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EXAMINING THE ECOLOGICAL FUNCTION OF SMALL-SCALE LIVING
SHORELINES IN GALVESTON BAY

by

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EXAMINING THE ECOLOGICAL FUNCTION OF SMALL-SCALE LIVING
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Dedication

This work is dedicated to my family. To my mother, who has always supported me and provided me with everything I need to be my best self, I am so grateful for all that you have given me. I know you are proud of me. To my sister Chelsea, thank you for always keeping me motivated and being a source of inspiration when I need to be reminded of what I am capable of accomplishing. To my partner, Natalie, thank you for being my rock throughout this, I could not have done this without you. Your constant love and support get me through my toughest moments. Lastly, to my late grandmother, thank you for instilling in me a love for education and learning. You taught me that few things in life can't be taken from you and that an education is one of them, no one will ever take this from me.

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ABSTRACT

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Living shorelines are an increasingly popular shoreline protection strategy. In contrast to traditional armoring techniques such as concrete bulkheads, living shorelines are designed to provide the many ecological functions and benefits of natural coastal wetlands. Despite a wealth of knowledge on coastal wetland restoration, studies verifying ecological function in living shorelines are limited. The objective of this study was to provide a comprehensive ecological assessment of three living shoreline projects in the Galveston Bay system. This study collected data on stem density, percent cover, and root-biomass to characterize plant communities. Data was also collected on the abundance and community structure of benthic and nekton organisms. Additionally, sediment heavy metal concentrations were examined. Living shoreline data was compared to both natural and armored shorelines ultimately confirming the suspected hypothesis that living shoreline sites function similarly to natural shorelines and improved over armored shorelines.

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CHAPTER I:
INTRODUCTION

Coastal Wetlands

It is established that coastal wetlands provide a suite of ecological services that translate to direct benefits for more than the third of the world's population living in coastal areas (Barbier 2019, Barbier et al. 2011, Van Coppenolle and Temmerman 2020). These benefits include sediment stabilization, valuable aquatic habitat, and improved water quality.

Sediment stabilization is provided primarily by plant communities attenuating wave energy and holding together sediments through root formations. A 2015 study concluded that increased plant stem density resulted in a 35% reduction in wave energy (Manis et al. 2015). Similarly, below-ground biomass was determined to be a significant predictor of soil erosion rates, and increased root biomass reduced lateral erosion (Lo et al. 2017). Further, plant species diversity is positively associated with soil stability in salt marshes (Ford et al. 2016). Together these characteristics of coastal wetlands provide defense against erosion, rising sea levels, and the more frequent threat of storm surges. An in-depth analysis post-Hurricane Sandy revealed that over 600 million dollars were saved in direct flood damages due to coastal wetlands (Narayan et al. 2017).

Coastal wetlands also have considerable recreational and commercial habitat value. Many species utilize coastal wetlands as a habitat intended to protect and nourish their offspring. Plant communities of coastal wetlands provide nutrients and shelter for juvenile shrimp, crabs, and fish; many of which are economically important. NOAA estimates that 68% of all commercial landings are species dependent on coastal wetlands and estuarine systems (Lellis-Dibble et al. 2008). Many bird species also seek out the

protection of coastal wetlands to feed and nest during migration (Withers 2002, Pickens and King 2014).

Along with the benefit of stabilizing shorelines and providing valuable habitat, coastal wetlands can also improve water quality by serving as sinks for various molecules. Coastal wetlands are able to mitigate some of the impacts of increased nitrogen loads from agricultural and residential developments during large runoff events (Adame et al. 2019). Contaminants, such as heavy metals and organics, are known to sequester in sediments of industrialized estuarine systems through a variety of mechanisms, thus reducing downstream impacts on both ecosystems and humans (Matagi et al. 1998, Sinicrope et al. 1992, Zhang et al. 2010). Of considerable note is the capacity of coastal wetlands to store carbon, at rates over 100 gram of carbon per square meter every year, more than 20 times that of tropical forest habitat which sequesters carbon at a rate of 4 grams per square meter every year (McLeod et al 2011, Li et al. 2018). Given the current climate crisis, not only preserving but expanding wetland habitat coverage will remain a critical task in managing global carbon budgets and achieving sustainable coastal civilizations.

While the benefits of coastal wetlands are now well studied, centuries of mismanagement have led to nearly a 40% loss in global coastal wetland acreage just in the past century (Davidson 2018). Due to the global extent of coastal wetlands, the driving force behind these losses can be dependent on a variety of factors but generally, these losses stem from a mix of anthropogenic pressures. Direct development of wetland habitats for agricultural, industrial, and residential purposes has led to significant losses (Coleman 2008, Hu 2017). Altering the natural hydrology of estuarine systems by diverting upstream water can impact the sediment supply to coastal wetlands and can indirectly lead to considerable losses in wetland acreage (Syvitski 2005). Similarly,

subsidence, a process which can be the result of tectonic activity as well as a process in which surface sediments compact due to groundwater withdrawal greater than recharge and petrochemical extraction, has not only led to major losses in wetland acreage but also led to coastal settlements around the globe facing the increasing threat of flooding (White and Tremblay 1995).

Despite facing intense anthropogenic pressures, coastal wetlands are known as one of the most resilient ecosystems on the planet. In the face of issues like subsidence and sea level changes, these ecosystems will migrate landward and maintain their position along the aquatic and terrestrial interface, if there is room to migrate. Over the past century, human activity has encroached upon this transition zone into the sea, built urbanized infrastructure and armored shorelines that prevent wetlands from migrating landward and causes these habitats to be converted into less valuable open water habitat, a global phenomenon known as “coastal squeeze” (Doody 2004, Pontee 2013).

Shoreline Armoring

While flooding related to sea level rise and subsidence seem like a relatively modern problem, the practice of shoreline armoring has been evidenced as old as 7000 years (Galili et al. 2019). Numerous structures have been engineered to protect coastal settlements from becoming inundated, ranging from concrete barriers that are miles long such as the seawall on Galveston Island to the small-scale residential bulkheads that armor coastal properties across the United States. Using material such as wood, vinyl, concrete, or crushed rock to armor shorelines has become common practice in areas where erosion, sea level rise, or subsidence may be threatening loss or damage of property (Gardner and Johnston 2021).

During the past century, coastal populations significantly increased as did the practice of armoring shorelines, leading to approximately 14% of the United States

shoreline currently being armored, with that number exceeding 50% in many large estuarine systems (Gittman et al. 2015). Along the east coast of the United States, approximately 10% of the open Atlantic shoreline has been armored, with nearly 15% of the interior shorelines of bays and tidal systems armored (Gittman et al. 2015). In heavily developed, urban areas, the percentage of shoreline armoring can reach 90%, with population density being a key predictor in the amount of shoreline armoring present. Additionally, it has been determined that adjacent shoreline armoring significantly increases the likelihood that neighboring property owners would also choose to armor their shoreline, resulting in a cascading effect that increases the overall amount of armoring (Beasley and Dundas 2021, Peterson et al. 2019, Scyphers et al. 2015).

Over the past few decades, studies have shown that traditional methods of shoreline armoring can reduce the many benefits that natural shorelines provide and even exacerbate the issues faced by shoreline property owners (Gittman et al. 2016b, Seitz et al. 2006). Armored shorelines reflect wave energy rather than absorbing it, sometimes leading to erosion on adjacent shorelines (Chapman and Underwood 2011, Fletcher et al. 1997). Additionally, when armored shorelines eventually fail and are submerged during storm events or from rising sea levels, the result can be a drastic retreat of shoreline and increased erosion compared to natural vegetated shorelines (Gittman et al. 2014, Lennon 1991).

Armored shorelines do not offer the shallow water protected nursery habitat that is provided by plant communities of natural shorelines (Balouskus and Targett 2018, Kornis et al. 2017, Toft et al. 2007). Similarly, armored shorelines are less likely to be used as nesting habitats for sea turtles and foraging habitats for shore birds (Dugan et al. 2008, Rizkalla and Savage 2011). Further, armored shorelines have been shown to have

lower abundance and species richness of benthic infauna, thus reducing prey availability and habitat quality (Lovall et al. 2017, Morley et al. 2012, Sobocinski et al. 2010)

There is less evidence explicitly suggesting direct negative impacts on water quality adjacent to armored shorelines. However, there is ample evidence showing that impervious urbanized infrastructure significantly increases runoff of stormwater and the accompanying nutrients and contaminants into estuarine systems (Carle et al. 2005, Freeman et al. 2019). It is reasonable to assume that to some extent, armored shorelines created may reduce water quality by allowing direct runoff of stormwater.

Collectively, these characteristics of armored shorelines contribute to the previously discussed anthropogenic pressures faced by coastal wetlands. Armored shorelines also require continuous maintenance or reconstruction making them costly and hardly a sustainable pursuit, with installation costs ranging from \$457-\$966 per linear foot and maintenance costs \$100 -\$500 per linear foot in the case the whole bulkhead doesn't need to be replaced (NOAA 2022). Since the negative ecological impacts of shoreline armoring have become apparent, alternative strategies have been developed. "Living shorelines", a term used to describe shoreline protection strategies that incorporate natural features and processes to maintain or enhance existing shorelines, became a common alternative to shoreline armoring beginning in the early 2000s (Currin et al. 2010).

Living Shorelines

A primary focus of shoreline protection is sediment stabilization and reducing wave energy impacting the shoreline. Living shorelines typically involve establishing native coastal wetland plant communities on shorelines. The above-ground mass of these plant communities attenuates wave energy allowing sediments to accumulate while the roots work to stabilize sediments along the shoreline. Along the Gulf of Mexico and

Atlantic coast, the primary species used to establish living shoreline projects is smooth cordgrass, *Spartina alterniflora* (syn. *Sporobolus alterniflorus*). This species is widespread and dominates along low marsh shorelines and has long been used to restore coastal wetlands due to its ability to tolerate a wide range of salinities and inundation (Broome et al. 1986, Broome et al. 1988, Craft et al. 1999, Travis and Grace 2010). Living shorelines projects have also used mangroves and common wetland plant species such as black needlerush, *Juncus roemerianus* (Fillyaw et al. 2021, Myszewski and Alber 2016, Peters et al. 2015). Designing living shorelines may involve planting multiple species strategically along the shoreline slope. *Spartina alterniflora* is typically planted at lower elevations near the water, while *Spartina patens* or similar high-marsh species may be used in the high marsh at higher elevations towards the upland boundary (GLO 2020).

Living shorelines in moderate or high energy wave environments will typically be designed to include breakwaters to reduce wave force on the shoreline. Breakwaters are placed a short distance off the shoreline and commonly consist of rock, concrete, or shell material. The reduction of wave energy provided by breakwaters of living shorelines reduces the pressure on plant communities to migrate upland and allows expansion of marsh vegetation at lower elevations (Martin et al. 2021, Cebrian and Sparks 2021). Breakwaters not only serve as a structural component of living shorelines but also can play a large role in the biology of living shoreline design. Oyster reefs, a common breakwater option for living shorelines, are known to serve as habitat for juvenile fish and crustaceans while also improving water quality (Coen et al. 1999, Grabowski and Peterson 2007).

A principle behind living shorelines is viewing shoreline environments along a spectrum ranging from soft, sloping, vegetated shorelines to hard, steep, concrete shorelines. Techniques used in the construction of living shorelines should be applied to

fit the situation and can be most effective when used in combination. For example, shorelines in high energy environments may not benefit from just softer ‘green’ techniques such as planting vegetation and may require harder ‘gray’ techniques such as rock or oyster shell breakwaters to achieve desired benefits (GLO 2020). Together, the structural and biological components of these engineered shorelines aim to mimic the many faculties that natural coastal wetlands provide such as sediment stabilization, providing habitat, and water filtration.

While the majority of living shoreline projects in the United States have been developed along the Atlantic and northern Gulf of Mexico coasts, living shoreline techniques are being applied on the Pacific coast and all over the world (Parker and Boyer 2019, Smith et al. 2020, Toft et al. 2021). While shoreline property owners are concerned about both habitat degradation and erosion, their decision on shoreline protection strategies is influenced by many other factors including the monetary cost and perceived benefits (Scyphers et al. 2020). By studying the ecology of living shorelines, coastal wetland conservationists can provide property owners and regulators with tools and knowledge to make informed decisions on creating sustainable shorelines.

