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Tools To Advance Environmental Monitoring, Wetland Restoration And Education In The Desert Southwest

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TOOLS TO ADVANCE ENVIRONMENTAL MONITORING, WETLAND RESTORATION AND EDUCATION IN THE DESERT SOUTHWEST

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ABSTRACT

The relatively rare freshwater ecosystems in the southwestern United States serve as biodiversity hotspots, yet they are among the most threatened systems in the world due to human impacts and climate change. Despite their importance to this arid landscape, the aquatic communities of desert wetlands remain relatively understudied. To restore and create new wetland habitats, effluent is becoming a more commonly used water source for these habitats. However, the effects of byproducts within the treated wastewater on these unique systems have not been well studied. In this study, we aim to better understand the factors that drive water quality and macroinvertebrate community composition of wetlands of the US desert Southwest. In addition, we focused on a local, restored wetland (Rio Bosque Wetlands), to better understand how water quality and community assemblages change with the increased use of treated effluent as a water source. Finally, in an effort increase awareness of habitat conservation and restoration we created an ecology-based virtual CURE (vCURE) that was implemented to non-science majors attending El Paso Community College.

Water quality and macroinvertebrate data were collected over three years from 14 different wetland and riparian sites spanning across West Texas, New Mexico, and Arizona. Results indicated that salinity related variables such as chloride, sulfate, and conductivity were the greatest drivers of environmental variance. Subsequently, nutrients were shown to have the greatest impact on macroinvertebrate communities with wetlands receiving treated wastewater showing a more uneven distribution of functional feeding groups (sites dominated by filter feeders) and lower Simpson Index scores. Increased salinity levels were also shown to correlate with lower Simpson Index scores thus, a decline in macroinvertebrate diversity and evenness.

To track the restoration of the Rio Bosque Wetlands, data collected in 2014, before a change in water regime, and data collected after (2016-2019) was used to determine differences in water quality and macroinvertebrate communities. The increased water inputs during the growing season in 2016-2019, established more permanent bodies of water which affected macroinvertebrate communities by allowing taxa with limited dispersal abilities time to build larger populations. Differences in assemblages within the park were also heavily influenced by the increased nutrients associated with effluent water. Overall, Rio Bosque Wetlands is displaying succession patterns similar to those of other, more established desert wetlands flooded with treated effluent water, with a growing community of filter feeders (Chapter 1). As a result, it is suggested that managers of these valuable created aquatic habitats try to find less nutrient-rich water sources, such as groundwater, to enhance the water quality in their sites. With reduced nutrient levels, we would expect to see an increased in sensitive taxa, predators, and collector-gatherers, among others. Though the macroinvertebrate community in created or restored sites, may not resemble those of a natural site due to the use of treated effluent water, these systems provide much needed habitat for aquatic flora and fauna within the desert landscape.

While the scientific community largely recognizes the importance the role of ecology plays in habitat preservation and combating the effects of climate change, much of the general population do not. To increase public understanding of preservation efforts for desert wetlands and other at-risk ecosystems, science literacy skills must increase within the community. Course-based Undergraduate Research Experiences (CUREs) have been used to improve science literacy and attitudes for large groups of students. In 2020 the COVID-19 pandemic and stay-at-home orders forced many college courses to switch to virtual learning which led me to create an ecology-based virtual CURE (vCURE). With the Undergraduate Research Student Self-Assessment (URSSA)

and the Test of Science Literacy Skills (TOSLS), we investigated the effects of participation in a vCURE on the science literacy skills, attitudes, and perceived gains of non-science majors and El Paso Community College. Our results showed that students were able to improve their overall TOSLS scores and increase their confidence levels in several general science and research related activities. In open ended responses, students felt that the course helped them improve skills that would be beneficial to them in the future, including communication, collaboration, and critical thinking. This shows that non-science majors can still benefit from CUREs though they do not intend to pursue a science related career. This CURE model can be modified to enhance students' knowledge of habitat conservation by creating an in-person wetland-themed CURE to further track the restoration of the Rio Bosque Wetlands.

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INTRODUCTION

Around the world, freshwater ecosystems are under continuous threat due to anthropogenic pressures and altered weather patterns due to climate change (Robert T. Brooks 2009; Mekonnen and Hoekstra 2020; Woodward, Perkins, and Brown 2010a) (Robert T. Brooks 2009; Mekonnen and Hoekstra 2020; Woodward, Perkins, and Brown 2010b). These changes have led to a decline in aquatic biodiversity that exceeding that of terrestrial systems due to the increasing demand for fresh water (Vörösmarty et al. 2010). Over time, changes to hydrological regimes will likely impact the flora and fauna of these systems due to fluctuating timing and magnitude of wetland inundation (Pitchford et al. 2012). With these challenges expected to become more severe, this becomes extremely problematic for wetland ecosystems, which rely on water availability to maintain basic wetland functions (Strayer and Dudgeon 2010).

Wetlands of the Desert Southwest

The southwest United States, though normally arid, has seen a drastic increase in drought conditions over the past 10 years due to changes in precipitation patterns (McKinnon, Poppick, and Simpson 2021; Overpeck and Udall 2020). In addition, rising temperatures pose a threat to these unique habitats by increasing evapotranspiration rates and the potential for prolonged megadrought conditions (Overpeck and Udall 2020; Strzepek et al. 2010; USDA Forest Service 2010).

The freshwater ecosystems in the arid southwestern United States serve as biodiversity hotspots, supporting a disproportionately high share of landscape diversity (Dinerstein et al. 2001; Stanislawczyk et al. 2018). These systems function as refugia for aquatic taxa such as macroinvertebrates (Griffis-kyle et al. 2019; Moorhead, Hall, and Willig 1998), fishes (Zengel and Glenn 1996), and macrophytes (Karpiscak et al. 2001). They also serve as nesting habitat for migratory birds (García et al. 2017). Recent studies have begun to highlight the novel communities within these habitats, emphasizing the presence of endemic and cryptic taxa (Griffis-kyle et al. 2019; Seidel, Lang, and Berg 2009; Stanislawczyk et al. 2018). Recently, there has been a push to better understand these assemblages and what drive this community composition (Bogan et al. 2014; Colombetti et al. 2020; Esposito 2012; Sei, Lang, and Berg 2009; Stanislawczyk et al. 2018). For example, Stanislawczyk et al. 2018, found that geographic distance between desert springs was a better predictor of macroinvertebrate community composition than abiotic parameters, likely due to isolation and limited dispersal between sites. These results are consistent with similar studies in desert springs in the Mojave Desert (Sada, Fleishman, and Murphy 2005).

While the threat to these arid wetlands has been understood since the 1980's (Hendrickson and Minckley 1985), they remain among the most understudied systems in the world (Nieto et al. 2017). For example, there are 15,000 springs in the southwest United States that have been identified and are being monitored, with nearly none having recorded historical data (USGS, 2018).

The biodiversity within these freshwater systems is especially vulnerable to climate change due to their relative isolation and fragmentation, leaving species with limited opportunity to disperse (Davis et al. 2013; Erwin 2009). Along with climate change, urban wetlands of the southwest face human related disturbances such as agriculture run off, vegetation removal, changes to water levels and drainage patterns; all of which contribute to their vulnerability. Recently, there has been a growing interest in restoring or creating freshwater habitats with the use of effluent water from wastewater treatment plants (Hamdhani, Eppehimer, and Bogan 2020; Hsu et al. 2011; O'Geen et al. 2010; Rodriguez and Lougheed 2010). Though this method usually provides a constant water source for these systems, the long-term effects of exposure to the high nutrient levels in the effluent water remains to be seen (B. W. Brooks, Riley, and Taylor 2006; Hamdhani, Eppehimer, and Bogan 2020).

Assessing created or restored wetlands

Along with this push for restoration comes the need for ways to track the restoration and the health of these wetlands. Many methods have been developed and used to track the restoration of created or restored wetlands. There have are several assessments using wetland plants including monitoring the abundance of native species (Adamus and Brandt 1990; Taddeo and Dronova 2018) in addition to plant biomass and tolerance to disturbance (Lopez and Fennessy 2002; Zhao et al. 2016). Wetland fauna have also been used as indicators of restoration. For example, the abundance of small fish, crustaceans and wading birds have been used as measures of healthy food web relationships in restored areas of the Everglades (Trexler and Goss 2009). Diversity indices are also commonly used as indicators of restoration, with most studies only focusing on the assemblages of one group of organisms: typically plants or vertebrates for conservation projects (Ruiz-Jaen and Aide 2005; Sebastián-González and Green 2016). Finally, ecological processes, such as nutrient cycling, are used less often than vegetation or diversity indices because they are usually slower to recover from disturbance and require multiple measurements over time (Ruiz-Jaen and Aide 2005).

Some studies have attempted to identify the macroinvertebrate metrics that would be best used for tracking wetland restoration but they have proven to be inconclusive at indicating success (Marchetti, Garr, and Smith 2010; Meyer and Whiles 2008; Ruhí et al. 2012). Others have shown that macroinvertebrate diversity (Simpson Diversity Index and Invertebrate Community Index) of created wetlands was significantly lower when compared to natural wetlands (Acharyya and Mitsch 2000; Spieles and Mitsch 2000; Swartz et al. 2019) and that dissolved oxygen and specific conductivity were the best predictors for species diversity (Spieles and Mitsch 2000). In another study focusing on wastewater ponds, drivers of community composition were identified as pH, vegetation structure, and pollution levels (Becerra et al. 2009). When comparing created, impacted and reference wetlands, it was determined that the amount of vegetation had the greatest influence on macroinvertebrate taxonomic richness (Swartz et al. 2019). While these few studies give some insight to what may drive community composition, none of them were conducted in desert wetlands, where water characteristics are very different, especially in salinity and hydroperiod, and where different "core" assemblages of macroinvertebrates may be found (Ruhí, Batzer, and Ruhí 2013). Therefore, it cannot be assumed that they will respond the same way to restoration efforts.

Aquatic macroinvertebrates as bioindicators

While macroinvertebrates are not often used as indicators of wetland restoration, they have, for decades, been used as a means of assessing water quality within freshwater systems due to the fact that they are in constant contact with water and sediment where many pollutants accumulate (Mandaville 2002). In past studies, macroinvertebrates communities have been used as biological indicators of heavy metals (Ordonez et al. 2011), nutrient enrichment (Cortelezzi et al. 2015; Søndergaard and Jeppesen 2007), land use (Anderson and Vondracek 1999; Sada, Fleishman, and Murphy 2005) vegetation cover (Death and Collier 2010; Lawrence et al. 2016), salinity (Dunlop et al. 2008; Sowa, Krodkiewska, and Halabowski 2020), and overall biomonitoring of freshwater habitats (Cairns and Pratt 1993; Johnson, Wiederholm, and Rosenberg 1993; López-López and Sedeño-Díaz 2015; Lougheed et al. 2007; Serrano Balderas et al. 2016; R. C. Sharma and Rawat 2009).

Many biological indices have been developed as a measure of organic and nutrient pollution within freshwater systems based on the presence or absence of tolerance and/or sensitive species. For example, the Hilsenhoff Biotic Index assigns pollution tolerance levels to macroinvertebrate families. The degree of organic pollution can then be determined based on the average tolerance level of the macroinvertebrates collected from that site (Hilsenhoff 1987). Another reliable biotic index used to assess water quality is the EPT Index, which meassures the richness of the most sensitive macroinvertebrate groups: Ephemeroptera (mayflies), Plecoptera (stoneflies) and Tricoptera (caddisflies) (Lenat 2016). The absence or presence of these orders can then be used to evaluate the quality of the water (Lenat and Science 1988). Other available indices that may be used include the Simpson Diversity Index as well as the Invertebrate Community Index (Spieles and Mitsch 2000). Several studies have also used macroinvertebrate species composition within multivariate statistical analysis to help understand patterns in communities composition along environmental gradients (Gleason and Rooney 2018; Lougheed et al. 2008; Moreno, Angeler, and De las Heras 2010; Zimmer, Hanson, and Butler 2011).

While many of these indices and metrics have proven to be reliable in wetlands and streams in temperate regions, it is unknown whether macroinverbreate assemblages in desert wetlands respond to distubance in the same way. This comes as a recent study has higlighted disparities of using the same indices across differing systems (Mazor et al. 2016).

Course-based Undergraduate Research Experiences (CURE)

While the scientific community largely recognizes the importance that ecology plays in habitat preservation and combating the effects of climate change, many outside of this group do not. To intensify preservation efforts for desert wetlands and other at-risk ecosystems, we must increase science literacy skills within the community. One way to better increase science literacy is to have students conduct meaningful research at the undergraduate level (Sadler et al. 2010). While having all students conduct research at some point in their academic careers is ideal, it is not often feasible due to limited undergraduate research positions available (Desai et al. 2008). These opportunities are often very competitive and the vast majority of students at four year universities will not be able to obtain a research position; this number is even less at the community college level (Auchincloss et al. 2014; Bangera and Brownell 2014; Kloser et al. 2013; Weaver, Russell, and Wink 2008).

One method of overcoming the lack of research positions available to students is with a Course-based Undergraduate Research Experience (CURE). A CURE typically occurs in the lab portion of science course where the whole class is involved in addressing a research topic (Auchincloss et al. 2014). Over the course of the semester, students will design and implement their own research projects with the result being a poster they can present to the class, or even at conferences. Since CUREs are integrated into the course, students who may typically not have the opportunity to conduct research through internships will gain the relevant experience. Differing from inquiry projects, CUREs increase the value of science communication and literacy as students will be required to read and cite research articles along with having to present a research poster as the final assessment (Dolan 2016). CUREs have been shown to be useful tools for improving science literacy and attitudes for large groups of students (Auchincloss et al. 2014; Dolan 2016) though students a hands-on and immersive experiences.

