LONG-TERM VEGETATION CHANGES IN A TEXAS COASTAL PRAIRIE WETLAND COMPLEX WITH INSIGHTS FOR MANAGERS

A Thesis

by

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ABSTRACT

Restoration efforts can enhance the value of wetlands and their ecosystem services, yet they are often performed without understanding the long-term trajectory of plant community development . To describe this trajectory for Texas coastal prairie wetlands composed of interspersed mima mounds and ponds, plant species and community composition were recorded over 9 years in 2 restored ponds at Sheldon Lake State Park. An analysis of similarity (ANOSIM) across all years and depths showed that plant communities changed substantially over time, regardless of depth, such that the plant communities from later years (2017-2019) did not resemble the communities present in earlier years (2012-2013) . Similarity percentage analysis (SIMPER) showed that the largest contributors to the community change were the native natural recruits, representing 60.8% to 83.0% of the total dissimilarity between years. These results suggest that native plants in Texas coastal prairie wetlands will recruit, establish, and dominate regardless of whether vegetation is planted by managers. However, planting can still accomplish key restoration goals, including a boost to overall vegetation diversity and the replacement of extirpated plant species that may be no longer present in the seed bank.

DEDICATION

This thesis is dedicated to Rosemary Elizabeth Kline, whose love of wetland plants and their identification, will never be forgotten.

ACKNOWLEDGEMENTS

I would like to thank Texas A&M AgriLife Extension Service for allowing me to utilize the collected monitoring data from the Sheldon Lake Wetland Restoration project, as part of the synthesis described in this thesis. The vision of Texas A&M AgriLife Extension Service includes being a "leader in providing science-based information and solutions in agriculture and health to every Texan". The monitoring efforts and this evaluation of that work, I believe, achieve this important vision and moreso provide insight into the science of wetland restoration.

Many thanks are owed to Mr. Paul Roling who spent countless hours in the field during the deep chill of winter mornings and blazing heat of summer afternoons, walking miles to collect all the data from every plant present in our plots (minus the spiders and crickets); and to Mary Carol Edwards who showed me that the labor of collecting data could also be "pond therapy."

I also owe personal thanks to my husband, Mr. Cris Broecker, who supported me through this whole process and especially when I was felt lost in 1.2 million data points. Much love and gratitude is also owed to my parents, Anna Balli Llosa and Luis Eduardo Llosa, who paved the way forward for me with their commitment from day one.

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Contributors

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Additional contributor included Mr. Paul Roling, who participated in all monitoring and data collection activities during the Sheldon Lake State Park Wetland Restoration Project.

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1. INTRODUCTION

Wetlands play a critical role on our natural landscape, providing essential ecological services—those services or goods which provide a benefit to human populations (Constanza et.al. 1997) and to nature. Some of the most critical services include water quality abatement and critical habitat refugia (Johnston 1991, Mitsch and Gosselink 2000). However, the continued loss of wetland resources from past activities coupled with inadequate protection policies (e.g., lack of proper enforcement of Section 404 of the Clean Water Act) in the United States (Bedford 1999, Dahl 1990) have created an obligatory need for wetland restoration. The driving motivation to restore biodiversity and ecosystem functions has propelled wetland restoration forward (Zedler 2000). However, wetland restoration practices, particularly for freshwater prairie wetland systems, have long implemented a minimalist effort (e.g., manipulation of a small number of specific abiotic factors such as removal of drainage pipes) consistent with the theory of self-design, where species assemblages develop on their own after key abiotic conditions are reestablished (Mitsch et. al. 1998). In the 1980s, many freshwater wetlands in the prairie pothole region of North America were restored via the sole re-establishment of abiotic factors such as hydrological conditions (Galatowitsch and van der Valk 1996). Several later studies (Seabloom and van der Valk 2003, Mulhouse and Galatowitsch 2003) concur that the self-design or "efficient community" hypothesis failed to demonstrate a convergence of restored wetlands to the reference natural wetlands, even 12 years after initial establishment. In their open letter, Streever and Zedler (2000) argue that a dependence on self-design methods for wetland restoration can fall dramatically short of success based on plant coverage criteria. Given the importance of wetlands, any insights into the factors that lead to – or prevent – failure are paramount. In parallel, Zedler (2007) points to the lack of restoration efforts proceeding without a clear understanding of the long-term success of the restored plant community, thus highlighting a compelling need for long-term vegetation monitoring—greater than 5 years—of these systems.

Typically, wetland restoration projects are constructed and then planted or seeded with a three to five-year mandatory monitoring period which does not provide any long-term insight into community changes and may misrepresent the change in the community by ceasing monitoring prior to community stabilization (Matthews et. al. 2009). Observing the long-term plant community changes in a restored wetland can be an important means of capturing the stochastic changes in the observed community and allowing the created wetland or "system" time to develop. Therefore, time—both in development of the natural system and in observation of the system--is an essential component of wetland restoration (Mitsch and Wilson 1996). As such, the short-term monitoring typically required for wetland mitigation—5 years or less--cannot capture the complete picture of a restored wetland community. To further support the need for long-term (and frequent) monitoring, Campbell et. al. (2002) compared 12 created wetland sites—from 2 years to 18 years—to natural reference sites and found that created sites were different from the natural sites, but not in a manner trending towards the natural sites, suggesting that the time needed for created systems to mature is unknown, or minimally greater than the typical 5-year monitoring protocol.

Long-term data collection at restored site can reveal complex series of changes within the community (e.g., wet years to dry years, seasonal variation over multiple years and progress of invasions). As an example, during the initial establishment of a restored wetland, these systems can experience invasion from noxious, introduced species (Zedler and Kercher 2002), or fluctuations in diversity as planted species are displaced by naturally recruited species (Budelsky and Galatowitsch 2004). Non-native (or invasive) species can be particularly problematic, as they tend to grow with greater vigor and higher reproductive capacity than native species (Galatowitsch et. al. 1999), and therefore may out-compete the restored plant community over time. Infrequent and short-term monitoring cycles would likely misrepresent these community changes as an insignificant presence (vegetative cover) of invasive species within the monitoring plots. Similar patterns may emerge with prolific native recruits that adapt to the restored site, potentially displacing planted/restored species (Armitage et. al 2006). Therefore, in the initial years of establishment, the short-term monitoring data potentially indicates that natural recruits and restored plants are equally successful within the site; however, the longer monitoring data may capture the complex, dynamic changes in abundance of planted species, non-native (invasive) recruits, and native recruited species. The dynamic shifts (i.e., dominance, displacement, establishment) between planted species, native recruits, and invasive recruits can ultimately determine the trajectory of the restored wetlands, and its success or failure.

The long-term monitoring data necessary to elucidate the maturation processes of restoration sites are rare (Zedler and Callaway 1999). Hagen and Evju (2013) argue that short term monitoring suffices for the necessary management of restoration sites; coupled with adaptive management measures for the restored wetland, the data can lead to long-term success. However, for reasons argued above, short-term monitoring falls short of identifying key fluctuations which can completely alter restored wetlands and support a misguided management technique. Amplified, any mismanagement of a restored wetland system can lead to a significant loss of function and ecosystem services, which, in turn, is a loss of significant invested costs.

This study intends to utilize long-term (9 years) vegetative cover data from a freshwater restoration project at Sheldon Lake State Park [Texas, USA] to identify potential key fluctuations or dynamic shifts which can guide restoration techniques and management of coastal prairie wetlands.

As funding to implement wetland restoration has increased since the implementation of the Section 404 of the Clean Water Act (1972), the restoration of these critical habitats has received national attention and scrutiny (Zedler 2000). With millions of dollars invested, funding organizations such as the U.S. Fish and Wildlife Service have placed high demands on wetland managers and practitioners to produce successful restoration projects within a restrictive time frame (Madsen 1986, U.S. Fish and Wildlife 2019). Yet, there remains uncertainty about best practices for restoration methods (Galatowitsch 2006). For example, a project in prairie pothole wetlands habitat in the midwest region followed the “self-design” premise, depending on natural recruitment to complete the plant assemblage. However, after 12 years, certain targeted wetland plant guilds remained poorly represented, strongly indicating the need for active planting of restoration sites to ensure a complete assemblage. Failures in the self-design approach may be widespread, as numerous studies have highlighted the depauperate presence of long-term detailed monitoring data to support the use of more labor-intensive but potentially more effective strategies such as targeted planting methodologies (Galatowitsch 2006, Streever and Zedler 2000). Thus, it is not surprising that many practitioners continue to follow—out of cost or lack of plant resources—the self-design method, rather than consider the better success of a planted restoration site (Prach et. al. 2001, Suding et. al. 2004)). More readily available short-term data often cannot reveal restoration trajectories, and therefore, the power of short-term data to assess success is minimal. It is within the context of elucidating the long-term changes in the characteristics of restored wetland, or the indicatory restoration trajectories, that the Sheldon Lake State Park wetland restoration (Sheldon) effort and monitoring protocol can play a critical role. The Sheldon monitoring protocol provides an opportunity to compare vegetation changes between seasons and across a larger arc of time to identify the successional variation and patterns between planted, native, and invasive recruits within the community. Utilizing this temporal trend data, an analysis of the Sheldon monitoring data can yield insight into restoration trajectories, thus potentially informing future projects and guarding against restoration failures.

2. METHODS

2.1 Restoration Project Background

In 2003, the wetland restoration effort at Sheldon Lake State Park began with an internal Texas Parks and Wildlife Department (TPWD) grant to “construct a Habitat Restoration project…[as part of the process] of implementing a Master Plan for the development of Sheldon Lake State park as a regional environmental learning center.” The end goal included “restoring and enhancing the landscape, returning it from years of crop production to a naturally functioning mosaic of prairie, seasonal wetlands and ponds” (TPWD 2003). The natural mosaic to be restored included poorly drained lowlands embedded in upper coastal prairie, described in Diamond and Smeins (1984). This was the “gold standard” to which the Sheldon wetland restoration aspired.

The project site had experienced significant alteration, primarily due to agricultural development, since the 1930s. The original landscape of the project site included the typical mosaic of mima mounds and lowlands (Figure 1a); however, by 1978, the consistent plowing by tenant farmers had levelled the landscape (Figure 1b), filling the lowlands with the soils of the mima mounds. By 2002, the agricultural development had all but erased any surface indicators of the former mima mounds and lowlands (Figure 1c). Mima mounds are semi-circular mounds of aeolian and fluvial origin, comprised of highly differentiated soil horizon layers underneath an overly thickened A (surface) horizon and typical of coastal prairie complex in southeastern Texas (Diamond and Smeins 1984, Carty et.al. 1988). The only remaining remnant of the original landscape was the lighter soil signatures (white dots in Figure 1) which indicated the former bases of the mima mounds.

