The Efficacy of Living Shorelines for Restoring Shoreline Habitat and Stability

Final Report

GLO Contract No. 20-040-000-B748
September 2022

Prepared by:
Jim Dobberstine, Principal Investigator (PI) [1]
Amanda Hackney, Co-PI [1,3]
Cindy Howard, Co-PI [2]
Christine Miller, Co-PI [1]
Yihfen Yen, Co-PI [1]
and
Emily Blumentritt, Undergraduate Research Assistant [1]
Ryan Gilbert, Graduate Research Assistant [2]

Technical and content contributions by:
Todd Cameron (Microbiome bioinformatics)[4]
Myrah Urquidez (GIS Image analyses)[1,2]
Ashley Van Weiren (GIS Image analyses)[1,2]

[1] Lee College, Baytown TX; [2] The University of Houston-Clear Lake, Houston, TX, [3] BlackCat GIS & Biological LLC, Houston TX, [4] The University of Texas Health Science Center, Houston, TX

Submitted to:

The Texas General Land Office
1700 Congress Ave.
Austin, TX 78701-1495

A report funded by a Texas Coastal Management Program Grant approved by the Texas Land Commissioner pursuant to National Oceanic and Atmospheric Administration Award No. NA19NOS4190106.

Table of Contents

EXECUTIVE SUMMARY

Living Shoreline restoration efforts are based on the premise that vegetated wetlands form a buffer between high-energy water and adjacent land, limiting or reversing shoreline erosion. Additionally, research suggests that the fringing marsh (wetland edge) is exceptionally important habitat for many important fishery species. Unfortunately, erosion along Galveston Bay's shoreline has exceeded 4 feet per year in many areas. A common response to erosion is to armor the shoreline with a hardened structure such as a bulkhead that offers limited habitat benefits and may increase erosion on adjacent shorelines. As the Galveston Bay system has lost as much as 8% of estuarine emergent wetlands and more than 50% of the freshwater emergent wetlands present in the 1950's through erosion, ground subsidence, and habitat conversion, efforts to restore and protect these important aquatic habitats are a priority under the Galveston Bay Plan.

Like much of the Texas coast, a large proportion of aquatic habitat in Galveston Bay occurs along private land on relatively small individual parcels, making successful restoration of these shoreline habitats collectively important. Also, small projects may be quite large in terms of the amount of linear feet of critical shoreline habitat (i.e., fringing marsh) restored or created. Much effort and funding has been expended to restore and protect these habitats, and that effort is likely to increase moving forward based on programmatic efforts by state and federal agencies, and NGOs including the Galveston Bay Foundation, Restore America's Estuaries, and others. Thus, comprehensive data that can lead to improvements in project success is important toward ensuring funds expended (public and private) toward such efforts result in robust, resilient projects. These shoreline restoration/protection projects have the potential to reduce erosion along shorelines across a very large area in Galveston Bay alone, a priority under the Galveston Bay Plan and that of several local, state, and federal agencies.

Pilot data collected by the PI suggests Living Shoreline projects are an ecologically beneficial option for erosion control and property protection. However, much of the scientific data in the literature regarding ecologic function comes from larger scale habitat restoration projects rather than smaller, privately owned sites reflective of many Living Shoreline sites along the Texas coast. Data pertaining to ecologic function and resiliency has been requested by the Galveston Bay Estuary Program, Texas Parks and Wildlife, and the NOAA Restoration Center.

In this study, we hypothesize that the restored sites will perform similarly to natural sites across the parameters measured, although time may be required before community development achieves ecologic parity with comparable natural sites. To test this hypothesis, data was collected at three Living Shorelines sites within the Galveston Bay system to attempt to assess the resiliency and functional aspects (biotic and abiotic) of these small-scale restoration projects. This data was compared to unrestored natural marsh reference sites and traditionally armored sites near each Living Shoreline site, finding as follows.

1. Sediment heavy metals and organic contaminants are below the Effects Range Low (ERL) for all contaminants measured at all living shoreline sites (LVS) and their corresponding natural and armored reference sites, apart from copper measured at the Armand bayou living shoreline, which measured 0.8 ppm over the ERL. This suggests that neither heavy metals nor organic compounds would have a meaningful impact on the biotic aspects of these sites.
2. GIS analyses of aerial imagery indicate that all of the LVS sites have remained stable along the shorelines, with no measurable change of the shoreline(s) found. This was in contrast to one of the reference sites, and signs of early failure at one of the armored sites (both at Trinity Bay).
3. Plant community data, especially species importance values, species diversity, and total abundance indicate that the LVS sites have undergone successional change, and appear to be trending toward values more replicative of the reference sites over time. This was also noted in GIS comparisons of aerial images of the sites.
4. Sediment macrobenthic community data also exhibit evidence of successional development at the LVS sites, especially in measures of taxa richness and total abundance of organisms.
5. Sediment microbiologic community data are mixed, indicating key similarities for key species important to nitrification between the natural reference site and the living shoreline in Armand bayou, but also point to potential disturbance across sites at both Trinity Bay and West Galveston Bay, altering the microbiologic community across treatments.

Overall, the data support the hypothesis that living shorelines function similarly to their natural counterparts in their ecological functions, and are trending toward further parity over time through processes of ecological succession. They also appear to outperform armored sites across the same metrics. Both the biological data and analyses of aerial imagery at the sites also suggest that the LVS sites may be more resilient over time, in light of relative sea level rise and shoreline erosion prominent in many of the areas studied, in part as a result of construction methodology and resultant topographic features of the selected sites that provide buffers against erosion and upslope areas to which to migrate over time if needed. These features would in turn protect ecologically and economically important coastal features, including private lands and coastal aquatic habitat.

As much of the Galveston Bay (and other Texas coastal) shorelines are coupled to private lands, the cumulative benefit associated with numerous small-scale projects systemwide could be substantive. This research provides valuable information for the purpose of adaptive management to coastal restoration managers and the public, and supports the application of this approach to shorelines as a means of stabilizing erosion and restoring aquatic habitat, potentially providing ecologic and economic benefits to the bay system that would benefit the adjacent land owner and extend those benefits to other users of bay resources through incremental improvements to water quality, fishery resources, and related economic benefits associated with these ecosystem functions.

OUTREACH EFFORTS

The project has offered opportunities for outreach across multiple platforms.

1. The project has enriched the educational and professional experience of numerous students across two institutions, including funding the thesis work for one M.S. student, paid research assistantships for 11 undergraduate students, and degree-required practicum work for 10 students. The students are from diverse socioeconomic and ethnic backgrounds, supporting our joint institutional goals as minority serving institutions (M.S.I.). Further, the project aided in providing a transfer corridor between the institutions, allowing students to remain on the project without disruption.
    a. Graduate student participants: Ryan Gilbert, B.S. (UHCL)
    b. Undergraduate student participants: Jordan Anderson (UHCL), Emily Blumentritt (LC), Elizabeth Cornwell (UHCL), Renee Deng (UHCL), Steve Eggleston (UHCL), Jeffrey Fato (UHCL), Kevin Goldstein (UHCL), Courtney Melo (UHCL), Kristy Smith (LC & UHCL), Emily Torres (LC & UHCL), Belinda Torres (LC), Myrah Urquidez (LC & UHCL), Ashley Van Weiren (UHCL), Aiden Weaver (LC)
2. Presentations (Scientific)
    a. Dobberstine, J., and Howard, C. "The Efficacy of Living Shorelines for Restoring Shoreline Habitat and Stability in Galveston Bay, Texas". Restore America's Estuaries Biennial Coastal and Estuarine Summit 2022. New Orleans, LA, USA, December 5, 2022.
    b. Gilbert, R., Smith, K., Weaver, A., Miller, C., and Howard. C. "Examining Sediment Contaminant Levels in Living Shorelines in Comparison to Both Natural and Armored Shorelines in Galveston Bay, Texas." Restore America's Estuaries Biennial Coastal and Estuarine Summit 2022. New Orleans, LA, USA, December 5, 2022.
    c. Blumentritt, E., Van Wieren, A., Weaver, A., Cameron, T., Yen, Y., Dobberstine, J., Howard, C. "Examining Sediment Macrobenthic, Microbial, and Nekton Communities in Living Shoreline Restoration Sites in Galveston Bay, Texas." Restore America's Estuaries Biennial Coastal and Estuarine Summit 2022. New Orleans, LA, USA, December 5, 2022
    d. Van Weiren, A., Blumentritt, E., Urquidez, M., Gilbert, R., Hackney, A., Dobberstine, J. "Examining Shoreline Stability and Restored Plant Communities of Living Shorelines in Galveston Bay, Texas. Restore America's Estuaries Biennial Coastal and Estuarine Summit 2022. New Orleans, LA, USA, December 5, 2022
    e. Dobberstine, J., and Leija, H. "Evidence for Small-Scale Living Shoreline Viability in Galveston Bay, Texas". Quarterly meeting of the Monitoring and Research Subcommittee, Galveston Bay Estuary Program. June 8, 2022.
    f. Dobberstine, J., and Leija, H. "Evidence for Small-Scale Living Shoreline Viability in Galveston Bay, Texas". Quarterly meeting of the Natural Resource Use Subcommittee, Galveston Bay Estuary Program. June 22, 2022.
3. Presentations (Public)

a. Dobberstine, J., and Leija, H. “Evidence for Small-Scale Living Shoreline Viability in Galveston Bay, Texas”. Restore America’s Estuaries Coastal Resilience Webinar Series. June 15, 2022.
b. Dobberstine, J., Howard, C., and Leija, H. “Evidence for Small-Scale Living Shoreline Viability in Galveston Bay, Texas”. Galveston Bay Foundation Lunch and Learn. August 31, 2022.

LIST OF FIGURES

ACKNOWLEDGEMENTS

We would like to thank the following organizations and individuals for their support throughout the project:

- The Texas Coastal Management Program for their funding support, advice, and technical assistance throughout the project, including:
    - Jessica Chappell
    - Julie McIntire
    - Andrea Walmus
    - Sharon Moore
    - Maria Saenz
- The landowners who provided us with access and permission to conduct our study on their sites, including:
    - Warren Sullivan
    - Alice Anne O'Donell
    - Paul Abadie
    - Chris Saddler, Harris County Precinct 2
    - Diane Hume and the Clear Lake Forest Homeowners Association
- Our supporters with the Houston Advanced Research Center, USDA NRCS Baytown, USFWS Clear Lake Office, Moody Gardens, the Texas Coastal Watershed Program of Texas A&M University, the Armand Bayou Nature Center, and the Galveston Bay Estuary Program.
- Students and personnel that directly supported this research with their work and expertise in the field and laboratory, including:
    - Jordan Anderson
    - Emily Blumentritt
    - Elizabeth Cornwell
    - Robert Cummings
    - Renee Deng
    - Steve Eggleston
    - Lauren Eagle
    - Jeffrey Fato
    - Abby Ficklin
    - Becky Freeman
    - Trey Gammon
    - Ryan Gilbert
    - Kevin Goldstein
    - Zach Martin
    - Courtney Melo
    - Thomas Provost
    - Kristy Smith
    - Emily Torres
    - Belinda Torres
    - Myrah Urquidez
    - Ashley Van Weiren
    - Aiden Weave

- A very special thank you to Regina Barrera, Lee College Science Lab Coordinator, for her tireless effort procuring materials and equipment, managing onboarding of student researchers at LC, and assisting with field work when needed!
- Another very special thank you to Bill Gammel, Lee College Biology Instructor, for providing boat support to access sites as needed, ongoing guidance and assistance to students in the field, and valuable input on the research approach and methods!

INTRODUCTION

It is estimated that roughly 20% of the four million people that reside in the five-county area surrounding the Galveston Bay Estuary live within two miles of a bay and its tidally influenced tributaries (Lester and Gonzalez, 2002). The bay system is both ecologically rich and economically important to the State of Texas, and the United States (USEPA 2004). For example, the total economic impacts of recreational fishing in Galveston Bay is estimated to contribute more than 1600 jobs generating $55 million in income, $152 million in sales output, and $87 million in value added to the Texas GDP. These impacts were estimated to be twice that of any other bay system in Texas (Ropicki et al. 2016).

However, the system has endured substantial impacts to habitat that supports ecologic and economic inputs. Early studies of the Galveston Bay Estuary Program indicate that the Galveston Bay system lost more than 50% of emergent wetlands present in the 1950's through erosion, ground subsidence, and habitat conversion (White et al. 1992, Ward 1993). Further, the Galveston Bay shoreline has been eroding at a rate of 2.4 feet per year since 1932, affecting about 78% of the total shoreline (Lester and Gonzales 2002b). Previous studies have indicated that abiotic stressors to the aquatic environment can negatively impact habitat and resident biologic communities (Osland et al. 2022, Fujiwara et al. 2019, Mukaimi et al., 2018, Van Diggelen and Montagna 2016, Minello and Webb 1997). Further, as much of the adjacent land use bordering the upper bay system is urban, suburban, and/or industrial in nature, there is potential for toxicants including heavy metals and persistent organic contaminants to sequester to sediments, subsequently impacting aquatic biological populations directly or downstream through sediment migration from erosion (HARC 2020, EPA 2004, Guillen et al. 1999).

A common response to erosion is to armor the shoreline with a hardened structure such as a bulkhead that offers limited habitat benefits and may increase erosion on adjacent shorelines in addition to losing stability from undermining wave energy (Gittman et al. 2015; Rella and Miller, 2012; Dugan et al. 2008). Previous estimates indicate that approximately 10% of the Galveston Bay shoreline has been modified by hardened structures (e.g., bulkhead, etc.), and more than 19% is classified as developed (HARC 2020). These forms of shoreline have reduced biologic function relative to natural marsh, and have been implicated in changes to water quality , sediment distribution, and wave energy (HARC 2020, Gittman et al. 2016, Seitz et al. 2006).

Efforts to restore and protect aquatic habitats are a priority under the Galveston Bay Plan, resulting in several millions of dollars of created and restored wetlands along the Texas coast, including within Galveston Bay (EPA 2013). Under this plan, Living Shoreline (LS) restoration efforts have been developed, which are based on the premise that vegetated wetlands form a buffer between high-energy water and adjacent land, limiting or reversing shoreline erosion (TGLO 2020, GBF 2014). Additionally, research suggests that the fringing marsh at the wetland edge is exceptionally important habitat for many important fishery species, generating an estimated $3B per year to the local economy (Minello 2004, Whaley and Minello 2002, HARC 2020, USEPA 2004). It was reported to the Texas Commission on Environmental Quality (TCEQ) that the replacement value of wetlands within the Galveston Bay system would approximate

$5.7 billion (Ko 2007). Numerous such projects have been implemented across the Galveston Bay system. However, evaluation of Living Shorelines ecologic function is typically short-term and has not been evaluated for any enhancement of ecosystem services in comparison to natural and hardened shorelines (Gittman et al. 2016). Long-term data consistently drawn from the same sites is of particular value to understanding trends across biotic and abiotic factors, more so than data drawn on a single point in time. These trends can inform on a range of issues, including shoreline dynamics (e.g., erosion, elevation, etc.) allowing restoration managers to better apply adaptive management approaches that are both resilient and cost effective.

Pilot data collected by the Primary Investigator (PI) suggests LS projects within the Galveston Bay system are an ecologically beneficial option for erosion control and property protection (Torres et al. 2020). They may also have positive impacts on water quality (i.e., nonpoint source pollutant loads from sheet flow runoff) and aquatic habitat relative to armored shorelines (Gittman et al. 2016). This is of importance as much of the Galveston Bay watershed is heavily influenced by urbanization, which has been shown to impact contaminant loads on adjacent water quality (HARC 2020).

However, much of the scientific data in the literature regarding ecosystem function comes from larger scale habitat restoration projects rather than smaller, privately owned sites reflective of many LS sites along the Texas coast. This is important data to gather, as these smaller sites may serve an underrepresented, but important, ecologic role in aggregate. Locations within the Galveston Bay watershed have historical sea level rise, increasing the need for studies of this kind, as shoreline wetlands are likely to become further stressed from inundation and erosion as a result of ongoing sea level rise (Feagin, et al. 2005, Titus, 2000). Also, monitoring conducted by constructing entities post-construction is typically limited to basic plant establishment data over a short time frame. Ecologic function, water quality, and ecosystem resiliency data has previously been requested by the Galveston Bay Estuary Program (GBEP), Texas Parks and Wildlife, TGLO Coastal Management Program, NOAA Restoration Center, and others (Dobberstine et al. 2006). Measures including biologic community composition, total abundance, and other metrics across various trophic levels have been used to establish functional success for restored coastal marsh (Thayer et al. 2003). Examination of the chemical and physical aspects of shorelines that may impair or otherwise affect biological communities and critical aspects of the physical environment on which they depend has been used to aid in better understanding of resilience of coastal marsh (Thayer et al. 2005).

This project proposed to collect comprehensive data at Living Shorelines sites throughout the Galveston Bay system to attempt to assess the resiliency and functional aspects (biotic and abiotic) of these small-scale restoration projects. Data collected could also serve as a baseline to assess observable change and trends across metrics over time moving forward for the purpose of informing science-based decision making and adaptive management approaches for living shorelines within Galveston Bay and similar locations. We tested the hypothesis that the Living Shorelines sites will compare more similarly to natural sites across the parameters measured than to armored sites, with the caveat that time may be required before community development achieves ecologic parity with comparable natural sites.