Living Shorelines Research

A common theme among living shorelines research is the determination of whether living shorelines effectively stabilize sediments. Smith et al. (2020) conducted a thorough examination and concluded just over 50% of the living shoreline studies have addressed this topic (Smith et al. 2020). Polk and Eulie (2018) found that 12 of 17 living shorelines examined reduce erosion; 6 of those shorelines accumulating sediment rather than continuing to erode after installation of the living shoreline. The sites from this study were followed up on after Hurricane Florence and it was determined that compared to control sites, the living shoreline sites were more resilient and experienced significantly

less erosion (Polk et al. 2022). These findings are reinforced by other studies examining sediment stabilization along living shorelines (Chowdhury et al. 2019, Smith et al. 2018). A reduction in erosion and accretion of sediment has been associated with natural marsh systems primarily as a function of the plant community; accordingly, plant communities of living shorelines are designed with the hope of providing these benefits (Currin et al. 2017).

Early research on plant communities of living shorelines found that three years after construction living shorelines had lower stem densities than natural reference shorelines and only one out of the three living shorelines had reached equal stem densities as the reference shorelines (Currin et al. 2008). Later studies found that by 8 years after construction, living shorelines had similar or higher stem densities than natural reference shorelines (Currin et al. 2017, Gittman et al. 2016a). Given the evidence that plant communities of living shorelines may take a few years to develop, studying the development of these plant communities is useful in understanding the timeline in which benefits provided by living shorelines may be realized. Despite not being the focus of many studies, reported data of plant communities on living shorelines typically include stem densities and species cover in relation to more commonly studied biological aspects such as habitat usage.

Habitat usage of living shorelines is a primary focus of biological studies on living shorelines. Due to the importance of natural tidal wetland habitat, it is important to address whether or not living shorelines are offering similar habitat as natural shorelines. Living shoreline projects in North Carolina showed higher abundances and diversity of nekton on living shoreline sites compared to both reference and armored sites, suggesting that living shorelines provide enhanced nursery habitat (Gittman et al. 2016a). This conclusion was supported by more recent research finding that living shorelines offer

similar habitat as natural shorelines for both nekton and crustaceans (Guthrie et al. 2022). One study found that while habitat usage in the marsh of living shorelines was on par with natural marshes, sampling for nekton seaward of the breakwater of living shorelines did not reveal differences in habitat usage among shoreline types (Smith et al. 2021).

Along with nekton, a few living shoreline studies looked at the benthic, or sediment-dwelling, community. Benthic infauna are a primary source of food for nekton and have long been studied as indicator organisms for water quality and are a key aspect of restored wetlands achieving congruity with natural habitats (Spieles and Mitsch 2000). Davenport et al. found that benthic biomass at one living shoreline site increased from 3.7 g/m² to 7.4 g/m² over the three-year study period, an almost 100% increase while on the natural and armored shorelines increases in biomass were observed at 37% and 16% respectively (Davenport et al. 2018). Other studies on benthic communities of living shorelines indicated lower or similar abundance to natural shorelines (Bilkovic and Mitchell 2013). The benthic environment, often used as indicator of habitat quality, is known to be greatly influenced by urbanization and has noticeable community shifts in the presence of certain metals and pollutants (Inglis and Kross 2000). Understanding how benthic communities of living shorelines compare to natural sites is not only important in determining the quality of habitat provided but also can offer insight into nutrient, sediment, and contaminant dynamics.

Research into determining if living shorelines function similar to natural shorelines in terms of improving water quality is very limited in scope. There is evidence suggesting that nitrogen dynamics function similarly to natural shorelines and that living shorelines, like natural tidal wetlands, can mitigate the impacts of nutrient loading (Onorevole et al. 2018). Other studies examined the ability of living shorelines to improve water quality by focusing on oyster reefs and their ability to filter water. For

example, one study focused on chlorophyll levels in the water column upstream and downstream of oyster reefs (La Peyre et al. 2014). Another calculated the amount of water filtered based on density of oysters (Milbrandt et al. 2015). There are currently no studies reporting direct measurements of contaminants on living shorelines.

The previously discussed research of living shorelines is continuing to grow. Some of the current gaps in living shoreline research are due to the relatively young age of many living shoreline projects. 31 of the 41 papers included in Smith et al (2020) were examining living shoreline sites that were less than 10 years old at the time. Certain components of coastal wetland restoration such as benthic invertebrates can take up to 10 or 20 years to reach parity with natural sites, while other components such as plant communities, only take a few years (Broome et al. 2019, Craft et al. 1999). Understanding the development of these different components will require evaluating these habitats on longer timescales. Other gaps in research include a lack of studies comparing living shorelines to both natural and armored shorelines. The Smith review also concluded that while the majority of living shoreline studies compared data to a natural reference site, only 20% of the relevant literature compared living shoreline data to armored shorelines. Collecting data on armored shorelines allows living shoreline researchers to demonstrate that these habitats are not only functioning similarly to natural habitats but that they can be an improved shoreline protection strategy over traditional methods. Smith also highlighted other gaps in knowledge by identifying that only 14% of research includes biological diversity data on both plants and animals, with a large percentage reporting on solely the later. Since a primary component of most living shoreline designs includes planting vegetation, including data on both plants and animals allows researchers to not only demonstrate that the success of these constructed habitats, but also determine to what degree design components contributed to that success.

Perhaps one of the biggest factors contributing to gaps in knowledge of living shorelines is that the majority of research on living shorelines has been concentrated in one region. The majority of comprehensive and long-term studies on living shorelines have taken place in the Chesapeake Bay region, although it is expanding into other estuaries along the United States Atlantic and Gulf Coasts, where numerous living shoreline projects have been implemented. Living shorelines studies in the Gulf of Mexico have been mostly limited to Alabama, Louisiana, and Florida, and focused on the sediment stabilization function of living shorelines and whether or not created oyster reefs were successful. Expanding living shoreline research into Texas, where considerable losses of tidal wetland habitat have occurred, would be extremely beneficial to coastal restoration managers and property owners alike.

Living Shorelines in Galveston Bay

Given the local nature of subsidence and erosion in the region, the Galveston Bay system is in need of shoreline solutions. The greater Houston area has historically subsided at some of the highest rates on the globe with some areas subsiding as much as 3 meters in the past 100 years or 3 cm/year (Galloway et al. 2003). While this rate has slowed to about 0.5 cm/year due to major shifts towards surface water usage, over 100 km² of coastal wetlands have already been lost (HGSC 2022, Galloway et al, 2003). Coupled with subsidence, the Galveston Bay shoreline has also experienced significant erosion with rates averaging 2.4 ft/year since 1932, higher than the 1.8 ft/year from 1850-1932 in which 12.5 square miles of land was lost (Lester et al. 2002). Over 10% of shorelines in Galveston Bay have been armored with concrete, wooden, and vinyl bulkheads or docks (HARC 2022). Since the late 1980's 114 restored shorelines sites have been developed on the Texas coast, amassing over 37,000 acres with a large percentage of these projects existing in Galveston Bay (GLO 2020). Based on GIS data

from a Texas General Land Office site suitability tool (Figure 1), large portions of the interior coast of Galveston Bay have the potential for living shorelines development. In addition to significant erosion and subsidence, habitat in the Galveston Bay system is perhaps the most valuable along the Texas coast, with nearly 50% of all commercial finfish catch in the state of Texas occurring adjacent to Galveston Bay (HARC 2022). Further, Galveston Bay is home to many large-scale industrial operations, leading to portions of the bay being under Texas Department of State Health Services (TDSHS) advisories for consuming seafood. Many harmful chemicals enter the bay system through runoff from industrial areas. The presence of these contaminants may influence the success of living shoreline projects. These characteristics of Galveston Bay make it a perfect candidate for evaluating the success of living shorelines.



Figure 1. Texas General Land Office Site suitability tool for varying types of living shorelines in Galveston Bay, Texas. Red indicates shoreline not suitable for any type of living shoreline. All other colors indicate suitability for some type of living shoreline.

There is a need to monitor and evaluate the ecological trends of current living shoreline projects in the Galveston Bay region to determine the primary factors influencing the success of these modified habitats. Galveston Bay shoreline, like much of Texas, is privately owned rather than public. As a result, much of the shoreline is fragmented small parcels. Consequently, studies focusing on small scale living shoreline projects may be most effective in demonstrating the viability of living shorelines in the Galveston Bay system. So far, research on small scale living shorelines in the Galveston Bay system have been limited to a few pilot studies (Martin et al. 2014, Torres et al. 2020). As evidenced from previous studies in Galveston Bay, where research on restored marshes found they did not support the same populations of various aquatic species as reference marshes, it is important to evaluate the biological function of the habitat and not just created acreage to determine the success of restoration projects (Minello and Webb Jr 1997). While no research has been published on living shoreline projects in Galveston Bay, there is a strong foundation of research studying natural marshes as well as large scale restored wetlands in the region. Dobberstine and Howard (2019) analyzed multiple approaches to estuarine habitat restoration in Galveston Bay and compares the biological community composition of various restorative design strategies with that of natural marshes, as a measure of functional success. Studying the long-term success and ecological trends of these engineered environments can provide great insight into future shoreline protection efforts and increase the benefits provided by living shorelines in the Galveston Bay system and globally.

Study Objectives

The goal of this research was to evaluate the ecological function of three living shoreline sites in the Galveston Bay system. This study sought to take a comprehensive approach to evaluating the success of these constructed habitats by examining a variety of biological and environmental metrics of living shorelines compared to both armored and natural shorelines. As an indicator of sediment stabilization and wave attention, this study examined plant communities using metrics such as percent coverage, stem density, chlorophyll concentrations and plant biomass. The habitat quality of living shorelines was assessed based on both nekton and benthic abundances and community structure. Additionally, this study evaluated the ability of living shorelines to serve as a sink for contaminants, an often-noted function of coastal wetlands, by examining the sediment concentrations of selected heavy metals based on their known involvement in human health issues as well as their known role in estuarine systems.

This study hypothesized that living shorelines in Galveston Bay will exhibit similar ecological characteristics to natural shorelines and improved ecological function over armored shorelines. The primary objectives of this study were to 1) characterize the plant communities of living shorelines in comparison to natural shorelines. 2) compare habitat usage amongst shoreline types by measuring abundance and diversity of both nekton and benthic organisms 3) compare the sediment concentrations of selected heavy metals amongst shoreline types.

CHAPTER II:
METHODOLOGY

Study Area/Site Characteristics

The three sites selected in this study, shown in Figure 2, were sampled in pilot studies conducted by Lee College (Martin et al. 2018, Torres et al. 2020). Study sites were selected to represent fresh, brackish, and saltwater marshes. Each study site is within the Galveston Bay system. Within each study site, there are three shoreline types: a living shoreline (LVS), a natural reference shoreline (REF), and an armored shoreline (ARM). Data on plants, benthic communities, and sediment heavy metals was collected in June of 2021. Nekton data was collected during June and October of 2021. A brief summary of each site is provided in Table 1.