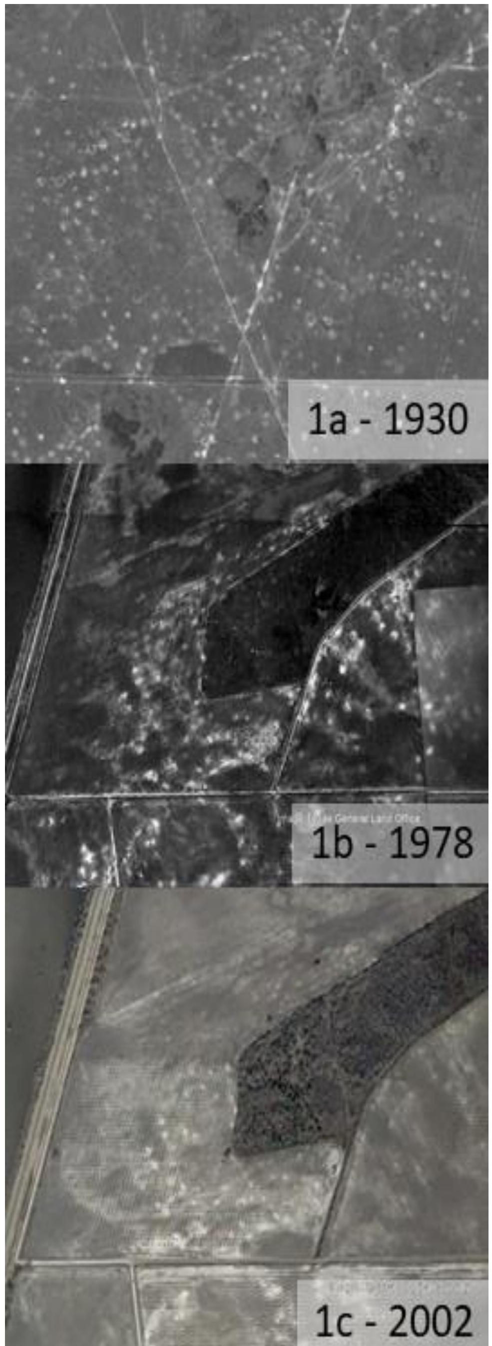


Figure 1 (a-c) - Progression of land alteration at Sheldon Lake State Park

The construction method for the wetland restoration consisted of excavating the lowlands that had been previously filled through agricultural processes. Locations of all excavated ponds were identified by the examination of aerial imagery and historical topographic maps with subsequent field investigation to verify remnant hydric soil conditions. This process of excavation was intended to re-expose the wetland soils to hydrologic conditions consistent with freshwater coastal prairie wetlands, and thus would allow for the support of wetland vegetation native to this community. In an effort to jump start the wetland plant community, the ponds were planted with nursery-reared plants. The palette of plant species (Table 1) used in the project was based on reference sites adjacent to the park.

As part of the initial restoration effort (Phase 1), a total of four wetland ponds were restored. In subsequent years, additional grants were awarded to complete Phases 2 and 3 at the Park, totaling 215 acres of restored wetlands embedded within coastal prairie. Four phases of restoration occurred as part of the overall effort, including the introduction of approximately 150,000 plants of 29 different species (Table 1 and 2) into the entire restoration area.

Site management upon completion of the restoration included an integrated process including targeted herbicide applications (e.g. glyphosate, sulfosulfuron, Imazapic/imazapyr, 2-4 D picloram, triclopyr), mowing, discing, and seed-head removal. Mowing included up to 3 cycles from Spring to Fall, depending on staff and available resources. No pre-emergent herbicides were utilized prior to excavation and planting (K. Norrid, personal communication, October 15, 2021).

Table 1- Original plant list for Sheldon Lake State Park Wetland Restoration

	Common Name	Scientific Name	Family	Functional Group	Duration	Wetland Status*
1	Coastal Water-hyssop	*Bacopa monnieri*	Scrophulariaceae	Forb/herb	Perennial	OBL
2	Water Canna	*Canna glauca*	Cannaceae	Forb/herb	Perennial	OBL
3	Shoreline sedge	*Carex hyalinolepis*⁺	Cyperaceae	Graminoid	Perennial	OBL
4	Jamaica saw-grass	*Cladium mariscus spp. jamaicense*	Cyperaceae	Graminoid	Perennial	OBL
5	Swamp lily	*Crinum americanum*	Liliaceae	Forb/herb	Perennial	OBL
6	Jointed Flatsedge	*Cyperus articulatus*⁺	Cyperaceae	Graminoid	Perennial	OBL
7	Green Flatsedge	*Cyperus virens*	Cyperaceae	Graminoid	Perennial	FACW
8	Common spikerush	*Eleocharis palustris*⁺	Cyperaceae	Graminoid	Perennial	OBL
9	Mountain spikerush	*Eleocharis montana*	Cyperaceae	Graminoid	Perennial	OBL
10	Sand spikerush	*Eleocharis montevidensis*	Cyperaceae	Graminoid	Perennial	FACW
11	Square-stem spikerush	*Eleocharis quadrangulata*	Cyperaceae	Graminoid	Perennial	OBL
12	Woolly rose mallow	*Hibiscus lasiocarpos*	Malvaceae	Forb/herb/ subshrub	Annual/ Perennial	OBL
13	Blue waterleaf	*Hydrolea ovata*	Hydrophyllaceae	Forb/herb	Perennial	OBL
14	Spider lily	*Hymenocallis liriosme*	Liliaceae	Forb/herb	Perennial	OBL
15	Virginia Iris	*Iris virginica*	Iridaceae	Forb/herb	Perennial	OBL
16	Southern blue flag	*Iris virginica var. shrevei*	Iridaceae	Forb/herb	Perennial	OBL
17	Common Rush	*Juncus effusus*	Juncaceae	Graminoid	Perennial	OBL
18	Grassleaf Rush	*Juncus marginatus*	Juncaceae	Graminoid	Perennial	FACW
19	Stout rush	*Juncus nodatus*	Juncaceae	Graminoid	Perennial	OBL
20	Lesser creeping rush	*Juncus repens*	Juncaceae	Graminoid	Perennial	OBL
21	Southern cutgrass	*Leersia hexandra*	Poaceae	Graminoid	Perennial	OBL
22	American white water-lily	*Nymphaea odorata*	Nymphaeaceae	Forb/herb	Perennial	OBL
23	Maidencane	*Panicum hemitomon*	Poaceae	Graminoid	Perennial	OBL
24	Switchgrass	*Panicum virgatum*⁺	Poaceae	Graminoid	Perennial	FAC
25	Pickerelweed	*Pontederia cordata*	Pontederiaceae	Forb/herb	Perennial	OBL
26	Anglestem Beaksedge	*Rhynchospora caduca*	Cyperaceae	Graminoid	Perennial	OBL
27	White-top sedge	*Rhynchospora colorata*	Cyperaceae	Graminoid	Perennial	FACW
28	Horned Beakrush	*Rhynchospora corniculata*	Cyperaceae	Graminoid	Perennial	OBL
29	Indianola Beaksedge	*Rhynchospora indianolensis*⁺	Cyperaceae	Graminoid	Perennial	FACW
30	Longbarb arrowhead	*Sagittaria longiloba*	Alismataceae	Forb/herb	Perennial	OBL
31	Nipplebract arrowhead	*Sagittaria papillosa*	Alismataceae	Forb/herb	Perennial	OBL
32	Delta arrowhead	*Sagittaria platyphylla*	Alismataceae	Forb/herb	Perennial	OBL
33	American bulrush	*Schoenoplectus pungens*	Cyperaceae	Graminoid	Perennial	OBL
34	Marsh-hay cordgrass	*Spartina patens*	Poaceae	Graminoid	Perennial	FACW
35	Gulf Cordgrass	*Spartina spartinae*	Poaceae	Graminoid	Perennial	OBL
36	Powdery alligator-flag	*Thalia dealbata*	Marantaceae	Forb/herb	Perennial	OBL
37	Eastern Gamagrass	*Tripscacum dactyloides*⁺	Poaceae	Graminoid	Perennial	FAC
38	Yellow eyed grass	*Xyris laxifolia var. iridifolia*⁺	Xyridaceae	Forb/herb	Perennial	OBL

*Wetland status are defined as obligate (OBL), facultative wet (FACW) or facultative (FAC) for the Atlantic and Gulf Coastal Plain region
⁺Designated planted species which were never recorded in any part of the monitoring data

Table 2- Planted species plug quantity by pond from April 5, 2011 to October 8, 2014*						
	Common Name	**Plant Species**	**Plug Quantity**	**Pond**	**Plug Quantity**	**Pond**
1	Coastal waterhyssop	*Bacopa monnieri*	30	17	25	11
2	Water Canna	*Canna glauca*	149	17		
3	Shoreline sedge	*Carex hyalinolepis*+	50	17		
4	Jamaica saw-grass	*Cladium mariscus spp. jamaicense*	89	17		
5	Swamp Lily	*Crinum americanum*	66	17	26	11
6	Jointed Flatsedge	*Cyperus articulatus*+	50	17		
7	Green Flatsedge	*Cyperus virens*	232	17	3	11
8	Common spikerush	*Eleocharis palustris*+	50	17		
9	Sand spikerush	*Eleocharis montevidensis*	250	17		
10	Mountain Spikerush	*Eleocharis montana*	342	17		
11	Square-stem spikerush	*Eleocharis quadrangulata*	455	17		
12	Woolly rose mallow	*Hibiscus lasiocarpos*	20	17		
13	Blue waterleaf	*Hydrolea ovata*	1095	17		
14	Spider lily	*Hymenocallis liriosme*	139	17	175	11
15	Virginia iris	*Iris virginica*	1045	17		
16	Southern blue flag	*Iris virginica var. shrevei*	-	-	-	-
17	Soft rush	*Juncus effusus*	468	17	810	11
18	Grassleaf rush	*Juncus marginatus*	2	17		
19	Stout rush	*Juncus nodatus*	6	17	114	11
20	Lesser creeping rush	*Juncus repens*	25	17		
21	Southern cutgrass	*Leersia hexandra*	160	17	100	11
22	American white water-lily	*Nymphaea odorata*	26	17		
23	Maidencane	*Panicum hemitomon*	375	17		
24	Switchgrass	*Panicum virgatum*+	810	17		
25	Pickerelweed	*Pontederia cordata*	275	17	150	11
26	Anglestem beaksedge	*Rhynchospora caduca*	160	17		
27	White-top Sedge	*Rhynchospora colorata*	10	17		
28	Horned beakrush	*Rhynchospora corniculata*	609	17		
29	Indianola beaksedge	*Rhynchospora indianolensis*+	48	17		
30	Longbarb arrowhead	*Sagittaria longiloba*	68	17		
31	Nipplebract arrowhead	*Sagittaria papillosa*	230	17		
32	Delta arrowhead	*Sagittaria platyphylla*	2654	17		
33	American bulrush	*Schoenoplectus pungens*	115	17		
34	Marsh-hay cordgrass	*Spartina patens*	1888	17	1700	11
35	Gulf Cordgrass	*Spartina spartinae*	490	17	520	11
36	Powdery alligator-flag	*Thalia dealbata*	25	17		
37	Eastern Gamagrass	*Tripsacum dactyloides*+	961	17		
38	Yellow-eyed grass	*Xyris laxifolia var. iridifolia*+	44	17		

*Table does not reflect planting effort prior to April 5, 2011

The planting records from the restoration efforts identified additional plugs of planted species that were added to both monitored ponds through 2014 (Table 2).

2.2. Restoration monitoring and data collection

A monitoring and data collection protocol was established to document long-term patterns of vegetation establishment, recruitment, and succession within the project site. The protocol assigned random sampling plot locations based on GPS coordinates in two representative ponds (ponds 11 and 17) where planting had been completed in 2010 (Figure 2). These ponds were selected because of their relatively large size, placement within the landscape to capture rainfall, and the similar maximum depths (> 12 inches) of the ponds. Both ponds were treated and constructed in the same manner: pre-construction herbicide treatment, tilling of the fallow land prior to construction, excavating to specified pond depths by the same satellite-guided tractor, and planting with the same palette of native plants by volunteers from 2010 to 2014.

