Mangrove Restoration in Galveston Bay: Ecological Benefits and Effective Restoration Techniques

Final Report

Prepared by:
Department of Marine Biology
Texas A&M University at Galveston (TAMUG)

Principal Investigator (PI):
Anna R. Armitage

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Questions concerning this report should be directed to:

Dr. Anna R. Armitage
Associate Professor
Texas A&M University at Galveston
200 Seawolf Parkway
Galveston, Texas 77553
(409) 740-4842
armitaga@tamug.edu

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Executive Summary

Landscape-level shifts in plant species distribution and abundance can subsequently alter ecosystem structure and function. Such shifts are occurring on the Texas Gulf Coast, where in recent years, mangroves have been encroaching into some areas occupied by salt marsh plants. To assess whether this change in the dominant plant community should alter restoration practices, a series of surveys was conducted at restored and reference sites with and without mangroves within Galveston Bay, Texas. Plants, soil, and fauna were surveyed at 12 sites in spring 2016, fall 2016, and spring 2017.

Study objective 1: Compare mangrove planting techniques. Existing and restored mangrove stands were surveyed around Galveston Bay, and compared to existing and restored salt marshes (primarily *Spartina alterniflora*). The study sites included relatively established stands (sites with mangroves that were not actively planted), and newly established areas (sites with mangroves that were planted as part of habitat restoration projects). A variety of plant health metrics for ecosystem functions were measured at all study sites. Surveys of plant cover and species composition were conducted within predefined elevation ranges, based on field-tested, standard protocols for monitoring wetland plant communities. Within restored wetlands, the survival and vitality of individual trees was measured at various stages of development.

Study objective 2: Quantify the ecological benefits of mangrove restoration in Galveston Bay. This project objective sought to define which ecological functions and ecosystem services are most likely to be enhanced by mangroves, and which would benefit more from marsh restoration. This information will allow coastal industries such as fisheries and tourism to be adaptively managed as part of a comprehensive, ecosystem-level Watershed Management Plan. The data generated will help practitioners decide how, and where, to balance the benefits of mangrove planting in planning future restoration projects. At the sites described above, a variety of ecosystem functions were measured. Small soil cores and plant tissue samples for nutrient (nitrogen and phosphorus) analysis were collected and analyzed. To assess the contributions of mangrove and marsh vegetation to estuarine food webs, a small number of common herbivorous and carnivorous nektonic species were collected and their stable isotope ($\delta^{15}N$, $\delta^{13}C$) concentrations were measured. To assess the benefits of mangrove restoration for birds, wildlife cameras were installed at a subset of sites.

This report includes subsections that encompass the Data Collection & Analysis Report, Mangrove Planting Techniques Report, and Ecological Benefits of Mangroves Report.

Restoration practice

In general, few of the planted mangroves at restoration sites were thriving. Most of the planted shrubs remained small (less than 1 m) for several years after planting, though many were reproductive. The healthiest mangroves were usually at relatively high elevations, near the upper edge of the *Spartina alterniflora* zone. Mangroves are actively recruiting to this high elevation zone at numerous locations around Galveston Bay, independent of planting efforts. Based on these observations, the recommendation for restoration practice is to focus planting efforts on fast growing species such as *Spartina alterniflora* and allow natural recruitment of mangroves to occur gradually over time.

Ecological benefits of mangroves

Marsh plant diversity was lower at sites with high mangrove cover. Within marsh or mangrove stands, fish and invertebrate densities were generally similar, though the species composition differed. Stable isotope analysis indicates that marsh vegetation is more important than mangroves in supporting coastal wetland food webs. At low densities, mangroves did not substantially alter wading bird or shorebird abundances. Mangroves generally increased carbon retention in the soil. Overall, our results revealed that salt marshes and mangroves support different plant and animal assemblages, and that mangrove encroachment is likely to have complex influences on ecosystem processes.

Introduction

The overall objective of this project was to determine if, when, and where mangrove restoration should be implemented in Galveston Bay. Most previous mangrove restoration research has been based in tropical regions (e.g., Indonesia, Caribbean), where mangrove forests are speciose, dense, and never exposed to freezing temperatures. In contrast, mangroves in Texas and the rest of the northern Gulf of Mexico are species poor, form patchy stands of dwarf trees, grow in arid, conditions, and are occasionally exposed to freezing temperatures. Within the Gulf of Mexico, some mangrove restoration research has been performed in Louisiana, where mangroves are more widely established than in Texas. Texas and Louisiana differ substantially in terms of hydrology, geomorphology, climate, rainfall, and salinity. Therefore, the limited body of available mangrove restoration work in the Gulf of Mexico is not directly applicable to Texas.

Some of the many questions about the ecological benefits of planting mangroves include: Will the taller stature of planted mangroves augment migratory bird roosting habitat and shorebird foraging habitat? Will mangroves be more – or less – resilient to fluctuations in sea level than marsh vegetation? Will mangroves reduce coastal erosion? Will mangroves increase – or decrease – fishery nursery value and trophic support for estuarine food webs? How will mangroves improve nonpoint source (NPS) mitigation by changing nutrient cycling and storage? In order to determine if, when, and where mangrove restoration should be implemented in Galveston Bay, quantitative answers to these questions are needed. Restoration practitioners will further benefit from quantitative studies to identify best practice mangrove restoration techniques, including optimal elevation ranges, propagule vs. seedling transplant, and single vs. clustered transplants.

By coordinating with the Texas Parks and Wildlife Department (TPWD) and other state, federal, and non-profit project partners, this project within Galveston Bay has the potential to be leveraged and applied at state and regional scales. At a local scale, the project outcomes will directly influence the practice of wetland restoration in Galveston Bay by providing concrete recommendations regarding if, when, where, and how to restore mangroves in Galveston Bay to local project partners and other restoration practitioners. At a state-level scale, this work converged with a large-scale manipulative and observational interdisciplinary study that is examining the ecological implications of mangrove expansion on the central Texas coast. Information from the current project on food webs and animal use of restored and established mangrove and marsh stands will contribute to broader questions about the ecological implications of mangrove expansion across the Texas coast. At the Gulf of Mexico scale, the data from this project will, at no additional cost, contribute to the development of a newly-initiated Gulf-wide Mangrove Migration Network. This United States Geological Survey (USGS) initiative seeks to monitor changes in mangrove and marsh distribution across the Gulf over the coming years.

This project will assist the state in implementing The Galveston Bay Plan over the next 10 years by (1) identifying effective mangrove restoration strategies, and (2) providing quantitative information on the ecological benefits of mangrove planting in various locations throughout the Bay. These data will help ensure effective and efficient resource management by collecting scientific data to inform best-practices for mangrove restoration, and provide recommendations as to if and when restoration efforts should focus on mangrove plantings.

The project will provide scientific information that will be used to protect, sustain, and restore the health of critical natural habitats and ecosystems, specifically the following hydrologic unit codes (HUC) and Segment IDs in the Galveston Bay Watershed (Texas) (see also Appendix A):

- West Bay; Segment 2424, HUC 12040204
- Lower Galveston Bay; Segment 2439, HUC 12040204

Project Significance and Background

Coastal wetlands provide many ecosystem services, including vital trophic support for iconic wildlife and for fishery species that use estuaries as spawning grounds or nursery habitat, and the mitigation of NPS pollution. However, these habitats are being lost at an alarming rate, particularly on the Gulf Coast of the United States. In response, coastal management commonly includes restoration projects that will augment the NPS mitigation performance and other ecological functions of coastal wetlands. This question is particularly topical in coastal wetland ecosystems of Texas, where restoration is a high priority management strategy, yet the foundation plant community in the coastal wetland landscape is undergoing a state change (Guo et al. 2013). Along the Gulf of Mexico coastline, tidal wetlands comprise a mosaic of habitat types, including salt marshes interspersed with mangroves and salt flats. Black mangroves (*Avicennia germinans*) are becoming increasingly common throughout the region (Figure 1; Montagna et al. 2007, Armitage et al. 2015). These shifts are echoed across the Gulf of Mexico coastline and may continue to progress in response to environmental drivers including warm winter temperatures and sea level rise (Krauss et al. 2011, Osland et al. 2013).

Figure 1. Black mangroves colonizing a salt marsh in Galveston Bay, Texas.

To date, most restoration projects on the Texas Gulf Coast have focused on low elevation marsh habitat (e.g., smooth cordgrass *Spartina alterniflora*), which can substantially augment the production of some fisheries species (Rozas et al. 2005). However, given the regime shift from herbaceous marshes to woody mangrove swamps across the region, black mangroves (*Avicennia germinans*) are more frequently occurring in restoration projects, whether planted purposefully or through natural recruitment. In addition, coastal managers in Galveston Bay have recently increased the use of black mangroves (*Avicennia germinans*) in restoration projects. Therefore, there is increasing interest in understanding whether the presence of black mangroves in restoration projects will improve coastal wetland functions and increase restoration success.

