The Long-term Development of Wetland Plant Communities with Water Deliveries in a Created Wetland in the Desert Southwest

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Abstract

In the middle of the Rio Grande, valuable wetlands and riparian areas have been replaced by urban and agricultural land, as well as the channelization of the river along the US-Mexico border. The Rio Bosque wetlands were constructed in El Paso, TX to mitigate the loss of these habitats. Unfortunately, these wetlands faced issues with water availability until 2016, when they began to receive the resource more frequently. Vegetation was allowed to establish naturally over time in response to winter water deliveries (2005–2014) and summer water deliveries (2016-7). This study investigated how changes in water delivery influenced the plant community development over more than a decade of restoration activities. We observed that water depth was an important predictor of increasing the relative frequency and cover of wetland plants, as well as macrophyte diversity. An NMDS ordination showed a transition of the vegetation community from limited water years to more regular water availability. The plant community changed from one dominated by upland plants, including invasive tumbleweed (*Salsola tragus*), to one dominated by tolerant, competitive wetland plants (*Lemna* spp and *Polygonum lapathifolium*). Deeper areas had a greater proportion of wetland plants, while facultative wetland plants occurred on the edges of ponds. In general, this 12-year study has helped to understand how the Rio Bosque ecosystem has changed after more regular water availability and will assist managers in tracking the future recovery and restoration of this valuable desert wetland.

1. Introduction

Riparian ecosystems in the southwestern US have been drastically altered over the past century, through diversion, groundwater pumping, damming, and anthropogenic inputs (Patten, 1998; Stromberg, 2001). Restoration of these habitats is complicated by several factors, including competing pressures on water resources, such that environmental needs can be at odds with irrigated agriculture and other consumptive uses. Strategies that could be used to restore arid region riparian ecosystem structure and function include releasing treated municipal effluent into stream channels, utilizing agricultural return flows to promote riparian growth, and diversion of partially treated waters into wetland treatment cells (Stromberg, 2001).

With the widespread loss of wetlands and riparian zones, many valuable ecosystem functions, such as nutrient retention, sediment attenuation, biodiversity support, and flood abatement, have also been lost (Zedler & Kercher, 2005). Wetland mitigation aims to replace wetlands that are destroyed or degraded by human development. Macrophyte species are often used as a key indicator of wetland restoration success, which can depend on many factors (Albert & Minc, 2004; Taddeo & Dronova, 2018). For example, in desert wetland restoration processes, the change in vegetation composition from xerophytes to aquatic species could indicate the return of vital factors, such as the availability of water (Sher et al., 2002; Long et al., 2003). Unfortunately, however, recreating ecosystem functions in wetland restoration or creation projects has often met with a low level of success (Race & Fonseca, 1996).

In the absence of plantings, the establishment of vegetation in created wetlands (CW) depends both on hydrology as well as seed dispersal (by waterfowl or others) or existing seedbanks. The new plant composition is often similar to nearby natural wetlands and may increase over time (Galatowitsch & van der Valk, 1996). However, CWs impose environmental limitations on their vegetation. Plant composition is affected by competitive skill, resource availability, and tolerance to residual pollution (Hadad et al., 2006). Because of often nutrient-rich and artificial conditions, some plant species may fail to develop or persist in CW (Saggaï et al., 2017). The frequency of flooding and extended drying may alter the structure and composition of aquatic communities (Wassens et al., 2017). Studies on ecological dynamics need to be conducted in desert CW, where water availability and precipitation are often limited. It is also vital to understand this ecosystem's vulnerability for effective management restoration and conservation (Lindenmayer et al., 2012).

The Rio Grande, as it passes through the Chihuahuan Desert, was formerly rich in riparian forests and wetlands but has been channelized for agricultural irrigation and stabilization of the international boundary between the US and Mexico (Watts et al., 2002). Wetland and riparian areas have largely been replaced by urban and agricultural areas in the middle Rio Grande (Roelle & Hagenbuck, 1995). Few natural wetlands or riparian habitats exist in the highly urbanized area surrounding the El Paso (USA) – Ciudad Juárez (Mexico) metroplex, one of the largest border communities in the world. The loss of riparian vegetation has been blamed in part for elevated nutrient levels throughout the middle Rio Grande (Oelsner et al., 2007; Rodriguez & Lougheed, 2010), which is contaminated by heavy metals (Rios-Arana et al., 2004; Ordonez et al., 2011), dominated by tolerant macroinvertebrates (Ordonez et al., 2011), and exceeds state standards for nutrient and bacterial pollution and total dissolved solids (IBWC 2008).

The Rio Bosque (RB) wetlands in El Paso, Texas, were created to mitigate wetlands and other habitats lost in the construction of irrigation canals along the Rio Grande. Located in the Rio Bosque Wetland Park, which was designed to recreate examples of river valley habitats in the region (Watts et al. 2002), these wetlands are unique and valuable models of riparian ecological communities in the southwest. For example, the park now hosts more than 260 species of birds in both upland and riparian habitats. In the early 2000s (2001–2002), water was delivered to the park almost year-round; however, the park then faced numerous challenges, not the least of which is that for more than a decade (2003–2015), the wetlands received water, in the form of treated wastewater effluent, only during the fall and winter months (Nov-Jan). During the rest of the year, most of the water in the region was allotted for agricultural use. Precipitation is not an important inflow to the Rio Bosque, located in the Chihuahuan Desert, and has an average annual rainfall of approximately 22 cm per year. In 2016, however, the RB secured additional water for the park, so the wetlands are largely flooded with effluent during winter (Nov-Jan) and spring/summer (May-Sept). During most of this time, smaller amounts of water from groundwater wells were also delivered into riverine channels in the park during spring and summer. This study examines the impact of the changes in water deliveries on plant community development within the wetland ponds in the park over a 12-year period (2005–2017). In particular, we aim to determine whether unique plant communities have developed in wetland cells and how bathymetry plays a role in plant community development.

2. Methods

2.1 Vegetation surveys

A 50x50m grid was created across the RBWP landscape to help select a subset of sample locations in the ponds and pond edge areas. This study focuses on a series of 53 1m^2 quadrats found within wetland ponds 1 and 2 (Fig. 1). Not all quadrats were sampled every year (2005 n = 48; 2009 n = 14; 2014 n = 28; 2016 n = 29; 2017 n = 32). This includes quadrats flooded regularly and those on the edges of the wetland cells. Edges were defined as quadrats that may have been flooded during the winter but were reported to have zero water depth during the summer sampling season. The quadrats were sampled in August, beginning in 2005 and continuing in 2009, 2014, 2016, and 2017. The cover of plants within a 1-m^2 quadrat was recorded using a modified Braun-Blanquet cover scale. Plants were identified on site or voucher specimens were taken to the lab for further analysis. The Wetland Indicator Status of each plant was determined using the USACE's Regional Wetland Plant List for the Arid West Region (Lichvar et al., 2016). This classification is based on the species occurrence in wetlands under natural conditions; obligate (OBL) species appear almost always in wetlands, facultative wetland (FACW) plants usually appear in wetlands but are occasionally found in non-wetlands, facultative (FAC) plants occur equally in wetlands and non-wetlands, facultative upland (FACU) plants mostly appear in non-wetlands and obligate upland (UPL) plants grow in non-wetlands. We calculated the relative cover and relative frequency of wetland plants, which were classified as those with OBL or FACW designations.

