FINAL REPORT

Project Name: CMP 18 – Maximizing the ecological value of coastal wetland restoration: comparisons among restoration techniques

Principal Investigator: Dr. Anna Armitage, TAMUG

Co-principal Investigators: Dr. Antonietta Quigg, Dr. Susan Knock, TAMUG

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Task 1: Wetland Comparisons

Status of the task during this reporting period: ☐in progress ☒completed

Major accomplishments and findings:

Our overarching project objective was to identify restoration techniques that maximize wetland restoration success. Our analyses sought to identify the influences of construction techniques and the surrounding landscape on ecosystem restoration success.

Approaches to wetland restoration vary in construction technique and placement within a larger landscape matrix of wetland habitat. Engineered marshes are often constructed by placing soil in terrace or mound formations, whereas a beneficial uses (BU) approach deposits dredge material to fill continuous areas to emergent marsh elevation. Either construction approach can be planted with native species, or colonization can occur naturally. Likewise, either type of wetland can be isolated in a degraded area, or be situated within a network of relict and restored marshes. We investigated how restoration success was influenced by the localized configuration of individual restoration sites and by the placement of each site within a wetland matrix.

Our study areas included *engineered* sites, multiple *beneficial use* sites, and additional unmanaged (*reference*) sites, in the Lower Neches and J.D. Murphree Wildlife Management Areas (Fig. 1). *Engineered marshes* contained mound or terrace formations that form a mixture of aquatic habitat, emergent marsh habitat, and "edge" habitat at the vegetation-water interface. *Beneficial uses marshes* (BUDM) were created by depositing dredge material to fill continuous areas to emergent marsh elevation, creating areas with expansive emergent marsh habitat. Subsets of both areas were actively planted with emergent marsh vegetation, and others were left unplanted. *Reference marshes* were adjacent unmanaged areas with comparable elevation and tidal influence.

In October 2013 and 2014, we surveyed emergent plant characteristics in planted engineered and BUDM sites that varied in size, isolation, and proximity to urban developments near Sabine Lake, TX. Plant biomass, cover, and species richness in BUDM marshes were similar to reference conditions (Figs. 2-4). In contrast, emergent plant biomass and cover were frequently over 70% lower in engineered marshes than in BUDM and reference marshes. Restoration failure (emergent plant cover < 10 % and biomass < 0.5 kg/m^2) occurred only in small (< 0.5 km^2) sites, though not all small sites failed (Fig. 5). Plant species richness was up to 2x higher in more altered sites that were close (< 1 km) to roads or urban development (Fig. 6). Individual restoration sites were highly dissimilar to each other, and some were failures in terms of emergent plant cover. However, when the failed sites were within a relatively large surrounding matrix of successful restored and reference sites, the ecosystem effects of that failure were minimized. Our analysis shows that construction method is less important than the placement of restoration projects within a fairly large wetland matrix in ensuring restoration success.

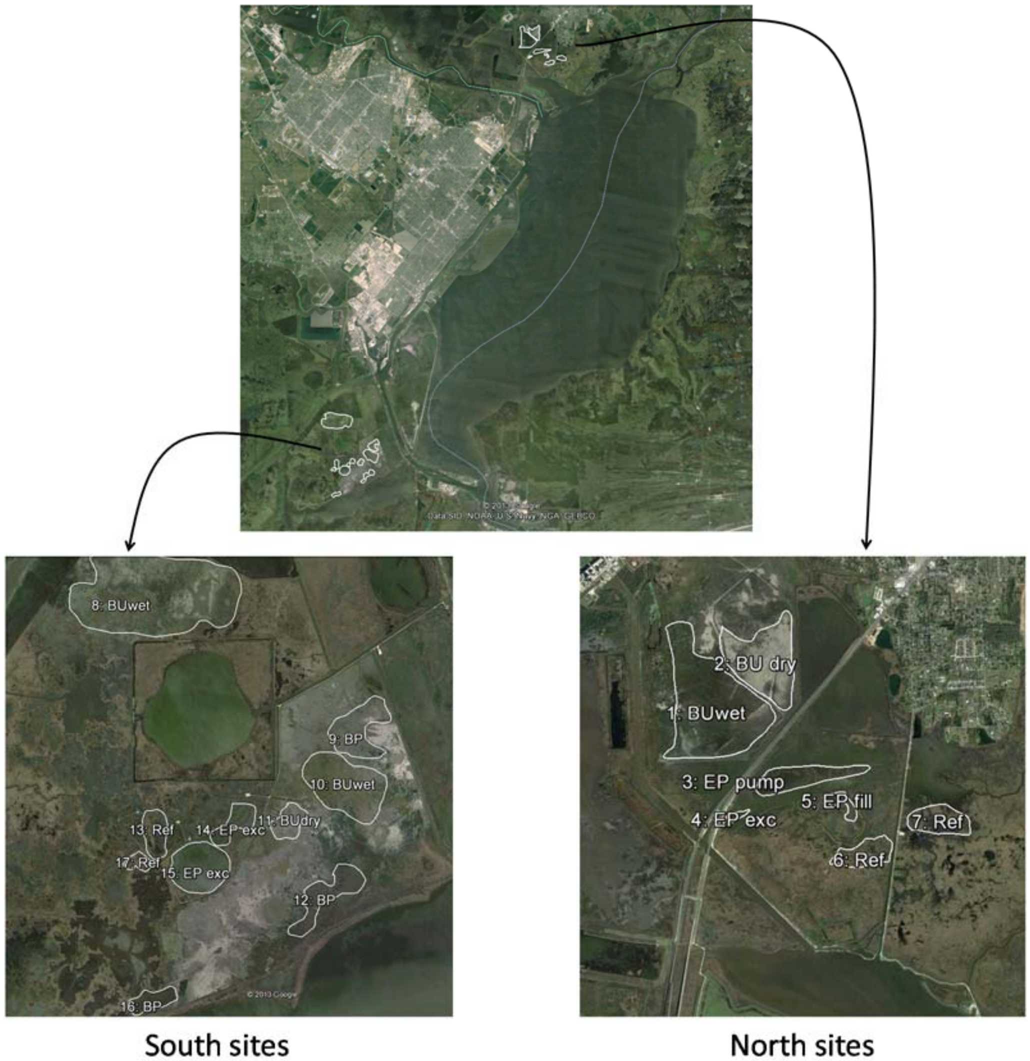

Figure 1: Study sites relative to Sabine Lake (top panel). Close-up views of site groups (lower panels). EP = Engineered, planted; BP = Beneficial uses, planted; BU dry = Beneficial uses, unplanted, high elevation; BU wet = Beneficial uses, unplanted, low elevation; Ref = Reference sites. EP areas are subdivided into different structural configurations: exc = dedicated on-site excavation into terrace or mound formations; pump = addition of off-site dredge material, creating mound formations; filled = excavations to create mounds followed by the addition of dredge material.

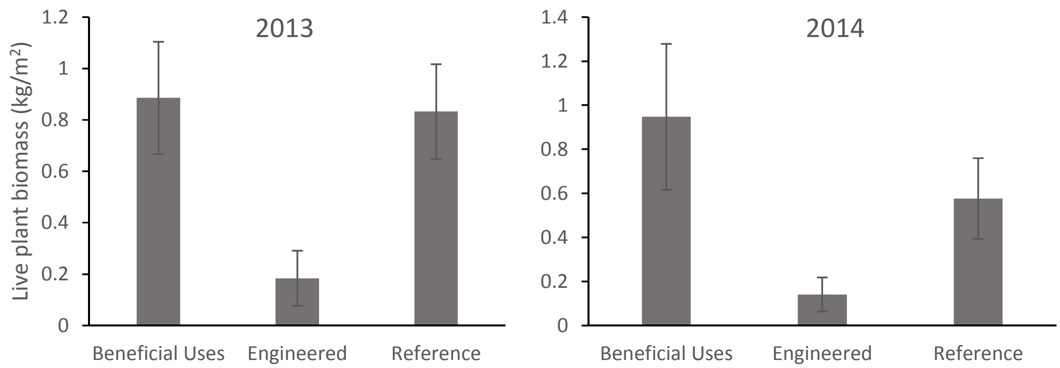

Figure 2: Average live plant biomass cover in planted beneficial uses sites, planted engineered sites, and reference sites, in October 2013 and October 2014. Error bars represent standard error.

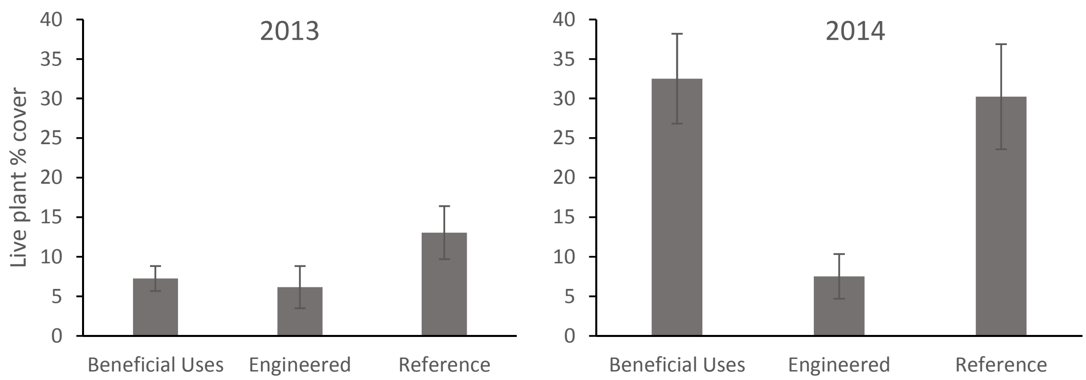

Figure 3: Average live plant percent cover in planted beneficial uses sites, planted engineered sites, and reference sites, in October 2013 and October 2014. Error bars represent standard error.

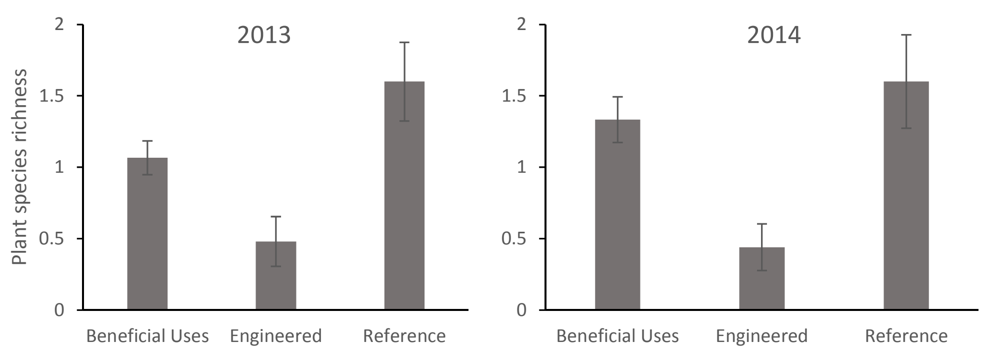

Figure 4: Average plant species richness in planted beneficial uses sites, planted engineered sites, and reference sites, in October 2013 and October 2014. Error bars represent standard error.

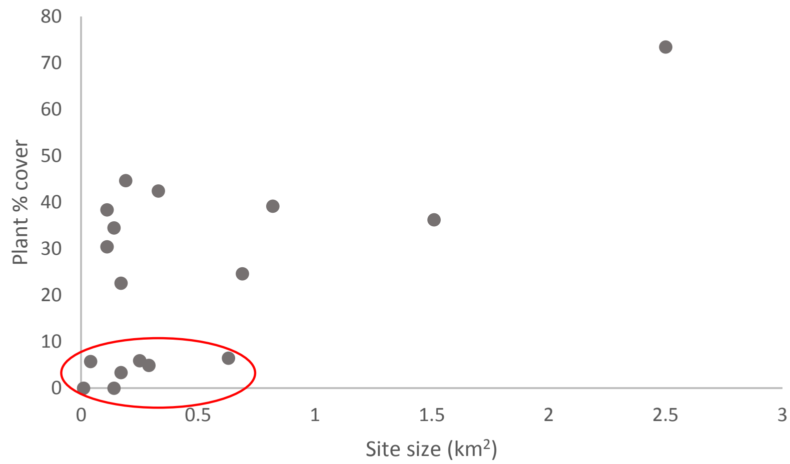

Figure 5: The relationship between live plant percent cover and site size in October 2014. The red circle highlights failed restoration sites, defined as < 10% emergent plant cover.

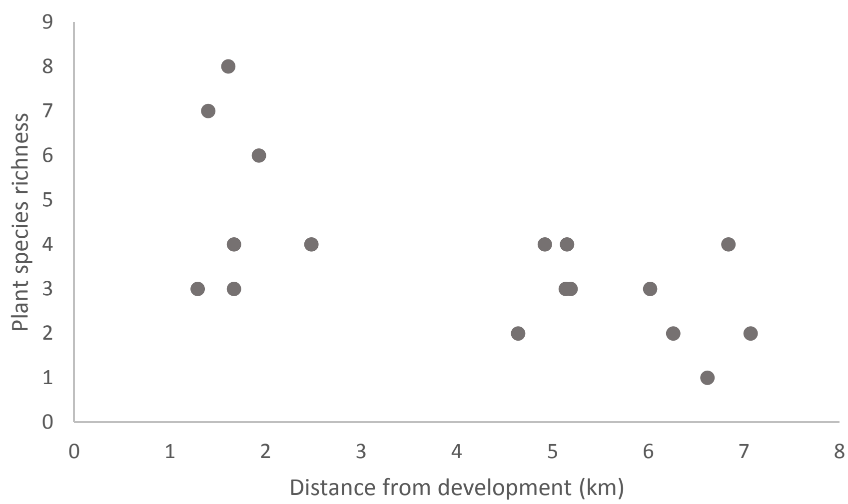

Figure 6: The relationship between plant species richness and distance from urban development in October 2014.

Deliverables completed during this reporting period:

The final report for this project was posted on the PI's webpage (http://www.tamug.edu/armitage/Current_Projects.html).

The following oral and poster presentations related to this project were or will be given at local, national, and international scientific conferences. The abstracts are included as documentation of deliverable completion.

(* indicates graduate student; ◊ indicates postdoc)

Quigg A., ◊Kinney E.L., Bowers K., Ho C.-K., Madrid E.N., Bell M.T., Armitage A.R. November 2013. Did acute drought affect ecosystem development in a restored brackish marsh? Coastal and Estuarine Research Federation 22nd Biennial Conference.

Abstract: Extreme events like droughts can dramatically alter ecosystem functions – the effects may be most profound in transitional (e.g., brackish) habitats or in the early-successional stages in developing ecosystems, such as recently restored habitats. As part of a long-term wetland restoration monitoring program, we were able to document the effects of the 2011 exceptional drought in Texas and to follow near-term recovery of ecosystem functions in restored brackish marshes in the NW Gulf of Mexico. As a result of the year-long drought, salinities increased to three-times greater than normal for several months, but then returned to normal (< 10 ppt) in 2012. Immediate effects on emergent plants were minimal, but lower plant biomass in 2012 suggested long-term drought effects. Drought and the corresponding high salinity had dramatic and long-term effects on submerged aquatic vegetation (SAV). The invasive Eurasian milfoil (*Myriophyllum spicatum*) was essentially eradicated during the drought and remained rare throughout 2012. Drought effects were less dramatic for native *Ruppia maritima*, which slightly increased in biomass in 2011 and 2012, possibly due to a release from competition from *Myriophyllum*. Drought may have caused a long-term loss of total SAV biomass and a possibly beneficial shift in species composition by disproportionately impacting the invasive species. Effects on phytoplankton were immediate during the drought, with a shift in community composition from diatom- to cyanobacteria-dominated. Many aquatic fauna, including snails, grass shrimp, and fish, declined dramatically during the drought. This may have been a direct response to salinity, as well as a trophic response to the substantial changes in the producers at the base of the food web. The findings from this study, which include pre-, during-, and post-drought surveys, provide insight into how acute drought affects ecosystem development in restored brackish marshes.

