Effect of Erosion Control Structures on Shoreline Marsh Species Populations

Final Report

Marsh sampling in Galveston Bay examining biological effects of erosion control structures.

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Abbreviations

GBEP	Galveston Bay Estuary Program
TCEQ	Texas Commission on Environmental Quality
TAMUG	Texas A&M University at Galveston
GB	Galveston Bay
POC	Percent Organic Content
ECS	Erosion Control Structure(s)

Executive Summary

Salt marsh restoration protects shorelines from storm surges and erosion, improves water quality, and provides critical nursery habitat and trophic support for commercially and recreationally important fishery species. Despite these important values, more than 8,000 acres of salt marshes have been lost across Galveston Bay (GB) (Texas, USA). Efforts to restore coastal wetlands in GB have used various restoration techniques, including the installation of erosion-control structures such as geotubes or breakwaters to help reduce wave action and sediment erosion. We investigated how these structures affect restored marsh community composition and productivity. We surveyed emergent marsh vegetation, epifauna, soil characteristics, and benthic microalgae in restored marshes with and without erosion control structures (ECS) in May and November 2022.

In both the fall and spring, aboveground plant biomass was higher at sites without ECS and belowground plant biomass was similar between sites with and without ECS. Total plant cover did not differ between sites with and without ECS in the spring but was higher at sites without ECS in the fall. The concentrations of the two abundant benthic microalgal groups, cyanobacteria and diatoms, were higher at sites without ECS. There was no significant difference in soil percent moisture between sites with and without ECS in either the spring or fall. Percent soil organic content was higher at sites without ECS in the fall, but did not differ between site types in the spring. In both time periods, snail density was higher at sites with ECS and crab burrow densities were similar between site types. There were few effects of ECS on the zooplankton or nekton communities in either spring or fall, though a few species varied among site types.

These findings indicate that ECS have stronger effects on benthic fauna and microalgae than on emergent vegetation. In addition, this provides further evidence that assessments of restored salt marshes that focus just on emergent plant cover may yield an incomplete picture of restoration success.

Introduction

Coastal inundation in the GB region is increasing in magnitude and frequency as more extreme storms (leading to both storm surge and heavy rainfall events) interact with onshore winds, greater flood tides, and erosion. Shoreline marsh areas are critical habitats that provide protection from erosion and floods, and they appear to perform better than bulkheads in maintaining sediment and elevation after major storms (Gitman et al. 2014, Smith et al. 2017). Even narrow, fringing marshes can dramatically attenuate floodwaters (Shepard et al. 2011). Marshes are also key nursery grounds for a wide variety of fauna, including juvenile stages of fisheries species and their prey items such as grass shrimp, penaeid shrimp, various crabs and snails, killifish, and juvenile drum.

Shoreline marshes are sometimes protected with breakwaters or sills—typically rock, concrete, or oyster shell—meant to further reduce erosion from overtopping storm waves, severe floods, or in some locations, vessel-generated waves. To date, the evidence that breakwaters enhance marsh habitat resiliency itself is mixed. After hurricanes Isabel and Irene, marsh areas in North Carolina with and without sills maintained elevation (or gained it), and vegetation itself was remarkably resilient regardless of sill presence or absence (Currin et al. 2008, Gittman et al. 2014). Yet other researchers have found that rock-based sills increased marsh resilience to Hurricane Matthew (Smith et al. 2018). Oyster-shell breakwaters promoted sediment accretion and marsh development under monsoon conditions in Bangladesh (Chowdhury et al. 2019). Given this mixed evidence, managers may understandably want to know how breakwaters influence the marsh community more broadly when designing projects and deploying scarce resources. Importantly, managers are lacking critical information needed to assess where and when breakwaters are protective of or detrimental to biological activity, diversity, and productivity in marshes.

Project Significance and Background

Breakwaters and sills are designed to reduce the exchange of momentum such as wave impacts and other hydrodynamic forces and materials, especially sediment, between the open bay and the marsh. Because of this, they could also affect populations of animals living in marsh habitats. Yet whether such impacts are beneficial or detrimental remains poorly explored. It has often been assumed (but rarely tested) that as long as vegetation remains intact, whether there is a breakwater present or not, resident fauna populations remain stable and the composition of the community does not change. Yet there is almost no evidence that this assumption is valid. One study on marsh nekton in North Carolina suggests marshes with sills may have higher nekton densities (Gittman et al. 2016), but the study site was located in a region with predictable tide-driven water levels and comparatively rare flooding. It remains unclear whether that work can be generalized to GB, where tidal flooding dynamics are unpredictable and weather-driven. Since breakwaters reduce material exchange with the open bay, they could also limit nutrient and zooplankton transport into the marsh, including juveniles (larvae) of marsh invertebrates. This could reduce recruitment and productivity of marshes with breakwaters relative to those without. However, sills could also serve to retain larvae produced locally within the marsh. Research in Alabama suggests that modified breakwaters with open channels can be particularly important for nekton egress and ingress to marsh habitats (Sharma et al. 2016). The relatively limited and mixed research on these topics makes it difficult for managers to assess where and when breakwaters might limit or enhance marsh fauna, complicating cost-benefit analyses.