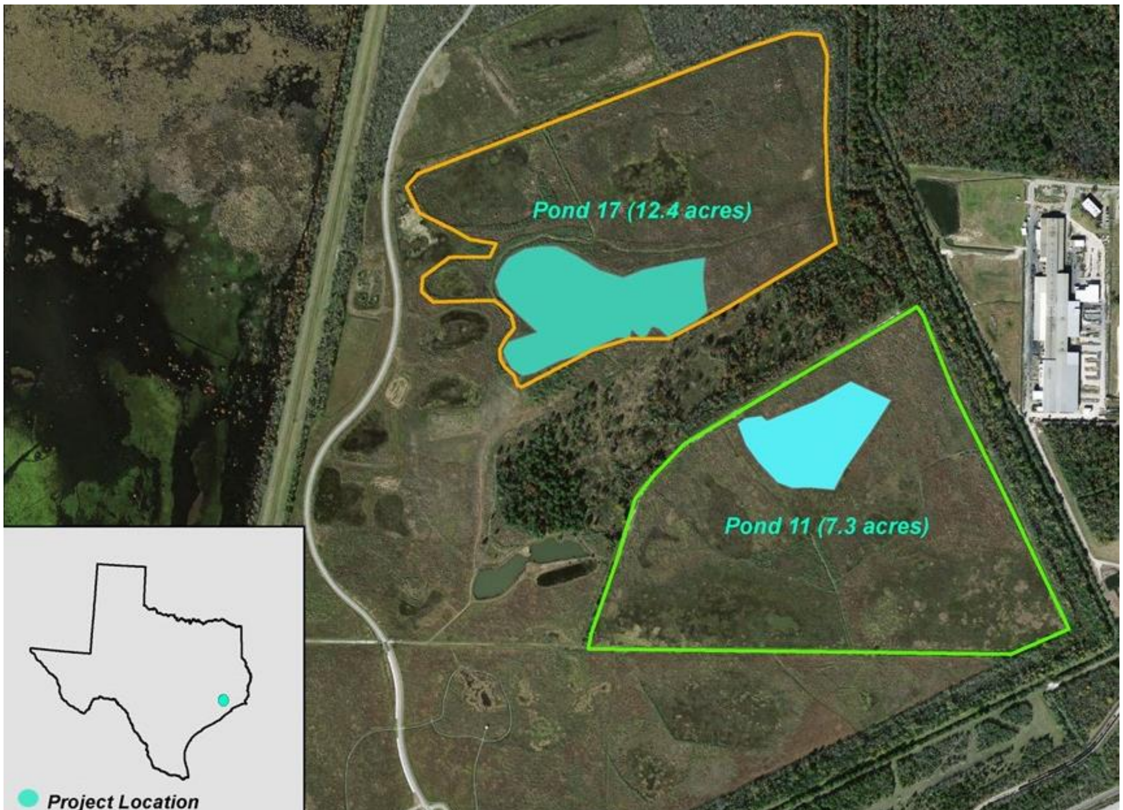


Figure 2 - Monitored ponds (11 and 17) within Phase 2 (orange) and Phase 3 (green) boundary lines

Vegetation was monitored 2-4 times a year from 2011-2019 (Table 3). This monitoring program provided an ideal opportunity to conduct a robust assessment of successional changes in the community.

Table 3 - Monitoring data sets available for Ponds 11 and 17

	2011	2012	2013	2014	2015	2016	2017	2018	2019
Spring			March	April	April	March	March		April
Summer	July	June	June	June		June		June	July
Fall	September	September	September		September				
Winter	December		December	January 2015	January 2016	December		December	

Permanent monitoring plots (1 m^2) were established at 58 locations (identified by randomly selected GPS points) within Pond 11, and 98 selected locations within Pond 17 (Figure 3). Sampling plots were spread across all potential restored emergent wetland habitat within the restoration boundaries, and irrespective of whether the plot had been initially planted. Plots in Pond 11 were evenly tiered across three water depths (0”, 6” and ≥12”), and plots in Pond 17 were tiered across four water depths (0”, 4”, 8”, and ≥12”). Originally, the recorded depth zones were labelled differently between ponds 11 and 17. Therefore, to standardize depth zones between the ponds, the zones were divided into three categories: shallow (0-3”), intermediate (4-11”), and deep (>12”). The regrouping of the depths was based on the dominant presence of plants adapted to temporary inundation typical of the shallower range of the intermediate zone (e.g., *Iva annua*). This was in contrast to the relative higher presence of more permanent wet soil/water conditions in the deeper (“11”) intermediate zone (e.g., *Eleocharis montana*).

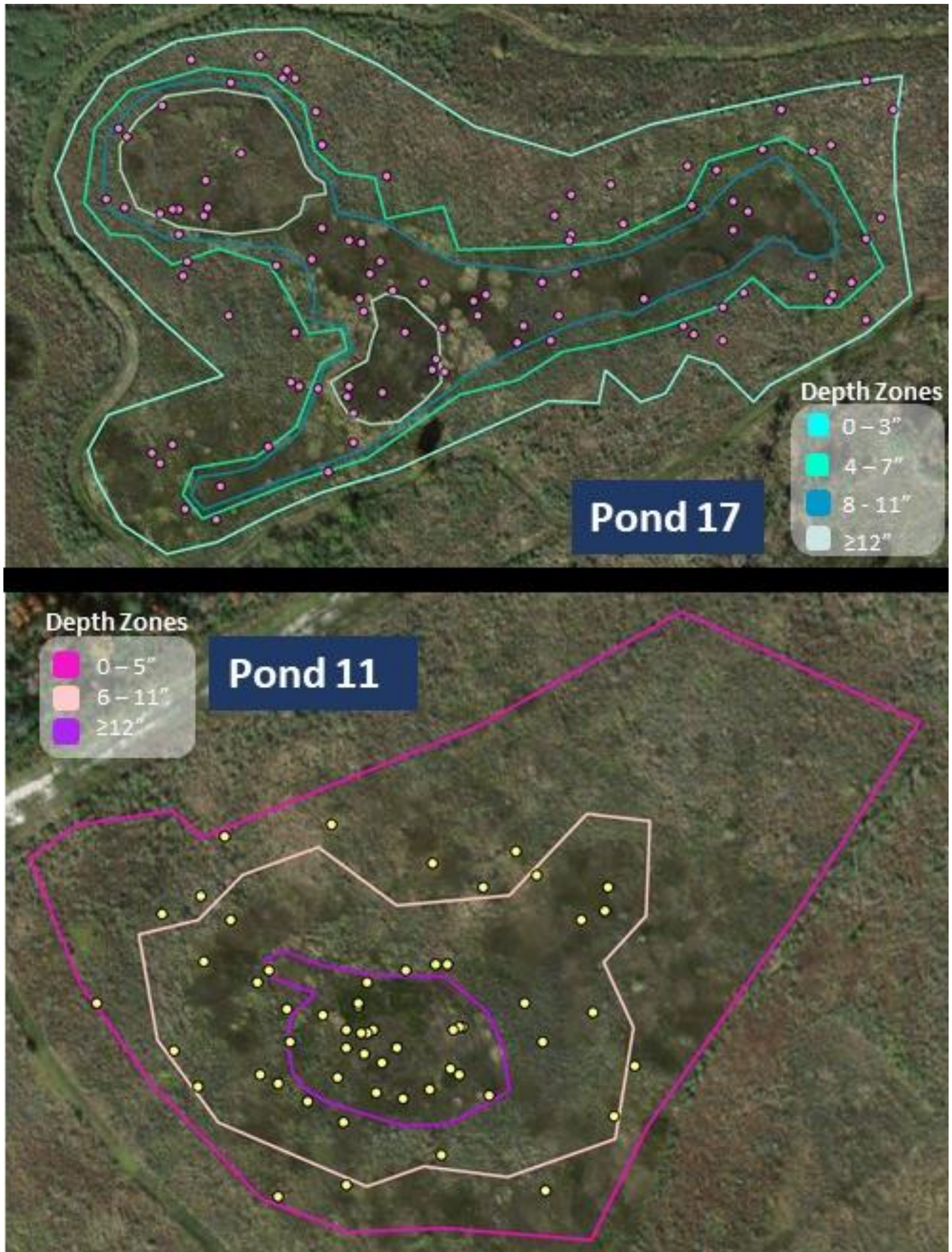


Figure 3 - Monitoring (sampling) plot locations within Pond 17 (pink dots) and Pond 11 (yellow dots)

In each plot, a visual estimate of the percent coverage of each plant species was recorded in one of 7 categories: 0, 1-4%, 5-29%, 30-69%, 70-94%, 95-99%, 100%. Of the 334 recorded species, 303 were native and non-native recruits that colonized over the course of the study.

2.3 Data Analysis

All percent cover data for all 334 recorded species was sorted based on depth (shallow, intermediate, deep), year (2012-2019), and season (spring, summer, fall, winter). Site level average percent cover for species present was calculated using the cover category value (i.e. 0 for 0% cover; 1 for 1-4% cover, 5 for 5-29% cover) and averaged over all plots within a specified season and year (see Results Table 8). Standard error was calculated as the standard deviation of all the specified plots divided by the square root of total number of sampled plots (see Results Table 8).

All plant species were organized into three different recruitment classes to determine if there were broader patterns of native or non-native species recruitment. The three classes were:

- *Planted* species (31 recorded species) were intentionally transplanted into the site as part of the restoration effort. A total of 7 planted species identified in Table 1 ($^{+}$) did not survive and were never recorded;
- *Native recruits* (249 species) were native species that established on their own during the study period;

- *Invasive recruits* (54 species) were primarily non-native species to the region that colonized the site during the study period. For the purposes of this analysis, the native species *Typha latifolia* was grouped within the invasive species, as it is widely considered an extremely aggressive recruiting species in restoration projects.

Although planting was completed in 2010, the final compiled dataset excluded data from all seasons in 2011. This year was a record drought year for the State of Texas. Beginning in October 2010 and extending through September 2011, the total rainfall for the state was recorded at 15.20 inches, falling below the previous 1956 record minimum of 15.40 inches (Nielsen-Gammon 2011). Coupled with a record high summer temperatures, "over 2°F above the previous Texas record", the lack of rainfall created exceptional drought conditions (Nielsen-Gammon 2011). By June 28, 2011, the U.S. Drought Monitor reported that 72.32 percent of the entire state, including the coastal zone location of this restoration project, was in D4 or Drought-Exceptional conditions (Nielsen-Gammon 2011, http://drought.unl.edu/droughtplanning/InfobyState.aspx). The Texas coastal landscape, including this restoration site, was highly affected by this lack of rainfall (Klockow et.al. 2018, Xu et. al. 2017). As a result, additional planting events (using the same original palette of plants) occurred through 2014. Therefore, this analysis considered monitoring data collected starting in 2012, in order to assess the effects of the restoration work itself, rather than the effect of a significant drought event (See Results Table 8 for drought year cover values).

All analyses were conducted in PRIMER v.6 (PRIMER-E Ltd., Plymouth Marine Laboratory, United Kingdom). To assess changes in plant communities over time, the compiled dataset was split into four seasonal subsets (spring, summer, fall, winter). Each seasonal dataset was square root transformed, and then changes in plant community composition over time were initially analyzed with a 2-way Analysis of Similarity (ANOSIM) based on a Bray-Curtis resemblance matrix, where the factors were year (2012, 2013, etc.) and depth (shallow, intermediate and deep). A dummy variable was added to the ANOSIM test to reduce the influence of absent species on the dissimilarity calculations (Clarke 2006). ANOSIM analyses generated an R-statistic that represents dissimilarity among species assemblages; R values greater than 0.25 indicate that communities are distinct from each other, and R values less than 0.25 indicate substantial similarities between groups (Clarke and Warwick 2001). Nonmetric multidimensional scaling (nMDS) ordination was used to represent average dissimilarities among years in Euclidean two-dimensional space, and the SIMPER (SIMilarity PERcentages) routine was used as an exploratory technique on selected pairwise comparisons between years (see Results Table 6) to identify which plant species most strongly contributed to the MDS ordination.