Armitage A.R., Bowers K., Bergren, R., Quigg A. 2015. Maximizing wetland restoration success: the influences of construction techniques and the surrounding landscape. Coastal and Estuarine Research Federation 23rd Biennial Conference.

Abstract: Approaches to wetland restoration vary in construction technique, planting strategy, and placement within a larger landscape matrix of wetland habitat. Engineered marshes are often constructed by placing soil in terrace or mound formations, whereas a beneficial uses (BU) approach deposits dredge material to fill continuous areas to emergent marsh elevation. Either construction approach can be planted with native species, or colonization can occur naturally. Likewise, either type of wetland can be isolated in a degraded area, or be situated within a network of relict and restored marshes. We investigated how restoration success was influenced by the localized configuration of individual restoration sites and by the placement of that site within a wetland matrix. In October 2014, we surveyed emergent plant characteristics in planted engineered and BU sites along with unplanted BU tidal brackish marshes that varied in size, isolation, and proximity to urban developments near Sabine Lake, TX (USA). Plant biomass, cover, and species richness in BU marshes were similar to reference conditions, regardless of planting technique. In contrast, emergent plant biomass and cover were over 70% lower in engineered marshes than in BU and reference marshes. Restoration failure (emergent plant cover < 10 % and biomass < 0.5 kg/m^2) occurred only in small (< 0.5 km^2) sites, though not all small sites failed. Plant species richness was up to 2x higher in more altered sites that were close (< 1 km) to roads or urban development. Individual restoration sites were highly dissimilar from each other, and some were failures in terms of emergent plant cover. However, when the failed sites were within a relatively large surrounding matrix of successful restored and reference sites, the ecosystem effects of that failure were minimized. Our analysis shows that construction method is less important than the placement of restoration projects within a fairly large wetland matrix in ensuring restoration success.

Bowers K., Armitage A.R., Bergren, R., ◊Kinney E.L., ◊Ho C.-K., ◊Madrid E.N., *Bell M.T., Quigg A. 2015. Resilience versus vulnerability: Prolonged consequences of an exceptional drought in a brackish marsh. Coastal and Estuarine Research Federation 23rd Biennial Conference.

Abstract: Extreme events like droughts can substantially alter ecosystem functions, and the effects may be particularly dramatic in transitional habitats such as brackish marshes. As part of a six-year restoration monitoring program, we documented the ecosystem effects of an exceptional drought in Texas in 2011. We measured emergent and aquatic habitat characteristics over two years before the drought, during the drought, and over a three-year recovery period. The emergent plant community was resilient to drought conditions; plant cover, biomass, and productivity did not significantly differ among years. In contrast, characteristics of the aquatic community, including water quality, submerged aquatic vegetation (SAV) biomass, and total fish and invertebrate densities were markedly different among pre-, during-, and post-drought periods. Prior to the drought, salinities were low (< 10 ppt), invasive Eurasian milfoil (*Myriophyllum spicatum*) dominated the SAV, and faunal densities frequently exceeded 25/m^2. During the 2011 drought, salinity was > 25 ppt, leading to an eradication of milfoil and a drop in fish and invertebrate densities to less than 5/m^2. By 2012, one year after the drought, salinity returned to brackish levels. However, milfoil remained absent and was replaced by native widgeongrass (*Ruppia maritima*) and filamentous green algae. The fish and invertebrate communities were relatively resilient, and recovered to comparable pre-drought density and composition, though with high interannual variation. Overall, the brackish marsh exhibited complex responses to an exceptional drought, with high resilience within the emergent plant assemblage. Aquatic plants did not return to pre-drought conditions, but drought effects on aquatic fauna were temporary. These findings highlight the prolonged consequences for brackish ecosystems as a result of an exceptional drought.

Armitage A.R., Bowers K., Bergren, R., Quigg A. January 2016. Maximizing wetland restoration success in Galveston Bay: lessons on the influences of construction techniques and the surrounding landscape. 10th State of the Bay Symposium.

Abstract: Approaches to wetland restoration vary in construction technique, planting strategy, and placement within a larger landscape matrix of wetland habitat. Engineered marshes are often constructed by placing soil in terrace or mound formations; this approach is common in Galveston Bay. In contrast, the relatively less widespread beneficial uses (BU) approach deposits dredge material to fill continuous areas to emergent marsh elevation. Either construction approach can be planted with native species, or colonization can occur naturally. Likewise, either type of wetland can be isolated in a degraded area, or be situated within a network of relict and restored marshes. We investigated how restoration success was influenced by the localized configuration of individual restoration sites and by the placement of that site within a wetland matrix. In October 2014, we surveyed emergent plant characteristics in planted engineered and BU sites along with unplanted BU tidal brackish marshes that varied in size, isolation, and proximity to urban developments near Sabine Lake, TX (USA). Plant biomass, cover, and species richness in BU marshes were similar to reference conditions, regardless of planting technique. In contrast, site-level emergent plant biomass and cover were over 70% lower in engineered marshes than in BU and reference marshes. Restoration failure (defined as emergent plant cover < 10 % and biomass < 0.5 kg/m^2) occurred only in small (< 0.5 km^2) sites, though not all small sites failed. Plant species richness was up to 2x higher in more altered sites that were close (< 1 km) to roads or urban development. Individual restoration sites were highly dissimilar from each other, and some were failures in terms of emergent plant cover. However, when the failed sites were within a relatively large surrounding matrix of successful restored and reference sites, the ecosystem effects of that failure were minimized. Our analysis shows that construction method is less important than the placement of restoration projects within a fairly large wetland matrix in ensuring restoration success. These lessons are directly applicable to the design and implementation of future wetland restoration projects in Galveston Bay.

Were there any problems or obstacles encountered during this reporting period (e.g., delays, remedial action taken, schedule revision). ☒Yes ☐No If yes, please explain:

A three-month no-cost extension was granted to extend the project end date to 6/30/15. All tasks were completed by that end date.

Task 2: Planting Technique Comparisons

Status of the task during this reporting period: ☐in progress ☒completed

Major accomplishments and findings:

As part of the monitoring performed for Task 1, we compared a subset of the sites by focusing on beneficial uses sites with and without plants added during the initial restoration effort. We measured plant biomass and cover at these sites.

Plant biomass was lower in unplanted marshes than in planted marshes and reference sites, particularly in 2013 (Fig. 7). The difference between unplanted and reference conditions was relatively small in 2014. Plant biomass in planted and unplanted marshes was similar to reference conditions in 2013 and 2014 (Fig. 8).

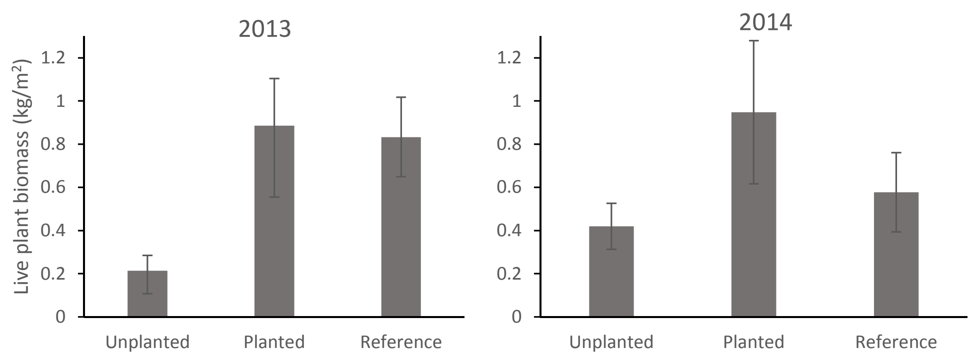

Figure 7: Average live plant biomass in planted and unplanted beneficial uses restored sites and reference sites in October 2013 and October 2014. Error bars represent standard error.

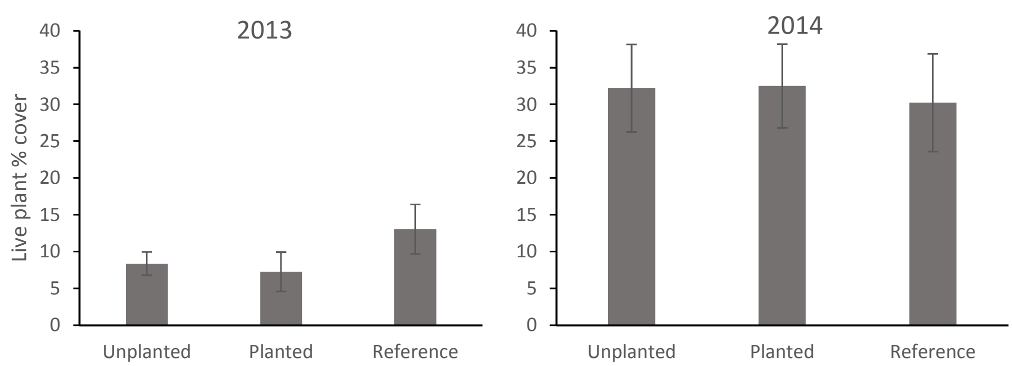

Figure 8: Average live plant percent cover in planted and unplanted beneficial uses restored sites and reference sites in October 2013 and October 2014. Error bars represent standard error.

Deliverables completed during this reporting period: See Task 1.

Were there any problems or obstacles encountered during this reporting period (e.g., delays, remedial action taken, schedule revision). ☒Yes ☐No If yes, please explain:

A three-month no-cost extension was granted to extend the project end date to 6/30/15. All tasks were completed by that end date.

Task 3: Secondary production: Aquatic fauna and shorebirds

Status of the task during this reporting period: ☐in progress ☒completed

Major accomplishments and findings:

Many wetland bird species forage in unvegetated areas called mudflats, yet these mudflats are rarely included in wetland restoration design. The beneficial uses (BUDM) restoration method uses dredge material to create a continuous area of wetlands. This approach can yield small variations in elevation, soil moisture, and the amount of vegetation. These variations may increase the number and diversity of birds using the restored area. We compared populations of birds, and the small, mud-dwelling animals (infauna) that they eat, among planted, unplanted + high elevation (dry), and unplanted + low elevation (wet) areas of a restored coastal marsh near Port Arthur, TX. We hypothesized that unplanted, wet areas would have the most infauna, and would therefore support more birds. Replicate cores were taken from each habitat type, sieved, and infauna were identified to the lowest taxonomic group possible. To monitor bird use, time-lapse wildlife cameras were deployed in each habitat type during the overwintering and spring and fall migration periods in 2013 and 2014. The pictures were later analyzed and birds were identified to species. Infaunal density in unplanted areas was 109 times higher than in planted areas (Fig. 9). Infaunal density was 68 times higher in wet unplanted areas than in dry unplanted areas. Likewise, birds were most frequently observed in wet, unplanted areas (Fig. 9). Snails (Class Gastropoda) were the most common type of fauna found, followed by bivalves (Class Pelecypoda) and worms (Class Oligochaeta) (Fig. 10). Oligochaetes were only found in wet, unplanted areas. The most common birds in wet areas were ibis, herons, egrets, and ducks (Fig. 11). In conclusion, when restoring wetlands, including areas that are wet and unplanted is important for providing habitat for the infauna that will support residential and migratory bird populations.

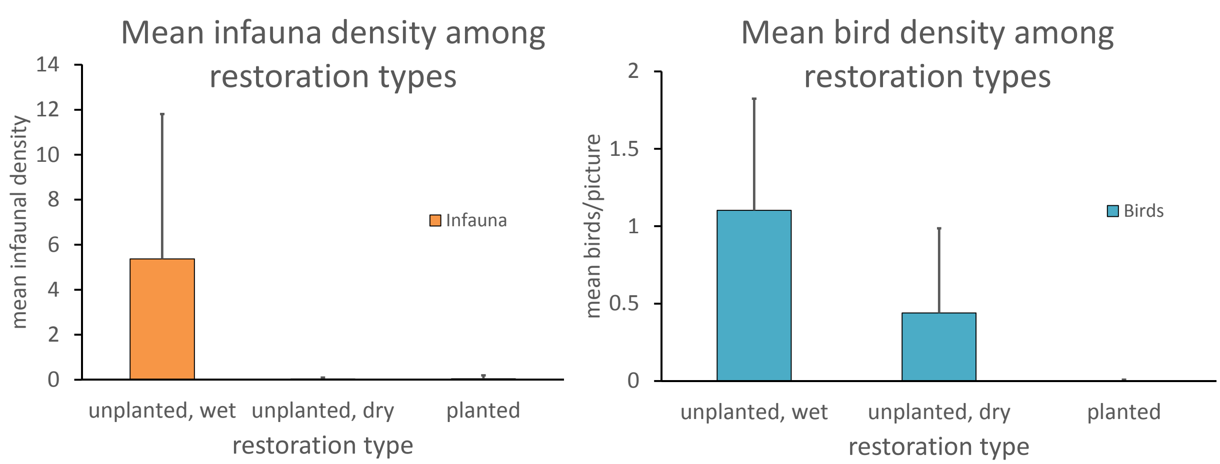

Figure 9: Average infaunal and bird densities in wet (low elevation) and dry (high elevation) unplanted beneficial uses restored sites, and in planted restored sites, in spring 2014. Error bars represent standard error.

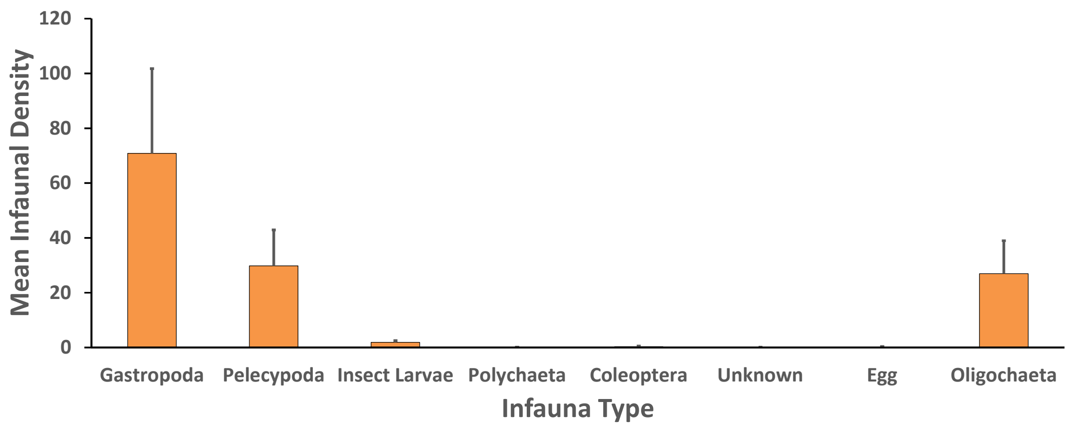

Figure 10: Average densities of infaunal groups, pooled across all restored wetland types, in spring 2014. Error bars represent standard error.

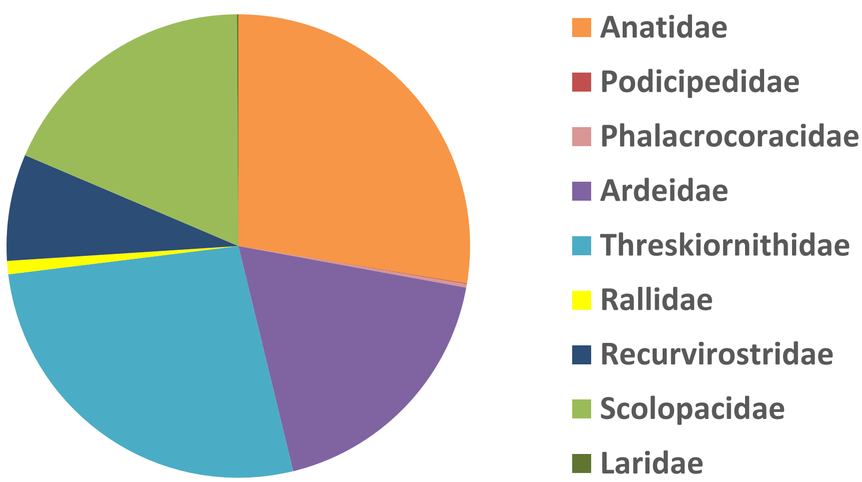

Figure 11: Relative family composition of bird assemblages, pooled across all restored wetland types, in spring 2014.