CUREs serve as a way to give research opportunities to more students, this in turn also helps to break down some of the barriers these students may face, thus increasing inclusivity (Bangera and Brownell 2014). Reasons for this loss include: lack of awareness of existing research opportunities and their benefits, differences in cultural norms, and financial or person barriers

(Bangera and Brownell 2014). This loss of retention is increased in students who attend non-4 year universities at the beginning of their undergraduate careers. With approximately 34% of students nationwide beginning their higher education careers at community colleges, this means a large proportion of students fall through the cracks every year (Community College Research Center, 2017).

Moreover, a high percentage of these students are coming from lower socioeconomic and underrepresented populations. In many community colleges there are little to no opportunities for their students to participate in undergraduate research; many of their students then transfer to 4 year universities with no research experience and no knowledge of how find or apply for research programs (Bangera and Brownell 2014). This becomes problematic as independent research is quickly becoming an unofficial prerequisite for admission to graduate school.

In 2020 the COVID-19 pandemic and stay-at-home orders forced many college courses to switch to virtual learning. CURE courses, which are recognized for their hands on activities student interactions, were now moved to an online setting. This major change, however, challenged educators to develop unique and innovated virtual CURES (vCURE) that still engaged students and allowed for hands on activities in a safe manner (Corson et al. 2021; Majka et al. 2021). Many developed what is now known as "CURE in a box" where students receive all the supplies necessary to conduct laboratory activities at home (Bennett et al. 2021). While this work great with more lab-based microbiology courses, they are not ideal for ecology-themed CUREs. While ecology-themed CUREs our outnumbered by their microbiology counterparts, previous research highlights the benefits of these types of courses (Kloser et al. 2013).

Goals and Objectives

In this study I aimed to fill gaps in knowledge regarding the drivers of macroinvertebrate community composition within desert wetlands of the southwest United States. In addition, we focused on a local, recently restored wetland, to better understand how water quality and community compositions change with the addition effluent water as a water source. Finally, I created an ecology-based virtual CURE (vCURE) that was implemented to non-science majors and El Paso Community College. This study will address the following objectives and underlying questions:

1. Identify drivers of macroinvertebrate community composition in wetlands of the desert Southwest of varying water sources [Chapter 1].
2. Determine how water quality and macroinvertebrate community composition in the Rio Bosque Wetlands have responded to wetland restoration efforts [Chapter 2].
3. Implement and investigate what effects participation in a virtual ecology- themed Course-based Undergraduate Experience had on the science literacy skills, attitudes, and perceived gains on non-science majors at a community college [Chapter 3].

This information can be used to better understand the succession that these unique systems go through during the restoration process. In turn, we sought to highlight key factors that could lead to better management practices of restored or created wetlands. With the vCURE we hope to adapt it for future use to better increase positive attitudes towards science, improve science literacy skills and create greater accessibility of research experiences for non-traditional students.

CHAPTER 1: ARE NUTRIENTS OR SALINITY THE DRIVERS OF MACROINVERTEBRATE COMMUNITY COMPOSITION IN WETLANDS OF THE DESERT SOUTHWEST?

1.1 Introduction and Background

The loss of global biodiversity is occurring at an exceedingly rapid rate due to climate change and overexploitation by humans (Dawson et al. 2011). While terrestrial ecosystems are often in the spotlight, aquatic ecosystems surpass their rate of loss of biodiversity due to declines in water quality, changes in nutrient availability and increasing temperatures (Association of State Wetland Managers 2015; Van De Waal et al. 2010; Xi et al. 2021). Arid region wetlands are especially vulnerable due to altered precipitation patterns related to climate change and declining groundwater flow as a result of overuse (Burkett and Kusler 2000; Taylor et al. 2013; Richey et al. 2015). As biodiversity hotspots, these oases are habitat for many organisms and provide critical habitat connectivity within the desert landscape (Dinerstein et al. 2001; Bogan et al. 2014; Drake et al. 2017). While freshwater habitats are known to support ~10% of all species, including many endangered and endemic species, arid region wetland ecosystems worldwide remain understudied and under-recognized when it comes to wetland ecology and conservation (Hershler and Liu 2010; Minckley et al. 2013; Murphy et al. 2013; Nieto et al. 2017; Stanislawczyk et al. 2018; Strayer and Dudgeon 2010; Walsh et al. 2009). Due to the rapid loss of habitat, there has been a recent push to protect and restore these rare freshwater ecosystems.

In the southwest United States, many wetlands have been restored or created to replace those wetlands that have been lost. Some wetland sites use the delivery of wastewater to mitigate or restore areas that were previously lost or degraded due to river channelization or agricultural use

(O'Geen et al. 2010; Rodriguez and Lougheed 2010). These sites create new habitats for migrating birds and aquatic organisms and well as areas of cultural value such as city parks (Andrade et al. 2018; Hamdhani et al. 2020; Bogan et al. 2020). These habitats are often used to further purify effluent water through the uptake of nutrients (i.e., nitrogen and phosphorus) and contaminants by wetland macrophytes and microalgae before replenishing groundwater sources (Whitton et al. 2016; Matamoros et al. 2017; Zhuang et al. 2019). While studies have shown these wetlands to be effective at reducing excess nutrients and contaminants from wastewater, the initial presence of these byproducts may have lasting effects on freshwater biota (R. T. Brooks 2000). In some non-arid created wetlands, increased nutrients cause shifts in community composition with an increase in pollution-tolerant macroinvertebrate taxa (Pinto et al. 2014). However, due to variables relatively unique to arid regions (i.e., extreme heat, irregular and rare precipitation), it is unknown if macroinvertebrates in arid wastewater wetlands respond the same way as those in non-arid regions.

In freshwater ecosystems, macroinvertebrates have historically been used as indicators of water quality and wetland health (Hilsenhoff 1987; Mandaville 2002). As bioindicators, aquatic macroinvertebrates serve as a low-cost and useful tool for monitoring wetland health and function due to their constant contact with water and sediment (Hilsenhoff 1987; Cairns and Pratt 1993; Bartell 2006; Siddig et al. 2016; McIntosh et al. 2019). By monitoring the abundance, diversity, and reproductive success of these organisms we can determine habitat response to change or disturbance (Foote and Rice Hornung 2005; Siddig et al. 2016; Wu et al. 2017). While these biotic indices are easily applied to non-arid region habitats, it should not be assumed that macroinvertebrates in arid habitats will respond the same way to environmental stressors. Recent studies have even highlighted the possible disparities of using the same biotic indices across differing systems (Mazor et al., 2016; Serrano Balderas et al., 2016).

When examining wetlands in non-arid regions the differences in macroinvertebrate community composition have often been attributed to vegetation community composition (Balcombe et al. 2005; Stewart and Downing 2008; Becerra Jurado et al. 2009; Swartz et al. 2019) and water quality associated with development (Carew et al. 2007; Kobingi et al. 2009; Lougheed et al. 2008). In contrast, other have pointed to hydroperiods and desiccation cycles (Esposito 2012; Gleason and Rooney 2018; Moraes et al. 2014; Pires, Stenert, and Maltchik 2019) or wetland isolation and dispersal limitations (Stanislawczyk et al. 2018) as the driving factor of macroinvertebrate community composition. While both these arid region studies identified differences in nutrient chemistry or salinity among sites, neither identified water chemistry as a predictor of macroinvertebrate community structure, perhaps because of the limited number of sites sampled, or small gradients examined. Salinity, in particular, may be elevated in arid region water bodies due to high evaporation rates and inconsistent water availability (Borrok and Engle 2014; Nielsen et al. 2003) and may increase in importance during dry periods (Jolly, McEwan, and Holland 2008; Lahr 1997). Furthermore, it is largely unknown what gradients of water quality organisms in desert wetlands of the US southwest are exposed to as there have been no broad scale studies to examine these environmental gradients.

The primary objective of this study was to determine how water chemistry varies among wetlands of the US desert southwest, and how this may drive macroinvertebrate community composition within these rare habitats. Specifically, we assess whether metrics of macroinvertebrate diversity, tolerance and functional feeding groups are related to water source (i.e., wastewater sites vs. non-wastewater sites) or salinity. We expect that wastewater effluent and highly saline water sources of many desert wetlands will negatively affect sensitive taxa due to their vulnerability to anthropogenic factors (Ocon & Capítulo, 2004) and lead to homogenization of functional feeding groups as shown in similar studies in non-arid regions (Lougheed et al. 2008).

Table 1.1: Sample sites, location of site, water source, and approximate area for 14 wetlands sampled in the Chihuahuan and Sonoran deserts. Sites 1 – 12 were visited in 2018 and 2019. Sites 13-14 were added in 2019. Only sites located in El Paso, Texas were also sampled in 2020 due to travel restrictions. Code names appear in Fig. 1b. * Indicates ephemeral wetlands.

#	Name	Location	Code Name	Water Source	Area (ha)
1	Tres Rios Wetlands	Phoenix, AZ	TR1, TR2	wastewater	91
2	Sweetwater Wetlands	Tucson, AZ	SW1, SW2	wastewater*	6
3	Las Palomas Marsh	Las Palomas, NM	LP	non-wastewater*	3
4	Rio Grande 1	Las Palomas, NM	RG1	non-wastewater	<1
5	La Mancha Wetlands	Las Cruces, NM	LM	non-wastewater*	<1
6	Rio Grande 2	Las Cruces, NM	RG2	non-wastewater	<1
7	Keystone Wetlands	El Paso, TX	KS	non-wastewater	1
8	Crossroads Pond	El Paso, TX	CR	non-wastewater*	3
9	Ascarate Lake	El Paso, TX	AS	wastewater	16
10	Rio Bosque Wetlands	El Paso, TX	RB1, RB2	wastewater*	11
11	Sandia Springs	Balmorhea, TX	SS1, SS2, SS3	non-wastewater*	1
12	BJ Bishop Wetlands	Presidio, TX	BJ	wastewater*	1
13	Cattail Falls	Big Bend National Park, TX	CF	non-wastewater*	<1
14	Manzanita Springs	Guadalupe Mountains National Park, TX	MS	non-wastewater	<1

Study Sites

We sampled wetland sites throughout the southwest United States, primarily in the Chihuahuan Desert and some in the neighboring Sonoran Desert (Figure 1). Most sites were sampled twice, once a summer in two different years, however, Cattail Falls and Manzanita Springs were only sampled once due to being added later in the project and COVID-19 travel restrictions. Sites located in El Paso, TX were sampled once every summer during the three sampling years. Some sites, such as the Rio Bosque Wetlands, were sampled in more than one area, as indicated in by multiple code names in Table 1.1 (i.e., RB1, RB2). Different areas sampled within one wetland were usually associated with separate ponded areas.

Rainfall in the Chihuahuan desert averages 247 mm annually and occurs primarily during the summer months (June-September) when peak ambient temperatures average 36°C (J. A. Matthews 2014). The Sonoran Desert receives between 75 to 380 mm of rain per year and with peak summer temperatures reaching up to 49°C (U.S. National Park Service 2019). During 2018 and 2019, the southwest received near-below to below average precipitation and experienced above average temperatures (NOAA 2019, 2020). Sites sampled in 2020 experienced near average precipitation with much above average temperatures (NOAA 2021).

Water depths for the sites ranged from 0.3 meters to greater than 1.5 meters, however areas sampled were in wadable depths (<0.5 meters). Sites were grouped by water sources: either wastewater (effluent water from treatment plants) or non-wastewater (i.e., Rio Grande, spring, or stormwater) (Table 1.1). Wastewater sites generally received continuous amounts of effluent water throughout the growing seasons. Non-wastewater sites included those that were flooded with water from the Rio Grande (Las Palomas, La Mancha, Rio Grande 1, Rio Grande 2); however, these were floodings and not considered riverine wetlands. Crossroads Pond differed by additionally receiving stormwater inflow sporadically throughout the year, especially during the summer monsoon season.

Figure 1.1: Map of all sites sampled in Arizona, New Mexico, and Texas during the summer months of 2018-2019.

1.2 METHODS

Macroinvertebrate Sampling

Prior to sampling, we qualitatively identified the three dominant macrophyte types in each wetland. Macroinvertebrate samples were then collected with three successive dips using a 250μm d-frame kick net from each of these three habitats. Contents from all dips were pooled into 1 composite sample. Because all sites were sampled with the same effort (3 dips in 3 different habitats for a total of nine dips per wetland), abundances are reported as catch per unit effort (CPUE) and are directly comparable. Macroinvertebrates were counted and identified in the field with some specimens kept for further identification in the lab. Specimens were preserved in 70% ethanol, stored at room temperature, and identified to the lowest possible taxonomic level. Many groups were identified to the genus level with some being identified to species names, however, order and family were used in analysis due to some samples not being identified past family (Merritt and Cummins 1996; Smith 2001).

Using these data, a variety of metrics of macroinvertebrate community composition were calculated, including those that summarized taxonomic richness, composition, and functional feeding groups. A full list of taxa with designated functional feeding guilds can be found in the Appendix. Ephemeroptera, Odonata and Tricoptera (EOT) composition was used as measure of diversity and water quality (Mereta et al. 2013). Similar metrics including Plecoptera (i.e., EPT) were not included due their absence in our study areas. Using abundance data, Simpson's Diversity Index (λ) was calculated for each sampling visit as a measurement of macroinvertebrate diversity (Simpson 1949). Both λ scores and the percentages of functional feeding groups were computed for each site visit, then averaged for sites that were sampled more than once (Anderson and Davis 2013).