To fill this research gap, this project asks how the presence of ECS influences biological populations and productivity within shoreline marsh areas. To address this question, we surveyed nekton and zooplankton in GB shoreline marsh areas with and without breakwater or sill structures, and examined productivity patterns by assessing plant cover, soil organic matter, above and belowground biomass, and benthic microalgae concentrations.

Relevance to Galveston Bay Plan Priorities

The 2017 Galveston Bay Report Card indicates that living resources associated with GB's wetlands are experiencing multiple stressors, including erosion (see Galveston Bay Estuary Program website[1]). To mitigate this stress, biotic and abiotic erosion control strategies have been implemented around the bay. The direct consequences of these strategies on floral and faunal species are unknown. Therefore, this project aimed to implement RES-3 (Conduct Physical Stressor Monitoring and Research) by quantifying the effects of a critical physical stressor (erosion) and erosion mitigation strategies on living resources (primary producer and invertebrate and larval fish relative abundance). In addition, this project supports RES-8 (Complete Coastal Resiliency and Acclimation Studies) by examining how well erosion mitigation strategies boost the potential productivity of the GB's living resources. This project addresses the Action Plan "Protect and Sustain Living Resources" through direct study of marsh species populations, habitat function, and productivity. The project also provides a first look at generally missing information on the biological impacts of key management actions (ECS) to enable effective science-based management of these critical coastal wetland habitats, thereby implementing the "Inform Science-Based Decision Making" Action Plan.

[1] https://gbep.texas.gov/

Methods

- Study sites and sampling plan
- General controls
- Field collections
    - Vascular plant cover
    - Benthic microalgal biomass
    - Aboveground and belowground plant biomass
    - Soil organic content
    - Benthic epifauna
    - Nekton
    - Larvae and zooplankton
- Data analysis

Study Sites and Sampling Plan

Eight study sites were selected in west GB; four sites were directly behind ECS (breakwaters or berms) and four were not behind such structures (Figure 1).

At each sampling site, a 50-meter transect was established parallel to the shoreline within two meters of the water-vegetation interface. Five replicate 0.25 square-meter plots were randomly placed along each transect. All samples and measurements were collected along these transects.

Field sampling was conducted in May 2022 and October 2022.

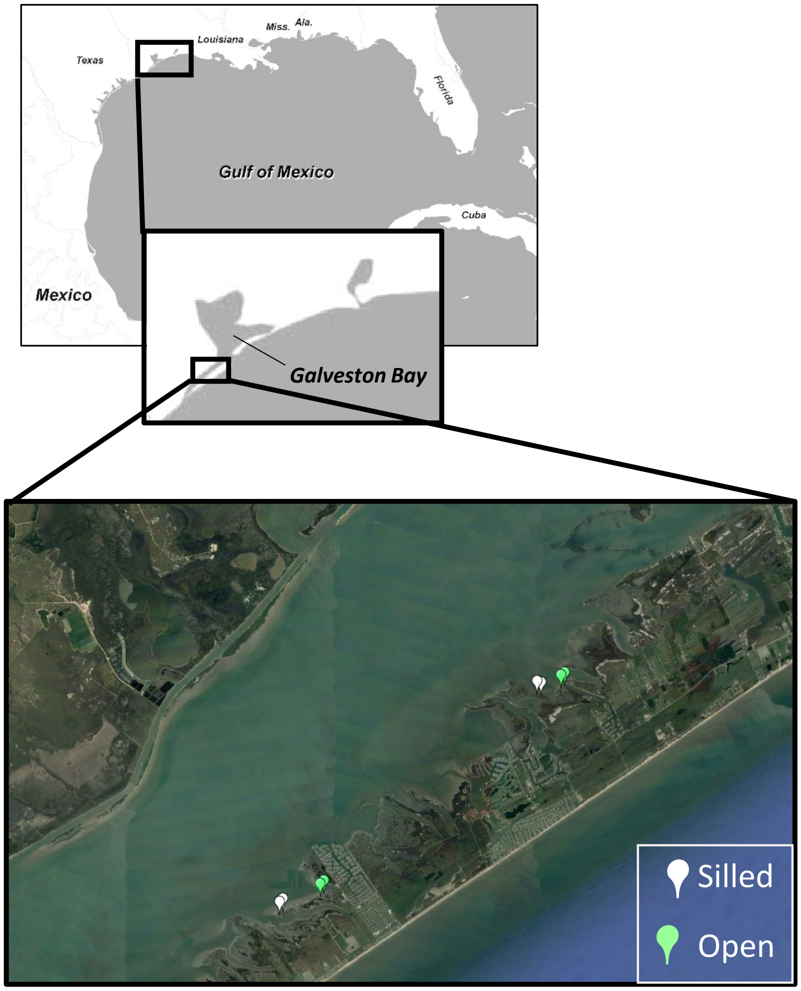

Figure 1. Map of study sites in west Galveston Bay.

General Controls

All controls outlined in the Quality Assurance Project Plan (QAPP 2022) were followed.