Deliverables completed during this reporting period:

The following poster presentations related to this project were given at local, national, and international scientific conferences. The abstracts are included as documentation of deliverable completion.

(* indicates graduate student; # indicates undergraduate student)

#Morrison R.V., *Whitt A.A., *Weaver C.A., Armitage A.R. April 2014. Bird density and family composition in restored brackish marshes. Texas A&M University at Galveston Student Research Symposium.

Abstract: Coastal wetlands provide critical foraging and roosting habitat for many species of migratory and resident birds. Wetland restoration usually focuses on emergent vegetation, and seldom includes the mudflat or subtidal habitat that is essential for these species, but an innovative restoration project near Port Arthur, TX

included some unvegetated mudflat habitat. We compared bird density and species richness between engineered marshes (EM) surrounded by extensive aquatic habitat and beneficial uses marshes (BUDM) with extensive mudflat. Data were recorded using time lapse digital cameras that took two pictures every 30 minutes from dawn to dusk, for five day periods in October and December 2013. Families Anatidae (ducks), Areidae (herons, egrets), Podicipedidae (grebes), Recurvirostridae (black-necked stilts), Scolopacidae (sandpipers), and Threskiornithidae (ibis and roseate spoonbills) were more abundant in the BUDM area, whereas families Pelecanidae (pelicans) and Phalacrocoracidae (cormorants) were more abundant in the EM area. The data are consistent with the feeding behavior of the birds observed; herons, egrets, and ibises forage in shallowly flooded mudflats, like those at the BUDM sites, for gastropods, bivalves, and crustaceans. Other birds, like cormorants and pelicans, prefer to eat fish and will dive for them in deeper waters like those at the EM site. These findings demonstrate the ecological benefit of including a variety of subtidal and mudflat habitats into coastal wetland restoration.

*Whitt A., #Morrison R., Rathjen M., #Norris A., Armitage A.R. July 2014. Migrating bird use of brackish marshes: Does restoration technique matter? 2014 Conference on Ecological and Ecosystem Restoration.

Abstract: An important goal of coastal wetland restoration, particularly from the public perspective, is to rehabilitate impacted wildlife populations, including migratory and resident birds. If successful, such restoration projects will revitalize migratory flyways for waterfowl (ducks) and shorebirds by providing trophic support and roosting habitat. There are many different approaches to wetland restoration, including variations in construction technique and planting strategy. *Engineered* marshes are often constructed by placing soil in terrace or mound formations, creating aquatic habitat that can be used by waterfowl. In contrast, *beneficial uses (BUDM)* marshes are created by depositing dredge material to fill continuous areas to emergent marsh elevation. Neither the *engineering* nor the *BUDM* approaches explicitly incorporates mudflat habitat, despite its importance for the charismatic shorebirds that are iconic coastal wetland species. Mudflat habitat may be created in *BUDM* areas by reducing or delaying marsh planting.

We investigated how these varied restoration techniques altered migratory and wintering bird usage of restored brackish marshes in the J. D. Murphree Wildlife Management Area near Port Arthur, TX (USA). Our research addressed two questions: (1) What is the value of restored brackish marshes to migrating birds? (2) Does bird density and species composition differ among marsh restoration techniques? We deployed time-lapse game cameras for two-week periods in fall 2013 and winter 2014 in a native undisturbed marsh and two constructed marshes, one with engineered mounds and the other an unplanted poured dredge slurry. We assessed bird utilization of the restored brackish marshes by comparing the frequency of bird presence and bird species richness among habitat types. Shorebirds and waterfowl preferred the less vegetated restored marshes, relative to the heavily vegetated native marsh. Of the two constructed marsh types, there was a higher abundance of birds in the *BUDM* marsh than in the marsh with engineered mounds. However, there were different species of waterfowl, wading and shorebirds utilizing each restored marsh type. For instance, *Eudocimus albus* (white ibis) and *Limnodromus* sp. (dowitcher) were more commonly seen along the edges of the engineered mounds, whereas *Himantopus mexicanus* (black-necked stilts) and *Anas discors* (blue-winged teal) utilized the *BUDM* marsh. These data will inform management decisions by showing that an ideal restoration design incorporates both aquatic and mudflat habitats which can be utilized by various species of waterfowl and shorebirds.

#Norris, A.E., Armitage, A.R. March 2015. Identifying coastal wetland restoration techniques to maximize benefits to bird populations. Texas Undergraduate Research Day at the Capitol.

Abstract: Many wetland bird species forage in unvegetated areas called mudflats, yet these mudflats are rarely included in wetland restoration design. Therefore, wetland restoration projects that include small variations in elevation, soil moisture, and the amount of vegetation may increase the number and diversity of birds using the restored area. Our objective was to compare populations of birds, and the small, mud-dwelling animals (infauna) that they eat, among high elevation (dry), low elevation (wet), planted, and unplanted areas of a restored coastal marsh near Port Arthur, TX. We hypothesized that unplanted, wet areas would have the most infauna, and would

therefore support more birds. Infauna were collected from soil cores, and game cameras captured bird pictures every 30 minutes and were also triggered by motion. Unplanted areas had 109 times more infauna than planted areas. Wet areas had 68 times more infauna than dry areas. There was a positive correlation between the number of infauna and number of birds within an area. The most common birds in wet areas were ibis, herons, egrets, and ducks. When restoring wetlands, including areas that are wet and unplanted is important for providing habitat for the infauna that will support residential and migratory bird populations.

#Norris, A.E., Armitage, A.R. April 2015. Identifying coastal wetland restoration techniques to maximize benefits to bird populations. 2015 Texas Bays and Estuaries Meeting.
See above.

#Norris, A.E., Armitage, A.R. April 2015. Identifying coastal wetland restoration techniques to maximize benefits to bird populations. TAMUG Research Symposium.
See above.

Were there any problems or obstacles encountered during this reporting period (e.g., delays, remedial action taken, schedule revision). ☒Yes ☐No If yes, please explain:

A three-month no-cost extension was granted to extend the project end date to 6/30/15. All tasks were completed by that end date.

Task 4: Education and Outreach

Status of the task during this reporting period: ☐in progress ☒completed

Major accomplishments and findings:

Over the course of the project, we partially supported one graduate student (Whitt) who is conducting thesis work related to this project. We supported two undergraduate research interns (Norris, Morrison) to assist with bird image processing and analysis, and employed two undergraduate student workers to assist with sample processing.

Deliverables completed during this reporting period:

See Task 3.

Were there any problems or obstacles encountered during this reporting period (e.g., delays, remedial action taken, schedule revision). ☒Yes ☐No If yes, please explain:

A three-month no-cost extension was granted to extend the project end date to 6/30/15. All tasks were completed by that end date.

Task 5: Project Reporting and Data Transfer

Status of the task during this reporting period: ☐in progress ☒completed

Major accomplishments and findings:

Progress reports have been filed on schedule.

Deliverables completed during this reporting period:

Quarterly progress reports have been filed on schedule.

Were there any problems or obstacles encountered during this reporting period (e.g., delays, remedial action taken, schedule revision). ☒Yes ☐No If yes, please explain:

A three-month no-cost extension was granted to extend the project end date to 6/30/15. All tasks were completed by that end date.

Acknowledgements: Logistical support, access to the study site (Old River Unit of the Lower Neches Wildlife Management Area), and additional support was provided by the Wildlife Division of the Texas Parks and Wildlife Department, under Wildlife Division Director Clayton Wolf. In particular, Jim Sutherlin and Mike Rezsutek from TPWD provided extensive logistical support.

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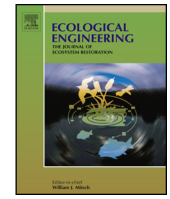

The influence of habitat construction technique on the ecological characteristics of a restored brackish marsh

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ABSTRACT

The primary goal of most habitat restoration projects is to improve ecosystem functions as compensation for habitat loss or degradation, but the optimal engineering approach to achieve that outcome is not always known a priori. Restored coastal wetlands are frequently engineered to create mound and terrace formations at low marsh elevations, but there have been few opportunities to quantitatively compare the ecological characteristics of construction methods that differ in soil source and configuration. Our study took place in a restored (2008) brackish marsh (Texas, USA) that included mounded formations built from on-site soil, off-site dredge material, or a combination of soil sources. We used a two-year (2009–2010) dataset from a restoration monitoring program that included emergent plant, water, soil, aquatic plant, and aquatic faunal characteristics to address two questions: (1) Do construction methods combining different soil sources and dredging techniques confer unique ecological characteristics? (2) Is there an ecological benefit to incorporating heterogeneity in a restored site by employing multiple construction methods? Our analyses revealed that plant root biomass and soil nitrogen and phosphorus concentrations were two times higher in the reference area than in any of the restored areas. Among the restoration construction methods, ecosystem characteristics were similar to each other within two years of restoration. In general, ecosystem characteristics were affected more by temporal variation between years than by construction method. Differences between years were driven by water characteristics; unusually high tides in 2010 doubled salinity and decreased water chlorophyll *a* concentration and dissolved inorganic nitrogen by half. Although the restored areas did not achieve all reference characteristics during the early development of the site, the differences among engineering approaches were relatively subtle. Therefore, the recommendation for practice is to use the approach that is most cost-effective for a specific site. In our study area, the dredging methods yielded the largest area of emergent marsh per unit effort, but the on-site soil excavations created more aquatic habitat. When ecological integrity is defined as the provision of a wide range of biotic and abiotic conditions at a landscape scale, then the use of different engineering approaches at different sites within a region creates habitat heterogeneity, thus conferring regional-level ecological benefits.

1. Introduction

The goals of ecological restoration can include the creation of new habitat or enhancing the integrity of existing habitat. Whatever the ultimate ecological objective, the first steps of the restoration process involve careful engineering to yield appropriate hydrology, elevation, and soil characteristics (Mendelssohn and Kuhn, 2003; Mitsch and Cronk, 1992; Turner and Streever, 2002). Specific details of the construction approach are often dictated by logistical constraints, such as the available sources of sediment and the topography of the site. The plant species used in restoration are usually intended to resemble reference populations, though the logistics of cultivating and transplanting plants often limits practitioners to using a few hardy species. Associated animal populations are seldom actively restored, under the broad assumption that they

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will colonize after soil, hydrology, and plant characters are suitable (Palmer et al., 1997).

Ecological restoration is particularly relevant in wetlands – a habitat where widespread loss has necessitated the development of regional management strategies and the initiation of scores of restoration projects across the continental United States. Wetland loss has been particularly extensive in the Gulf of Mexico due a number of contributing factors, including impoundments, hydrological diversions, and subsidence following the withdrawal of groundwater (Dahl, 1990; Moulton et al., 1997; Turner, 1990). Many projects in this region must add sediment to create habitat at an appropriate emergent marsh elevation. Soil sources may include dredge material from off site or dedicated on-site excavations (Turner and Streever, 2002). Over the last 20 years, many projects in the northern Gulf of Mexico have utilized some form of "terracing", where emergent marsh is created by excavating bay-bottom sediments and shaping them into elongated or circular mounded structures (Turner and Streever, 2002). This approach creates both emergent and aquatic habitat, which augments habitat value for waterfowl and fish (La Peyre et al., 2007; O'Connell and Nyman, 2010). Other projects have backfilled slurries of off-site dredge material to the aquatic habitat around terraces, or have substituted slurries for excavated terraces; these slurry amendments to excavated sediments often accelerate plant reestablishment (Schrift et al., 2008; Slocum et al., 2005; Stagg and Mendelssohn, 2010; Streever, 2000).

Most wetland restoration projects include some assessment of success relative to reference wetlands (e.g., Edwards and Mills, 2005; Rozas and Minello, 2009). However, there have been few opportunities to quantitatively compare the ecological characteristics of construction methods that use terrace excavation, slurried backfill, or a combination of both techniques. Baustian et al. (2009) evaluated dredged canals that were restored to wetland elevation by filling with different soil sources but found few differences in the plant communities among treatment and reference sites. Other studies have compared different terrace spatial configurations, but have usually focused on specific functions, particularly fishery value (Merino et al., 2010; Rozas et al., 2005; Rozas and Minello, 2001, 2009). At a broader ecosystem level that incorporates a range of biotic and abiotic characteristics, the differences in ecological characteristics among construction approaches have not been well quantified.

In coastal wetland landscapes, small differences in elevation, on the scale of a few centimeters, can have dramatic effects on flooding frequency, and subsequently on soil characteristics and plant species composition. Therefore, using a variety of terracing and backfilling restoration techniques that introduce topographical heterogeneity can potentially also support a wide variety of ecosystem functions (Bell et al., 1997). For example, wetlands that include emergent vegetation, mudflats, and tidal creeks have higher productivity at multiple trophic levels, from microalgae and vascular plants to macrofauna, fish, and birds (Armitage et al., 2007; Callaway, 2005; Janousek, 2009; Larkin et al., 2009; Morzaria-Luna et al., 2004). Ecosystem functions such as nutrient cycling and water quality remediation are also improved in more topographically heterogeneous habitats (Wolf et al., 2011). Terracing approaches can be successful in creating topographical heterogeneity in restored wetlands by explicitly integrating aquatic habitat into the design (Hough-Snee et al., 2011; O'Connell and Nyman, 2010), but additional ecological value may be conferred by using multiple construction techniques within a single restoration site. To define the nature of those ecological benefits, restoration practitioners must first understand if there are differences in ecological benefits among construction approaches.

We used a two-year (2009–2010) dataset from a brackish marsh restoration monitoring program to address two questions: (1) Do construction methods combining different soil sources and dredging techniques confer unique ecological characteristics? (2) Is there an ecological benefit to incorporating heterogeneity in a restored site by employing multiple construction methods? In our study, we define "ecological benefit" as the multivariate similarity of a suite of biotic and abiotic characteristics to a reference area.

2. Methods

2.1. Study site

The study site was within a wetland complex in the Texas Parks and Wildlife Department's Old River Unit of the Lower Neches Wildlife Management Area (LNWMA), Texas, USA (30°00 N, 93°51 W) (Fig. 1). As part of the Chenier Plains ecosystem, the site historically consisted of hydraulically connected shallow lakes and small bayous. Natural processes (e.g., hurricane surges) and human activities, primarily the construction of navigation channels, introduced saline tidal waters to this ecosystem. By the mid-1970s, the influx of saline water and concurrent natural subsidence had contributed to a substantial loss of freshwater marsh vegetation throughout the area.

As part of a mitigation project, open water areas of the LNWMA were restored to emergent marsh elevation in 2008 (Fig. 1). Hydrological restoration involved the installation of earthen plugs (30 cm above mean sea level) in access canals in 1997 and 2007 in an effort to limit salt water input to the site and create uniform tidal input across the study area. In 2007, three construction methods were used to create emergent mounds (1–3 m in diameter) in a variant of the "terracing" restoration technique (Turner and Streever, 2002). All structures were engineered to achieve elevations of 0.2–0.4 m above mean sea level, which is an optimal elevation for low marsh vegetation (McKee and Patrick, 1988) and was the approximate elevation of the reference area. All mounds were planted with cultivated *Spartina alterniflora* cv. Vermilion, which is a common, salt tolerant, low marsh species in Gulf of Mexico coastal marshes, which typically have low plant species diversity.