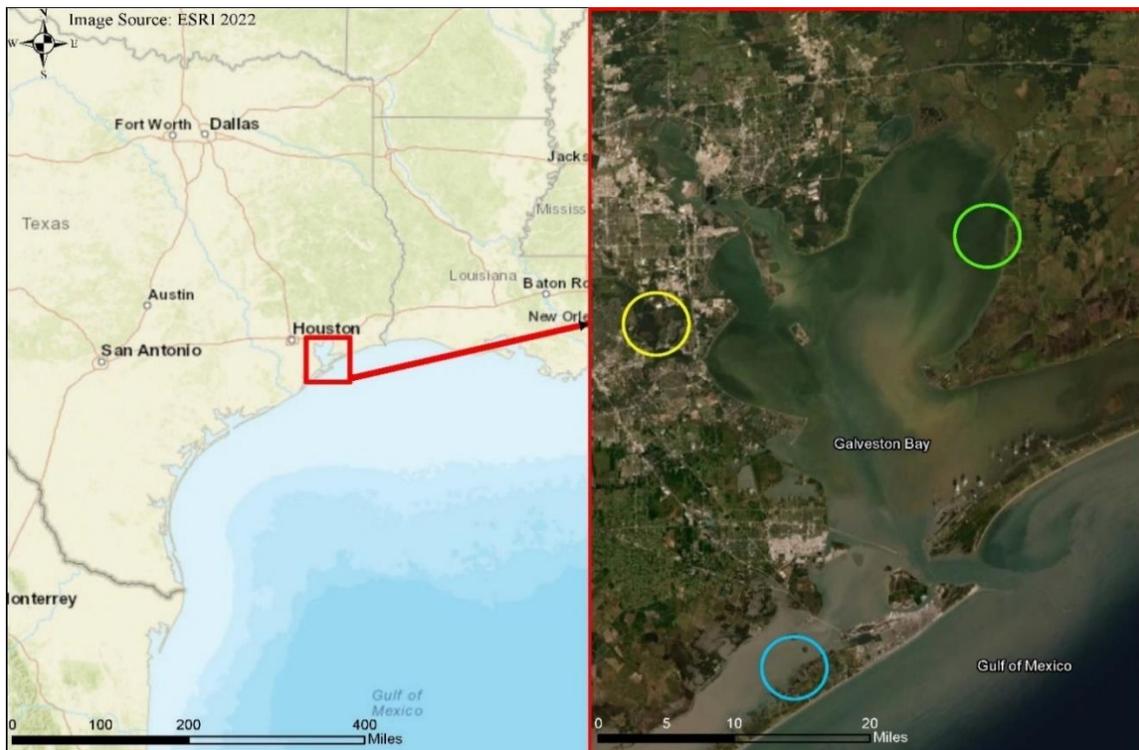


Figure 2. Left: Vicinity map of Study Area; Right: Armand Bayou sites circled in yellow, Trinity Bay sites circled in green, West Galveston Bay sites circled in blue.

Table 1

Site coordinates, age, and features of natural reference (REF), living shoreline (LVS) and armored (ARM) study sites.

Site Location	Site Name	Coordinates (lat/long) decimal degrees	Age at time of sampling (years)	Features/Description
Armand Bayou	(AB-REF)	29.594348, -95.084634	20+	Natural fringe.
	(AB-LVS)	29.577984, -95.068216	10	650 ft Riprap breakwater, spartina plantings.
	(AB-ARM)	29.567069, -95.069986	20+	Wooden bulkhead, frequently overtopped.
Trinity Bay	(TB-REF)	29.666186, -94.696561	20+	Exposed fringe adjacent to Trinity Bay.
	(TB-LVS)	29.665812, -94.696471	8	900 ft rock breakwater, spartina plantings.
	(TB-ARM)	29.665144, -94.697100	10+	Vinyl bulkhead put in place of failed bulkhead from Hurricane Ike.
West Galveston Bay	(WGB-REF)	29.255434, -94.917530	20+	Natural salt marsh, popular fishing spot.
	(WGB-LVS)	29.222317, -94.934097	16	350 ft low profile breakwater (concrete bags), spartina planting.
	(WGB-ARM)	29.229460, -94.919325	20+	Concrete bulkhead lining canal, developed residential area.

Armand Bayou Site Characteristics

Armand Bayou is a tidally influenced tributary to Galveston Bay, located along the western side of the upper reaches of Galveston Bay. The Armand Bayou watershed receives drainage from heavily developed areas southeast of Houston, including the primarily industrial areas of Pasadena, Deer Park, and La Porte. Due to the heavily developed nature of its surroundings, shoreline wetland losses in Armand Bayou stem from a mixture of factors but most notably from historical subsidence. Due to extensive ground water usage in the mid-20th century, significant subsidence was observed in the Armand Bayou system resulting in a 91% decrease in acreage of wetlands by 1979

(McFarlane 1991). While subsidence has slowed, Armand Bayou is a great example of how subsidence can convert fringing wetland habitats into open water, as McFarlane also noted that Mud Lake (recently renamed as Lake Mark Kramer), in the lower reaches of Armand Bayou, expanded from 100 acres to 325 acres. Study sites in the Armand Bayou system, shown in Figure 3, were located in Mud Lake, where Armand Bayou widens and flows into Clear Lake, and eventually, Galveston Bay. Historical data shows that the surface salinity values for Mud Lake average about 6.7 ppt from 1998 to 2003 (HGAC 2006). This study area was chosen to be representative of freshwater or low salinity shoreline sites.

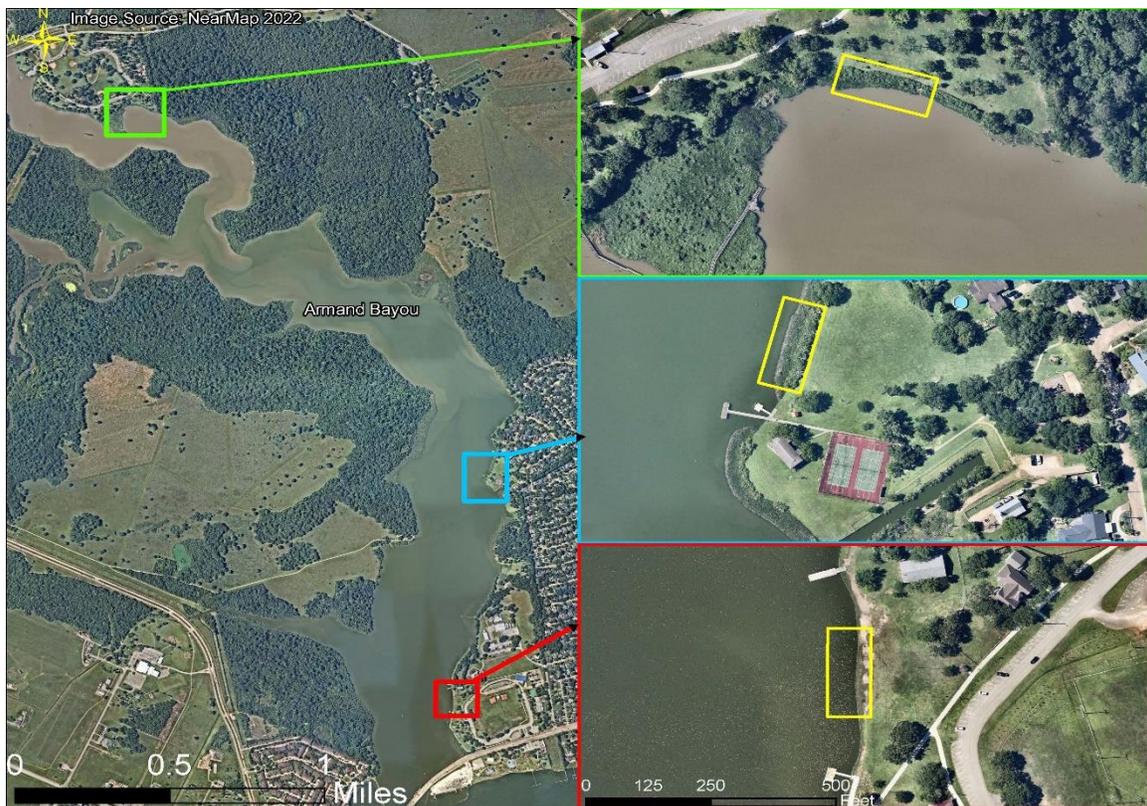


Figure 3. Left: Map of Armand Bayou Site; Right top: reference shoreline, Right middle: living shoreline, Right bottom: armored shoreline; sample areas shown in yellow.

The natural reference shoreline was located the furthest upstream along the eastern bank of Armand Bayou just south of Bay Area Park. Wetland habitat along this stretch has remained relatively undisturbed for at least forty years. The living shoreline site for this study area was located to the south along the same eastern bank of Armand Bayou and was constructed in 2011 (GLO 2020). Engineered features of this living shoreline site include 650 ft of concrete riprap as a breakwater and planted *Spartina alterniflora* and *Spartina Patens*. Fill material was brought in behind the breakwater to create a sloping shoreline, where previously there was open water. The armored shoreline site for this study area was the furthest south, just near the Mud Lake and Clear Lake confluence along the eastern shoreline. This site is composed of a wooden bulkhead that has been in place for over twenty years.

Trinity Bay Site Characteristics

Trinity Bay is in the northeastern portion of Galveston Bay and receives drainage from the Trinity River. Unlike Armand Bayou, Trinity Bay shorelines are not as developed as other areas of Galveston Bay, however the nearly 10 miles of fetch length over the bay and being near the outflow of the Trinity River create strong erosional forces. While Trinity Bay is tidally influenced, mean salinity values for the bay are 12 ppt due to the bay's proximity to freshwater inflow from the Trinity River (Rayson et al. 2015). As a result of Trinity Bay's salinity levels being largely dependent on freshwater inflow, the conditions at this location can greatly vary on monthly or yearly basis. Due to the dynamic salinity observed in the Trinity Bay, this site was chosen to represent brackish shoreline sites.

The shorelines selected within Trinity Bay, shown in Figure 4, were along the eastern coast of the bay. The natural reference shoreline was the northern most site and has remained undeveloped for over forty years, but it was clear upon site visit that the site

is experiencing significant stress from erosion and based upon observations from the previous pilot study the shoreline has retreated up to ten meters (Jim Dobberstine, verbal communication). The living shoreline site located directly adjacent to the south of the reference site was constructed in 2013 (GLO 2020). This site featured over 900 feet of shoreline planted with *Spartina alterniflora* and protected by rock breakwater. The armored shoreline site is to the south and consists of vinyl bulkhead constructed in 2008 in place of previous bulkhead that failed during Hurricane Ike.

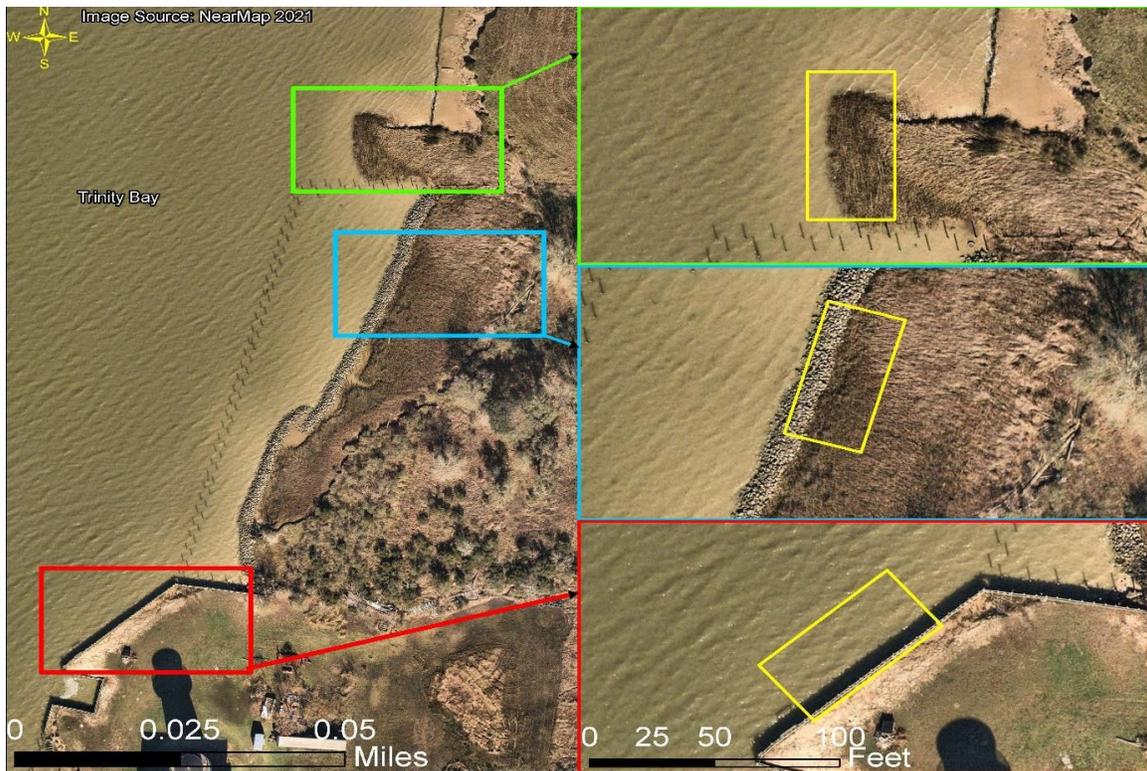


Figure 4. Left: Map of Trinity Bay Site; Right top: reference shoreline, Right middle: living shoreline, Right bottom: armored shoreline; sample areas shown in yellow.