STUDY AREA

The study sites reside within the Galveston Bay complex, located on the upper Texas coast consisting of three sub bays (e.g., Trinity Bay, East Bay, and West Bay), and numerous tributary systems including the Trinity and San Jacinto rivers, which flow into Galveston and Trinity Bays from the northwest. The system exits to the Gulf of Mexico through Bolivar Roads located between Bolivar Peninsula and Galveston Island, and San Luis Pass located between Galveston and Follett's islands. Together these features form a barrier island type estuary with a salinity gradient ranging from very fresh (e.g., 0 practical salinity units [psu] salt concentration), to nearly marine (e.g., 30 psu) (HARC 2020). This gradient, combined with adjacent and submerged land features, creates a variety of opportunity and stressors for the aquatic habitat and shoreline features across the approximately 1600 square kilometer system.

Thus, it was determined necessary to select study sites that were located in different areas across the bay system, choosing a site in the upper bay representing oligohaline (e.g., 0.5-5.0 psu), mesohaline (5.0-18.0 psu), or polyhaline (e.g., 18.0-30.0 psu) as the prevailing condition (Montagna 2020), as salinity is frequently a primary driver for floral development and composition in estuarine systems, and can impact other chemical conditions, including dissolved oxygen, impacting productivity (Montagna 2020 and citations therein).

Each study site was also selected by the availability of a natural reference marsh and an armored site within the same waterbody (preferable) or a nearby waterbody exposed to similar water chemistry conditions. This was done so that data gathered from each could be used as controls against which to compare data gathered from the study site, while attempting to minimize factors that would serve as covariables, complicating analyses.

The study sites and their paired reference sites were as follows (Figure 1).

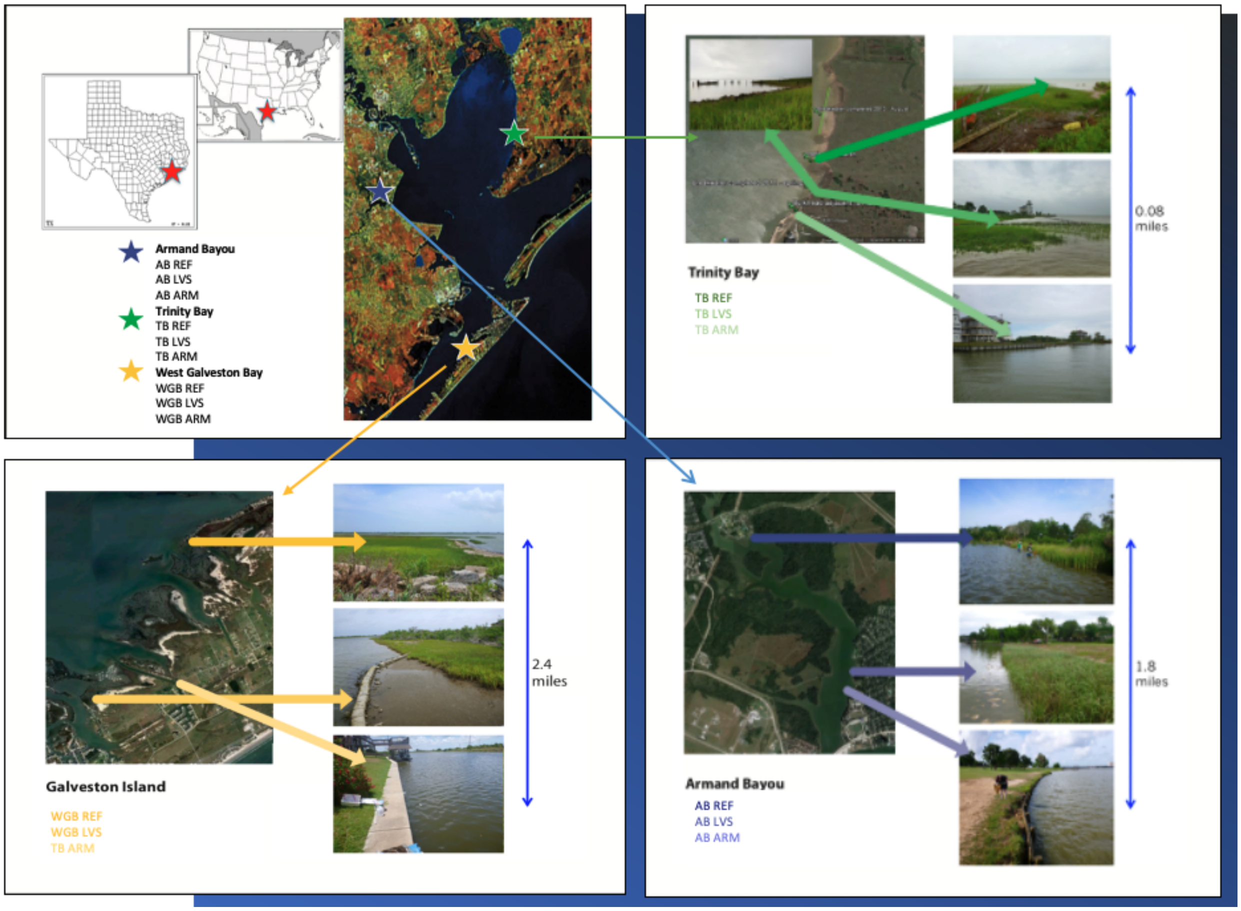


Figure 1. Study site locations.

<u>Armand Bayou (AB)</u>

Located on the west side of Galveston Bay, within the Clear Creek subwatershed.

- The study site is located adjacent to a residential neighborhood park, constructed by the homeowner's association in cooperation with the Galveston Bay Foundation. The Living Shoreline restoration was constructed in 2011 as a high-profile breakwater using rock (riprap) to provide a foot to retain clean fill, which was subsequently sprigged with Gulf Cordgrass (*Spartina alterniflora*) on approximate 3-foot centers.
- The natural reference marsh is located adjacent to a county park located approximately 1.5 miles upstream (northwest) of the study site. It is composed predominantly of a mix of Common Threesquare (*Schoenoplecus pungens*) and Gulf Cordgrass (*Spartina alterniflora*). The marsh is a small, emergent remnant section of the subsided shoreline. The site dates to the early 2000s, prior to which it was part of an upland forested complex dating to the 1940s.
- The armored site is located adjacent to a county park, approximately 0.75 mile downstream of the study site (southeast). The site is a wooden seawall (bulkhead).

Trinity Bay (TB)
Located along the east shoreline of Trinity Bay, downstream of the Trinity River and upstream of Double Bayou.

- The study site is located adjacent to private property and constructed by the landowner in cooperation with the Galveston Bay Foundation. The Living Shoreline restoration was constructed in 2011 as a high-profile breakwater using rock (riprap) to provide a retaining foot for the restoration marsh, which was subsequently sprigged with Gulf Cordgrass (*Spartina alterniflora*) on approximate 3-foot centers.
- The natural reference marsh is located immediately adjacent (south) to the study site, composed predominantly of a mix of Common Threesquare (*Schoenoplecus pungens*) and Gulf Cordgrass (*Spartina alterniflora*). The marsh is a small, emergent remnant section of the subsided and badly eroded shoreline. The site dates to the mid-1990s, prior to which the site existed as submerged mudflat as far back as 1970.
- The armored site is also located immediately adjacent to the study site (north). The site is a vinyl sheet pile seawall (bulkhead) fitted to an upland, residential property.

West Galveston Bay (WGB)
Located on the north side of Galveston Island.

- The study site is located adjacent to a residential property on Eckert Bayou, constructed by the property-owner in cooperation with the Galveston Bay Foundation. The site is part of a stretch of Living Shoreline constructed cooperatively by a number of property owners along the same shoreline extending approximately 0.3 miles from the study site toward West Galveston Bay along the east side of Eckert Bayou. The Living Shoreline restoration was constructed in 2005 as a low-profile breakwater using stacked concrete to provide a breakwater along the existing sandy shoreline. The site was subsequently sprigged with Gulf Cordgrass (*Spartina alterniflora*) on approximate 3-foot centers.
- The natural reference marsh is located adjacent to a small residential community located approximately 2.45 miles east of the study site and part of the Gangs Bayou complex on the bayside of Galveston Island. It is composed predominantly of Gulf Cordgrass (*Spartina alterniflora*). The site dates to at least the 1950s.
- The armored site is located between the study and reference sites, adjacent to a residential subdivision incorporating a system of dead-end canals approximately 1 mile east of the study site. The site is a concrete seawall (bulkhead) at the transition point from the bulkheaded Spanish Grant canal leading out to Starvation Cove on the bayside of Galveston Island.

METHODS

Field Sampling: Flora, fauna, sediment
All of the sites were sampled during June and/or July 2021. Nekton and ambient water conditions were also sampled in October 2022. The study site, reference site, and armored site

for each location (e.g., Trinity Bay, Armand Bayou, and West Galveston Bay) were sampled concurrently, as follows:

Day 1

- Ambient atmospheric conditions: Temperature, wind direction, and wind speed were measured using an ExoTech Thermo-Anemometer model 45118. Relative humidity was measured using a Bacharach 0012-7043 Sling Psychrometer.
- Ambient water conditions: Salinity (PSU), temperature (°C), dissolved oxygen (mg/L), and pH were measured using a Hanna Instruments HI98194 multiparameter meter. Turbidity was measured using a 60cm field turbidity tube.
- Plot-transect placement: A 10-meter transect line was stretched from a point on the waterside of each site (origin) in a transverse orientation across the marsh to the landward side (terminus). The origin and terminus were marked with a PVC stake and the GPS coordinates of the origin were recorded. $1m^2$ PVC plots were placed at each end of the transect (landward side of the transect) and one at the midpoint of the transect (waterward).
- Plant community: Each species of plant found within the $1m^2$ plot was identified, and stem count and relative coverage of each species was recorded for each plot.
- Spartina chlorophyll productivity: Leaf chlorophyll was measured at the 2nd leaf on 5 plants chosen at random from within each plot using an atLEAF chlorophyll CHL STD meter and recorded.
- Spartina biomass productivity: A 1/8 m^2 plot was placed at the midpoint of each $1m^2$ plot (e.g., origin, midpoint, or terminus), on the opposite side of the transect line. The roots and shoots of the plants from within the 1/8 m^2 plot were dug out using a sharpshooter shovel, bagged, and returned to the lab where they were sorted to remove the Spartina stems and roots, and washed using fresh water. The stems were then cut from the roots at the root collar for each plant. The roots and shoots were placed in separate aluminum dissecting trays, labeled, and dried in a Quincy Labs Model 40 GC drying oven at 90°C for 24 hours.
- Nekton community: Four 10"x10"x18" Promar collapsible minnow traps with 2-inch opening at each end were deployed at each site by placing them as close to the edge of the marsh as possible. At the armored site, placement was at a similar distance from the bulkhead to the trap as was the case with the study site. At each site, two traps were positioned parallel to the shoreline, and two were positioned parallel to the shoreline.

Day 2

- Nekton Community: Traps were collected 1 at a time at approximately 24 hours following deployment. Each was emptied prior to the next being retrieved, and specimens identified and enumerated, photographed in a Carolina Biological glass viewer, then released to the water. In instances where field identification was not possible, representative specimens were fixed in 10% buffered formalin and removed to the laboratory for later identification.
- Benthic macroinvertebrate community: 2"x 4" sediment core samples were taken using a Wildco stainless steel hand core sampler. Five cores were taken randomly

from within each $1m^2$ plot along the transect, equaling 15 samples total per transect. Whole sediment samples were visually inspected, placed in 6 ½" x 6" plastic "Ziploc" style bags, preserved with 10% buffered formalin, and transported to the lab for processing, staining, and sorting.

- Sediment microbial community: Two milliliters of soil/sediment was collected for each sample from the top one-inch layer of earth using sterile Eppendorf tubes pushed directly into the soil. For sediment submerged in water, a corer was used to help obtain the samples from the top layer of the sediment. Three cores were taken randomly from within each 1 m^2 plot along the transect, equaling 9 samples total per transect. Upon collection, samples were kept on ice in coolers and returned to the lab within the same day and stored at -20 C until processing.
- Sediment heavy metal contaminants: 2"x 4" sediment core samples were taken using a Wildco stainless steel hand core sampler equipped with a plastic sleeve and plastic core tip. 1 core was taken randomly from within each $1m^2$ plot along the transect, equaling 3 samples total per transect. Whole sediment samples were visually inspected, placed in 6 ½" x 6" plastic "Ziploc" style bags, preserved in a cooler on ice, and transported to the lab for where they were stored at 0°C until they could be processed and analyzed.
- Sediment organic contaminants: 2"x 4" sediment core samples were taken using a Wildco stainless steel hand core sampler and stainless steel core tip. 1 core was taken randomly from within each $1m^2$ plot along the transect, equaling 3 samples total per transect. Whole sediment samples were visually inspected, placed in 6 ½" x 6" plastic "Ziploc" style bags, preserved in a cooler on ice, and transported to the lab for where they were stored at 1.1°C until they could be processed and analyzed.

Laboratory processing: Flora, fauna, sediment

- Plant biomass productivity: Dried samples were removed from the drying oven at approximately 24 hours, and weights were recorded using an O'Haus Navigator XL electronic scale Model NVL1101/1. Shoot and root weight was recorded for each sample into an Excel-based data sheet for later analysis.
- Nekton Community: Any specimens returned to the lab from the field were identified using a dissecting microscope, and relevant taxonomic keys. All specimen identification data was then transferred to an Excel-based data sheet.
- Benthic macroinvertebrate community: Whole sediment samples were carefully washed individually through a #35 (0.5 mm) mesh sieve, using a gentle stream of tap water. All material remaining on the sieve was transferred to a sealed plastic jar and was re-preserved in 10% buffered formalin and stained with 50% Eosin B and 50% Sudan IV to facilitate sorting the organisms. Prior to sorting, the samples were rewashed over a #200 mesh sieve and re-preserved in ethanol. All benthic samples were sorted under low power on a stereo dissecting scope; organisms were identified to the lowest possible taxon, enumerated, recorded

and stored in vials. Data from lab data sheets was then recorded for each sample to an Excel-based datasheet for later analyses.

- Sediment microbiome: Genomic DNA of sediment microbes was extracted in triplicate from each of the 27 sediment samples using the Qiagen DNeasy PowerLyzer PowerSoil Kit according to the manufacturer's instructions. Specifically, frozen sediment samples were thawed at room temperature for 15 min, and 250 mg of sediment was placed into a bead tube containing 0.1 mm glass beads. Using a BeadBug (Benchmark Scientific), the sediment suspended in provided buffers was beaten at 4,000 rpm in four cycles of one-min bead beating and one-min ice-bath incubation. The extracted genomic DNA was further purified using a Zymo PCR Inhibitor Removal Kit and eluted with 100 ul of UltraPure DNase/RNase-free distilled water (ThermoFisher). DNA was quantified and quality-assessed using NanoDrop (ThermoFisher). Samples that have lower than the sequencing required concentrations were concentrated using a vacufuge (Eppendorf). PCR was subsequently performed to assess extraction efficiency by using 2X Taq Master Mix (New England Biolab), 5 ng/ul of extracted genomic DNA as template, and 10 uM of each 16s rRNA universal primer: 337F (5'-GACTCCTACGGGAGGCAGCAG-3') and 805R (5'-GACTACCAGGGTATCTAATCC-3'). PCR was performed in a 25 ul reaction with one cycle of denaturation at 95 C for 3 min, 25 cycles of denaturation at 95 C for 30 sec, annealing at 55 C for 30 sec, and extension at 68 C for 30 sec, followed by one cycle of five-minute extension. A total of 81 samples of microbial genomic DNA were then submitted to the University of Houston NextGen Sequencing Center (Texas, USA), which performed pre-library QC, Illumina 16s v4/v5 library preparation, Illumina MiSeq 600 cycle V.3 sequencing with pair-ended reads, and phylogenetic analysis. Sequences of the pooled 81 samples were demultiplexed to obtain sequence reads in each sample. All reads were trimmed to an average length of 275 base-pairs and then loaded onto the Silva 16S v32 Database for Operational Taxonomic Unit (OTU) clustering analysis. The filtering was set at an OTU clustering algorithm with 97% similarity percentage and the following cutoffs: Minimum Occurrences of 2, Chimera Crossover Cost of 3, Kmer Size of 6, Mismatch cost of 1, Minimum Score of 40, Gap cost of 4, and Minimum unaligned end mismatches of 5. The output data were further organized and displayed using the program R.
- Sediment heavy metal contaminants: Sediment samples were analyzed for the following heavy metals: cadmium chromium, copper, lead, nickel, and zinc. Sediment cores were dried at 95°C for at least 24 hours. All plant and shell material was removed from each sample, and then the sediment pulverized using a mortar and pestle. Sediments were digested following EPA Method 3050B to extract metals into an aqueous solution. Samples were then diluted to 50-ml with Optima Water and stored in the dark until analyzed by ICP. ICP analysis was conducted using a Perkin-Elmer Optima 7000 ICP AES.
- Sediment organic pesticide contaminants: 27 sediment samples (with at least 2 duplicates each) were processed via Solid Phase Extraction (following a

QuEChERS protocol), and the organic eluate of each was analyzed by GC-MS. The GC-MS results were compared specifically to the GC-MS trace for a low-level concentration mixture of 20 organochlorine pesticides (EPA mixture 8081), which had been completed as 200 ppm, 100 ppm, 100 ppm, 50 ppm, 10 ppm, 1 ppm and 0.1 ppm solutions in hexane. The GC-MS results were also compared to the standard Shimadzu Mass Spectral Library for similar structures.