3. RESULTS

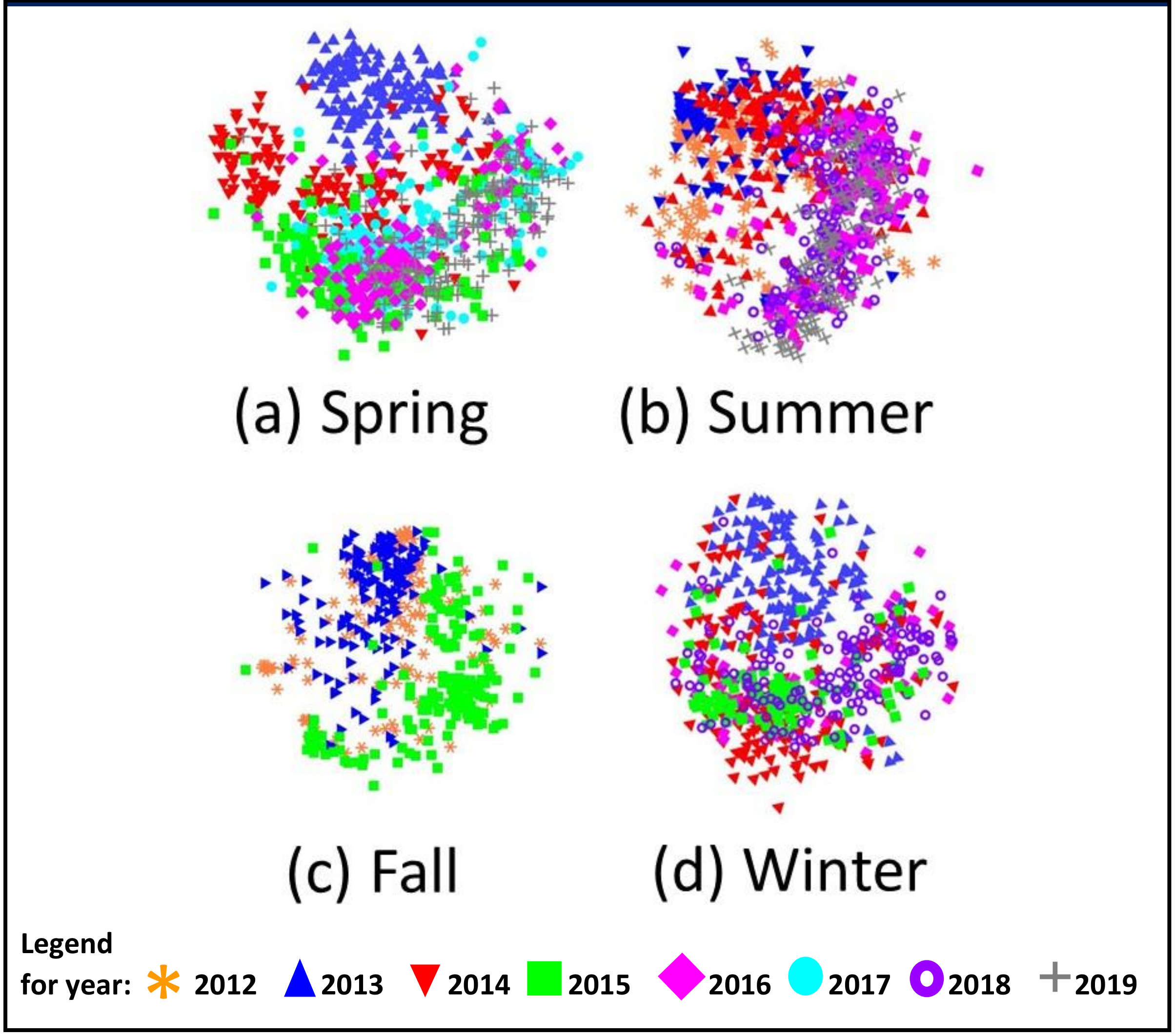
Plant communities changed substantially over time, regardless of depth. In the spring, there was a substantial difference in plant communities among years (global R = 0.49) but not across depths (global R = 0.213) (Table 4). Similar patterns were detected for all seasons, where differences among years were larger than differences among depths ($R_{Year} > R_{Depth}$ for all seasons; Table 4).

Table 4 - Global R Values from two-way ANOSIM tests using the factors YEAR and DEPTH individually, per season

FACTORS	SPRING	SUMMER	FALL	WINTER
YEAR	0.49	0.421	0.336	0.342
DEPTH	0.213	0.252	0.233	0.162

Visual examination of plant assemblages in the spring revealed that the early established plant community (2013) diverged from and was exceptionally dissimilar to the later succession community (2016-2019; Figure 4a). There was a clear progression of change in the spring plant assemblages over time, with each year incrementally different than the year prior (Figure 4a). Similar patterns of change were apparent in the summer (Figure 4b), with distinct changes over time from early (2012-2013) to later (2017-2019) successional assemblages. In the fall, successive years were also distinct from each other (Figure 4c), though there were no fall data points in later succession years (2016 or later). In the winter, there was more variability within and among years, but early (2013) and late (2018) successional assemblages were largely distinct from each other (Figure 4d).

Figure 4: The non-metric Multidimensional Scaling graphs for spring, summer, fall and winter display clear dissimilarities in community structure between years.



Detailed changes over time were further explored with pairwise ANOSIM tests between all pairs of years within each season. Overall, there was substantial dissimilarity among years, especially when comparing earlier (2012-2014) to later (2016-2019) years (Table 5). Pairs of years with R values greater than 0.25 indicate substantial dissimilarity (Clark 2006); values below this threshold were not considered in further analysis.

In the spring, pairwise comparisons revealed substantial differences between most years, with the exception of a few pairs spanning 1-2 years (Table 5). For example, the year pair 2013-2019 had an R value of 0.819, indicating substantially different plant community composition between those years. There was little overlap of recorded plants present in spring 2013 compared to those present in spring 2019 (Figure 4a). Summer, like spring, showed a similar divergence between early and later years for community structure (Figure 4b). In particular, there was a very large dissimilarity in the summer of 2013 compared to 2019 (R = 0.758), suggesting that a very different community existed at the end of the monitoring versus the beginning (Table 5). Although the fall season had fewer years to compare, the larger pattern of change over time remained clear (Figure 4c), with high R values (>0.4) for comparisons between early and later years. There were also long-term changes in the plant communities in the winter, with the largest year pair R values recorded for pairs of years separated by the longest time ($R_{2013-2018} = 0.581$).

Table 5 - Global R values for paired years from two-way ANOSIM pairwise test, displayed by season (R values over 0.25 indicated in bold, plain text indicate values below threshold, and greyed year pairs contain no data)

YEARS PAIRS	SPRING	SUMMER	FALL	WINTER
2012-2013		0.175	0.115	
2012-2014		**0.296**		
2012-2015			**0.421**	
2012-2016		**0.646**		
2012-2017				
2012-2018		**0.56**		
2012-2019		**0.666**		
2013-2014	**0.661**	**0.256**		**0.378**
2013-2015	**0.827**		**0.495**	**0.568**
2013-2016	**0.764**	**0.704**		**0.615**
2016-2017	**0.763**			

Table 5 Continued - Global R values for paired years from two-way ANOSIM pairwise test, displayed by season (R values over 0.25 indicated in bold, plain text indicate values below threshold, and greyed year pairs contain no data)

YEARS PAIRS	SPRING	SUMMER	FALL	WINTER
2013-2018		**0.653**		**0.581**
2013-2019	**0.819**	**0.758**		
2014-2015	**0.362**			0.086
2014-2016	**0.539**	**0.404**		0.211
2014-2017	**0.509**			
2014-2018		**0.447**		**0.26**
2014-2019	**0.655**	**0.528**		
2015-2016	0.184			**0.251**
2015-2017	**0.261**			
2015-2018				0.164
2015-2019	**0.403**			
2016-2017	0.06			
2016-2018		0.139		0.246
2016-2019	**0.274**	0.177		
2017-2018				
2017-2019	0.236			
2018-2019		0.042		

One-way SIMPER tests identified the species that contributed the most to the dissimilarity between pairs of years within each season, pooled across depths (Table 6). Each contributing plant species was grouped into one of three classes (planted, native recruits, invasive recruits), and the change between pairs of years (increase or decrease) was noted. In most cases, the changes over time were attributable to increases in native recruits (Table 6). However, in the summer, the invasive *Alternanthera philoxeroides*, a species of management concern, had increased substantially over time, becoming a notable component of the community (Table 6). Only one of the originally planted species, *Eleocharis quadrangulata*, consistently increased in cover over time.

Table 6 - Top contributing plant species to dissimilarity in identified year pairs (R>0.5) per one-way SIMPER test using YEAR as factor (arrows indicate an increase or decrease respectively between years in the identified year pair; top ten contributors are further listed in Appendix)

Year pairs of interest	R value	Avg Diss	Species	% Contribution to Dissimilarity by Species		Recruitment Group
Spring 2013-2014	0.661	92.8	*Limnosciadium pumilum*	9.9	↑	Recruit
			Isoetes melanopoda	7.0	↑	Recruit
			Eleocharis obtusa	6.0	↓	Recruit
Spring 2013-2015	0.827	96.16	*Ludwigia palustris*	8.5	↑	Recruit
			Isoetes melanopoda	5.7	↑	Recruit
			Eleocharis obtusa	4.8	↓	Recruit
Spring 2013-2016	0.764	95.09	*Juncus validus*	9.1	↑	Recruit
			Cyperus virens	6.3	↑	Planted
			Ludwigia palustris	6.3	↑	Recruit
Spring 2013-2017	0.763	94.87	***Cyperus virens***	7.8	↑	Planted
			Juncus validus	6.2	↑	Recruit
			Eleocharis obtusa	5.6	↓	Recruit
Spring 2013-2019	0.819	95.94	*Eleocharis obtusa*	5.7	↓	Recruit
			Eleocharis quadrangulata	5.7	↑	Planted
			Limnosciadium spp.	5.1	↓	Recruit
Spring 2014-2016	0.539	90.24	*Limnosciadium pumilum*	9.4	↓	Recruit
			Juncus validus	9.3	↑	Recruit
			Ludwigia palustris	6.9	↑	Recruit
Spring 2014-2017	0.509	89.51	*Limnosciadium pumilum*	9.7	↓	Recruit
			Cyperus virens	7.9	↑	Planted
			Juncus validus	6.3	↑	Recruit
Spring 2014-2019	0.655	92.93	*Limnosciadium pumilum*	9.4	↑	Recruit
			Isoetes melanopoda	6.3	↓	Recruit
			Eleocharis quadrangulata	5.6	↑	Planted
Summer 2012-2016	0.646	94.73	*Iva annua*	14.3	↓	Recruit
			Eleocharis quadrangulata	7.2	↑	Planted
			Ludwigia palustris	7.2	↑	Recruit
Summer 2012-2018	0.56	93.57	*Iva annua*	15.2	↓	Recruit
			Eleocharis quadrangulata	9.0	↑	Planted
			Alternanthera philoxeroides	5.2	↑	Invasive
Summer 2012-2019	0.666	94.65	*Iva annua*	13.9	↓	Recruit
			Eleocharis quadrangulata	9.8	↑	Planted
			Alternanthera philoxeroides	4.3	↑	Invasive
Summer 2013-2016	0.704	93.29	*Iva annua*	8.7	↓	Recruit
			Symphyotrichum divaricatum	6.5	↓	Recruit
			Ludwigia palustris	6.3	↑	Recruit
Summer 2013-2018	0.653	92.88	*Iva annua*	9.1	↓	Recruit
			Eleocharis quadrangulata	7.6	↑	Planted
			Symphyotrichum divaricatum	6.8	↓	Recruit
Summer 2013-2019	0.758	94.25	***Eleocharis quadrangulata***	8.4	↑	Planted
			Iva annua	8.4	↓	Recruit
			Symphyotrichum divaricatum	6.3	↓	Recruit
Summer 2014-2019	0.528	90.63	***Eleocharis quadrangulata***	9.6	↑	Planted
			Iva annua	5.2	↓	Recruit
			Ludwigia palustris	4.6	↓	Recruit
Winter 2013-2015	0.568	92.79	*Juncus validus*	10.5	↑	Recruit
			Cyperus virens	9.7	↑	Planted

Year pairs of interest	R value	Avg Diss	Species	% Contribution to Dissimilarity by Species		Recruitment Group
			Limnosciadium pumilum	7.4	↓	Recruit
Winter 2013-2016	0.615	94.76	***Cyperus virens***	9.5	↑	**Planted**
			Juncus validus	7.7	↑	Recruit
			Eleocharis quadrangulata	7.1	↑	**Planted**
Winter 2013-2018	0.581	94.39	***Eleocharis quadrangulata***	8.9	↑	**Planted**
			Limnosciadium pumilum	7.4	↓	Recruit
			Cyperus virens	5.2	↑	**Planted**

Most of the dissimilarity between year pairs was attributable to native recruits (Table 7). Planted and invasive species were relatively low (between 8.1-29.7%) contributors to dissimilarity between years (Table 7). However, in most year pair comparisons, planted species contributed more to dissimilarity than did invasive species (Table 7). Specifically, for year pair comparisons where R > 0.5, planted species were often double the value of invasive recruits—an important observation for management purposes (Table 7).