Despite the increased frequency of mangrove planting in the Bay, many questions about this practice remain. With regards to NPS mitigation, how will mangroves change nutrient cycling and storage? Will the taller stature of planted mangroves augment migratory bird roosting habitat and shorebird foraging habitat? Will mangroves increase – or decrease – fishery nursery value and trophic support for estuarine food webs? This is a particularly pertinent question given the number of living marine resources on the Texas coast – including commercial and recreational fishery species such as red drum (*Sciaenops ocellatus*) and brown shrimp (*Farfantepenaeus aztecus*), and iconic wildlife such as shorebirds – depend directly on coastal wetlands for food and habitat (e.g., Engle 2011). In order to determine if, when, and where mangrove restoration should be implemented in Galveston Bay, quantitative answers to these questions are needed. Furthermore, restoration practitioners will benefit from quantitative studies to identify best practice mangrove restoration techniques, including optimal elevation ranges, propagule vs. seedling transplant, and single vs. clustered transplants.

The objectives of this project were:

- *Objective 1:* To identify mangrove planting techniques that yield near- and long-term mangrove restoration success at sites throughout Galveston Bay.
- *Objective 2:* To compare specific ecosystem functions between existing mangrove stands and salt marshes in Galveston Bay.

Methods

Study sites

Twelve study sites in Galveston Bay were selected, including established and restored black mangrove (*Avicennia germinans*) stands, and mature and restored salt marshes (primarily comprised of smooth cordgrass, *Spartina alterniflora*) (Table 1). The study sites with mature stands of vegetation were dominated by mangrove and/or marsh species that were not actively planted. Restored areas were primarily comprised of mangrove and/or marsh vegetation that was planted as part of habitat restoration projects (see maps in Appendix A).

Table 1. Sites were accessible by foot from public roadways or by kayak. Sites are listed below from west to east.

Site name	Site type	Coordinates
Terra Mar Restoration	Restored mangrove	N 29.134477°, W 95.070083°
Terra Mar Reference	Reference marsh	N 29.133948°, W 95.067884°
Isla del Sol	Restored mangrove	N 29.141061°, W 95.057531°
Sunset Cove	Reference marsh	N 29.151475°, W 95.037469°
McAllis Point	Restored marsh	N 29.177585°, W 95.013759°
Indian Beach	Reference marsh	N 29.174083°, W 95.008659°
Dalehite marsh	Restored marsh	N 29.223735°, W 94.943473°
Dalehite mangrove	Reference mangrove	N 29.226033°, W 94.942221°
Reitan	Restored marsh	N 29.316250°, W 94.918066°
Sweetwater	Restored mangrove	N 29.271067°, W 94.883045°
East End	Reference mangrove	N 29.331085°, W 94.753875°
Bolivar	Reference mangrove	N 29.376481°, W 94.733716°

At each site, four 100-m^2 plots were established (Figure 2). Plots targeted the representative mangrove and/or marsh vegetation within the area. The plots were circular with radii of 5.65 m. A temperature logger was installed at the center of each plot.

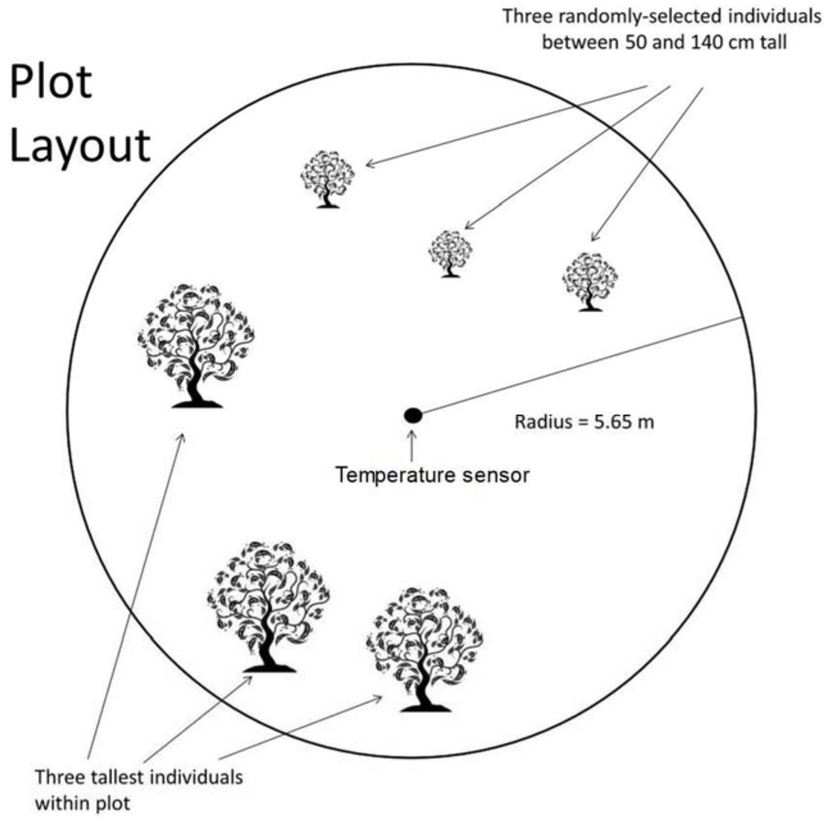

Figure 2. Graphic of study plot layout.

Sampling schedule

Sampling was conducted in spring 2016 and fall 2016, coinciding with the beginning and peak growing season periods, respectively. A subset of measurements was also taken in spring 2017 to measure plant fitness after the winter senescence season.

Data Collection and Analysis

<u>Plant community composition and structure measurements</u>

Within each 100-m^2 plot, the percent cover of mangrove and marsh species was estimated for the entire plot at three strata: (1) less than 1.4 m elevation from the soil surface; (2) height greater than or equal to 1.4 m elevation from the soil surface; and (3) all vegetation strata (i.e., from the soil surface to the tallest canopy). The 1.4 m elevation was derived from protocols used by project partners at the USGS Wetland and Aquatic Research Center to monitor mangroves around the Gulf of Mexico, and is based on the premise that freeze events will often differentially affect tall and short mangroves; a height of 1.4 m provides a good point for quantifying these structural development-specific effects.

In addition to the percent cover estimates, a total of up to six mangrove plants were tagged at the start of the study period within each 100-m^2 plot for long-term monitoring. The tagged mangrove plants consist of the three tallest individuals in the plot and three randomly selected individuals that are between 50 and 140 cm tall (Figure 2). At some of the mangrove sites, fewer than six mangrove individuals were present in the study plots; in those cases, all mangroves in the plot were tagged regardless of their height. The total number of tagged mangroves at each reference mangrove site (see Table 1 for a list of sites) was: Bolivar: 24; Dalehite mangrove: 22; East End: 24. The total number of tagged mangroves at each restored mangrove site was: Isla del Sol: 6; Sweetwater: 17; Terra Mar restored: 9. For each of the three sampling events (Spring 2016, Fall 2016, Spring 2017), the following measurements were recorded on each tagged mangrove plant: (1) plant height; (2) crown diameter 1 (CD1; the largest diameter); and (3) crown diameter 2 (CD2; the largest diameter perpendicular to CD1). All of these measurements were used to quantify mangrove performance and response to extreme winter temperature events.

Temperature measurements

Black mangrove mortality may occur following freezing events where temperatures are lower than -4°C for several consecutive hours (Cavanaugh et al. 2015). In dense stands, mangroves may insulate each other from such freeze stress. To assess the potential insular properties of mangroves, air and soil temperature sensors were placed in a subset of plots for the duration of the contract period. However, there were no freeze events of sufficient severity or duration to cause mangrove mortality during the contract period. Therefore, temperature was not a relevant abiotic factor during the contract period and is not reported in detail.

Plant and soil carbon, nitrogen, and phosphorus (CNP)

In spring 2016, three soil samples were collected from each plot, 50 cm from the base of each marked tall tree (if present). Soil cores were 5 cm in diameter and 15 cm deep. Soils were dried at 60°C and homogenized prior to nutrient analyses. Two new (apical) leaves were collected from each of three *Avicennia* trees (if present) and three stems of *Spartina alterniflora*. Leaves were rinsed to remove salts, dried at 60°C and homogenized prior to nutrient analyses.

Investigators measured the total carbon (C) content of the soils nitrogen (N), and phosphorus (P) content of plant tissue. Carbon and nitrogen contents were determined using a CHN analyzer (Perkin-Elmer 2400 CHN Analyzer), which reports nutrients as a percent of dry weight. Phosphorus contents were determined by a dry-oxidation, acid hydrolysis extraction followed by a colorimetric analysis of phosphate concentration of the extract (Fourqurean et al. 1992).

Plant and animal isotopes

Basal resources that support food webs is often determined by measuring stable isotopic ratios ($\delta^{15}N$, $\delta^{13}C$) in plant and animal tissue. Stable isotopic ratios in plants vary among species based on productivity, metabolism, and differences in fractionation rates (Peterson and Fry 1987). Mangroves and marsh plants incorporate these isotopic signatures over time. Stable isotopes reveal information about food web relationships because the isotopic ratios of animals reflect their food sources (Peterson and Fry 1987, Armitage and Fourqurean 2009). Specifically, $\delta^{13}C$ in consumers is very similar to their food sources, and $\delta^{15}N$ is slightly but predictably heavier than their food (Peterson and Fry 1987).