2.2 Additional environmental parameters

The bathymetry of the ponds was surveyed in 2009, 2016, and 2017. Depth was surveyed on a 50x50 (2009) or 20x20m (2016–2017) grid; locations were confirmed using a sub-meter accuracy GPS unit. Water depth was recorded at each point and related to a common landmark depth each year. Surface sediment samples were collected during plant surveys in 2016. Organic matter content was analyzed after drying to a constant weight and ashing samples at 550°C.

2.3 Data analysis

All statistical analyses were performed in RStudio (Version 3.6.3, 2020). Because the data could not be normalized, a non-parametric Kruskal-Wallis test equivalent to an ANOVA with Dunn's post hoc tests was used to find any differences among groups (Dunn, 1964). We used simple linear regression in RStudio (Version 3.6.3) to determine relationships between the selected variables. The relative abundance of all taxa with more than one individual in two different periods: before (2005–2014) and after (2016–2017) restored water availability was calculated to determine if the dominance of wetland vegetation has changed. The Simpson index, a measure of species richness and evenness, was calculated each year. Chi-square analysis was used to establish any association between the type of soil distribution and the vegetation cluster.

Non-metric Multi-dimensional Scaling (NMDS) was used to ordinate a 2D representation of plant community composition. The analysis was done using the R-package vegan (Oksanen et al., 2019), and the results were plotted in ggplot2 (Wickham, 2016). The distance between the NMDS points of each quadrat over the years was calculated to quantify the amount of change and analyze the influence of water depth on the average change using a linear regression model. The non-parametric test Analysis of Similarities (ANOSIM) was used to analyze differences in multivariate space among ordered groups (periods and sites) (Clarke, 1993).

Permutational multivariate analysis of variance (PERMANOVA) was used to determine significant differences among the continually flooded sites and ponds' edges. We used the adonis function from the vegan package in RStudio (Version 3.6.3, 2020) (Oksanen et al., 2019). The adonis.pair function from the EcolUtils package (Salazar, 2018) was used to determine pairwise differences in community composition.

To identify communities of plant species from 2016 and 2017, a Ward cluster analysis in RStudio (Version 3.6.3) was used. The relative cover (RC) of 21 dominant plant species, those species that were observed more than once in 2016 and 2017, were used to run the cluster analysis.

2.4 Map creation

ArcMap (version 10.5.1) was used to digitize the perimeters of ponds 1 and 2 based on photographs downloaded from Google Earth Pro from a date that showed maximum pond extent (Imagery date: 01-26-2021). These photos were uploaded to ArcMap, georeferenced, digitized, and projected in WGS_1984_UTM_Zone_13N. Maps of relative cover and bathymetry were created using the interpolation tool from ArcToolbox with inverse distance weighting (IDW) or kriging.

3. Results

3.1. Changes in water depth and wetland plants

Water was not present in the ponds during summers prior to 2016. There was a significant increase in water depth in Pond 1 in 2016 (Fig. 2a), with a slight decline in 2017. In Pond 2, there was a delayed response in that we only saw a significant increase in water depth in 2017 ($p = 0.001$) (Fig. 2d). Water depth in 2017 was generally shallow, with an average of 7 cm and 9 cm in ponds 1 and 2, respectively. However, each pond had deeper areas where water tended to remain longer after water deliveries stopped.

Relative frequency and relative cover of wetland plants tended to increase following the change in water delivery; however, the timing of the response differed among the ponds. In pond 1, there was a significant increase in wetland plant frequency in 2016 as compared to 2005 and 2014 ($p = 0.001$) (Fig. 2b **& c**); however, by 2017, both cover and frequency were greater than in all other years. The relative frequency of wetland plants rose to 44% by 2017, and the relative cover to 42%. In Pond 2, the relative

frequency of wetland plants increased to 55% by 2017 and the relative cover to 49%; both were significantly higher in 2017 compared to all other years (p = 0.001). The modest presence of wetland plants in 2005 in pond 2 was likely remnant populations remaining after summer water delivery in 2002.

The relative cover and frequency of wetland plants were positively and significantly related to water depth in almost all cases examined (Table 1). In Pond 1, the relative frequency (r^2 = 0.74–0.76) and cover (r^2 = 0.30–0.74) of wetland plants was strongly related to water depth in both years; however, in Pond 2, it was not until the second year following water delivery in summer that we saw a similar relationship between depth and both wetland plant cover (r^2 = 0.27) and frequency (r^2 = 0.35). The relationship between depth and plant communities was always weaker in Pond 2.

Table 1
Summary of statistics for regressions relating water depth with relative cover and relative frequency of wetland plants in Pond 1 and Pond 2 over 2 years (2016, 2017)

Year	Measure	Pond 1	Pond 2
2016	Relative frequency	r2 = 0.74, p = 0.0001	r2 = 0.24, p = 0.04
2017	Relative frequency	r2 = 0.76, p = 0.0001	r2 = 0.35, p = 0.01
2016	Relative cover	r2 = 0.30, p = 0.033	r2 = 0.03, p = 0.46
2017	Relative cover	r2 = 0.74, p = 0.0001	r2 = 0.27, p = 0.04

Differences in water depth varied spatially in the ponds, and when interpolated across space, they showed slight differences from the averages reported above. The bathymetric interpolation results (Fig. 3) showed an average depth of 0.09 m in Pond 1, as opposed to 0.24m in Pond 2. In Pond 1, the northern section included an expanse of relatively shallow water (0–0.1 m), with the deepest area in the west central area of the pond (up to 0.33 m). Pond 2 included fewer shallows and was between 0.20 and 0.46 m deep across most of the cells (west, north, and south of the pond), with shallower areas in the east (0.1–0.15 m).

Similar to Fig. 2, the interpolation maps showed a gradual change in the relative cover of wetland plants (OBL and FACW) (Fig. 4). In 2016, wetland plants covered the western part of Pond 1, with smaller patches in the east. Pond 2 has only two small patches of wetland plants in the middle and north. Wetland plant cover increased in 2017. In Pond 1, the model showed how wetland plants increased in cover in the western and eastern regions, while more upland plants dominated the middle of the pond and some edges. In Pond 2, the coverage of wetland plants increased, almost completely covering the entire pond, leaving only the edges of the eastern part uncovered.

3.2 Plant community changes

The dominant taxa changed from facultative upland plants prior to 2015 to wetland plants after 2016. From 2005 to 2014, the species with the highest relative abundance were *Atriplex rosea* (22%), *Salsola tragus* (21%), and *Chenopodium album* (12%), which are facultative upland species. Only one of the ten

dominant species was classified as a wetland plant (*Rumex maritumus*, FACW), which was relatively uncommon (5%). For the period of 2016–2017, the dominant species were *Lemna* (15%), *Polygonum lapathifolium* (13%), *and Echinochloa crus-galli* (9%), which are obligate or facultative wetland species. Half of the dominant taxa during this period were wetland plants. Only two upland species (*C. album*, *M. canescens*) from 2005–2014 remained in the top 10 taxa over time. In addition to changes in composition, the Simpson index showed more diversity in the 2016–2017 period (D = 13.45) than in the period of 2005–2014 (D = 7.7). An increase in species richness is also observed from 23 to 28 species.