Table 7 - Total contribution toward the average dissimilarity per year pair of interest by recruitment category (R>0.5)

Year Pair of interest	Recruitment Group	% Contribution to Avg Diss
Spring 2013-2014	Native Recruit	81.64
	Invasive	10.19
	Planted	8.12
Spring 2013-2015	Native Recruit	72.34
	Invasive	7.59
	Planted	20.10
Spring 2013-2016	Native Recruit	72.19
	Invasive	10.29
	Planted	17.49
Spring 2013-2017	Native Recruit	68.55
	Invasive	9.89
	Planted	21.58
Spring 2013-2019	Native Recruit	65.87
	Invasive	12.51
	Planted	21.60
Spring 2014-2016	Native Recruit	70.02
	Invasive	10.69

Table 7 Continued - Total contribution toward the average dissimilarity per year pair of interest by recruitment category (R>0.5)		
Year Pair of interest	**Recruitment Group**	**% Contribution to Avg Diss**
	Planted	19.27
Spring 2014-2017	Native Recruit	66.03
	Invasive	10.51
	Planted	23.47
Spring 2014-2019	Native Recruit	64.45
	Invasive	12.68
	Planted	22.86
Summer 2012-2016	Native Recruit	65.15
	Invasive	11.05
	Planted	23.79
Summer 2012-2018	Native Recruit	58.92
	Invasive	15.55
	Planted	25.60
Summer 2012-2019	Native Recruit	57.71
	Invasive	12.95
	Planted	26.27
Summer 2013-2016	Native Recruit	68.51
	Invasive	11.25
	Planted	20.23
Summer 2013-2018	Native Recruit	62.56
	Invasive	15.18
	Planted	22.19
Summer 2013-2019	Native Recruit	64.85
	Invasive	12.62
	Planted	22.51
Summer 2014-2019	Native Recruit	58.38
	Invasive	12.60
	Planted	28.98
Winter 2013-2015	Native Recruit	69.85
	Invasive	10.73
	Planted	19.39
Winter 2013-2016	Native Recruit	54.10
	Invasive	16.23
	Planted	29.68
Winter 2013-2018	Native Recruit	57.76
	Invasive	13.42
	Planted	28.80

At a landscape level, native recruits dominated the plant community. While planted species did not out-compete natural recruits (Table 6 and 7), these same species persisted at the restored site through 2019 (Table 8), indicating that planting increased diversity beyond the species pool of native recruits. Specifically, 25 of the original 31 planted species as part of the restoration project survived initial planting, and persisted within the plant community until

2019. Four of the 25 surviving planted species had cover values exceeding 1% demonstrating that the plant species survived and thrived (Table 8). Consistently, those four planted species were also among the top ten species contributing to dissimilarity when observing community changes between different years (Appendix). It is important to note that the site-level sampling design and the high diversity of plant species generated relatively low absolute percent cover values for recorded species.

Table 8 - Average percent cover per species in a drought year and in two late successional time periods (native recruits in green, planted species in purple; invasive recruits in red)

	Drought Year		Late Successional Year			
	Fall 2011		Spring 2019		Summer 2019	
Species Name	Average	Std Error	Average	Std Error	Average	Std Error
Agalinis heterophylla	0	0	0.013	0.009	0.006	0.006
Alternanthera philoxeroides	0.115	0.048	2.635	0.471	1.814	0.499
Ambrosia psilostachya	0.109	0.056	0.013	0.009	0.013	0.009
Ampelopsis arborea	0	0	0.006	0.006	0.006	0.006
Anagallis minima	0	0	0.013	0.009	0	0
Andropogon glomeratus	0.186	0.07	0.192	0.043	0.679	0.280
Azolla caroliniana	0	0	0.359	0.069	0.288	0.055
Baccharis hamimifolia	0	0	0.045	0.033	0.038	0.033
Bacopa monnieri	0	0	0.000	0.000	0.006	0.006
Calitriche peploides	0	0	0.013	0.009	0	0
Canna glauca	0	0	0.224	0.195	0	0
Carex longii	0	0	0.071	0.035	0.026	0.013
Carex triangularis	0	0	0.006	0.006	0	0
Centella erecta	0	0	0.199	0.192	0	0
Chamaesyce maculata	0.141	0.057	0	0	0	0
Chloris canterai	0.083	0.046	0	0	0	0
Cladium jamaicense	0	0	0	0	0.038	0.033
Crinum americanum	0.006	0.006	0	0	0	0
Croton capitatus	0.192	0.192	0	0	0	0
Cyclospermum leptophyllum	0.000	0.000	0.006	0.006	0	0
Cynodon dactylon	0.679	0.642	0.019	0.011	0.026	0.013
Cyperus acuminatus	0	0	0	0	0.006	0.006
Cyperus entrerianus	0	0	0.263	0.197	0.269	0.197
Cyperus haspen	0	0	0	0	0.109	0.048
Cyperus virens	0.391	0.086	1.981	0.307	1.641	0.239
Digitaria ciliaris	0.038	0.033	0	0	0	0
Diodia virginiana	0.006	0.006	0.019	0.011	0.026	0.013
Echinochloa colona	5.429	1.257	0	0	0	0
Echinochloa crusgalli	0.462	0.449	0	0	0	0
Echinochloa spp.	0.244	0.195	0	0	0	0
Eclipta prostrata	4.391	0.926	0.013	009	0.064	0.020
Eleocharis montana	0	0	0.045	0.033	0.026	0.013
Eleocharis montevidensis	0	0	0.686	0.454	0.513	0.206
Eleocharis quadrangulata	0.058	0.034	2.686	0.436	12.353	1.612
Eleocharis obtusa	0.192	0.192	0.147	0.057	0	0

Table 8 Continued - Average percent cover per species in a drought year and in two late successional time periods (native recruits in green, planted species in purple; invasive recruits in red)

	Drought Year		Late Successional Year			
	Fall 2011		Spring 2019		Summer 2019	
Species Name	Average	Std Error	Average	Std Error	Average	Std Error
Eryngium hookeri	0	0	0	0	0.006	0.006
Eupatorium capillifolium	0.006	0.006	0	0	0.006	0.006
Fimbristylis autumnalis	0	0	0	0	0.026	0.013
Fimbristylis miliacea	0	0	0	0	0.038	0.033
Galium tinctorium	0	0	0.006	0.006	0	0
Helianthus angustifolis	0	0	0	0	0.013	0.009
Heliotropium indicum	0.013	0.009	0	0	0	0
Heliotropium procumbens	1.590	0.532	0	0	0	0
Hibiscus lasiocarpos	0	0	0.006	0.006	0.006	0.006
Hydrolea ovata	0	0	0.071	0.035	0.179	0.065
Iris virginica	0.192	0.192	0	0	0	0
Iris virginica var. shrevei	0	0	0	0	0.006	0.006
Isoetes melanopoda	0	0	0.327	0.075	0.090	0.023
Isolepis carinata	0	0	0	0	0.051	0.034
Iva annua	6.462	1.359	0.058	0.019	0.263	0.193
Jacquemontia tamnifolia	0.006	0.006	0	0	0	0
Juncus acuminatus	0	0	1.449	0.155	0.564	0.057
Juncus diffusisimus	0	0	1.391	0.391	0.288	0.046
Juncus effusus	0	0	0.378	0.202	0.135	0.057
Juncus marginatus	0	0	0.058	0.034	0.026	0.013
Juncus nodatus	0	0	0.058	0.034	0.045	0.017
Juncus repens	0	0	0.327	0.202	0.122	0.056
Juncus validus	0	0	0.391	0.095	0.269	0.068
Krigia caespitosa	0	0	0.006	0.006	0	0
Landoltia punctata	0	0	0.013	0.009	0.051	0.018
Leersia hexandra	0	0	0.840	0.285	2.506	0.775
Lemna aequinoctialis	0	0	0.282	0.062	0.160	0.041
Leptochloa nealleyi	0	0	0	0	0.019	0.011
Limnosciadium pumilum	0	0	0.077	0.021	0.006	0.006
Lindernia dubia	0	0	0	0	0.045	0.017
Ludwigia glandulosa	0	0	0	0	0.096	0.047
Ludwigia decurrens	0	0	0.045	0.033	0.006	0.006
Ludwigia linearis	0	0	0	0	0.032	0.032
Ludwigia palustris	0	0	2.000	0.467	1.705	0.464
Melochia corchorifolia	0.455	0.099	0	0	0	0
Mikania scandens	0	0	1.103	0.235	2.288	0.471
Marsilea vestita	0	0	0.256	0.067	1.090	0.341
Oxalis dillenii	0.179	0.071	0.006	0.006	0	0
Oxalis spp.	0.064	0.019	0	0	0	0
Panicum dichotomiflorum	0.455	0.448	0.006	0.006	0	0
Panicum hemitomon	0.096	0.055	0.154	0.064	0.199	0.072
Panicum repens	3.955	1.044	0.000	0.000	0.609	0.609
Paspalum acuminatum	0	0	0.205	0.192	0.026	0.013
Paspalum urvillei	0.321	0.202	0.609	0.122	1.295	0.391
Paspalum denticulatum	0	0	0	0	0.006	0.006
Phyla nodiflora	0	0	0.058	0.034	0.071	0.035
Phyllanthus pudens	0.006	0.006	0	0	0	0
Physalis angulata	0.006	0.006	0	0	0	0
Physalis longifolia	0.013	0.009	0	0	0	0
Plantago virginica	0	0	0.006	0.006	0	0
Polygonum lapathifolium	0	0	0.122	0.056	0	0
Polygonum hydropiperoides	0	0	0.058	0.019	0.308	0.085
Polypremum procumbens	0.006	0.006	0	0	0	0
Pontederia cordata	0	0	1.955	0.439	2.647	0.821

Table 8 Continued - Average percent cover per species in a drought year and in two late successional time periods (native recruits in green, planted species in purple; invasive recruits in red)