Aerial Imagery

All of the sites were sampled during July 2021 (average water, peak floral coverage), and again in February 2022 (low water, floral senescence). The study site, reference site, and armored site for each location (e.g., Trinity Bay, Armand Bayou, and West Galveston Bay) were sampled concurrently, as follows:

- Ground control points: 25cm discs (frisbees) mounted on 1.3m x ¾ inch PVC poles were labeled A-F and were used as ground control points (GCPs) for the purpose of establishing visible, measurable, elevation-correctable reference points within each image. 1 GCP was placed at the origin and another at the terminus of the 10m transect. The remaining four GCPs were placed at locations around the 10m transect, in a non-linear formation to enhance accuracy for elevation measures. Each GCP pole was inserted into the sediment until the height of the GCP above the sediment line was 1m. The GPS location and elevation was recorded using a Trimble R1 GPS device from the top center point of each GCP.
- Drone imagery: A DJI Phantom 4 Pro drone equipped with a 1”, 20MP, CMOS true color camera was used to collect low altitude, high-resolution images at each site. A Parrot BEBOP-Pro Thermal drone equipped with FLIR One-Pro thermal imaging camera with detection limits between -20°C to 400C was used to take thermal images at each site. Flights were conducted by a licensed drone operator following Federal Aviation Administration (FAA) Part 107 requirements.
    - June 2021: Thermal images were captured over the site at altitudes of 30 and 60 meters to aid in determining saturation of each respective site. True color images were also captured as a practice run to work out methods of image correction using the GCPs.
    - February 2022: True color images were captured using a pre-programmed flight path for each site at an altitude of 148ft. The flight path for each site was laid out using DJI Pilot software to optimize image quality and overlap for integration within the GIS. Two fights were made at each site to provide duplication of imagery and selection of the best images when processing the images.
- GIS processing: The GPS points collected in the field were added to a blank project in ArcGIS Pro. Drone images of each site were selected based on clarity and coverage of the area and added to the ArcGIS Pro project with auto calculated statistics. Multiple photos were used for each site to ensure full

coverage of the area. The coordinate system for each image was set to NAD 1983 UTM Zone 15N. Each image was then scaled, oriented, and georeferenced with a 1st Order Polynomial (Affine) transformation.

After georeferencing, the multiple images per site were then combined into a single raster using the Create Mosaic Dataset tool. This tool was used to make a new raster dataset file, and then georeferenced drone images were added into the mosaic. To manage cell size on the high quality images, the Aggregate tool and the Mean aggregation technique was used, setting Cell Factor to 25 or less. Cell size was also altered in the Environments menu to 0.25m. Rasters were then reclassified into 5 classes using the Iso Cluster Unsupervised Classification tool and converted into polygons using the Raster to Polygon tool with the Simplify Polygon option on. Symbology of the polygon was assigned according to the previous raster grid code. Each polygon was then edited to trim away unnecessary areas as needed, such as open water beyond the shoreline.

NAIP images from 2016 and 2018 were added to the project for each site. The historical NAIP images were already georeferenced and consisted of one image per site per year, so the above methods were followed from the Aggregation step onwards to turn these images into polygons for analysis. Because each NAIP image covered a large area, an extent was set around each site to focus processing on only the relevant areas for the study. NAIP images were coarser quality than the current drone images. This resulted in more "blocky" appearing habitat polygons that are less precise than the 2021/2022 shapefiles but still represent basic habitat types well.

All polygon files were examined individually to assign habitat type values. Categories for habitat types included bare ground (BG), low marsh (LM), high marsh (HM), mowed lawn (LAWN), and human constructed structures like bulkheads, jetties, buildings, etc, (STRUCT). Area of BG, LM, and HM polygons were summed to determine net gain or loss per site per year. All calculations used the NAD 1983 UTM Zone 15N coordinate system to determine area.

Analysis of data

Data from all field and laboratory analyses were collected, and analysis of variance was used to determine significant differences ($p \leq 0.05$) between the study sites and the respective reference and armored sites. Plant community data was analyzed by comparing the transect average for stem density, percent coverage of each species, and *Spartina alterniflora* root and shoot biomass. A two-sample t-test was used to detect significant differences in means. Shannon-Wiener species diversity index values and importance values were tabulated for plant communities. For chlorophyll production, the average chlorophyll concentration of the three readings for each plant was used as one of five replicates for each plot on the transects. One way ANOVA testing along with Tukey's pairwise t-test was used to detect significant differences in chlorophyll concentrations among living shorelines and reference shorelines. One way ANOVA and Tukey's pairwise t-test was also used to detect significant differences amongst shoreline types for sediment heavy metal concentrations and nitrospirae percent composition. Shannon-Wiener species diversity index and Pielou's evenness index were calculated for benthic macroinvertebrate communites. All statistical analyses were conducted using Minitab® v20.4 (2021) software.

RESULTS

Plant community:

- Density: Stem densities at Armand Bayou LVS (75 stems per m^2), shown in Figure 2, were about half of the amount found at REF (138). Trinity Bay stem density was 65 and 45 stems per square meter, for LVS and REF, respectively. Spartina alterniflora was the only observed plant species for both shorelines at the West Galveston Bay site. LVS at West Galveston Bay had much higher stem density of nearly 300 stems per m^2, double the amount of the reference shoreline's 150 stems.

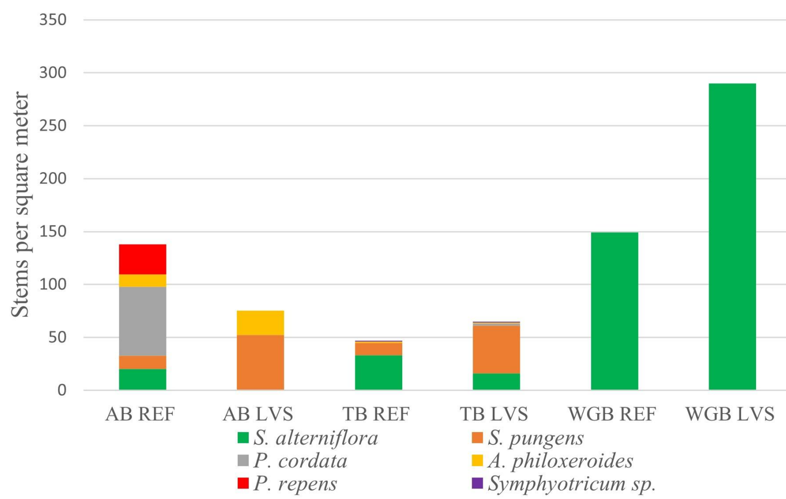


Figure 2: Stem densities of plant communities. Stem densities at AB LVS were approximately one half of that found at AB REF, perhaps alluding to the relative age differences between the two sites. WGB REF had approximately half of the stem density compared to WGB LVS, possibly reflecting negative impacts of ground subsidence at REF. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

- Coverage: Total coverage at Armand Bayou, shown in Figure 3, was higher at REF than LVS. REF plots were dominated by *Spartina alterniflora* (20% coverage), while none was found at LVS. *Schoenoplectus pungens* was the dominant plant species at LVS, while only a small amount was observed at REF. At Trinity Bay, LVS had greater total coverage (47.7%), than REF (15%). However, similar to Armand Bayou, LVS was dominated by *Schoenoplectus pungens*, and REF was dominated by *Spartina alterniflora*. At West Galveston Bay higher total coverage was observed at LVS (86.7%) than REF (31.3%). LVS in West Galveston Bay was the only shoreline to be observed with greater than 50% total coverage. Spartina alterniflora was the only observed plant species for both shorelines at the West Galveston Bay site.

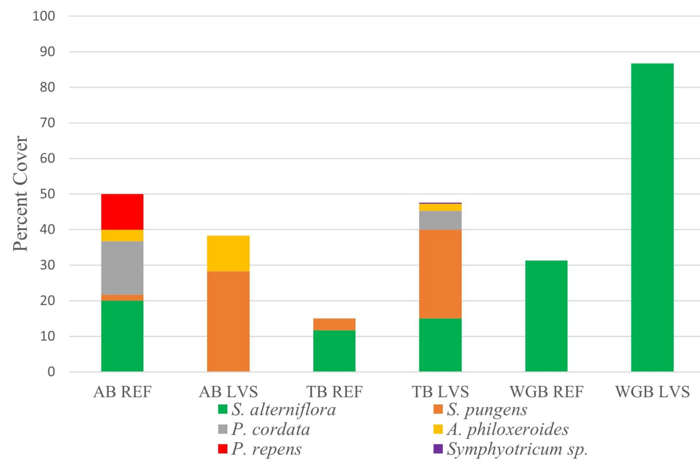


Figure 3: Plant coverage. Total coverage was different (p≤ 0.05) between REF and LVS at WGB, perhaps indicating negative impacts from ground subsidence at REF that are mitigated by the breakwater and shoreline topography at LVS. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

- <u>Shannon-Wiener species diversity index</u>: Species diversity values, shown in Figure 4, among Armand Bayou shorelines were vastly different with H' equaling 1.39 for REF, and 0.62 for LVS. For Trinity Bay, LVS had higher species diversity (0.78), than REF (0.57). Species diversity for West Galveston Bay was zero for both shorelines due to *Spartina alterniflora* being the only species present.

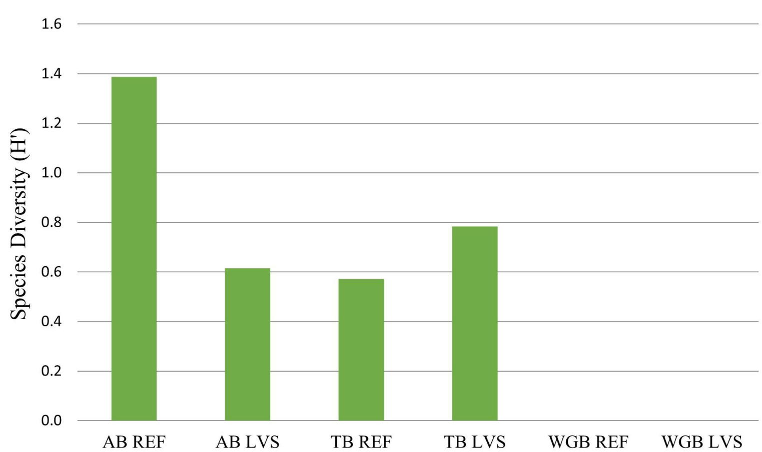


Figure 4: Plant species diversity index. Diversity was greatest at AB REF among all of the sites and locations. Diversity was greater at TB LVS than TB REF due to the presence of *P. cordata*, *A. philoxeroides*, and *Symphyotrichum sp*. that was absent at the adjacent REF site. AB

= Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

- <u>Plant importance values</u>: *Spartina alterniflora* was the most important plant represented at the reference site for each of the three locations (e.g., TB. AB, WGB), and at the study site at WGB, as shown in Figure 5. As noted previously, *S. alterniflora* was the only species present within the plots at both the reference and study sites at WGB. *S. pungens* was the most important plant at the study site for both AB, and TB.

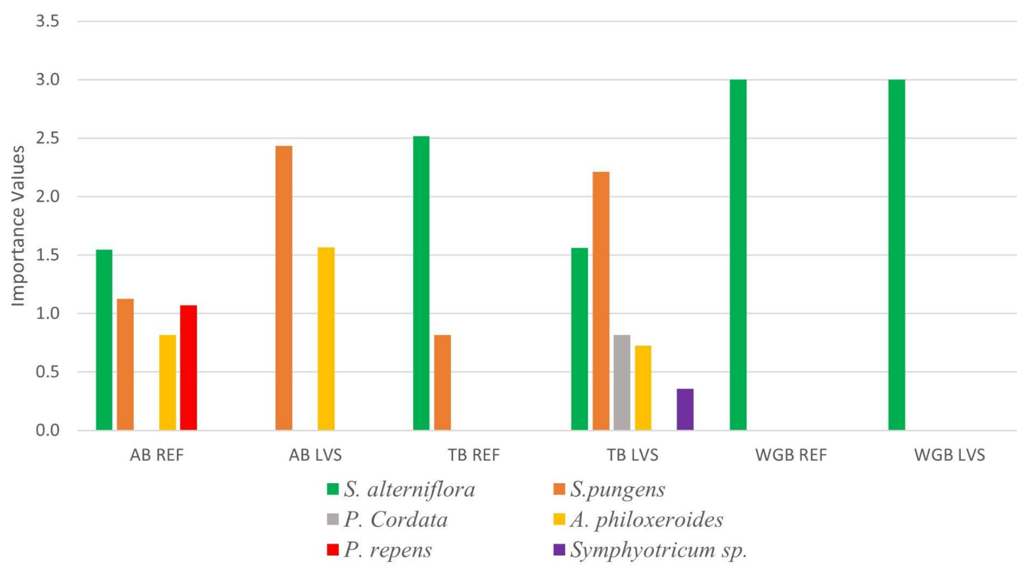


Figure 5: Plant species importance values. *S. alterniflora* was present at all sites across all locations, although sparse and not present within the plots on the transect at AB LVS. *S. pungens* was the most important plant at AB LVS and TB LVS. *S. alterniflora* was the only species measured at WGB REF and WGB LVS, where salinities are higher. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

- <u>Plant productivity</u>:
  - <u>*Spartina* biomass</u>: For Armand Bayou and Trinity Bay, biomass of both roots and shoots was similar between LVS and REF sites. For WGB, root biomass was significantly higher at LVS compared to REF. Biomass of roots exceeded shoots for

all shorelines except for Armand Bayou LVS, as shown in Figure 6.

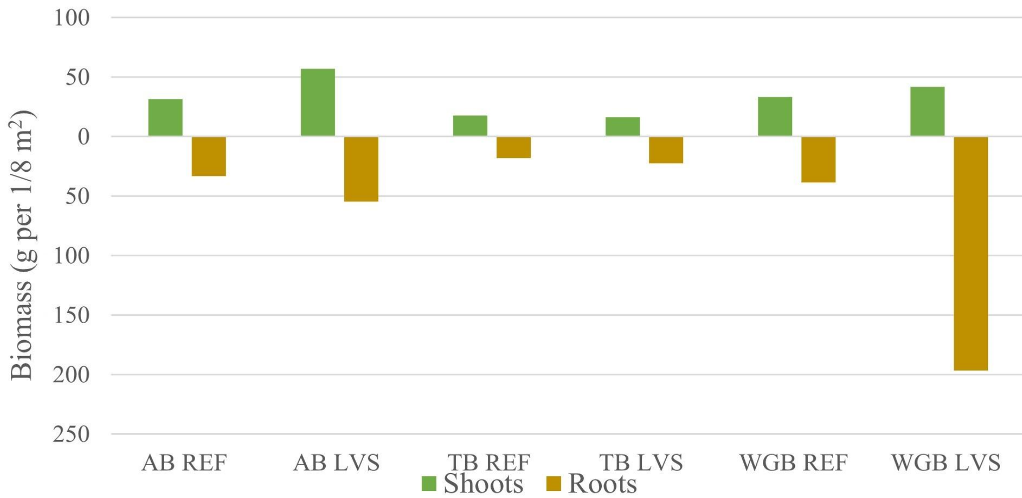


Figure 6: *Spartina* biomass for shoots and roots in grams dry mass per ⅛ m. *S. alterniflora* root biomass was higher at WGB LVS than WGB REF ($p \leq 0.05$). AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

- *Spartina* chlorophyll: *Spartina* chlorophyll concentrations, shown in Figure 7, were significantly higher at LVS than REF at both Trinity Bay and West Galveston Bay. Due to the absence of *Spartina alterniflora* in the square meter plots at Armand Bayou LVS, no chlorophyll values were measured at this site.

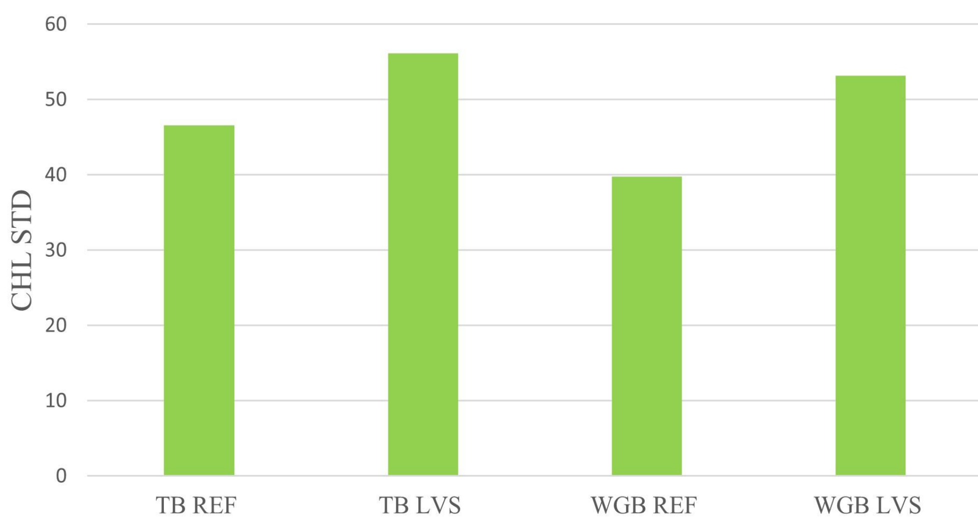


Figure 7: *Spartina* leaf chlorophyll in CHL Std. Chlorophyll concentrations were higher at the LVS sites at TB and WGB than either of the respective REF sites for those locations. No *S. alterniflora* was found within the plots along the transect at AB LVS, and was very sparsely located within the site as a whole ($p \leq 0.05$). TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

Nekton community: The total abundance of organisms varied by site and season (Figure 8). Among the Armand Bayou sites, total abundance was highest at the REF site, followed by the LVS site during October compared to June, driven primarily by White Shrimp (*Litopenaeus setiferus*). AB ARM abundance was very low during both sample periods. At Trinity Bay, total abundance was higher at all sites in June when compared to October, with the highest abundance at the LVS site, driven primarily by Grass Shrimp (*Palaemonetes vulgaris*). At WGB, total abundance followed opposite trends between the two sample periods. At the REF site, total abundance was nearly the same in June and October, driven by Grass Shrimp (*Palaemonetes vulgaris*) in both periods. Total abundance was higher at the LVS and ARM sites in June, driven by white and grass shrimp at LVS, and comb jelly (*Ctenophora*) at the ARM site.