The NMDS ordination shows all sampled plots and periods in a two-dimensional solution, which had a final stress of 0.06, indicating a good fit (Fig. 5). Pond 1 and Pond 2 appeared to overlap in community composition during all periods, with a slight tendency for Pond 1 to be lower on NMDS axis 2. Quadrats sampled in 2005, 2009, and 2014 ("pre") overlap on the left side of the plot. After water is delivered in the summer, starting in 2016, we see quadrats from 2016 ("post16") spread across the plot, while 2017 quadrats ("post17") are located largely on the right side of the graph. This plot indicates a gradual transition in vegetation community composition from the limited water years on the left to the more regular water availability on the right. This transition or change in plant community structure over periods is confirmed by the ANOSIM test (p = 0.001, R = 0.3101).

Table 2. Comparison of top 10 dominant plant species in each study period in order of relative abundance (acronym used in NMDS graph)

2005-2014	2016-2017
Atriplex rosea (ATRROS; 22%)	*Lemna* spp (LEMNA; 15%)*
Salsola tragus (SALIBE; 21%)	*Polygonum lapathifolium* (POLLAP; 13%)*
Chenopodium album (CHEALB; 12%)	*Echinochloa crus-galli* (ECHCRU; 9%)*
Hymenoxys odorata (HYMODO; 9%)	*Typha domingensis* (TYPDOM; 9%)*
Heliotropium curassavicum (HELCUR; 7%)	*Machaeranthera canescens* (MACCAN, 6%)
Malva neglecta (MALNEG; 5%)	*Leptochloa fusca* ssp. *fascicularis* (LEPFF; 5%)
Rumex maritimus (RUMMAR; 5%)*	*Atriplex canescens* (ATRCAN; 5%)
Chenopodium desiccatum (CHEDES; 5%)	*Leptochloa fusca kunth* (LEPFK; 5%)*
Machaeranthera canescens (MACCAN; 4%)	*Lepidium alyssoides* (LEPALL; 4%)
Hoffmannseggia glauca (HOFGLA; 3%)	*Chenopodium album* (CHEALB; 3%)

* Indicate OBL or FACW species

The average distance between NMDS scores for each quadrat between pre-water and post-water years was calculated to visualize the major changes in the sampled quadrats. A regression model showed that the water depth explained 53% (p = 0.001) of the average distance or change between pre-water and post-water NMDS points (Fig. 6). The biggest community change occurred in the deepest water.

Table 3. Pairwise differences in community compositions between pond and pond edges across years (pairwise adonis)

	Pond 1		Pond 2	
Year	**R^2**	**P value**	**R^2**	**P value**
2005	0.11	0.051	0.07	0.016*
2009	0.11	0.759	0.53	0.202
2014	0.11	0.483	0.08	0.243
2016	0.22	0.017*	0.05	0.576
2017	0.14	0.039*	0.21	0.016*

* Indicate significant differences

Adonis analysis indicated a significant difference in the composition of the plant community among the continually flooded quadrats compared to those located on the ponds' edges (Table 3). In Pond 1, we see a transition from no difference among the edges and areas that flood during the summer (adonis; $p > 0.05$) to substantial differences between the two areas in 2016 and 2017. In pond 2, differences between the ponds and edges are most evident in the 2nd year of restoration (2017). Finally, we also found a significant difference between edges and the summer flooded areas in 2005 in pond 2 (adonis; $p < 0.05$), with a difference that approached significance in pond 1 ($p = 0.051$).

Focusing solely on summer water delivery years (2016 and 2017), we analyzed the distribution pattern of 21 dominant plant species using a cluster analysis. The analysis showed six different groups (Table 4). Table 4 displays the most representative species of each group or cluster based on more than 3% averages of the relative cover of each species in each cluster.

Table 4
2016 and 2017 Grouped plant species into 6 clusters

Groups	Code	Species	Wetland plant	Cover $\bar{X}$	Cover σ
1	LEMNA	*Lemna*	OBL	0.991	0.016
2	LEPFF	*Leptochloa fusca ssp. Fascicularis*	FACW	0.377	0.418
2	POLLAP	*Polygonum lapathifolium*	OBL	0.359	0.389
3	ATRARG	*Atriplex argentea*	FAC	0.894	0.196
4	SALIBE	*Salsola tragus*	FACU	0.897	0.115
5	ECHCRU	*Echinochloa crus-galli*	FACW	0.887	0.132
6	CHEALB	*Chenopodium album*	FACU	0.131	0.282
6	HLLCIL	*Helianthus ciliaris*	FAC	0.096	0.260
6	LEPALL	*Lepidium alyssoides*	N/I	0.113	0.272
6	LEPPM	*Leptochloa panicea mucronata*	FACW	0.054	0.170
6	MACCAN	*Machaeranthera canescens*	UPL	0.103	0.277
6	SUESUF	*Suaeda suffrutescens var. suffrutescens*	N/I	0.063	0.242
6	TYPDOM	*Typha domingensis*	OBL	0.076	0.245
6	CHERUB	*Chenopodium rubrum*	FACW	0.034	0.181

[a]N/I Means pending to classify

Cluster 1 was dominated by the free-floating obligate (OBL) *Lemna*. The OBL plant *Polygonum lapathifolium* and the facultative wetland (FACW) grass *Leptochloa fusca ssp fascicularis* dominated cluster 2. Cluster 3 included the facultative plant *Atriplex argentea*, while Cluster 4 was dominated by the facultative upland (FACU) *Salsola tragus*. Cluster 5 included sites dominated by the FACW grass *Echinochloa crus-galli*. The last group 6 showed a combination of a variety of plant types, including OBL (*Typha domingensis*), FACW (*Leptochloa panicea mucronate, Chenopodium rubrum*), FAC (*Helianthus ciliaris*), FACU (*Chenopodium album*), UPL (*Machaeranthera canescens*), and two plants without indicator status in the National Wetland Plant List (*Lepidium alyssoides*, *Suaeda* suffrutescens *var. suffrutescens*).

We calculated the percent occurrence of each cluster within the quadrats located on the pond edges and those found in the flooded pond quadrats. Clusters dominated by OBL or FACW plants (Clusters 1, 2, 5) were found in the flooded quadrats 73–100% of the time, while the FAC (3) and FACU (4) clusters were found on the edges at least 80% of the time. The mixed cluster (6) was equally likely found in flooded or edge quadrats. (Table 5). This analysis also highlights differences among ponds and years. In particular,

clusters 2 (plant *P. lapathifolium, L. fusca*) and 4 (*Salsola tragus*) were more common in Pond 1, with cluster *4* being exclusively located in the quadrats sample there. Cluster 1 (*Lemna*) was found only in pond 2. Cluster 5 (*Echinochloa crus-galli*) only appeared in 2017.

