson et al. (2015) found that porewater nutrient content after high-density algal additions result in significantly higher nutrient concentration. These results suggest that macroalgal accumulation does not have an automatic effect on porewater, but the impact may be dependent on macroalgal density.

GREENHOUSE GAS FLUXES

In this study, we are examining the impact that macroalgal accumulation has on net greenhouse gas fluxes within a salt marsh environment to determine whether macroalgal coverage influences salt marsh carbon flux dynamics. Due to slow decomposition processes and high rates of carbon dioxide uptake by vegetation, salt marshes are important carbon sinks (Reddy and DeLaune 2010). Salt marshes produce a negligible amount of methane due to the competitive interaction of sulfate reducing bacteria and methanogens (Reddy and DeLaune 2010; Poffenbarger et al. 2011). However, nutrient enrichment (via macroalgal decomposition) can enhance carbon dioxide and methane production in salt marshes potentially creating a carbon source (Moseman-Valtierra 2013; Chmura et al., 2016). The decomposition of organic matter, including macroalgae, can supply nutrients needed for microorganisms, which enhances the generation of carbon dioxide (via respiration; Reddy and DeLaune 2010) and methane (via methanogenesis; Poffenbarger et al. 2011). In addition, soil oxygen is depleted during decomposition, thus exacerbating anoxic conditions creating a more hospitable environment for methanogens and anaerobic bacteria, which can lead to the greater fluxes; these patterns were seen in our experiment (Valiela et al. 1997; Reddy and DeLaune 2010). Watson et al. (2015) found that high-density *Ulva* accumulation resulted in significantly higher carbon dioxide fluxes. These trends were observed under our low-density algal treatments (Fig. 5, Fig 6.), but were not significant. Our results suggest that these relatively low-density yet current macroalgal accumulations do not show to significantly impact a salt marsh's carbon flux dynamics.

Previous studies have demonstrated the threat of *Ulva* coverage on salt marsh vegetation (Watson et al., 2015; Wasson et al., 2017). The blade morphology of *Ulva* may create a barrier between the soil-atmospheric interface, yielding a more anoxic environment that is suitable for anaerobic bacterial activity (Reddy and DeLaune 2010). Increased plant and soil respiration due to plant stress caused by macroalgal smothering, macroalgal respiration, and an increase in resource availability from plant and macroalgal detritus could be responsible for the trend of greater carbon dioxide and methane fluxes seen in the treatments containing *Ulva* (Flexas et al., 2006; Neubauer, 2010; Ryan, 1991). The *Fucus* treatment showed similar fluxes to the control (negative carbon dioxide flux and lower methane flux), which could be explained by its branched morphology and slow decomposition that minimizes shading stress and nutrient release. Further investigation is warranted to see if this pattern remains in higher density *Fucus* accumulation.

Macroalgal coverage at current densities showed patterns of increased carbon dioxide and methane emissions relative to the control, but did not result in lasting negative effects on *S. alterniflora*, indicating *S. alterniflora* resiliency to macroalgal coverage at these densities. However, negative implications of macroalgal coverage (initial stem density decline, trends of higher greenhouse gas fluxes) are manifesting under low-density accumulations. Furthermore, the impacts of higher density accumulation (420 g/m²; Newton and Thornber 2012) may result in negative implications on *S. alterniflora* survival and significantly higher greenhouse gas emissions (e.g. Watson et al., 2015; Wasson et al., 2017). As macroalgal densities

increase with climate change progression, they may significantly influence vegetation survival and greenhouse gas production (Doney et al., 2012; Sinha et al., 2017).

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Table 1. Ambient seawater average nutrient content. These are the mean values ($\pm$ 1 standard error) from June-September 2016, the duration of the experiment. Data is obtained from Narragansett Bay Long-Term Plankton Time Series at <https://web.uri.edu/plankton/data/> at Narragansett Bay Station Two

<u>Nutrient</u>	<u>Average Value (μmol) +/- Standard Error</u>
Ammonium	1.38 ± 0.38
Dissolved Inorganic Phosphorous	1.30 ± 0.10
Nitrate+Nitrite	0.60 ± 0.16
Nitrate	0.069 ± 0.11
Nitrite	0.13 ± 0.027
Dissolved Inorganic Nitrogen	0.62 ± 0.14

Table 2. Statistical result of *Spartina alterniflora* core porewater analysis

<u>Nutrient</u>	<u>Average (μmol) +/- Standard Error</u>	<u>Repeated Measures ANOVAs results</u>
Ammonium	Control: 13.12 $\pm$ 3.28 <i>Fucus</i> 6: 9.35 $\pm$ 3.63 <i>Ulva</i> 6: 13.13 $\pm$ 2.98 Mix 3: 22.51 $\pm$ 3.59 Mix 6: 13.33 $\pm$ 3.28	Model: $F_{12,31} = 1.70$, $p=0.12$
Nitrite	Control: 0.44 $\pm$ 0.042 <i>Fucus</i> 6: 0.23 $\pm$ 0.047 <i>Ulva</i> 6: 0.35 $\pm$ 0.038 Mix 3: 0.44 $\pm$ 0.046 Mix 6: 0.43 $\pm$ 0.042	Model: $F_{12,31} = 9.60$, $p<0.0001$ Treatment: $F_{2,31} = 1.58$, $p=0.22$ Time: $F_{1,31} = 49.71$, $p<0.0001$ Treatment*Time: $F_{6,31} = 4.10$, $p=0.0038$
Nitrate	Control: 3.20 $\pm$ 0.51 <i>Fucus</i> 6: 2.10 $\pm$ 0.56 <i>Ulva</i> 6: 1.49 $\pm$ 0.46 Mix 3: 2.39 $\pm$ 0.56 Mix 6: 1.75 $\pm$ 0.51	Model: $F_{12,31} = 1.33$, $p=0.25$
Nitrite+Nitrate	Control: 3.63 $\pm$ 0.53 <i>Fucus</i> 6: 2.34 $\pm$ 0.59 <i>Ulva</i> 6: 1.84 $\pm$ 0.48 Mix 3: 2.83 $\pm$ 0.58 Mix 6: 2.18 $\pm$ 0.53	Model: $F_{12,31} = 1.22$, $p=0.31$
Phosphate	Control: 1.92 $\pm$ 0.46 <i>Fucus</i> 6: 1.28 $\pm$ 0.51 <i>Ulva</i> 6: 1.83 $\pm$ 0.42 Mix 3: 2.76 $\pm$ 0.50 Mix 6: 1.74 $\pm$ 0.46	Model: $F_{12,31} = 1.04$, $p=0.44$

Table 3. Tukey post hoc test results of the *Spartina alterniflora* stem nitrogen content analysis. Conditions not connected by same letter represent significant differences.

<u>Treatment</u>	<u>Week</u>	<u>Letter</u>
Control	0	C, D
Control	4	G, H
Control	8	G, H
Control	12	D, E, F, G
<i>Fucus</i> 6	0	B
<i>Fucus</i> 6	4	G, H
<i>Fucus</i> 6	8	C, D, E
<i>Fucus</i> 6	12	C, D
<i>Ulva</i> 6	0	E, F, G
<i>Ulva</i> 6	4	A
<i>Ulva</i> 6	8	H
<i>Ulva</i> 6	12	D, E, F
Mix 3	0	C, D
Mix 3	4	C, D
Mix 3	8	G, H
Mix 3	12	B, C
Mix 6	0	C, D
Mix 6	4	H
Mix 6	8	F, G, H
Mix 6	12	B, C

Figure 1a. Schematic showing experimental mesocosm design, with *Spartina alterniflora* and varying macroalgal wrack treatments, n=5 (left to right: control, *Fucus* 6, *Ulva* 6, Mix 3, Mix 6). The green-blade alga is *Ulva* and the branched alga is *Fucus*.