	Drought Year		Late Successional Year			
	Fall 2011		Spring 2019		Summer 2019	
Species Name	Average	Std Error	Average	Std Error	Average	Std Error
Potamogeton diversifolius	0	0	0.026	0.013	0.006	0.006
Ptilimnium capillaceum	0	0	0.006	0.006	0	0
Pyrrhopappus pauciflorus	0	0	0	0	0.006	0.006
Ranunculus pusillus	0	0	0.013	0.009	0	0
Rhynchospora caduca	0	0	0.199	0.072	0.077	0.021
Rhynchospora colorata	0	0	0.006	0.006	0	0
Rhynchospora corniculata	0	0	0.199	0.072	0.340	0.090
Riccia stenophylla	0	0	0.038	0.015	0.032	0.014
Richardia spp.	0	0	0.006	0.006	0	0
Rorippa teres	0	0	0	0	0.013	0.009
Rotala ramosior	0	0	0	0	0.026	0.013
Rubus trivialis	0	0	0.250	0.195	0.064	0.045
Rudbeckia texana	0	0	0	0	0.192	0.192
Saccharum giganteum	0	0	0.231	0.195	0.282	0.197
Sagittaria graminea	0	0	0.051	0.034	0	0
Sagittaria longiloba	0	0	0	0	0.006	0.006
Sagittaria papillosa	0	0	0.058	0.019	0.051	0.034
Sagittaria platyphylla	0	0	0.346	0.085	0.891	0.286
Salvia lyrata	0	0	0.006	0.006	0.051	0.034
Schizophyllum commune	0	0	0.006	0.006	0	0
Schoenoplectus pungens	0	0	0.013	0.009	0.006	0.006
Sesbania drummondii	0	0	0.013	0.009	0.199	0.192
Sesbania herbacea	3.224	0.064	0.026	0.013	0.096	0.037
Solidago altissima	0	0	0.013	0.009	0	0
Spartina patens	0	0	0.686	0.610	0.551	0.451
Spirodela polyrhiza	0	0	0.103	0.024	0.096	0.037
Steinchisma hians	0	0	0.006	0.006	0.006	0.006
Stenotaphrum secundatum	0	0	0.224	0.195	0	0
Strophostyles spp.	0	0	0	0	0.808	0.637
Symphyotrichum divaricatum	0.910	0.287	0.019	0.011	0.006	0.006
Triadica sebifera	0	0	0.449	0.449	0.455	0.449
Typha domingensis	0	0	0.128	0.063	0.359	0.204
Typha latifolia	0	0	0.699	0.131	1.577	0.396
Urochloa platyphylla	0.737	0.221	0	0	0	0
Utricularia gibba	0	0	0.962	0.284	5.378	1.282
Utricularia radiata	0	0	0.205	0.052	4.981	1.303
Verbena brasiliensis	0.994	0.633	0	0	0	0

4. CONCLUSIONS AND DISCUSSION

Plant communities undergo change over time, driven by natural successional processes over time scales of months to years (Walker and Wardle 2014). Restored plant communities similarly change over time, but may require longer intervals (years to decades) to reach a target plant assemblage dominated by native recruits and/or targeted (i.e., planted) plant species (Choi 2004, Zedler and Callaway 1999, Matthews and Spyreas 2010). The analysis presented herein corroborates this assertion, where there was substantial change in the plant community from 2012 to 2019 across all seasons (Tables 4, 5). This analysis indicated that the restored plant community continued to change several years after restoration efforts had ended, and suggests that changes may continue to occur in subsequent years.

Across the board, the spring season demonstrated the highest R values (i.e. $R_{2013\text{-}2019}$ = 0.819), representing the most change. For the spring season, the plant species contributing most to the change in community include: *Eleocharis obtusa*, *Eleocharis quadrangulata*, and *Limnosciadium* spp. Annual recruits like *Eleocharis obtusa* and *Limnosciadium* spp. largely declined in relative abundance, whereas the most common planted species *Eleocharis quadrangulata* increased over time (Table 6). Colonizing annual recruits typically diminish as opportunities for recruitment decline (e.g., less disturbed area to colonize) while both native and planted perennial species recruited and persisted (Seabloom and van der Valk 2003). Plants like *Cyperus virens* and *Juncus acuminatus* are representative examples of species which could displace annual recruits and lead to substantial changes over time as was observed in this study (Appendix, spring).

The plant community changed dramatically from initial planting to the final monitoring, and the change may be largely attributed to species-specific life histories and growth forms. The top contributors to dissimilarity among years were *Eleocharis obtusa*, *Eleocharis quadrangulata* and *Limnosciadium* spp. Of those species, *Eleocharis obtusa* and *Limnosciadium* spp. were native, natural recruits, whereas *Eleocharis quadrangulata* was classified as a planted species. Both native recruits are considered annuals and were observed to decrease in presence from the earliest monitoring year to the latest, versus the planted perennial plant which increased in presence during the same period (Table 6). Both annual species are low-growing plants which do not compete well with taller, larger perennial plants, like *Eleocharis quadrangulata* (Correll and Correll 1972). As larger perennial plants grow and expand their range, the available area for low growing annuals to recruit, establish and grow diminishes (Shipley et.al. 1989), as this competitive edge of recruited perennial (native recruits and planted species) outcompetes other species. In addition, the planted perennial species had an automatic advantage over native recruits as they were planted at the adult stage, bypassing any need to establish from the seed bank, grow and develop roots. These interspecific competitive strategies, over time, can lead to substantial plant community flux, where natural recruits could give way to persistent native recruits and planted perennial species.

Planting a restored site, in contrast to natural recruitment, is an important part of the restoration "jump start" process (Young et. al. 2001), where planted species compete with both native and invasive natural recruits for establishment and/or dominance within the plant community. Observing the change over time across recruitment classes, the natural recruits contributed more to community change than did planted species or invasive recruits (Table 7), representing anywhere from 54.1% to 81.64% of the total dissimilarity between years, typically as a result of increasing abundance over time. The planted species represented between 8.12 to 29.68% of the total dissimilarity (Table 7), a notably lower value than native recruits. Intentionally planting the restoration site may have reintroduced these species to the landscape, but the reintroduction did not result in community dominance, as anticipated or desired.

As suggested previously, the different recruitment classes—native recruits, invasive recruits and planted species—compete and interact within the restoration site, changing its structure over time. Planted species are typically perennial (Table 1), and when planted into the restoration site, are plugs (small subdivisions) of a fully grown plant (Methods section). Each plug is also planted with a high amount of root material per plug. Once placed, the planted plug with intact roots and fully developed leaf and stem structure retains important characteristics to jump start and ensure their survival. Thus, planted species are afforded the benefits of already-developed plant structures. In contrast, native natural recruits are dependent on their successful life history strategies to establish within a restoration site. These strategies, such as persistent seed bank presence, prolific seed production and effective seed dispersal, provide necessary “tools” for native natural recruits to enter the restoration site, and remain within the site over time (Galatowitsch 2006). For a restoration site like Sheldon Lake State Park, where wetlands had previously occurred at the site and the preserved seed banks were re-exposed to ideal light and water conditions during the restoration process, long-dormant seeds of native species could potentially re-emerge and re-establish.

In contrast, invasive plants owe their potential overall success to their inherent ability to out-compete other native recruits—and potentially planted species--and dominate the landscape (Zedler and Kercher 2002) over time. Typically, their competitive edge arises from mechanisms such as excessive seed production or higher growth rates driven by faster resource utilization, which increase their recruitment and/or long-term establishment competitive edge (Ren and Zhang 2009, Spencer and Coulson 1976). It was originally expected that invasive recruits would surpass all native recruits and planted species over time, due to invasive species' more aggressive recruitment and establishment strategies. However, the invasive recruits contributed relatively little (less than 20%) to the overall dissimilarity between compared years (Table 7). Several of the most aggressive invasive species which utilize these aggressive strategic mechanisms, such as *Paspalum urvillei* or *Alternanthera philoxeroides*, had relatively low abundances and did not change substantially over time (Appendix). This outcome was not expected; the hypothesized dominance of invasive recruits simply did not occur within the ten-year sampling period.

The overall low presence of the invasive recruits could be attributed to the initial proliferation of native natural recruits once the seed bank was re-exposed suitable ideal environmental conditions. The wetland seed bank had been buried by agricultural and other development processes in the 1940s at Sheldon Lake State Park, and at such time, many of the noted invasives were present in the United States (Commonwealth Agricultural Bureaux International 2021) but did not occur in the immediate area, and therefore would not have recruited to the seed banks of these isolated ephemeral wetlands. Thus, invasive recruits, although aggressive competitors, would have lagged behind the initial colonization by native recruits and the intentional placement of planted species.

Similarly, the exceptional drought of 2011 may have greatly inhibited the establishment of the invasive recruits. In river basins, hydrological drought can last longer than a corresponding meteorological drought episode, where aquatic systems may require an extended period to recover from the drought (Liu et. al. 2020). The extensive length (16 months) of the drought in this case would reduce the successful water-borne recruitment of any viable seed and minimize favorable conditions for establishment (e.g., moist soil). In addition, most invasive recruits depend on prolific seed production as a recruitment strategy, but the drought likely reduced invasive species fitness and reduced seed output. Therefore, coupled with targeted management of invasive recruits (see methods), the drought may have effectively impeded invasive introduction and establishment at this restoration site.

Likewise, the 2011 drought may have temporarily slowed the growth and reproduction of species that were planted prior to the drought. Aquatic plants, like *Eleocharis quadrangulata* and *Canna glauca*, from the planted recruitment class could experience significant drought stress with decreased turgor and slowing or halting of cell growth (Touchette et. al 2007) while also experiencing a stimulation for root growth (Molyneux and Davies 1983). The potential stimulation ensured that these planted species could endure the long nature of the drought by retaining critical root structure. Even with desiccation stress on the uppermost part of the plants, these species could recover from the changes in their physiology with an extended recovery period. Given the "most intense one year drought in Texas" (Nielsen-Gammon 2011), the extended recovery period slowed the planted species ability to re-establish and out-compete native natural recruits.

For some planted species like *Canna glauca,* an obligate deep-water species, the drought's effects—complete water loss--would have been devastating over the 1-year time frame. This type of planted species would simply not have survived the extended lack of water. *Crinum americanum*, a deep water planted species was initially planted within the project site, however, it was no longer recorded after 2014 (pers. obs.) and is consistent with other regional prairie wetland observations (van der Valk 2005). Further, other deep-water species were not recorded in later successional years, including *Thalia dealbata* and *Nymphaea odorata* (Table 8). Thus, deep-water planted species disproportionately experienced the significant effect of drought during the project time line.

The drought event, while significant, could be considered a disturbance event to which those prior-established plants, both native recruits and planted species, are adapted (Brock et. al. 2003). The recovery of both recruitment groups may be a reflection of their adapted resilience to the fluctuating wet-dry cycles of these restored wetlands, which are ephemeral systems by nature. The sexual reproduction of seeds which in turn generates multigenerational propagules (Bornette and Puijalon 2011, Brock et. al 2003) and the "long-lived" ability of seed to resist desiccation (Humphries and Baldwin 2003) are both mechanisms toward the long-term establishment and success for prairie wetland species. Specifically, the plants *Cyperus virens* (planted), *Eleocharis quadrangulata* (planted) and *Juncus validus* (native recruit) are capable of producing many viable, persistent seeds (Correll and Correll 1972). Those same seeds which dropped into the soils of the restored wetland basins upon initial restoration and subsequent germinating years from both the prior existing seed bank *and* the planted species, likely remained dormant until the arrival of rain upon the conclusion of the drought (Table 8).

Even with the harsh effects of the drought, a majority of the planted species continued to survive (Table 8), only partly supported by additional plantings (Table 2). This supplemental planting likely contributed to the persistence of planted species in the later successional years (Table 8).

The large discrepancy in cover and expansion among the different recruitment classes, where native recruits exceeded both invasive and planted species, could suggest to the practitioner that any planting effort within a wetland restoration project is not necessary. Certainly, this would be consistent with observations and recommendations made to date (Prach et. al. 2001, van der Valk 1981) and are magnified by guidelines where only “revegetation potential” is considered (Galatowitsch and van der Valk 1994). Further, Mitsch et al. (1998) advocates that planting a restored site is unnecessary given that the trajectories of a planted and unplanted wetland eventually converge. However, in the case of wetland restoration projects where buried seed bank material may have unknown viability due to the length of burial time (Erlandson 1987, van der Valk et. al. 1992), and barriers to wetland seed dispersal (i.e., surrounding upland forests), planting a restored site may not only be a necessity but warranted.