Table 5
Comparison of the percentage of each cluster in the ponds and edge of RB wetlands

Cluster	Wetland Plant	% Edges	% Ponds	Pond 1-2016	Pond 2-2016	Pond 1-2017	Pond 2-2017
1	OBL	0	100	0	0	0	100
2	FACW-OBL	26.7	73.3	26.7	13.3	53.3	6.7
3	FAC	83.3	16.7	16.7	16.7	33.3	33.3
4	FACU	80	20	20	0	80	0
5	FACW	0	100	0	0	50	50
6	Mix	53.3	46.7	30	43.3	13.3	13.3

3.3 Effect of organic matter and types of soils on plant communities in 2016 and 2017

Soil organic matter has no significant relationship with the relative cover and frequency of wetland plants ($p = 0.46$ and 0.91, respectively). Additionally, a comparison among the 8 USDA soil types underlying the Rio Bosque has no significant effects on relative cover and frequency ($p = 0.40$ and $p = 0.45$, respectively; not shown).

4. Discussion

This project documents the development of wetland plant communities following increased water availability during the growing season in two constructed wetlands in the US desert southwest. Similar to other studies in both temperate (Mitsch et al., 2005; Morimoto et al., 2022) and arid (Norman et al., 2014; Bateman et al., 2015) regions, we found that macrophytes colonized and became dominant in the RB wetlands within 2 years of establishing favorable hydrological conditions. Without human intervention in the form of planting, macrophytes likely grew from the germination of buried seeds and seeds dispersed by wind, water, and birds. Desert floodplains can have large, diverse, and persistent soil seed banks (Capon & Brock, 2006; Capon, 2007; Stromberg et al., 2009). Although no seed bank studies have been performed, the floodplain soils of the historic meander of the Rio Grande on which the RB wetland cells were constructed presumably hold a persistent seed bank of riparian species that have emerged under suitable conditions.

Wetland plant communities tended to form in the deepest and more permanently flooded areas of the ponds. Depth influences light, nutrient availability, gas exchange, species germination, and seedling composition, affecting plant distribution and wetland plant diversity (Keddy, 1983; Seabloom, 1998;

Casanova & Brock, 2000). Other studies have similarly found a greater proportion of wetland plants in deeper, flooded areas, transitioning to more facultative plants along edges and upland areas (Haag et al., 2005; Meyer et al., 2010; Galal et al., 2021). Comparable zonation can be seen in both natural and restored wetlands (Seabloom & Van Der Valk, 2003), with obligate plants, in general, responding quickly to increased water availability after restoration (Sonnier et al., 2023). However, in some areas, wetland plant cover was not evident, possibly due to a delay in vegetation response to fluctuating water levels or because some quadrats do not always receive water (Van Der Valk & Welling, 1988). Zonation along a depth or elevation gradient is often attributed to differential germination of the seeds of better-adapted species (Van Der Valk & Welling, 1988; Seabloom, 1998).

Open water regions of the RB wetlands became more dissimilar from the edges over time, with differences occurring more slowly in Pond 2. This is similar to Meyer et al (2010), who found that restoration towards natural conditions occurred more quickly in the deeper, wetter sites than on the site margins. In addition to gradients in moisture and anoxic conditions, the transition from open water to edges in dryland habitats may be linked with a gradient to higher salinity margins (Sanderson et al., 2008). Salinity, in particular, can reduce germination on wetland margins in the absence of flooding (Brock et al., 2005). However, we have no evidence that salinity varies between the middle and edges (Pond 1 2016, $p = 0.1353$, t-test).

While mitigation sites often meet criteria related to the establishment of wetland vegetation, they may be dominated by exotic or weedy species (Matthews & Endress, 2008). Prior to summer water delivery, but with the provision of water to the wetland cells during winter, the ponds had an overabundance of tumbleweed (*Salsola tragus*), which was the second most abundant plant found in the ponds. The exotic invasive tumbleweed commonly invades disturbed habitats of the southwest and has remained a nuisance throughout upland and wetland habitats in the Rio Bosque. However, tumbleweed is generally intolerant of extended saturation periods and soils with well-established native vegetation (Young, 1991) and was quickly eliminated from the open water area of the wetland cells in 2016.

The remaining dominant plants before 2016 were Saltbush (*Atriplex rosea*), White goosefoot (*Chenopodium album)*, Bitterweed (*Hymenoxys odorata*), and Salt heliotrope (*Heliotropium curassavicum*) and were all well adapted to the pre-water conditions. These taxa are FACU annual forbs that grow in disturbed or man-made sites like the RB. They are highly salt-tolerant and, while they prefer well-drained soil, salt heliotrope also grows well in moist soil (Tumlison & Serviss, 2018; Tang et al., 2022; Todorović et al., 2022).

Plant diversity in the wetland ponds nearly doubled after the provision of the summer water source. A diverse macrophyte community may contribute positively to nutrient cycling and soil moisture, increase productivity, and stop the proliferation of invasive species (Means et al., 2017, Zhang et al., 2015). Macrophyte diversity is also related to increased microhabitats for other biotic communities to thrive (Yofukuji et al., 2021). Notably, dominant species may be more important in determining ecosystem properties (Doherty et al., 2011; Zhang et al., 2015; Means et al., 2017). The dominant vegetation in RB

changed almost entirely post-water delivery, with the four most abundant taxa classified as wetland plants. This change was likely directly associated with the hydrology of the ecosystems, much like Meyer et al., 2010.

The wetlands species that most dominated the RB after 2016 were *Lemna spp*., *Polygonum lapathifolium*, *Echinochloa crus-galli*, and *Typha dominguensis.* Several of these taxa are noted for their important habitat benefits and as an important source of food for waterfowl (e.g. Gibson et al., 2002; Owen et al., 2020). However, they are all largely tolerant of disturbance and highly competitive species. For example, duckweed (*Lemna spp*.), a floating plant with a worldwide distribution, tends to dominate in nutrient-enriched and urban wetlands (Lougheed et al., 2007, 2008), where it may outcompete submerged plants and algae for nutrients and light (Scheffer et al., 2003). Notably, *Lemna* was also observed in Pond 1, but at less than 12% cover. Furthermore, it was present in both ponds during recent visits in the summer of 2024. *Polygonum lapathifolium* also occurs in disturbed environments and is tolerant of a range of moisture levels (Sultan et al., 1998). Because of its fast growth rate, it is often used in phytoremediation (Li et al., 2020). *Echinochloa crus-galli is* also resilient to harsh conditions and is often invasive due to its high competitive potential for nutrients, light, and water (Khanh et al., 2007), and *maybe* phytotoxic to other plants through allelopathy (Xuan et al., 2006; Bajwa et al., 2015). Finally, the cattail *Typha domingensis* is characterized by its worldwide distribution and fast growth rate. While cattails offer ecological services such as removing nutrients, water purification, and habitat and food for some species, they are also often invasive because they may colonize quickly and form dense monospecific habitats, reducing native species of plants, fish, and waterbirds (Bansal et al., 2019).