Figure 1b. Photo of experimental set up at the Marine Science Research Facility at the University of Rhode Island (URI)'s Narragansett Bay Campus in Narragansett, RI.

Figure 2. Mean percent *Spartina alterniflora* stem density change from initial measurements per treatment, ± 1 standard error. The initial average stem values per core (prior to algal additions) were as follows. Control: 31.2, *Fucus* 6: 31.6, *Ulva* 6: 31, Mix 3: 32.8, and Mix 6: 35. Measurement were taken every 4 weeks through duration of experiment.

Figure 3. Mean percent change in *Spartina alterniflora* stem height from initial height per treatment, ± 1 standard error. The initial average stem height values (prior to algal additions) were as follows. Control: 41.6cm, *Fucus* 6: 39.4cm, *Ulva* 6: 39.1cm, Mix 3: 39.4cm, and Mix 6: 38.3cm. Measurement were taken every 4 weeks through duration of experiment.

Figure 4. *Spartina alterniflora* stem percent nitrogen content per treatment, $\pm$ standard error. Measurements were taken every 4 weeks throughout the duration of the experiment. Week 0 represents the percent nitrogen content before macroalgal addition.

Figure 5. Mean carbon dioxide fluxes per treatment, $\pm$ standard error.

Figure 6. Mean methane fluxes per treatment, ± 1 standard error.

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CHAPTER 2

MONITORING SALT MARSH VEGETATION AND SOIL RESPONSE TO RUNNEL AND SEDIMENT ENHANCEMENT INTERVENTIONS

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Abstract

Sea level rise within New England is accelerating at a rate faster than the global average, leaving salt marshes susceptible to degradation. Sediment enhancement and hydrological alterations (e.g. runnel installations) projects are two types of climate change adaptation techniques that have been implemented in New England to combat the effects of sea level rise within salt marshes. Sediment enhancement is used to increase the elevation and runnels (shallow ditches) to enhance drainage in drowning marshes. In this study, we investigated the impacts of these climate change adaptation methods on soil properties, vegetation composition, and greenhouse gas fluxes (methane and carbon dioxide). *Spartina alterniflora* stem density in certain runnel areas increased after runnels were installed and *Spartina patens* (high marsh plant) were able to persist under this treatment. The sediment enhancement treatment resulted in low belowground biomass, soil % organic matter, and plant recolonization, suggesting that the added sediment material inhibited vegetation growth. There was a significant difference in carbon dioxide uptake rates among treatments, with the unmanipulated (Control) areas having the highest uptake rates.

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. 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 (Ashton et al., 2008; 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 and 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 and DeLaune, 2010). They are also a vital food source, breeding habitat, and nursery ground for birds (including the vulnerable salt marsh sparrow, *Ammodramus caudacutus*) and aquaculture species including fish and shellfish (Hanson and Shriver, 2006; Raposa and Roman, 2006; Bayard and Elphick, 2011). These environments also provide flood abatement and help prevent coastal erosion (Leonard and 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). 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 (Hartig et al., 2000; 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., 2017b).

Although climate change mitigation research (e.g. efforts to reduce greenhouse gas emissions) serves an important purpose, climate change adaptation 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). One climate adaptation approach to build salt marsh resiliency is sediment

enhancement (SE), also known as thin layer deposition, where dredged sediment material is added to the salt marsh surface (Cahoon et al., 2019). The purpose of this technique is to raise the salt marsh platform to an elevation that can withstand future projections of SLR.

Another technique to offset SLR impacts is a type of hydrological climate change adaptation, where shallow dredged ditches (runnels) are dug into the existing marsh platform to drain excess water from high marsh areas that have ponded. Since the 1930s, human-made ditches have become a prominent feature of Atlantic Coast salt marshes (Corman et al., 2012). Historically, deep ditches have been used for mosquito control and agriculture purposes (Dale et al., 1993; Breitfuss and Connolly, 2004; Dale and Knight, 2006; Dale, 2008). However, our study describes an alternate use of ditches as shallow runnels that helps to preserve marsh ecological functions in light of climate change. Runnels constructed in this study are shallow (0.15–0.5 m wide by 0.2–0.5 m depth) and strategically placed by unnatural pools, which helps to mitigate some of the negative effects (marsh subsidence, decrease in natural pools) of historical ditches of greater depths (Dale et al., 1993). These unnatural pools are a result of excess tidal inundation that leads to vegetation die off and displacement of high marsh plants (Raposa et al., 2017b).

The U.S. Fish and Wildlife Service and Rhode Island's Save the Bay implemented SE and hydrological (shallow runnel) projects at salt marsh sites in Narragansett, RI. The purpose of this study was to assess salt marsh habitat response, including vegetation, soil, and greenhouse gas fluxes, of the treated vs. control (untreated) areas. We expected that runnels would promote plant recovery, including

increases in belowground biomass, and allow for the persistence of high marsh plant species. By contrast, the SE project would demonstrate more gradual signs of plant recovery. The impacts of these projects will have implications on future coastal resiliency initiatives within New England and offer potential methods to mitigate SLR impacts globally.

Methodology

Our field sites were located along fringing salt marshes in the Narrow River in Narragansett, Rhode Island USA, where dredged runnels (2015) and 10cm thick, fine sand SE (2016) projects were conducted by the U.S. Fish and Wildlife Service and Save the Bay. The runnels were created using hand shovels and an excavator. The sediment for the SE project was distributed using a hydraulic dredge pipeline and excavator. Save the Bay (STB) performed the initial preliminary site assessments in 2014, including vegetation surveys, before the runnels were implemented. As these methods were different from those used in this study to track recovery over time, we used STB results as a reference but do not include them in statistical analyses. In May 2017, an initial planting effort of *Spartina alterniflora* and *Distichlis spicata* occurred to jumpstart vegetation recolonization for the SE area, but most plants did not survive by the end of the 2017 growing season (October).

For our post-manipulation study, we had four treatments: control (unaltered), sediment enhancement, soft sediment runnels, and vegetated runnels. We separated runnels into two different treatments based on their field characteristics and elevation at the start of this study in 2017; soft sediment runnels were surrounded by unconsolidated sediment, bare, and at a lower elevation, while vegetated runnels were

surrounded by vegetation, more stable sediment, and at a higher elevation. The Control and the Soft Sediment Runnel were at similar elevations, and the Sediment Enhancement was at a significantly higher elevation than the other 3 treatments ($F_{3,35}=22.9$, $p<0.01$). Average elevations for treatments were as follows: 1) Control: $30.9\text{cm} \pm 5.1$ (1.0ft NAVD88); 2) Soft Sediment Runnel: $24.2\text{cm} \pm 0.8$ (0.8ft NAVD88); 3) Vegetated Runnel: 39.6 ± 10.1 (1.3ft NAVD88); and 4) Sediment Enhancement: $51.4\text{cm} \pm 6.4$ (1.7ft NAVD88).