Field Collections

Vascular Plant Cover

Vascular plant percent cover was visually estimated in each plot (five/transect). To minimize observer bias, all cover values were recorded by a single observer.

Benthic Microalgal Biomass

Benthic microalgal biomass was measured once per plot (five/transect) using a BenthoTorch. The BenthoTorch uses the in vivo fluorescence of algal cells to measure algal biomass. Cell pigments are excited by different colors (wavelengths) of light and will emit red fluorescence light as a response to light stimulation. The BenthoTorch records the intensity of the chlorophyll fluorescence and uses an internal algorithm to calculate total algal biomass as well as the relative abundance of different algal groups, namely green algae, blue-green algae (cyanobacteria) and diatoms. After each reading, the results are shown on the display and stored in an internal memory bank. An encapsulated USB port enables the data to be transferred to a personal computer.

Aboveground and Belowground Plant Biomass

In each plot (five/transect), representative above- and belowground biomass samples were collected using a 10-centimeter diameter core. Aboveground biomass samples were clipped from the surface within the 10-cm diameter corer and transferred to plastic zip-close bags for transport to Texas A&M University at Galveston (TAMUG) for analysis. The 10-cm corer was then pushed into the sediment to a depth of 15-cm and the below-ground biomass within the core will be collected in zip-close bags for transport to TAMUG for analysis. In the laboratory at TAMUG, aboveground biomass samples were separated by species, rinsed with freshwater to remove salt and adhered sediment, dried at 60 degrees Celsius (˚C) for 48 hours, and weighed to determine biomass. Belowground biomass samples were rinsed through a two-millimeter sieve to remove salts and sediment, dried at 60˚C, and weighed to determine biomass.

Soil Organic Content

Soil samples were collected from each plot (five/transect) using a corer with a five-cm diameter and to a depth of five cm. The cores were transferred into zip-close bags for transport to TAMUG for analysis. In the laboratory, soil was dried at 60˚C for at least 48 hours, homogenized and weighed, and then the samples were burned in a furnace at 500˚C for eight hours and reweighed. The loss in weight on ignition was used to calculate the organic content of the soil.

Benthic Epifauna

In each plot (five/transect), an observer counted all epifauna of a size >one cm and fiddler crab burrows. This survey focused on the most common epifauna present in the marsh, snails, and fiddler crabs. Due to the mobile nature of fiddler crabs, standard practice is to use burrow density as a relative measure of crab abundance.

Nekton

Nekton samples (five/transect) were collected from the water adjacent to each plot using a cast-net (1.52-m diameter, 6.4-mm mesh). Nekton samples were placed in zip-close bags and transported back to TAMUG for analysis in the laboratory using a dissecting microscope. The samples were identified to the lowest practical taxonomic level, which varied based on size and age of the individuals, with resolution no less than phylum.

Larvae and Zooplankton

Larvae and zooplankton samples (five/transect) were collected from the water adjacent to each randomly placed quadrat using a hand pump. Ten liters of water were pumped through 300-micrometer mesh to collect larvae and zooplankton. The zooplankton sample collected in the cod end of the mesh was rinsed into a clean plastic jar with 70% ethanol and brought back to TAMUG for analysis using a dissecting microscope. The larvae/zooplankton samples will be processed at TAMUG on a dissecting microscope. The samples were sorted to phylum using the most recently published keys available.

Data Analysis

Above and belowground biomass data, percent plant cover, soil organic content, percent soil moisture, and benthic microalgal data were transformed as appropriate and then separate Analysis of Variance (ANOVA) tests were performed, using site type (ECS present or ECS absent) as the independent variable with site nested within the explanatory variable to account for differences among sites.

For data that could not be transformed to meet normal or homoskedastic assumptions, we performed Kruskal-Wallis nonparametric rank sum tests to determine differences between site types for spring data: percent soil organic content, crab burrow counts, and green algae concentration. In the fall, we applied this test to the following data: percent soil moisture, percent total plant cover, snail counts, crab burrow counts, and green algae concentration. For ANOVA and Kruskal-Wallis tests, differences between sites with and without ECS were designated as significant at $p < 0.05$.

Each season, the two most abundant zooplankton groups—amphipods and copepods across both seasons—were analyzed with separate quasi-poisson general linear models (GLM), where relative abundance was the response variable and site type was the explanatory variable. All other zooplankton and all nekton were analyzed using Kruskal-Wallis nonparametric rank sum tests on presence/absence data because of the low number of individuals found within each taxa identified. To determine differences between zooplankton and nekton community assemblages at sites with and without ECS, we used analysis of similarity (ANOSIM) on zooplankton and nekton separately for each season. ANOSIM generates an R-statistic between zero and one that is an indicator of effect size, where values < 0.25 indicate a high degree of overlap among assemblages (Clarke & Warwick 2001).

Differences between site types and site locations were determined significant at $p < 0.05$.

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Results and Observations

Spring

Aboveground plant biomass was higher at sites without ECS (ANOVA df = 1; F-stat = 4.245; p = 0.04) (Figure 2A). Belowground biomass was similar between sites with and without ECS (ANOVA df = 1; F-stat = 0.978; p = 0.33) (Figure 2). Percent plant cover did not differ between sites with and without ECS (ANOVA df = 1; F-stat = 1.894; p = 0.152) (Figure 3A).