It is worth noting the somewhat elevated number of wetland plants in wetland cell 2 in 2005, where the edge of the pond was also seen to have a different composition than the middle. We attribute this to the summer water availability in 2001 and 2002, during which time multiple obligate and facultative wetland plants also became established in the ponds, including *Polygonum lapathifolium*, *Echinochloa crus-galli*, and *Typha dominguensis* (Watts et al., 2002). A small number of *Polygonum lapathifolium* remained in Pond 2 into 2005, supporting the resilience of this species to a wide range of moisture (Sultan et al., 1998); however, its loss after longer-term summer drought conditions is noted here. While higher water levels promote the spread of aquatic plants, if the water levels decrease for an extended period, edge habitats may be invaded once again by upland and woody plants (Wilcox et al., 2002).

For several reasons, the vegetation community in Pond 2 may have developed differently than in Pond 1. First, water delivery was delayed to Pond 2 in 2016, so the deepest conditions in this pond were not observed until 2017. Second, while well-drained floodplain soils underline both ponds, each pond's predominant soil types differ. A significant proportion of Pond 1 is underlain by sandy loam, while more of Pond 2 is underlain by silty loam (Nachtergaele, 2001). Detailed water level monitoring in both ponds between 2006 and 2008 indicated that Pond 2 drained more slowly once water inputs ceased (unpubl. data), perhaps due to these differences in soil type. Soil texture influences wetland hydrologic patterns, organic matter, and plant nutrient retention and availability, affecting the growth and distribution of vegetation. The finer soils, such as silt and clay, tend to retain better water, organic matter, and nutrients,

benefiting the aquatic plants, while soils with more sand particles have greater permeability (Stolt et al., 2000; Huang et al., 2013; Jackson et al., 2019; Wankmüller et al., 2024). Campbell et al. (2002) reported differences in plant communities and vegetation cover between created and natural wetlands derived, among other things, from the type of soil having less sand and more silt and clay in the natural sites. Hossler & Bouchard (2010) also found lower plant biomass in created wetlands than in natural wetlands due to the lack of capacity to hold carbon in the sandiest soils. Although we are not comparing our two ponds to natural wetlands, they have different soil characteristics that may affect the plant community. In wetlands like RB, where there are fluctuations in water availability, having soils that can retain water for longer and maintain soil moisture would be an advantage for preserving wetland plants.

The history and construction history of the RBWP also likely impacted plant community development throughout the RBWP. While both the Pond 1 and Pond 2 sites were completely cleared, excavated, and re-contoured as part of the construction of the wetland project in 1997, important differences existed. Prior to the construction of the wetland project in 1997, most of the upland areas and the site of the future Pond 1 had been in agricultural production in the recent past. The Pond 1 region was much more open and supported a very different plant community than the future location of Pond 2. For example, a massive salt cedar bosque occupied the site of the future Pond 2. The exotic salt cedar (*Tamarix ramosissima*) has been associated with soil salinization (Ladenburger et al., 2006); however, we saw no lasting impact of the dominance of this plant on the salinity of the soil in Pond 2 (not shown).

During the construction of wetlands, excavation may alter soil texture and, therefore, the percentage of organic matter. The organic matter content is lower in CWs than in natural ones. However, if water conditions are favorable, the organic matter levels and the clay content will increase over time. The decomposition of organic matter decreases in saturated soil, which produces carbon accumulation. Organic matter is an essential soil structure and a significant predictor of the growth and distribution of macrophytes (Stolt et al., 2000; Hossler & Bouchard, 2010) and often the two are positively related (Ogdahl & Steinman (2015); Squires & Lesack (2003). However, we did not find a significant relationship between wetland plant cover and soil organic matter. However, the fact that constructed wetlands generally have less organic matter than natural wetlands may delay the restoration and functionality of constructed wetlands. Organic matter in the RB ponds averaged 4.1 ± 2.6% in Pond 1 and 6.9 ± 2.5% in Pond 2. This is comparable to other studies that show 5–6% organic matter in created wetlands, as opposed to often double that in created wetlands (Campbell et al. (2002), Bishel-Machung et al. (1996))

5. Conclusion

The importance of conserving, restoring, and replacing aquatic ecosystems has been of great interest due to the significant loss of these ecosystems that we have faced in recent years due to different human activities. RB is an example of extended efforts by a large community of people to recreate wetlands where they once existed and try to reestablish some of their ecosystem services. This study documented the long-term (more than a decade) development of water resources and wetland plant communities in the two cells of the RB. We observed that once water was established, there was a rapid

shift from upland plants to more aquatic plants and increased plant diversity. RB is in the process of restoration, but this process requires time, and emphasis must be placed on water availability. Although the availability of water in the park has increased, the continued delivery of water continues to be a challenge, preventing the park from being fully restored. Furthermore, this valuable habitat is under threat from highway expansion and construction in the region. Although the management and permanent availability of water in the RB may be a problem, we recommend finding and discussing new strategies for managing and conserving water in the RB to continue recovering and restoring this ecosystem.

Declarations

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Data Availability: "The datasets generated during and/or analyzed during the current study are available in the Garcia, E. D., V. Lougheed (2025). The Long-term Development of Wetland Plant Communities with Water Deliveries in a Created Wetland in the Desert Southwest, HydroShare, http://www.hydroshare.org/resource/83de61ef19c34547b55cf0a463b4c6fe