We collected post-manipulation data using two salt marsh sites for each treatment, with one linear transect (12m) per site, except for the SE transects, which were both located at the same site (~68m apart), due to the limiting areas of sediment placement. Along each transect, we established six circular plots (26cm in diameter), spaced 2m apart. Plot measurements were taken in August and October of 2018 and 2019; these times represented the beginning, middle, and end of each salt marsh growing season in New England. We were not able to sample during June or July due to closure of our field sites for salt marsh sparrow nesting. Soil depth (5cm) pH was measured at each plot using an Electronic Soil Tester pH Analyzer (FLIR Commercial Systems Inc., Nashua, New Hampshire). Within each plot, the *S. alterniflora* stems were counted and the percent covers of the remaining plant species were measured. 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, percent organic matter, percent moisture, and bulk density. The core samples were weighed for wet weight, dried at 30°C for 3 days, weighed for dry weight, and then sieved (2mm sieve). The sieve separated the root belowground biomass and sediment.

The belowground biomass was then weighed and the sediment (10-15g) was burned at 400 °C for 16 hours. The percent organic matter of the sample was determined using the Loss on Ignition Method (Ball 1964). Percent moisture was calculated using the wet weight and dry weight of each soil sample.

Greenhouse gas (GHG) fluxes (carbon dioxide and methane) of each plot were measured in monthly from August to October of 2017-2019. Carbon dioxide (CO₂) and methane (CH₄) 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. Chamber deployments were maintained for each mesocosm for 4 minutes. 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 to test for differences in treatments over time for the following variables: *S. alterniflora* stem count, species percent cover, belowground biomass, organic matter, salinity, percent moisture, pH, and gas fluxes.

Results

Pre/Post treatment community composition

Open water within the runnel treatment decreased from 27% (prior) to 0% after the runnels were implemented (data collected by Save the Bay). The unvegetated areas (combined bare and open water areas) in the Soft Sediment Runnel were 11% less in August 2019 than August 2014 before the runnels were implemented (Figure 1). In the Control, unvegetated areas were 51% greater in August 2019 than in August 2014. The *S. alterniflora* percent cover of the Soft Sediment Runnel was approximately 32% and 42% higher in August 2018 and 2019, respectively, than the Runnel area in 2014 (Figure 2). The *S. patens* percent cover in the Vegetated Runnel was 49% and 54% higher in August 2018 and 2019, respectively, than the Runnel area in 2014 (Figure 3). The *S. patens* of the Control was 84% and 87% lower in August 2018 and 2019, respectively, than the Control area in 2014 (Figure 3). The *S. alterniflora* of the Control was 15% and 14% higher in August 2018 and 2019, respectively, than the Control area in 2014 (Figure 2).

Post treatment vegetation

From August 2018 to August 2019, the bare areas in the Soft Sediment Runnel and Vegetated Runnel decreased by 38% and 12%, respectively, while the bare areas in the Control and Sediment Enhancement stayed relatively consistent (Figure 1). The bare areas in October 2017 compared to October 2019 decreased 78% and 28% in the Soft Sediment Runnel and Control, respectively, while the Vegetated Runnel and Sediment Enhancement stayed relatively consistent (treatment: $F_{3,117} = 78.99$, $p < 0.0001$; time $F_{4,117} = 15.06$, $p < 0.0001$; treatment*time: $F_{12,117} = 2.73$, $p = 0.0027$).

There was a mean increase in *S. alterniflora* stem density in the Soft Sediment Runnel and Control, 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, there was a steady decrease in *S. alterniflora* stem density over time in the Vegetated Runnel and minimal to no growth in the Sediment Enhancement (overall model $F_{71,224} = 12.72$, $p < 0.0001$; treatment: $F_{3,224} = 75.88$, $p < 0.0001$; time: $F_{6,224} = 19.32$, $p < 0.0001$; treatment*time: $F_{18,224} = 7.58$, $p < 0.0001$). There was no significant difference in *S. alterniflora* stem height among treatments, but there was an interaction effect (treatment: $F_{2,783} = 0.69$, $p = 0.51$; time $F_{4,783} = 89.63$, $p < 0.0001$; treatment*time: $F_{16,783} = 7.70$, $p < 0.0001$). The Sediment Enhancement was not included in this analysis for stem height, since there were not enough *S. alterniflora* stems present (average $n = 0.12$ per plot).

S. alterniflora percent cover was close to zero in October 2019 than 40% cover (~237x higher) in October 2017 in the Soft Sediment Runnel (Figure 2). The Control was ~14x greater in October 2019 than October 2017, while the Vegetated Runnel was ~7x greater in October 2019 than October 2017. There was minimal to no *S.*

alterniflora percent cover in the Sediment Enhancement from 2017-2019 (overall model $F_{59,118} = 10.16$, $p < 0.0001$; treatment: $F_{2,118} = 78.92$, $p < 0.0001$; time: $F_{3,118} = 7.41$, $p = 0.0001$; treatment*time: $F_{11,118} = 3.45$, $p = 0.0004$).

S. patens was not present in the Soft Sediment Runnel and Sediment Enhancement areas from 2017-2019, while it declined in the Control from 8% to 0% from 2017-2019 (Figure 3). By contrast, the percent cover of *S. patens* was significantly higher in the Vegetation Runnel treatment (~11.6x higher than the Control) compared to the other treatments (overall model $F_{69,194} = 9.19$, $p < 0.0001$; treatment: $F_{1,194} = 113.12$, $p < 0.0001$; time: $F_{5,194} = 3.58$, $p = 0.0040$; treatment*time: $F_{16,194} = 4.87$, $p < 0.0001$).

Belowground Biomass (BGB) and Organic Matter

Belowground biomass was 2.6x and 1.9x higher in the Control than the Soft Sediment Runnel and Vegetated Runnel areas, respectively. Sediment Enhancement had very low belowground biomass (mean = 0.04g; Figure 5), with a significant interaction (overall model $F_{53,111} = 8.83$, $p < 0.0001$; treatment: $F_{3,111} = 145.41$, $p < 0.0001$; time: $F_{3,111} = 7.71$, $p = 0.0001$; treatment*time: $F_{9,111} = 2.21$, $p = 0.026$).

Sediment Enhancement had ~99% less organic matter than the other treatments (overall model $F_{53,106} = 36.03$, $p < 0.0001$; treatment: $F_{3,106} = 453.85$, $p < 0.0001$; time: $F_{3,106} = 2.85$, $p = 0.041$; treatment*time: $F_{9,106} = 0.92$, $p = 0.51$; Figure 6).

Abiotic Soil Factors: Salinity, percent moisture, and pH

Sediment Enhancement had significantly higher salinity than the other three treatments. Sediment Enhancement salinity was 70% higher than the Control. The Control was 29% and 25% higher salinity than Soft Sediment Runnel and Vegetated

Runnel areas, respectively (overall model $F_{31,284} = 19.80$, $p < 0.0001$; treatment: $F_{3,284} = 165.53$, $p < 0.0001$; time: $F_{7,284} = 8.03$, $p < 0.0001$; treatment*time: $F_{21,284} = 1.36$, $p = 0.13$; Table 2).

Sediment Enhancement had at least 75% lower moisture than the other three treatments at each time point (Table 3; overall model $F_{53,111} = 750.33$, $p < 0.0001$; treatment: $F_{3,111} = 4087.68$, $p < 0.0001$; time: $F_{3,111} = 1.47$, $p = 0.23$; treatment*time: $F_{9,111} = 0.72$, $p = 0.69$). By contrast, the Soft Sediment Runnel had significantly higher moisture than all other treatments.

The soil pH was significantly higher in the Sediment Enhancement compared to the other three treatments (overall model $F_{23,219} = 29.89$, $p < 0.0001$; treatment: $F_{3,219} = 140.66$, $p < 0.0001$; time: $F_{5,219} = 20.71$, $p < 0.0001$; treatment*time: $F_{15,219} = 9.34$, $p < 0.0001$; Table 4).