West Galveston Bay Site Characteristics

The West Galveston Bay shoreline sites were located on the bay side of Galveston Island in the southwestern portion of the Galveston Bay system. Shorelines in this part of Galveston Bay are subject to numerous erosional forces including waves from vessel

traffic, tidal fluxes, and storm surge. Due to the proximity to the Gulf of Mexico, salinity at these sites is typically higher than either the Armand Bayou or Trinity Bay sites and average about 20 ppt (HARC 2022). These sites were chosen to be representative of saltwater shorelines.

The three shorelines chosen for the West Galveston Bay sites, shown in Figure 5, are not as close together as the Armand Bayou sites or the Trinity Bay Sites. All three sites were along the northern shoreline of Galveston Island are close enough that habitats, erosional forces, and general characteristics should be similar. The natural reference site was located off Sportsman Road and has remained mostly undisturbed for over forty years. The living shoreline site was located on the eastern shoreline of Eckert Bayou, a small inlet of West Galveston Bay. This shoreline features 340 feet of Sakrete concrete breakwater and marsh grasses planted in 2005 (Haille Leija, Galveston Bay Foundation, verbal communication,). The armored shoreline is a concrete bulkhead lining a canal in a residential area, this bulkhead has been in place for 20 years.



Figure 5. Left: Map of West Galveston Bay Site; Right top: reference shoreline, Right middle: living shoreline, Right bottom: armored shoreline; sample areas shown in yellow.

Field Methods

Plant Communities

In order to determine if plant communities of living shorelines are comparable to natural shorelines, methods adapted from Brower et al. (1998) and Thayer (2005) were employed to collect data on coverage, density, diversity, and productivity as an indicator of functional success. A 10 meter transect was placed diagonally through vegetation originating near the shoreline and extending towards open water, shown in Figure 6. A square meter plot was placed at the origin, center, and terminal ends of the transects. All plant species within the square meter plots, shown in Figure 7, were identified and their relative coverages were recorded. Stem counts of each plant species in the square meter plots were recorded to assess density. An atLEAF chlorophyll CHL STD meter was used

to measure chlorophyll concentrations of 5 randomly selected *Spartina alterniflora* plants within the plot. A 1/8 m² plot of *Spartina alterniflora*, both roots and shoots, were excavated from the opposite side of the transect as the 1 m² plots and returned to the laboratory for biomass analysis. Armored sites did not have vegetation along the shorelines therefore vegetation data was not recovered from these shoreline sites.



Figure 6. Photo of 10 meter transect placement and orientation for West Galveston Bay LVS site.



Figure 7. Photo of square meter plot used for plant community sampling at the Armand Bayou REF site.

Nekton Activity

Methods for comparing habitat usage amongst natural, armored, and living shorelines included measuring the abundance and diversity of nekton along the shorelines. Nekton data was collected using 10" x 18" minnow traps with one 2-inch opening on each end. Four traps were placed equidistant along a ten meter transect parallel to the shoreline. Traps were in place for 24 hours and then retrieved the following day, as to catch a full tidal cycle. Upon retrieval, nekton organisms were placed into glass viewing boxes for identification and enumeration and then released back into the wild.

Benthic Macroinvertebrate Communities

As another measure of habitat usage, data on the abundance and diversity of benthic macroinvertebrates inhabiting the top layer of sediment was also compared amongst shoreline types. Benthic communities were sampled by collecting sediment

cores along the shoreline. A total of five cores were taken from the 1 m² plots used for plant community analysis. For armored sites, cores were taken along a 10 meter transect parallel to the shoreline. Each core was approximately 10 centimeters long and 5 centimeters in diameter. Upon recovery of each core, the resulting sediment was immediately placed in a Ziploc bag and preserved in 10% neutral buffered formalin. Samples were gently massaged to ensure formalin reached all organisms in the sample and then stored in a cooler until returned to the laboratory.

Sediment Heavy Metals

Sediment cores for heavy metal analysis were collected from the same locations as benthic cores. However, only one 5 by 5 centimeter core was collected from each plot, totaling 3 cores per transect. A plastic core tip and insert was placed inside of the coring device to ensure samples were not contaminated by contacting the metal surface of the coring device. Cores were retrieved, placed into Ziploc bags, stored on ice, and then refrigerated until laboratory analysis.

Laboratory Methods

Plant Communities

Excavated *Spartina alterniflora* samples were washed to remove sediments leaving only plant material. The plants were then cut at the root collar to separate the roots from the shoots. Both roots and shoots were then weighed separately. After initial weight was measured, all plant material was lightly wrapped in aluminum foil and dried in an oven at 100°C for at least 24 hours, or until all moisture was removed. Once dried, the roots and shoots were once again weighed separately.

Benthic Macroinvertebrate Communities

Sediment cores collected for benthic community analysis were washed through a #60 (0.5 mm) mesh sieve, using a gentle stream of tap water to remove all sediment. All

material remaining on the sieve was transferred to plastic containers and preserved in 10% buffered formalin. Samples were then placed in solution of 50% Sudan IV and 50% Eosin B for a minimum of 72 hours to stain organisms for sorting and identification. After staining, samples were washed on #200 (0.062 mm) mesh sieve and preserved in formalin until ready to be sorted. Upon beginning the sorting process, samples were washed through the #200 (0.062 mm) mesh sieve to remove formalin and then placed in ethanol while sorting to limit formalin exposure.

During sorting, a small amount of the sample was poured into a 4-inch square plastic grid tray and viewed under low power on a stereo dissecting scope. The trays were worked from left to right, following the grid lines from the top to the bottom of the tray. The trays were then rotated 90° and reworked to ensure all organisms were removed. This process was repeated for another aliquot of the sample until the entire sample was sorted. Benthic organisms were identified to the lowest taxon possible, using the appropriate dichotomous keys. All possible efforts were made to identify polychaetes, crustaceans, insects, and mollusks to species, whereas other taxa were only able to be identified to genus or order. Only pieces with intact heads were enumerated on the data worksheet.

Sediment Heavy Metals

Sediment cores were dried at 95°C for at least 24 hours. Sediment was then pulverized using a mortar and pestle and any plant and shell material was removed from the sample using forceps. EPA Method 3050B was used to digest sediments and extract metals into an aqueous solution as follows. Three 1-gram samples were taken from each sediment core and placed into labeled 50-ml HotBlock™ digestion tubes. 10 ml concentrated nitric acid (trace metal grade) was added to each HotBlock™ digestion tube and heated on a HotBlock™ digester at 95°C for 30 minutes. Samples were then cooled

to room temperature and an additional 2.5 ml concentrated nitric acid was added and the samples were again heated on the HotBlock™ digester for another 30 minutes. This process of nitric acid additions, heating and cooling was repeated until no brown fumes were generated and sediment was thoroughly solubilized. Samples were then cooled completely and 1 ml of 30% hydrogen peroxide was added, and samples were heated for 30 minutes. After cooling, samples were 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 samples were analyzed for the following heavy metals: cadmium chromium, copper, lead, nickel, and zinc.

Data Analysis

Microsoft Excel and Minitab® v. 20.4 (2021) software packages were used to manage and analyze collected data. 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. For chlorophyll production, the average chlorophyll concentrations of the three readings for each plant was used as one of five replicates for each plot on the transects. A two-sample t-test was used to detect significant differences in means for plant community metrics of natural reference and living shoreline site

For nekton abundance, the sum of individuals from all traps along a transect for both months was combined. Nekton species richness was calculated for each shoreline as the total number of species observed across all traps for both seasons. Similarly, taxa richness for benthic macroinvertebrate communities was calculated as the total number of individuals observed across all sample plots for each shoreline. Total benthic abundance was calculated for each shoreline as the total number of individuals observed along a transect. Mean benthic abundance values were calculated for each shoreline as the average number of individuals found within 1 m² plots on that shorelines transect. One

way ANOVA and Tukey's pairwise comparisons were used to detect significant differences in mean benthic abundance between locations and shoreline types. For plant, nekton, and benthic community analysis, Shannon-Weiner species diversity Index (H) was calculated using the following formula:

$$H = -\sum[(pi) \times \ln(pi)]$$

where pi is equal to the proportion of the entire community made up of species i . so that for each species, pi is equal to the number of individuals of that particular species divided by the total number of individuals of the community.

For benthic community analysis, Piuolo's Evenness index was calculated using the following formula:

$$J' = \frac{H'}{\ln(S)}$$

where J' is community evenness on a scale of 0 to 1 and H' is the Shannon-Wiener diversity index and S is the number species observed at that site. When J' is equal to 1 all species are equally represented in the community, values closer to 0 indicate dominance by certain species.

Sediment heavy metal concentrations were analyzed by performing one-way ANOVA and Tukey's pairwise comparisons to detect significant differences in the concentrations of each metal and combined metals amongst shoreline types at each location. Sediment heavy metal concentrations were compared against the Long and Morgan Effects Range Low criteria (Long and Morgan 1990, Long et al. 1995). The Long and Morgan ERL is a widely established screening benchmark recommended by NOAA and reflects the lowest concentrations at which a contaminant will start to impact sensitive species (Buchman 1999).

For each metal whose concentration exceeded the ERL, a linear regression model was performed to determine whether that metal's concentration was a predictor of benthic macroinvertebrate abundance.

CHAPTER III:

RESULTS

Comparison of Plant Communities

Density, Coverage, and Diversity

Stem densities at Armand Bayou LVS (75 per m²), shown in Figure 8, were about half of the amount found at REF (138). Armand Bayou REF was dominated by *Spartina alterniflora* (20% coverage, shown in Figure 9), while none was found at the LVS. At the LVS, *Schoenoplectus pungens* was the dominant plant species (28.3%), while only a small amount was observed at REF (1.7%). REF had greater total coverage (50%), compared to LVS (38.3%).

At Trinity Bay stem density was slightly higher at LVS (65 per m²) than REF (45 stems per m²). Trinity Bay LVS was dominated by *Schoenoplectus pungens* (25%) and REF was dominated by *Spartina alterniflora* (11.7%). Total coverage at Trinity Bay LVS (47.7%) was higher than at REF (15%).

The West Galveston Bay LVS had a stem density of nearly 300 stems per m², double the amount of REF (150). *Spartina alterniflora* was the only observed plant species for both shorelines at the West Galveston Bay site. At West Galveston Bay significantly higher total coverage was observed at LVS (86.7%) than REF (31.3%) ($p \leq 0.05$). LVS in West Galveston Bay was the only shoreline to be observed with greater than 50% total coverage.