Water Quality Sampling

At the time of macroinvertebrate collection, physicochemical conditions such as pH and conductivity were collected in the field using a YSI® 556 multi-probe (YSI Incorporated Yellow Springs, OH, USA). Dissolved organic carbon (DOC) and total dissolved nitrogen (TDN) samples were determined after filtration through pre-ashed GF/F filters and stored in precombusted amber glass bottles at 4°C until analysis (APHA 1998). Both were determined using a Shimadzu TOC-L analyzer with TMN module. Water samples for additional water chemistry were collected from an open water location using acid washed HDPE bottles. Anion concentrations (Cl^-, SO_4^{2-}, NO_3^-, PO_4^{3-}) were measured on a Dionex 2100 ion chromatograph. Alkalinity was measured using a Mettler Toledo G20 auto-titrator. Turbidity was measured in triplicate using a Hach 2100 turbidimeter. Percent organic matter was determined using a "loss on ignition" method in which a subsample of the sediment was dried at 100°C for one hour. The sample was then weighed and heated in a muffle furnace at 550°C for fifteen minutes and reweighed (APHA 1998). Percent organic matter was calculated from the mass lost after ashing.

Chlorophyll-a concentration, as an estimate of algal biomass, was quantified for both phytoplankton and periphyton. To measure phytoplankton, a known volume of water (between 150-1000 mL) was collected from open water and filtered through a GF/C filter to collect algae floating in the water column. Filters were frozen until analysis. Periphyton was collected from pond sediment surfaces at three haphazard locations in each pond using a spatula and an inverted petri dish. All three periphyton samples were combined into one composite sample. Algae were separated from the sediment by rinsing with distilled water, pouring off and retaining the algal-rich supernatant solution and repeating ten times, at which point the solution typically became clear. A subsample of the resulting algal suspension was stored in a test tube, wrapped in foil and frozen until the analysis for chlorophyll. Chlorophyll a (CHLa) was extracted into 90% acetone for 24 h in the freezer. Absorbance of the extract was measured with a Genesis 10 UV spectrophotometer (APHA 1998). Concentrations were calculated on a volumetric basis for phytoplankton ($\mu g\ L^{-1}$) and by area sampled for periphyton ($\mu g\ cm^{-2}$). Phytoplankton CHLa was corrected for turbidity and phaeopigments by acidification (Wetzel and Likens 2002); total CHLa refers to uncorrected CHLa values.

Data Analysis

All statistical analysis and graphing were performed in R (Version 4.1.2). A Principal Component Analysis (PCA) was used to describe underlying gradients in the environmental data. All environmental data, including physicochemical properties and algal biomass were entered into the analysis. The PCA analysis was conducted using the "princomp" function and data were transformed and standardized as required, to approximate a normal distribution (McCune and Grace 2002). Graphing of the PCA was performed with the "factoextra" package. Simpson Diversity Indices were calculated using the "vegan" package. Water quality and macroinvertebrate metrics were compared between wastewater and non-wastewater sites using Wilcoxon rank-sum tests, due to non-normality of data. Pearson correlation coefficients were determined to relate Simpson's Diversity Index scores and PCA scores for all sites. Normality of residuals was confirmed for all regression analyses.

1.3 Results

Environmental Gradients

Environmental conditions ranged from nutrient-poor (non-detectable levels of NO_3^- and PO_4^{3-}) to nutrient-rich, with relatively high levels of water column chlorophyll (maximum 352 ug/L), DOC (maximum 75ppm) and nutrients (Table 1.2). There was also a large gradient of salinity-related variables such as Cl^- and SO_4^{2-} ranging from non-detectable amounts to 828.5 and 5309 ppm, respectively. Water clarity ranged from clear (1.8 NTU) to highly turbid (208.3 NTU). Sites generally had largely inorganic sediments with the highest percentage of organic matter only 9%.

Table 1.2: Median, standard deviation, and range of water physio-chemical variables for wetlands sampled in the Chihuahuan and Sonoran deserts. Phytoplankton CHLa was corrected for turbidity and phaeopigments by acidification (Wetzel and Likens 2002); Total CHLa refers to uncorrected CHLa values.

	Median	SD	Min	Max
Conductivity (mS/cm)	3.30	3.89	0.21	16.40
Alkalinity (meq/L)	200.11	130.80	21.98	457.62
Turbidity (NTU)	24.3	38.3	1.8	208.3
pH	7.4	0.8	6.3	9.3
Dissolved Organic Carbon (ppm)	13.84	17.46	0.29	75.04
Organic Matter %	1.3	3.0	0.00	9.0
Periphyton (µg cm^{-2})	0.00	0.01	0.00	0.02
Total Phytoplankton CHLa (µg L^{-1})	17.58	34.91	0.16	352.28
Corrected CHL (µg L^{-1})	21.82	60.63	0.00	146.68
Cl^-(ppm)	281.84	290.23	0.00	828.53
SO_4^{2-} (ppm)	536.38	1073.27	0.00	5309.00
NO_3^- (ppm)	1.62	2.79	0.00	9.00
PO_4^{3-} (ppm)	2.63	4.97	0.00	26.00
Total Dissolved Nitrogen (ppm)	2.84	3.85	0.00	7.00

The PCA yielded two dimensions explaining more than 50% of variation in the environmental data: PCA 1 accounted for 31.9% of the variability, and PCA 2 accounted for 22.1%. For PCA1, DOC was the greatest driver of variance, along with salinity-related variables such as Cl^-, SO_4^{2-}, alkalinity and conductivity. Both total and corrected phytoplankton CHLa were also related to this axis (Figure 2a; Table 1.4). This axis contrasted urban ponds with high salinity, such as Keystone and Crossroads, to more remote sites, such as Manzanita Springs and Cattail Falls, with relatively low salinity levels. Nutrients such as NO_3^-, PO_4^{3-} and TDN, as well as soil organic matter, were the greatest drivers of variance along PCA 2 (Figure 2a; Table 1.3). This axis

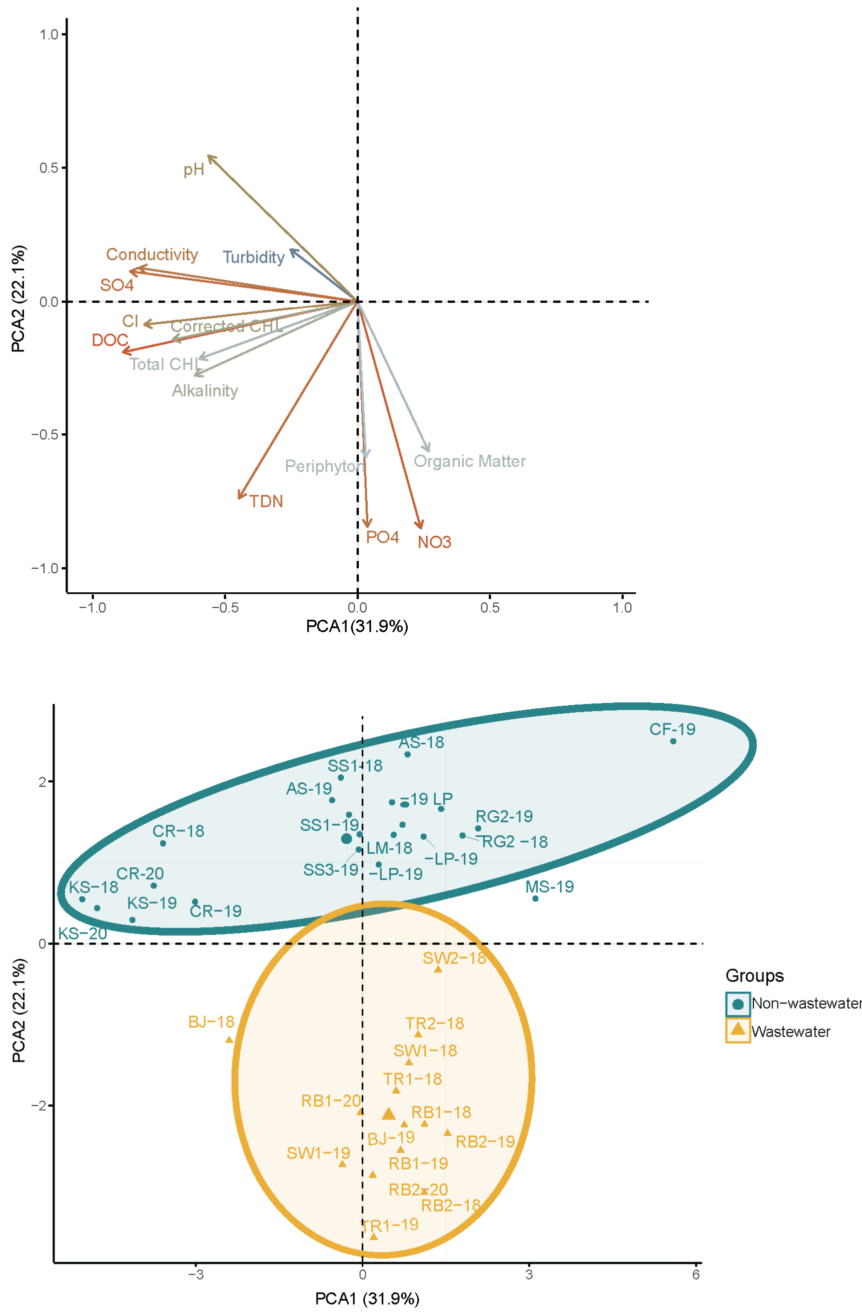

Figure 1.2: Plots of PCA scores of environmental data collected from 14 wetlands in the Chihuahuan and Sonoran deserts with (a) environmental vectors, where longer arrows indicate stronger correlations with the axis scores, and (b) sites grouped by water source. Sites codes are listed in Table 1.1 and appear with the last two digits of the year they were sample

contrasted sites flooded with effluent water (Rio Bosque Wetlands, Sweet Water, Tres Rios and BJ Bishop) to all other sites. Wetland sites flooded with water from the Rio Grande (Rio Grande

1 & 2, Las Palomas, La Mancha) were shown to have relatively low levels of nutrients (Table 1.5). Differences based on sites flooded with wastewater versus those flooded with non-wastewater is especially apparent, as they occupied distinct groups on the PCA plot (Figure 2b)

Table 1.3: Correlation coefficients (*r*) of water physiochemical parameters with PCA1 and PCA2 scores from wetlands sampled in the Chihuahuan and Sonoran deserts. Significance: ***p<0.0001, **p<0.01, *p<0.05

	PCA1	PCA2
Conductivity (mS/cm)	-0.8250***	0.1259
Alkalinity (meq/L)	-0.6148**	-0.2793
Turbidity (NTU)	-0.2529	0.1949
pH	-0.5632*	0.5456*
Dissolved Organic Carbon (ppm)	-0.8855***	-0.1917
Organic Matter %	0.2686	-0.5627*
Periphyton (µg cm^{-2})	0.0318	-0.5853*
Total Phytoplankton CHL (µg L^{-1})	-0.5972*	-0.2161
Corrected CHL (µg L^{-1})	-0.6981**	-0.1450
Cl^{-}(ppm)	-0.8052***	-0.0884
SO_4^{2-} (ppm)	-0.8586***	0.1127
NO_3^{-} (ppm)	0.2392	-0.8511***
PO_4^{3-} (ppm)	0.0374	-0.8458***
Total Dissolved Nitrogen (ppm)	-0.4484	-0.7393***

Macroinvertebrate Metrics

In total, 13,760 macroinvertebrate individuals were collected over the time of the study. Total abundances ranged from 15 to more than 1000 per unit effort, the latter being sites that were dominated by mostly ostracods and Cladocera, while the number of taxa groups found at each site ranged from 2 to 10, depending on the site.

When grouped by water type, many metrics were significantly higher in sites that were fed with non-wastewater, including both tolerant and sensitive taxa (Table 1.4). % EOT, which was used as a measure of both diversity and water quality, was also high in site receiving non-wastewater, as were the percentage of predators and collector-gatherers (Table 1.4). Non-wastewater sites also had a more even representation by functional feeding groups, notably collectors, predators and filterers, while wastewater sites were largely dominated by filterers (Figure 4). Similarly, within the non-wastewater sites (low nutrients), results showed multiple taxa with relatively even percent abundances (10-15%), including Ephemeroptera, Odonata, Hemiptera, Coleoptera and Amphipoda (Table 1.4). There were no correlations when diving Odonata into subgroups: Anisoptera and Zygoptera. Conversely, wastewater fed sites were dominated by filterers (Figure 4; Table 1.4), largely represented by significantly more ostracods (62%) and cladocerans (12%).

Table 1.4: Means and standard error of macroinvertebrate metrics from wetlands in the Chihuahuan and Sonoran deserts grouped by non-wastewater and wastewater source type. Wilcoxon rank sum significant difference between groups ***p<0.0001, **p<0.01, *p<0.05, + <0.10, without asterisks indicate non- significance. EOT= Ephemeroptera, Odonata, Tricoptera

	Non-Waste	**Waste**
Total taxa	7.16 (0.46)	9.78 (1.12)+
No. of orders	5.39 (0.42)	6.50 (0.81)
No. of families	6.95 (0.42)	8.50 (0.81)
Simpson Diversity Index	0.57 (0.20)*	0.39 (0.22)
% Ephemeroptera	13.56 (3.16)	3.17 (1.56)
% Odonata	10.86 (2.40)+	3.67 (1.12)
% Amphipoda	11.49 (23.07)	11.05 (20.19)
% Gastropoda	7.56 (9.7)	4.06 (7.96)
% Hemiptera	11.05 (3.59)*	1.55 (0.93)
% Coleoptera	12.54 (4.10)**	0.30 (0.15)
% Diptera	5.75 (1.30)*	2.00 (0.60)
% Chironomidae	4.49 (6.79)	1.70 (2.14)
% Cladocera	5.84 (11.7)	12.26 (26.55)+
% Decapoda	1.34 (3.71)	0.11 (0.33)
% Ostracoda	20.33 (5.62)	61.75 (7.64)**
% EOT	24.46 (3.93)**	6.85 (2.34)
% Predators	32.51 (4.98)**	5.53 (1.43)
% Scrapers	8.91 (11.58)	4.18 (8.16)
% Filterers	25.77 (6.00)	74.02 (5.78)***
% Collector-gatherers	29.38 (4.58)+	15.93 (5.35)

λ scores were found to be positively associated with both PCA1 ($r^2 = 0.11$, $p = 0.04$) and PCA 2 ($r^2 = 0.16$, $p = 0.01$) axes (Figure 3) indicating that increased salinity and nutrient levels resulted in a decline in macroinvertebrate community diversity and evenness. When comparing the λ scores of wastewater sites and non-wastewater sites, there was a significant difference with non-wastewater sites displaying higher macroinvertebrate diversity scores (Table 1.4). There were no significant correlations between percent abundances of taxa or functional feeding groups and either of the PCA axes after corrections for multiple comparisons.