The three construction methods varied in terms of soil source and aquatic habitat depth (Fig. 2). *Excavated* mound construction was based on conventional terracing techniques, where mounds were created from dedicated on-site sediment adjacent to the mound construction and were surrounded by a water depth of approximately 1 m. *Filled* mounds were created in a novel combination of two steps: first, excavated dedicated sediment was used to construct mounds as above, and then off-site material was dredged from a nearby canal ("direct dredge"), mixed with water to form a slurry, and backfilled around the mounds to create shallow (< 0.5 m depth) aquatic habitat. *Pumped* mounds were constructed by creating a dredge material slurry; this material was pumped into mound formations ("slurry and pump") that were surrounded by shallow water habitat (<0.50 m depth). A *reference* marsh with similar tidal influence was selected near the restored areas to act as a standard of comparison; this marsh was not actively managed or altered. The three mound construction methods varied in soil source, but initial soil tests suggested that soil texture and trace nutrient concentrations were similar among all soil sources (A.R. Armitage, unpublished data). Therefore, the primary functional difference among construction methods was the depth of the surrounding aquatic habitat.

We used georeferenced aerial site photos to randomly select ten sampling stations in each of the four habitat types (*excavated*, *filled*, *pumped*, *reference*). At each station, we measured a range of

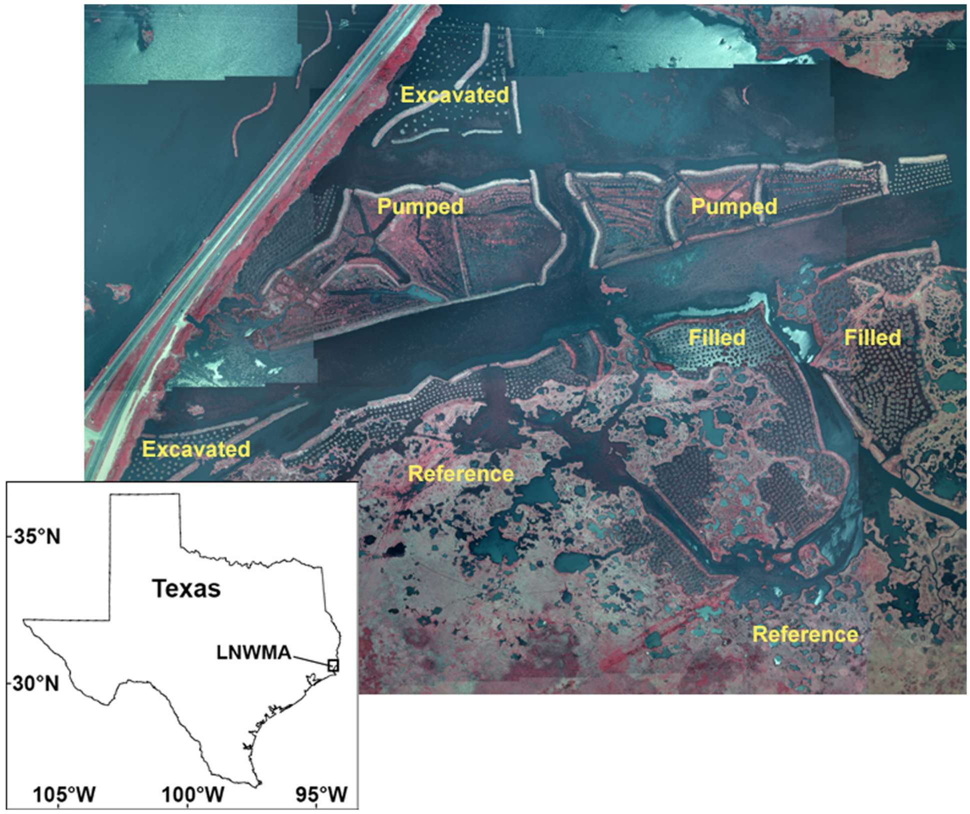

Fig. 1. Infrared aerial image taken during a flyover of the study site in the Lower Neches Wildlife Management Area (LNWMA) in September 2009. Representative areas of each of the three restoration construction methods (excavated, filled, pumped) and the reference marsh are labeled.

plant, soil, and water characteristics in summer (June–September) 2009 and 2010. Archived weather data were acquired from the NOAA National Climatic Data Center for the sampling dates to verify that climatic conditions were similar between years. Average air temperatures over the sampling periods were 30.0 °C in 2009 and 29.3 °C in 2010. The one-month cumulative rainfall at the site prior to the sampling events was 13.6 cm in 2009 and 14.2 cm in 2010. However, tidal patterns varied substantially between

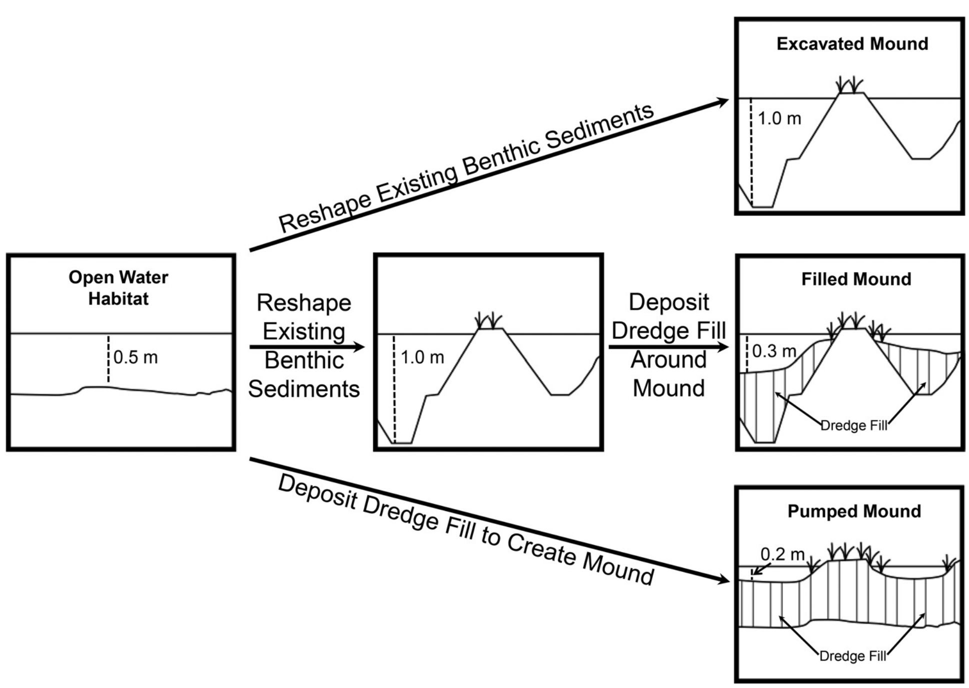

Fig. 2. Graphic of restoration construction methods. All mound types had similar elevations (0.2–0.4 m above mean sea level) and emergent habitat dimensions (1–3 m diameter). Dashed lines indicate approximate water depth at mean high tide.

years – in 2009, tidal height ranged from 10 cm below to 24 cm above mean sea level, but it ranged from 24 to 72 cm above mean sea level in 2010. Higher tides in 2010 were likely driven by low pressure from Tropical Storm Hermine, which was located in the southern Gulf of Mexico during that time period, although the eye of the storm was several hundred miles from our study site.

2.2. Emergent plant characteristics

A monopod camera extension was used to take a digital photo of each sampling station; the camera was parallel to the ground and one meter above the plant canopy. Percent live plant cover was calculated using the spectral analysis program VegMeasure (Oregon State University). All of the stems within a representative 10 cm × 20 cm quadrat were clipped at the sediment surface and brought back to the lab; stem density was calculated as stems/m^2. We then rinsed and dried all tissue and calculated aboveground biomass as dry weight (kg/m^2). We measured nitrogen content of *S. alterniflora* leaf tissue with a CHNSO analyzer and phosphorus content using a dry-oxidation acid hydrolysis extraction followed by colorimetric analysis of the extract (Fourqurean et al., 1992). We measured the chlorophyll *a* content of freshly collected *S. alterniflora* leaf tissue with a SPAD-502 portable leaf meter (Konica Minolta Corporation, USA). This device measures the transmission of 650 and 940 nm red light through leaves and is a reliable tool for inferring chlorophyll *a* content (Madrid et al., 2012a; Netto et al., 2002). We rinsed soil cores (7.5 cm diameter, 20 cm depth) through a 2-mm sieve, collected all root tissue, and calculated belowground biomass as dry weight (kg/m^3).

2.3. Aquatic plant characteristics

To determine submerged aquatic vegetation (SAV) biomass, a 16-tine metal rake was dragged over a one meter area extending perpendicular from the emergent vegetation line, covering an area of 0.082 m^2 (modified from Spears et al., 2009). This collection was repeated twice over the same area in order to ensure full collection of biomass. In the lab, SAV was sorted by species, dried, and weighed to determine biomass (g/m^2). Logistical constraints limited SAV collections to 3–5 stations per habitat type per year.

2.4. Soil characteristics

A soil core (4 cm diameter, 10 cm depth) was collected from each sampling station on each date. Grain size was determined using the hydrometer method of Bouyoucos (1962). Organic content was determined by loss on ignition. Soil nitrogen and phosphorus contents were measured as described above for plants.

2.5. Water characteristics

Surface (top 20 cm) salinity was recorded with a YSI 30 salinity/conductivity/temperature meter. Water samples were collected in acid washed polycarbonate bottles and kept in a cooler for transport to the laboratory. Samples were immediately filtered (GF/F; Whatman) onto a filter under low vacuum (<130 kPa) pressure. The filtrate was stored in an acid cleaned HDPE bottle (125 mL; Nalgene) which was triple rinsed with extra filtrate before keeping the final sample for analysis. Filtrate was frozen prior to dissolved nutrient (NO_3^-, NO_2^-, NH_4^+ and HPO_4^{3-}) analysis. Nutrient analyses were performed following standard procedures at the Geochemical and Environmental Research Group (GERG) located at Texas A&M University (College Station, TX). Dissolved inorganic nitrogen (DIN) concentrations were calculated by summing NO_3^-, NO_2^- and NH_4^+ concentrations.

Filters from the nutrient analysis were frozen at −80 °C for chlorophyll *a* analysis. Chlorophyll *a* concentrations were measured using a Turner 10-AU fluorometer according to Arar and Collins (1997) with several modifications. Filters were extracted with a 60/40 solution of 90% acetone/DMSO and kept overnight in the dark at 4 °C. Filters were removed and samples centrifuged for 5 min to remove any particulates. After measuring the initial fluorescence, samples were acidified with 10% HCl and the fluorescence measured a second time.

Total suspended solids (TSS) were measured using Method 2540 D of Standard Methods (APHA, 1998). Filters were pre-combusted (500 °C for 5 h) and pre-weighed. After filtration of a known volume of sample water, filters were dried in an oven at 60 °C for no less than 48 h and then reweighed.

2.6. Aquatic fauna characteristics

Aquatic fauna were collected using a throw trap following Kinney et al. (2013). Briefly, a 1-m^2 throw trap was deployed on aggregations of SAV, and the contents of the trap were sorted in order to remove all fish and invertebrates. All fish were euthanized on ice on site in accordance with TAMU Animal Use Protocol 2009-31. All animals were placed in coolers for transport to the lab, where they were sorted by species and tallied to determine density. Logistical constraints limited aquatic faunal collections to five stations per habitat type in 2009.

2.7. Data analysis

In order to identify differences among habitat types, and to assess whether restored areas were approaching reference conditions over time, we used a variety of multivariate analytical techniques (PRIMER v.6, PRIMER-E Ltd., Plymouth Marine Laboratory, United Kingdom). The permutational analyses in the PRIMER program do not require normal data distribution, but data for water nutrients, TSS, and stem density were log transformed to minimize the influence of outliers. All emergent plant, soil, and water data were then normalized to a common scale of −1 to +1, where the mean = 0. We performed two-way Analysis of Similarity (ANOSIM), based on a Euclidean resemblance matrix, where the factors were year (2009 and 2010) and habitat type (*excavated*, *filled*, *pumped*, and *reference*). We used MDS (nonmetric multidimensional scaling) ordination to represent average dissimilarities among years and habitat types in Euclidean two-dimensional space. We then used the SIMPER (SIMilarity PERcentages) routine as an exploratory technique to identify the variables that most strongly contributed to the MDS ordination.

Aquatic fauna and SAV had lower replication, precluding inclusion in the ANOSIM model with emergent plant, water, and soil characteristics. We analyzed total fish and invertebrate density and the biomass of two common SAV species separately with two-way permutational multivariate ANOVA (PERMANOVA+ add-on to PRIMER) based on a zero-adjusted Bray–Curtis resemblance matrix. This analysis does not assume normal data distribution, but we confirmed similar data dispersion in multivariate space across treatments with the PERMDISP routine. We also compared fish and invertebrate species assemblages between years and among habitat types with a two-way analysis of similarity (ANOSIM) based on a Bray–Curtis resemblance matrix. We used MDS ordination to represent these dissimilarities in Euclidean two-dimensional space.

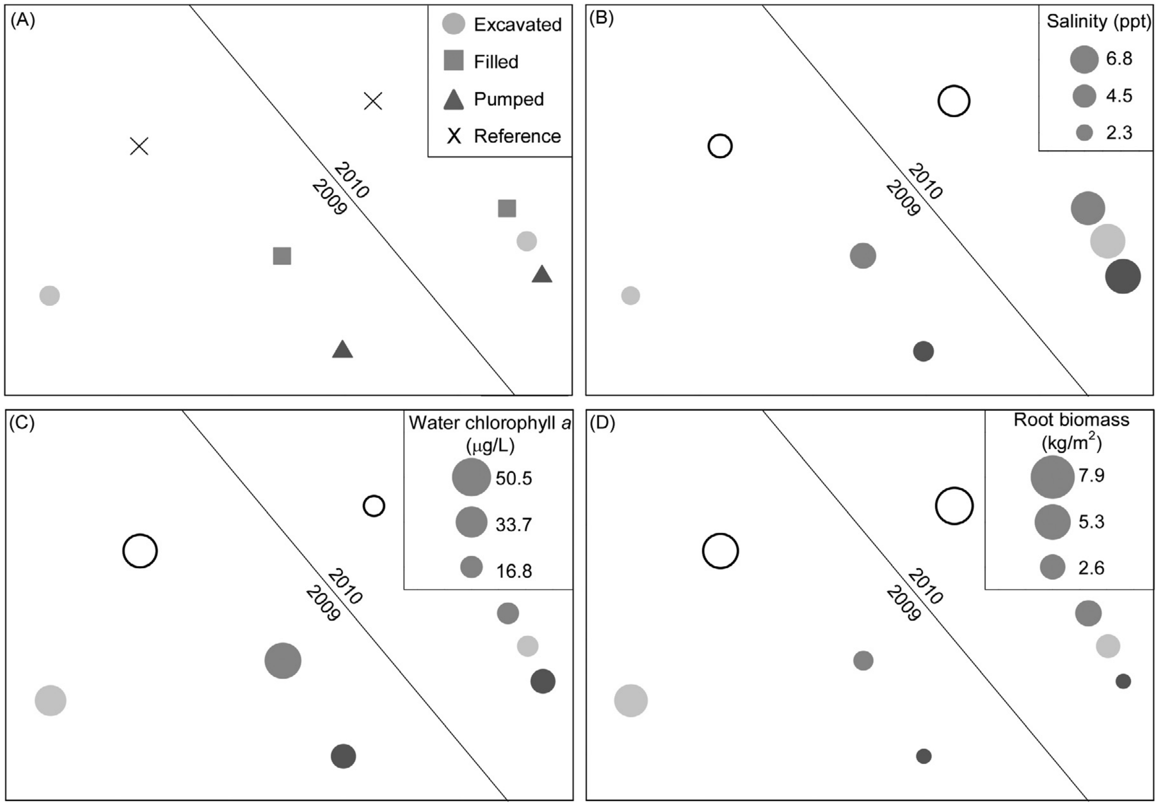

Fig. 3. (a) MDS ordination plot representing average dissimilarities among years and habitat types (excavated, filled, pumped, reference). (b–d) Bubble plots overlaid on MDS ordination; (b) water column salinity, (c) water chlorophyll *a* concentration, (d) belowground plant biomass.