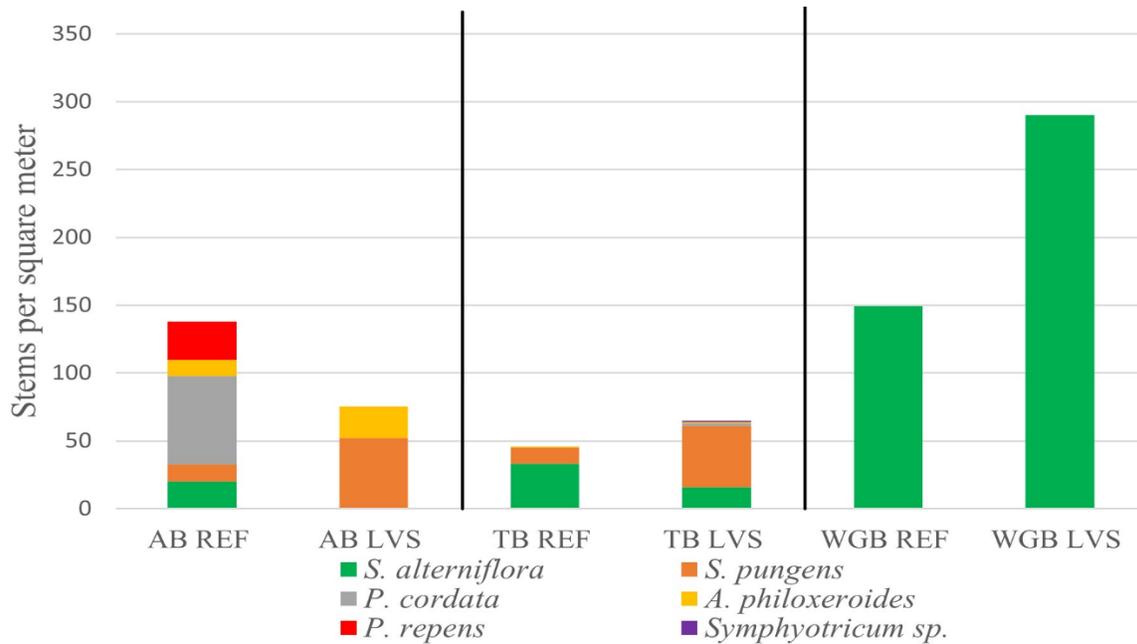


Figure 8. Stem densities of plant communities, AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = Reference shoreline, LVS = Living shoreline.

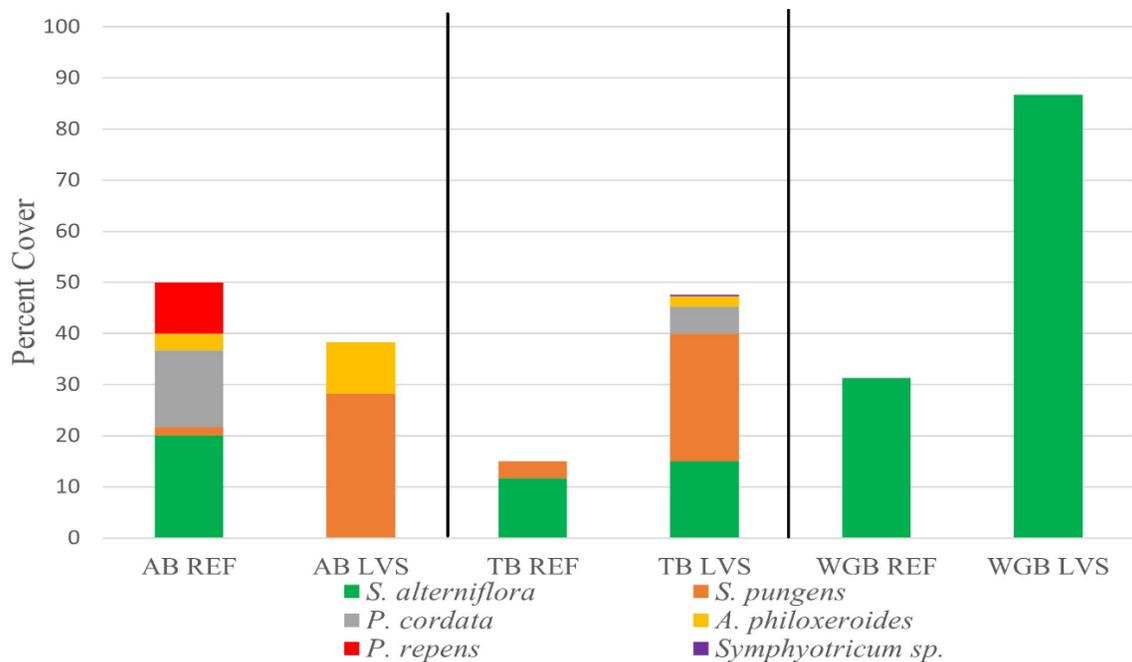


Figure 9. Species percent cover of plant communities, AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site. For WGB total percent cover was significantly higher at LVS than REF ($p \leq 0.05$).

Species diversity at Armand Bayou, shown in Figure 10, was higher at REF ($H' = 0.6024$) than LVS ($H' = 0.26722$). For Trinity Bay, species diversity was highest at LVS ($H' = 0.36947$), and lowest at REF ($H' = 0.24854$). At West Galveston Bay, species diversity at both LVS and REF was zero due to *Spartina alterniflora* being the only species present.

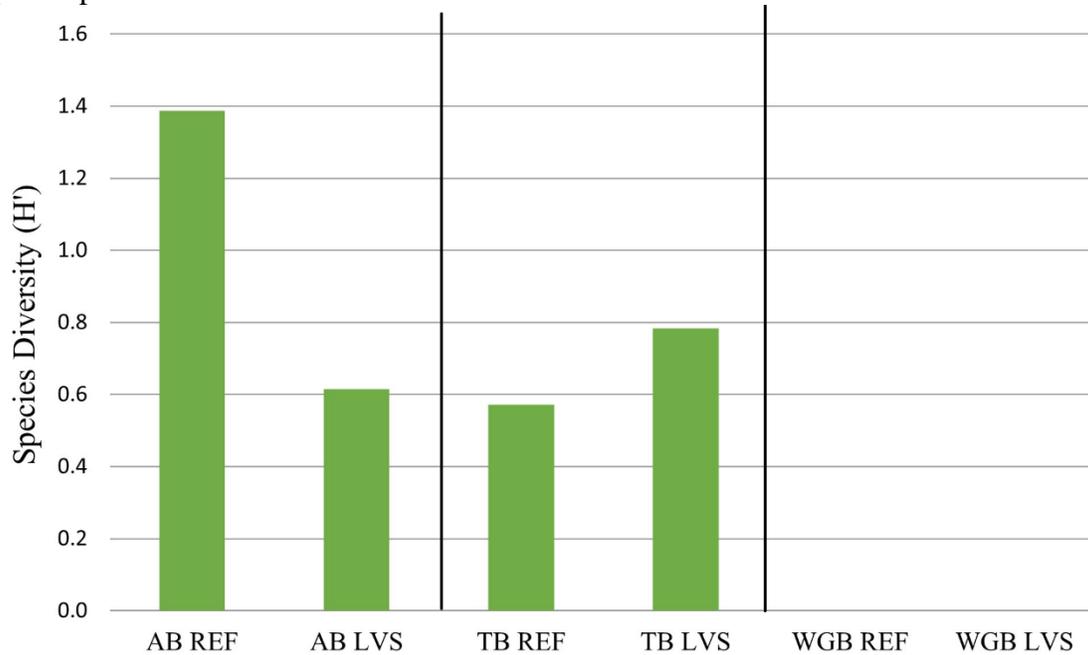


Figure 10. Plant Community Species Diversity calculated using Shannon Wiener Index. Diversity was zero at WGB due to *S. alterniflora* being the only species present.

Spartina Productivity

For Armand Bayou and Trinity Bay, biomass of both roots and shoots was similar between LVS and REF sites. For West Galveston Bay, 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 11.

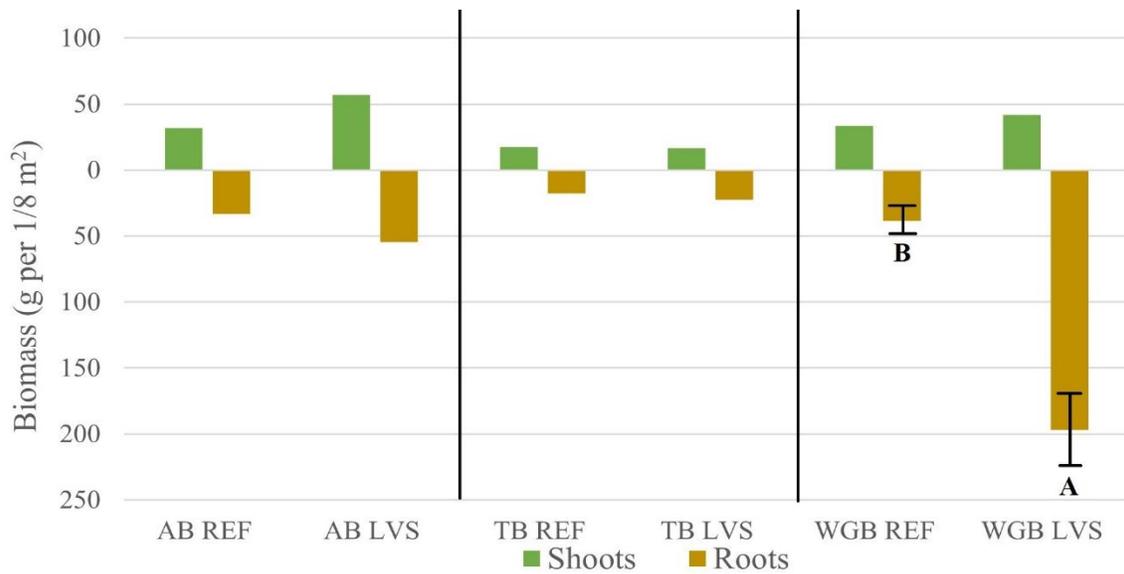


Figure 11. *Spartina alterniflora* shoot vs root biomass. AB = Armand Bayou, TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference site, LVS = living shorelines site. Error bars indicate significant differences detected for that site, shorelines with different letter groupings are significantly different.

Spartina chlorophyll concentrations, shown in Figure 12, were significantly higher at LVS than REF for 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.

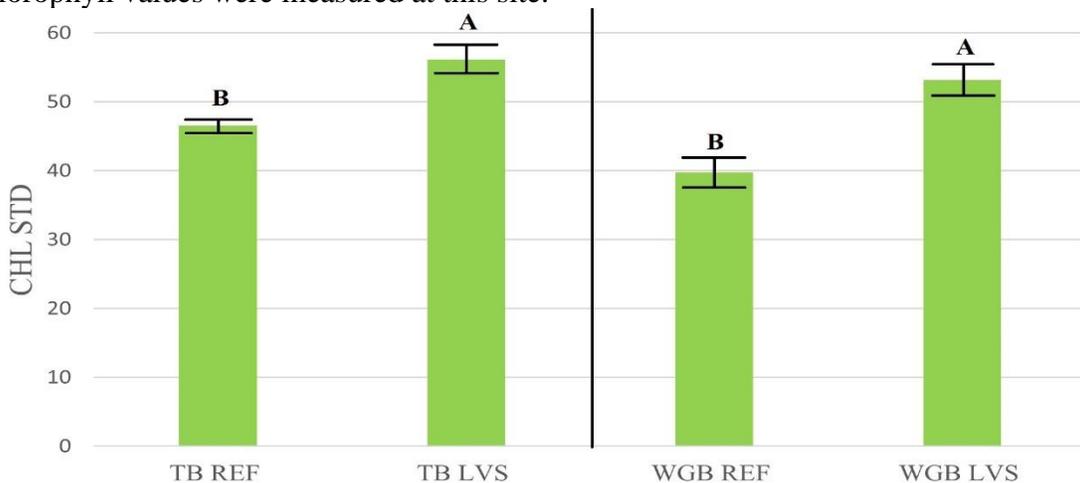


Figure 12. *Spartina alterniflora* chlorophyll readings. TB = Trinity Bay, WGB = West Galveston Bay, REF = natural reference shoreline, LVS = living shoreline. Error bars indicate significant differences detected for that site, shorelines with different letter groupings are significantly different.

Habitat Usage Amongst Shorelines

Nekton

A total of 648 individual nekton organisms and 12 different taxa were observed across all sites, shown in Table 2. Half of nekton taxa were observed in small abundances, with 6 of the 12 observed taxa having less than 10 total individuals observed across all sites. Crustaceans (shrimp and crabs) accounted for 599 of the total individuals observed. Fish species were observed in low abundances across all sites with a total of only 48 individuals.