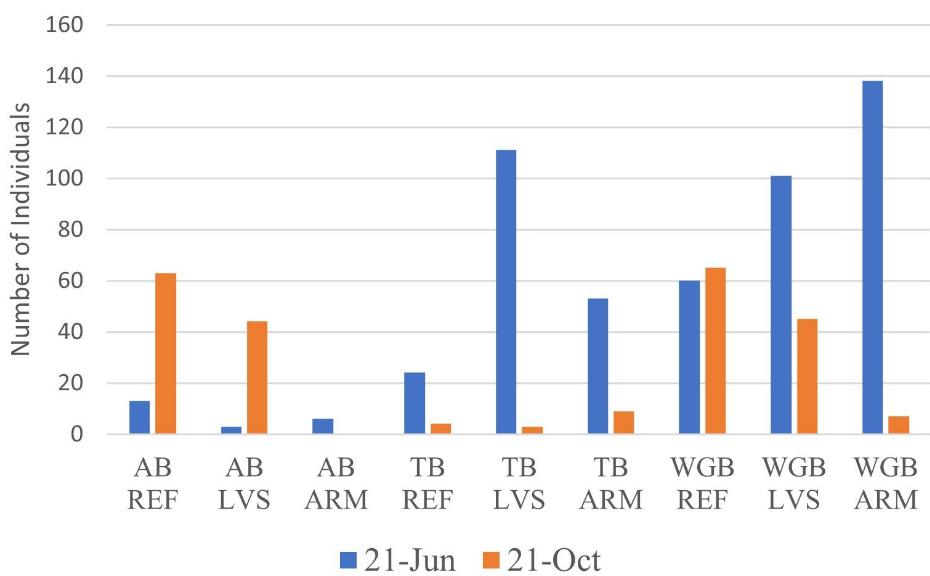


Figure 8. Total abundance of nekton species. The total abundance of organisms varied by site and season. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site, ARM = armored sites.

Total species richness across both periods at Armand Bayou was highest at REF (9 species) followed by LVS and then ARM (Figure 9). Trinity Bay exhibited the opposite, with the lowest species richness at the REF site (2 species) and the highest at the ARM site (6). WGB REF exhibited the highest species richness (8 species) while there was little difference between the LVS and ARM sites.

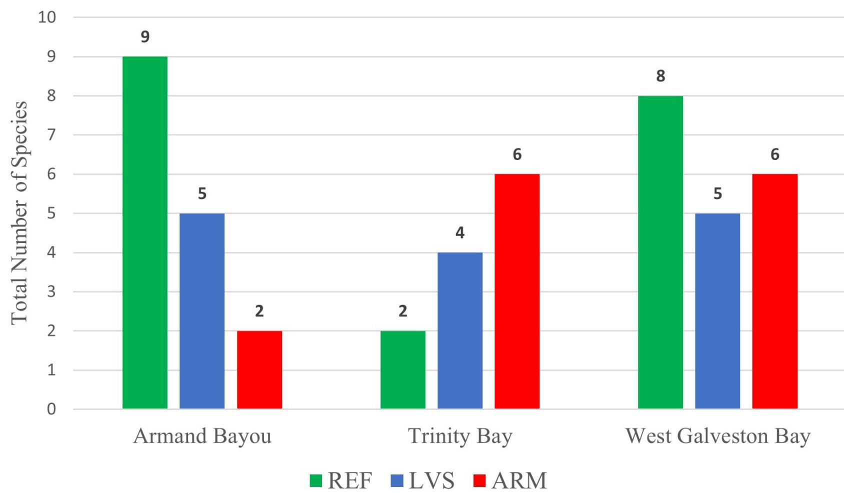


Figure 9. Total richness of nekton species. The total richness of organisms was highest at the REF for both AB and WGB, but was reversed at TB. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site, ARM = armored sites.

<u>Benthic macroinvertebrate community</u>: Total number of taxa identified ranged from a high of 22 (AB REF) to as few as 5 (TB ARM) (Figure 10). Total abundance of organisms ranged from 2077 (TB REF) to 51 (TB ARM) (Figure 11). In all cases total taxa richness and total abundance were highest at the REF site for each respective location (e.g., AB, TB, WGB). The lowest total taxa richness and total abundance was uniformly at the ARM site for each location.

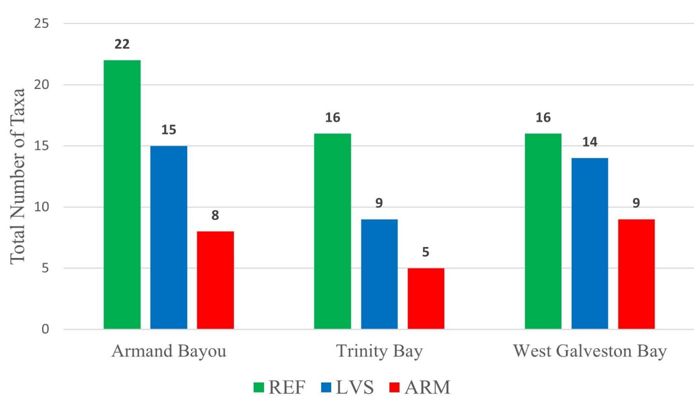


Figure 10: Total taxa richness of benthic macroinvertebrates. The number of taxa were highest at the REF site at each location ($p \leq 0.05$). AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site, ARM = armored sites.

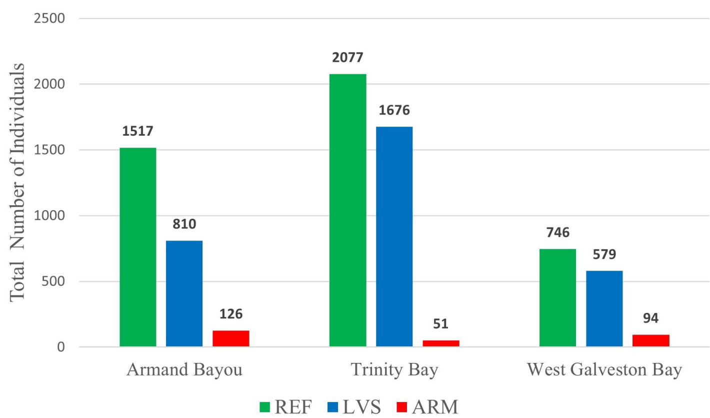


Figure 11: Total abundance of benthic macroinvertebrates. Total abundance of organisms was uniformly highest at the REF site and lowest for the ARM site at each location ($p \leq 0.05$). AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

Taxa diversity (Figure 12), was highest at the REF site and lowest at the ARM site for Armand Bayou. For Trinity Bay, REF had the highest taxa diversity while the LVS was the lowest. In West Galveston Bay, LVS had the highest taxa diversity and REF had the lowest. Benthic macroinvertebrate community evenness (Figure 13), was similar amongst shoreline types at Armand Bayou, with ARM community evenness slightly higher at 0.70 than REF and LVS, both of which were 0.60. For Trinity Bay, the LVS had a much lower evenness (0.22) than either REF (0.52) or ARM (0.60) . Community evenness at West Galveston Bay sites ranged from 0.43 at REF to 0.68 at LVS.

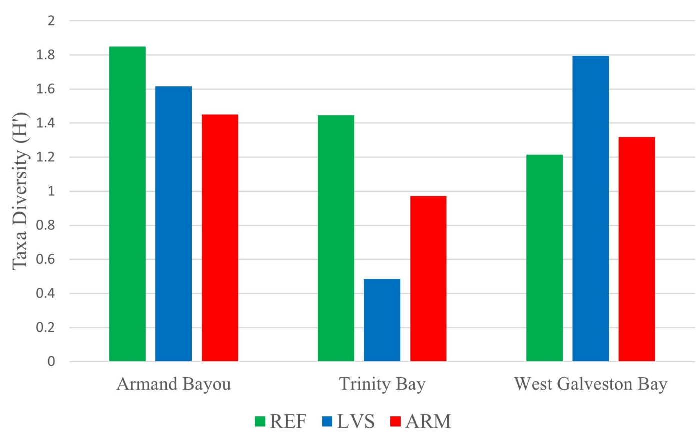


Figure 12: Taxa diversity of benthic macroinvertebrates. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site, ARM = armored site.

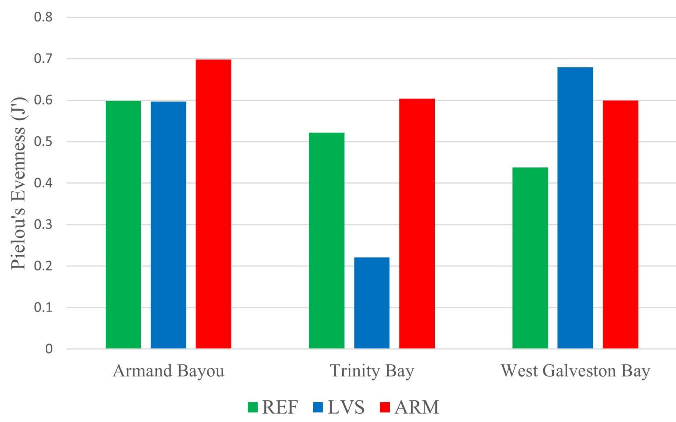
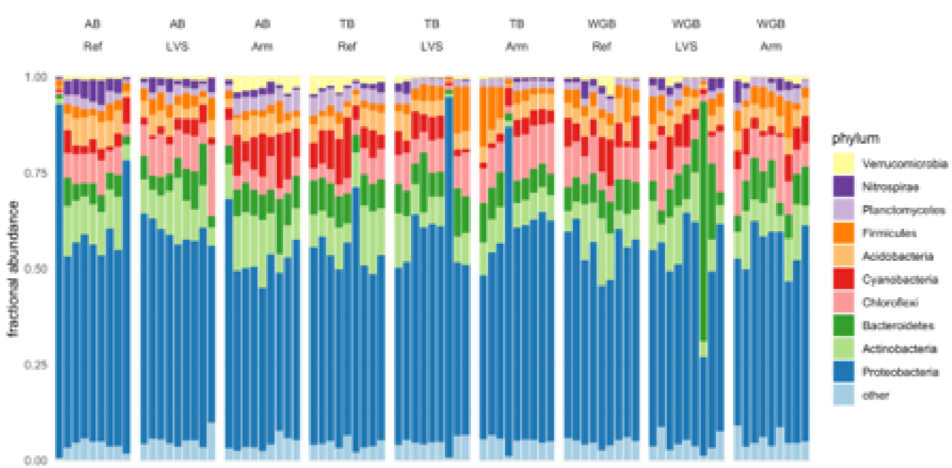


Figure 13: Evenness of benthic macroinvertebrate communities. For Trinity Bay, the LVS had a much lower evenness than either REF or ARM, possibly a factor of ongoing community successional processes. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site, ARM = armored site.

<u>Benthic microbiome</u>: Approximately half of a million reads were recovered from each sequenced sample, and about 60,000 of them matched with the database as bacterial OTUs. Analysis of the recovered OTUs indicates the following 10 bacterial phyla constituting the highest relative abundance in all collecting sites (Figure 14): Proteobacteria, Actinobacteria, Bacteroidetes, Chloroflexi, Cyanobacteria, Acidobacteria, Firmicutes, Planctomyces, Nitrospirae, and Verrucomicrobia.



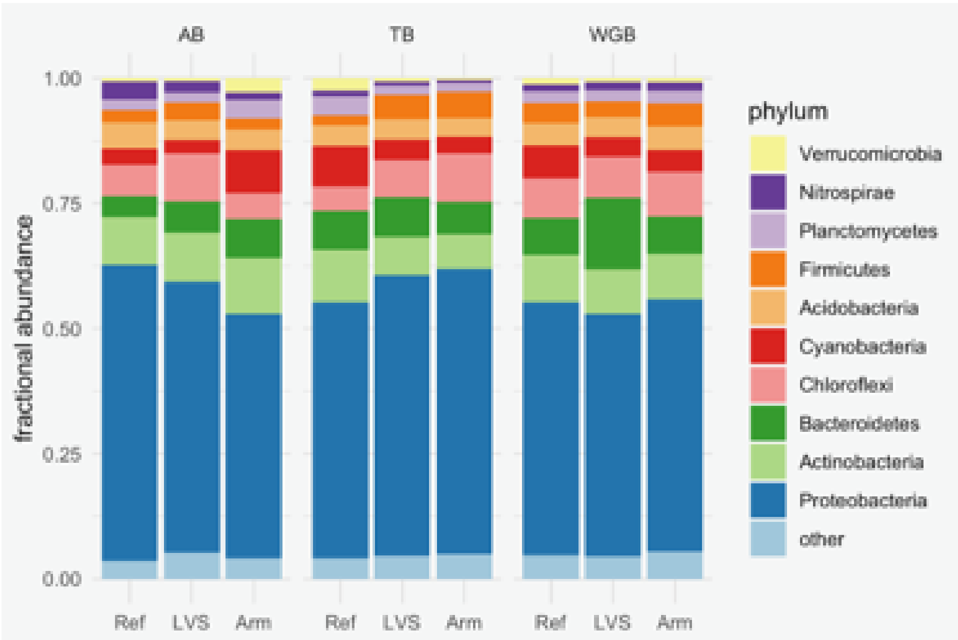


Figure 14: The 10 bacterial phyla exhibiting the highest levels of relative abundance in the three collecting locations are shown. (Upper graph) Fractional abundance of bacteria found in the nine sample replicates collected from each location. (Lower graph) Average fractional abundance of bacteria found in each site and location. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

When analyzing the percentage composition of these phyla, we found proteobacteria to be the most relatively abundant phylum in all collecting locations, comprising 48-59% of all bacteria (Figures 14 & 15). The remaining lower abundant phyla exhibit less consistent representation in each collecting site; however, the following trends are observed: at the Armand Bayou location, the bacterial abundance profile of the LVS restored site is more similar to that of the natural reference site than to the armored site. For instance, both the LVS and natural reference sites at Armand Bayou have Proteobacteria, Actinobacteria, and Chloroflexi as their top three relative abundant phyla, and Planctomyces and Gemmatimonadetes as the least abundant bacteria of the top 10 phyla (Figure 15). In contrast to these sites, the armored site of Armand Bayou contains Cyanobacteria rather than Chloroflexi as a top abundant phylum, and Firmicutes and Nitrospirae as its low abundant phyla (Figure 11). This trend is reversed at the Trinity Bay location, where the bacterial representations in the LVS restored site are more similar to those of the armed site than the natural reference site. For instance, phyla Actinobacteria, Cyanobacteria, and Plantomycetes all have lower percentages of composition in the LVS (7.6%, 4.2%, and 1.9%, respectively) and armored sites (6.8%, 3.3%, and 1.9%, respectively) than in the natural reference sites (10.5%, 8.1%, and 3.8%, respectively) (Figure 15). Furthermore, phyla Chloroflexi and Firmicutes both have higher levels of abundance in the LVS (7.5% and 4.9%, respectively) and armored sites (9.7% and 5.3%, respectively) than in the natural reference sites (4.9% and 2.0%, respectively) at the Trinity Bay location (Figure 15). As for the West Galveston Bay location, we could not discern notable differences in bacterial representations among the restored, natural, and bulkhead sites.

A.

AB-Ref	AB-LVS	AB-Arm
Proteobacteria (59.03%)	Proteobacteria (54.05%)	Proteobacteria (48.67%)
Actinobacteria (9.48%)	Actinobacteria (9.80%)	Actinobacteria (11.21%)
Chloroflexi (6.27%)	Chloroflexi (9.61%)	Cyanobacteria (8.54%)
Acidobacteria (5.22%)	Bacteroidetes (6.25%)	Bacteroidetes (7.77%)
Bacteroidetes (4.26%)	Acidobacteria (3.88%)	Chloroflexi (5.16%)
Nitrospirae (3.61%)	Firmicutes (3.53%)	Acidobacteria (4.18%)
Cyanobacteria (3.17%)	Cyanobacteria (2.72%)	Planctomycetes (3.73%)
Firmicutes (2.55%)	Nitrospirae (2.20%)	Verrucomicrobia (2.88%)
Planctomycetes (2.06%)	Planctomycetes (2.05%)	Firmicutes (2.18%)
Gemmatimonadetes (0.84%)	Gemmatimonadetes (0.86%)	Nitrospirae (1.48%)

B.