Table 1.5: Means and standard error of water quality parameters grouped by water type. Wilcoxon rank sum difference between groups ***p<0.0001, **p<0.01, *p<0.05, without asterisks indicate non- significance.

	Non-wastewater	**Wastewater**
Conductivity (mS/cm)	4.76 (0.98)	2.28 (0.49)
Dissolved Organic Carbon (ppm)	15.45 (4.65)	7.75 (1.23)
Alkalinity (meq/L)	223.65 (24.55)	245.60 (32.32)
Corrected Phytoplankton CHLa (µg L^{-1})	28.76 (15.94)	21.28 (11.72)
Cl^- (ppm)	358.51 (71.29)	155.86 (23.76)
SO_4^{2-}(ppm)	951.65 (285.07)	122.31 (12.97)
Total CHL (µg L^{-1})	14.55 (6.10)	22.54 (11.58)
Total Dissolved Nitrogen (ppm)	1.39 (0.40)	5.20 (1.33) **
NO_3^- (ppm)	0.23 (0.06)	4.64 (0.89) ***
PO_4^{3-}(ppm)	0.15 (0.05)	7.34 (1.72) ***
Periphyton (µg cm^{-2})	0.001 (0.0003)	0.008 (0.002) **
Organic Matter %	2.0 (0.3)	0.05 (0.005) **
pH	7.7 (0.2) **	6.89 (0.14)
Turbidity (NTU)	30.5 (9.6)	14.05 (4.39)

1.4 Discussion

Wetlands in this study tended to vary along a gradient of either salinity or nutrient enrichment, with salinity appearing to explain more among-site variability. While salinity may be the greatest driver of environmental variation amongst desert wetlands, nutrient loads from wastewater appears to be the greatest driver of variation within macroinvertebrate communities.

This follows the trajectory of other studies listing anthropogenic disturbances as greater drivers of community composition over salinity loads (Moreno et al. 2010). Overall, our hypotheses correctly indicated that increased levels of nutrients, such as those found in wastewater from treatment sites has negative effects on macroinvertebrate diversity and abundances in sensitive taxa. Furthermore, this has shown to cause changes in distribution of functional feeding groups, specifically leading to communities dominated by filter feeders. While salinity also led to reduced diversity of macroinvertebrate taxa, results did not show an effect of elevated salinity on any taxonomic group or functional feeding group.

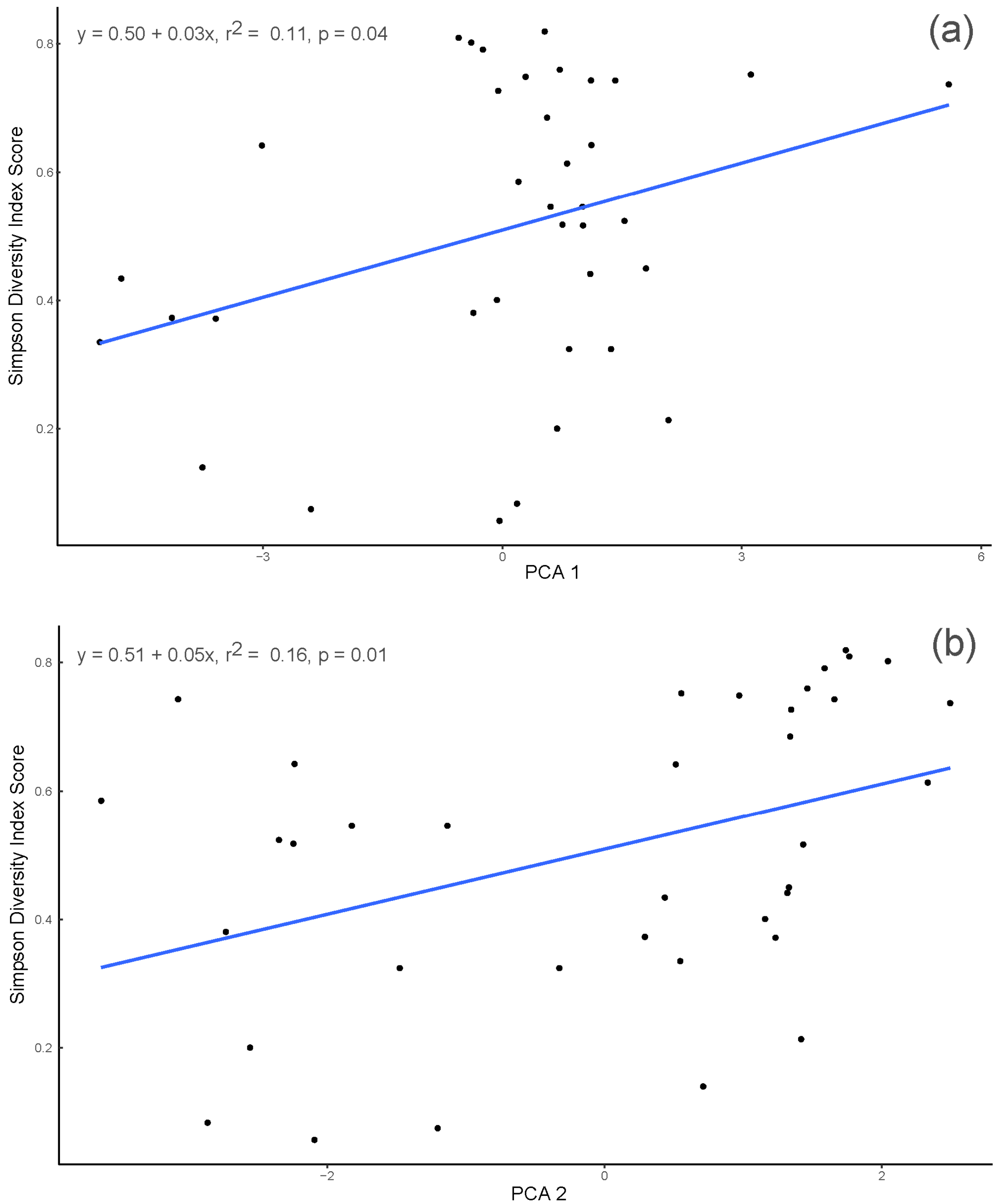

Figure 1.3: Regression plots depicting significant associations ($p<0.05$) of Simpson Diversity Index scores with (a) PCA1 and (b) PCA2 axes scores for all 14 wetlands in the Chihuahuan and Sonoran deserts

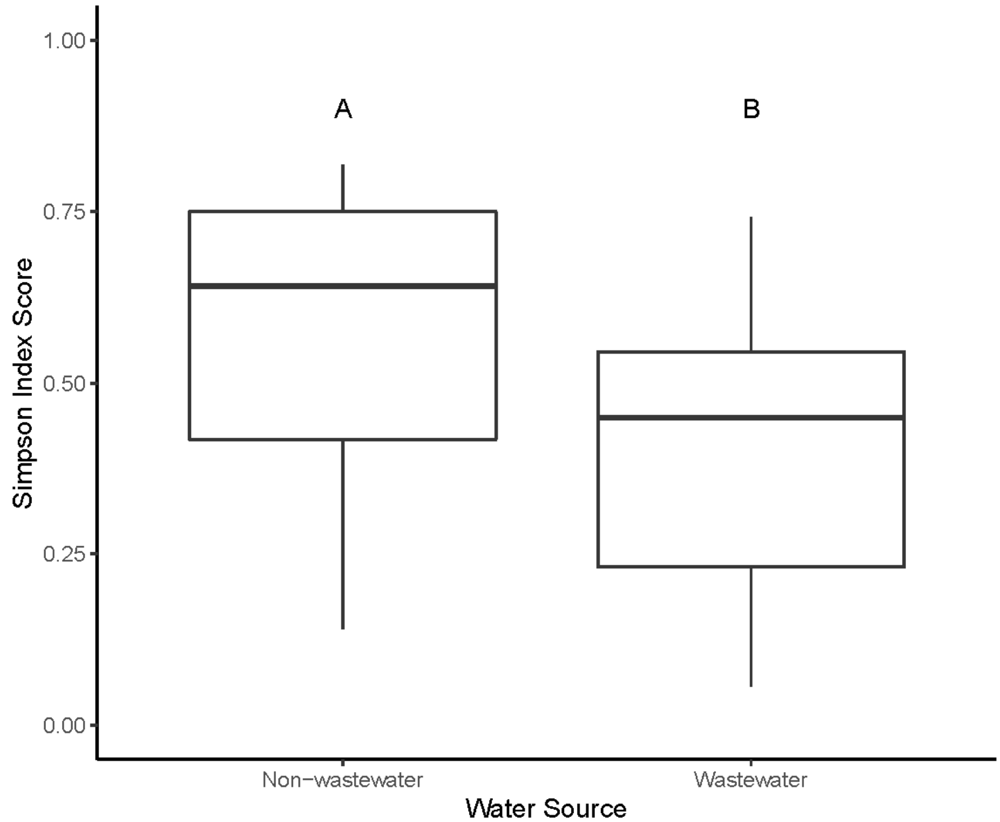

Figure 1.4: Boxplot depicting average Simpson Index Scores for wetlands in the Chihuahuan and Sonoran deserts grouped by water source type: non-wastewater and wastewater. Letters indicate statistical differences (p = 0.02).

Salinity

The salinity gradient contrasted permanent and isolated spring sites such as Cattail Falls and Manzanita Springs, with low chloride, sulfate, and conductivity levels, to known naturally high saline sites within El Paso, TX city limits, such as Keystone and Crossroads. The relatively high levels of salinity within these two sites are likely due their location. These arid region wetlands are both highly dependent on the regional, saline water table to maintain water levels. Groundwater is known to have high levels of salts and sulfate in the region (Hiebing et al. 2018). Irregular influx of water and rising temperatures could lead to high evaporative conditions, which could contribute to the high levels of salinity within these sites (Jolly et al. 2008; Borrok and Engle 2014).

DOC and chlorophyll-a were also shown to vary along the salinity gradient. Sites that are highly saline have been shown to have suppressed microbial activity (including those which take up DOC) which may explain the higher levels of available DOC within these sites (Straathof et al. 2014; Yang et al. 2018). In some studies, the increase in chlorophyll-a levels within highly saline sites was related to SO_4^{2-} and salt-induced aggregation of suspended matter, which can lead to increase light penetration of the water column and thus, high rates of photosynthesis (Donnelly et al. 1997; Nielsen et al. 2003). However, given that there was no effect of water clarity in our study, this is unlikely.

While the salinity gradient explained most of the environmental variability among sites, there were relatively few significant associations between salinity and metrics of macroinvertebrate community composition. Sites that were higher in salinity tended to have a lower Simpson Index Scores, thus lower macroinvertebrate diversity and evenness. This remains consistent with similar studies showing negative relationships between macroinvertebrate taxonomic richness and functional evenness with increasing levels of salinity and related parameters (Kefford et al. 2004; Chemers et al. 2011; Ordonez et al. 2011; Cuthbert et al. 2020; Muresan et al. 2020). Although other studies within Chihuahuan desert freshwater systems have found that amphipods are adapted to high levels to salinity (Cuthbert et al. 2020; Dinger et al. 2005; Gervasio et al. 2004) and coleopterans, in general, are tolerant of high salinity within freshwaters (Colombetti et al. 2020; Garrido and Munilla 2008; Lancaster and Scudder 1987; S. Sharma, Sharma, and Pir 2019), we were unable to verify these trends with our data.

Nutrients

Not surprisingly, there was a distinct difference in physiochemical features between sites flooded with wastewater and those flooded with non-wastewater. The sites flooded with wastewater were significantly higher in nutrients such as NO_3^-, PO_4^{3-} , and TDN, typical of effluent water (Hamdhani, Eppehimer, and Bogan 2020; Zhuang et al. 2019). Periphyton was also significantly higher in the wastewater sites, likely due to the high levels of nutrients, which are often a limiting factor of benthic algal communities (Power 1992; Francoeur et al. 1999).