At the time of restoration planning for the Sheldon Lake project, it was unknown whether the wetland seed bank would be present or viable. The project site had been under agricultural production for over 70 years and the low-lying wetlands buried for the same length of time, suggesting to the project planners that any remnant seed bank would have been eradicated by repeated tilling or prolonged burial. Therefore, a lacking seed bank coupled with minimal natural recruitment from surrounding remnant wetlands (the state park is surrounded by industrial development to the north, south and west, and residential development to the east) were indicators that it could be necessary to plant native species in order to ensure those target species were components of the restored plant community.

Additional support warranting the planting of the restoration site arises from the persistence of planted species throughout the monitoring cycle. Even though seven planted species (Table 2) were never recorded in the monitoring data, 25 of the recorded 31 planted species—representing more than half of the original targeted species list- persisted and several key species like *Eleocharis quadrangulata* and *Pontedaria cordata*, thrived (Table 8, late successional year). Despite The presence of the planted species, at a minimum, increased the diversity of the overall site, ensuring certain species, typical of coastal prairie wetlands, survived.

At Sheldon Lake, one of the main goals of the restoration for the landowner (TPWD) was the "restoration of the historic landscape" which had been significantly altered or removed, such that less than one percent of the original 2.5 million acres remained (U.S. Fish and Wildlife Service and U.S. Geological Survey 2000). A secondary goal was the provision of refugia for plant species and associated bird species (Texas Parks and Wildlife Department 2003). For plant species removed or extirpated by the previous 70 years of agricultural development, the Sheldon Lake State Park wetland restoration project was the opportunity to replace "lost" species like *Eleocharis quadrangulata*, *Leersia hexandra* and *Pontedaria cordata* back into the landscape (Table 8). These plant species provide critical habitat for bird species such as American Bitterns and Little Blue Herons (VanRees-Siewert and Dinsmore 1996, Weber et. al. 1982). This refugia goal is especially critical when considering that freshwater wetland dependent bird populations (world-wide) have been and continue to deteriorate (Millennium Ecosystem Assessment Program 2005).

The restored coastal prairie wetland complex similarly provided refugia for additional species, including small mammals and amphibians. Approximately two years post restoration planting, a short-term baseline study of rodents was conducted by TPWD Urban Wildlife biologist which captured various rodents (e.g. Hispid rat, fulvous harvest mouse) traversing around and between restored ponds within Phase 2 and 3 (Norrid 2015). Additionally, multiple observation of Ranid species, specifically *Pseudacris fouquettei* and *Pseudacris crucifer*, were observed within the restoration project site, confirming the use of these amphibian population (K. Norrid, personal communication, October 15, 2021). These supplemental observations further support the restoration effort.

The two primary restoration goals—landscape-level and species-level restoration--were accomplished and quantified within this project. The collected monitoring data suggested a predominance of native recruits with a thriving supplement of planted species. These restored wetlands, in turn, provided a basic means for colonization and usage of local native fauna. It is the combination of this accrued desired vegetation and subsequent faunal usage within the landscape which compellingly support the whole restoration effort as a necessity and a success.

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APPENDIX

TOP TEN CONTRIBUTING PLANT SPECIES TO TOTAL DISSIMILARITY BY SEASON

SPRING					
Year pairs of interest	R value	Species	% Contribution to dissimilarity		Recruitment Group
2013-2014	0.661	*Limnosciadium pumilum*	9.85	↑	Recruit
		Isoetes melanopoda	7.01	↑	Recruit
		Eleocharis obtusa	6.01	↓	Recruit
		Limnosciadium spp.	5.35	↓	Recruit
		Callitriche heterophylla	4.65	↓	Recruit
		Ranunculus spp.	3.8	↓	Recruit
		Leptochloa nealleyi	3.1	↓	Recruit
		Ambrosia psilostachya	2.87	↑	Recruit
		Iva annua	2.66	↓	Recruit
		Ludwigia palustris	2.61	↑	Recruit
2013-2015	0.827	*Ludwigia palustris*	8.45	↑	Recruit
		Isoetes melanopoda	5.74	↑	Recruit
		Eleocharis obtusa	4.84	↓	Recruit
		Cyperus virens	4.74	↑	Planted
		Juncus diffusissimus	4.56	↑	Recruit
		Limnosciadium spp.	4.44	↓	Recruit
		Juncus marginatus	3.96	↑	Planted
		Eleocharis quadrangulata	3.65	↑	Planted
		Limnosciadium pumilum	3.63	↑	Recruit
		Callitriche heterophylla	3.4	↓	Recruit
2013-2016	0.764	*Juncus validus*	9.13	↑	Recruit
		Cyperus virens	6.34	↑	Planted
		Ludwigia palustris	6.33	↑	Recruit
		Eleocharis obtusa	5.54	↓	Recruit
		Limnosciadium spp.	4.97	↓	Recruit
		Eleocharis quadrangulata	3.91	↑	Planted
		Callitriche heterophylla	3.83	↓	Recruit
		Paspalum urvillei	3.57	↑	Invasive
		Ranunculus spp.	3.49	↓	Recruit
		Isoetes melanopoda	3.22	↑	Recruit
2013-2017	0.763	*Cyperus virens*	7.76	↑	Planted
		Juncus validus	6.16	↑	Recruit
		Eleocharis obtusa	5.61	↓	Recruit
		Limnosciadium spp.	5.07	↓	Recruit
		Eleocharis quadrangulata	3.98	↑	Planted
		Callitriche heterophylla	3.83	↓	Recruit
		Ludwigia palustris	3.69	↑	Recruit
		Ranunculus spp.	3.57	↓	Recruit
		Paspalum urvillei	3.16	↑	Invasive
		Isoetes melanopoda	2.91	↑	Recruit
2013-2019	0.819	*Eleocharis obtusa*	5.71	↓	Recruit
		Eleocharis quadrangulata	5.66	↑	Planted
		Limnosciadium spp.	5.1	↓	Recruit
		Alternanthera philoxeroides	4.6	↑	Invasive
		Cyperus virens	4.3	↑	Planted
		Juncus acuminatus	3.98	↑	Recruit
		Callitriche heterophylla	3.97	↓	Recruit
		Ranunculus spp.	3.59	↓	Recruit
		Ludwigia palustris	3.42	↑	Recruit
		Pontedaria cordata	2.95	↑	Planted
2014-2015	0.362	*Limnosciadium pumilum*	9.88	↓	Recruit
		Ludwigia palustris	9.04	↑	Recruit
		Isoetes melanopoda	6.53	↑	Recruit
		Cyperus virens	5.14	↑	Planted
		Juncus diffusissimus	4.91	↑	Recruit

SPRING

Year pairs of interest	R value	Species	% Contribution to dissimilarity		Recruitment Group
		Eleocharis quadrangulata	4.25	↑	Planted
		Juncus marginatus	4.24	↑	Planted
		Callitriche heterophylla	3.2	↓	Recruit
		Marsilea vestita	2.8	↑	Recruit
		Alternanthera philoxeroides	2.5	↑	Invasive
2014-2016	0.539	*Limnosciadium pumilum*	9.41	↓	Recruit
		Juncus validus	9.28	↑	Recruit
		Ludwigia palustris	6.92	↑	Recruit
		Cyperus virens	6.41	↑	Planted
		Isoetes melanopoda	6.06	↓	Recruit
		Eleocharis quadrangulata	4.28	↑	Planted
		Callitriche heterophylla	3.42	↓	Recruit
		Alternanthera philoxeroides	3.34	↑	Invasive
		Paspalum urvillei	3.12	↑	Invasive
		Juncus diffusissimus	2.98	↑	Recruit
2014-2017	0.509	*Limnosciadium pumilum*	9.69	↓	Recruit
		Cyperus virens	7.86	↑	Planted
		Juncus validus	6.3	↑	Recruit
		Isoetes melanopoda	6.19	↓	Recruit
		Ludwigia palustris	4.71	↑	Recruit
		Eleocharis quadrangulata	4.28	↑	Planted
		Callitriche heterophylla	3.58	↓	Recruit
		Alternanthera philoxeroides	3.16	↑	Invasive
		Marsilea vestita	3.05	↑	Recruit
		Paspalum urvillei	2.71	↑	Invasive
2014-2019	0.655	*Limnosciadium pumilum*	9.41	↑	Recruit
		Isoetes melanopoda	6.33	↓	Recruit
		Eleocharis quadrangulata	5.57	↑	Planted
		Alternanthera philoxeroides	4.72	↑	Invasive
		Cyperus virens	4.35	↑	Planted
		Ludwigia palustris	4.35	↑	Recruit
		Juncus acuminatus	3.93	↑	Recruit
		Callitriche heterophylla	3.35	↓	Recruit
		Pontedaria cordata	3.07	↑	Planted
		Ambrosia psilostachya	2.4	↓	Recruit
2015-2017	0.261	*Ludwigia palustris*	9.03	↓	Recruit
		Cyperus virens	8.07	↑	Planted
		Juncus validus	6.16	↑	Recruit
		Eleocharis quadrangulata	5.45	↑	Planted
		Isoetes melanopoda	5.21	↓	Recruit
		Juncus diffusissimus	4.95	↓	Recruit
		Juncus marginatus	4.28	↓	Planted
		Limnosciadium pumilum	4.04	↓	Recruit
		Marsilea vestita	3.52	↑	Recruit
		Paspalum urvillei	3.07	↑	Invasive
2015-2019	0.403	*Ludwigia palustris*	8.73	↓	Recruit
		Eleocharis quadrangulata	6.28	↑	Planted
		Cyperus virens	5.78	↓	Planted
		Juncus diffusissimus	5.68	↓	Recruit
		Isoetes melanopoda	5.63	↓	Recruit
		Alternanthera philoxeroides	4.36	↑	Invasive
		Juncus marginatus	4.24	↓	Planted
		Limnosciadium pumilum	3.87	↓	Recruit
		Juncus acuminatus	3.6	↑	Recruit
		Pontedaria cordata	3.06	↑	Planted
2016-2019	0.274	*Juncus validus*	9.64	↓	Recruit
		Ludwigia palustris	7.45	↓	Recruit
		Cyperus virens	6.99	↓	Planted
		Eleocharis quadrangulata	6.61	↑	Planted
		Alternanthera philoxeroides	5.28	↑	Invasive
		Juncus diffusissimus	4.36	↓	Recruit

SPRING

Year pairs of interest	R value	Species	% Contribution to dissimilarity		Recruitment Group
		Pontedaria cordata	4.24	↑	Planted
		Juncus acuminatus	4.11	↑	Recruit
		Paspalum urvillei	3.78	↓	Invasive
		Isoetes melanopoda	3.53	↓	Recruit