TB-Ref	TB-LVS	TB-Arm
Proteobacteria (51.12%)	Proteobacteria (56.14%)	Proteobacteria (57.05%)
Actinobacteria (10.48%)	Bacteroidetes (7.88%)	Chloroflexi (9.68%)
Cyanobacteria (8.08%)	Actinobacteria (7.60%)	Actinobacteria (6.84%)
Bacteroidetes (7.73%)	Chloroflexi (7.51%)	Bacteroidetes (6.50%)
Chloroflexi (4.85%)	Firmicutes (4.90%)	Firmicutes (5.27%)
Acidobacteria (4.19%)	Acidobacteria (4.06%)	Acidobacteria (3.75%)
Planctomycetes (3.82%)	Cyanobacteria (4.02%)	Cyanobacteria (3.27%)
Verrucomicrobia (2.55%)	Planctomycetes (1.87%)	Planctomycetes (1.86%)
Firmicutes (1.98%)	Gemmatimonadetes (0.96%)	Gemmatimonadetes (1.16%)
Nitrospirae (1.05%)	Nitrospirae (0.74%)	Calditrichaeota (0.78%)

C.

WGB-Ref	WGB-LVS	WGB-Arm
Proteobacteria (50.52%)	Proteobacteria (48.43%)	Proteobacteria (50.29%)
Actinobacteria (9.51%)	Bacteroidetes (14.13%)	Actinobacteria (9.23%)
Chloroflexi (8.00%)	Actinobacteria (8.99%)	Chloroflexi (9.03%)
Bacteroidetes (7.23%)	Chloroflexi (8.30%)	Bacteroidetes (7.31%)
Cyanobacteria (6.52%)	Acidobacteria (4.03%)	Acidobacteria (4.81%)
Acidobacteria (4.48%)	Cyanobacteria (3.85%)	Firmicutes (4.47%)
Firmicutes (3.99%)	Firmicutes (3.08%)	Cyanobacteria (4.29%)
Planctomycetes (2.39%)	Planctomycetes (2.24%)	Planctomycetes (2.51%)
Nitrospirae (1.33%)	Nitrospirae (1.68%)	Nitrospirae (1.70%)
Verrucomicrobia (1.30%)	Gemmatimonadetes (0.86%)	Gemmatimonadetes (1.01%)

Figure 15: Percentage composition of each of the top 10 bacterial phyla found in each of the three collecting locations. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

Sediment heavy metal contaminants: Sediment concentrations of total heavy metals (µg/g dry sediment), shown in Figure 16, were higher at Armand Bayou than either the Trinity Bay or West Galveston Bay sites. For Armand Bayou, REF and LVS were significantly higher in total metals than ARM. At Trinity Bay, LVS and ARM had significantly higher total heavy metal concentrations than REF. For West Galveston Bay, total heavy metals were significantly higher at REF compared to ARM, LVS was not significantly different from REF or ARM.

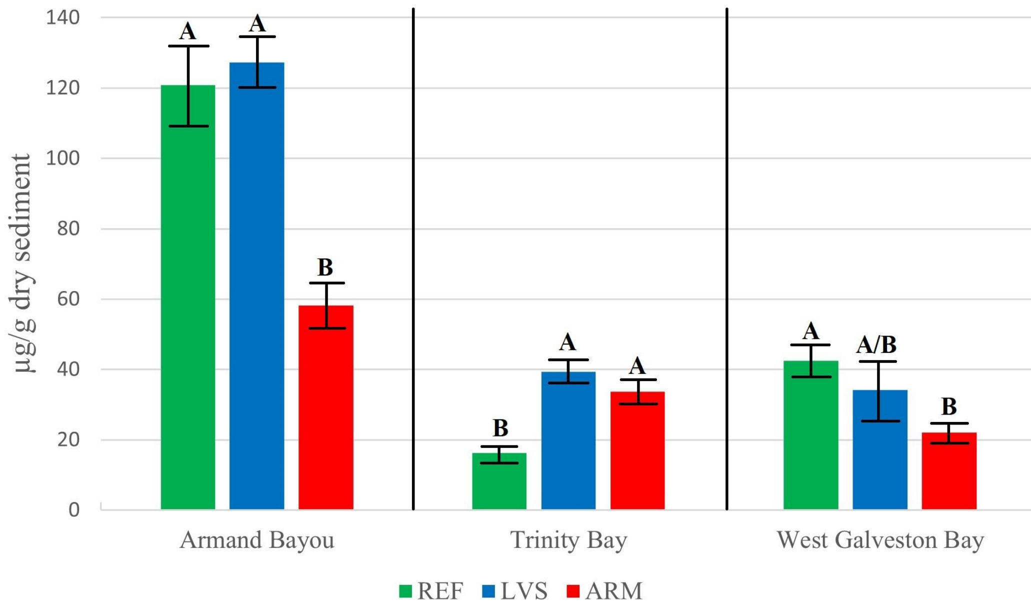


Figure 16: Sediment Concentrations of Total Heavy Metals. AB REF and AB LVS exhibited higher concentrations than at AB ARM ($P \leq 0.05$). TB REF exhibited lower concentrations than either TB LVS or ARM ($P \leq 0.05$). WGB RREF concentrations exceeded WGB ARM but not WGB LVS ($P \leq 0.05$). Error bars indicate significant differences detected for that site, shorelines with different letter groupings are significantly different. REF = natural reference site, LVS = living shorelines site, ARM = armored site.

Individual metals varied between sites and treatments. For example, Zinc concentrations at Armand Bayou, shown in Figure 17 were the highest among selected heavy metals across all shoreline types and were significantly higher at both LVS (48.7) and REF (54.0) than ARM (24.5). Nickel sediment concentrations were also significantly higher at both LVS (16.7), and REF (16.7) compared to ARM (6.6). Copper and Chromium concentrations were significantly higher at LVS than REF and ARM, and REF sites were significantly higher than ARM sites. Only copper was found at levels that exceed the Long and Morgan Effects Range Low (ERL) concentration for any contaminant studied. However, the detected level of 34.8 ppm for copper exceeds the ERL by only 0.8ppm (Long and MacDonald 19956; Long and Morgan 1990),

Heavy metal concentrations for Trinity Bay and West Galveston Bay varied within the sites, but at levels below the ERL for each contaminant measured.

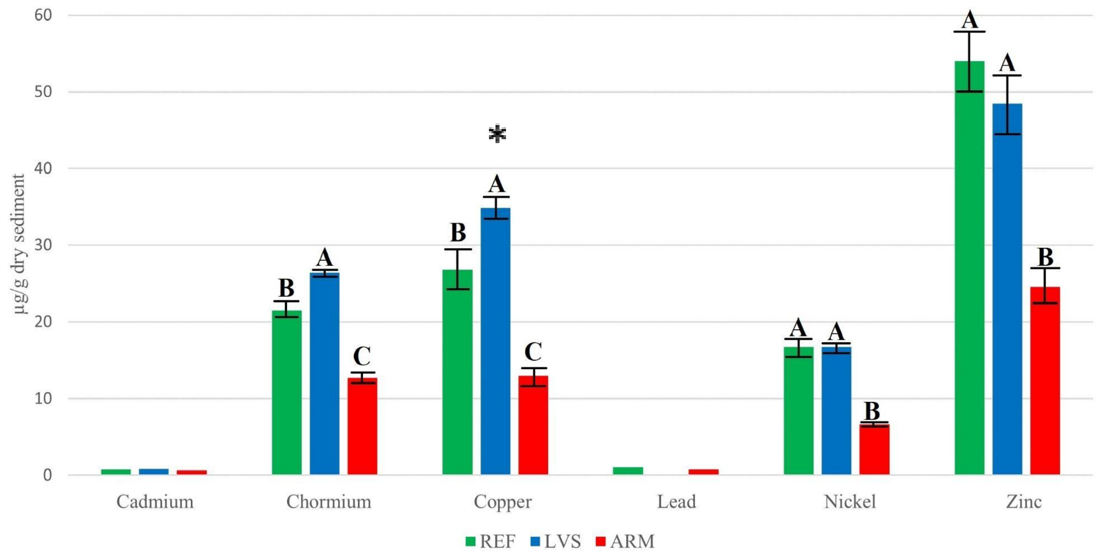


Figure 17: Armand Bayou (AB) sediment heavy metal concentrations. Differences were found in individual metals concentrations across AB REF, LVS, and ARM ($p \leq 0.05$). However, only the concentration for copper exceeded the Morgan and Long Effects Range Low (ERL) for any contaminant measured. Error bars indicate significant differences detected between site treatments for each metal, shorelines with different letter groupings are significantly different. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site. ARM = armored site.

<u>Sediment organic pesticide contaminants</u>: Organic contaminants were below detectable limits in all samples. Figure 18 shows an example chromatogram of the EPA reference sample alongside a chromatogram from AB LVS. From these results, we observe no distinction between shoreline types (i.e., REF, LVS, ARM)) at any of the locations.

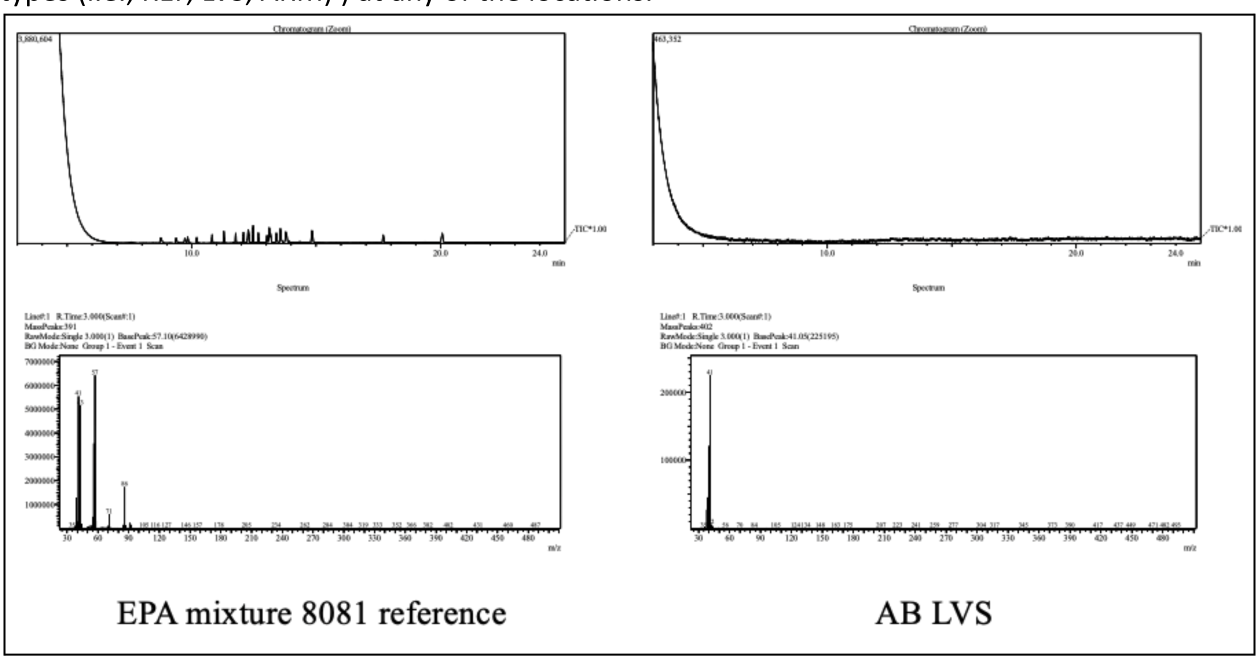


Figure 18. Sediment organic pesticide contaminant chromatography examples: EPA 8081 and AB LVS. All sites exhibited sediment organic contaminant levels at or below the detection limit, indicating that organic pesticide loads to the sediments are unlikely to pose stress to the biological community at these sites. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

Aerial imagery/shoreline stability: All three living shoreline sites (i.e., AB LVS, TB LVS, WGB LVS) exhibited shoreline stability throughout the period from 2016-2021 when examining the location of the shoreline (Figures 19 -21). This was also true at two of the reference sites (e.g., AB REF and WGB REF). However, the reference sites at TB REF exhibited approximately 10 meters of retreat toward the shoreline (Figure 22) . Regarding the ARM sites, all remained in place with no apparent degradation within the transect location. However, at TB ARM, a 20-meter section of the vinyl sheet pile bulkhead just outside of our study boundary collapsed in 2017 and then subsequently eroded approximately 5 meters shoreward by 2021. Additionally, the bulkhead continued to collapse outward toward each end for about 60 meters in that same timeframe, approaching the boundary of our study area (Figure 23).

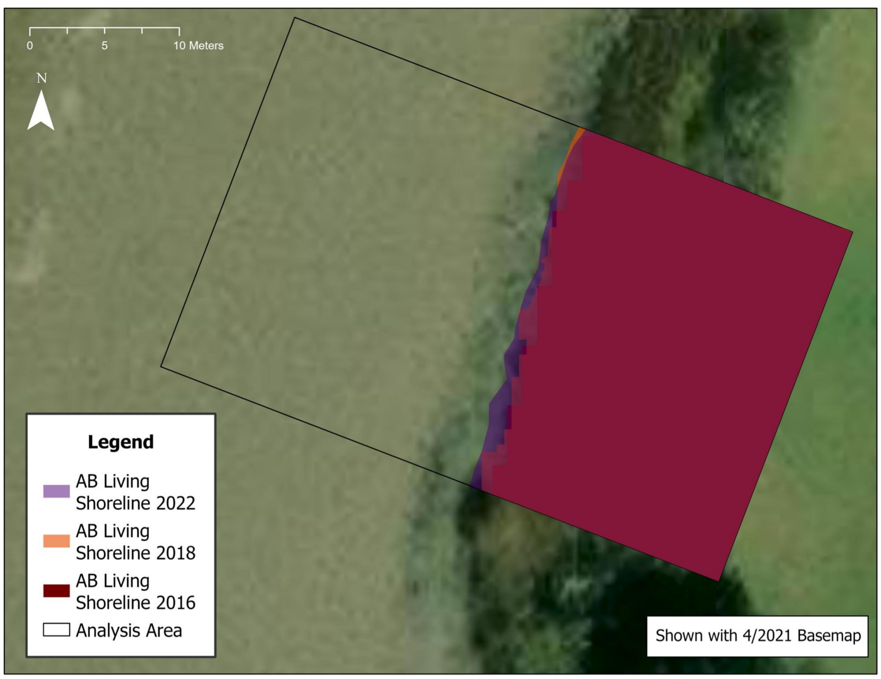


Figure 19. AB LVS Shoreline change from 2016 - 2022. Imagery indicates little change at the LVS over the timeframe analyzed. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

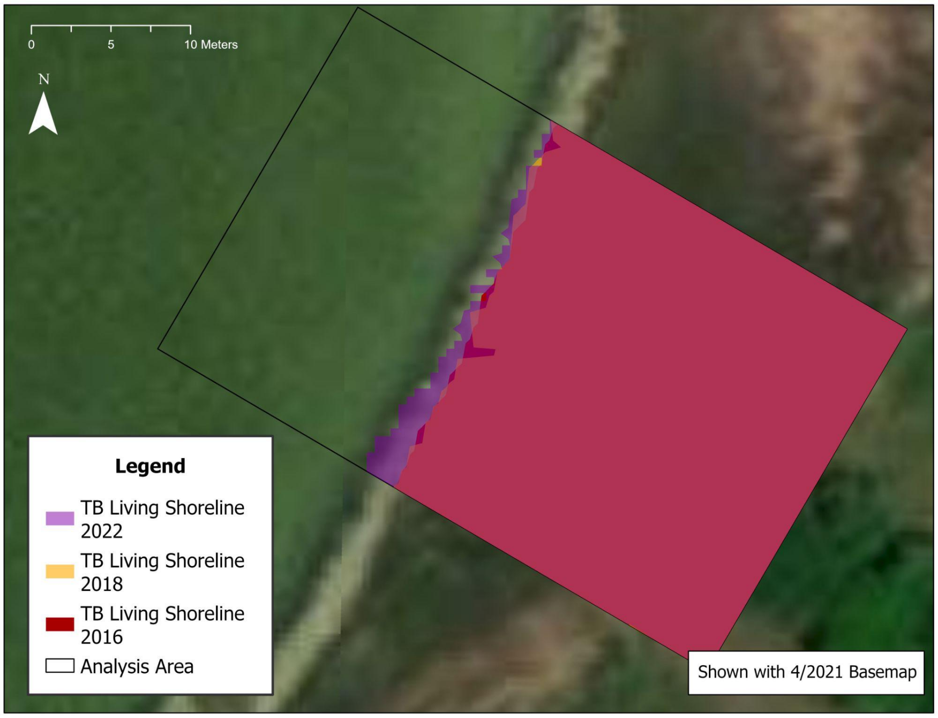


Figure 20. TB LVS Shoreline change from 2016 - 2022. Imagery indicates little change at the LVS over the timeframe analyzed. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