Greenhouse gas fluxes

There was not a significant difference in methane flux among treatments (overall model: $F_{84,209} = 0.44$, $p = 1.0$). There was significantly higher uptake in carbon dioxide in the Control than the other three treatments (12.6x Sediment Enhancement, 2.1x Soft Sediment Runnel, and 1.9x Vegetated Runnel; Figure 7; overall model $F_{75,156} = 9.60$, $p < 0.0001$; treatment: $F_{1,156} = 13.75$, $p = 0.0003$; time: $F_{3,156} = 35.73$, $p < 0.0001$; treatment*time: $F_{19,156} = 9.62$, $p < 0.0001$). Both runnel treatments took up significantly more carbon dioxide than Sediment Enhancement. The Soft Sediment Runnel showed more CO₂ uptake in 2018 and 2019 than in 2017 (Figure 7).

Discussion

A prevalent issue within Rhode Island salt marshes is that low marsh species are displacing high marsh plants due to sea level rise impacts (Raposa et al., 2017b). This pattern is seen in the Control as *S. patens* in the Control gradually decreased from 2017-2019, as well as a decrease from initial conditions in *S. patens* coverage in 2014. By contrast, the Vegetated Runnel shows a decline in *S. alterniflora* in the Vegetated Runnel and the persistence of *S. patens* from 2017-2019, which suggests runnel effects in combination with higher elevations can help *S. patens* outcompete *S. alterniflora*. The results of this study suggest that runnel impacts have the potential to prevent the displacement of high marsh plants by low marsh plant species, but further monitoring is necessary to see if this pattern persists. 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 and 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 within salt marshes (Crain et al. 2009; Tonjes, 2013). In this study, the runnel hydrological manipulation demonstrated an ability to combat SLR impacts and preserve high marsh habitat. This has positive implications on the vulnerable bird species, *Ammodramus caudacutus* or salt marsh sparrow, that nests within *S. patens* habitat (Bayard and Elphick, 2011). Without intervention, the impacts of SLR and excess tidal inundation may result in the continued decline in high marsh habitat, which will be detrimental to the salt marsh

sparrow and other bird and invertebrate species that rely on high marsh habitat for survival (Bayard and Elphick, 2011; Zajac et al., 2017).

In a drainage enhancement study, Raposa et al. (2019) used linear 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 linear creek areas. However, in our study the high marsh plants decreased from 2017-2019 in the Control while Soft Sediment Runnel did not contain *S. patens* throughout that time period, but showed significant increase in *S. alterniflora* over time. This suggests that drainage treatments (linear creeks vs shallow runnels) can lead to varying results. There were some parallels between studies, as Raposa et al. (2019) found that linear creeks had the greatest impact on bare areas that were later revegetated. Although shallower runnels were used in this study, we found similar results, as the bare areas within the Soft Sediment Runnel decreased at a faster rate than the Control areas. The Control and the Soft Sediment Runnel increased in *S. alterniflora* percent cover from October 2017 to October 2018, but the rate of increase was ~15x greater in the Soft Sediment Runnel than the Control suggesting a drainage impact.

This study showed an increase of belowground biomass in the Soft Sediment Runnel and decline in the Control from August 2018 to August 2019, but these are short term effects that needs longer term monitoring to determine if this pattern continues and will have lasting implications on the salt marsh sediment stability. However, the Control represented higher overall belowground biomass than the other treatments. Although the Control and Soft Sediment Runnel of this study were located

at similar elevations, the Control represented better initial conditions (vegetated, stable sediment) than the Soft Sediment Runnel (bare and unconsolidated sediment) at the beginning of the study. The Soft Sediment Runnel areas were located by pools before the runnels were placed and the Control was not located by pooled areas, which could have resulted in these varying initial conditions. For future studies, an untreated control area by undrained pools would more accurately compare and depict the impacts of runnels. There was less of an increase in the Vegetated 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 and organic matter in the Vegetated Runnel compared to the Control, due to the difference in dominant vegetation.

One of the goals of climate change adaptation is to preserve coastal wetlands ecosystem functions for environmental purposes (Hartig et al., 2000; Barbier et al., 2011; Weston, 2014b; Linhoss et al., 2015; Crosby et al., 2016; Watson et al., 2017; 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 and Moseman-Valtierra, 2015). The Soft Sediment Runnel had higher carbon dioxide uptake in 2018 and 2019 than in 2017, which suggests an improvement in this ecological function in coordination with vegetation recovery. The Control showed the highest rates of CO₂ uptake in 2018 likely due to the higher *S. alterniflora* stem count than the other treatments. The Control did not represent conditions of ponded areas, which likely led

to better response of vegetation and CO₂ uptake when compared to the Soft Sediment Runnel areas that were ponded before the runnels were implemented creating initial conditions that were more severely degraded.

Due to the higher elevation of the Vegetated Runnel compared to the Control and Soft Sediment Runnel we expected to see the differences in vegetation and CO₂ uptake. Moseman-Valtierra et al. (2016) showed that *S. alterniflora* dominated areas removed CO₂ from the atmosphere at a faster rate than *S. patens* dominated areas. Due to this, it was not unexpected that the CO₂ fluxes are lower in 2019 than 2017 in the Vegetated Runnel, as there were less *S. alterniflora* stems in 2019. Carbon dioxide uptake was lower in all treatments in 2019 than 2018, which suggests that abiotic factors rather than treatment are having the greatest impact on CO₂ fluxes during that period of measurements (Portnoy and Valiela, 1997; Poffenbarger et al., 2011; Wilson et al., 2015; Martin and Moseman-Valtierra, 2017; Wang et al., 2017).

The Narrow River SE project did not progress as expected, as the plots within this experiment had no natural revegetation growth throughout the study. This project deposited ~10 cm of sediment, which is less than another SE project at Ninigret Pond in Charlestown, RI (~40cm) that occurred at the same time. The Ninigret Pond project experienced close to full revegetation by the third growing season, despite having much greater sediment deposition depth than the Narrow River project (W. Ferguson, personal communication). In other studies, Mendelssohn and Kuhn (2003) deposited ~60cm of sediment and reported full vegetation after two years. Slocum et al. (2005) found that moderate levels of sediment deposition (5-12cm) produced the most beneficial and long-lasting effects on vegetation and soil. Therefore, it is likely that the

amount of sediment deposited at the Narrow River site is not the main cause of hindered vegetation growth.

The sediment size used for this SE project was classified as fine sand (Hardy unpublished data), which could have been too fine for vegetation growth and promote soil drainage. Larger grain sized sediment, coarse sand, has been used in other SE projects within Rhode Island, which resulted in greater vegetation recolonization (C. Chaffee, personal communication). Croft et al. (2006) used medium sized sand in their sediment placement project and found vegetation to return after two growing seasons. Hardisky and Adams (1978) found that fine sediment placement material resulted in less stem growth than the use of larger grain sediment. Therefore, grain size may be an important influencer of vegetation recovery.

Due to the bare SE areas, evaporation was likely a large factor within this environment, which explains the high salinity and low moisture of this treatment, which caused an additional barrier to plant revegetation (Bertness et al., 1992; Bertness, 1992). Halophytes, such as *Salicornia*, were also seldom seen within these plots although having a high salinity tolerance. The SE areas often resulted in insignificant CO₂ fluxes, which was expected given the lack of vegetation. Minimal microbial activity, which is suggested by the low organic matter and belowground biomass within the treatment, could also explain the lack of soil respiration (measured via CO₂ emission) and insignificant flux (Reddy and DeLaune, 2010).