Total algal concentration was higher at sites without ECS (ANOVA df = 1; F-stat = 6.318; p = 0.01) (Figure 4A). Cyanobacteria concentrations were higher at sites without ECS (ANOVA df = 1; F-stat = 12.997; p = 0.001) (Figure 4C). Green algae concentration did not differ between site types (Kruskal-Wallis df = 1; X^2 = 0.22; p = 0.63) (Figure 4E). Diatom concentrations were higher at sites without ECS (ANOVA df =1; F-stat = 8.297; p = 0.007) (Figure 4G), though there was substantial variation within sites without ECS (ANOVA df = 6; F-stat = 5.814; p = 0.0003).

There were no significant differences in percent soil moisture between sites with and without ECS (ANOVA df = 1; F-stat = 0.154; p = 0.69) (Figure 5A). ECS did not affect soil organic content (Kruskal-Wallis df = 1; X^2= 0.38; p = 0.53) (Figure 5C).

Snail density was higher at sites with ECS (ANOVA df = 1; F-stat = 18.561; p = 0.0001) (Figure 6A). Crab burrow density did not differ between sites with and without ECS (Kruskal-Wallis df = 1; X^2= 1.25; p = 0.2) (Figure 6C).

Overall, zooplankton communities were similar between sites with and without ECS (ANOSIM R = -0.002). However, a GLM indicated that copepods were more abundant when ECS were present (df = 39; t = 2.113; p = 0.04). All other zooplankton abundances did not differ between sites with and without ECS (Table 1).

Overall, nekton communities were similar between sites with and without ECS (ANOSIM R = 0.03). Of the nekton found, only Gulf menhaden (Kruskal-Wallis df = 1 X^2= 5.57; p = 0.01) and ctenophores (Kruskal-Wallis df = 1 X^2= 5.44; p = 0.01) differed between sites with and without ECS, with more found at sites without ECS. All other nekton abundances did not differ between sites with and without ECS (Table 2).

Fall

Plant aboveground biomass was higher at sites without ECS (ANOVA df = 1, F-stat = 4.298; p = 0.04) (Figure 2B). Belowground biomass was similar between sites with and without ECS (ANOVA df = 1; F-stat = 3.419; p = 0.07) (Figure 2D). There was a higher plant cover at sites without ECS (ANOVA df = 1; X^2= 7.066; p = 0.007) (Figure 3B).

Sites without ECS had a higher total algal concentration (ANOVA df = 1; F-stat = 7.036; p = 0.01; Figure 4B). Cyanobacteria concentrations were higher at sites without ECS (ANOVA df = 1; F-stat = 18.60; p = 0.0001) (Figure 4D). Green algae concentration did not differ between site types (Kruskal-Wallis df = 1; X^2= 0.21; p = 0.64) (Figure 4F). Diatom concentrations were higher at sites without ECS (ANOVA df =1; F-stat = 8.77; p = 0.005) (Figure 4H), but there was significant variation among sites without ECS.

There were no significant differences in percent soil moisture between sites with and without ECS (Kruskal-Wallis df = 1; X^2= 1.97; p = 0.15) (Figure 5B). Sites without ECS had higher soil organic content (ANOVA df = 1; F-stat = 6.91; p = 0.01) (Figure 5).

There were more snails at sites with ECS (Kruskal-Wallis df = 1; X^2= 3.77; p = 0.052), though the absolute magnitude of those differences was small (Figure 6B). Crab burrow density did not differ between sites with and without ECS (Kruskal-Wallis df = 1 X^2= 1.82; p = 0.17) (Figure 6D).

Overall, zooplankton communities were largely similar between sites with and without ECS (ANOSIM R = 0.08). These differences were largely attributed to chaetognaths (Kruskal-Wallis df = 1; X^2= 6.88; p = 0.008) and polychaetes (Kruskal-Wallis df = 1; X^2= 8.27; p = 0.004), which were more abundant at sites without ECS. All other zooplankton abundances did not differ between sites with and without ECS (Table 3).

Overall, nekton communities were similar between sites with and without ECS (ANOSIM R = 0.03). Of the nekton found, only ctenophores (Kruskal-Wallis df = 1; X^2= 12.65; p = 0.0003) and shrimp (Kruskal-Wallis df = 1; X^2= 5.75; p = 0.01) differed between sites with and without ECS with more of each group found at sites without ECS. All other nekton abundances did not differ between sites with and without ECS (Table 4).