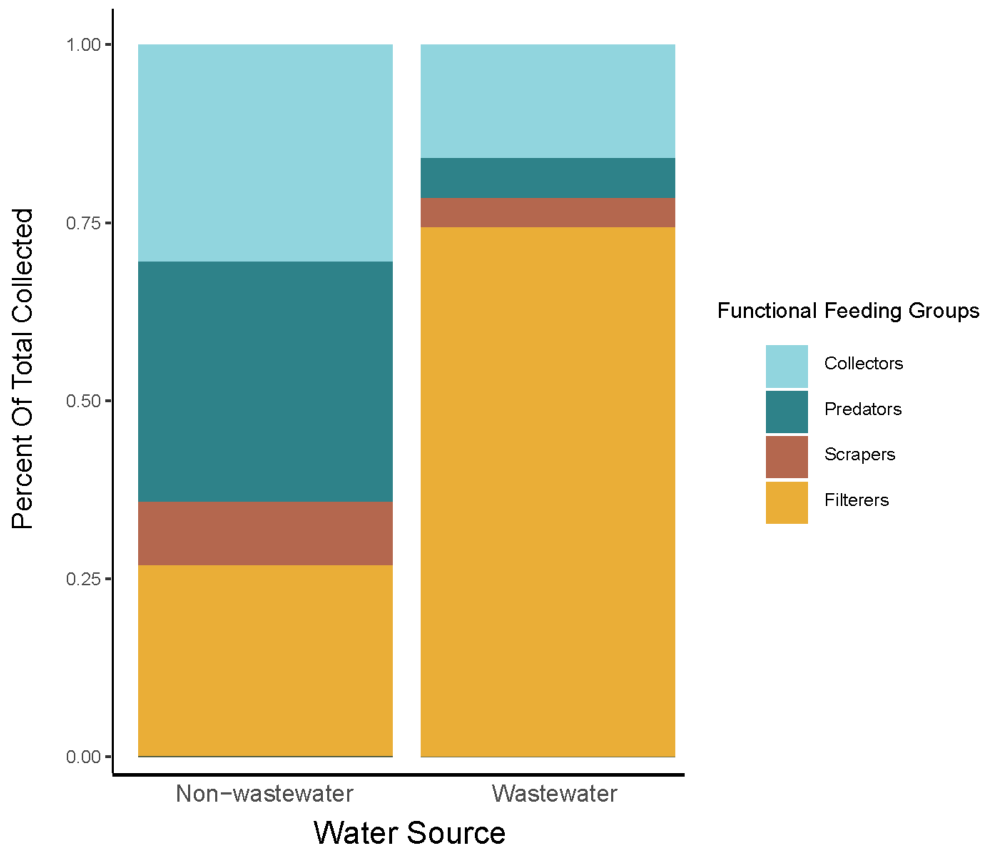

Figure 1.5: Relative abundances of functional feeding groups from wetlands in the Chihuahuan and Sonoran deserts grouped by water source types: non-wastewater and wastewater.

Sites with lower nutrient levels had more diverse and even macroinvertebrate communities. Lougheed et al. (2008) found that wetlands in less developed, nutrient-poor locations had increased diversity of multiple taxonomic groups. This is consistent with multiple studies finding homogenization of macroinvertebrate communities with increased nutrient levels, some stating total phosphorus as the main driver of decline in diversity (Spieles and Mitsch 2000; Hsu et al. 2011; Ouyang et al. 2018; Qu et al. 2019). Along the nutrient gradient, there was a clear contrast in macroinvertebrate community structure between wastewater sites and non-wastewater sites. The presence of multiple taxa with relatively even percent abundances (10-15%) agrees with findings of increased evenness in non-wastewater or low nutrient sites compared to wastewater wetlands, specifically with the increase in more sensitive taxa such as Ephemeropterans (Becerra Jurado et al. 2009; Hsu et al. 2011). The percent EOT increased significantly within non-wastewater sites, likely due to their sensitivity to anthropogenic impacts (Kutcher and Bried 2014; Ode, Rehn, and May 2005). The increase in predators in the absence of wastewater was also found by other studies relating declines in predators because of increased nutrients and anthropogenic disturbances (Fu et al. 2016; Zhang et al. 2019). Corixidae, in particular, have been commonly observed in other studies in Rio Grande habitats (Bain et al.2011, Burdett et al. 2015), which were generally lower in nutrients than wastewater fed sites.

Functional feeding groups were also evenly represented in the absence of wastewater, with collectors, predators, and filterers each forming approximately one-third of the composition. In contrast, filterers (ostracods in particular) dominated the community in wastewater sites, representing more than 60% of the total abundance, and increased in abundance along the PCA nutrient gradient. Increased relative abundance of filter feeders in high nutrient sites could be due to increased periphyton algae levels within these sites (Hillebrand and Kahlert 2001).

Other studies indicated plant diversity as being the main driver of diversity and habitat selection in macroinvertebrates (Hsu et al. 2011; Perron and Pick 2020; Perron et al. 2021). Although we did not quantitatively evaluate plant species richness, there appeared to be a similar trend with macroinvertebrate richness increasing within sites that tended to have higher plant diversity, many of which are non-wastewater sites.

Results from this investigation could be an important consideration for maintaining or restoring biodiversity to macroinvertebrates in arid region wastewater wetlands. More research is needed to confirm whether prolonged nutrient inputs in wastewater fed wetlands leads to further homogenization of macroinvertebrate communities, or whether this becomes an alternative stable state for these sites. Recent work has shown that the creation of wetland habitats fed by wastewater can substantially alter and improve aquatic macroinvertebrate community composition in a desert site relative to non-wetland aquatic habitats (Chapter 2). Thus, while wastewater sites are substantially different than their more natural counterparts, creation of these sites can benefit landscape level diversity (Stanislawczyk et al. 2018). It is suggested that, where possible, managers of these valuable created habitats might try to find less nutrient-rich water sources, such as groundwater, to enhance the water quality in their sites. With reduced nutrient levels, we would expect to see an increased proportion of EOT, predators and collectors, among others. Further investigation is required to determine if other trophic levels are equally impacted by salinity and nutrient levels within these arid wetland ecosystems.

CHAPTER 2: CHANGES TO A WATER DELIVERY SYSTEM AND ITS EFFECTS ON A DESERT WETLAND.

2.1 INTRODUCTION

Since 1700, it is estimated that 87% of wetlands have been lost worldwide; a rate 3 times faster than that of natural forests (Ramsar Convention on Wetlands, 2018). Despite many restoration programs in effect, there is still projected to be more wetland loss in the future due to climate change and increasing demand for freshwater (Vörösmarty et al. 2010; Xi et al. 2021). In arid regions, the hydrology of freshwater habitats is especially vulnerable due to the higher evapotranspiration rates and drought conditions (Overpeck and Udall 2020; Strzepek et al. 2010). This has led to the use of alternative sources of freshwater to restore, maintain or create freshwater habitats.

Effluent from wastewater treatment plants has been reused around the world for agricultural irrigation and is becoming more popular as constant water source for freshwater systems (Hamdhani, Eppehimer, and Bogan 2020; Toze 2006). This includes rivers (Bogan et al. 2020; Hamdhani, Eppehimer, and Bogan 2020), streams (Luthy et al. 2015), wetlands (Hsu et al. 2011; Matamoros, Rodríguez, and Bayona 2017; Quanz et al. 2021), lakes (Lasee et al. 2017) and ponds (Becerra et al. 2009). While this provides a much needed water source to these habitats, byproducts from the wastewater may have effects on the systems water quality and biota (Hamdhani, Eppehimer, and Bogan 2020). Most often addressed are the high levels of nutrients found within the wastewater due to ecosystem uptake of these compounds (Karpiscak et al. 2001; Metzeling et al. 2003; Whitton et al. 2016). However, the long term exposure of effluent byproducts on these habitats remains to be seen (Hamdhani, Eppehimer, and Bogan 2020). While some studies have reported differences in macroinvertebrate communities between wastewater and non-wastewater sites, these systems still act as refugia to many aquatic taxa (Becerra Jurado et al. 2009; Hsu et al. 2011). With the increase in habitat restoration or creation using these methods, comes the need to be able to assess restoration success.

Currently, there is little information about wetland restoration trajectories, even less concerning desert wetlands. While some patterns with restoration have been observed, most trajectory models prove to be too simple and unrealistic (J. W. Matthews, Spyreas, and Endress 2009; J. B. Zedler et al. 1999). Additionally, most models created for one wetland type are not easily transferred to others; creating a need for habitat specific restoration models (J. Zedler 2000). While evaluating the success of wetland restoration projects is difficult, most are only monitored for 2 to 5 years if at all (Cole and Shafer 2002; U.S. Government Accountability Office 2005). When it does occur, the focus is primarily on the establishment of wetland vegetation and hydric soils which may be poor indicators of wetland function (Kihslinger 2008).

Biological indicators of wetland restoration have included native plant abundance and biomass (Adamus and Brandt 1990; Lopez and Fennessy 2002), population size of wetland fauna including fishes, crustaceans, and birds (Trexler and Goss 2009), and diversity indices (Ruiz-Jaen and Aide 2005). Less popular methods include ecological processes, such as nutrient cycling, due to that fact they are slower to recover from disturbance and require multiple measurements over time (Ruiz-Jaen and Aide 2005). Though they have often been utilized as indicators of wetland health and water quality, macroinvertebrate community assemblages have not been commonly used to measure restoration success in wetlands (Ruiz-Jaen and Aide 2005; Swartz et al. 2019). Some studies have observed broader patterns of succession in macroinvertebrates such as initial colonization of generalist active dispersers followed by establishment of more specialist passive dispersers (Brown, Smith, and Batzer 1997; Ruhí et al. 2012; Sartori et al. 2015). Similar patterns are observed in functional feeding groups with the increase in niche availability over time (Coccia et al. 2021). Along with biodiversity, functional diversity can provide a more complete representation of community responses to restoration (Perez Rocha et al. 2018).

Though attempts have been made to track restoration though the use of macroinvertebrates in temperate regions, most have proven to be inconclusive in identifying specific metrics to indicate success (Marchetti, Garr, and Smith 2010; Meyer and Whiles 2008; Ruhí et al. 2012). However, wetlands in arid regions have been shown to have differing “core” macroinvertebrate assemblages than non-arid wetlands (Ruhí, Batzer, and Ruhí 2013), so it cannot be assumed that they will respond the same way to restoration.

Though macroinvertebrate trajectories remain unclear, several studies have highlighted differences in assemblages between created and natural wetlands. For example, it was determined that macroinvertebrate diversity (Simpson Diversity Index and Invertebrate Community Index) of created wetlands was significantly lower when compared to natural wetlands (Acharyya and Mitsch 2000; Spieles and Mitsch 2000; Swartz et al. 2019) with dissolved oxygen and specific conductivity being the best predictors for species diversity (Spieles and Mitsch 2000). In wastewater ponds, drivers of community composition were identified as pH, vegetation structure and pollution levels (Becerra et al. 2009). When comparing created, impacted and reference wetlands, it was determined that the amount of vegetation had the greatest influence on macroinvertebrate taxonomic richness (Swartz et al. 2019). While these few studies give some insight to what may drive community composition, none of them were conducted in desert wetlands, where water characteristics are very different, especially in salinity and hydroperiod.

One desert wetland that recently has been restored is the Rio Bosque Wetlands, which are located along the Rio Grande River where it marks the US-Mexico border in El Paso, TX (Figure

Figure 2.1: Map of the Rio Bosque Wetlands in El Paso, Texas. Sample sites from 2016-2019 are represented by a yellow star. Map from the Center for Environmental Resource Management: Rio Bosque Wetlands webpage.

2.1). The area in which the park lies was drastically changed in mid-1930s when the Rio Grande was channelized, preventing the river water from reaching the site. In 1997, the U.S. section of the International Boundary Water Commission (IBWC) began to rebuild the wetland park using wastewater from the adjacent Roberto Bustamante Wastewater Treatment Plant (Watts, Sproul, & Hamlyn, 2002). In the years leading up to 2015, the park received an average of 124 days of water per year, largely outside of the growing season and mostly within the channels. Since 2015, the average number of water days has increased to 272, which includes water delivered during the summer growing season. In 2016, there was a significant increase in water availability to the site, allowing water to fill the wetland cells for the first time during the growing season in 12 years (CERM, 2016). Currently, the park continues to receive treated wastewater from the Roberto

Bustamante Water Treatment plant, as well as water from the irrigation canals and groundwater pumps (Figure 2.2). Recent attempts to track restoration success of the Rio Bosque Wetlands highlighted the scarcity of information on indicators of wetland quality in the U.S. southwest.

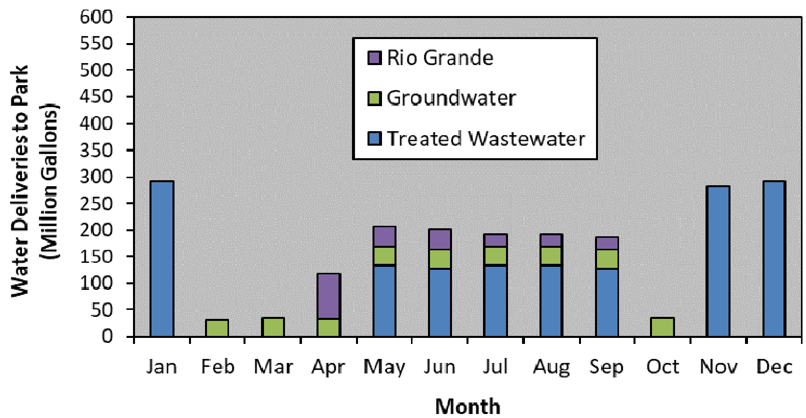

Figure 2.2: Figure from the Center for Environmental Resource Management: Rio Bosque Wetlands webpage depicting the current typical water availability pattern at the Rio Bosque Wetlands.

In this study, we examine the use of macroinvertebrates to track wetland restoration and how changes in water quality and quantity are impacting the desert macroinvertebrate communities that play an integral part in the ecology of these isolated wetland habitats. We suspected that increased water availability during the growing season will lead to an increase in macrophyte abundance, an increase in passive disperser abundance and an overall change in macroinvertebrate assemblage. Subsequently, we aim to determine if and how the community compostion between the ponds and channels differ from each other after the change in water availability. By determining these successional patterns, we hope to be able better track the restoration of these invaluble habitats and highlight key factors that could lead to better management practices of restored or created wetlands.

2.2 METHODS

Study Site

The Rio Bosque Wetlands are a restored riparian wetland that is part of a 372-acre City of El Paso park the University of Texas at El Paso (UTEP) manages through the Center for Environmental Resource Management (CERM**).** The Rio Bosque Wetlands Park was initially constructed in 1997 and designed to include 40 acres of wetland habitat within 2 wetland cells or ponds. The park is enclosed by irrigation canals and drains on three sides, and the western boundary of the park lies adjacent to the Rio Grande,

which forms the international border between the U.S. and Mexico in this area (Watts, Sproul, & Hamlyn, 2002). Water quality and macroinvertebrate data was collected from the Rio Bosque Wetlands flooded channels during the summer of 2014. Sampling during subsequent years is varied based on water availability; however, largely occurred from May through September, within both channels and wetland ponds. Macrophyte data were also collected as part of related studies during the summers of 2014, 2016, and 2017.