The SE area also did not have well established drainage. The high evaporation rates removed moisture from the sediment, but salt remained in the system leading to hypersaline areas. Although not represented in the study plots, SE areas closest to

natural creeks and runnels had lower salinity, ~30ppt, and more positive vegetation regrowth (personal observation). The runnel treatments within this experiment had significantly lower salinity than the Control, matching results from other runnel studies (Dale, 2008; Dale and Knight, 2006). Incorporating runnels into SE projects has been practiced at the Ninigret SE project, which produced positive vegetation results that were partially attributed to the runnel impact (W. Ferguson personal communication). This suggests that establishing runnels within SE sites could serve as an important component to prevent hypersaline areas that hinder vegetation growth.

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Table 1. Average percent cover of open water, *Spartina alterniflora* (dominant low marsh species) and *Spartina patens* (dominant high marsh species) before the runnels were placed in 2014.

Treatment	Habitat Composition	Average Percent Cover $\pm$ 1_standard error
Control	Open water	13.56 $\pm$ 0.14
Control	<i>Spartina alterniflora</i>	53.3 $\pm$ 2.91
Control	<i>Spartina patens</i>	26.0 $\pm$ 1.95
Control	Other Vegetation	12.89 $\pm$ 5.99
Control	Bare	2.0 $\pm$ 1
Runnel	Open water	27.0 $\pm$ 9.11
Runnel	<i>Spartina alterniflora</i>	39.6 $\pm$ 0.71
Runnel	<i>Spartina patens</i>	21.0 $\pm$ 0.61
Runnel	Bare	8.6 $\pm$ 0.58
Runnel	Other Vegetation	12.08 $\pm$ 1.81

Table 2. Average salinity (psu) per treatment $\pm$ 1 standard error.

Treatment	Salinity (psu)	Month and Year
Control	35.5 ± 2.8	August 2018
Sediment Enhancement	51.5 ± 3.3	August 2018
Soft Sediment Runnel	22.8 ± 4.0	August 2018
Vegetated Runnel	26.6 ± 3.3	August 2018
Control	29.1 ± 3.9	October 2018
Sediment Enhancement	54.5 ± 4.3	October 2018
Soft Sediment Runnel	21.5 ± 5.5	October 2018
Vegetated Runnel	23.6 ± 3.9	October 2018
Control	31.9 ± 0.9	August 2019
Sediment Enhancement	64.2 ± 1.6	August 2019
Soft Sediment Runnel	27.3 ± 1.3	August 2019
Vegetated Runnel	25.7 ± 0.9	August 2019
Control	24.8 ± 2.2	October 2019
Sediment Enhancement	48.2 ± 2.2	October 2019
Soft Sediment Runnel	22.4 ± 3.1	October 2019
Vegetated Runnel	21.7 ± 2.2	October 2019

Table 3. Average percent moisture per treatment $\pm$ 1 standard error.

Treatment	Moisture (%)	Month and Year
Control	85.0 ± 0.7	August 2018
Sediment Enhancement	19.1 ± 0.7	August 2018
Soft Sediment Runnel	87.8 ± 1.0	August 2018
Vegetated Runnel	84.2 ± 0.8	August 2018
Control	85.3 ± 0.6	October 2018
Sediment Enhancement	20.1 ± 0.6	October 2018
Soft Sediment Runnel	88.4 ± 0.9	October 2018
Vegetated Runnel	84.4 ± 0.6	October 2018
Control	85.8 ± 0.9	August 2019
Sediment Enhancement	20.4 ± 0.9	August 2019
Soft Sediment Runnel	88.1 ± 1.2	August 2019
Vegetated Runnel	84.6 ± 0.9	August 2019
Control	86.3 ± 0.5	October 2019
Sediment Enhancement	20.7 ± 0.5	October 2019
Soft Sediment Runnel	86.6 ± 0.7	October 2019
Vegetated Runnel	84.8 ± 0.5	October 2019

Table 4. Average pH (5cm depth) per treatment $\pm$ 1 standard error.

Treatment	pH	Month and Year
Control	5.2 ± 0.1	August 2018
Sediment Enhancement	7.6 ± 0.1	August 2018
Soft Sediment Runnel	6.4 ± 0.2	August 2018
Vegetated Runnel	5.9 ± 0.2	August 2018
Control	5.2 ± 0.2	October 2018
Sediment Enhancement	7.3 ± 0.2	October 2018
Soft Sediment Runnel	6.2 ± 0.2	October 2018
Vegetated Runnel	5.9 ± 0.2	October 2018
Control	6.6 ± 0.1	August 2019
Sediment Enhancement	7.1 ± 0.1	August 2019
Soft Sediment Runnel	6.4 ± 0.1	August 2019
Vegetated Runnel	6.2 ± 0.1	August 2019
Control	6.7 ± 0.06	October 2019
Sediment Enhancement	7.8 ± 0.06	October 2019
Soft Sediment Runnel	6.8 ± 0.09	October 2019
Vegetated Runnel	6.4 ± 0.06	October 2019

Figure 1. Mean percent bare $\pm$ 1 SE (per 26cm diameter plot). Red line represents unvegetated areas (combined open water and bare) in 2014 before the runnels were constructed. Red line only represents previous conditions of runnel and control sites and does not refer to Sediment Enhancement previous conditions.

Figure 2. Mean *Spartina alterniflora* percent cover + 1 SE (per 20cm diameter plot). Pre-Runnel represents the entire runnel area before the runnels were constructed (data collected by Save the Bay).

Figure 3. Mean *Spartina patens* percent cover + 1 SE (per 26cm diameter plot). Pre-Runnel represents the entire runnel area before the runnels were constructed (data collected by Save the Bay).

Figure 4. Mean *Spartina alterniflora* stem density $\pm$ 1 SE (per 26cm diameter plot).

Figure 5. Mean belowground biomass per core $\pm$ 1 SE.

Figure 6. Mean belowground percent organic matter ± 1 SE.

Figure 7. Mean carbon dioxide fluxes per treatment ± 1 SE. Sediment Enhancement does not have error bars or values if it did not produce significant fluxes at time of collection ($R^2 < 0.9$).

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CHAPTER 3

SUCCESSFUL IMPLEMENTATION OF ADAPTIVE MANAGEMENT INTO A CLIMATE CHANGE ADAPTATION STRATEGY

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Abstract

Due to climate change and other anthropogenic stressors, future conditions and impacts facing coastal habitats are unclear to coastal resource managers. Adaptive management strategies have become an important tactic to compensate for the unknown environmental conditions that coastal managers and restoration ecologists face. Adaptive management requires extensive planning and resources, which can act as a barrier to achieve a successful project. These barriers also create challenges in incorporating adaptive management into climate change adaptation strategies. The Rhode Island Coastal Resource Management Council overcame these challenges and implemented a successful adaptive management approach to restore a drowning salt marsh using the climate adaptation strategy, sediment enhancement, at Quonochontaug Pond in Charlestown, RI. Through effective communication and active stakeholder involvement, this project successfully incorporated interdisciplinary partner and stakeholder collaboration and developed an iterative learning strategy that highlights the adaptive management method.

1. Introduction

Much research has been conducted on climate change mitigation, but comparatively less attention has focused on implementation of adaptive management strategies to protect environments impacted by climate change (IPCC., 2014). Accelerated relative sea level rise (hereafter referred to as SLR) rates are a major effect of climate change and are a serious threat to coastal environments throughout the Northeast USA (Ashton et al., 2008; Carey et al., 2017; Weston, 2014). New England itself is facing SLR rates that are three or four times the global average (Sallenger et al., 2012). These elevated rates are likely to cause increased flooding, damage to infrastructure in low-lying and coastal areas, decreased resiliency to storms, and loss of coastal wetlands, including salt marshes (Ashton et al., 2008; Wigand et al., 2017). Climate adaptation focuses on enhancing resilience to current and future climate change impacts including SLR, which will help in managing and maintaining coastal ecosystems such as salt marshes (Stein et al., 2013; Wigand et al., 2017).