At Armand Bayou, total abundance was highest at REF (76 individuals), shown in Table 2, followed by LVS (46), catch was composed primarily of White Shrimp, (*Litopenaeus setiferus*) for both REF (45) and LVS (40). The ARM site had the lowest abundance with only 6 individuals being found. For Trinity Bay total nekton abundance was highest at LVS (117), twice the amount found at ARM (62), with catch amongst all shoreline types dominated by Grass Shrimp (*Palaemonetes vulgaris*). The lowest total abundance was observed at REF (28). For WGB, total abundance was similar between LVS (146) and REF (126), and much lower at ARM (43) The most abundant taxa found at all three shorelines was Grass Shrimp.

Table 2

Total abundance of nekton collected in traps during June and October 2021. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline.

Scientific name	Common name	Armand Bayou			Trinity Bay			West Galveston Bay			Total
		REF	LVS	ARM	REF	LVS	ARM	REF	LVS	ARM	
<i>Palaemonetes vulgaris</i>	Grass Shrimp	2	0	5	25	106	47	84	97	27	393
<i>Litopenaeus setiferus</i>	White Shrimp	45	40	0	0	0	1	12	42	9	149
<i>Callinectes sapidus</i>	Blue Crab	10	4	1	3	8	8	12	3	5	54
<i>Armases cinereum</i>	Squarebacked marsh crab	0	1	0	0	1	0	0	0	0	2
<i>Pagurus sp.</i>	Hermit Crab	0	0	0	0	0	0	1	0	0	1
<i>Lagodon rhomboides</i>	Pin Perch	4	0	0	0	2	2	3	0	0	11
<i>Ctenogobius boleosoma</i>	Darter Goby	0	0	0	0	0	3	11	0	2	16
<i>Paralichthys lethostigma</i>	Southern Flounder	0	0	0	0	0	1	0	0	0	1
<i>Micropogonias undulatus</i>	Atlantic Croaker	1	1	0	0	0	0	3	2	0	7
<i>Gambusia affinis</i>	Western Mosquitofish	10	0	0	0	0	0	0	0	0	10
<i>Fundulus grandis</i>	Gulf Killifish	2	0	0	0	0	0	0	0	0	2
<i>Syngnathus sp</i>	Pipefish	1	0	0	0	0	0	0	0	0	1
Grand Total		75	46	6	28	117	62	126	144	43	648

Nekton taxa richness, shown in Figure 13, was highest at the REF site for Armand Bayou, with 9 different taxa being observed. Only 4 species were observed at the Armand Bayou LVS, and half that found at the ARM site. For Trinity Bay, the highest taxa richness was found at the ARM site (6), followed by LVS (4) and REF (2). For WGB, taxa richness was similar amongst shoreline type with the highest being observed at REF (7) and the lowest at ARM and LVS (4 for both).

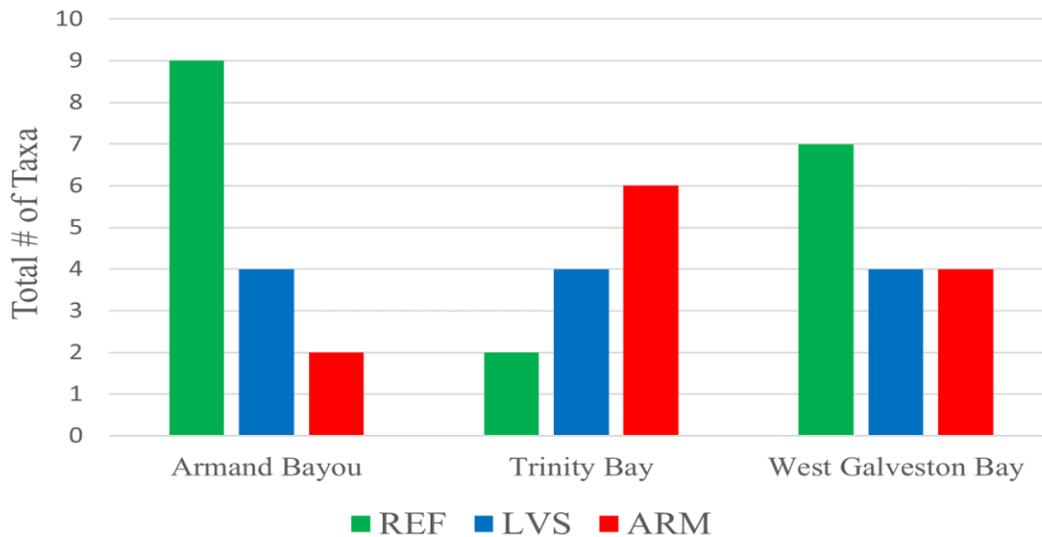


Figure 13. Total number of different nekton taxa observed for June and October 2021. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline.

Benthic Macroinvertebrates

Benthic macroinvertebrate abundance was one of few metrics in which there was not significant difference among locations (i.e. Armand Bayou, Trinity Bay, West Galveston Bay). This allowed for some comparison of shoreline types using data from all locations pooled together. Pooled data from all locations revealed both REF and LVS shoreline types had significantly higher mean benthic abundance than ARM sites ($p \leq 0.05$), as shown in Figure 14.

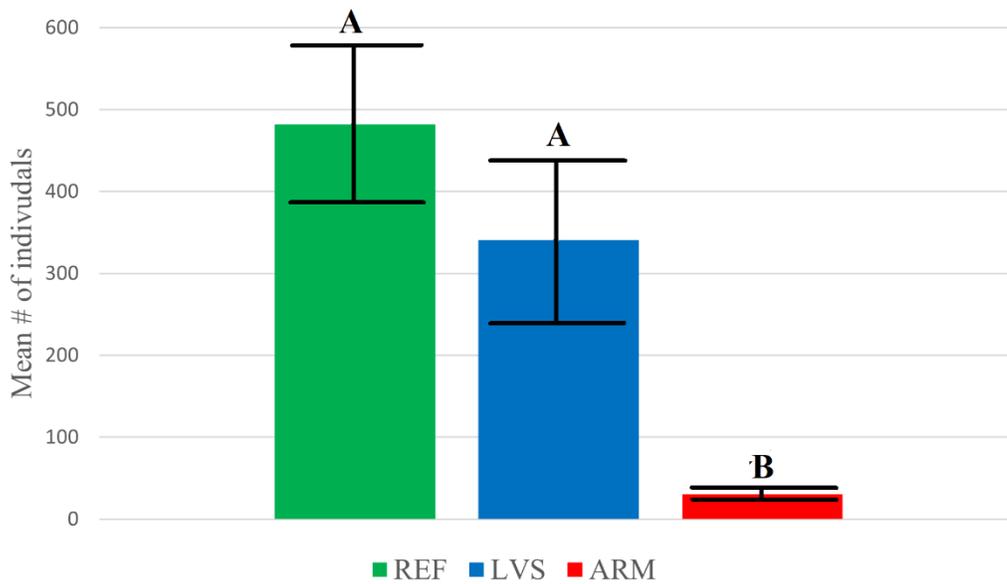


Figure 14. Comparison of mean benthic abundance of shoreline types across all locations. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline. Shorelines with different letter groupings are significantly different ($p \leq 0.05$).

Additionally results from benthic macroinvertebrate community analysis showed that both total abundance and taxa richness, shown in Figures 15 and 16, was highest at REF and lowest at ARM for all three locations. Total abundance ranged from a high of 2077 individuals at TB REF and a low of 51 individuals at TB ARM. A summary of identified taxa is provided in Table 3. The highest taxa richness was observed at AB REF (22), and the lowest at TB ARM (5).

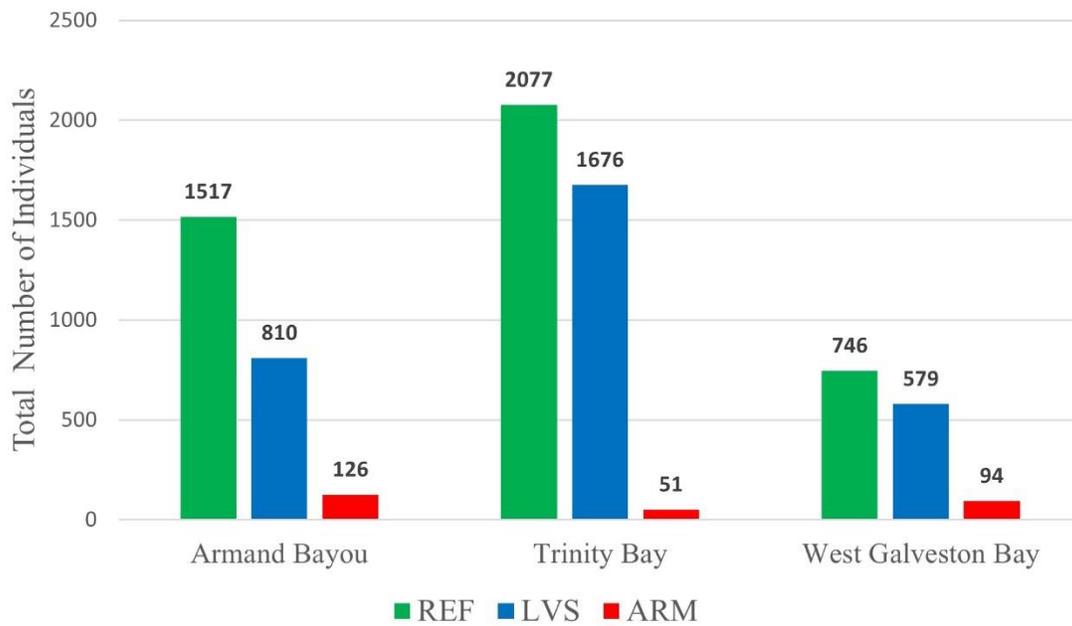


Figure 15. Benthic macroinvertebrate abundance. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline.

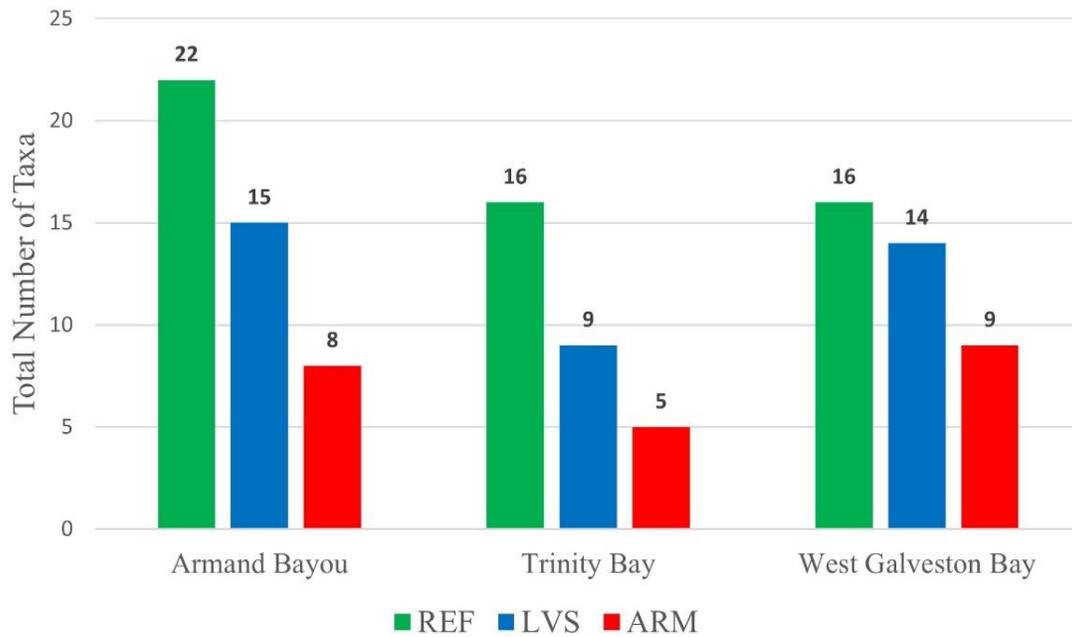


Figure 16. Total number of benthic macroinvertebrate taxa. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline.

Table 3

List of benthic macroinvertebrate taxa identified across all samples. Listed taxa were used in taxa diversity calculations. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline.