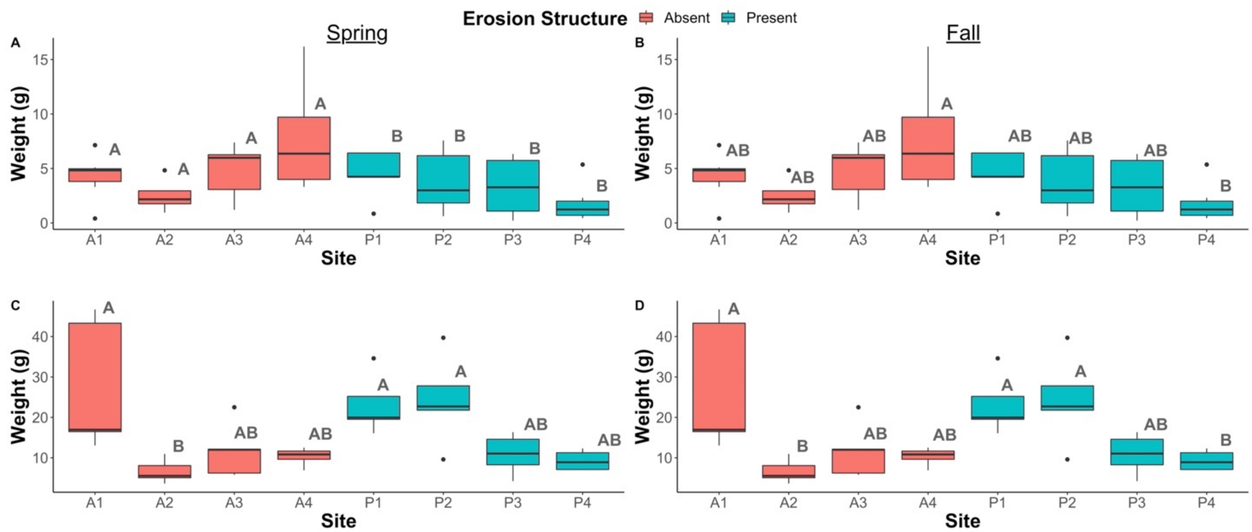

Figure 2. Boxplots of aboveground (A and B) and belowground (C and D) biomass. Panels A and C show data collected in spring 2022 and panels B and D show data collected in fall 2022. Peach bars represent sites without ECS and teal bars are sites with ECS present. Bars that do not share the same letters indicate significant differences based on ANOVA.

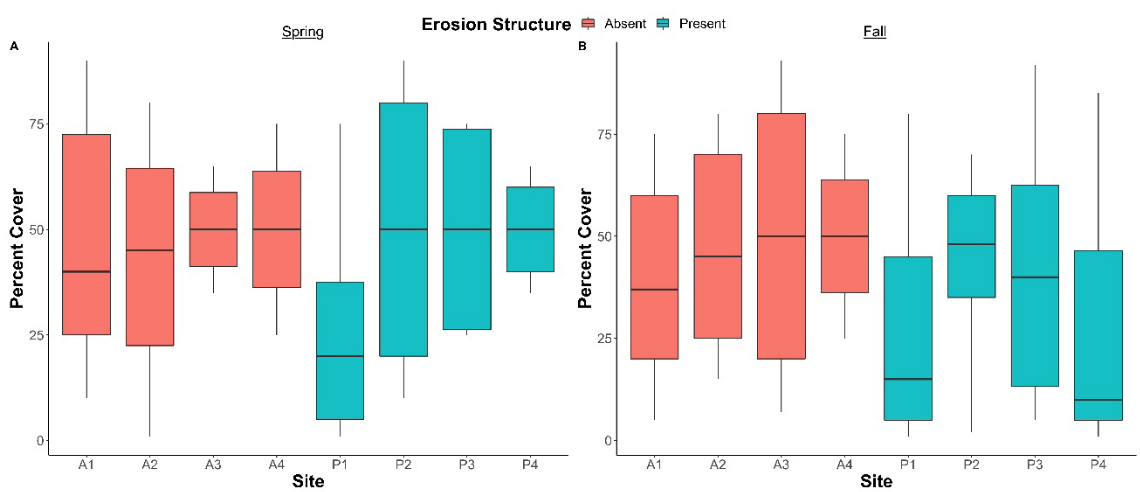

Figure 3. Boxplots of total percent plant cover in spring 2022 (A) and fall 2022 (B). Bars represent the total cover of all plant species present. Peach bars represent sites without ECS and teal bars are sites with ECS present. Percent cover was measured using a 0-100% scale. For spring, there were no significant differences among sites. For fall, there was significantly higher plant cover at sites without ECS, largely driven by the low cover at P1 and P4.