3. Results

3.1. Ecosystem characteristics

The analysis of ecosystem characteristics that included emergent plant, water, and soil metrics revealed a larger difference between years than among habitat types. ANOSIM generates an *R* statistic that is essentially an indicator of effect size. For differences between years (2009 and 2010), the *R* value was 0.412 ($p = 0.001$), suggesting that ecosystem characteristics were relatively distinct between years. The MDS plot of average dissimilarities showed a clear separation between years, with low stress (0.01) (Fig. 3a). In contrast, the *R* statistic for habitat type was 0.257. Although this was significantly greater than zero ($p = 0.001$), this relatively low *R* value suggested a large amount of overlap among habitat types.

SIMPER analysis was used as an exploratory tool to identify the variables that most strongly contributed to the MDS ordination. The largest differences between years were found in water salinity, dissolved inorganic nitrogen concentration, and chlorophyll *a* concentration. Salinity was 2–3 times higher in 2010, likely caused by the unusually high tides driven by Tropical Storm Hermine (Fig. 3b, Table 1). This marine influence was also reflected in lower chlorophyll *a* content (Fig. 3c) and dissolved inorganic nitrogen (Table 1), both of which were lower by at least half in 2010.

Although there was a substantial amount of overlap among habitat types, SIMPER analysis suggested that some soil characteristics differed among habitat types, particularly between reference and restored areas. Plant belowground biomass was up to two times higher in reference sites than in all restored areas (Fig. 3d, Table 1). Similarly, reference soil nitrogen and phosphorus concentrations were higher than in all restored areas (Table 1).

3.2. Aquatic plant characteristics

The two most common SAV species were *Ruppia maritima* (widgeongrass) and *Myriophyllum spicatum* (Eurasian watermilfoil). In three of the four habitat types, *Myriophyllum* biomass was more than an order of magnitude higher in 2009 than in 2010, but there was no significant difference among restored or reference habitat types (Tables 1 and 2, Fig. 4a). *Ruppia* biomass was extremely variable, and was altogether absent from many sampling stations, particularly in 2009 (Fig. 4b). Although there was a significant difference in *Ruppia biomass* between years (Table 2), this effect was largely due to differences in data dispersion in multivariate space (PERMDISP, $p = 0.056$). In other words, the PERMDISP analysis indicates that variability in *Ruppia* biomass was not consistent between years.

3.3. Aquatic fauna characteristics

Fish and invertebrate densities were significantly higher in 2010 than in 2009 (Tables 1 and 2, Fig. 4c, d). The significant difference in invertebrate density between years was influenced by differences in data dispersion in multivariate space (PERMDISP, $p = 0.036$). This indicates that variability in invertebrate density was not consistent between years, though there were clearly many more invertebrates throughout the study area in 2010 (Fig. 4d). The most common fish species were rainwater killifish (*Lucania parva*), sailfin molly (*Poecilia latipinna*), and sheepshead minnow (*Cyprinodon variegatus*). The most common invertebrate species were a small hydrobid snail (*Probythinella protera*) and daggerblade grass shrimp (*Palaemonetes pugio*). A complete list of species collected is reported in Bell (2011).

Table 1
Average values (±SE) for each measured variable in restored (excavated, filled, pumped) and reference sites over two years.

	Emergent plant characteristics						
	Percent cover	Live stem density (#/m^2)	Aboveground biomass (kg/m^2)	Belowground biomass (kg/m^2)	*Spartina* leaf % nitrogen	*Spartina* leaf % phosphorus	*Spartina* chlorophyll *a* (SPAD)
2009							
Excavated	14.9 ± 3.8	785.0 ± 168.3	1.7 ± 0.6	4.1 ± 1.1	1.6 ± 0.2	0.17 ± 0.02	0.18 ± 0.02
Filled	38.9 ± 4.3	800.0 ± 87.1	2.7 ± 0.6	2.1 ± 0.7	1.4 ± 0.2	0.14 ± 0.02	0.22 ± 0.03
Pumped	57.8 ± 7.7	1437.5 ± 171.1	3.8 ± 0.6	0.9 ± 0.3	1.5 ± 0.1	0.13 ± 0.01	0.24 ± 0.01
Reference	52.2 ± 3.1	1405.0 ± 247.5	2.1 ± 0.4	5.8 ± 1.3	1.2 ± 0.1	0.09 ± 0.01	0.23 ± 0.02
2010							
Excavated	63.2 ± 9.1	930.0 ± 109.6	3.3 ± 0.5	1.4 ± 0.8	1.5 ± 0.1	0.13 ± 0.01	0.19 ± 0.02
Filled	68.8 ± 7.7	440.0 ± 54.7	2.4 ± 0.3	2.6 ± 0.5	1.6 ± 0.1	0.12 ± 0.01	0.23 ± 0.02
Pumped	54.3 ± 6.9	1245.0 ± 102.1	5.6 ± 0.8	0.8 ± 0.3	1.6 ± 0.1	0.17 ± 0.01	0.25 ± 0.01
Reference	55.3 ± 7.2	945.5 ± 336.7	2.3 ± 0.4	5.6 ± 0.6	1.4 ± 0.2	0.16 ± 0.02	0.28 ± 0.03

	Soil characteristics			
	% Sand content	Soil % organic content	Soil % total nitrogen content	Soil % total phosphorus content
2009				
Excavated	47.0 ± 6.5	10.5 ± 1.4	0.28 ± 0.05	0.016 ± 0.002
Filled	45.8 ± 2.9	9.1 ± 0.7	0.22 ± 0.02	0.021 ± 0.002
Pumped	31.3 ± 2.9	5.3 ± 0.3	0.11 ± 0.01	0.024 ± 0.002
Reference	58.6 ± 5.2	14.0 ± 1.4	0.42 ± 0.05	0.034 ± 0.004
2010				
Excavated	25.0 ± 4.3	11.0 ± 1.5	0.26 ± 0.05	0.018 ± 0.004
Filled	26.8 ± 4.7	13.0 ± 3.0	0.25 ± 0.06	0.020 ± 0.003
Pumped	28.0 ± 4.4	9.0 ± 2.5	0.15 ± 0.03	0.023 ± 0.002
Reference	52.1 ± 5.2	13.8 ± 1.8	0.34 ± 0.05	0.040 ± 0.002

	Water characteristics				
	Salinity (ppt)	Chlorophyll *a* (μg/L)	Total suspended solids (g/L)	HPO_4 (mg/L)	DIN (mg/L)
2009					
Excavated	2.4 ± 0.6	29.1 ± 5.2	0.05 ± 0.01	0.011 ± 0.006	0.098 ± 0.020
Filled	4.9 ± 0.2	40.3 ± 5.3	0.09 ± 0.05	0.004 ± 0.000	0.089 ± 0.010
Pumped	3.1 ± 0.3	19.1 ± 2.3	0.13 ± 0.10	0.008 ± 0.004	0.115 ± 0.035
Reference	4.4 ± 0.1	37.7 ± 2.8	0.06 ± 0.01	0.005 ± 0.001	0.070 ± 0.011
2010					
Excavated	9.1 ± 0.1	12.3 ± 3.1	0.07 ± 0.01	0.007 ± 0.001	0.039 ± 0.012
Filled	8.8 ± 0.1	12.6 ± 2.9	0.09 ± 0.02	0.012 ± 0.002	0.022 ± 0.005
Pumped	9.5 ± 0.1	14.5 ± 2.7	0.07 ± 0.01	0.008 ± 0.001	0.052 ± 0.012
Reference	7.2 ± 0.2	13.8 ± 1.6	0.05 ± 0.00	0.024 ± 0.002	0.044 ± 0.006

	Aquatic characteristics			
	Myriophyllum biomass (g/m^2)	*Ruppia* biomass (g/m^2)	Total fish density (g/m^2)	Total invertebrate density (g/m^2)
2009				
Excavated	562.1 ± 200.4	0.1 ± 0.1	18.8 ± 3.9	8.6 ± 2.5
Filled	572.6 ± 140.7	0	18.7 ± 4.1	13.0 ± 8.1
Pumped	319.5 ± 124.8	121.6 ± 121.6	24.8 ± 11.0	7.7 ± 3.6
Reference	543.4 ± 129.0	1.1 ± 0.7	39.2 ± 5.5	12.0 ± 4.7
2010				
Excavated	12.2 ± 3.4	64.2 ± 63.2	78.1 ± 33.6	338.2 ± 189.4
Filled	37.8 ± 29.4	2.1 ± 1.4	94.5 ± 21.7	555.3 ± 196.6
Pumped	243.4 ± 151.4	37.2 ± 30.4	49.2 ± 9.4	525.6 ± 117.0
Reference	46.7 ± 17.7	1.5 ± 1.5	53.8 ± 25.9	332.7 ± 155.1

The ANOSIM comparing fish community composition among treatments suggested a large amount of overlap between years ($R = 0.345$, $p = 0.001$) and habitat types ($R = 0.143$, $p = 0.002$). The p values indicate that these R statistics were significantly greater than zero, but the low (<0.4) absolute value of the R statistic suggests a small treatment effect size, and a large amount of overlap among all treatments (Fig. 5a). In contrast, the invertebrate community did not vary at all among habitat types ($R = 0.066$, $p = 0.055$), but differed markedly between years ($R = 0.685$, $p = 0.001$; Fig. 5b). This temporal difference was partly driven by a large increase in the hydrobid snail *P. protera* in 2010.

4. Discussion

The central goal of most restoration efforts is to improve ecosystem functions in order to compensate for habitat loss or degradation. Targeted ecosystem functions or characteristics can be broad (e.g., increase habitat area) or specific (e.g., provide habitat for a particular species) (Kentula, 2000). The restoration project in our study did not explicitly focus on any specific function or species, but rather sought to broadly improve the ecological integrity of the landscape. The goal of "ecological integrity" can be interpreted in different ways. If defined as similarity to reference conditions,

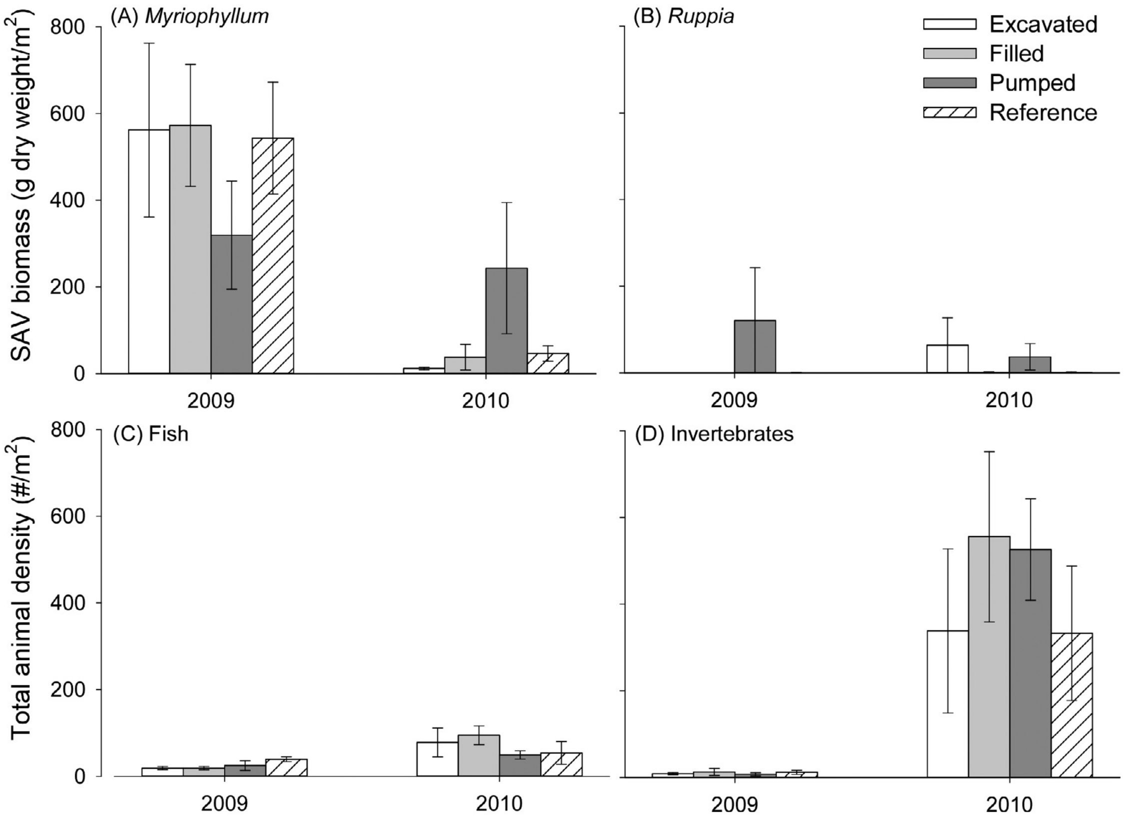

Fig. 4. Responses of submerged aquatic vegetation (SAV) and fauna to three restored (excavated, filled, pumped) and one reference habitat type in two years. (a) *Myriophyllum spicatum* biomass; (b) *Ruppia maritima* biomass; (c) total fish density; (d) total invertebrate density.

then the restored areas were similar but not identical to those target conditions; they differed primarily in terms of belowground plant and soil characteristics. There was no strong evidence that these characteristics of the restored areas were converging on reference conditions over the two-year study period, though the restored areas were still relatively young. Alternatively, ecological integrity can be defined as the provision of a wide range of biotic and abiotic features that increase the landscape-level heterogeneity of the ecosystem. In our study site, the diversity of construction approaches introduced topographical heterogeneity that integrated aquatic habitat into the design (Hough-Snee et al., 2011; O'Connell and Nyman, 2010), subsequently increasing the range of habitats available for biota at the site and contributing to the broad ecological success of this restoration project.

Table 2
Results of two-way permutational ANOVA (PERMANOVA) for the effects of year and habitat type on aquatic plant biomass and faunal densities.

	df	MS	Pseudo-*F*	*p*
***Myriophyllum spicatum* biomass**				
Year	1	16,720	18.01	0.001
Habitat type	3	654	0.71	0.621
Year × Habitat Type	3	1731	1.87	0.091
Residual	26	928		
***Ruppia maritima* biomass**				
Year	1	4547	4.79	0.018*
Habitat type	3	1046	1.10	0.337
Year × Habitat Type	3	786	0.83	0.528
Residual	26	949		
Total fish density				
Year	1	2579	5.48	0.014
Habitat type	3	318	0.68	0.625
Year × Habitat Type	3	504	1.07	0.347
Residual	45	470.4		
Total invertebrate density				
Year	1	28,930	33.07	0.001*
Habitat type	3	1846	2.11	0.066
Year × Habitat Type	3	906	1.04	0.394
Residual	45	875		

* Significant differences in dispersion (PERMDISP) among treatments.