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Figures

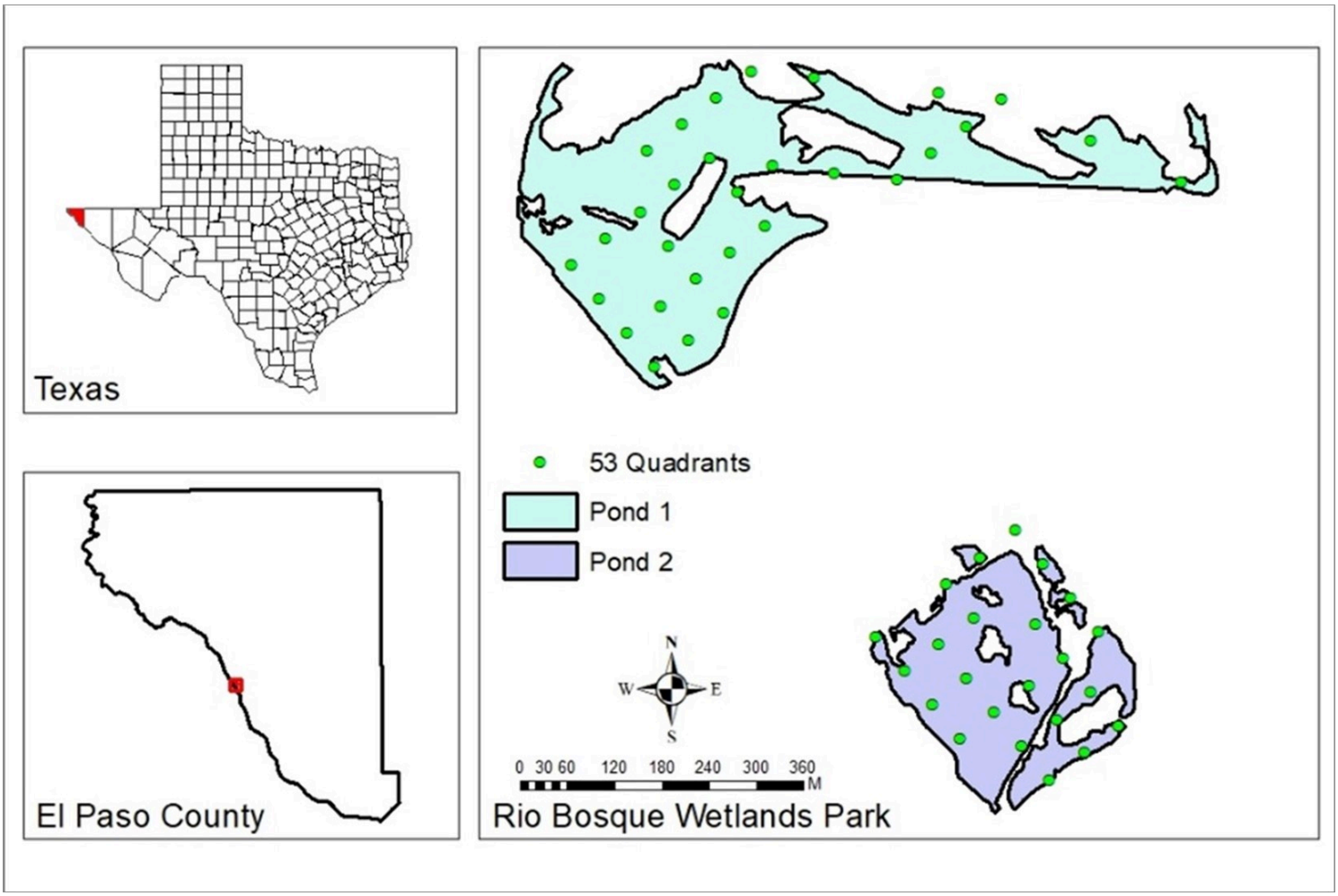

Figure 1

Location of the Rio Bosque Wetland Park in El Paso County, TX, and location of sampled quadrats within wetland ponds 1 and 2 in the park

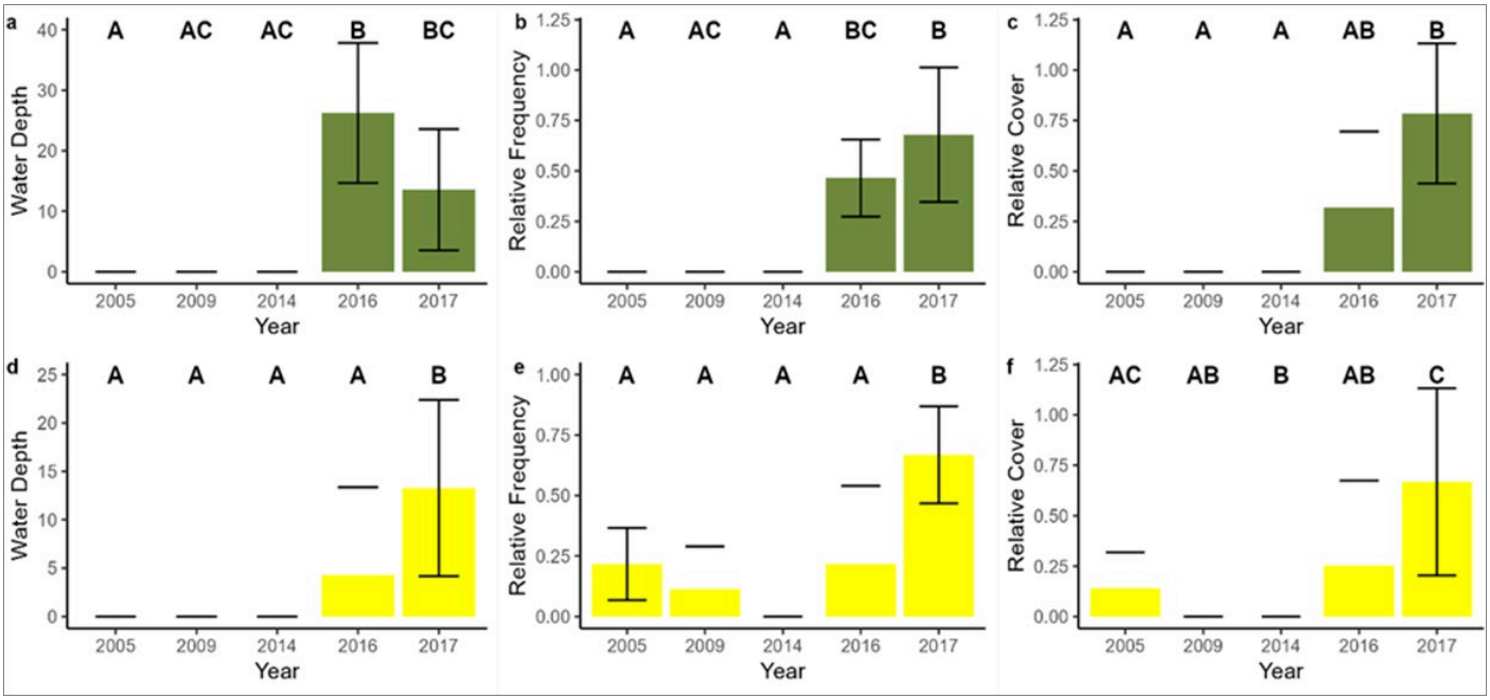

Figure 2

The bar plots a, b, and c show how water depth, relative frequency, and relative cover of wetland plants (OBL & FACW) change through time in Pond 1, while d, e, and f show the changes in Pond 2. The years with the same letters (A, B, and C) indicate no statistical difference; however, the years with different letters differ significantly