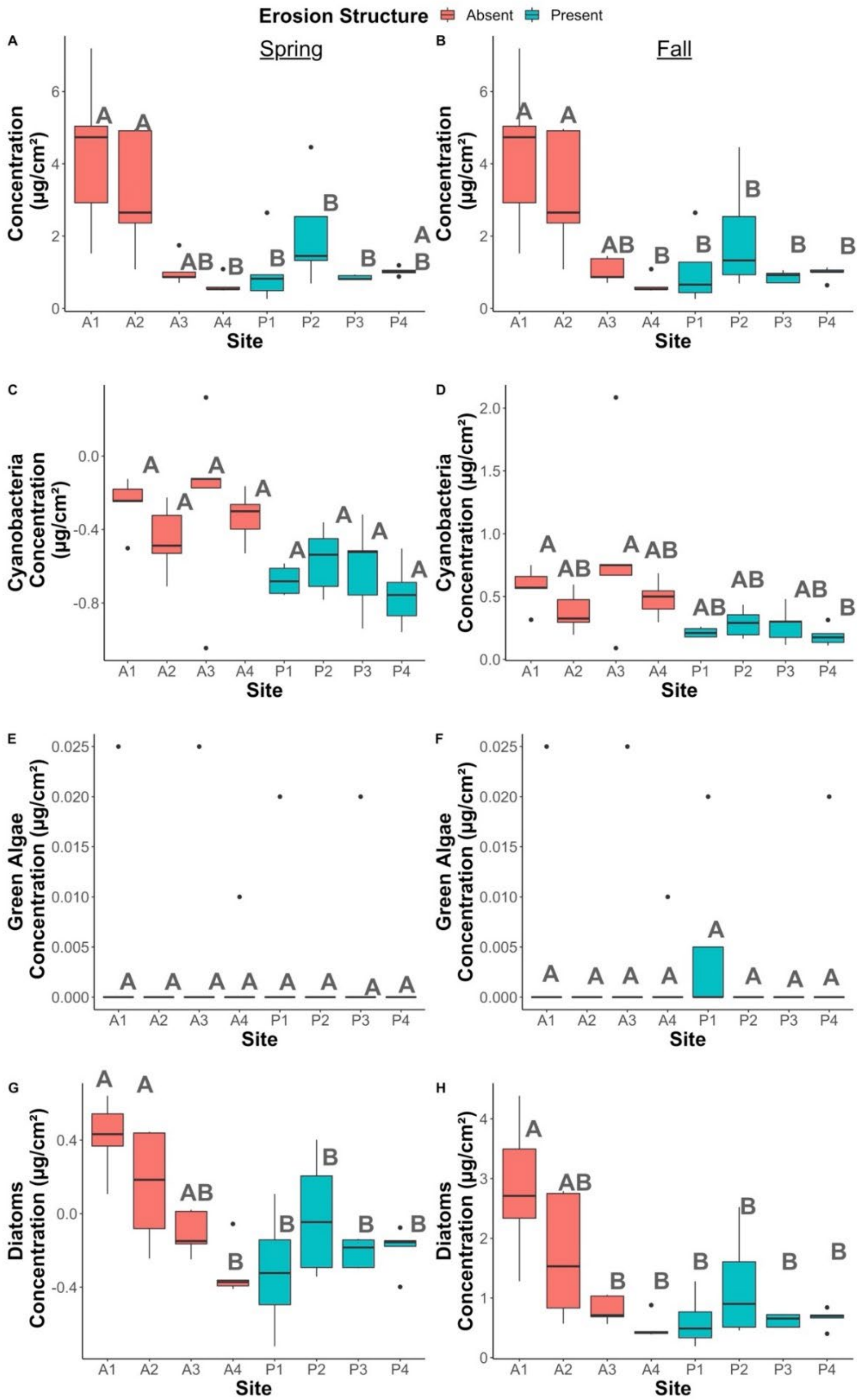

Figure 4. Boxplots of total benthic microalgal concentration (A and B), cyanobacteria concentration (C and D), green algae concentration (E and F), and diatom concentration (G and H). Panels in the left column show data collected in spring 2022 and panels in the right column show data collected in fall 2022. Peach bars represent sites without ECS and teal bars are sites with ECS present. Bars that do not share the same letters indicate significant differences based on ANOVA.