Many previous studies have shown that terracing or backfilling approaches can augment and accelerate the restoration of plant and animal assemblages in wetlands (e.g., Hough-Snee et al., 2011; Schrift et al., 2008; Slocum et al., 2005; Stagg and Mendelssohn, 2010; Streever, 2000). However, there have been few previous opportunities to explicitly compare the ecological success of different construction approaches that utilize different soil sources and are engineered in different configurations. Our study provided the unique insight that although the restored areas did not achieve all reference characteristics during the early development of the site, the ecological differences among engineering approaches were relatively subtle. Most previous work evaluating the success of engineered wetland restoration projects has focused on comparisons to reference sites, and the degree of success often depends on project-specific goals. Terraced restoration sites seek to increase edge habitat for fishery species that rely on the marsh–water interface for food and shelter (Rozas et al., 2005). In this respect, engineered terraces are usually successful, with few discernible differences relative to reference areas (Feagin and Wu, 2006; Merino et al., 2010; Rozas et al., 2005; Rozas and Minello, 2001, 2009). In some cases, restored nekton populations may have lower diversity or a smaller size structure than in reference areas (La Peyre et al., 2007; Thom et al., 2004), yet overall fishery values are improved relative to unrestored areas. However, other ecosystem functions may not be as intact. For example, terraced projects often have a

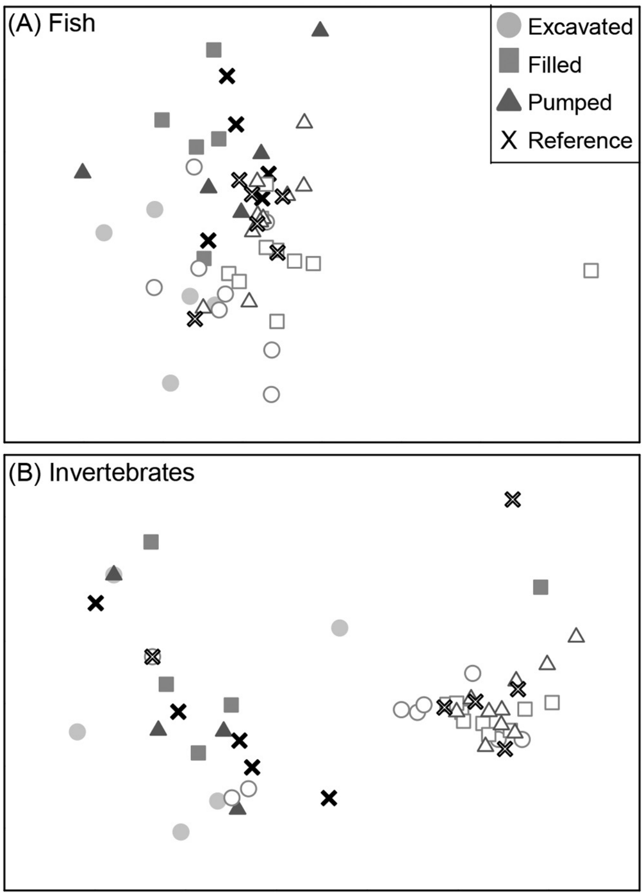

Fig. 5. Composition of aquatic fish (a) and invertebrate (b) assemblages in response to three restored (excavated, filled, pumped) and one reference habitat type in two years. The MDS ordination is a representation of dissimilarities among treatments based on a Bray–Curtis similarity matrix. Closed symbols: 2009; open symbols: 2010.

smaller spatial extent of emergent marsh and lower habitat diversity (Feagin and Wu, 2006), which ultimately decreases the capacity of restored marshes to sequester carbon (Madrid et al., 2012b).

The metrics most commonly used to assess restoration "success" are linked to the plant community, particularly plant cover (Cole and Shafer, 2002; Kentula, 2000). The plant canopy in our study developed rapidly – a common occurrence in restored marshes (Craft et al., 2003; Cui et al., 2009; Edwards and Proffitt, 2003; Webb and Newling, 1985). Our analysis indicated that plant metrics such as percent cover and aboveground biomass were not clearly linked to restoration status (restored vs. reference), construction method (*excavated, filled, pumped*), or year. Reference areas differed somewhat from the restored areas in terms of belowground biomass, though this distinction explained a relatively small portion of the statistical dissimilarity among habitat types. Lower belowground biomass is common in restored areas, where root systems have been shown to take five or more years to develop (Castillo et al., 2008; Edwards and Mills, 2005; Webb and Newling, 1985).

Soil characteristics, especially nitrogen and organic carbon pools, are usually very slow to develop in restored wetlands (Craft et al., 1999; Edwards and Proffitt, 2003; Langis et al., 1991; Lindau and Hossner, 1981; Minello and Webb, 1997; Staszak and Armitage, 2013). Based on this substantial body of previous work, and the variety of soil sources used in the construction of our restored site, we expected that soil characteristics would vary among construction types and between reference and restored areas. However, we were surprised to find that, based on the parameters we measured, the soil was largely homogeneous across all habitat types, though soil nutrient concentrations were somewhat higher in the reference area.

The homogeneity of the soils across habitat types may have been driven by the substantial hydrological modifications at the site. The LNWMA, part of the Gulf Coast Chenier Plain ecosystem, has been subject to Structural Marsh Management over the last several decades. Structural Marsh Management includes the installation and removal of levees, water control structures, and impoundments (Bolduc and Afton, 2004). The most recent alterations to the study site include the 1997 and 2007 installation of earthen plugs, intended to reduce tidal salt water input to the area and restore its brackish character. These hydrological alterations may regulate soil nutrient concentrations at the landscape level. Furthermore, the soils may have been homogeneous because the site was fairly eutrophic – nitrogen and organic content levels were high relative to many other sites in the region (Madrid et al., 2012a).

Water characteristics were also largely similar among habitat types. Despite the differences in water depth among construction methods, and between restored and reference areas, periodic tidal flushing through the extensive channel network appears to have been sufficient to homogenize water characteristics across habitat types within each year. There were some differences in water characteristics between years, particularly an increase in surface water salinity in 2010, though that change was small relative to the large drought-induced increase in salinity in 2011 (Kinney et al., 2013). The salinity measurements in our study were "snapshots" – one-time measurements that did not integrate conditions over entire growing seasons. However, local meteorological conditions were remarkably similar between sampling events in 2009 and 2010, and were within the normal range of past conditions, so it is unlikely that periods of low rainfall or high temperatures were responsible for the salinity increase. Rather, the salinity difference between years was attributable to a remote tropical storm that drove tides higher than normal during the 2010 sampling, overtopping the earthen plugs and flooding the study site with salt water. The concurrent decrease in chlorophyll *a* and dissolved inorganic nitrogen reflected that marine input (Howarth and Marino, 2006).

The frequent tidal inundation and salinity increase in 2010 was a likely driver of the concurrent changes in the aquatic plant and animal community. The aquatic plant *Myriophyllum* is sensitive to high salinities (Cho and Poirrier, 2005; Frazer et al., 2006), suggesting that the 2010 decrease in biomass may have been a negative response to salinity. However, *Myriophyllum* can sometimes tolerate salinities up to 15 ppt without substantial detriment to growth (Martin and Valentine, 2012), so other factors may have contributed to the *Myriophyllum* decline in 2010, such as the concurrent decrease in dissolved inorganic nitrogen.

Myriophyllum, *Ruppia*, and other SAV species provide food and refuge for a variety of aquatic fauna (Valinoti et al., 2011). Therefore, we expected that the large loss of SAV biomass in 2010 would correspond with a decrease in aquatic fauna. Rather, the opposite occurred – there was a large increase in invertebrate density and, to a lesser degree, fish density. This pattern was probably not a direct, positive salinity response, because all of the most common species had relatively wide salinity tolerances (Kinney et al., 2013). Our analysis of similarity, particularly for fish, showed a high degree of overlap between years, further suggesting that the faunal species in our site could tolerate higher salinity. The invertebrate assemblage showed more separation between years, but this was mostly driven by an increase in snails in 2010. The snails tend to prefer low salinities, but both years were within their documented salinity range (Brown et al., 2000). It seems likely that our sampling effort was inadvertently timed to correspond with a peak in snail abundance, possibly during a reproduction event, and that the increase in other invertebrates was more modest, similar to fish. Regardless, we argue that the faunal increase over time is not an indicator of recovery in the restored areas because the reference area exhibited a similar temporal pattern.

The different engineering approaches did not appear to have a strong influence on the ecological condition of our study sites, as measured by plant, soil, water, or faunal characteristics. We concede that our metrics were recorded at a relatively small, plot-level spatial scale. Some ecosystem functions, such as carbon sequestration, are more appropriately assessed at a landscape scale, and are likely to be influenced by landscape-level differences in restored site design, such as the ratio of emergent marsh to aquatic habitat (Madrid et al., 2012b). Faunal communities, including aquatic and emergent assemblages, may also be influenced by landscape design at a larger spatial scale (Murphy et al., 2012). Regardless, our dataset clearly demonstrated that on the near term, there was no marked difference among construction methods. Engineered mounds such as ours often provide some degree of topographical heterogeneity (Bruland and Richardson, 2005; Hough-Snee et al., 2011), which is generally considered to be an ecosystem benefit (e.g., Connell, 1978; Wiens, 1977). In our dataset, variability in multidimensional space represented some degree of ecological heterogeneity, though this heterogeneity could not be consistently linked to a specific engineering approach.

Based on the ecological similarities among engineering approaches, and the minimal differences in ecosystem characteristics between restored and reference areas, the primary implication for practice is that any given restoration project should choose the most cost-effective and logistically feasible engineering method for that locale. In our project, no direct records of construction cost are available, but local practitioner estimates suggest that all construction methods cost between approximately $11–33 per cubic yard of sediment placed, a range similar to other intertidal wetland projects (Yozzo et al., 2004). The *filled* and *pumped* approaches at our site eventually began to fill in and create more emergent marsh habitat (Madrid et al., 2012b). Therefore, if the specific goal of a restoration project is to create emergent marsh, then either of those methods is acceptable and should be selected based on site-specific equipment and labor costs. However, the aquatic habitat in the *excavated* area also provided a critical ecological niche for fish, invertebrates, and submerged aquatic vegetation (La Peyre et al., 2007; O'Connell and Nyman, 2010). This type of heterogeneity, including emergent and aquatic habitat, may be important at a larger landscape scale (Bell et al., 1997; Hough-Snee et al., 2011). Therefore, we recommend the use of the single most cost-effective technique within a site, but the use of different engineering approaches at different sites within a region may confer an important, larger-scale ecological benefit.

5. Conclusions

The restored areas did not achieve all reference characteristics during the early development of the site, particularly in terms of belowground plant biomass and soil nutrient concentrations. However, the differences among engineering approaches within the restored area were relatively subtle. Therefore, the recommendation for practice is to use the construction approach that is most cost-effective and logistically feasible for a specific site. In our study area, the dredge slurry methods yielded the largest area of emergent marsh per unit effort, but the on-site soil excavations created more aquatic habitat. Using multiple engineering and integrating aquatic habitat into the design yielded topographic and habitat heterogeneity across the wetland landscape. When ecological integrity is defined as the provision of a wide range of biotic and abiotic conditions at a landscape scale, then the use of different engineering approaches at different sites within a region created habitat heterogeneity, thus conferring regional-level ecological benefits.

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Acute Effects of Drought on Emergent and Aquatic Communities in a Brackish Marsh

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Abstract Plants and animals in brackish marshes are adapted to live within a wide, yet finite, range of conditions. Events that shift the environmental state beyond that range can dramatically alter habitats and, potentially, the numerous ecosystem services they provide. A prolonged exceptional drought in Texas (October 2010–January 2012) provided a unique opportunity to understand how brackish wetland habitats respond to an extreme environmental event. We examined marshes in the Lower Neches Wildlife Management Area (Texas, USA) that fell within the drought affected area, including restored areas and an adjacent reference marsh. To test our hypothesis that the brackish marsh community would be sensitive to drought conditions, we quantified emergent plant and submerged aquatic vegetation (SAV) and animal (invertebrates, fish) characteristics in summer 2010 and 2011. In spite of its severity, the exceptional drought of 2011 did not have a negative impact on emergent plant communities: biomass, stem density, and chlorophyll *a* concentrations were the same in pre-drought and drought years in all restored and reference areas. In contrast, SAV biomass was reduced by up to 100 % in the drought year. Some fish and invertebrate densities were also reduced by an order of magnitude or more, possibly due to the loss of SAV. Aquatic faunal species composition was markedly different in the drought year, largely due to the loss of the hydrobiid snail *Probythinella protera* and the gain of some marine species, including Gulf menhaden (*Brevoortia patronus*), brown shrimp (*Farfantepenaeus aztecus*), and white shrimp (*Litopenaeus setiferus*). By altering aquatic the plant and animal community, this drought event may subsequently reduce trophic support for higher consumers, or contribute to a decline in water quality. Restoration monitoring programs that only focus on relatively stress-resistant, emergent wetland plant communities may underestimate the sensitivity of these ecosystems to extreme environmental events like droughts.

Keywords *Spartina alterniflora* cv. Vermilion · *Ruppia maritima* · *Myriophyllum spicatum* · Clupeidae · Penaeid shrimp · Restoration · Texas · Gulf of Mexico · Chenier Plain

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Introduction

Among the many factors that influence community structure in marsh habitats, the effects of extreme environmental events (e.g., drought, hurricanes or tropical storms) are arguably among the most dramatic (Ciais et al. 2005). The magnitude and timescale of recovery from extreme events is typically linked to the type and duration of impact. Short-term fluctuations (days–weeks) in weather conditions, even when extreme, often have temporary effects on plant/animal communities. Responses to short-term, stochastic periods of rainfall, for example, may involve fluctuations between wet and dry-tolerant plants (Wetzel and Kitchens 2007; Forbes et al. 2008). More prolonged environmental events (e.g., 1+ year) can have substantial long-term effects, especially if certain key species do not recover (Cho and Poirrier 2005) or if succession patterns are altered (Parsons et al. 2006), potentially modifying community structure and decreasing biodiversity (Nielsen and Brock 2009; Zedler 2010). Environmental events of particular concern are those that may be increasing in frequency and severity in conjunction with climate change, particularly droughts (Meehl and Tebaldi 2004).

Drought impacts are well documented in many terrestrial and riparian habitats, where there can be dramatic, negative impacts on primary production and aquatic and terrestrial food webs (Hinckley et al. 1979; Ciais et al. 2005; Livingston 2007). In coastal habitats, the effects of drought on ecosystem functions may be equally severe. Although it is normal for estuaries to experience predictable and stochastic variations in temperature and salinity, drought events can dramatically reduce freshwater input to estuaries, causing sudden and/or prolonged increases in salinity. Hence, the areas within estuaries that are more heavily dependent on freshwater, such as the brackish tidal marshes near the inland extent of tidal influence, are most likely to be affected by droughts. Drought in estuaries has been associated with large-scale dieback (McKee et al. 2004; Hughes et al. 2012), loss of diversity (Nielsen and Brock 2009; Zedler 2010), and shifts in plant and animal communities (Gascón et al. 2007; Forbes et al. 2008; Wedderburn et al. 2012). Drought induced stress has also led to species composition shifts in wetland fish (Martinho et al. 2007), seagrass (Cho and Poirrier 2005; Koch et al. 2007), and emergent plant communities (Visser et al. 2002; White and Alber 2009), and likely contributed to habitat loss as a result of marsh dieback in Georgia, South Carolina and Louisiana, USA (Alber et al. 2008). However, much of this previous work has focused on marine wetland habitats, and there is less quantitative information about community-level responses of brackish marshes to drought conditions. Furthermore, few studies have examined the effects of drought on both emergent and aquatic assemblages in these brackish wetland habitats.

Droughts impacting coastal habitats occur with some regularity on the Gulf of Mexico coast (Montagna et al. 2009), but 2011 brought record-breaking drought conditions to the region, particularly to Texas. Most of Texas, including the Gulf of Mexico coastal ecoregion, experienced an "exceptional drought" or D4, the most severe classification by the U.S. Drought Monitor (Tinker et al. 2011), which is equivalent to less than −5 on the Palmer Drought Severity Index (http://droughtmonitor.unl.edu/classify.htm). By October of 2011, almost 88 % of Texas was classified as experiencing exceptional drought (Nielsen-Gammon 2011). This classification is defined by widespread crop/pasture losses and shortages of water in reservoirs, streams and wells. Some areas would have ranked even higher if the U.S. Drought Monitor range was extended past D4 (Travis County 2011). In addition to being dry, it was also the hottest summer on record with average temperatures from June through August ≥ 3 °C above the long-term average (Nielsen-Gammon 2011). Widespread tree and vegetation die-offs contributed to a record fire season in Texas (Travis County 2011). The cumulative damage on native plant and animal species is still being studied, including the effects on wetlands and coastal systems. Preliminary data show that freshwater flow at gauged rivers was the lowest ever recorded (including the record-setting drought of 1953–1954) (Carla Guthrie, Texas Water Development Board, pers. comm.) and a decline in *Spartina alterniflora* abundance was observed on the Nueces River Delta (Stachelek 2012).