This project compared marsh and mangrove subsidies to wetland food webs by measuring the stable isotope ratios ($\delta^{15}N$, $\delta^{13}C$) of herbivorous and omnivorous species that are ubiquitous in Galveston Bay. Consumers collected were primarily grass shrimp (*Palaemonetes* spp.), penaeid shrimp (*Litopenaeus setiferus*, *Farfatepenaeus aztecus*), and blue crabs (*Callinectes sapidus*). Specimens were collected in spring and fall 2016, corresponding with peak seasonal abundance and spawning periods (Pullen 1963, Scharf and Schlicht 2000, Ganz and Knowlton 2002). Most fauna were collected with a 6-m bag seine (6 mm mesh) that was pulled along three replicate 20-m transects parallel to the shoreline at each study site. Bag seine collections were supplemented with dip nets to obtain adequate tissue for analysis. All past and future collections are permitted under the TPWD Scientific Research Permit No. SPR-0708-303 (exp. 7/2017). Although no vertebrates were targeted in this study, project personnel followed the Texas A&M University Animal Use Protocol 2015-0114 (exp. 5/2018) by euthanizing specimens over an ice slurry and then freezing them until further analysis in the lab. To prepare frozen animal samples for isotopic analyses, muscle tissue was removed when practical. For crabs, shells were discarded and soft tissue was retained. Brown and white shrimp were bisected behind the thorax, the shell was removed, and only the posterior muscle section was retained for analysis. For grass shrimp, bisection was not practical due to their small size; therefore, entire specimens were retained for analysis. To reduce potential analysis bias introduced by variable fat content among animal species (Post et al. 2007), lipids were extracted from all animal samples with an Accelerated Solvent Extractor (Dionex) prior to isotope analysis.

For isotope analyses, leaf samples were collected in spring and summer 2016 in addition to those described above for nutrient analyses. Two new (apical) leaves were collected from each of three *Avicennia* trees and three stems of *Spartina alterniflora*. Leaves were rinsed to remove salts, dried at 60°C and ground and homogenized prior to stable isotope analyses.

All isotopic measurements were performed with standard elemental analyzer isotope ratio mass spectrometer procedures at the Stable Isotope Facility at University of California Davis (http://stableisotopefacility.ucdavis.edu/13cand15n.html). The samples' isotopic ratios (R) were reported in the standard delta notation: δ (‰) = $[(R_{sample}/R_{standard})-1]\times 1{,}000$‰. These results were reported relative to the international standards of atmospheric nitrogen and Vienna Pee Dee belemnite for carbon using the secondary standards International Atomic Energy Agency (IAEA) N-3 for $\delta^{15}N$ and IAEA CH-6 for $\delta^{13}C$. Comparing the $\delta^{13}C$ and $\delta^{15}N$ signatures between primary producers and consumers helped to determine whether the food web at each site was based on autochthonous (from within the site) marsh or mangrove vegetation, or allochthonous (from another site; e.g., marine or algal subsidies) vegetation.

Bird camera deployment

To assess the benefits of mangrove restoration for birds, wildlife cameras (owned by the Armitage lab) were deployed at a subset of sites. Cameras were trained on individual mangrove trees in order to capture roosting behavior by migratory passerines and other birds, and on the marsh-bay interface in order to capture the foraging behavior of estuarine-dependent bird species. In order to minimize the risk of equipment loss and/or tampering, cameras were deployed at relatively isolated locations or on private property (Sweetwater and East End, see Table 1).

Cameras were deployed for two-week periods, with photos taken every 30 minutes from sunrise until sunset, with additional motion-activated photos. The photos were downloaded to a computer and examined to identify and quantify any birds present.

Mangrove Planting Techniques

To address objective 1 and identify mangrove planting techniques that yield near- and long-term mangrove restoration success at sites throughout Galveston Bay, analyses focused on comparisons between mature and restored mangrove stands (Table 1). Data collection and analyses were performed as described above.

Ecological Benefits of Mangrove Restoration

To address objective 2 and identify the ecological benefits of mangrove restoration, analyses focused on comparisons between mature mangrove and marsh stands, and between restored mangrove and marsh stands (Table 1). Data collection and analyses were performed as described above.

Results and observations

Plant cover

Marsh plant cover was significantly and inversely related to mangrove cover (linear regression p < 0.001; Figure 3). There was relatively high variability in marsh cover, especially at sites with low mangrove cover ($r^2 = 0.227$), but in all plots with more than 40% mangrove cover, there was less than 30% marsh cover. Total wetland plant cover in plots at sites with high mangrove cover was comparable to the marsh-only plots (Figure 4). Since total plant cover was not changed by mangroves, this suggests that mangroves are displacing and replacing marsh vegetation.

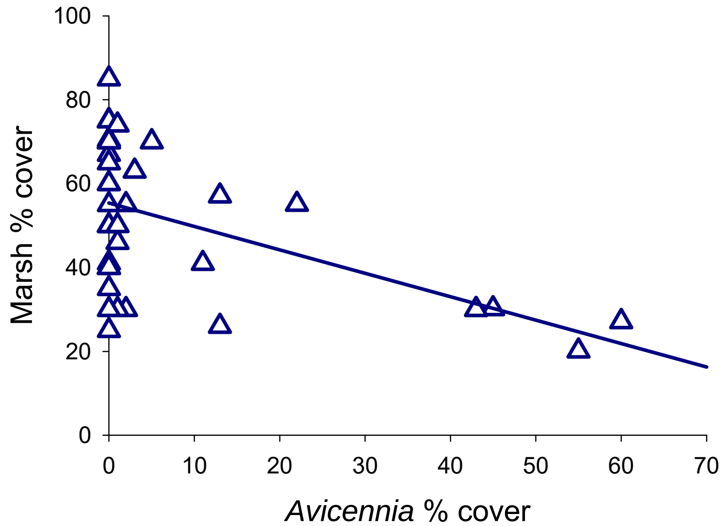

Figure 3. Relationship between total cover of all marsh plants and black mangrove cover in Galveston Bay.

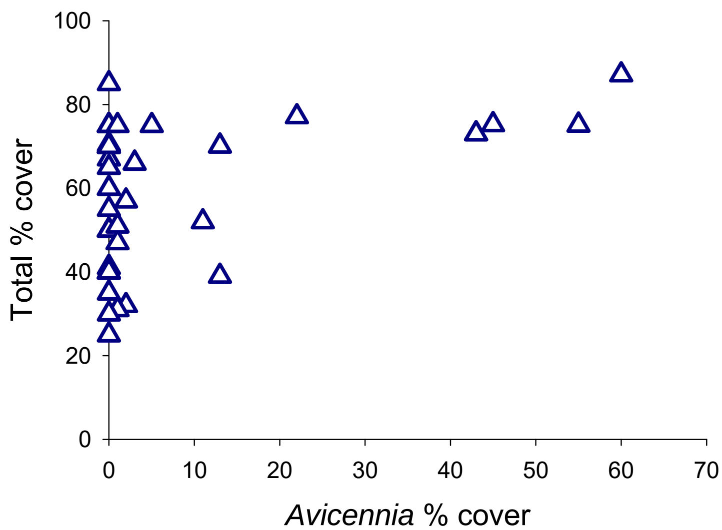

Figure 4. Total wetland plant cover (marsh + mangrove) relative to black mangrove cover in Galveston Bay.

Plant species richness

Marsh plant species richness significantly declined as mangrove cover increased (linear regression $p = 0.032$, $r^2 = 0.170$), but this pattern only emerged at the site-level when sites with higher mangrove cover in the Port Aransas region were included (Figure 5, Appendix A). Above a threshold of about 60% mangrove cover, no more than two species of marsh plants co-occurred with the mangroves in the intertidal zone.

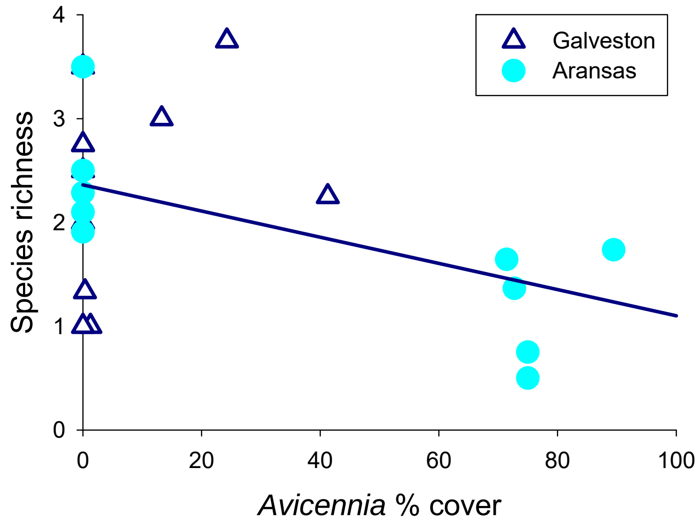

Figure 5. Relationship between marsh species richness and mangrove cover at sites in Galveston and Aransas Bays, Texas.

Planted mangroves

The height of tagged individual mangroves did not change substantially over time in either reference (Figure 6) or restored (Figure 7) sites. This lack of change in height over time was expected given the relatively short time period of the study (one full growing season). It was notable, however, that the mangroves planted in the restored site did not increase in height, despite their relatively young age. Further, all mangroves in the restored sites were less than about one meter in height, whereas mangroves in the reference sites were often twice that height. The persistence of a short mangrove canopy suggests that the mangroves in the restored sites were surviving (only one mortality was noted during the study period) but were not thriving.