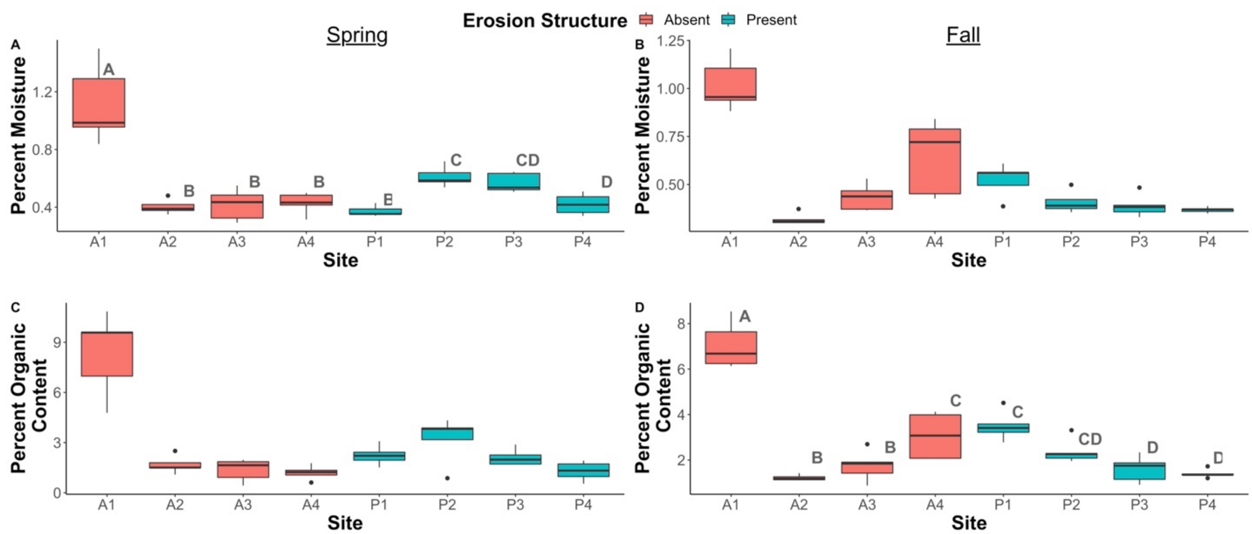

Figure 5. Boxplots of percent soil moisture (A and B), and percent organic content (C and D). Panels A and C show data collected in spring 2022 and panels B and D show data collected in fall 2022. Peach bars represent sites without ECS and teal bars are sites with ECS present. Bars that do not share the same letters indicate significant differences based on ANOVA (spring) or Kruskal-Wallis tests (fall).

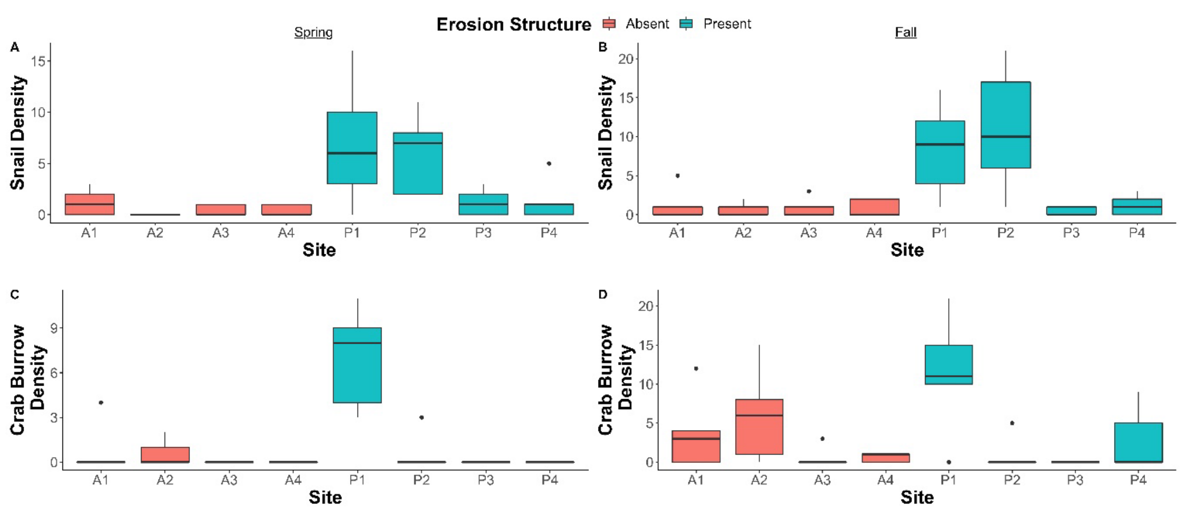

Figure 6. Boxplots of snail counts (A and B), and crab burrow counts (C and D). Panels A and C show data collected in spring 2022 and panels B and D show data collected in fall 2022. Peach bars represent sites without ECS and teal bars are sites with ECS present. There were no statistically significant differences among sites with and without ECS.

Table 1. Zooplankton caught in spring 2022. Mean abundances and standard error at sites with and without ECS. N/A indicates the taxa reported was not found within site type.

Taxa	Copepod	Decapod crab zoea	Decapod Shrimp Larvae	Megalops	Nematode	Amphipod	Mysid Shrimp	Jellies	Barnacle Nauplii	Polychaete
ECS	31.75 (+/- 2.61)	0.50 (+/- 0.41)	0.75 (+/- 0.55)	0.50 (+/- 0.41)	0.25 (+/- 0.50)	1.00 (+/- 0.58)	2.00 (+/- 0.87)	0.25 (+/- 0.50)	N/A	0.25 (+/- 0.50)
No ECS	8.75 (+/- 1.12)	0.25 (+/- 0.50)	0.25 (+/- 0.50)	N/A	N/A	2.50 (+/- 0.98)	1.00 (+/- 0.71)	N/A	0.25 (+/- 0.50)	0.25 (+/- 0.50)

Table 2. Nekton caught in spring 2022. Mean abundances and standard error at sites with and without ECS. N/A indicates the taxa reported was not found within site type.

Taxa	Pinfish	Menhaden	Mullet	Killifish	Rough Silverside	Sea Robin	Ctenophore	Shrimp
ECS	0.60 (+/- 0.82)	N/A	0.25 (+/- 0.55)	0.05 (+/- 0.22)	N/A	N/A	1.25 (+/- 1.62)	1.40 (+/- 1.76)
No ECS	0.45 (+/- 0.60)	0.70 (+/- 1.38)	0.40 (+/- 0.82)	0.05 (+/- 0.22)	0.35 (+/- 1.35)	0.05 (+/- 0.22)	2.35 (+/- 2.50)	1.05 (+/- 1.39)

Table 3. Zooplankton caught in fall 2022. Mean abundances and standard error at sites with and without ECS. N/A indicates the taxa reported was not found within site type.