As part of a long-term wetland restoration monitoring project (est. 2009) in a brackish marsh, we quantified emergent plant and aquatic plant and animal community structure before and during the 2011 drought event. A secondary objective was to compare the effects of drought on restored and reference coastal marshes. We hypothesized that changes in salinity would lead to changes in emergent vegetation, SAV, and aquatic fauna (invertebrates, fish) abundance and community composition. We also expected drought effects to be more pronounced on newly restored areas than on more established reference marshes.

Methods

Our study site was an area of restored and reference brackish marshes within the Texas Parks and Wildlife Department's Old River Unit of the Lower Neches Wildlife Management Area (LNWMA) (Texas, USA, 30°00 N, 93°51 W; Fig. 1). The hydrological rehabilitation of the restored area was completed in 2007, following the installation of a series of culverts and earthen plugs to restrict salt water inflow. In 2008, construction of emergent marsh habitat resulted in three distinct habitat types: "Excavated" mounds were created from dedicated sediment (adjacent to the mound construction) and surrounded by a water depth of 1–2 m (Restored 1); "Filled" mounds were constructed with dedicated, excavated sediment that was then surrounded by off-site dredge material, creating shallow (<0.5 m) aquatic habitat (Restored 2); "Pumped" mounds were created with off-site dredge material using a backfilling technique, and were surrounded by shallow water habitat (0.25–0.50 m depth) (Restored 3). All of the restored areas were planted with *Spartina alterniflora* cv. Vermilion. A reference marsh with similar tidal influence and remnant brackish marsh vegetation was selected near the restored areas to act as a standard of comparison; this marsh was not actively managed or altered (see Fig. 1). Reference marshes contained relatively dense stands of salt marsh vegetation such as *Spartina alterniflora*, *Spartina patens*, *Distichlis spicata*, *Schoenoplectus robustus*, and *Iva frutescens*.

To verify local drought conditions, archived weather data were acquired from the NOAA National Climatic Data Center for Port Arthur, Texas for 2009, 2010, and 2011. Total annual rainfall in 2011 was nearly half of normal conditions, and average summer maximum temperatures were 2.0 °C higher than the long-term average (Table 1). These data confirmed

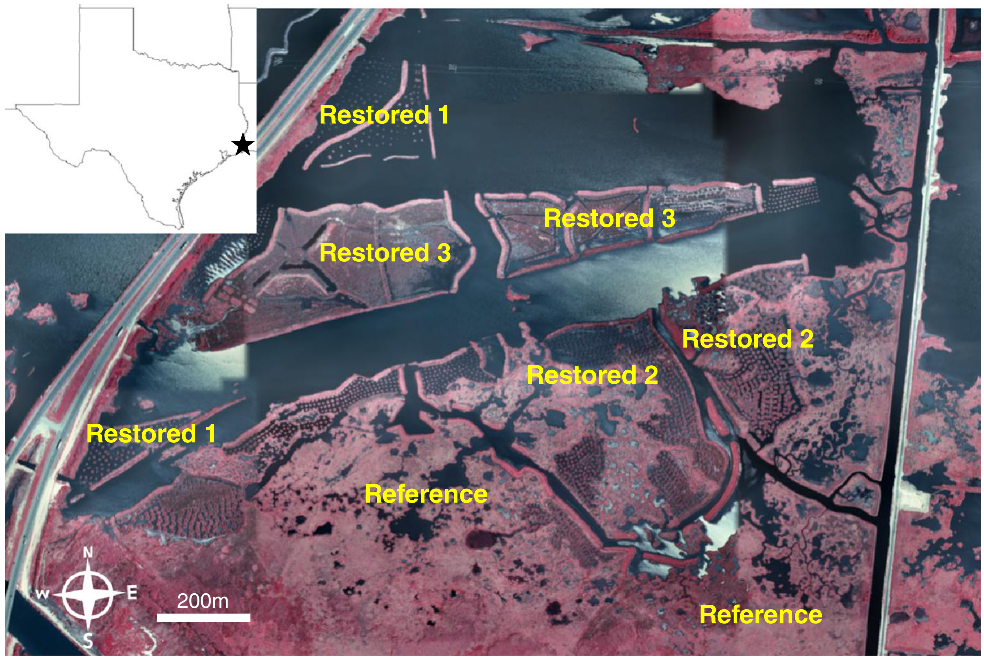

Fig. 1 Aerial view of wetland restoration site and adjacent reference area in the Lower Neches Wildlife Management Area, Texas, USA, taken August 2011 (30°00 N, 93°51 W). *Inset*: outline of Texas; *star* marks the approximate location of study site

the local severity of the statewide drought conditions reported by Nielsen-Gammon (2011). As part of a monitoring program in the restored marshes, we established ten randomly located sampling stations within each of the three restored areas and one reference area. Each station included an area of emergent marsh at least 7 m^2 in area and adjacent aquatic habitat. At each station, we measured characteristics of the emergent and aquatic communities. Surface water salinity was recorded quarterly using a YSI 85 probe between January 2010 and January 2012.

Emergent vegetation was clipped at the sediment surface from within a 10×20 cm quadrat located one meter from the marsh–water interface in September 2010 (pre-drought) and September 2011 (drought). Plants were sorted by species, and then heights and stem densities were recorded. Live plant tissue was washed and dried at 60 °C and weighed to determine biomass. Chlorophyll *a* concentration was measured in the field using a SPAD-502 portable leaf meter (Konica Minolta Corporation, USA) on live *S. alterniflora* leaves. This device measures the transmission of 650 and 940 nm red light through live leaves and is a reliable tool for inferring chlorophyll *a* content (Markwell et al. 1995; Bullock and Anderson 1998; Netto et al. 2002). Measurements were converted to mg chlorophyll *a* per mg leaf according to the relationship established by Madrid et al. (2012).

SAV was sampled in 2010 and 2011 by dragging the head of a 16-tine metal rake twice over a one meter area extending perpendicular from the marsh vegetation–water interface, covering an area of 0.082 m^2 (modified from Spears et al. 2009). The SAV included both *Ruppia maritima* (widgeongrass) and *Myriophyllum spicatum* (Eurasian watermilfoil). In the lab, SAV was sorted by species, dried at 60 °C and weighed to determine species-specific biomass.

Aquatic fauna were collected twice in the summer of each year (June and September). Fauna were sampled with a throw trap, which consisted of two horizontal square 1-m^2 frames (one PVC frame and one metal rod frame), connected in parallel by a fine mesh net. Each sampling station was approached by an airboat drifting towards the station with the motor turned off. The trap was tossed from the airboat into the water on top of a haphazardly selected SAV aggregation at each station. A two-person team entered the water behind the trap and slid another square mesh net underneath the throw trap; the whole trap and its enclosed contents were raised on to the deck of the airboat for cleaning and sorting. All fishes were removed from the vegetation on-site and placed in plastic bags containing water from the study site that was chilled to <4 °C. These bags were placed in a large cooler containing an ice slurry to euthanize all fishes (in accordance with TAMU Animal Use Protocol 2009–31). All samples were transported back to the lab in coolers and frozen pending further analysis. In the lab, all invertebrates that were

Table 1 Annual rainfall and average summer temperature maxima for pre-drought and drought years

	Annual rainfall (cm)	Rainfall deviation from normal (cm)	Average maximum summer temperature (°C)	Temperature deviation from normal (°C)
2009	151.3	−0.8	33.6	0.8
2010	118	−34.1	33.7	1.3
2011	78.8	−74.8	35.6	2.0

not removed in the field were manually removed from the SAV and tallied.

SAV biomass, total fish density, and total invertebrate density were log transformed due to unequal variances. Monte Carlo analysis was performed on SAV biomass due to lack of normality, and two-way mixed model analyses of variance (ANOVA) were performed on emergent vegetation characteristics, total fish density, and total invertebrate density, where the factors were year (pre-drought September 2010 and during drought September 2011) and habitat type (reference and three restored habitat types). We also used two-way ANOVA with habitat type and year to examine changes in density of the five most common faunal species: a small hydrobiid snail (*Probythinella protera*), daggerblade grass shrimp (*Palaemonetes pugio*), rainwater killifish (*Lucania parva*), sailfin molly (*Poecilia latipinna*), and sheepshead minnow (*Cyprinodon variegatus*).

To examine the similarity of faunal assemblages between years and among habitat types, we used PRIMER v6 (PRIMER-E Ltd., Plymouth Marine Laboratory, UK) to create a Bray–Curtis similarity matrix among treatments. We then used two-way analysis of similarity (ANOSIM) to identify differences in faunal community composition, where the factors were year (September 2010 and September 2011) and habitat type (reference and three restored habitat types). We used NMDS (nonmetric multidimensional scaling) ordination to represent these dissimilarities among years and habitat types in Euclidean two-dimensional space based on the similarity matrix.

Results

Emergent Vegetation

Average water column salinity (psu) in the study site increased from 8.8 in 2010 to 22.1 in September 2011 (Fig. 2). Emergent vegetation biomass, stem density, and plant fitness as estimated by *Spartina alterniflora* chlorophyll *a* concentration, were not significantly different between pre-drought (2010) and drought (2011) years (Fig. 3a,b,d; ANOVA all $p>0.1$). Emergent vegetation canopy height was about 25 cm shorter in 2011 than in 2010 (Fig. 3c; $p=0.02$). There were no significant differences in vegetation characteristics among restored and reference areas (all $p>0.05$).

Submerged Aquatic Vegetation

Myriophyllum spicatum biomass averaged 110 kg/m^2 in September 2010 but was absent in September 2011 (Monte Carlo $p<0.001$; Fig. 4a). There was no difference in *M. spicatum* biomass among restored and reference areas (Monte Carlo $p=0.11$). *Ruppia maritima* also had very low biomass in the

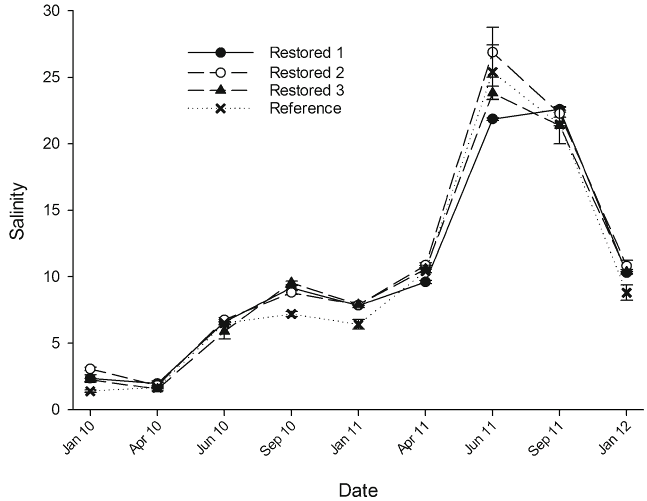

Fig. 2 Average (±SE) salinity at restored and reference sites from January 2010 to January 2012

drought year (Fig. 4b). However, *R. maritima* was less common relative to *M. spicatum*, so there was no significant difference in *R. maritima* biomass between years and no significant difference among restored and reference areas (Monte Carlo, all $p>0.05$).

Fauna

Total fish density decreased by an order of magnitude in the drought year (2011) compared to the pre-drought sampling (2010) (Fig. 5; ANOVA, $p<0.01$) but did not differ significantly among restored and reference areas ($p=0.75$). The most common species, *L. parva* and *P. latipinna*, decreased significantly (all $p<0.01$), while *C. variegatus* decreased, but not significantly ($p=0.34$) (Table 2). Total invertebrate density decreased by nearly 99 % between 2010 and 2011 (Fig. 6; $p=0.02$) but did not differ among restored and reference areas ($p=0.31$). *Palemonetes pugio* decreased by nearly 90 % in 2011 ($p<0.01$). The snail *P. protera* decreased significantly from thousands of individuals per square meter in 2010 to none in 2011 ($p<0.01$). When *P. protera* was removed from the analysis, there was still an 85 % decline in invertebrate abundance between 2010 and 2011, though that result was only marginally significant, likely due to high variability in 2010 (Fig. 6, inset; $p=0.07$).

Total species richness was similar between years; 20 species (eight invertebrate and 12 fish) were present in summer 2010 and 22 species (five invertebrate and 17 fish) were present in summer 2011 (Table 2). Eleven species occurred in both years, but there were several species that were gained or lost in 2011. Nine species (five invertebrate and four fish) were lost in 2011, and 11 species (two invertebrate and nine fish) were gained (Table 2). Many of these species gains or losses occurred in species that were uncommon, but notable

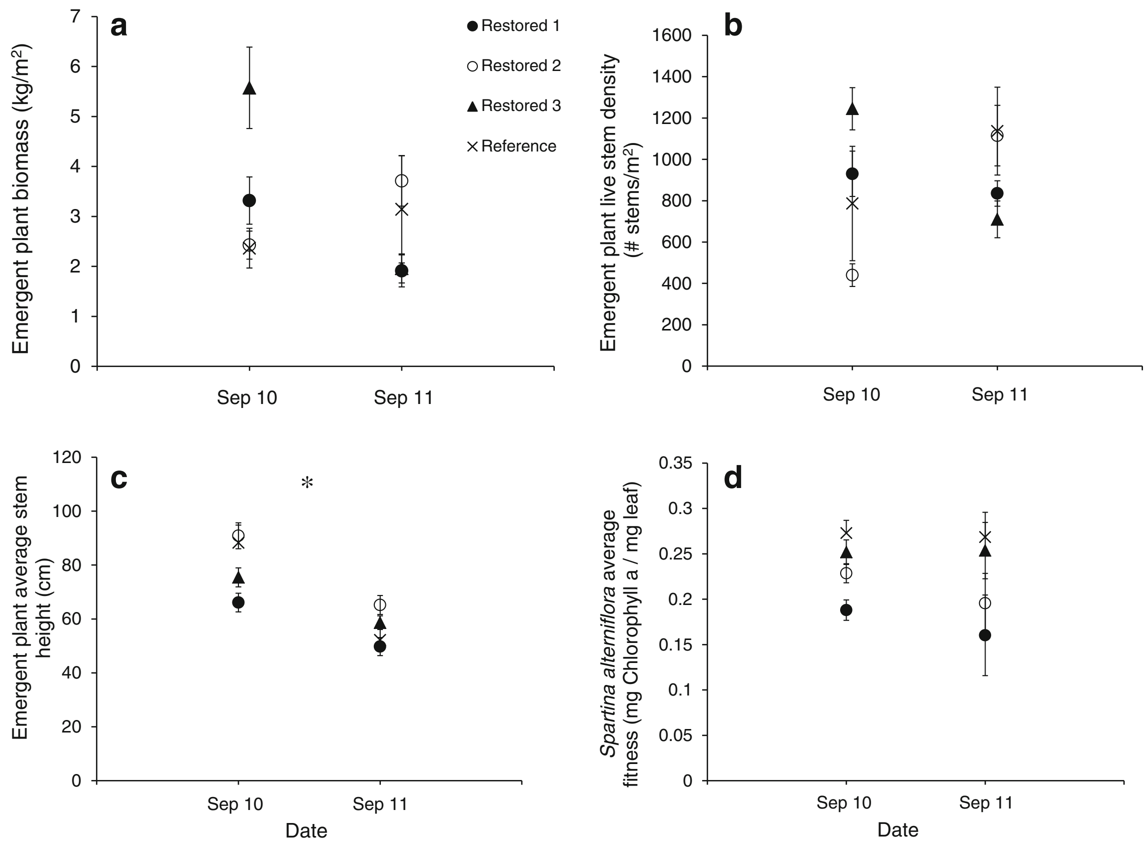

Fig. 3 Emergent plant characteristics (average ± SE) in restored and reference areas in pre-drought (2010) and drought (2011) years. **a** Biomass, **b** stem density, **c** average stem height, **d** *Spartina alterniflora* fitness, as estimated by chlorophyll *a* concentration. *Asterisk* (*) indicates significant differences between years

exceptions included the loss of the extremely abundant hydrobiid snail *P. protera*, and the gain of several marine species, Gulf menhaden (*Brevoortia patronus*), brown shrimp (*Farfantepenaeus aztecus*), white shrimp (*Litopenaeus setiferus*). These shifts in species composition were reflected in the ANOSIM analyses, which generate an *R* statistic that indicates the degree of overlap among communities (Clarke and Warwick 2001). Aquatic faunal assemblages in pre-drought (September 2010) and drought (September 2011) years were significantly distinct from each other, with a high R value ($R=0.628$, $p=0.001$), suggesting little overlap between years. The NMDS plot showed a clear separation between years, with low (0.12) two-dimensional stress (Fig. 7). Among communities in restored and reference areas, the R statistic was relatively low ($R=0.174$). Although significantly greater than zero ($p=0.001$), this low *R* value suggested a large amount of overlap among restored and reference habitat types.