The contrast between reference and restored mangrove sites was similarly pronounced for canopy area. There was no substantive change in canopy area over time at any site, but the areal ground coverage of individual tagged trees in restored sites was a fraction of that in reference sites (Figure 8, Figure 9).

Despite their small size, many of the mangroves in restored sites were reproductive at some point during the study period; over 70% of the tagged trees were observed to have flowers or propagules. It is unknown whether these small trees were producing viable propagules, if any of the propagules successfully germinated, or how far the released propagules disperse.

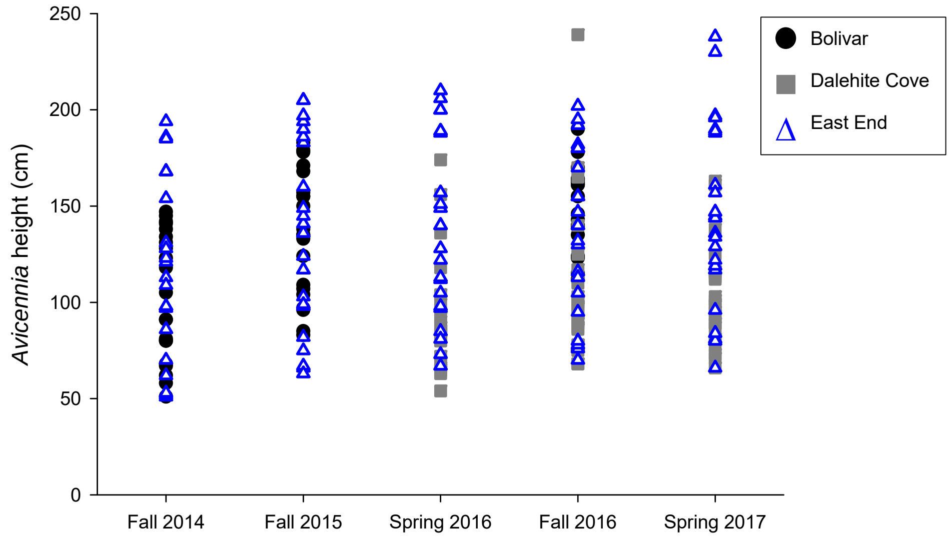

Figure 6. Height of tagged individual *Avicennia germinans* over time at reference sites with mature stands of mangroves. Measurements collected prior to the contract period (2014 and 2015) were taken in cooperation with project partner USGS as part of an unfunded collaboration.

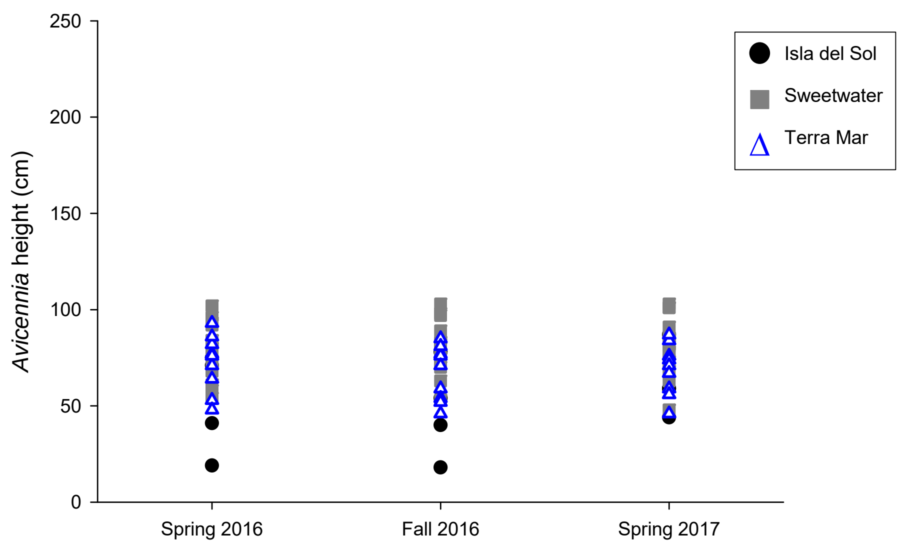

Figure 7. Height of tagged individual *Avicennia germinans* over time at restored sites with mangroves that were planted circa 2010.

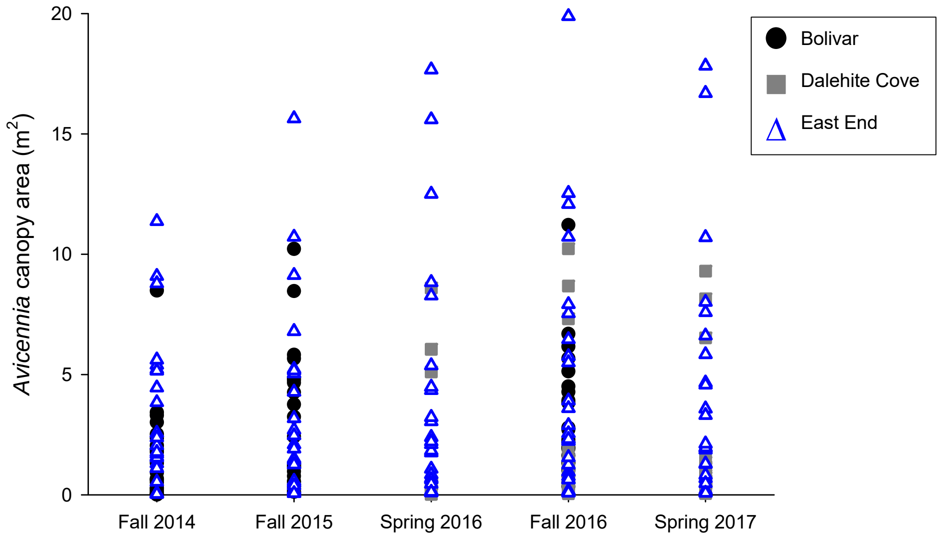

Figure 8. Canopy area of tagged individual *Avicennia germinans* over time at reference sites with mature stands of mangroves. Measurements collected prior to the contract period (2014 and 2015) were taken in cooperation with project partner USGS as part of an unfunded collaboration.

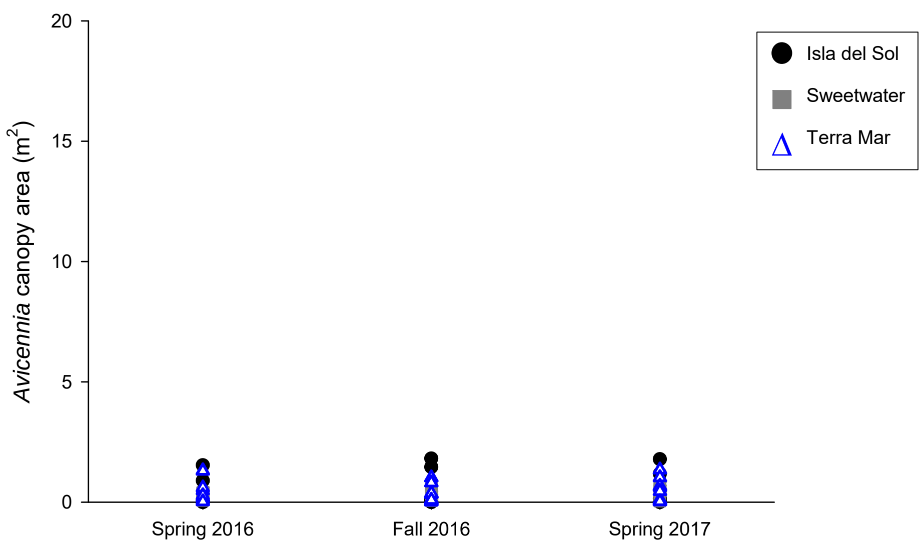

Figure 9. Canopy area of tagged individual *Avicennia germinans* over time at restored sites with mangroves that were planted circa 2010.

Plant CNP

The nutrient content of *S. alterniflora* leaves did not differ across sites; all carbon, nitrogen, and phosphorus values were similar regardless of restoration status or the presence of mangroves (Figure 10). Likewise, leaf nutrient content of *A. germinans* was consistent across sites, regardless of restoration status (Figure 11).