Macroinvertebrate Sampling

Prior to sampling, we qualitatively identified the three dominant macrophyte types in each wetland pond. Macroinvertebrate samples were then collected with three successive dips using a 250µm d-frame kick net from each of these three habitats. Contents from all dips were pooled into 1 composite sample. Because all sites were sampled with the same effort (3 dips in 3 different habitats for a total of nine dips per wetland), abundances are reported as catch per unit effort (CPUE) and are directly comparable. Macroinvertebrates collected with the net were counted and identified in the field with some specimens kept for further identification in the lab. Specimens were preserved in 70% ethanol, stored at room temperature, and identified to the lowest possible taxonomic level. Many groups were identified to the genus level with some being identified to species names, however, order and family were used in analysis due to some samples not being identified past family (Merritt and Cummins 1996; Smith 2001). Abundance data was summarized and coded by rarity of taxa and used to calculate approximate percent abundances and macroinvertebrate metrics for each site visit (Table 2.1). Coded numbers were used in the calculation of approximate percent abundances. Percent abundances of macroinvertebrates, functional feeding groups and active/passive disperser taxa were grouped and compared by time and area of collection (channels vs. ponds) (Merritt and Cummins 1996; Wiggins, Mackay, and Smith 1980).

Table 2.1: Code number, abundance and rarity used to summarize macroinvertebrate abundance data.

Code Number	Abundance	Rarity
1	< 5	Rare
2	5-10	Occasional
3	10-20	Common
4	20-50	Abundant
5	>50	Dominant

Water Quality Sampling

At the time of macroinvertebrate collection, physicochemical conditions such as pH and conductivity were collected in the field using a YSI® 556 multi-probe (YSI Incorporated Yellow Springs, OH, USA). Turbidity was measured in triplicate using a Hach 2100 turbidimeter. Water chemistry samples were be collected in acid washed bottles from each wetland subsite. Total phosphorus (TP) was determined using the ascorbic acid method following persulphate digestion (APHA, 1998).

Chlorophyll-a concentration, as an estimate of algal biomass, was quantified for both phytoplankton and periphyton. To measure phytoplankton, a known volume of water (between 150-1000 mL) was collected from open water and filtered through a GF/C filter to collect algae floating in the water column. Filters were frozen until analysis. Periphyton was collected from pond sediment surfaces at three haphazard locations in each pond using a spatula and an inverted petri dish. All three periphyton samples were combined into one composite sample. Algae were separated from the sediment by rinsing with distilled water, pouring off and retaining the algal-rich supernatant solution and repeating ten times, at which point the solution typically became clear. A subsample of the resulting algal suspension was stored in a test tube, wrapped in foil and frozen until the analysis for chlorophyll. Chlorophyll a (CHLa) was extracted into 90% acetone for 24 h in the freezer. Absorbance of the extract was measured with a Genesis 10 UV spectrophotometer (APHA 1998). Concentrations were calculated on a volumetric basis for phytoplankton (μg L-1) and by area sampled for periphyton (μg cm-2). Phytoplankton CHLa was corrected for turbidity and phaeopigments by acidification (Wetzel and Likens 2002); total CHLa refers to uncorrected CHLa values.

Table 2.2: Rio Bosque Wetlands sampling year, frequency or month of collection, type of data collected and whether samples were collected pre or post increase in water availability. Samples were collected June-August except in the case of those only sampled once.

Year	Sampling frequency or month	Habitats Sampled	Pre or Post water increase
2014	Weekly	channels	Pre increase
2016	Bi-monthly	ponds	Post increase
2017	Weekly	channels and ponds	Post increase
2018	June	ponds	Post increase
2019	July	ponds	Post increase

Data Analysis

A Principal Component Analysis (PCA) was used to describe underlying gradients in the environmental data. All environmental data, including physicochemical properties and algal biomass were entered into the analysis. The PCA analysis was conducted using the "princomp" function and data were transformed and standardized as required, to approximate a normal distribution (McCune and Grace 2002). Graphing of the PCA was performed with the "factoextra" package. Non-metric Multi-dimensional Scaling (NMDS) and Simpson Diversity Indices were calculated using the "vegan" package. An Analysis of Similarities (ANOSIM) based on abundance data was performed in order to identify significant differences in community composition between years sampled. ANOVAs and post-hoc Tukey-Kramer analyses were used to compare most water quality parameters, NMDS scores, to determine any differences in years sampled. Water quality parameters and macroinvertebrate abundances that could not be normalized were compared using Kruskal-Wallis rank sum test and post-hoc analysis with the "pgirmess" package. Kruskal-Wallis rank sum tests were also used to compare functional feeding group, active/passive disperser and macroinvertebrate abundances among sampling times and site types. Pearson correlation coefficients were determined to relate water physiochemical parameters and PCA scores with NMDS scores. Spearman correlation coefficients (r_s) were used to relate water physiochemical parameters and macroinvertebrate metrics that could not be normalized. Normality of residuals was confirmed for all regression analyses. All statistical analysis and graphing were performed in R (Version 4.1.2) (R Core Team 2021).

2.3 RESULTS

The PCA yielded two dimensions which explained 54% of the variation in the environmental data. PCA 1 accounted for 34.1% of the variability and PCA2 accounted for 20.3%

(Figure 2.3). PCA 1 contrasted channel sites visited in 2014, to pond sites visited in 2016, 2018 and 2019. Both ponds and channels were visited in 2017 which is likely why this year was plot overlapping 2014 and 2016. For this axis, nutrients such as NO_3^- and TP were the greatest drivers of variance, indicating higher levels of nutrients in sites sampled after the increase in water availability (2016-2019). The second strongest driver for PCA1 was conductivity, this time indicating higher levels in 2014, before the increase in water availability. Phytoplankton and pH were other strong drivers along PCA1 that were higher in 2014 compared to other years. Further analysis revealed a significant relationship between these two variables indicating as pH levels decreased over time, so did the concentration of phytoplankton ($r^2 = 0.22$, $p = 0.0003$). PCA2 was primarily driven by dissolved oxygen and temperature; there was no clear differences among years or sites along this axis.

Table 2.3: Mean and standard deviation of water physio-chemical variables for areas sampled in the Rio Bosque Wetlands during the summer months before (2014) and after (2016-2019) the increase in water availability. Letters indicate statistically significant differences ($p<0.05$; Kruskal-Wallis rank sum test and post-hoc analysis).

	Before Increase - Channels	**After Increase - Channels**	**After Increase - Ponds**
Conductivity (mS/cm)	2.41 (0.25) [a]	2.05 (0.30) [ab]	1.90 (0.26) [b]
Dissolved Oxygen (mg/L)	6.74 (5.02)	3.21 (1.36)	7.54 (16.57)
Temperature °C	23.79 (3.02)	23.86 (4.40)	24.38 (4.03)
Turbidity (NTU)	6.33 (6.56) [a]	9.42 (8.10) [ab]	25.07 (31.01) [b]
pH	7.86 (0.59) [a]	7.14 (0.35) [ab]	7.28 (0.63) [b]
NO_3^- (ppm)	0.13 (0.17) [a]	4.51 (3.14) [b]	5.39 (1.85) [b]
Total phosphorus (ppm)	0.56 (0.43) [a]	1.88 (0.69) [b]	2.61 (0.90) [b]
Periphyton CHLa (µg cm^{-2})	0.006 (0.004) [a]	0.015 (0.001) [b]	0.004 (0.006) [a]
Phytoplankton CHLa (µg L^{-1})	38.64 (38.5) [a]	25.04 (9.39) [ab]	18.09 (34.20) [b]

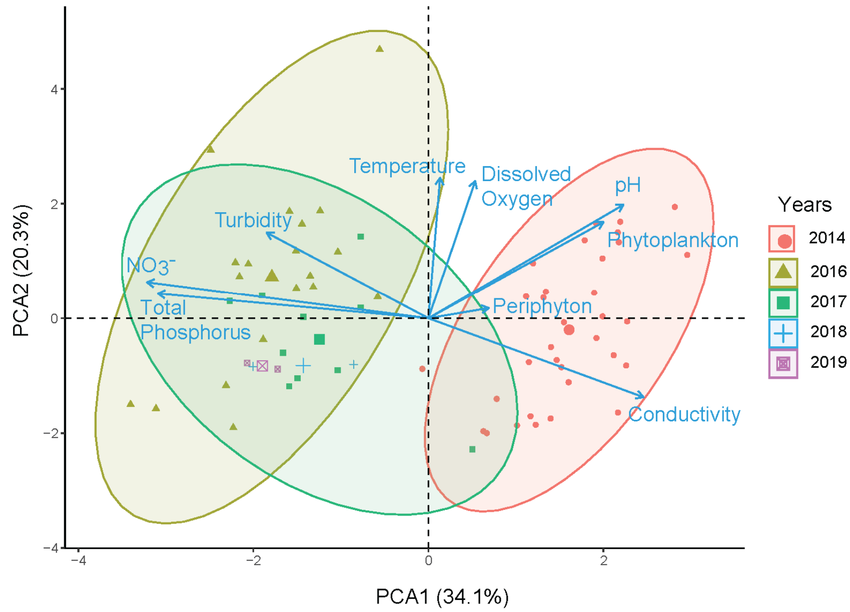

Figure 2.3: Plot of PCA scores of environmental data collected from the Rio Bosque Wetlands with environmental vectors, where longer arrows indicate stronger correlations with the axis scores, and points grouped by sampling year.

AVOVAs and post-hoc tests further confirmed the variation in water quality among years sampled. Nutrient levels (NO_3^-, TP) increased significantly in both the ponds and the channels over time (Figure 2.4, Table 2.3). Several variables did not change in the channels over time but were different in the newly flooded ponds relative to the channels. More specifically, both pH and conductivity levels were significantly lower in the ponds relative to the channels in 2014 (Figure 4, Table 3). We see a similar trend with phytoplankton, decreasing from an average of 38.6 µg L^{-1} in 2014 to 18.09 µg L^{-1} in ponds post water increase (Table 2.3). Conversely, turbidity was significantly higher in the ponds (25 NTU) as compared to the channels in 2014 (6 NTU). Finally, periphyton levels on average significantly increased within the channels (0.015 µg cm^{-2}), where they were higher than both the ponds (0.004 µg cm^{-2}) and the pre-water channel levels (0.006 µg cm^{-2}). Other parameters such as dissolved oxygen, and temperature fluctuated over the years but did not display any patterns related to the increase in water or site type.

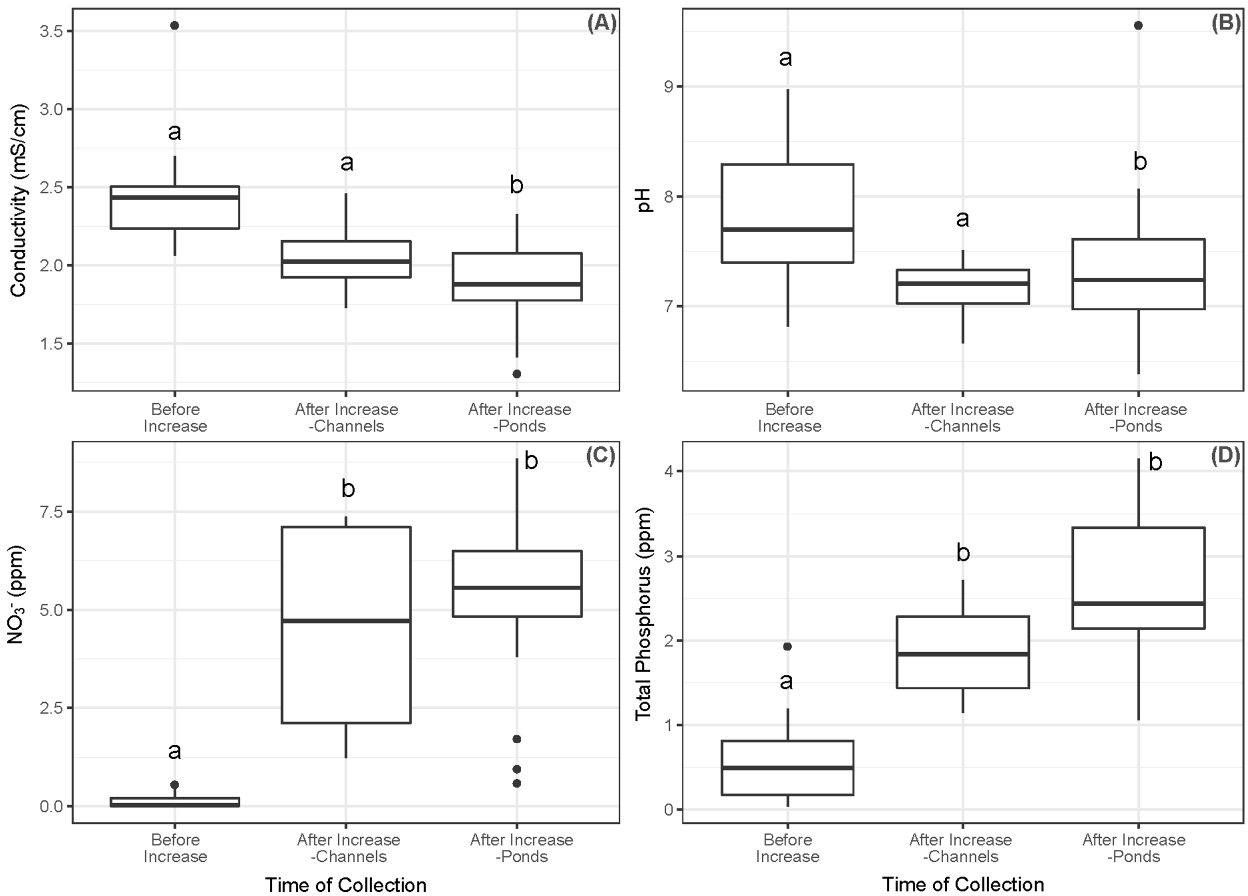

Figure 2.4: Mean conductivity (A) pH (B), NO_3^- (C) and total phosphorus (D) for water samples collected from the Rio Bosque Wetlands during the summer months of 2014, 2016, 2017, 2018 and 2019. Lowercase letters show significant differences among years as indicated by ANOVA, Tukey-Kramer and Kruskal-Wallis rank sum analyses.