Climate change and other anthropogenic impacts have lowered the resiliency of Northeast coastal marshes. Salt marshes serve as a carbon sink, food source, breeding habitat, and nursery ground for birds (including the vulnerable salt marsh sparrow, *Ammodramus caudacutus*), fish, and shellfish (Bayard and Elphick, 2011; Hanson and Shriver, 2006; Raposa and Roman, 2006). These environments also provide flood abatement and help prevent coastal erosion (Barbier et al., 2011; Leonard and Luther, 1995). Historically, lateral transgression and vertical accretion of New England marshes have been able to keep pace with SLR (Raposa et al., 2017; Redfield, 1972). However due to increased coastal development, reduced sediment

supplies (caused by urbanization, dam construction, and reforestation), and accelerating rates of SLR, marshes are no longer able to migrate or accrete at a rate fast enough to withstand SLR impacts (Sallenger et al., 2012; Weston, 2014; Watson et al., 2017). As a result of these impacts, Northeast marshes, including those in New England, have suffered from increased dieback areas, vegetation loss, peat subsidence, waterlogged soils, and ponding (Hartig et al., 2000; Alber et al., 2008; Raposa et al., 2017). SLR has also exacerbated salt marsh erosion as a result of increased crab burrows in high marsh areas, due to waterlogged soils (Crotty et al., 2017; Raposa et al., 2018). These combined effects further decrease salt marsh resiliency in light of storms and climate change impacts, which the Northeast is particularly susceptible to (Frumhoff et al., 2007; Kirwan and Megonigal, 2013; Crotty et al., 2017).

Climate change adaptation is a management strategy that addresses climate-related vulnerabilities of susceptible habitats and focuses on preparing for, coping with, and responding to the impacts of current and future system changes (Stein et al., 2013; Wigand et al., 2017). Investing in climate change adaptation projects can increase coastal resiliency to environmental threats and minimize damages (monetary and environmental) from storm events (Narayan et al., 2017; Sutton-Grier et al., 2015). Climate adaptation strategies have been implemented across the U.S. (including living shorelines, green infrastructure, green roofs, flood abatement strategies, irrigation efficiency for agricultural practices, etc.) on the federal, state, local/regional, and private sectors (Bierbaum et al., 2013). One climate adaptation approach to build salt marsh resiliency is sediment enhancement (SE), also known as thin layer deposition where dredged sediment material is added to the salt marsh surface (Cahoon et al.,

2019). The purpose of this technique is to raise the salt marsh platform to an elevation that can withstand future projections of SLR. Although climate adaptation strategies have been adopted nationwide, the incorporation of adaptive management within these projects is uncommon.

Adaptive management incorporates learning-based decision making into management actions (Salafsky et al., 2001; Allen and Gunderson, 2010; Williams, 2011). This strategy is an iterative learning process that allows management actions to proceed despite uncertainty and requires changes in action to improve the management strategy as knowledge and understanding increases (Allen and Gunderson, 2010; Williams, 2011). There is a benefit to this strategy that accounts for uncertain and unexpected responses of a management action, but adaptive management involves challenges that must be overcome. Lack of resources and communication, disorganized coordination and leadership, inherent lack of flexibility within institutions, minimized stakeholder engagement, and action procrastination and avoidance can inevitably lead to adaptive management failure (Adger et al., 2009; Allen and Gunderson, 2011; Bierbaum et al., 2013; McNeeley, 2012). Since adaptive management requires a monitoring component, a larger commitment of time and resources is needed, which can pose an additional challenge. These challenges provide barriers to incorporating adaptive management into climate adaptation projects and require intensive planning to overcome.

The Quonochontaug (Quonnie) project, a state-run and federally funded initiative lead by the Coastal Resource Management Council (CRMC), incorporates the SE climate change adaptation strategy and adaptive management while integrating

lessons learned from past SE projects. This paper describes the successful incorporation of adaptive management into the Quonnie SE project and highlights the use of collaboration and outreach in restoration initiatives. We discuss how adaptive management components: 1) Create a project model 2) Establish a clear and common purpose/action 3) Develop and implement a management and monitoring plan 4) Analyze results and iterate 5) Communicate results, were applied for the successful implementation of the Quonnie climate change adaptation project (Salafsky et al., 2001).

2. Quonochontaug (Quonnie) sediment enhancement project overview

Quonnie Pond is located along the southern coast of RI, in the towns of Charlestown and Westerly. SLR impacts caused degradation and drowning of salt marsh areas within Quonnie pond and other parts of southern New England (Watson et al., 2017). The purpose of the SE project was to restore 30 acres of salt marsh habitat and improve tidal flushing and increase eelgrass habitat via dredging, create conditions and elevations suitable of targeted vegetation species, and to improve salt marsh resiliency to SLR and climate change impacts. Dredging improved recreational access to the Pond by removing sediment that hinders navigation and limits water recreation.

3. Establishing the climate change adaptation project: Identifying stakeholders and partners

3.1 Establishing the salt marsh climate change adaptation and adaptive management team

For the Quonnie sediment enhancement adaptive management (Q-SEAM) project, the initial goal was to gather together organizations and people dedicated to salt marsh protection, including agencies experienced in assessing salt marsh vulnerability and condition and implementing restoration actions. This required the expertise of federal, state, and local agencies, as well as non-profit and non-government organizations (NGOs); all held specific roles and responsibilities (Table 1). The creation of this team occurred during the stage of initial assessment of salt marsh condition, prior to the SE implementation.

3.2 Initial salt marsh condition assessment

Rhode Island follows the Salt Marsh Monitoring and Assessment Program (SMMAP) (Raposa et al., 2016). SMAPP monitoring helped identify the degrading marsh conditions and provided the necessary data to support the SE initiative at the Quonnie Pond site (Figure 1). This monitoring involved the rapid assessment of marsh conditions with marsh site visits across the state. Monitoring showed an abundance of ponding and vegetation die-off areas and the displacement of high marsh plants by low marsh plant species within the Quonnie salt marsh (Cole Ekberg et al., 2017; Kutcher, 2019). This site was also identified to have relatively low surface elevation within the tidal frame and was characterized as an area of high disturbance (i.e. high density of human-made ditches, crab burrows, and edge erosion) (Kutcher, 2019).

The Sea Level Affecting Marshes Model (SLAMM) simulates the response of salt marsh areas to varying SLR rate scenarios (SLAMM, 2009). Results of the SLAMM model simulations help evaluate marsh migration potential and prioritize appropriate marsh adaption and restoration efforts (Cole Ekberg et al., 2017; Wigand

et al., 2017). The Quonnie SLAMM results predicted significant marsh loss with 1m of SLR within the next 40-50 years and recognized limited potential for salt marsh migration

(http://www.crmc.ri.gov/maps/maps_slamm/20150331_RISLAMM_Summary.pdf).

These results and the SMMAP monitoring helped determine the SE treatment as an appropriate climate adaptation strategy for this site.