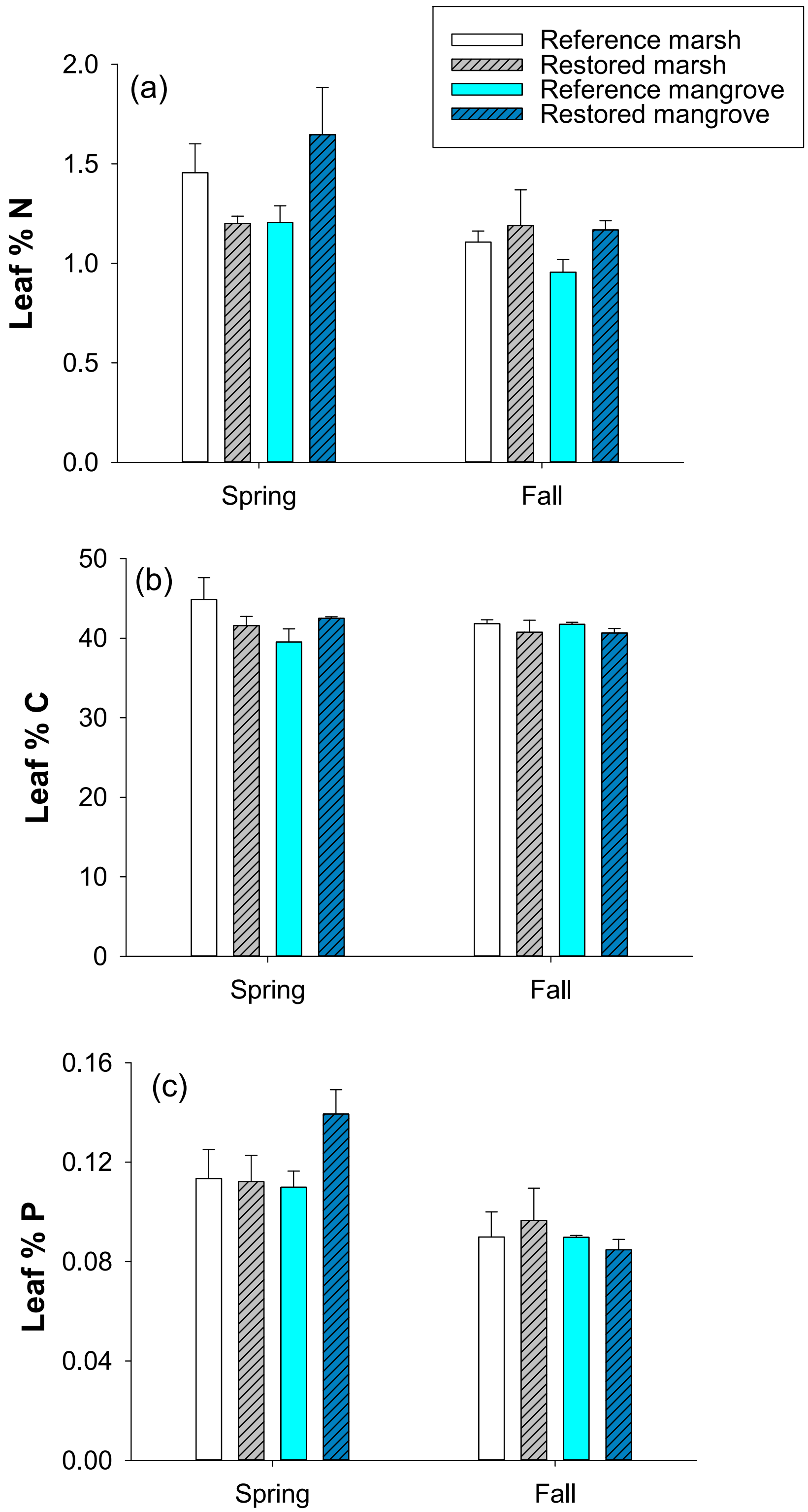

Figure 10. Nutrient content of *Spartina alterniflora* leaves at restored and reference sites with and without mangroves. (a) % nitrogen; (b) % carbon, (c) % phosphorus. Error bars represent standard error.

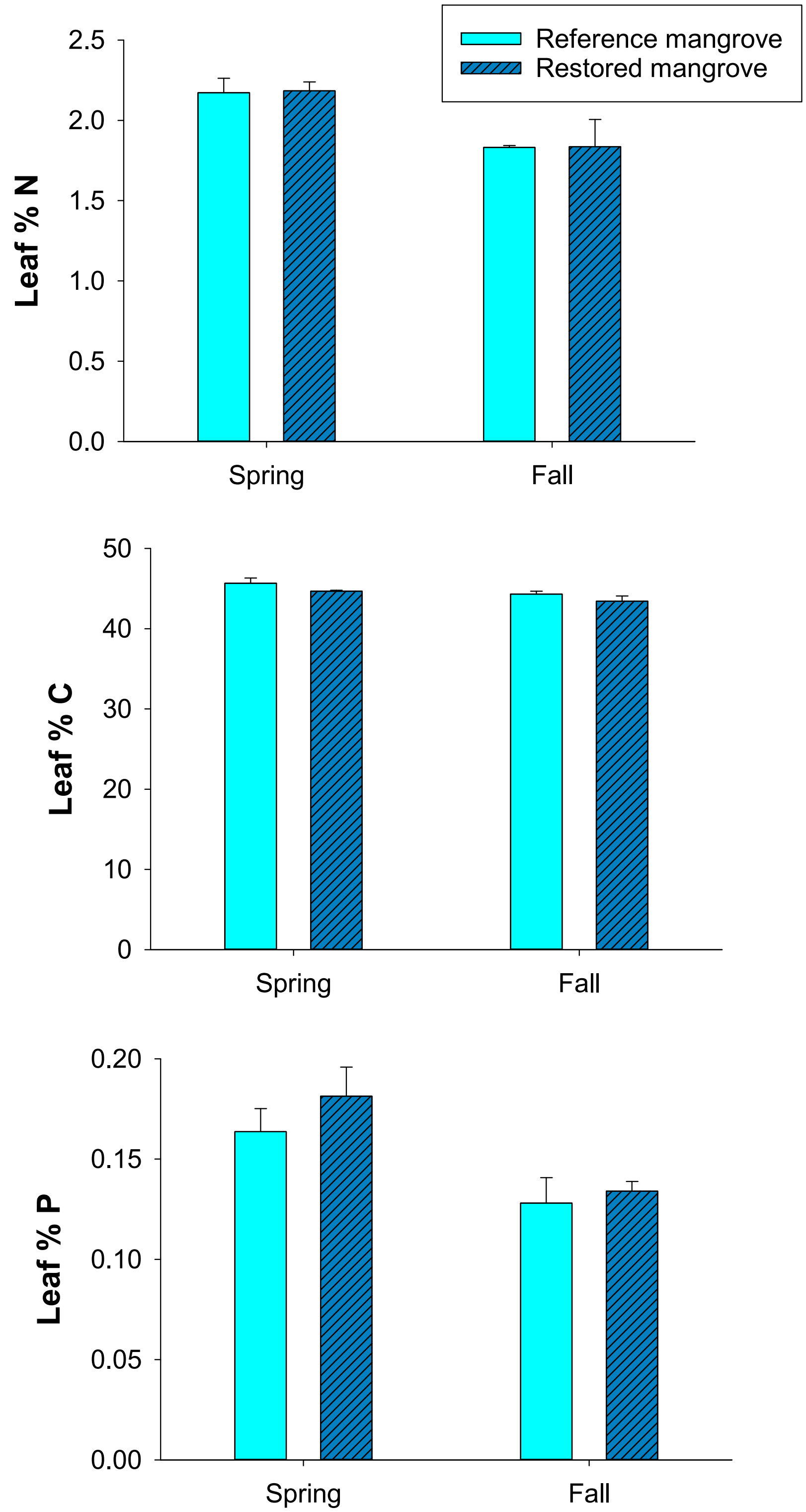

Figure 11. Nutrient content of *Avicennia germinans* leaves at restored and reference sites with and without mangroves. (a) % nitrogen; (b) % carbon, (c) % phosphorus. Error bars represent standard error.

Soil carbon

Total soil carbon was low at all sites, and were similar regardless of restoration status or the presence of mangroves (Figure 12). Although soil carbon appeared to be lower at sites with mangroves, there was high variability, and this difference was not significant (ANOVA, $p > 0.05$).

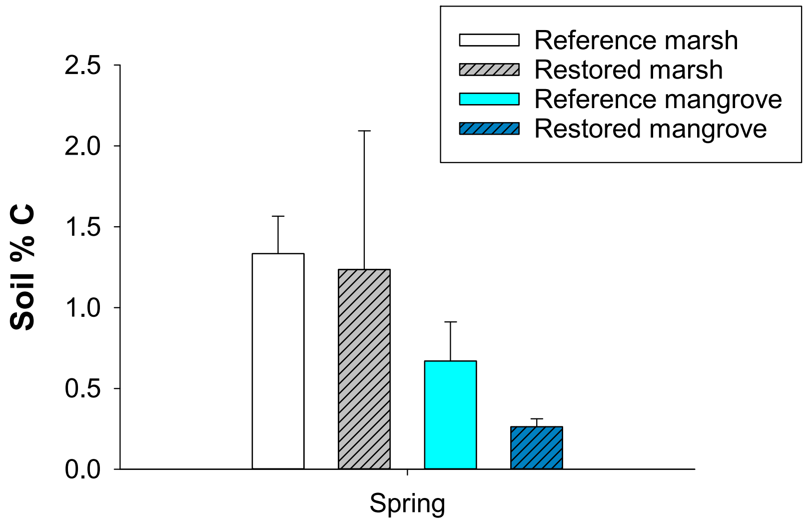

Figure 12. Total percent carbon in soils from restored and reference sites with and without mangroves. Error bars represent standard error.

Stable isotopes

Analysis of stable isotope signatures generally suggests that mangroves were not major contributors to the estuarine food web. In spring 2016, the consumers collected in the seine and dip nets included grass shrimp (*Palaemonetes vulgaris* and *P. pugio*) and penaeid shrimp (*Litopenaeus setiferus*, *Farfatepenaeus aztecus*). All of these consumers had $\delta^{13}C$ signatures that were more closely aligned with *S. alterniflora* signatures than with *A. germinans* signatures (Figure 13). Likewise, in fall 2016, all consumer signatures were more closely aligned with *S. alterniflora* signatures (Figure 14). Slightly different consumers were collected in spring and fall – blue crabs (*Callinectes sapidus*) were only found in the fall, and *P. vulgaris* were only found in the spring.