The NMDS plot revealed a marked difference along the NMDS1 axis in macroinvertebrate community composition among the years sampled, specifically between samples collected before (2014) and after (2016-2019) increase in water availability (Figure 2.5). Samples from 2014 had significantly lower NMDS1 scores than those sampled in 2016 and 2017 (ANOVA, $p < 0.0001$). Coleopterans were obviously associated with the negative end of NMDS1, while several taxa, including Gastropods, Ostracods, Zygoptera were found at the opposite end of the axis. There was no significant difference in years or sites along the NMDS2 axis; though, Dalyellidae and Amphipods varied along this axis.

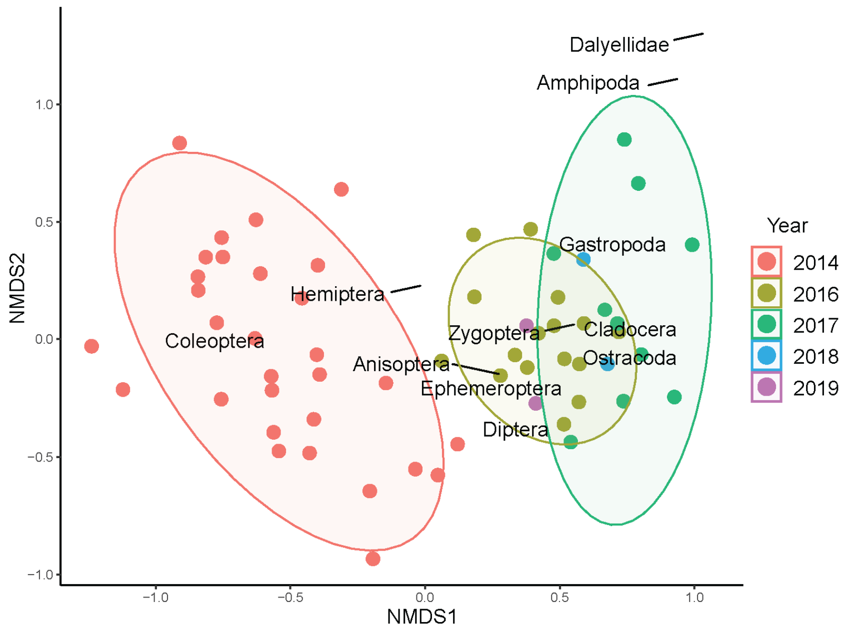

Figure 2.5: NMDS plot of macroinvertebrate taxa abundance data sampled before (2014) and after (2016-2019) the increase in water availability at the Rio Bosque Wetlands. NMDS stress = 0.16. ANOSIM R = 0.51, p-value = 0.0001.

NMDS1 was significantly associated with PCA1, conductivity (r^2 = 0.46, p = 1.152e-08), TP (r^2 = 0.53, p = 4.67154e-08) and NO_3^- (r^2 = 0.69, p = 7.149e-15) (Figure 2.6). These relationships reflected higher conductivity and lower nutrient levels within samples collected in 2014 compared to other years. None of the water quality parameters measured were correlated with NMDS2 axes scores (Table 2.4). However, there was a significant correlation with Julian date indicating samples collected earlier in the season were found towards the negative portion of NMDS2 and samples collected later in the season along the positive end (r = 0.30, p = 0.03).

Table 2.4: Correlation coefficients (r) of water physiochemical parameters and PCA scores with NMDS scores from areas samples in the Rio Bosque Wetlands during the summer months of 2014, 2016, 2017, 2018 and 2019. Significance: ***p<0.0001, **p<0.01, *p<0.05

	NMDS1	**NMDS2**
PCA1	-0.77***	-0.07
PCA2	0.26	-0.03
Conductivity (mS/cm)	-0.68***	-0.05
Dissolved Oxygen (mg/L)	-0.09	-0.16
Temperature °C	0.24	-0.11
Turbidity (NTU)	0.40	0.17
pH	-0.33	-0.08
NO_3^- (ppm)	0.83***	0.05
Total phosphorus (ppm)	0.73***	0.04
Periphyton CHLa (µg cm^{-2})	0.03	-0.17
Phytoplankton CHLa (µg L^{-1})	-0.38	0.14

Using the coded abundances, approximate percent abundance was calculated for each sample and grouped by pre-and post-increase in water availability. Post group abundances were further divided by sampling area: channels and ponds. (Figure 2.7). From the NMDS and percent abundances, we can see that the channels in 2014 were dominated by Coleopterans. In subsequent years, there were significantly lower relative abundances of Coleopterans in both the ponds and the channels (Table 2.5). The years after the change in water availability were significantly more abundant in Gastropoda at both site types, increasing from about 1% up to 17% abundance in the channels. There was also a change in Amphipoda abundances, however, this was limited to the channels (+27%) and was not observed in the ponds (+1.9%). Ostracoda (15.8%) and Zygoptera (17.4%) populations established in the ponds were found to be significantly greater than observed in the channels in 2014; while also increasing in abundance in the channels post water increase, this change was not significant. Finally, some taxa such as Anisoptera, Diptera and Ephemeroptera decreased somewhat in abundance within the channels, while being relatively high in the ponds, leading to significant differences between these two habitats post water increase.

Table 2.5: Means and standard error of macroinvertebrate relative abundances of habitats sampled before (2014) and after (2016-2019) the increase in water availability at the Rio Bosque Wetlands. Letters indicate statistically significant differences ($p<0.05$; Kruskal-Wallis rank sum test and post-hoc analysis)

	Before Increase - Channels	After Increase - Channels	After Increase - Ponds
% Amphipoda	0.0 [a]	27.0 (13.5) [b]	1.9 (5.1) [a]
% Anisoptera	9.2 (10.6) [ab]	1.7 (2.6) [a]	10.3(5.8) [b]
% Cladocera	0.4 (2.0)	0.0	1.7 (4.7)
% Coleoptera	40.9 (18.9) [a]	2.2 (3.5) [b]	6.1 (5.6) [b]
% Dalyellidae	0.4 (2.0)	8.2 (10.6)	0.0
% Diptera	14.4 (16.7) [ab]	5.3 (6.4) [a]	16.7 (6.4) [b]
% Ephemeroptera	9.4 (11.5) [ab]	4.0 (3.3) [a]	12.1 (7.9) [b]
% Gastropoda	1.6 (4.9) [a]	17.4 (10.7) [b]	7.8 (7.0) [b]
% Hemiptera	12.6 (16.3)	4.9 (4.7)	8.0 (6.6)
% Ostracoda	4.3 (8.6) [a]	15.8 (12.6) [ab]	17.9 (12.1) [b]
% Zygoptera	6.8 (9.0) [a]	13.5 (6.8) [ab]	17.4 (5.5) [b]

A comparison of functional feeding group distribution also revealed significant differences due to the change in water regime but also between habitat type (Figure 2.8). There was a significant increase in scrapers both within the channels (17%) and ponds (8%) when compared with the channels in 2014 (2%). Conversely, there was a change in predators with lower abundances in the ponds (41%) and channels (30%) with the increase in water (2014: 70%). Populations of filterers within the ponds (20%), was significantly greater than the channels before (4%) or after the increase in (16%) water. There was no significant difference in collectors-gatherers with the increase in water or between site type (2014 channels: 23%; after increase channels: 36%; after increase ponds: 30%).

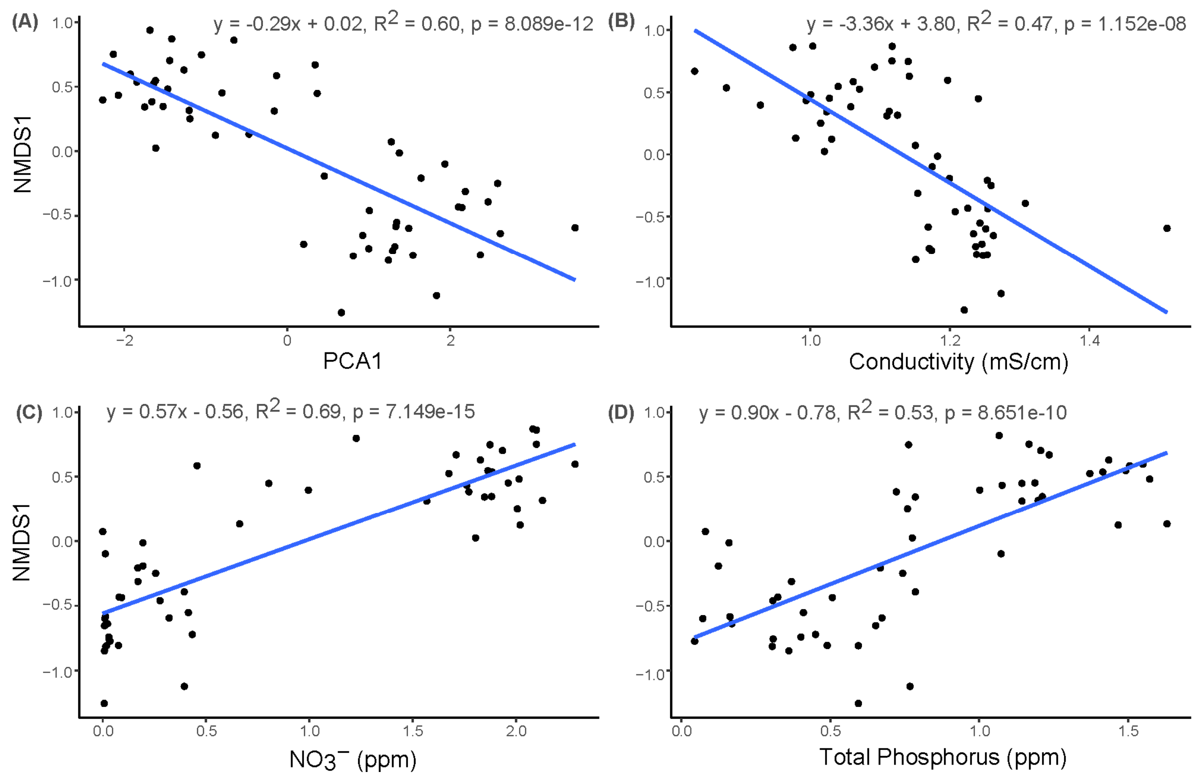

Figure 2.6: Regression plots depicting significant associations ($p<0.05$) of NMDS1 scores with PCA1 scores (A), Conductivity (B), NO_3^- (C) total phosphorus (D) for samples collected from the Rio Bosque Wetlands during the summer months of 2014, 2016, 2017, 2018 and 2019.

The abundance of the functional feeding groups was largely correlated with levels of nutrients within the water, especially for filterers (NO_3^-: r_s=0.68, p <0.01; TP: r_s=0.67, p <0.01) and predators (NO_3^-: r_s=-0.68, p <0.01; TP: r_s=-0.48, p <0.01). Increase levels of conductivity were negatively correlated with scraper abundance (r_s = 0.68, p=0.03). Collector-gatherer abundances were not significantly correlated with any water quality variable, but there were significant patterns related with Julian date (r_s=-0.70, p <0.01).

There were significant differences in composition of active and passive disperser taxa among both time of collection and site types. In 2014, communities had significantly more active disperser taxa (93%) than both the channels (32%) and ponds (70%) of subsequent years. Post water increase habitats were also significantly different from each other with the ponds having significantly more active dispersers and few passive dispersers than the channels (Figure 2.9).

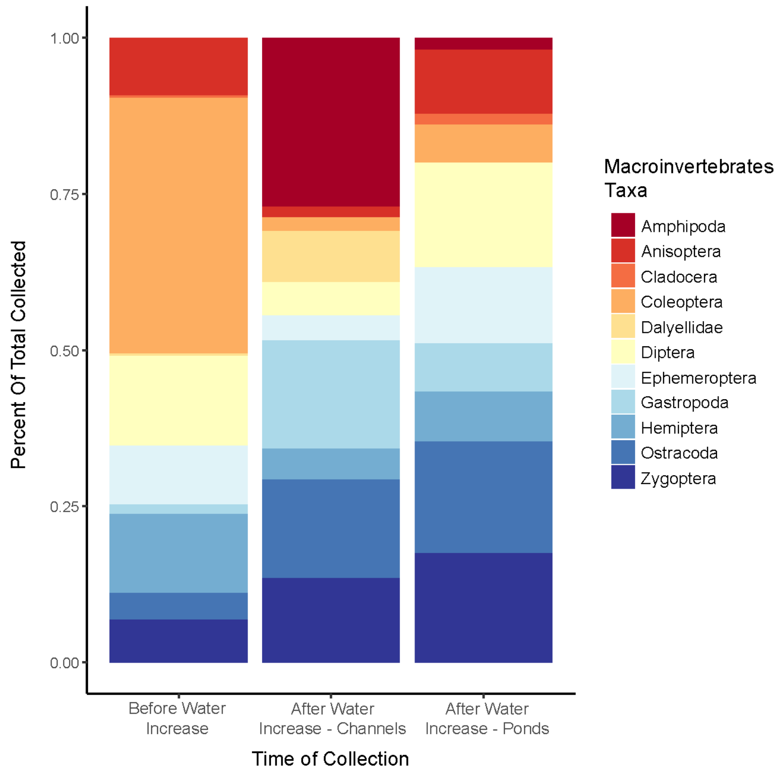

Figure 2.7: Relative abundances of macroinvertebrate taxa from the Rio Bosque Wetlands grouped by time and location of collection. Before water increase samples were collected in 2014 from channels within the park; after water increase samples were collected from 2016-2019 from channels and ponds.