Taxa	Armand Bayou			Trinity Bay			West Galveston Bay			Total
	REF	LVS	ARM	REF	LVS	ARM	REF	LVS	ARM	
<i>Nemertea</i>	6	0	1	1	0	0	1	0	1	20
<i>Nematoda</i>	154	292	8	514	1463	11	486	57	11	5992
<i>Oligochaeta</i>	364	149	18	176	165	34	123	201	15	2490
<i>Capitella capitata</i>	4	1	2	8	3	0	41	27	2	176
<i>Hobsonia florida</i>	40	4	0	0	0	0	2	1	2	98
<i>Streblospio benedicti</i>	3	0	21	9	0	1	47	13	56	300
<i>Laeonereis culveri</i>	14	0	0	3	0	0	5	1	0	46
<i>Alitta succinea</i>	27	23	0	4	3	0	19	174	1	502
<i>Hermundura fauveli</i>	0	2	0	3	0	0	1	0	0	12
<i>Polydora</i>	0	0	0	0	0	0	1	0	0	2
<i>Polychaete sp</i>	0	1	0	0	0	0	0	0	0	2
<i>Macoma</i>	1	3	0	0	0	2	0	0	0	12
<i>Rangia</i>	3	0	0	0	0	0	0	0	0	6
<i>Gastropoda</i>	24	6	0	15	0	0	0	2	5	104
<i>Uca spp</i>	0	0	0	0	0	0	0	1	0	2
<i>Ostracoda</i>	124	30	1	3	5	0	2	2	1	336
<i>Copepoda</i>	53	0	0	4	2	0	1	25	0	170
<i>Harpacticoida</i>	1	2	0	1	0	3	7	0	0	28
<i>Cyclopoida</i>	20	0	0	0	0	0	1	0	0	42
<i>Isopoda</i>	0	12	0	7	0	0	0	0	0	38
<i>Cladocera</i>	5	0	0	0	0	0	0	0	0	10
<i>Tanaidacea</i>	7	0	0	822	1	0	8	36	0	1748
<i>Amphipoda</i>	52	36	11	499	3	0	1	18	0	1240
<i>Decapoda</i>	1	0	0	0	0	0	0	0	0	2
<i>Mysidacea</i>	3	0	0	0	0	0	0	0	0	6
<i>Insecta (Chironomidae)</i>	610	246	64	8	31	0	0	21	0	1960
<i>Chordata (Tunicata)</i>	1	3	0	0	0	0	0	0	0	8
Grand Total	1517	810	126	2077	1676	51	746	579	94	15252

Taxa diversity at Armand Bayou, shown in Figure 17, followed a similar pattern as total abundance and richness with the highest diversity at REF and the lowest at ARM. Pielou's Evenness Index, shown in Figure 18, was similar amongst shoreline types for Armand Bayou, with ARM (0.7) being slightly higher than REF and LVS (both 0.6). For Trinity Bay, the highest diversity was observed at REF and the lowest at LVS. Pielou's Evenness Index for Trinity Bay was similar between REF and ARM, 0.61 and 0.52 respectively. The LVS at Trinity Bay had the lowest evenness across all locations and shorelines (0.22). For West Galveston Bay, taxa diversity was highest at LVS (1.8) with similar diversity values at REF and ARM, 1.2 and 1.3 respectively. Community evenness followed a similar pattern as diversity with the highest being LVS (0.67) followed by ARM (0.59), and REF (0.43).

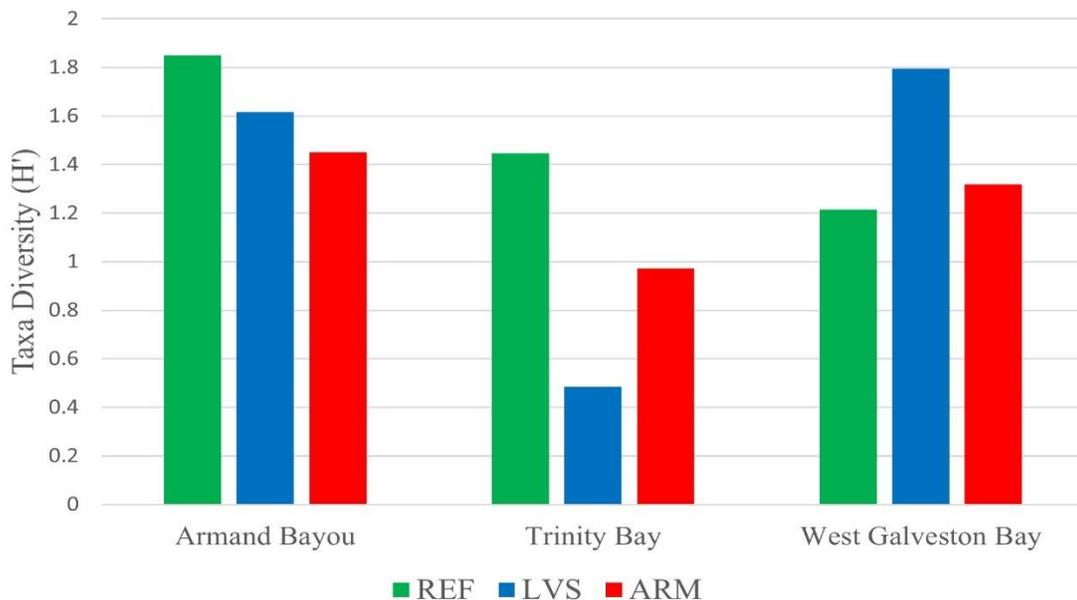


Figure 17. Shannon Wiener Species Diversity Index (H') for benthic macroinvertebrate communities. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline.

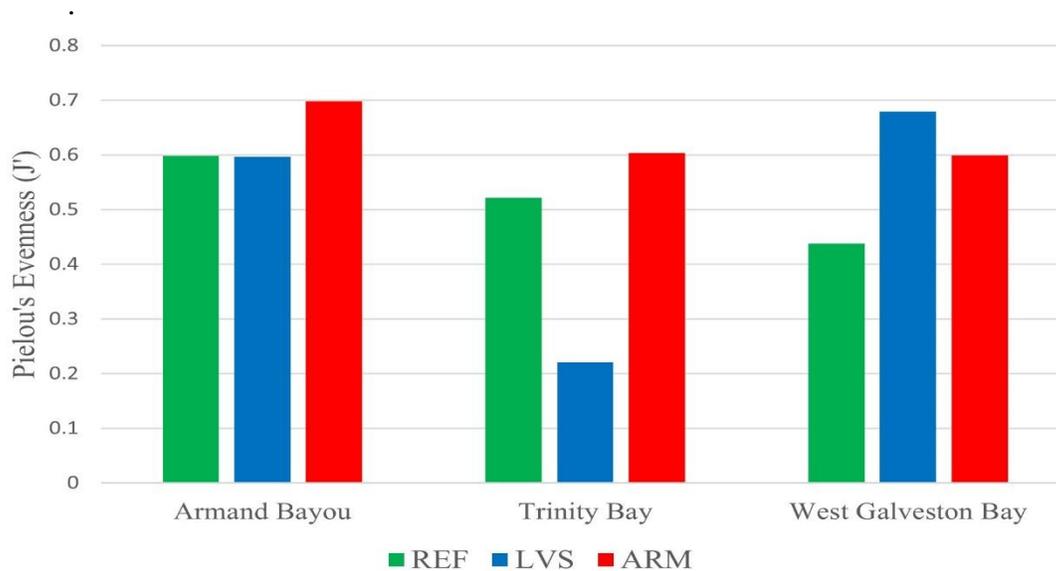


Figure 18. Pielou's Evenness for benthic macroinvertebrate communities. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline.

Comparison of Sediment Heavy Metal Concentrations

Armand Bayou sediments had higher combined concentrations of all measured heavy metals, shown in Figure 19, than either the Trinity Bay or West Galveston Bay sites ($p \leq 0.05$ for all significant differences). For Armand Bayou, sediments at REF and LVS had significantly higher concentration of combined metals than ARM. At Trinity Bay, sediments at LVS and ARM had significantly higher concentrations of combined metals than REF. For West Galveston Bay, REF sediments had a significantly higher concentration of combined metals than ARM.

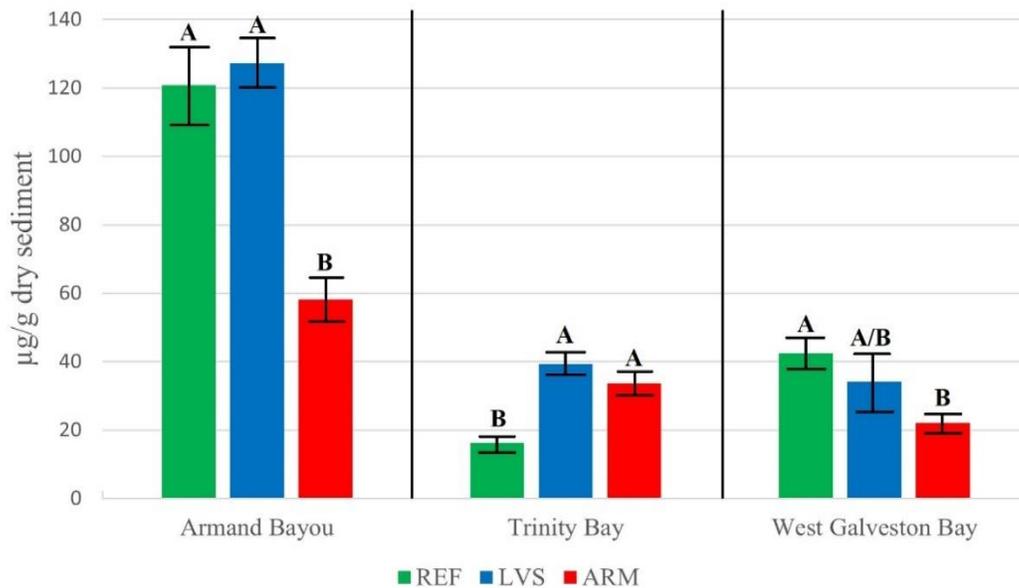


Figure 19. Combined concentrations of all measured heavy metals for each shoreline. REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline. Error bars indicate significant differences detected for that location, shorelines with different letter groupings are significantly different ($p \leq 0.05$).

Armand Bayou

For Armand Bayou sediments there was no significant differences amongst shoreline types for both cadmium and lead, shown in Figure 20. Zinc and nickel concentrations in Armand Bayou sediments were significantly higher at both LVS and REF than at ARM. Copper and chromium concentrations were significantly higher at LVS than both REF and ARM, and REF sites were also significantly higher than ARM sites. The concentration of copper at the LVS site was the only contaminant to exceed the Long and Morgan Effects Range Low and only exceeded the threshold by 0.8 µg/g dry sediment (Long and Morgan, 1990; Long et al., 1995). Results from a regression analysis of copper sediment concentrations compared to benthic taxa abundance did not indicate a relationship between the factors.

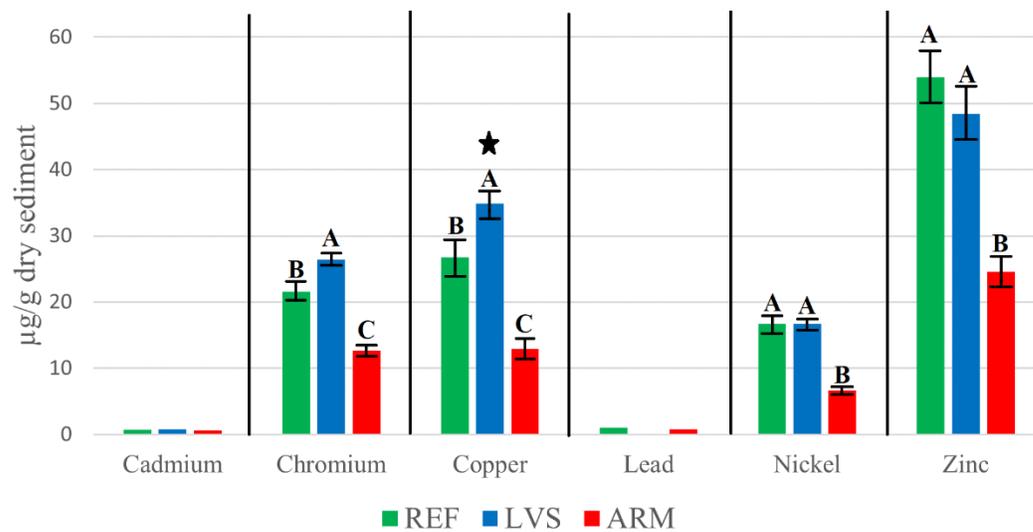


Figure 20. Armand Bayou sediment heavy metal concentrations, REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline. Error bars indicate significant differences detected between site treatments for each metal, shorelines with different letter groupings are significantly different ($p \leq 0.05$).