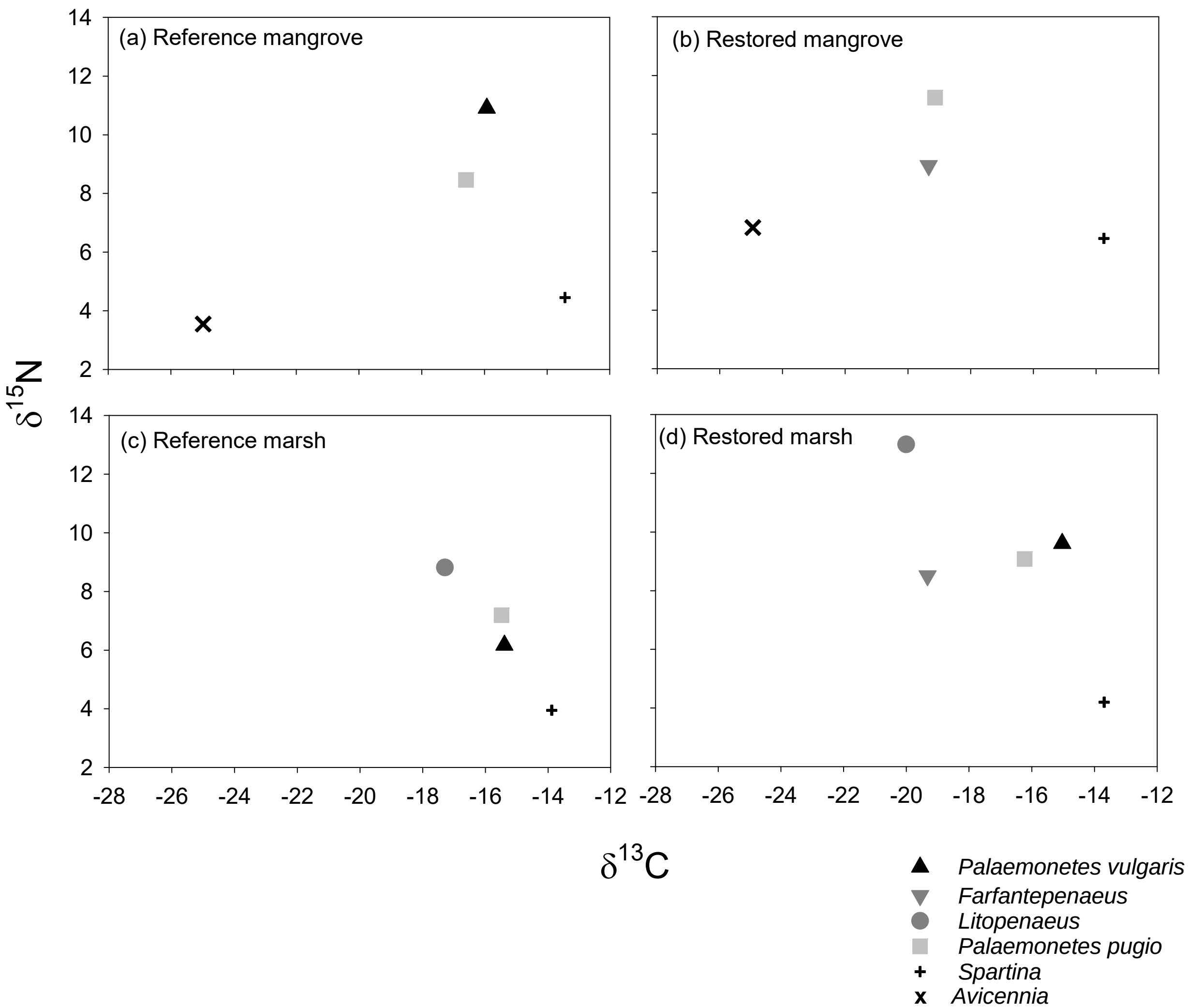

Figure 13. Stable isotope ratios of consumers relative to *Spartina alterniflora* and *Avicennia germinans* in spring 2016 in restored and reference wetlands in Galveston Bay, Texas.

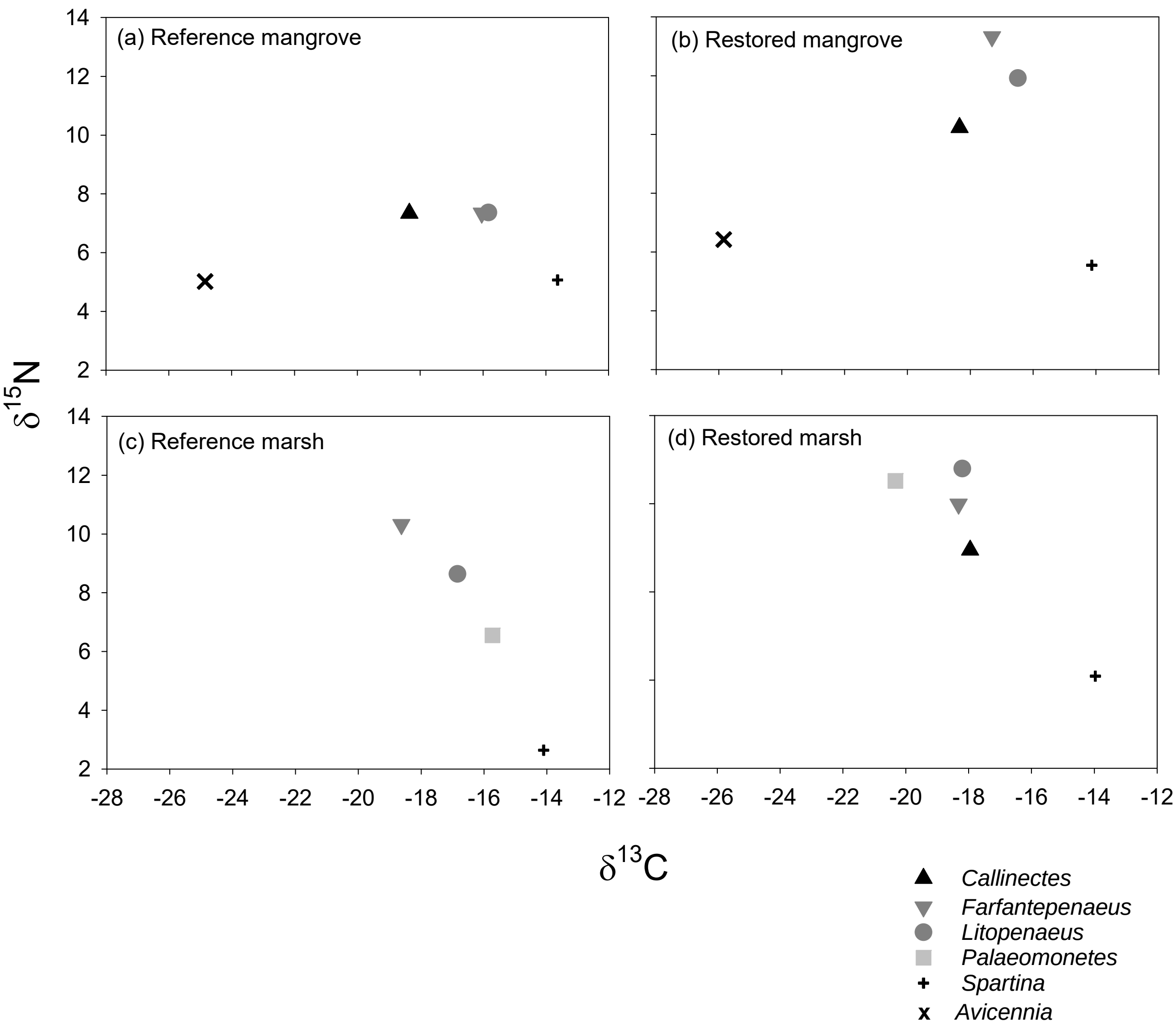

Figure 14. Stable isotope ratios of consumers relative to *Spartina alterniflora* and *Avicennia germinans* in fall 2016 in restored and reference wetlands in Galveston Bay, Texas.

Birds

In general, there were very few birds detected in the game camera images. Qualitatively, wading birds were observed to be using all sites, regardless of mangrove presence (Figure 15). Furthermore, when wading birds were observed foraging, they appeared to be exclusively foraging in marsh vegetation or at marsh edges. No passerines or shorebirds were observed in the game camera images, though it was likely that they were present but too small to trigger the camera.

Figure 15. Game camera images captured of wading birds. Top image: Great egret at a reference mangrove site (East End). Bottom image: Great blue heron at a restored mangrove site (Sweetwater).

Discussion

This study reinforced the emerging consensus that high levels of mangrove cover will decrease overall plant community diversity in coastal wetlands (Guo et al. 2017). This conclusion parallels findings from the comprehensive body of work on woody encroachment in grasslands (e.g., Ratajczak et al. 2012, Limb et al. 2014), The loss of plant diversity in terrestrial systems can have detrimental effects on ecosystem processes such as productivity and trophic support on near- and long-term time scales (Grime 1998). The consequences of species loss in coastal wetlands are not yet fully understood, though there are likely to be decreases in processes such as nutrient cycling and primary productivity (Gedan and Bertness 2009).