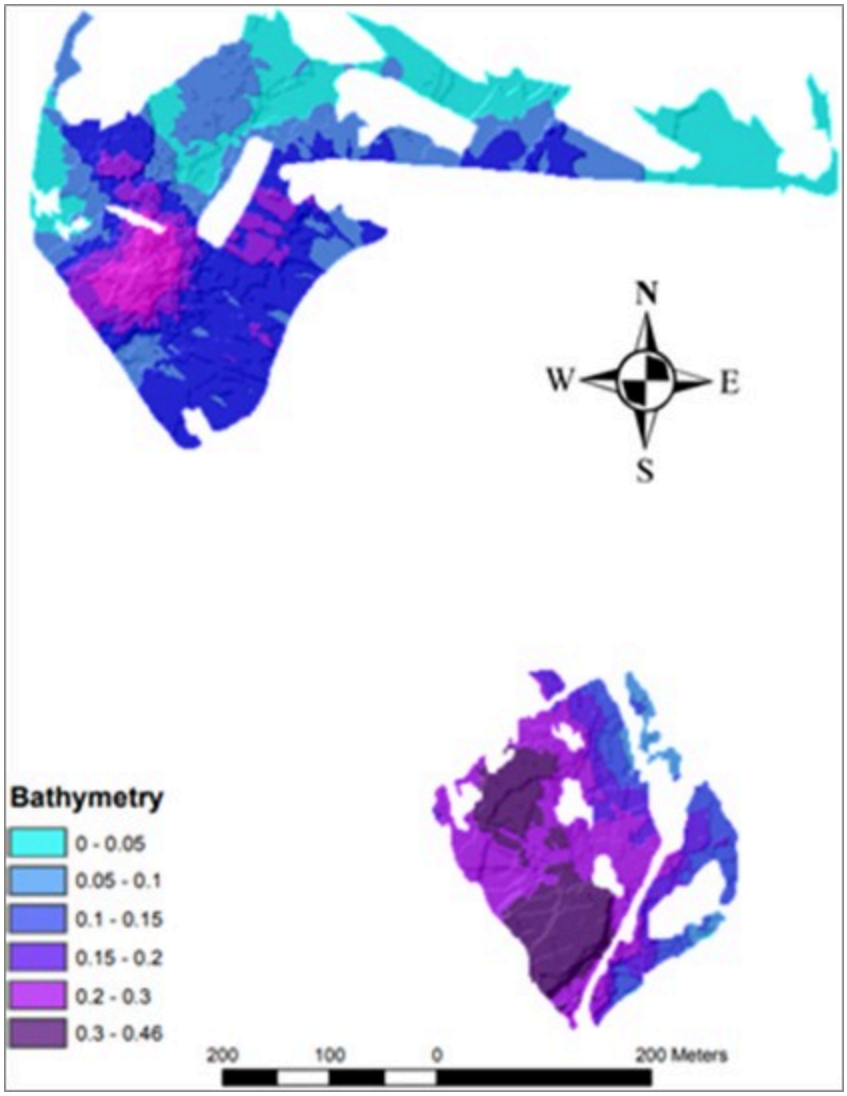

Figure 3

Bathymetry of the wetland ponds at the Rio Bosque.

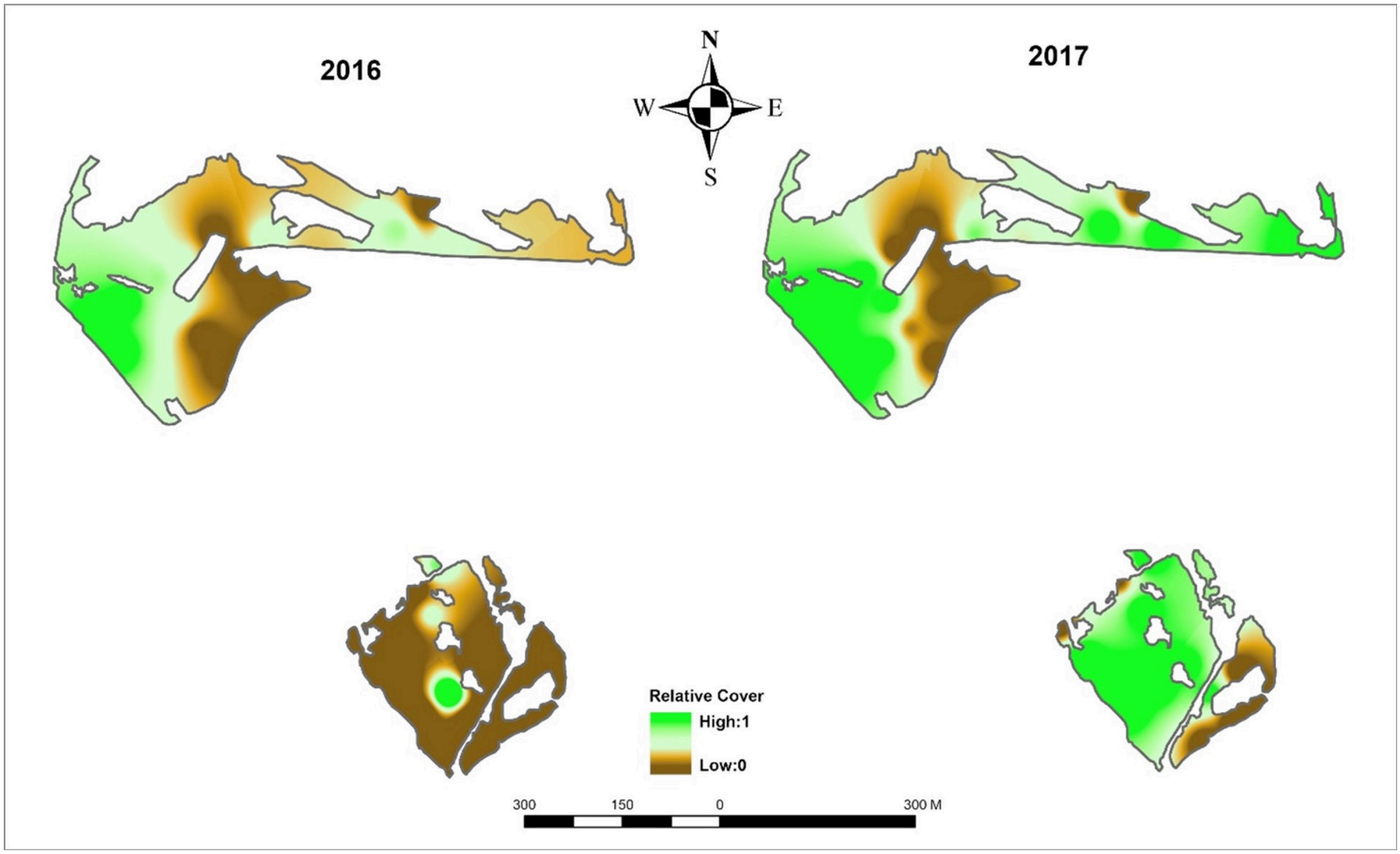

Figure 4

Distribution model of the relative cover of wetland plants (OBL and FACW) within the ponds at Rio Bosque in 2016 and 2017