Taxa	Copepod	Chaetognath	Barnacle Cyprid	Megalops	Cumacean	Amphipod	Mysid Shrimp	Jellies	Barnacle Nauplii	Polychaete
ECS	7.55 (+/- 15.56)	N/A	0.10 (+/- 0.45)	N/A	0.10 (+/- 0.31)	0.45 (+/- 1.61)	0.20 (+/- 0.70)	N/A	0.05 (+/- 0.22)	N/A
No ECS	15.45 (+/- 14.91)	0.40 (+/- 0.68)	0.05 (+/- 0.22)	0.05 (+/- 0.22)	N/A	1.70 (+/- 3.16)	0.50 (+/- 1.00)	0.05 (+/- 0.22)	0.05 (+/- 0.22)	0.50 (+/- 0.76)

Table 4. Nekton caught in fall 2022. Mean abundances and standard error at sites with and without ECS. N/A indicates the taxa reported was not found within site type.

Taxa	Speckled Trout	Killifish	Ctenophore	Shrimp
ECS	0.05 (+/- 0.22)	0.10 (+/- 0.31)	N/A	1.10 (+/- 1.37)
No ECS	N/A	N/A	3.55 (+/- 5.82)	0.25 (+/- 0.55)

Spring and Fall Comparisons

In both the fall and spring, aboveground biomass was higher at sites without ECS and belowground biomass was similar between sites with and without ECS. Total plant cover did not differ between sites with and without ECS in the spring but was higher at sites without ECS in the fall.

Cyanobacteria, green algae, and diatom concentrations followed the same patterns in the spring and fall. That is, the concentrations of the two abundant benthic microalgal groups, cyanobacteria and diatoms, were higher at sites without ECS. Green algae were relatively rare and did not differ between site types.

There was no significant difference in soil percent moisture between sites with and without ECS in either the spring or fall. Percent soil organic content was higher at sites without ECS in the fall but did not differ between site types in the spring.

Snail and crab burrow density trends were the same in the spring and fall, with higher snail density at sites with ECS and crab burrow densities similar between site types.

There were few effects of ECS on the zooplankton community in either spring or fall. In the spring, zooplankton community composition was similar between sites with and without ECS, though copepods were more abundant at sites with ECS. In the fall, overall zooplankton communities were similar between sites with and without ECS, though chaetognaths and polychaetes were more abundant at sites without ECS.

Overall, nekton communities did not differ between sites with and without ECS in the spring or fall, though there were some species-specific differences. In the spring, ctenophores and menhaden were significantly more abundant at sites without ECS, while ctenophores and shrimp were more abundant at sites without ECS in the fall.

Discussion

Overall, we found some evidence of differences between sites with and without ECS, but the results were inconsistent across seasons and high site-level variability may have masked more consistent differences.

Aboveground plant biomass was higher at sites without ECS in both seasons (and cover was higher at sites without ECS in the fall), though belowground biomass was similar. It is possible there is more tidal exchange at sites without ECS, which could enhance nutrients that boost aboveground growth (increased aboveground production is a known outcome of nutrient enrichment, Deegan et al 2012).

Soil characteristics were similar regardless of ECS presence. ECS do not appear to be facilitating or inhibiting the accumulation of organic matter. This is a bit surprising, but soil characters can be very slow to respond to environmental change (decades), so it may be that soil characteristics are more strongly linked to the restored nature of the site than the presence or absence of ECS on these timelines (e.g., all sites were created with sandy, low nutrient dredge material). Comparing sites of different ages with and without ECS would be useful if such sites exist in sufficient abundance.

Cyanobacteria and diatoms (benthic microalgae) were also found in higher concentrations at sites without ECS, which could again be related to greater tidal exchange and more nutrients. These patterns bear further examination and a larger project across more locations with a focus on quantifying tidal influx and nutrient influx is warranted.

For fauna, we found more snails at sites with ECS. It is possible that protected areas facilitate snails in some way; possibly via refugia from predators or stressors. Since there were more benthic microalgae at sites without ECS, the higher abundance of snails at sites with ECS is not likely to be food-resource related. We found no difference in crab abundances between sites with and without ECS.

There was substantial variability in zooplankton and nekton over space and time, but few community-level differences could be attributed to the presence of ECS. For those taxa that did differ by ECS presence, they tended to be more abundant in sites without ECS (with the exception of copepods in the spring, which may have experienced reduced dispersal or predation due to the ECS). The differences may have consequences for food webs, which is a subject that should be investigated further, though it may be hard to detect because of high mobility of predators.

Overall, there do appear to be some differences between sites with and without ECS, though it is difficult to draw conclusions about causation due to the pilot nature (low replication) of the experiment and high spatial and temporal variability. There is a clear need to target potential mechanisms with follow-up research both in the laboratory and field. The differences we did find between sites with and without ECS were relatively small in magnitude. Though detectable, they may not result in substantial differences in ecosystem function. This pilot study points to the need for a more rigorous and broader-scale study with more substantial investment, yet identifying suitable sites without ECS in the region to do such work is a challenge as ECS seem to be a standard part of most GB restoration projects.

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