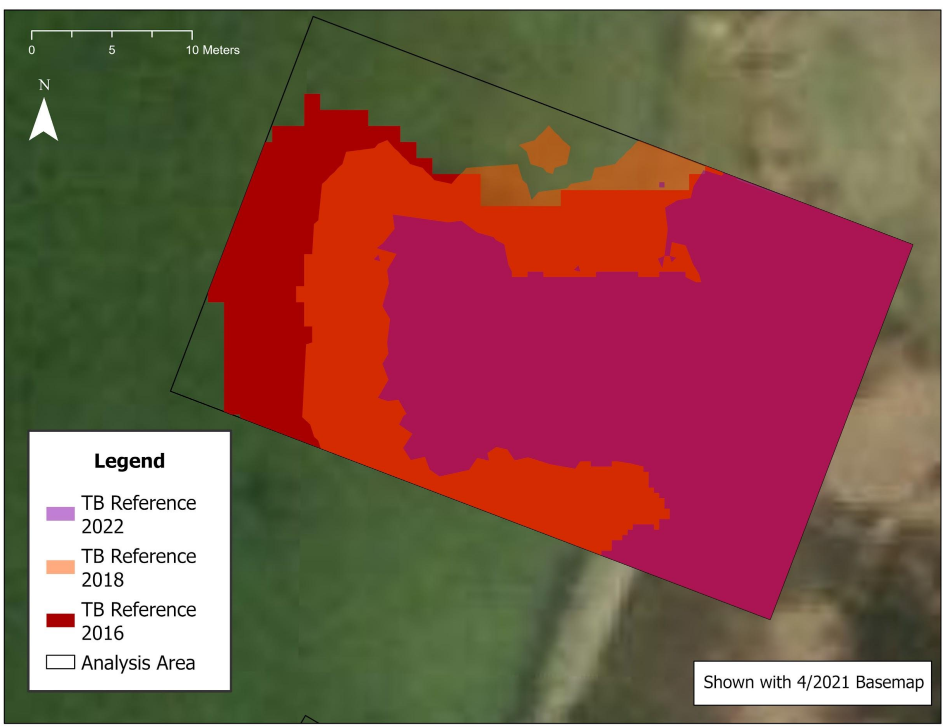


Figure 21. TB REF Shoreline change from 2016 - 2022. Imagery indicates 10 +/- meters of change at the REF shoreline over the timeframe analyzed. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

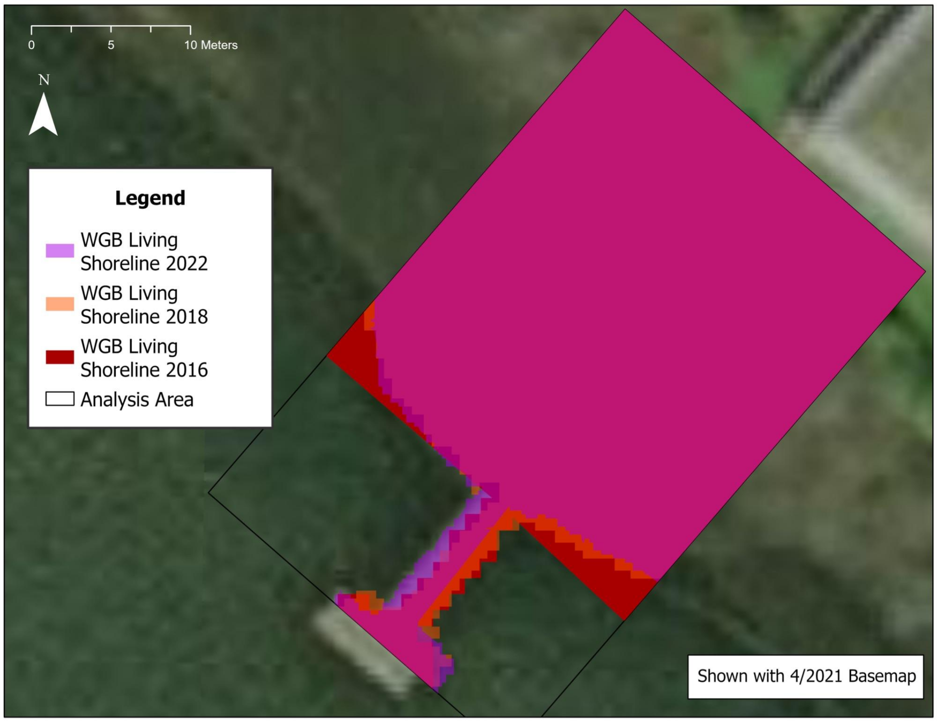


Figure 22. WGB REF Shoreline change from 2016 - 2022. Imagery indicates minimal change at the REF shoreline over the timeframe analyzed. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

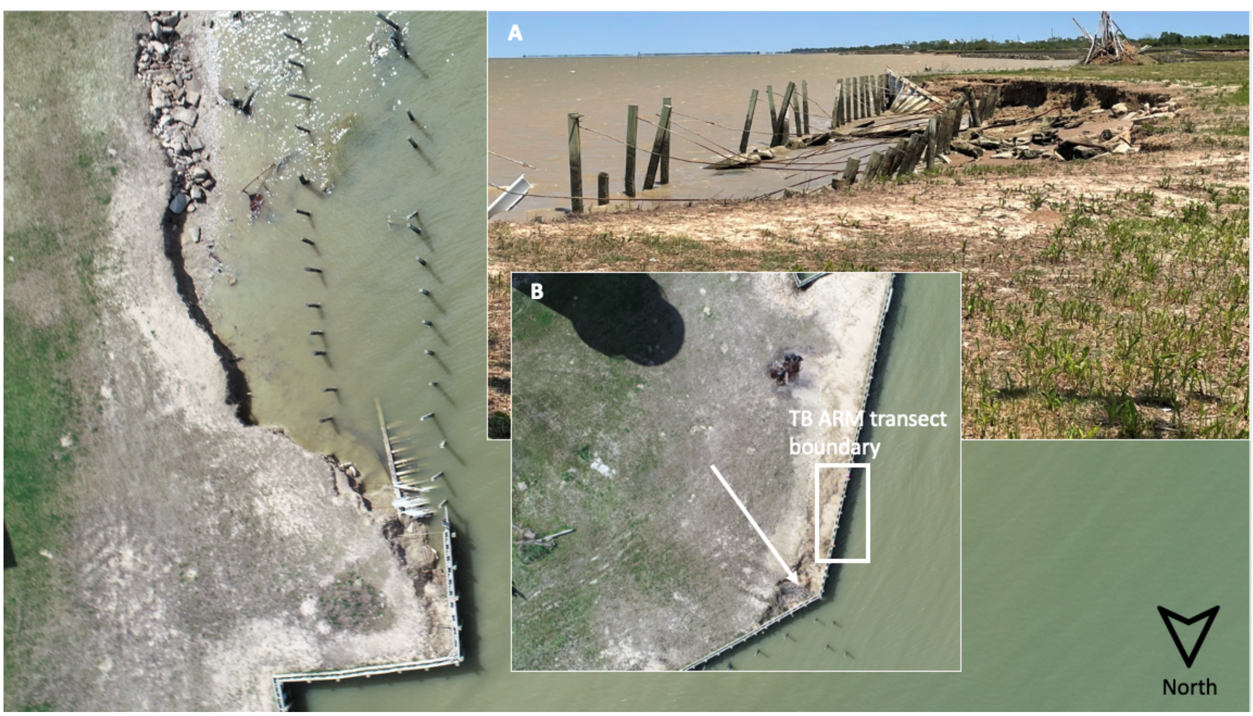


Figure 23. Failing sheet pile bulkhead at Trinity Bay. Inset A shows the damage from the ground, looking toward the south. The study boundary for TB ARM is immediately south of the damaged area (Inset B) , on the same bulkhead, where undermining is also becoming evident. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

As seen in Figures 24-26, ARM sites in all three bay systems saw minimal changes (6% or less loss). LVS sites were fairly stable with AB and TB gaining slightly and WGB having a slight (-3.9%) loss. These small percentages of change may be an artifact of poor resolution in historical photos used to generate 2016 and 2018 polygons and not necessarily indicate differences. AB LVS gained low marsh area from 2016-2022 (+142%), and had large gains in high marsh (+139%) and a complete loss of formerly bare ground habitat. This was due to vegetation expanding into bare habitat, providing more shoreline stabilization. TB LVS experienced a 31% increase in bare ground, a 426% increase in high marsh, and a 55% loss of low marsh. Loss of low marsh was possibly due to succession and establishment of high marsh vegetation. WGB LVS had bare ground decrease by -51%, indicating new vegetation growth. It also experienced a loss of low marsh (-40%) and a gain of high marsh vegetation (131%). Raster rendering was particularly coarse for the WGB LVS site due to a poor quality 2016 NAIP image and may overestimate bare ground habitat for that year.

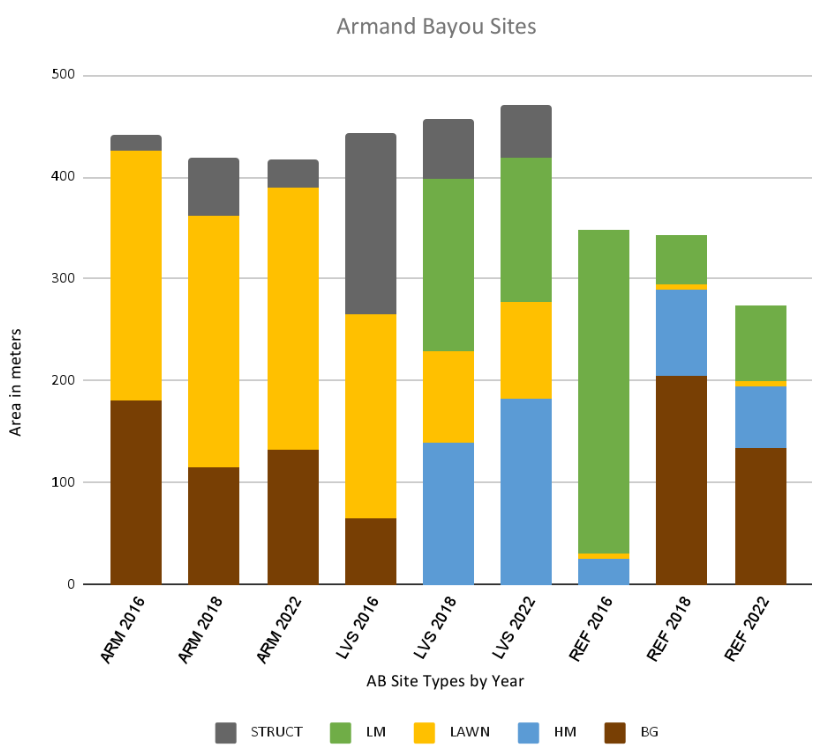


Figure 24. AB land cover change from 2016 - 2022. AB LVS gained low marsh area from 2016-2022 (+142%), and had large gains in high marsh (+139%) and a complete loss of formerly bare ground habitat. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

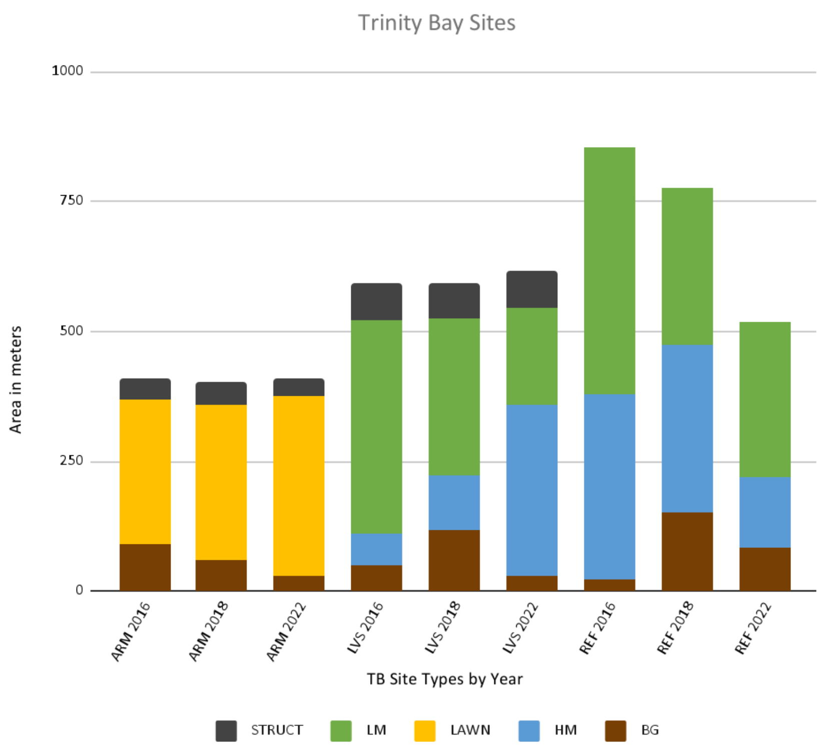


Figure 25. TB land cover change from 2016 - 2022. TB LVS experienced a 31% increase in bare ground, a 426% increase in high marsh, and a 55% loss of low marsh. Loss of low marsh was possibly due to succession and establishment of high marsh vegetation. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

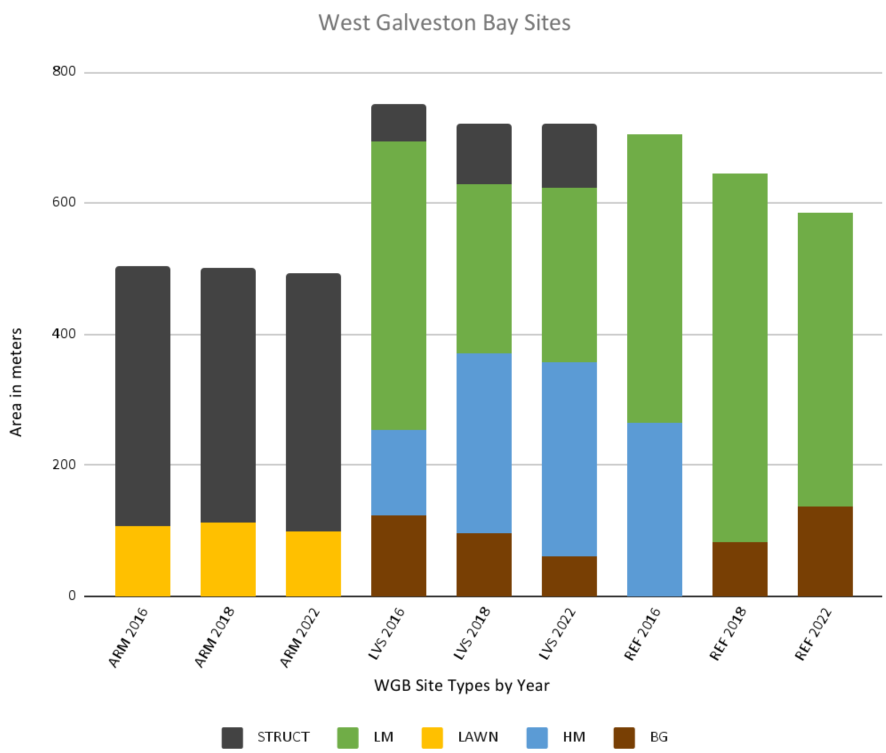


Figure 26. WGB land cover change from 2016 - 2022. WGB LVS had bare ground decrease by -51%, indicating new vegetation growth. It also experienced a loss of low marsh (-40%) and a gain of high marsh vegetation (131%). AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

REF sites across all three systems saw losses in total area (ARM REF -21.7%, TB REF -39.4%, WGB REF -16.8%). All REF sites gained bare ground habitat from 2016-2022, indicating loss of vegetation and associated increasing rates of erosion. AB REF increased high marsh slightly (gaining 33.8 m^2) but saw a loss of 77% of low marsh habitat. TB REF saw dramatic changes in habitat composition, losing 62% of high marsh and 37.2% of low marsh area. WGB REF appeared to have lost all 265.4 m^2 of its high marsh and saw a slight (+2.5%) increase in low marsh. Images from 2022 show the retreat of vegetation and widening of channels at this location as compared to 2016.

DISCUSSION

The most striking result of our study may reside in the remarkable differences we measured at the three different locations across the bay system. Of the three, the data from Armand Bayou (AB) most clearly support our hypothesis. Conversely, data from Trinity Bay (TB) and West Galveston Bay (WGB) indicate the presence of stressors and suggest comparisons may be challenged by aspects of ecological succession following planting at the Living Shorelines (LVS) sites. Also, intermediate disturbances at the LVS and the respective REF sites within each location may not be equivalent in magnitude or may be mitigated by the structural or topographic aspects of the LVS.

Armand Bayou (AB)

Biologically, the measures of the plant community composition indicate that the LVS site is undergoing successional processes, and has not yet achieved the same level of species richness or diversity as the corresponding natural reference site (AB REF). Further, while *S. alterniflora* scores the highest importance value of the three species found at REF, it was absent from the plots at the LVS site, although small stands were visible within the LVS off of the transect and were present in the 1/8 m^2 biomass plot. It is noteworthy that the site was sprigged with *S. alterniflora* in 2011. Further, *S. alterniflora* was measured at the site in a pilot for this study in 2018, at which time it was *S. alterniflora* was by far the most important plant in the AB LVS community as with REF (Torres et al. 2020). Importance values at both AB REF and AB LVS between 2018 and 2021 show a decline in *S. alterniflora* and increase in *S. pungens* during that time. Measures of salinity by the Texas Commission on Environmental Quality from 2018 to 2021 at the monitoring station upstream from the AB study sites were below 4 ppt, slightly lower than the 1998-2003 average of 6.7 ppt for Mark Kramer (formerly Mud) Lake where the LVS is located (TCEQ 2022, Masterson 2006)). Lower salinities may promote the presence of *S. pungens*, and may favor proliferation of the species in the LVS site relative to *S. alterniflora* under conditions where the marsh is still developing through the processes of ecological succession. Further, it is noteworthy that AB REF supported a greater diversity of plants, in addition to an overall higher relative coverage and stem density, likely indicative of the more established biological community at REF.