4. Quonnie sediment enhancement adaptive management project

4.1 Quonnie project model

Iteration is a major theme in adaptive management; Q-SEAM incorporated methods and lessons learned from a previous SE project at Ninigret Pond in Charlestown, RI. Q-SEAM adapted the same Before, After, Control, Impact (BACI) experimental design model as the Ninigret project, where the control (area where no management action took place) and impact (sediment enhancement) sites were monitored before and after treatment (Smith, 2014). The model incorporated monitoring that would occur for at least five years after sediment placement. It was hypothesized that the control would show signs of degradation (displacement of high marsh plants by low marsh plants, increase in vegetation die-off areas, loss of soil organic carbon, loss of habitat value) over time, while the impact area would gradually recolonize vegetation and nekton communities and accumulate soil organic matter over the five-year monitoring period. Project targets and metrics (Table 2) were incorporated into the BACI model to guide learning. To optimize results and enhance the project, communication, construction, and monitoring techniques learned from the Ninigret project were incorporated in the Q-SEAM plans (Table 3). Results learned

from the BACI monitoring and analyses would inform future decision making for Quonnie maintenance as well as future SE projects.

Important stakeholder communication techniques and construction and field strategies were learned and adapted for Q-SEAM to help gain project support and improve management strategies (Table 3). For example, dredging methods used at Ninigret were altered and improved for the Quonnie project (RTK mounted equipment and amphibious and low ground pressure equipment). Earlier monitoring at Ninigret taught the Q-SEAM team that intensive post-construction sediment grading (to ensure target elevations were met and establish drainage) was needed, that geese would use the area for foraging, and that excessive wind and sediment movement could impact the target elevations. By being aware of these potential issues, Q-SEAM project managers were able to incorporate actions (i.e. goose fencing; beach grass and dune fencing placement for wind protection and sediment stabilization) into the management plan, which were expected to have positive results on maintaining target elevations and subsequent plant colonization.

4.2 Establish a common purpose/action

An important initial adaptive management step was to create a clear project mission that was discussed and agreed upon by all stakeholders. Addressing and recognizing stakeholder goals early on helped to avoid future complications, and it held the partners accountable and committed to their project responsibilities. While addressing the major goals of the project stakeholders, the mission statement was manageable and conveyed realistic expectations (Figure 2). CRMC leaders ensured they were clear and forthcoming about the roles of each stakeholder, the logistics of

the project and their impacts on stakeholders' goals, which helped to manage stakeholder expectations. CRMC went through a negotiation process with the Town of Charlestown and the Salt Ponds Coalitions before agreeing on the amount of sediment to be dredged. Although concessions and compromises were made (Town of Charlestown provided more funds to dredge additional sediment and determined the dredging areas), CRMC ensured that the stakeholders' needs were heard and considered, which further helped to establish trust and commitment amongst the stakeholders and partners.

4.3 Development and implementation of a management and monitoring plan

CRMC and the monitoring partners collaborated to create the Quonnie Quality Assurance Project Plan (QAPP), which included a flexible management and monitoring plan that allowed for learning and monitoring plan adjustments, highlighting the adaptive management approach. The QAPP included project targets and metrics such as elevation, vegetation community, and wildlife community (Table 2) and methods to assess these targets. Monitoring these targets was essential to evaluate marsh function and restoration progress as well as for the learning needed to support future decision-making and management plan adjustments.

CRMC sought partner and stakeholder feedback and input throughout the development of the adaptive management plan via meetings and public presentations to municipal commissions. This allowed for stakeholders to voice concerns and identify issues early, and for the project team to address them in a manner that aligned with the project's goals and targets. CRMC maintained open and frequent communication with the project stakeholders, and shared project designs and plans as

they were developed. This transparency built trust with the stakeholders, and also allowed CRMC to address concerns early and rectify issues to prevent future conflict.

Having a clear management and construction plan to convey to the dredging company, J. F. Brennan Company, Inc. (hereafter J. F. Brennan), helped with communication and collaboration. CRMC ensured that the construction plans for J. F. Brennan were detailed enough for design implementation, but were flexible enough to incorporate contractor expertise and methodologies. CRMC and J. F. Brennan went through an iterative process throughout construction, where adjustments to the construction plan and design were made as necessary and as the project progressed. J. F. Brennan appreciated having their inputs valued. One of the lead constructors in an interview said, "They [CRMC] look to us for ideas and value our opinion...the process is made easier because they are open and upfront." Establishing two-way communication between hired contractors, where contractors' ideas and expertise were respected, considered, and incorporated, enhanced the outcome of Q-SEAM and highlights the learning/adaptive component of adaptive management.

The monitoring plan was helpful in establishing goals and parameters as well as the responsibilities of each partner, which in turn kept the partners accountable. Monitoring occurred during the peak growing season, between mid-August and mid-September before sediment placement and the first season after placement and was intended to continue for four additional growing seasons thereafter. Monitoring partner meetings were held before each salt marsh growing season to discuss the parameters that would be measured, monitoring methods, and timelines as well as a meeting after the growing season to discuss monitoring results and adjustments for the

next season. Meetings were then scheduled as needed throughout the growing season to address unexpected issues and adjustments to the original monitoring/management plans. Outside of these meetings, the monitoring partners were in open and continuous communication to address questions as they arose.

4.4 Analyze results and iterate

As data was interpreted and field conditions became clearer, CRMC and partners had to adapt and learn from unexpected challenges, which sometimes called for adjustments to the QAPP and data collection methods. For example, the Quonnie site was more accessible than previous SE sites and civilians used the area as a recreational space. In response to this, signage and fencing were placed on the borders of the site and a separate area was designated as a recreational location (Figure 3a &b). Monitoring changes were needed as well, which included adjusted pH and soil salinity sampling methods due to the low moisture content of the dredge material. During construction, the Q-SEAM team learned that the use of one dredge versus two dredges would make the handling/distribution of dredge material more manageable and prevent sediment buildup. As adaptive management calls for, management and monitoring plans were adjusted accordingly as this new information arose. The flexibility of each monitoring partner and efficient communication allowed for quick responses to these unexpected outcomes and adjustments to original methods.

4.5 Communicate results

The Q-SEAM monitoring data were made available throughout the monitoring process to provide transparency, cultivate public engagement, and provide project updates, via the CRMC ArcGIS Online Quonochontaug Data Gallery

(<https://crmcgis.maps.arcgis.com/apps/MinimalGallery/index.html?appid=bfd4d36733c43fa93874e09414457e4>). The CRMC communicated SE project results through regional conference presentations and site visits with the community and regional agencies, and is currently developing supplemental material such as restoration guidance and lessons learned documents. Making information readily available helped maintain public involvement and interest in the project as well as educated other agencies that were interested in learning more about the SE restoration technique. Agencies including NBNERR and EPA Atlantic Coastal Environmental Sciences Division, communicate with other NERRs and EPA facilities across the country to help to further develop SE best practices and apply them to other sites.

5. Community outreach and engagement

Throughout the Quonnie project, outreach and community engagement was a continuous priority. During the early stages of the project, Charlestown members were brought in for site visits, and CRMC presented SE plans at town council meetings to help gain support for the project and improve understanding of the project's purpose. A Quonnie planting event, organized and facilitated by Save the Bay, was one of the largest outreach initiatives that occurred after sediment placement in the early spring of 2019. This event brought together school groups, Save the Bay volunteers as well as volunteers from various town organizations, project stakeholders and partners, and Charlestown citizens. Planting events allowed citizen volunteers to make a physical contribution and connection to the project (Figure 3c & d). CRMC sponsored short promotional videos to highlight the restoration that occurred in the state (<http://www.crmc.ri.gov/>). The Salt Ponds Coalition published an article about the

project in its newsletter, Tidal Page, as well as produced videos focused on the SE projects within the state. CRMC and monitoring partners continue to present at local, regional and national meetings to share their experiences and results with the SE technique.