A landscape-level shift in the dominant plant community is likely to alter the value of wetlands as shelter and as critical trophic support for many wetland and estuarine fauna (Guo et al. 2017). Although both mangrove and marsh vegetation have the potential to support plant- and detritus-based trophic pathways (e.g., Duarte and Cebrian 1996, Silliman and Zieman 2001, Feller and Chamberlain 2007), this study provided evidence that mangroves are not supporting estuarine consumers in Galveston Bay. Although it is widely expected among regional practitioners that increasing abundance of mangroves will alter trophic dynamics (e.g., Montagna et al. 2007, Perry and Mendelssohn 2009, Osland et al. 2013), this study provided some of the first direct evidence of that trophic alteration.

Changes in plant composition may alter the value of coastal wetlands for many types of resident and migratory wetland-dependent birds. Related work by the PI (Appendix B) suggests that wintering shorebird diversity and relative abundance is higher in marshes than in mangroves on the central Texas coast where mangrove coverage is higher. However, mangroves provide nesting habitat for herons and egrets in Galveston Bay and other regions of the Texas coast (Appendix B). Therefore, a landscape that includes both marsh and mangrove vegetation may provide some benefits for coastal bird communities.

Restoration practice

Mangroves planted within restoration sites in Galveston Bay survived over the course of this study, but did not appear to be thriving. Furthermore, it is unknown what the initial planting density at each site was, and based on the generally poor condition of the surviving trees, it seems likely that there was substantial mortality of planted mangroves. Therefore, planting was not successful in creating a robust mangrove canopy within a reasonable (~5-year) time period.

At sites where planted mangroves survived, they remained small, and had highly localized effects, if any, on the surrounding flora and fauna. However, as *Avicennia* shrubs increase in size, this sphere of influence may increase in size and magnitude; studies in regions of the Texas coast with more expansive mangrove stands have found more pronounced effect (Guo et al. 2017). Therefore, the ecosystem benefits of mangrove restoration likely will be maximized at sites where mangroves are already established.

Anecdotally, mangroves are being found in more sites around Galveston Bay, usually at high elevations, as a result of natural dispersal. Based on these observations, the recommendation for restoration practice is to focus planting efforts on fast growing species such as *Spartina alterniflora* and allow natural recruitment of mangroves to occur gradually over time.

Ecological benefits of mangroves

Overall, sites dominated by mangrove or marsh vegetation were not identical in terms of structure or function. Mangroves reduced vascular plant richness, which may alter carbon supply to herbivores and detritivores. Despite the difference in trophic support, subtidal nekton relative abundance did not appear to strongly differ between sites with and without mangroves, though composition often differs (Appendix B; Johnston and Caretti 2017, Smee et al. 2017). At high densities, mangroves can lower the abundance and diversity of coastal bird communities, but these effects are only discernable at high (> ~60%) mangrove cover (Appendix B). Ultimately, a decline in coastal birds may decrease revenue from birdwatchers, though that outcome is likely to manifest over a longer time scale spanning 5+ years.

Coastal wetlands are important contributors to carbon stocks in marine environments, broadly referred to as blue carbon (Mcleod et al. 2011). Mangroves may increase soil carbon stocks (Doughty et al. 2015). However, this benefit of mangrove expansion was not apparent in Galveston Bay, where soil carbon pools are relatively low compared to other areas of the Gulf Coast (Moyer et al. 2016). Furthermore, mature mangroves in Galveston Bay are small in stature and do not have the capacity to store large amounts of carbon in their biomass. Therefore, the capacity for mangroves to augment soil carbon storage in this region is fairly small.

Summary

Restoration practice

In general, few of the planted mangroves at restoration sites were thriving. Most of the planted shrubs remained small (less than 1 m) for several years after planting, though many were reproductive. The healthiest mangroves were usually at relatively high elevations, near the upper edge of the *Spartina alterniflora* zone. Mangroves are actively recruiting to this high elevation zone at numerous locations around Galveston Bay, independent of planting efforts. Based on these observations, the recommendation for restoration practice is to focus planting efforts on fast growing species such as *Spartina alterniflora* and allow natural recruitment of mangroves to occur gradually over time.

Ecological benefits of mangroves

Marsh plant diversity was lower at sites with high mangrove cover. Within marsh or mangrove stands, fish and invertebrate densities were generally similar, though the species composition differed. Stable isotope analysis at the sites sampled indicated that marsh vegetation is more important than mangroves in supporting coastal wetland food webs. At low densities, mangroves did not substantially alter wading bird or shorebird abundances. Mangroves generally increased carbon retention in the soil. Overall, our results revealed that salt marshes and mangroves support different plant and animal assemblages, and that mangrove encroachment is likely to cause complex changes in ecosystem processes.

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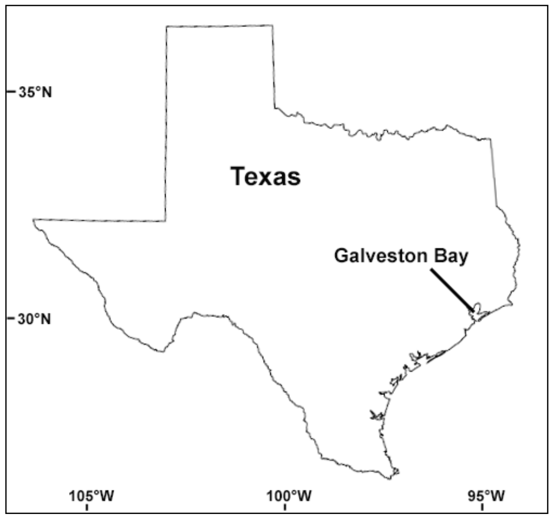
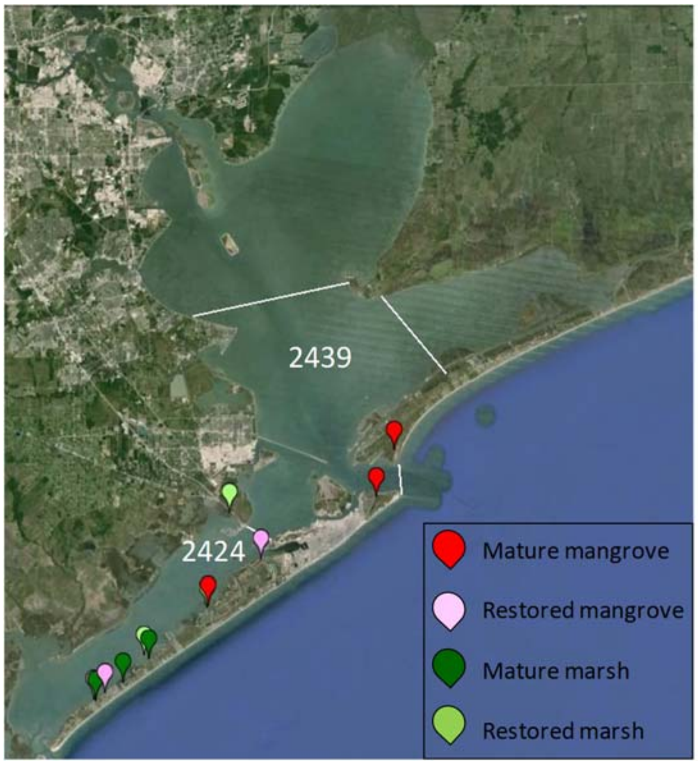
APPENDIX A. MAPS

Area maps depicting the location of Galveston Bay in Texas (A) and the location of study sites within Galveston Bay (B). Map C details the study area and depicts areas of mature and recently established (restored) mangrove and salt marshes. All sites were within the Lower Galveston Bay Watershed; section codes and approximate boundaries are denoted on maps B and C.

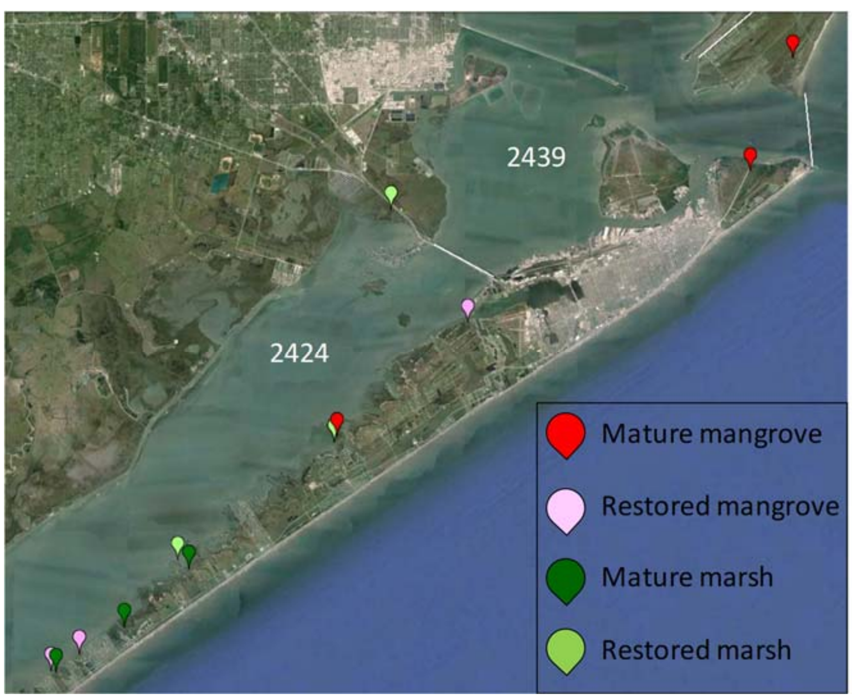
Map D depicts additional sites on the central coast of Texas. These sites were sampled as part of a separate project, but some parameters were compared to those measured in the current study.