SUMMER

Year pairs of interest	Global R Value	Species	% Contribution to Dissimilarity		Functional Category
2012-2014 Avg Diss = 88.43	0.296	*Iva annua*	15.65	↓	Recruit
		Eleocharis obtusa	4.74	↓	Recruit
		Ludwigia palustris	4.43	↑	Recruit
		Sesbania herbacea	3.87	↓	Recruit
		Cyperus virens	3.85	↑	Planted
		Alternanthera philoxeroides	3.68	↓	Invasive
		Paspalum urvillei	3.63	↑	Invasive
		Marsilea vestita	3.26	↑	Recruit
		Symphyotrichum divaricatum	3.23	↓	Recruit
		Eleocharis quadrangulata	3.23	↑	Planted
2012-2016 Avg Diss = 94.73	0.646	*Iva annua*	14.28	↓	Recruit
		Eleocharis quadrangulata	7.2	↑	Planted
		Ludwigia palustris	7.15	↑	Recruit
		Cyperus virens	6.18	↑	Planted
		Eleocharis obtusa	4.35	↓	Recruit
		Juncus acuminatus	4.14	↑	Recruit
		Alternanthera philoxeroides	3.97	↓	Invasive
		Paspalum urvillei	3.89	↑	Invasive
		Sesbania herbacea	2.89	↓	Recruit
		Isoetes melanopoda	2.88	↑	Recruit
2012-2018 Avg Diss = 93.57	0.56	*Iva annua*	15.19	↓	Recruit
		Eleocharis quadrangulata	8.95	↑	Planted
		Alternanthera philoxeroides	5.19	↑	Invasive
		Cyperus virens	5.1	↑	Planted
		Eleocharis obtusa	4.61	↓	Recruit
		Paspalum urvillei	4.37	↑	Invasive
		Sesbania herbacea	3.3	↓	Recruit
		Marsilea vestita	3.05	↑	Recruit
		Pontedaria cordata	2.64	↑	Planted
		Ludwigia palustris	2.46	↑	Recruit
2012-2019 Avg Diss = 94.65	0.666	*Iva annua*	13.88	↓	Recruit
		Eleocharis quadrangulata	9.8	↑	Planted
		Alternanthera philoxeroides	4.28	↑	Invasive
		Eleocharis obtusa	4.24	↓	Recruit
		Cyperus virens	3.72	↑	Planted
		Mikania scandens	3.29	↑	Recruit
		Utricularia radiata	3.08	↑	Recruit
		Utricularia gibba	2.95	↑	Recruit
		Paspalum urvillei	2.92	↓	Invasive
		Sesbania herbacea	2.89	↓	Recruit
2013-2014 Avg Diss = 85.69	0.256	*Iva annua*	9.85	↓	Recruit
		Symphyotrichum divaricatum	6.62	↓	Recruit
		Leptochloa nealleyi	5.98	↓	Recruit
		Oxalis spp.	4.81	↓	Recruit

SUMMER					
Year pairs of interest	Global R Value	Species	% Contribution to Dissimilarity		Functional Category
		Paspalum urvillei	4.63	↓	Invasive
		Ludwigia palustris	4.28	↑	Recruit
		Marsilea vestita	3.84	↑	Recruit
		Ambrosia psilostachya	3.81	↑	Recruit
		Coreopsis tinctoria	3.8	↓	Recruit
		Cyperus virens	3.65	↑	Planted
2013-2016 Avg Diss = 93.69	0.704	*Iva annua*	8.67	↓	Recruit
		Symphyotrichum divaricatum	6.48	↓	Recruit
		Ludwigia palustris	6.32	↑	Recruit
		Eleocharis quadrangulata	6.09	↑	Planted
		Cyperus virens	5.47	↑	Planted
		Leptochloa nealleyi	5.32	↓	Recruit
		Paspalum urvillei	4.63	↓	Invasive
		Oxalis spp.	4.39	↓	Recruit
		Juncus acuminatus	3.65	↑	Recruit
		Coreopsis tinctoria	2.91	↓	Recruit
2013-2018 Avg Diss = 92.88	0.653	*Iva annua*	9.09	↓	Recruit
		Eleocharis quadrangulata	7.56	↑	Planted
		Symphyotrichum divaricatum	6.75	↓	Recruit
		Leptochloa nealleyi	5.52	↓	Recruit
		Paspalum urvillei	4.95	↓	Invasive
		Oxalis spp.	4.61	↓	Recruit
		Cyperus virens	4.55	↑	Planted
		Alternanthera philoxeroides	3.81	↑	Invasive
		Marsilea vestita	3.45	↑	Recruit
		Coreopsis tinctoria	3.04	↓	Recruit
2013-2019 Avg Diss = 94.25	0.758	*Eleocharis quadrangulata*	8.43	↑	Planted
		Iva annua	8.43	↓	Recruit
		Symphyotrichum divaricatum	6.25	↓	Recruit
		Leptochloa nealleyi	4.97	↓	Recruit
		Oxalis spp.	4.24	↓	Recruit
		Paspalum urvillei	3.82	↓	Invasive
		Cyperus virens	3.35	↑	Planted
		Alternanthera philoxeroides	2.9	↑	Invasive
		Mikania scandens	2.88	↑	Recruit
		Coreopsis tinctoria	2.81	↓	Recruit
2014-2016 Avg Diss = 87.58	0.404	*Ludwigia palustris*	7.65	↑	Recruit
		Eleocharis quadrangulata	7.58	↑	Planted
		Cyperus virens	6.1	↑	Planted
		Iva annua	5.44	↓	Recruit
		Marsilea vestita	4.3	↓	Recruit
		Paspalum urvillei	4.27	↑	Invasive
		Juncus acuminatus	4.13	↑	Recruit
		Isoetes melanopoda	3.62	↑	Recruit
		Eleocharis montevidensis	3.59	↓	Planted
		Juncus validus	3.08	↑	Recruit
2014-2018 Avg Diss = 90.13	0.447	*Eleocharis quadrangulata*	8.81	↑	Planted
		Iva annua	5.56	↓	Recruit
		Cyperus virens	5.15	↑	Planted
		Ludwigia palustris	5.09	↓	Recruit
		Marsilea vestita	4.94	↓	Recruit
		Paspalum urvillei	4.47	↑	Invasive
		Alternanthera philoxeroides	4.2	↑	Invasive

SUMMER					
Year pairs of interest	Global R Value	Species	% Contribution to Dissimilarity		Functional Category
		Eleocharis montevidensis	3.18	↓	Planted
		Ambrosia psilostachya	2.81	↓	Recruit
		Pontedaria cordata	2.58	↑	Planted
2014-2019 Avg Diss = 90.63	0.528	*Eleocharis quadrangulata*	9.64	↑	Planted
		Iva annua	5.15	↓	Recruit
		Ludwigia palustris	4.62	↓	Recruit
		Cyperus virens	3.89	↑	Planted
		Marsilea vestita	3.87	↓	Recruit
		Alternanthera philoxeroides	3.26	↑	Invasive
		Utricularia gibba	3.23	↑	Recruit
		Paspalum urvillei	3.22	↓	Invasive
		Eleocharis montevidensis	3.2	↓	Planted
		Mikania scandens	3.18	↑	Recruit

FALL					
Year pairs of interest	R value	Species	% Contribution to Dissimilarity		Functional Category
2012-2015 Avg Diss = 90.74	0.421	*Iva annua*	13.95	↓	Recruit
		Cyperus virens	6.76	↑	Planted
		Eleocharis quadrangulata	6.47	↑	Planted
		Ludwigia palustris	6.2	↑	Recruit
		Marsilea vestita	5.39	↑	Recruit
		Eleocharis montevidensis	4.88	↑	Planted
		Sesbania herbacea	4.46	↓	Recruit
		Panicum dichotomiflorum	4.18	↓	Recruit
		Fimbristylis autumnalis	3.74	↓	Recruit
		Symphyotrichum divaricatum	3.58	↓	Recruit
2013-2015 Avg Diss = 90.80	0.495	*Iva annua*	11.14	↓	Recruit
		Symphyotrichum divaricatum	7.24	↓	Recruit
		Cyperus virens	6.73	↑	Planted
		Eleocharis quadrangulata	6.7	↑	Planted
		Ludwigia palustris	6.28	↑	Recruit
		Marsilea vestita	5.52	↑	Recruit
		Leptochloa nealleyi	4.47	↓	Recruit
		Eleocharis montevidensis	4.1	↑	Planted
		Paspalum urvillei	3.54	↑	Invasive
		Sesbania herbacea	3.3	↓	Recruit

WINTER					
Year pairs of interest	R value	Species	% Contribution to Dissimilarity		Functional Category
2013-2014 Avg diss = 92.21	0.378	*Ludwigia palustris*	9.44	↑	Recruit
		Limnosciadium pumilum	7.86	↓	Recruit
		Cyperus virens	5.94	↑	Planted
		Isoetes melanopoda	5.42	↑	Recruit
		Marsilea vestita	5.4	↑	Recruit
		Eleocharis quadrangulata	5.08	↑	Planted
		Juncus scirpoides	4.69	↑	Recruit
		Symphyotrichum divaricatum	4.37	↓	Recruit
		Juncus validus	4.02	↑	Recruit

WINTER					
Year pairs of interest	R value	Species	% Contribution to Dissimilarity		Functional Category
		Paspalum urvillei	3.1	↑	Invasive
2013-2015 Avg Diss = 92.79	0.568	*Juncus validus*	10.54	↑	Recruit
		Cyperus virens	9.7	↑	Planted
		Limnosciadium pumilum	7.39	↓	Recruit
		Marsilea vestita	7.33	↑	Planted
		Ludwigia palustris	5.21	↑	Recruit
		Isoetes melanopoda	5.15	↑	Recruit
		Symphyotrichum divaricatum	4.52	↓	Recruit
		Paspalum urvillei	3.82	↑	Invasive
		Andropogon glomeratus	3.73	↑	Recruit
		Alternanthera philoxeroides	2.95	↑	Invasive
2013-2016 Avg Diss = 94.76	0.615	*Cyperus virens*	9.52	↑	Planted
		Juncus validus	7.66	↑	Recruit
		Eleocharis quadrangulata	7.05	↑	Planted
		Limnosciadium pumilum	6.94	↓	Recruit
		Paspalum urvillei	6.13	↑	Invasive
		Isoetes melanopoda	4.33	↑	Recruit
		Symphyotrichum divaricatum	4.26	↓	Recruit
		Alternanthera philoxeroides	3.16	↑	Invasive
		Utricularia radiata	2.61	↑	Recruit
		Ambrosia psilostachya	2.32	↓	Recruit
2013-2018 Avg Diss = 94.39	0.581	*Eleocharis quadrangulata*	8.85	↑	Planted
		Limnosciadium pumilum	7.37	↓	Recruit
		Cyperus virens	5.15	↑	Planted
		Marsilea vestita	5.1	↑	Recruit
		Symphyotrichum divaricatum	4.56	↓	Recruit
		Alternanthera philoxeroides	4.3	↑	Invasive
		Juncus validus	3.92	↑	Recruit
		Ludwigia palustris	3.31	↑	Recruit
		Paspalum urvillei	2.93	↑	Invasive
		Pontedaria cordata	2.9	↑	Planted
2014-2018 Avg Diss = 87.54	0.26	*Ludwigia palustris*	9.48	↓	Recruit
		Eleocharis quadrangulata	9.42	↑	Planted
		Cyperus virens	6.9	↓	Planted
		Marsilea vestita	6.09	↓	Recruit
		Juncus validus	5.61	↓	Recruit
		Isoetes melanopoda	4.58	↓	Recruit
		Juncus scirpoides	4.51	↓	Recruit
		Alternanthera philoxeroides	4.08	↑	Invasive
		Andropogon glomeratus	3.29	↓	Recruit
		Paspalum urvillei	3.25	↓	Invasive
2015-2016 Avg Diss = 80.82	0.251	*Cyperus virens*	12.48	↑	Planted
		Juncus validus	11.33	↓	Recruit
		Paspalum urvillei	7.32	↑	Invasive
		Eleocharis quadrangulata	7.18	↑	Planted
		Marsilea vestita	6.07	↓	Recruit
		Ludwigia palustris	5.14	↓	Recruit
		Isoetes melanopoda	4.13	↓	Recruit
		Andropogon glomeratus	4.05	↓	Recruit
		Alternanthera philoxeroides	3.55	↑	Invasive
		Utricularia radiata	2.73	↑	Recruit