Trinity Bay

No significant difference amongst shoreline types was observed for lead concentrations at Trinity Bay, shown in Figure 21. Chromium and copper concentrations were significantly higher at LVS than both REF and ARM, ARM was also significantly higher than REF. Cadmium was significantly higher at both LVS and ARM than REF. Zinc concentrations were significantly higher at LVS than REF. Nickel concentrations were significantly higher at ARM than both LVS and REF, and LVS concentrations were significantly higher than REF.

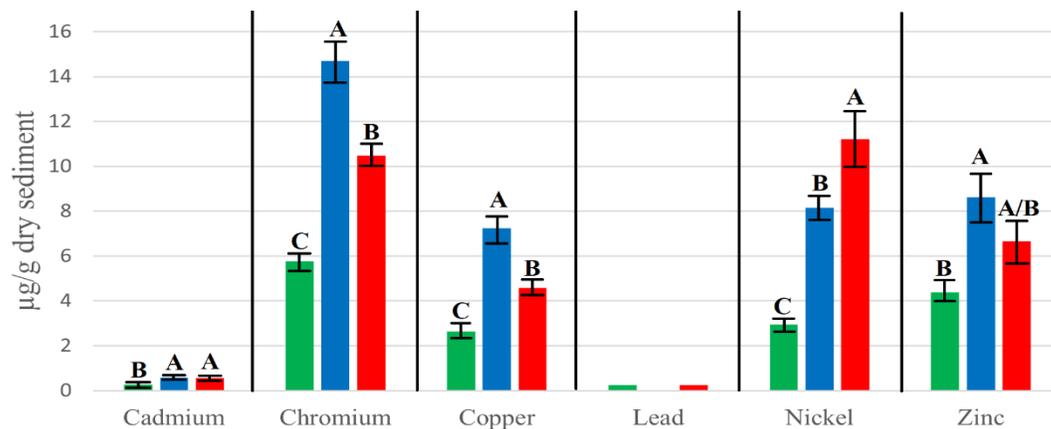


Figure 21. Trinity Bay sediment heavy metal concentrations, REF = Natural reference shoreline, LVS = living shoreline, ARM = Armored shoreline. Error bars indicate significant differences detected between site treatments for each metal, shorelines with different letter groupings are significantly different ($p \leq 0.05$).

West Galveston Bay

For West Galveston Bay, sediment concentrations of cadmium, shown in Figure 22, were not significantly different amongst shoreline types. Lead was found to be significantly higher at LVS than ARM; however, REF was not significantly different from LVS, or ARM. Concentrations of chromium, copper, and nickel were significantly higher at REF than ARM and LVS was not significantly different from REF or ARM.

Zinc concentrations were significantly higher at ARM than both REF and LVS.

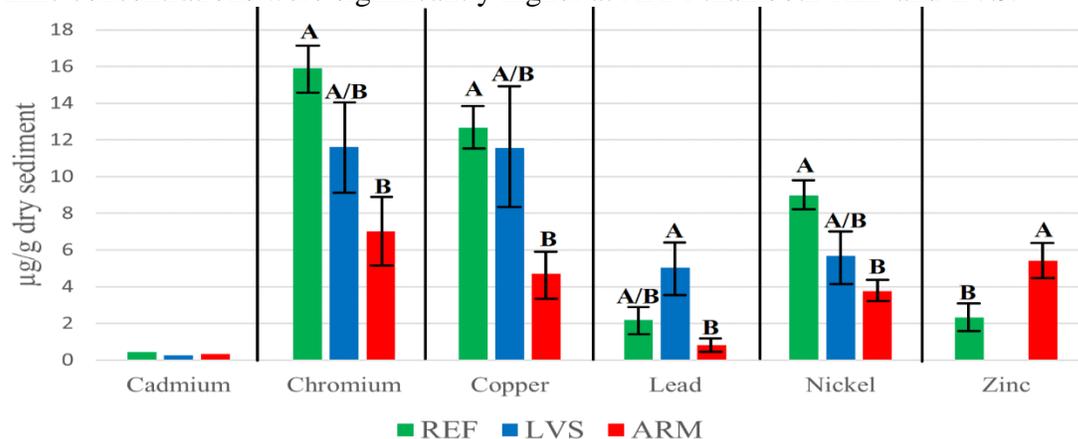


Figure 22. West Galveston Bay sediment heavy metal concentrations, REF = Natural reference shoreline, LVS = Living shoreline, ARM = Armored shoreline. Error bars indicate significant differences detected between site treatments for each metal, shorelines with different letter groupings are significantly different ($p \leq 0.05$).

CHAPTER IV:

DISCUSSION

This study aimed to evaluate the ecological function of three small-scale living shoreline sites in Galveston Bay. Galveston Bay, being the largest estuarine system along the Texas Gulf Coast, is large enough that there is great variability in both biotic and abiotic characteristics throughout the bay system. In consideration of this factor, it is important to assess the success of each living shoreline in relation to a similarly located natural shoreline before looking to identify any overarching trends among living shorelines across the Galveston Bay system. With this in mind, this study confirmed its hypothesis that these small-scale living shorelines in Galveston Bay are providing similar ecological function as natural reference shorelines and enhanced function over armored shorelines. Additionally, there is evidence from this study suggesting living shorelines may have the capacity to mitigate some of the stressors impacting natural shorelines in the Galveston Bay system.

Site Evaluations

Armand Bayou

For Armand Bayou, plant community metrics were lower than values observed at the natural reference site, however they were not significantly different. This finding may be indicative of ongoing successional development of the plant community at the living shoreline. While stem density and root biomass at the living shoreline were lower than the natural reference site, they may still be offering ecological function that is not offered by armored shorelines. Additionally, abundance and taxa richness for both nekton and benthic taxa were higher at REF and LVS than ARM for Armand Bayou, suggesting that both the living shoreline and the natural reference site are providing superior habitat than the armored site. Similarly, sediment concentrations of combined heavy metals was

higher at the living shoreline and natural reference shoreline than the armored shoreline, suggesting that both are providing improved contaminant sequestration services over the armored shoreline. Given that similar plant communities, habitat usage, and contaminant sequestration was observed between the living shoreline and the natural reference shoreline and that these values were higher than observed at the armored shoreline, it is evident that the Armand Bayou living shoreline is functioning similar to the natural shoreline and provides improved ecological function over the armored shoreline.

Trinity Bay

The Trinity Bay living shoreline had a more robust plant community (higher stem density, percent cover, productivity) than the natural reference shoreline, suggesting improved ecological function over not only the armored site but also the natural reference site. Further, nekton and benthic abundance was much higher at the living shoreline site compared to the armored site, suggesting it offers improved habitat. Total sediment heavy metal concentrations were similar between the living shoreline and armored site, although both chromium and copper concentrations were significantly higher at LVS than ARM, and only nickel was significantly higher at ARM than LVS. The natural reference site had much lower sediment heavy metal concentrations than either LVS or ARM, once again showing lower ecological function than the living shoreline. The living shoreline site at Trinity Bay showed signs of improved ecological function over both the REF site and the ARM site. Based on plant percent cover, nekton abundance, and sediment heavy metal concentrations, it is evident the natural reference site is experiencing decreased ecological function. Whereas the Trinity Bay living shoreline's breakwater may be providing more protection to plant communities from wave-induced erosional forces, allowing them to establish at low elevations. Of note, both benthic taxa diversity and evenness at the Trinity Bay living shoreline were lower than either the natural reference

or armored site due to the majority of individuals found being nematodes. While abundance and richness values were on par with the reference site, the domination by opportunistic taxa in the wake of construction has been observed in other living shoreline projects (Bilkovic and Mitchell 2013). Benthic communities of restored wetlands have been shown to develop over longer timespans than other aspects such as plant communities (Broome et al. 2019, Craft et al. 1999). This timeline of benthic community shifts may also apply to the high abundance and diversity observed at the natural reference site, despite plant, nekton, and heavy metal data suggesting reduced ecological function.

West Galveston Bay

The West Galveston Bay living shoreline was similar to the natural reference shoreline in terms of plant diversity but was higher in all other plant community metrics (density, cover, productivity). In terms of habitat usage, the living shoreline was much more similar to the reference shoreline than the armored. Both the living and reference shorelines had higher benthic abundance and taxa richness than the armored shoreline. Additionally, catch of nekton organisms at the living shoreline more closely resembled catch at the reference site, while ARM nekton catch was much lower. Sediment heavy metal concentrations at West Galveston Bay were higher for the LVS than the ARM for 4 out of 6 selected metals. While not as high as the concentrations found at the reference site, which were significantly higher than the armored site, the living shoreline is sequestering metals into sediments at higher rate than the armored site. In conjunction, a more robust plant community, along with similar habitat usage and contaminant sequestration indicate the West Galveston Bay living shoreline is providing similar if not improved ecological function as the natural reference site and improved function over the armored site.

Limitations and Future Studies

While this study does provide evidence that these living shoreline sites provide similar ecological function as their natural counterparts and improved over armored shorelines, the scope and scale of field efforts restrict the applicability of these findings from far reaching conclusions about the success of living shorelines in the Galveston Bay complex and elsewhere. This study was designed to examine small scale living shorelines, and by the nature of that pursuit, sample size is limited across numerous metrics. Transect length is constricted by the nature of Texas shoreline which consists of typically small privately owned parcels. Number of transects is constricted by the nature of shoreline sampling in which fringe wetland habitat may only be a few meters wide. Similarly, sample size substantially limited nekton data. Due to the transient and mobile nature of many nekton taxa, reliable nekton data must either be collected in large quantities provided by methods such as seining/trawling or be sampled more frequently, both of which present their own logistical issues and financial cost. While these limitations may contribute to a low confidence in these aspects of this study, other areas such as benthic macroinvertebrate and sediment analysis provide more stable measures of ecological function. Benthic macroinvertebrate abundance was found to be significantly higher at living shorelines than armored shorelines across all locations, a finding that contributes greatly to a subject of which only three studies have examined. While certain methodologies such as those used for nekton data may not have produced sufficient sample size for meaningful statistical analysis, that finding in itself may be beneficial to the living shoreline research community where methods for evaluating restoration success have yet to be widely established.

Perhaps the most important direction that future research of living shorelines in Galveston Bay can take is continued monitoring of these same sites. While it would be

beneficial to expand site location, this study provides a foundation for continued monitoring of these sites and may provide often sought answers about the timeline in which benefits provided from these projects are realized. Given that two of the projects in this study showed signs of improved ecological function over the natural sites, measuring the particular stressors experienced by some of the reference sites and linking that to functional success data may bolster the claim that living shorelines not only offer similar functions as natural shorelines but can also be more resilient. Investigating other characteristics of sediment such as organic carbon content, grain size, or texture in relation to the development of different biological components (i.e. plant or benthic communities) would be useful in understanding how physical factors are impacting living shoreline development.

Conclusion

This study is one of the first to examine the ecological function of small-scale living shoreline sites along the Texas coast. The finding that living shorelines function similar to natural shorelines and superior to armored shorelines in terms of plant communities and habitat usage, reinforces previous research from other regions. By examining sediment heavy metals, this study provided the first piece of the puzzle in understanding the dynamics of contaminants in living shorelines. Examining the ecological function of these restoration sites across multiple metrics is valuable for improving restoration methods and also providing credibility to the fruitfulness of restoration efforts. Living shoreline projects are becoming a focus of restoration efforts and receiving millions of dollars in funding. In order to see that ecological benefits from those investments are maximized, continued research into the function of living shorelines is essential.

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