Discussion

The severity of the drought was spatially uniform; salinity nearly tripled in the water column throughout our study site, and soil salinities were likely high as well due to the lack of rainfall and above-normal temperatures. Accordingly, drought stress had similar impacts on the plant and faunal communities in all restored and references areas. None of the restoration methods (excavated, pumped, filled) offered a clear advantage under exceptional drought conditions, nor a disadvantage relative to the reference marsh. However, there were variations in the drought effects among subhabitats. In particular, aquatic plant and animal communities were more strongly affected by drought than were emergent communities.

In spite of its severity, the exceptional 2011 drought had few strong effects on the emergent plant community. Although we did not examine porewater conditions at our site, the salinity range of tidal water during the drought was well within the tolerance range (up to 50) for *S. alterniflora* (Walkup 1991). Unlike the drought conditions that led to emergent marsh dieback in Louisiana (McKee et al. 2004), lack of flushing by rainfall in our study area did not appear to be detrimental to the emergent plant community. Other instances of *S. alterniflora* dieback attributed to extreme drought have followed drought periods exceeding more than 1 year (Swenson et al. 2004; Alber et al. 2008). In our case, the most severe drought conditions lasted for about a year. As a result, emergent vegetation biomass, density, and

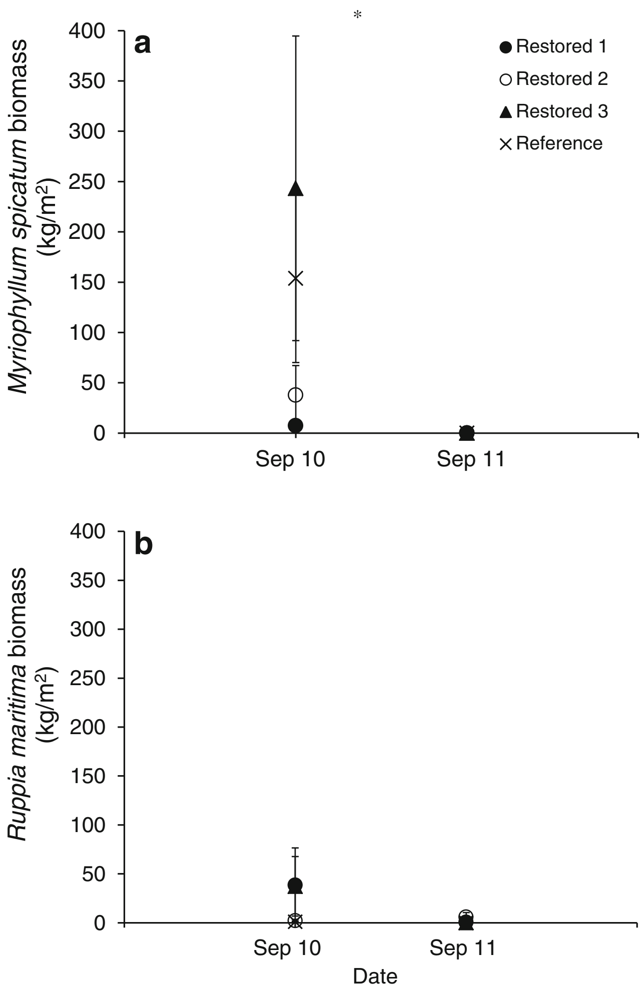

Fig. 4 Average (±SE) biomass in September 2010 and 2011: **a** *Myriophyllum spicatum*, **b** *Ruppia maritima*. *Asterisk* (*) indicates significant differences between years

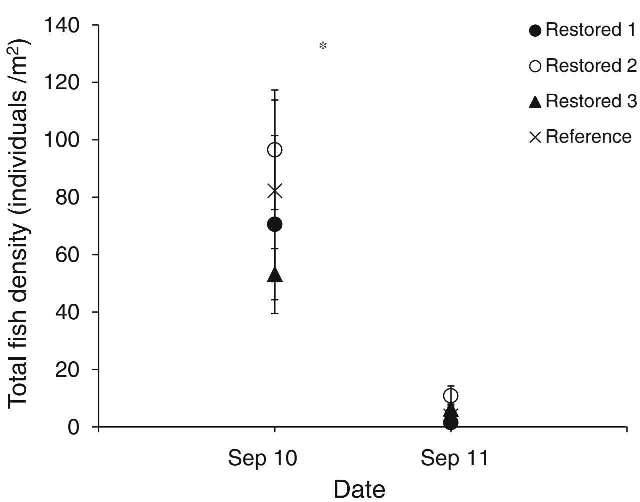

Fig. 5 Average (±SE) total fish density in September 2010 and 2011. *Asterisk* (*) indicates significant differences between years

fitness at our site showed few negative responses during the drought year, suggesting that there may be few long-term drought effects on the emergent plant community.

Overall, it appeared that both native vegetation and the *S. alterniflora* cv. Vermilion planted at our site were extremely drought tolerant, but results from other studies on height and biomass response of *S. alterniflora* to drought conditions are conflicting. Differences in canopy height in response to changes in rainfall have been attributed to reduced soil salinity (Wiegert et al. 1983) and elevated sulfide concentrations (King et al. 1982). However, Charles and Dukes (2009) found that experimental warming and drought conditions in a New England salt marsh increased *S. alterniflora* biomass but did not alter porewater chemistry. Stachelek (2012) found that emergent plant composition can be used as an indicator of drought and that *S. alterniflora* decreased during drought conditions, but the effects were slow to manifest. Some studies have documented shifts to more drought tolerant species (Miller et al. 2005; Forbes and Dunton 2006; Forbes et al. 2008; Nielsen and Brock 2009; Stachelek 2012), but our restored and reference areas were already dominated by salt marsh species that did not change in relative abundance in response to drought. Furthermore, both reference and restored salt marsh vegetation exhibited resistance to the extreme drought conditions in Texas, indicating that the native *S. alterniflora* in the reference plots was as hardy as the *S. alterniflora* cv. Vermilion planted in the restored plots.

Increased salinity had a much larger effect on the aquatic community than on the emergent plants. The dominant SAV species, *Myriophyllum spicatum*, transitioned from ubiquitous in 2010 to completely absent in 2011. Though *Ruppia maritima* was never abundant in our study area, it was also nearly absent in 2011. *Myriophyllum spicatum* is sensitive to salinities higher than 15, while *R. maritima* has been found to be more tolerant of elevated salinity (Merino et al. 2009). Cho and Poirrier (2005) found that when salinity increased in nearby Lake Ponchartrain, Louisiana, that *M. spicatum* declined and *R. maritima* became dominant. As salinities fell back to the normal range, *R. maritima* declined and *M. spicatum* recovered. Drought-induced shifts in aquatic plant communities can be temporary (Wetzel and Kitchens 2007) or result in long-term elimination of some species from the system and a potential shift in associated waterfowl (Cho and Poirrier 2005; Shili et al. 2007). Elimination of SAV can negatively impact water clarity and light penetration, slowing SAV recovery and potentially changing predator–prey dynamics (Moore 2004; Chaplin and Valentine 2009). Ongoing recovery of both SAV species will be monitored, as the shift away from the dominant *M. spicatum* could alter future SAV species composition and therefore the value of the SAV canopy as food and refuge for associated fauna (Valinoti et al. 2011).

Table 2 Total abundance of all faunal species across ten sampling stations in summer (June and September 2010 and 2011), in alphabetical order

	Scientific Name	Common Name	2010	2011	Species lost in 2011	Species gained in 2011
Invertebrates	Amphipoda	Amphipod	171	0	×	
	Anisoptera	Dragonfly larvae	10	0	×	
	Callinectus sapidus	Blue crab	7	3		
	Corixa spp.	Water boatmen	36	0	×	
	Farfantepenaeus aztecus	Brown shrimp	0	5		×
	Farfantepenaeus duorarum	Pink shrimp	15	16		
	Hydrobiomorpha casta	no common name	14	0	×	
	Litopenaeus setiferus	White shrimp	0	35		×
	Palaemonetes pugio	Daggerblade grass shrimp	983	295		
	Probythinella protera	Marsh snail	9491	0	×	
Fish	*Anchoa mitchilli*	Bay anchovy	0	1		×
	Bairdiella chrysoura	American silver perch	0	1		×
	Brevoortia patronus	Gulf menhaden	0	26		×
	Cyprinodon variegatus	Sheepshead minnow	158	153		
	Dorosoma cepedianum	Gizzard shad	5	0	×	
	Fundulus grandis	Gulf killifish	3	1		
	Fundulus jenkinsi	Saltmarsh topminnow	16	0	×	
	Fundulus pulvereus	Bayou killifish	0	2		×
	Fundulus similis	Longnose killifish	1	0	×	
	Gobiosoma robustum	Code goby	0	12		×
	Ctenogobius boleosoma	Darter goby	0	2		×
	Gobiosoma bosc	Naked goby	16	3		
	Lagodon rhomboides	Pinfish	0	1		×
	Lucania parva	Rainwater killifish	854	97		
	Menidia beryllina	Inland silverside	27	20		
	Microgobius gulosus	Clown goby	13	2		
	Mugil curema	Silver mullet	1	0	×	
	Ophichthus gomesii	Shrimp eel	0	1		v
	Poecilia latipinna	Sailfin molly	1011	203		
	Stellifer lanceolatus	Star drum	0	1		×
	Syngnathus scovelli	Gulf pipefish	14	1		

Species lost or gained in the drought year are noted

The largest drought-related change in animal density occurred in marsh snails (*P. protera*) and grass shrimp (*P. pugio*). *Probythinella protera* were abundant in 2010, reaching densities over 200 m^{-2}, but they were completely absent in 2011. High salinity is the most likely explanation for the disappearance of these snails, as they are sensitive to salinities greater than 25 (Brown et al. 2000). Grass shrimp (*P. pugio*) can tolerate a range of salinities from 0 to 44, but *P. pugio* densities declined by an order of magnitude in 2011. This was most likely an indirect drought response: high salinity caused die-off of SAV, and the subsequent loss of SAV habitat refuge may have resulted in increased predation (Duffy and Baltz 1998; Hitch et al. 2011; Valinoti et al. 2011).

The most abundant fish species (*Lucania parva*, *Poecilia latipinna*, and *Cyprinodon variegatus*) densities were reduced by over 70 % in the drought year. All of these species can tolerate a range of salinities from 0 to 44 and higher for *C. variegatus* (Gelwick et al. 2001), suggesting that the decrease in density was not a direct response to higher salinity. As in the case of *P. pugio*, SAV loss could have contributed to increased predation, or these motile species may have moved to refugia outside the study area where surviving SAV remained.

In addition to lower abundances, we also found several novel marine species appearing in 2011, most notably brown shrimp (*F. aztectus*), white shrimp (*L. setiferus*), and Gulf menhaden (*B. patronus*). Sabine Lake surveys by Texas Parks and Wildlife Department suggest that menhaden were 24 % more abundant in 2011 than in previous years (http://www.st.nmfs.noaa.gov/.txt). These shrimp species were even more

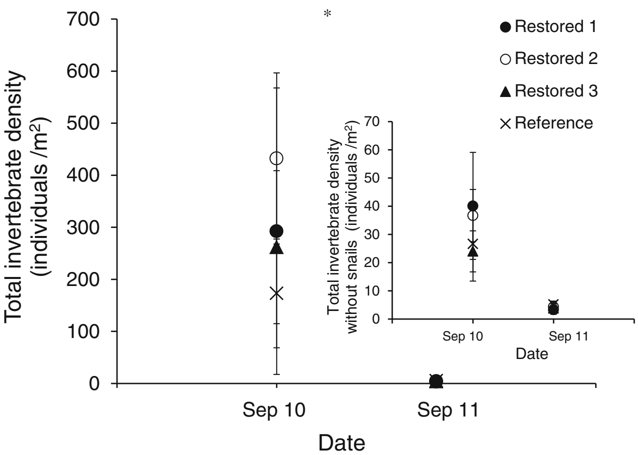

Fig. 6 Average (±SE) total invertebrate density in September 2010 and 2011. *Inset*: average invertebrate density without snails (*Probythinella protera*). *Asterisk* (*) indicates significant differences between years

abundant earlier in the drought year, corresponding with their reproductive cycle (A.R. Armitage, unpublished data). The increase in relative abundance of these marine species, in conjunction with decreases in other fish and invertebrate species, yielded very distinct aquatic faunal community composition in each year, as represented by the high R value in our Analysis of Similarity. These shifts may or may not persist post-drought, depending on the recovery of the SAV community and the population dynamics of the fish and invertebrate species. The drought conditions lessened in severity in 2012, but much of Texas continued to verge on drought. It remains to be seen if our study area will follow a recovery trajectory and return to a pre-drought assemblage, or if there has there been a permanent shift in community structure. To that end, we will continue to monitor vegetation and fauna to document the development trajectory throughout the restored and reference areas in the area.

Much of the perceived value of wetland systems is in their aquatic community structure and services to larger coastal

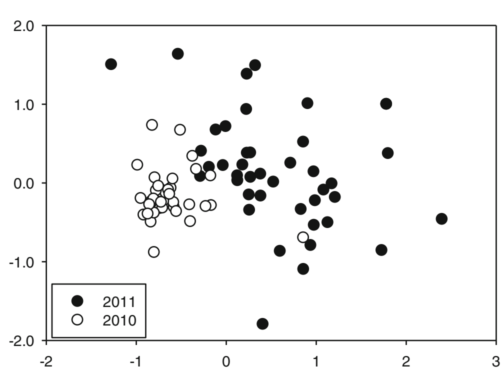

Fig. 7 Composition of aquatic faunal assemblages in pre-drought (2010) and drought (2011) years. The nonmetric multidimensional scaling (*NMDS*) ordination is a representation of dissimilarities among treatments based on a Bray–Curtis similarity matrix (2-D stress value=0.12)

systems. Potential impacts of drought on aquatic communities include loss of SAV and changes in the plant and animal species present, reduction in fish and invertebrate abundance, and changes in fish and invertebrate community structure. By altering plant and animal communities, extreme drought events can change habitat use higher up the food web (Gaines et al. 2000) or contribute to habitat loss (Miller et al. 1996). Restoration monitoring programs that only focus on emergent wetland plant communities, which may be relatively more resilient to drought, may underestimate the sensitivity of these ecosystems to extreme events like drought.

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