6. Conclusions

The Q-SEAM project demonstrated that effective collaboration, efficient communication, community involvement, and outreach were necessary to overcome adaptive management challenges and achieve success. Collaboration was an integral part of the adaptive management approach as the Quonnie project required the expertise of multiple disciplines. Partnership and collaboration came with benefits including resource and cost sharing, division of responsibilities, development of management plans, and implementation of monitoring. However, challenges were associated with collaboration, which CRMC was able to overcome with frequent and open communication with partners, and guided, productive monitoring and project meetings. The partners established and held similar goals, which led to accountability, commitment, and timely follow through with actions.

Community involvement and outreach were instrumental components of the Q-SEAM project. Therefore, establishing trust and actively involving the community in the adaptive management approach was essential for the success of the project. CRMC operated under full transparency with the Town of Charlestown and other stakeholders, addressing their concerns early on and managing expectations. Establishing trust early with the stakeholders, through site visits, town and project planning meetings, was essential to gain stakeholder support and assistance. Involving

the community throughout the project grants the public an invested interest in its success.

Rhode Island's successful use of an adaptive management strategy to implement the SE climate change adaptation project is expected to influence future decision-making on coastal marsh restoration in the Northeast USA and beyond. An adaptive management strategy is a valuable tool due to its flexible nature that accounts for unexpected results and adjustable management and monitoring plans. Incorporating adaptive management strategies within climate change adaptation and resiliency projects becomes increasingly important as climate change progresses and future conditions are more uncertain.

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Figure 1. Describes agencies' roles in the initial assessment and proposal development of the Q-SEAM project.

Figure 2. Describes the main goals of the project stakeholders and the derived overall project mission.

Figure 3. A. Signage placed at Quonnie restoration site B. Designated recreational area for civilians at the Quonnie restoration site. C & D. Quonnie salt marsh planting community event

Table 1. Partners, stakeholders, and decision-makers and their roles for the Quonnie sediment enhancement project

Stakeholders	Agency Type	Role of Partners
Coastal Resource Management Council (CRMC)	State	Lead and supervisory organization; Responsible for planning and implementation of the project; Performed dredge sediment testing for hazardous material; Applied for funding, permitting; Legal responsibility; Organizer of stakeholder meetings; Executed and managed contracts for construction, planting, adaptive management and monitoring
RI Department of Environmental Management	State	Property owner, manager of public fishing and boating access, permitting entity, provided equipment for post-construction excavation (Office of Mosquito Abatement).
Town of Charlestown	State	Dredge permit applicant, provided non-federal match funding, some technical and conceptual design assistance, coordination with Harbor Master and Police Department re: public safety during construction
National Oceanic and Atmospheric Administration (NOAA)	Federal	Lead funder from NOAA Coastal Resilience Grants (FFO #: NOAA-NOS-NRPO-2017-2005159)
National Fish and Wildlife Foundation	Federal	Funder; Award from Hurricane Sandy Coastal Resiliency Program (leveraged federal construction funding)
Salt Ponds Coalition	Non-Governmental Watershed	Provided non-federal matching funds, public outreach and

	Organization (NGO)	education, letters of support for funding application
J. F. Brennan Company, Inc.	Contractor	Contractor for dredging and placement of material
Monitoring Partners		
Save The Bay (Narragansett Bay)	Non-profit, NGO	Construction oversight, Vegetation monitoring; Habitat restoration expertise; Volunteer coordination; Planting; Adaptive management in coordination with RIDEM
Rhode Island Natural History Survey	NGO	Initial MarshRAM site assessment of salt marsh condition (pre-dredge placement); Monitoring of vegetation community recovery and rare plant species
Environmental Protection Agency, Atlantic Ecology Division	Federal	Soils monitoring; Technical support on salt marsh monitoring and assessment; Consulted through US Army Core of Engineers permit process.
University of Connecticut's Saltmarsh Habitat and Avian Research Program	Educational Institution	Avian monitoring
University of Rhode Island Environmental Data Center	Educational Institution	Elevation monitoring; Hydrology monitoring; Acquisition of Unmanned Aerial Systems (drone) imagery; Development of ArcGIS online-based project data portal

Table 2. Monitoring targets for Quonnie sediment enhancement project

Monitoring Metric	Target/ Monitoring Goals
Saltmarsh habitat restored	30 acres
Eelgrass habitat restored	3 acres
Low marsh plant community elevation range	0.15-0.23m (0.5-0.75ft NAVD88)
High marsh plant (<i>Spartina patens</i> , <i>Juncus gerardii</i> , <i>Distichlis spicata</i>) community elevation range	0.23-0.46m (0.75-1.5ft NAVD88)
<i>Iva frutescens</i> community elevation range	0.38-0.53m (1.25-1.75ft NAVD88)
Nekton species	Summer flounder, winter flounder, striped bass, river herring, menhaden, tautog, American eel, bluefish, and scup

Table 3. Communication tips for working with the town, public, and other stakeholders

1. Make clear how the project's goals align with their goals
2. Avoid the use of jargon and use terms they are familiar with
3. Explain how the project will benefit them. Relate the project to issues they care about.
4. When speaking with legislature, highlight how the project will address public health and safety
5. Listen to and address concerns. Make their voices and needs heard, which helps to establish trust.
6. Engage the community throughout the process with site visits, updates, and town meetings.
7. Communicate often with stakeholders and partners with meetings and updates

Table 4. Permits needed for the 30-acre Quonnie sediment enhancement dredge project

Agency Issued	Permit
US Army Corps of Engineers	Section 404 Category II General Permit
RI Department of Environmental Management	Dredging Permit (includes Section 401 Water Quality Certification)
Coastal Resource Management Council	Dredging Permit / Coastal Assent
NOAA served as lead federal agency	National Environmental Policy Act (NEPA) Compliance (includes sign-off from State and Tribal Historic Preservation Officers)

Table 5. Quonnie and Ninigret timeline of major events and project progression.

August 2011- Salt marsh condition assessment by Save The Bay
September 2012- Meeting with National Park Service about Jamaica Bay Thin Layer Deposition Project (Big Egg)
2013- Meetings with town, Salt Ponds Coalition and partners; Ninigret funding proposal development
May 2013- Funding proposal submitted for Ninigret construction, Quonnie design
July 2013- Site visit to Ninigret and Quonochontaug (Quonnie) with partners
January 2014- Regional thin layer deposition meeting on Long Island
October 2014- Ninigret award accepted
August 2015- Ninigret pre-restoration monitoring
September 2015- Ninigret consultant contracted for permitting and design
December 2016- Ninigret project designed, permitted and implemented
2017- Quonnie designs developed; Project team meetings for design review; Quonnie permit applications developed
July 2017- Applied for NOAA funding for Quonnie construction
November 2017- NOAA funding awarded
2018- Quonnie permits received

June 2018- Request for Proposal (RFP) issued for Quonnie construction work

August 2018- Quonnie pre-restoration monitoring

October 2018- Quonnie contractor hired, contract executed

November 2018- Mobilization of dredging equipment at Quonnie

December 2018- Quonnie dredging and placement

**January 2019- Demobilization of dredging equipment at Quonnie;
Quonnie As-built surveys**

March 2019- Post-construction adaptive management (excavation to ensure target elevations; drainage establishment)

May 2019- Quonnie planting event

August 2019- Quonnie post-restoration monitoring

Monitoring Activity

Outreach and Coordination Activity

Project Implementation Activity

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