Of course, no plants were present at the armored site (AB ARM). This may be reflected in the nekton, benthic macroinvertebrate, and microfaunal communities. For example, benthic macroinvertebrate taxa abundance were lowest at the ARM site at each location (i.e., AB, TB, WGB), suggesting that conditions on the open bayou/bay bottom in front of the respective bulkhead is not conducive to favorable conditions for these organisms. Similarly, benthic taxa diversity and community evenness for LVS was also more similar to REF than ARM. This is important in that benthic macroinvertebrates play a crucial role in geochemical conditions at the water-sediment interface, promote decomposition and nutrient cycling, and form an important link in the food web (Brown et al. 2000, Lerberg et al. 2000). Larger animals, including many commercially important fish, often feed upon these benthic organisms (Lerberg et al., 2000; Little, 2000). Because of their sessile nature, benthic macroinvertebrate species are typically found only where the conditions are suitable for that species' survival (Klemm et al. 1990, Rakocinski et al. 1997). The presence or absence of particular species, and the community overall make-up, can be used as an indicator of habitat degradation (Brown et al. 2000, Carr et al. 2000, Engle et al. 1994; Holland et al. 1973, Klemm et al. 1990, Lerberg et al. 2000, Rakocinski et al. 1997).

As with all of the locations studied in this work, potential abiotic stressors in the form of sediment contaminants in Armand Bayou fell below the ERL for all contaminants measured, with the exception of copper at the AB LVS site, which exceeded the ERL by only 0.8ppm (Long and MacDonald 1995, Long and Morgan 1990). These concentrations do not appear to have affected the total abundance of benthic macroinvertebrates at the LVS site, which is much more similar to the REF site in this measure than it is to ARM. It is probable that the benthic macroinvertebrate community is developing in concurrence with the plant community. Previous studies have noted rapid increases in biomass of benthic macroinvertebrate communities at LVS sites as they develop, which is important for the LVS to achieve functional parity with natural reference marshes (Spieles and Mitch 2000, Davenport et al 2018).

Additionally, bacterial phyla across the three study sites at Armand Bayou (i.e., REF, LVS, ARM) exhibit compositional similarity across the phyla represented, dominated at each site by Proteobacteria. However, the bacterial abundance profile of LVS is more similar to that of REF than to ARM. For instance, both the LVS and natural reference sites at Armand Bayou have Proteobacteria, Actinobacteria, and Chloroflexi as their top three relative abundant phyla, and Planctomyces and Gemmatimonadetes as the least abundant bacteria of the top 10 phyla (Figure 15). In contrast to these sites, the armored site of Armand Bayou contains Cyanobacteria rather than Chloroflexi as a top abundant phylum, and Firmicutes and Nitrospirae as its low abundant phyla (Figure 15). Interestingly, Phylum Nitrospirae in the Armand Bayou location is found most abundant in the natural reference site, moderately abundant in the LVS restored site, and least abundant in the armored site (Figure 27). Because Nitrospirae play a role in nitrification, they are crucial for nitrogen cycling in the ecosystem. Their abundance in the collecting locations might reflect the capacity of these sediment microbes in supporting vegetation growth, which may in turn impact the abundance of other community populations across trophic levels. Indeed, the microbiome data collected at the Armand Bayou location are thus far in line with the diversity of plants found in the three different sites of this location.

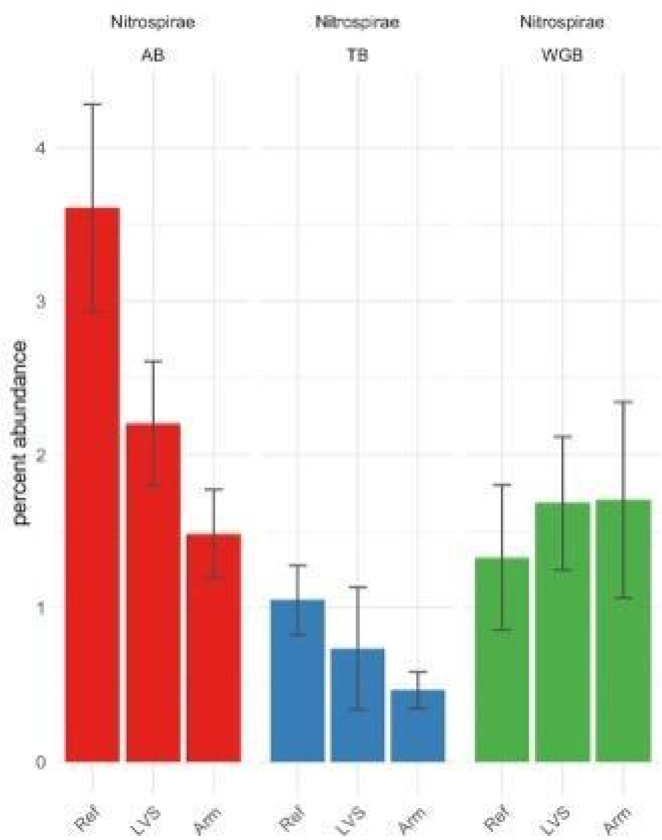


Figure 27. Relative abundance of Phylum Nitrospirae in all collected locations. Their abundance in the different collecting locations might reflect the capacity of these sediment microbes in supporting vegetation growth, which may in turn impact the abundance of other community populations across trophic levels. The data indicate a significant difference between AB REF both AB LVS and AB Arm, but not between AB LVS and AB Arm. TB REF was different than TB ARM, but neither was different from TB LVS. The data indicate no difference between the sites at WGB. ($p \leq 0.05$). AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site.

Nekton total abundance and species richness values are in line with plant, benthic macroinvertebrate, and microbial community metrics. However, we should note that site conditions make robust sampling for nekton challenging, due to the very shallow, varied bathymetry of the sites, the bottom of which can be exposed during low tides. Additionally, snags (e.g., dead trees, rocks, etc.) make secure placement of even small traps difficult in some locations. After numerous trial runs, it was decided to place traps over a 24-hour period immediately in front of the site(s) in an area that would remain submerged over the sample timeframe. This presents a compromise as in so doing, the traps would not be sampling nekton from within the marsh, but rather hopefully catching organisms associated with the marsh through ingress or egress, The most significant limitation is likely to be at the LVS sites, where traps had to be placed in front of the breakwater (WGB LVS) or in front of the riprap toe at the base of the marsh (AB LVS and TB LVS) (Smith et al 2021). N values were low across all site samples in June and October.

None-the-less, these combined data sets support our hypothesis for this site. The living shoreline restored site compares more similarly to the natural reference site across the parameters measured than to the armored site, although time may be required before community development achieves ecological parity with the natural marsh.

Trinity and West Galveston bays (TB and WGB)
Measures of the plant community at both sites indicate similar (WGB) or greater (TB) diversity and greater density (both) at the LVS site than their respective REF site. At WGB, *S. alterniflora* was the only species present in the plots at both the LVS and REF sites. However, the stem density, root biomass, and leaf chlorophyll were all higher at the LVS site than at the REF site for WGB. Similarly, at TB, root biomass and leaf chlorophyll values were greater at the LVS site than the REF site. The microbiome at the LVS sites did not exhibit strong similarities to their respective REF sites at either site. This may be explained by observed characteristics of disturbance evident at both locations (e.g., WGB, TB), although the nature and source of disturbance differs between the two locations.

Disturbances to marsh communities can be frequent along estuarine shorelines within the Galveston Bay system. These can include intensified wave fetch and erosional forces from winter cold fronts, flushing and scouring during high or extreme rainfall events, extreme temperatures (hot and cold) and salinity/dissolved oxygen fluctuations, deposition and redistribution of sediment from storms, and/or loss of elevation related to subsidence or relative sea level rise (HARC 2020).

Examining WGB, it appears differences we see in the plant community when examining stem density, percent cover, root biomass, and chlorophyll production may be attributed to disturbance. Field observations and collected data indicate conditions at the REF site have changed since 2018, where the measured plant community included *Salicornia virginica*, and *S. alterniflora* root biomass measured 88% greater than 2021 (Torres et al. 2020). Similar changes have been noted at the LVS site since 2018, including a reduction of root biomass (57%). Additionally, *Salicornia virginica* and *Batis maritima* were present in 2018, but not in 2021.

The WGB REF and LVS sites are located within an area of Galveston County that experienced between one to six feet of ground subsidence between 1906 and 2000, which has continued at a lesser rate along the coastal portion of the county through 2019 (HGSD 2019; Turco 2019). Changes in elevation could in turn affect the biologic community through changes to inundation related factors including salinity, dissolved oxygen, increased erosional forces, etc. (Osland et al. 2022, Fujiwara et al. 2019). However, the LVS site may be better positioned to adapt to this changing condition by virtue of the fact that it resides more immediately along a sloped shoreline, perhaps affording some opportunity for the marsh to migrate upslope for some period of time. Modeling suggests that spatial constraints may interfere with successional processes of coastal habitats (Feagin et al. 2005). This may also be true of marsh communities where slope or other space constraints may not allow for resilience to disturbance, such as erosion. None-the-less, this feature may benefit the WGB LVS site, at least in the short-term, relative to the REF site for this location. Additionally, benthic taxa diversity and community evenness for WGB was lowest at REF and highest at LVS, which further supports the inference that the LVS may be more stable and resilient to the disturbances observed at the REF site. Further, the LVS breakwater constructed in 2006 remains intact and likely will continue to provide some protection against wave fetch, allowing the plants to continue to modify and sequester sediments, further reducing erosion and slowing and perhaps maintaining elevation of the marsh over time (Coops et al. 1996, Feagin et al. 2009). It is also noteworthy that the breakwater also appears to be providing habitat for eastern oyster (*Crassostrea virginica*) on the structure and in the mud flat between the structure and the LVS marsh edge (Figure 28).

Figure 28. Eastern oysters growing on and adjacent to the breakwater the West Galveston Bay Living Shoreline site in 2021. Image: Haille Leija, GBF.

Disturbance is likely affecting the TB REF site as well. In this case, the site is located in an area that receives significant erosional forces from wave fetch and outflow from the Trinity River during rain events upstream, as evidenced by erosion rates approximating 2.5 meters per year since 2012, resulting in a cut shoreline littered with sizeable debris transported along the bay front from the river (Figure 29). The REF site has decreased in aerial coverage since 2011 retreating toward the adjacent shoreline, whereas the LVS site has remained stable during the same period. In this instance, the REF site is relatively exposed to the bay and its erosional forces, relative to the LVS site, which is better protected behind the riprap breakwater forming the front toe of the filled and planted “shelf”. Also, the REF marsh is entrapped between the bay and the cut shoreline behind it, and has little upslope area to which it could retreat. While this is also true for the LVS site, the breakwater feature may be providing conditions that are allowing the LVS to persist beyond what the REF marsh would be able to maintain. This is readily visible through infrared aerial imagery captured at the site on July 13, 2021 (Figures 30 and 31).

Figure 29. Looking northeast with TB REF in the background; cut shoreline visible behind the stand of marsh. Image: Cindy Howard, UHCL.

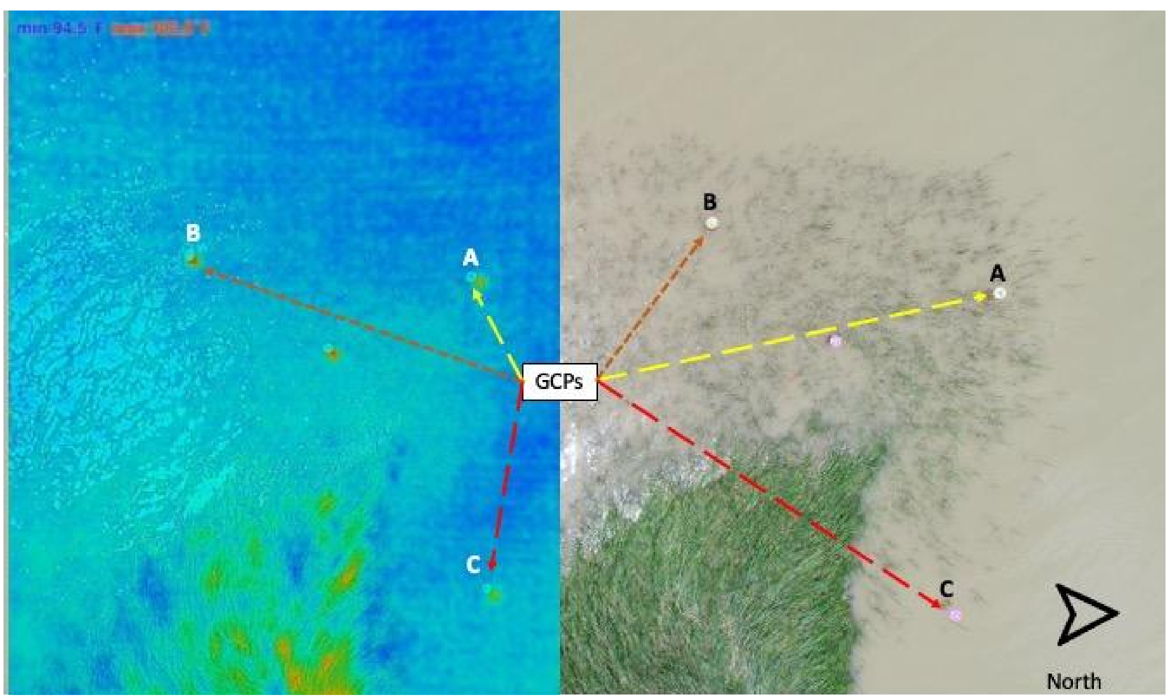


Figure 30. Low altitude drone image of the Trinity Bay reference site with georeferenced ground control points shown for orientation, July 13 2021. Left: FLIR infrared IR image where red depicts the highest emitted surface temperature (103.9°F) and blue depicts the coolest emitted surface temperature (94.5°F); warmer surfaces are higher in elevation and dryer, whereas cooler surfaces are wetter and lower elevation. Much of the reference marsh is inundated compared to the LVS site on the same date and time of day, potentially introducing stressors not present at the LVS.

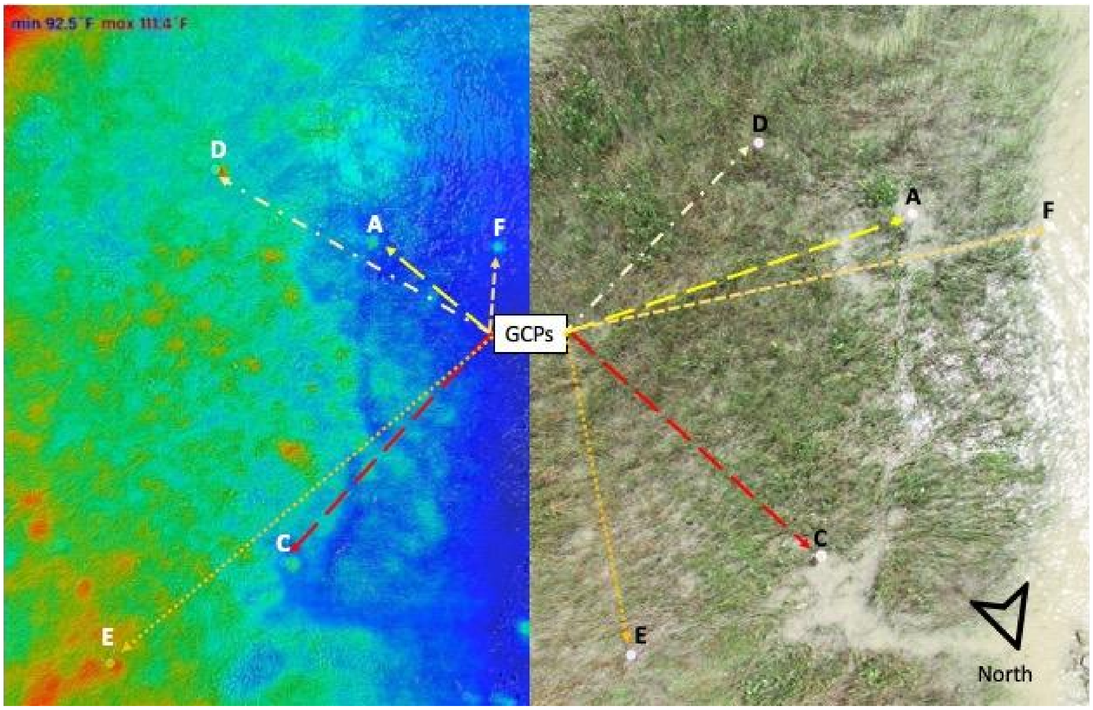


Figure 31. Low altitude drone image of the Trinity Bay Living Shoreline restoration site with georeferenced ground control points shown for orientation, July 13 2021. Left: FLIR infrared IR image where red depicts the highest emitted surface temperature (111.4°F) and blue depicts the coolest emitted surface temperature (92.5°F); warmer surfaces are higher in elevation and dryer, whereas cooler surfaces are wetter and lower elevation.