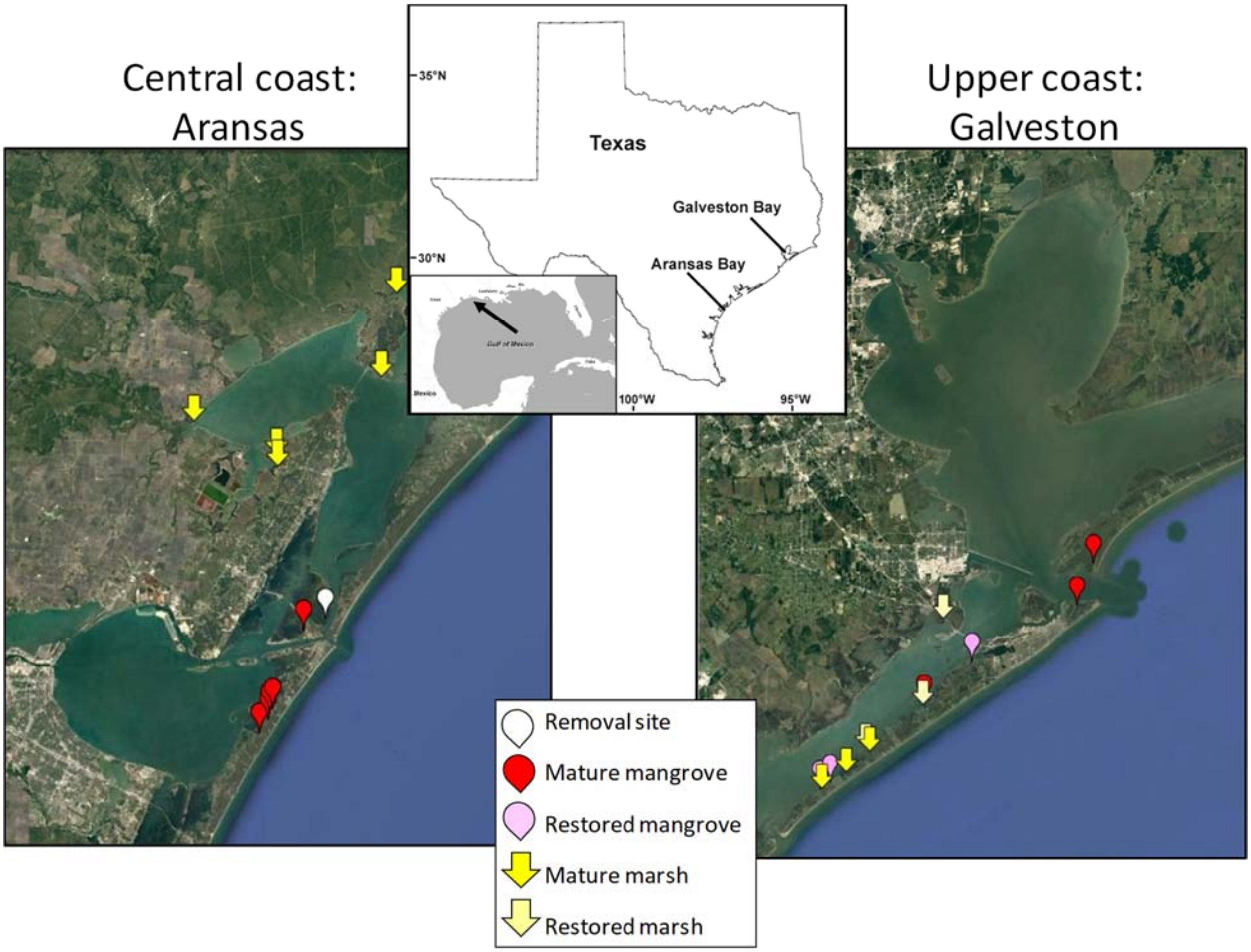
Map A


2439
2424
Mature mangrove
Restored mangrove
Mature marsh
Restored marsh

Map B

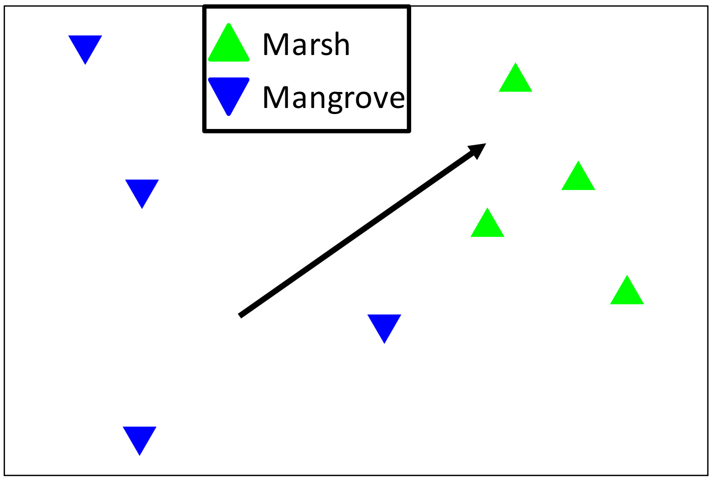

2439
2424
Mature mangrove
Restored mangrove
Mature marsh
Restored marsh

Map C


Central coast:
Aransas
Upper coast:
Galveston
35°N
30°N
Texas
Galveston Bay
Aransas Bay
Gulf of Mexico
Mexico
100°W
95°W
Removal site
Mature mangrove
Restored mangrove
Mature marsh
Restored marsh

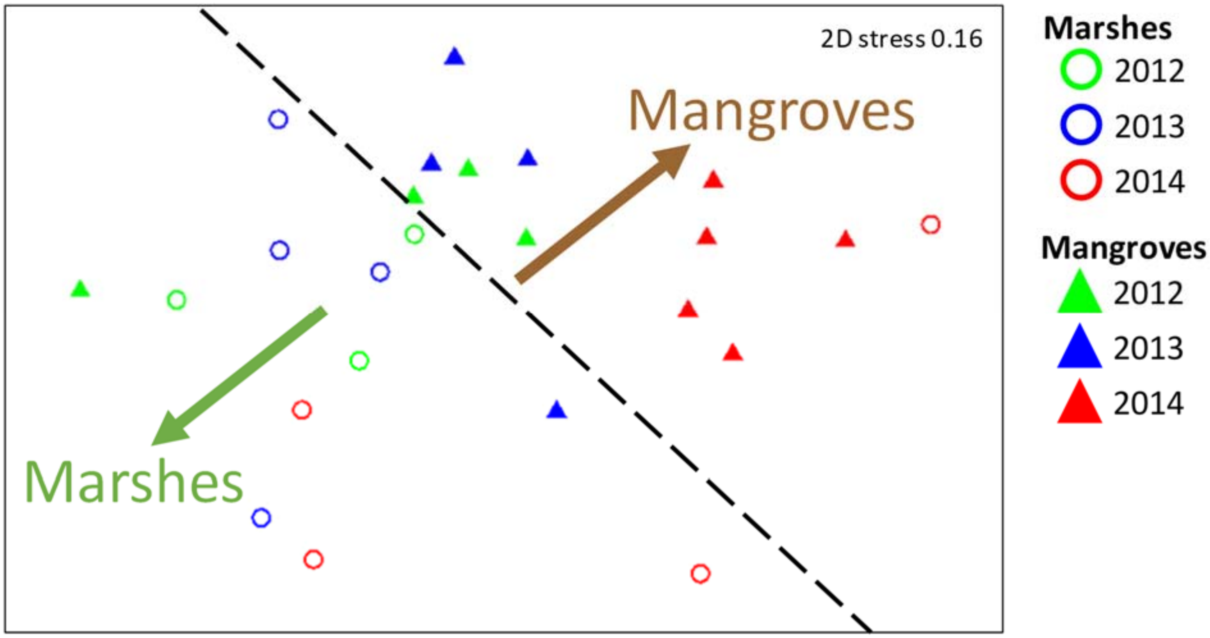
Map D

Appendix B. Related work



Non-metric multidimensional scaling plot depicting differences in shorebird assemblages at sites dominated by marshes or mangroves near Port Aransas, Texas. Arrow indicates direction of increasing abundance and higher species diversity. These results are from a parallel, related study conducted by the PI.

Photo of various species of herons and egrets nesting in a stand of mangroves in West Galveston Bay. Photo credit Jan Culbertson, TPWD.



nMDS plot of nekton assemblages adjacent to stands of marsh and mangrove vegetation from a pilot study conducted by the investigators in Port Aransas, Texas. Nekton relative abundance was similar but composition differed between vegetation types. These results are from a parallel, related study conducted by the PI.