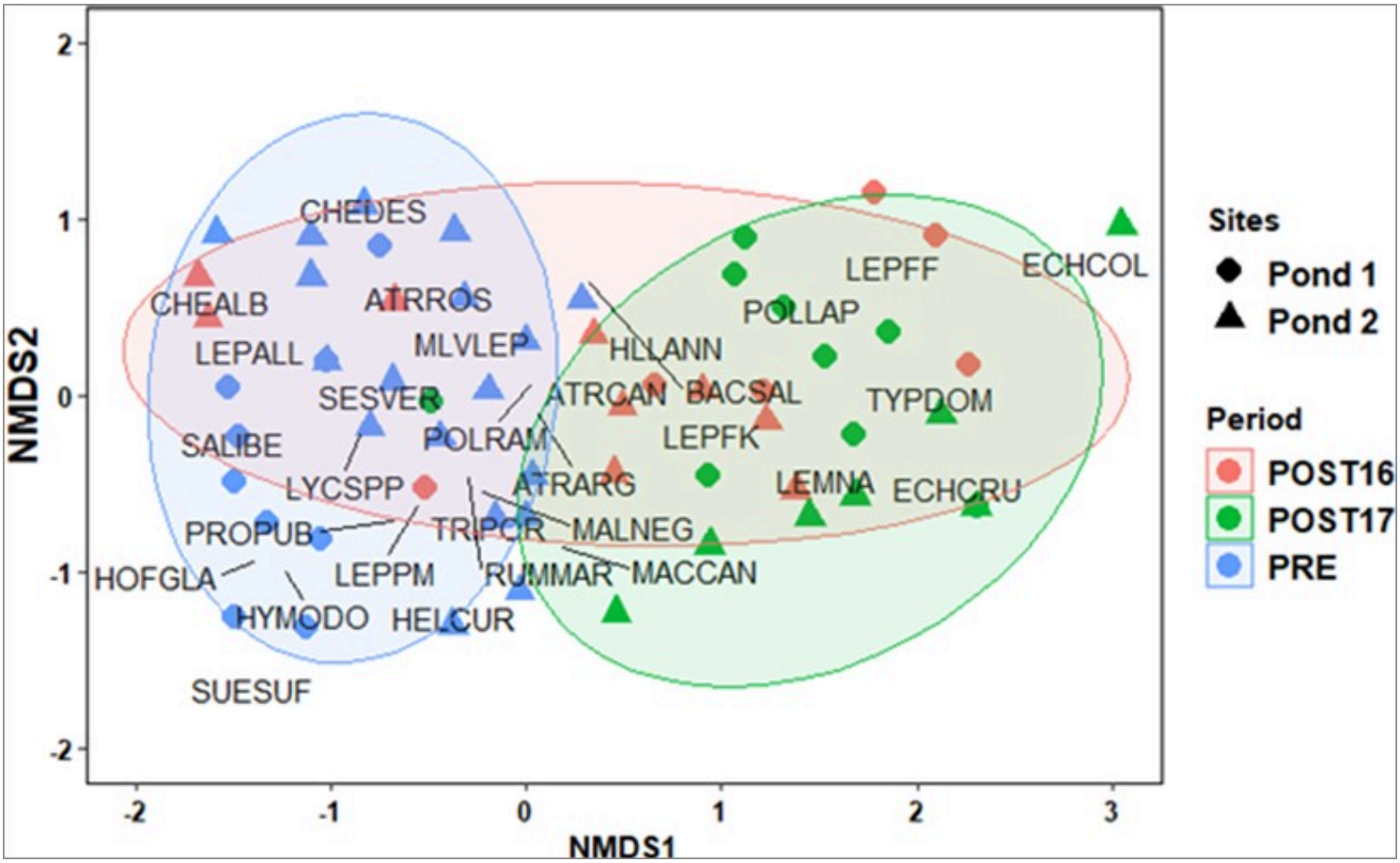

Figure 5

Non-metric multidimensional scaling (NMDS) of plant communities in the Rio Bosque Wetland Park over more than a decade (2005-2017) comparing data collected prior to regular water delivery during the growing season (Pre) and the 2 years immediately following summer water delivery (Post16 and Post17). Species codes are defined in **Table 2**.

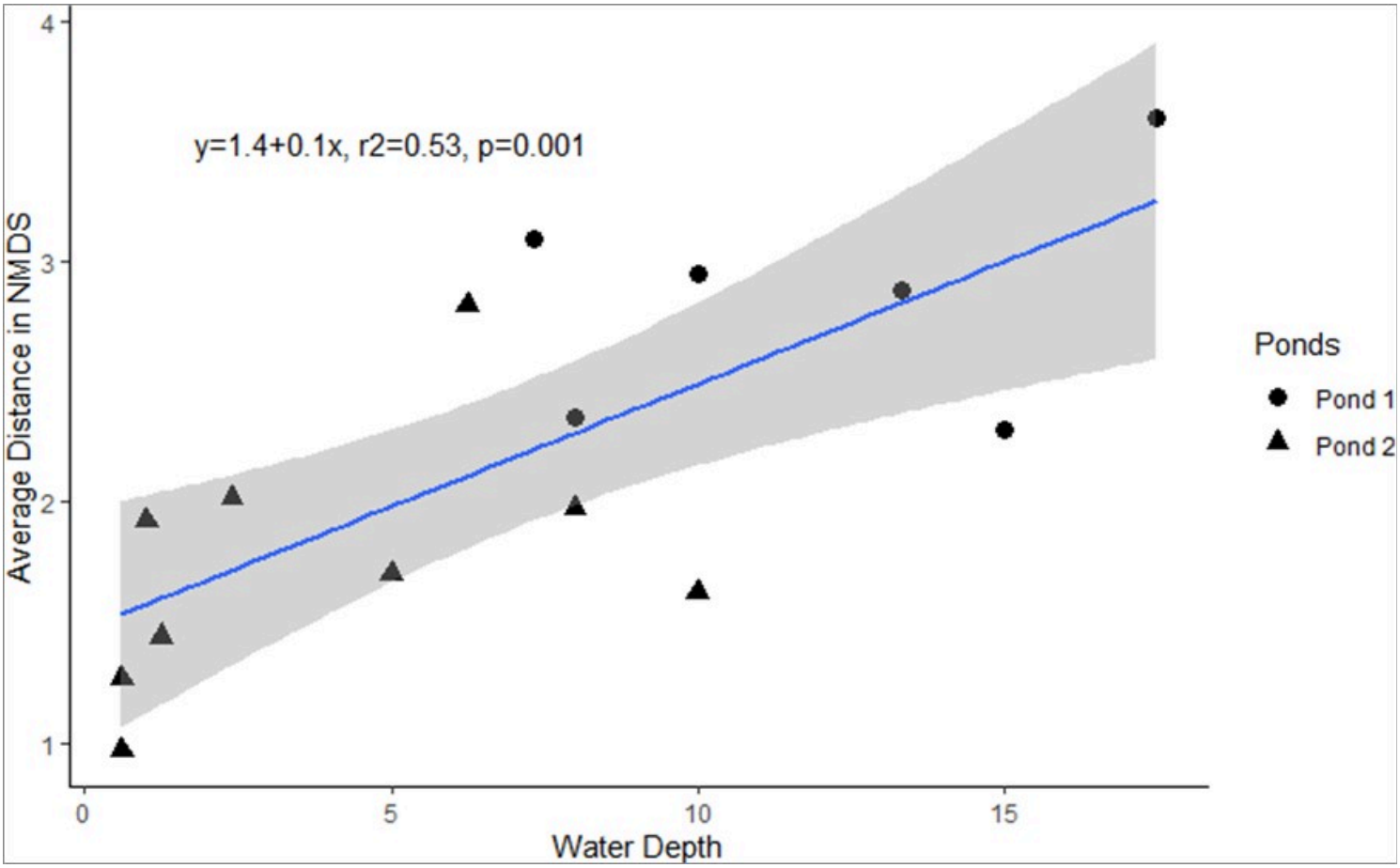

Figure 6

The regression linear model shows the influence of the 2016-2017 average water depth on the average distance in NMDS space between pre-water and post-water periods.