Here again, stress related to inundation, erosion, and related factors may explain differences in productivity measured between the TB REF and LVS sites, and may explain the relatively low abundance of Nitrospirae bacteria in the REF sediments (Figure 27). Conversely, the TB LVS plant community, similarly to the LVS at Armand Bayou, appears to be undergoing ecological succession based on changes in plant community composition since 2018, as the importance *S.pungens* and *S.alterniflora* increased, and the species diversity fell overall. For example, species such as *Eleocharis obtusa* present in 2018 were absent in 2021, most likely a result of competition and shading from taller species developing in the interim (Torres et al. 2020). Additionally, TB LVS benthic taxa diversity and community evenness is much lower than the REF despite abundance being similar, suggesting the benthic community of the LVS may still be developing and has not yet reached ecological parity with the REF site. As noted in similar studies of benthic communities of living shorelines, early establishment of benthic communities may be dominated by opportunistic taxa as a result of construction disturbance and development of community structure similar to natural shorelines may take time (Bilkovitch and Mitchell 2013).

Of note at all of the locations is that the reference sites exhibited the highest benthic macroinvertebrate species richness and total abundance. This would suggest that abiotic stressors affecting the plant community and microbiome have not yet impaired the benthic macroinvertebrate community at TB REF and WGB REF. These two sites also exhibited lower nekton species total abundance than their respective LVS sites.

While it is yet to be seen whether the LVS sites will eventually reach peak community functional status, the conditions appear favorable given the apparent stability of the site overall. This is reflected by the taxa and total abundance similarity between the REF and LVS sites, although the composition of the TB and WGB microbiome is less clear at this point. In every case, the LVS sites appear to be stable in terms of the physical shoreline, with little to no evidence of erosion or elevation change beyond that which would be expected for sediment as it undergoes settling and consolidation after placement (TB and WGB). The LVS sites appear to be more stable at TB and WGB that their respective REF sites, suggesting that application of Living Shorelines would be good candidates for restoration of degrading natural marsh, in addition to ecologic enhancements to private or public shorelines where erosion and land loss might otherwise be subject to armoring without aquatic habitat benefits. It is also noteworthy that the LVS site at Trinity Bay is exhibiting greater stability than its respective vinyl sheet pile armored site that was constructed in 2010, and collapsed across much of the front of the bulkhead in 2017, subsequently eroding behind the bulkhead significantly since that time.

SUMMARY

Our results suggest the efficacy of Living Shorelines applications can successfully promote the establishment of aquatic habitat replicative of natural habitat within the same system. In cases where more severe erosion of shoreline habitat is occurring, LS applications may also be a viable approach to both shoreline stabilization while also restoring habitat that is, or would be, lost because of erosion and/or relative sea level rise. In either case, the data show that habitat creation via this process may take some time to fully reach peak ecological function, as the processes of ecological succession unfold. This finding is consistent with studies on large-scale habitat restoration within the Galveston Bay system (Rozas et al. 2005, Rozas and Minello 2001, Minello and Webb 1997).

This data should be helpful to restoration managers and funding entities as considerations are made regarding shoreline restoration efforts. Consideration of successful outcomes for physical and biologic resiliency is important considering the implementation costs for construction. Further, demonstrating successful erosion management is often a key decision point when choosing among methods for shoreline stabilization, more so than ecological benefits that might be gained. Comprehensive data derived from aerial imagery, including habitat analyses, elevation and shoreline measurements, and multi-year comparison such as we have demonstrated in this study may be a cost-effective means of demonstrating effectiveness to potential landholders of effectiveness.

Unfortunately, in some instances due to site-specific issues such as strong erosional energy such as those at our Trinity Bay LVS site, the cost of implementing a living shoreline can exceed the cost of a traditional sheet pile bulkhead or similar. However, one of the benefits of soft shorelines over hardened shorelines is the lessened reverberation of wave energy and its erosional force to adjacent properties, benefitting the neighboring shoreline properties in addition to a site holder's shoreline employing such an approach. Under some circumstances, one landowner can be an advocate to neighboring property holders to make similar choices, such as what occurred in Eckert bayou adjacent to our WGB LVS site. Neighboring property owners choosing to also install living shorelines has ostensibly increased the overall beneficial impact of the approach in that area. Furthermore, there may be cost benefits gained from economies of scale where larger stretches of shoreline can be protected simultaneously using this approach.

Future studies are needed to expand these data over time, and at additional locations to develop better projections for ecological development across a variety of habitat types (e.g., estuarine, marine, and/or freshwater riparian). It would also be helpful to examine the long-term stability across different design approaches to living shoreline projects, which can range from simple planting to riprap reinforced planted marshes with breakwater features. These studies would be helpful in an effort to continue to improve applications of these stabilization methods across the broad range of conditions that can exist across coastal areas, and even across time at any given location. Also, studies of a wider range of abiotic factors that may impact site success, including emerging contaminants (e.g., microplastics), may shed further light on long term biologic adaptation of living shoreline restorations in urban or industrial areas. Finally, research investigating economic, social, and regulatory hurdles that might suppress application of living shorelines for shoreline stabilization and aquatic ecosystem enhancement may highlight opportunities to streamline processes or provide incentive for living shoreline development as the preferred alternative where options exist. Our results suggest that living shorelines can perform the function of shoreline stabilization as well or better than traditional methods, while providing secondary benefits to the local ecosystem and economy through enhanced aquatic habitat and the subsequent benefits provided to regional fisheries, water quality.

REFERENCES:

Bilkovic, D.M. and Mitchell, M.M. 2013. Ecological tradeoffs of stabilized salt marshes as a shoreline protection strategy: effects of artificial structures on macrobenthic assemblages. Ecological Engineering 61, 469-481.

Brown, SS, Gaston, GR, Rakocinski, CF, Heard, RW. 2000. Effects of sediment contaminants and environmental gradients on benthic macroinvertebrate community trophic structure in Gulf of Mexico estuaries. Estuaries. 23(3): 411-24.

Carr, RS, Montagna, PA, Biedenbach, JM, Kalke, R, Kennicutt, MC, Hooten, R, Cripe, G. 2000. Impact of storm-water outfalls on sediment quality in Corpus Christi Bay, Texas, USA. Environmental Toxicology and Chemistry. 19(3): 561-74.

Coops H, Geilen N, Verheij HJ, Boeters R, nan der Velde G. 1996. Interactions between waves, bank erosion and emergent vegetation: an experimental study in a wave tank. Aquatic Botany 53: 187-198.

Davenport, T.M., Seitz, R.D., Knick, K.E. and Jackson, N. 2018. Living shorelines support nearshore benthic communities in upper and lower Chesapeake Bay. Estuaries and Coasts 41(1), 197-206.

Dobberstine J, Smith P, Johnston S, Benson K. 2006. Successes and Challenges: An overview of community-based coastal marsh restoration in Galveston Bay. Presented at: the 3rd National Restore America's Estuaries National Conference; New Orleans, LA.

Dugan JE, Hubbard, DM, Rodil IF, Revell DL, and S Schoeter. 2008. Ecological effects of coastal armoring on sandy beaches. Vol. 29 (1). Pp 160-170.

Engle, VD, Summers, JK, Gaston, GR. 1994. A benthic index of environmental condition of Gulf of Mexico estuaries. Estuaries. 17(2): 372-84.

Environmental Protection Agency (EPA). 2013. Coastal Wetlands Initiative: Gulf of Mexico Review (EPA-843-R-10-005D). Office of Science and Technology Standards and Health Protection Division, Washington, DC. 278 pp.

Environmental Protection Agency (EPA). 2004. The Incidence and Severity of Sediment Contamination in Surface Waters of the United States, National Sediment Quality Survey: Second Edition (EPA-823-R-04-007). 174 pp plus appendices.

Feagin RA, Lozada-Bernarda SM, Ravens TM, Moller I, Yeager KM, Baird AH. 2009. Does vegetation prevent wave erosion of salt marsh edges? PNAS 106(25): 10109-10113.

Feagin, DA, Sherman, DJ, & Grant, WE. 2005. Coastal erosion, global sea-level rise, and the loss of sand dune plain habitats. Frontiers in Ecology and the Environment, 3(7), 359-364.

Fujiwara, M, Martinez-Andrade, F, Wells, RJD, Fisher, M, Pawluk, M, and M Livernois. 2019. Climate-related factors cause changes in the diversity of fish and invertebrates in subtropical coast of the Gulf of Mexico. Communications Biology. 2, 403.

Holland, JS, Maciolek, NJ, Oppenheimer, CH. 1973. Galveston Bay benthic community structure as an indicator of water quality." Contributions in Marine Science. 17: 169-188.

Gittman, RK, Fodrie, FJ, Popowich, AM, Keller, D., Bruno, JF, Currin, CA, Peterson, C. and MF Piehler. 2015. Engineering away our natural defenses: an analysis of shoreline hardening in the US. Frontiers in Ecology and the Environment 13(6), 301-307.

Gittman, RK, Peterson, CH, Currin, CA, Fodrie, FJ, Piehler, MF, & Bruno, JF 2016. Living shorelines can enhance the nursery role of threatened estuarine habitats. Ecological Applications, 26(1), 249-263.

Guillen, G., Whaylen, L., Broach, L., and S. Smith. 1999. Evaluation of Sediment Quality in the Vicinity of Permitted Discharges and Selected Land-use Practices: Galveston Bay. Unpublished TNRCC Report. 169 pp plus appendices.

HARC (Ed.) 2020. State of the Bay: A Characterization of the Galveston Bay Ecosystem (4th Edition). Texas Commission on Environmental Quality, Houston, Texas.

Klemm, DJ, Lewis, PA, Fulk, F, Lazorchak, JM. November, 1990. Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface waters. Washington D.C., USA: USEPA. USEPA/600/4-90/030.

Ko, J. 2007. The Economic Value of Ecosystem Services Provided by the Galveston Bay/Estuary System. Prepared for the Texas Commission on Environmental Quality Galveston Bay Estuary Program.

Lerberg, SB, Holland, AF, Sanger, DM. 2000. Responses of Tidal Creek Benthic macroinvertebrate Communities to the Effects of Watershed Development. Estuaries. 23(6): 838-853.

Lester J and L Gonzalez [a]. 2002. Chapter Four: The Human Role, Present. p. 39-54. In: The State of the Bay: A Characterization of the Galveston Bay Ecosystem. 2nd Edition. Austin (TX). The Texas Commission on Environmental Quality.

Lester J and L Gonzalez [b]. 2002. Chapter Five: Physical Form and Process. p. 67-94. In: The State of the Bay: A Characterization of the Galveston Bay Ecosystem. 2nd Edition. Austin (TX). The Texas Commission on Environmental Quality.

Little, C. 2000. The Biology of Soft Shores and Estuaries. New York (NY), USA: Oxford University Press 252 pages.

Long E. and L. Morgan. The Potential for Biological Effects of Sediment-sorbed Contaminants Tested in the National Status and Trends Program. 1990. NOAA Technical Memorandum NOS OMA 52. 235 pages.

Long E., Macdonald D., Smith S., and F. Calder. Incidence of Adverse Biological Effects within Ranges of Chemical Concentration in Marine and Estuarine Sediments. 1995. Environmental Management 19 (1): 81-97.

Masterson, C. (Ed). 2006. Armand Bayou Watershed Plan: Phase 1. A Report of the Coastal Coordination Council pursuant to National Oceanic and Atmospheric Administration Award No. NA170Z1140.

Minello TJ. 2004. A habitat-use model to determine essential fish habitat for juvenile brown shrimp (Farfantepenaeus aztecus) in Galveston Bay, Texas. Fishery Bulletin 102(2): 264-277.

Minello TJ., Webb J. 1997. Use of natural and created Spartina alterniflora salt marshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. Marine Ecology Progress Series 151: 165-179

Montagna, P. 2020. “Section 8.0: Salinity”. Freshwater Inflow: database and website developed by Texas A&M University - Corpus Christi. https://www.freshwaterinflow.org/about/ Accessed September 2022.

Mukaimi, M., Dellapenna, T., and J. Williams. 2018. Enhanced land subsidence in Galveston Bay, Texas: Interaction between sediment accumulation rates and relative sea level rise. Estuarine, Coastal and Shelf Science 207:183-193.

Osland, M., Hughes, A., Armitage, A., Scyphers, S., Cebrian, J., Swinea, S., Shepard, C., Allen, M., Feher, L,. Nelson, J., O’Brien, C., Sanspree, C., Smee, D., Snyder, C., Stetter, A., Stevens, P., Swanson, K., Williams, L., Brush, J., and R. Bardou. (2022). The impacts of mangrove range expansion on wetland ecosystem services in the southeastern United States: Current understanding, knowledge gaps, and emerging research needs. Global Change Biology. 28. 10.1111/gcb.16111.

Rakocinski, CF, Brown, SS, Gaston, GR, Heard, RW, Walker, WW, Summers, JK. 1997. Benthic macroinvertebrate responses to natural and contaminant-related gradients in northern Gulf of Mexico estuaries.” Ecological Applications. 7(4): 1278-98.

Rella, Andrew J., and Jon K. Miller. Engineered Approaches for Limiting Erosion along Sheltered Shorelines: A Review of Existing Methods. Sept. 2012. Prepared for: The Hudson River Sustainable Shorelines Project NYSDEC Hudson River National Estuarine Research Reserve.

Ropicki, R., Hanselka, D., Dudensing, R., Massey, J., O’Connel, J., and T. Looney. 2016. The Economic Impacts of Recreational Fishing in the Galveston Bay System. Texas A&M Agrilife Extension report TAMU-SG-16-517. 12 pages.

Rozas LP, Caldwell P, Minello TJ. 2005. The Fishery Value of Salt Marsh Restoration

Projects. J. Coastal Research Special Issue 40: 37-50.

Rozas LP, Minello TJ. 2001. Marsh terracing as a wetland restoration tool for creating fishery habitat. Wetlands 21(3): 327-341.

Seitz, R.D., Lipcius, R.N., Olmstead, N., Seebo, M.S. and Lambert, D.M. 2006. Influence of shallow-water habitats and shoreline development on abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. Marine Ecology Progress Series 326, 11-27.

Smith, C.S., Paxton, A.B., Donaher, S.E., Kochan, D.P., Neylan, I.P., Pfeifer, T., Van Hoeck, R.V. and Taylor, J.C. 2021. Acoustic camera and net surveys reveal that nursery enhancement at living shorelines may be restricted to the marsh platform. Ecological Engineering 166, 106232.

Spieles, D.J. and Mitsch, W.J. 2000. Macroinvertebrate community structure in high-and low-nutrient constructed wetlands. Wetlands 20(4), 716-729.

Texas General Land Office (TGLO). 2020. A Guide to Living Shorelines in Texas. Austin, Texas. https://cleancoast.texas.gov/documents/guide-to-living-shorelines-texas.pdf

Thayer, G.W., McTigue, T.A., Belmer, R.J., Burrows, F.M., Merkey, D.H., Nickens, A.D., Lozano, S.J., Gayaldo, P.F., Polmateer, P.J., and P.T. Pinit. 2003. Science-based Restoration Monitoring of Coastal Habitats, Volume One: A framework for Monitoring Plans under the Estuaries and Clean Water Act of 200 (Public Law 160-457). NOAA Coastal Ocean Program Descision Analysis Series No. 23. NOAA National Centers for Coastal Ocean Science, Silver Spring, MD. 35 pp. plus appendices.

Thayer, G.W., McTigue, T.A., Salz, R.J., Merkey, D.H., Burrows, F.M., S.J., and P.F. Gayaldo. 2005 (eds.). Science-based Restoration Monitoring of Coastal Habitats, Volume One: Tools for Monitoring Coastal Habitats. NOAA Coastal Ocean Program Decision Analysis Series No. 23, NOAA National Centers for Coastal Ocean Science, Silver Spring, MD. 628 pp. plus appendices.

Titus JG. 2000. Does the U.S. Government Realize that the Sea is Rising? How to Restructure Federal Programs so that Wetlands and Beaches Survive. Vol 30 Golden Gate U. L. Rev.

Torres, E., Blumentritt, E., Joy, R., and J. Dobberstine. 2020. Evidence for Small-Scale Living Shoreline Viability in the Galveston Bay Estuary. Research Poster: 11th State of the Bay Symposium. Galveston Bay Estuary Program (TCEQ), Galveston, TX January 222-23, 2020.

USEPA. December 2004. National Coastal Condition Report II. Office of Research and Development/Office of Water. Washington D.C. 311 pages.

Van Diggelen, A. and P. Montagna. 2016. Is Salinity Variability a Benthic Disturbance in Estuaries? Estuaries and Coasts 39: 967-980.

Ward GH, Armstrong NE. 1992. Ambient Water and Sediment Quality of Galveston Bay: Present Status and Historical Trends. Houston, TX, USA: Galveston Bay National Estuary Program. Report No. GBNEP- 22

Whaley SD, Minello TJ. 2002. The distribution of benthic infauna of a salt marsh in relation to marsh edge. Wetlands 22(4): 753-766.

White WA, Tremblay TA, Wermund Jr. EG, Handley LR. 1993. Trends and Status of Wetland and Aquatic Habitats in the Galveston Bay System, Texas. Houston, TX, USA: Galveston Bay National Estuary Program. Report No. GBEP-31.