2.4 Discussion

Between 2014 and 2016, the Rio Bosque Wetlands began receiving perennial water deliveries from the Roberto Bustamante wastewater treatment plant. This meant that, for the first time, the wetlands would be receiving significant water flow during spring and summer growing seasons. We observed significant changes in water quality and macroinvertebrate community composition in the park over this time, likely due to a combination of changes in the relative contribution of water sources, groundwater vs effluent, as well as the substantial growth of aquatic plants in the wetlands.

The biggest change in water quality parameters that was observed was an increase in nutrient concentrations after the changes in water availability. Since the park previously received mostly groundwater to the channels during the growing season, the increase in effluent water delivery led to the significant increase in NO_3^- and TP within the ponds, as it is known to be high in nutrients (Hamdhani, Eppehimer, and Bogan 2020; Zhuang et al. 2019). Moreover, the ponds also experienced a significant decline in conductivity, which can be an indicator of salinity levels. Groundwater, especially in the El Paso area, is known to have high levels of salinity, thus the reduced relative contribution of groundwater and increased flow from the effluent likely flushed out or diluted the salts within the ponds (Hiebing et al. 2018; Jolly, McEwan, and Holland 2008). Since the channels still receive groundwater, it is unsurprising that conductivity levels were not significantly different from 2014. After 2014, the pH levels of the water began to decrease in the ponds and the abundance of phytoplankton along with it; these two factors are likely linked to dissolved inorganic carbon (DIC). DIC is required for photosynthesis and can sometime be a limiting factor for phytoplankton populations during and after algae blooms (Hein 1997). As the phytoplankton uptake DIC, the acid-buffering capacity is reduced, thus leading to a decrease in the pH of the water (Alam et al. 2001; Carlson et al. 2001). The significantly lower populations of both phytoplankton and periphyton withing the ponds may also be linked to reduction in sunlight caused by overgrowth of *Typha* spp., *Polygonum* spp. *and Lemna* spp.

In 2014, the park's channels were primarily filled with groundwater year-round; meaning that, by summer, the accumulated salt levels were likely very high. During this time these areas were dominated by coleopterans, of which some species are known to be tolerant of low water quality and high salinity levels within freshwater ecosystems (Colombetti et al. 2020; Garrido and Munilla 2008; Lancaster and Scudder 1987; S. Sharma, Sharma, and Pir 2019). Over time, the relative abundance of Amphipoda, which are also adapted to high levels of salinity (Cuthbert et al. 2020; Dinger et al. 2005; Gervasio et al. 2004), significantly increased within the channels. However, work in regional wetlands did not show that these taxa are representative of saline wetlands (Chapter 1), and thus these trends may be limited to more channelized environments. While dipterans have not been shown to be directly affected by nutrient levels (Gresens et al. 2007), increased salinity levels have been shown to delay emergence to adulthood by 15-88% (Hassell, Kefford, and Nugegoda 2006). This, coupled with the growing population of mosquito fish we observed predominantly found in the channels may account for the large differences in Diptera abundances.

Though mayflies are usually highly sensitive to increased nutrients and turbidity (Cortelezzi et al. 2015; Stewart and Downing 2008) there was greater relative abundance of Ephemeroptera within the ponds as compared to the channels. The difference may related to the increased conductivity related to groundwater found within the channels as mayflies have also been found to be sensitive to salinity (Kefford 2019). Though only significantly increasing in the ponds over time, there were greater relative abundances of Ostracoda within both site types after 2014 (Figure 2.7).

Ostracods and gastropods increased throughout the park over time, most notably in the wetland cells. Similarly there was an association between ostracods and wastewater fed wetlands (Chapter 1). Ostracod populations have been found to be highly affected by many factors, including salinity, temperature, pH, dissolved oxygen (Ruiz et al. 2013), In the Rio Bosque Wetlands, abundances were significantly negatively correlated with pH ($r_s = -0.55$ $p = 0.00002$)

and conductivity (r_s = -0.40, p = 0.003), but positively correlated with NO_3^- (r_s = 0.64, p = <0.0001) and TP (r_s= 0.59, p = <0.0001). While many other factors are likely contributing to the change in abundance, ostracods seemed to thrive in the less saline, high nutrient effluent water. The overall difference in nutrient levels with the increase in effluent is also likely what led to the differences in gastropod abundances in both the ponds and the channels. Since gastropods make up our entire scraper population, we see this difference also reflected in the functional feeding group composition. Since we saw the greatest difference in gastropod abundance within the channels, we can also attribute this change to the significantly higher amounts of periphyton within this habitat (Saikia, Ray, and Mukherjee 2011).

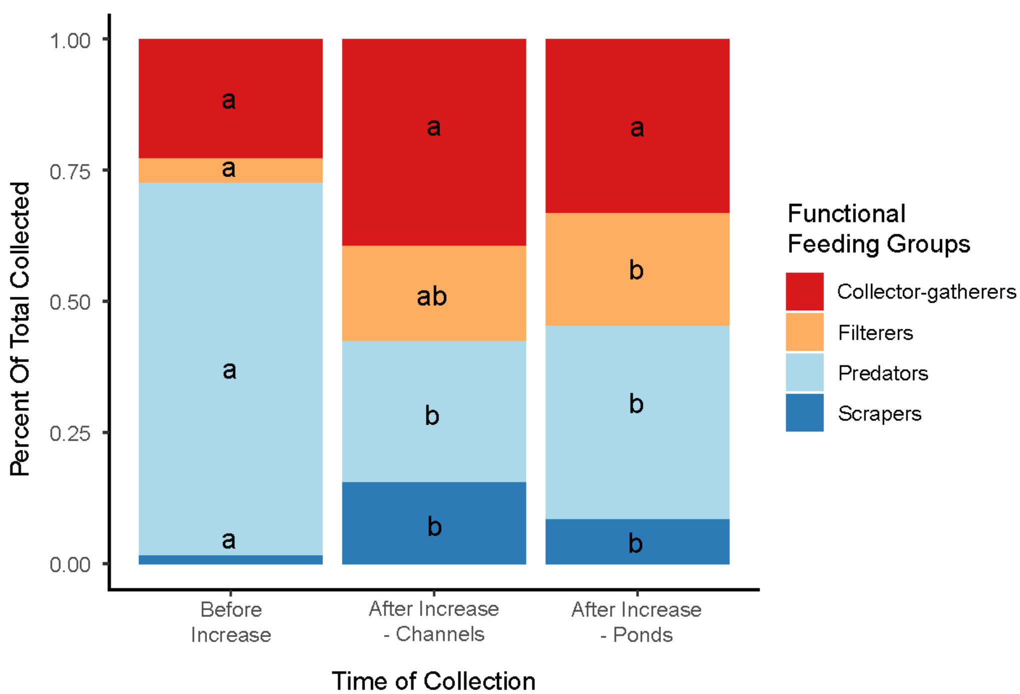

Figure 2.8: Relative abundances of functional feeding groups from the Rio Bosque Wetlands grouped by time and location of collection. Before water increase samples were collected in 2014 from channels within the park; after water increase samples were collected from 2016-2019 from channels and ponds. Letters indicate statistically significant differences (p<0.05; Kruskal-Wallis rank sum test and post-hoc analysis) among times of collection.

The large increase in amphipods in the channels over time may be explained by changes in predator abundances. Coleopterans decreased through time in the channels, where amphipods were highly successful. Changes in the location of predatory larval dragonflies (Anisoptera), which declined in the channels and were relatively high in the ponds, may also help explain the large change. Overall, the ponds had higher relative abundances of these predator taxa that are known feed on amphipods (Mikolajewski et al. 2010; Wellborn, Skelly, and Werner 1996).

Along with changes in overall functional feeding group ratios after 2014, there were also differences in relative abundance of taxa within groups, especially between habitat types. For example, coleopterans made up 60% of predators in 2014. In subsequent years there was this shift in dominance to Zygopterans in both the ponds (42%) and channels (48%). Overall, this group was less dominated by one taxa with the increase in other predator taxa. The same can be said about composition of collector-gatherers. While there was no significant difference among time of collection or habitat type, we know that post-increase collector-gatherer communities in the channels were dominated by Amphipoda (73%) whereas the ponds had significantly higher relative abundances of dipterans (56%) and ephemeropterans (36%). The trend of increase in filterers within the channels is similar to patterns seen with other, more established effluent wetlands of the desert southwest. The changes in the relative abundance of taxa within the functional feeding groups may also be an indication of increased ecosystem resilience though functional redundancy (Feit et al. 2019).

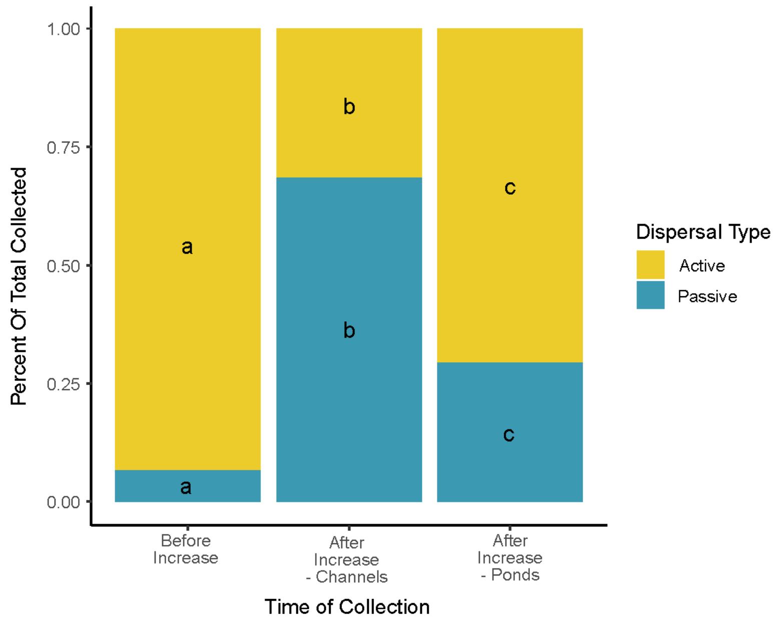

Figure 2.9: Relative abundances of active and passive disperser taxa from the Rio Bosque Wetlands grouped by time and location of collection. Before water increase samples were collected in 2014 from channels within the park; after water increase samples were collected from 2016-2019 from channels and ponds. Letters indicate statistically significant differences among times of collection ($p<0.05$; Kruskal-Wallis rank sum test and post-hoc analysis)

The increase of some of the functional feeding groups, including filterers, as well as passive dispersers in the channels and ponds over time may be an indication of less disturbance, particularly drying events, which can reduce habitat for taxa with weak overland dispersal abilities (Washko and Bogan 2019). The channels have consistently held water for a longer period of time than the ponds, allowing passive dispersers with longer colonization times to increase in abundance (Baber et al. 2004; Gleason and Rooney 2018; Moraes et al. 2014). With more constant water deliveries to the ponds, we should expect to also see populations of passive dispersers increasing over time.

Though the use of macroinvertebrates as indicators of restoration has been shown to be inconclusive when assessing specific metrics, studies suggest the presence of taxa that have longer life-cycle durations and non-insects with limited dispersal abilities could indicate advanced phases of succession (Meyer and Whiles 2008; Marchetti et al. 2010; Ruhí et al. 2012). Though the increase in water availability led to noticeable changes within the ponds and channels of the Rio Bosque Wetlands, there were significant increases in taxa that fit these criteria including Amphipoda, Gastropoda and Ostracoda in both these sampling areas. With the increase in effluent during the summer months, the Rio Bosque Wetlands saw fewer drying periods especially within the ponds. This likely broadly shaped the communities within the park with increased abundance of taxa with longer life-cycles such as amphipods (Esposito 2012; Porst et al. 2012; Schriever et al. 2015; Waterkeyn et al. 2008).

The Rio Bosque Wetlands is displaying succession patterns similar to those of other, more established desert wetlands flooded with effluent water (Chapter 1). Within these other sites, there were trends towards communities with increased filter feeder populations (i.e.,. ostracods); which are growing in abundance within the Rio Bosque Wetlands. Since this has been largely attributed to the nutrient enrichment from effluent water, it is suggested that the increased mixing of groundwater and effluent at the Rio Bosque Wetlands to dilute the amount of not only salinity also nutrients. With mixed water in both channels and ponds, we would expect to see an increase in both salinity and nutrient sensitive taxa. The change in the macrophyte community may be another factor leading to differences between the ponds and channels, specifically with phytoplankton levels. While wetland plant coverage fluctuated in 2016, subsequent years experienced 100% coverage of *Typha* spp., *Polygonum* spp. and *Lemna* spp. in some areas, leaving no open water. On average, relative cover of plants in the wetland cells ranged from an average of 40-50% in 2014 to 57-100% in 2016 and 2017 (Lougheed, unpublished data). The overgrowth of wetland plants in the Rio Bosque Wetlands is an issue that is currently being dealt with through manual removal to increase the amount of open water available for waterfowl. Over time, this may help diversify microhabitats within the ponds leading to further diversification of taxa.

Overall, the increased water during the growing season, helped the Rio Bosque Wetlands establish more permanent bodies of water, especially within the ponds. This change affected macroinvertebrate communities by allowing taxa with limited dispersal abilities and longer lifecycle durations time to build larger populations (Esposito 2012; Ruhí et al. 2012; Schriever et al. 2015). These results may bring additional insight as recently studies have highlighted the use of functional diversity as an indicator of restoration (Coccia et al. 2021; Feit et al. 2019). Differences in assemblages within the park were also heavily influenced by the differing water sources within the ponds and channels. Many studies have determined that it takes 10 years for macroinvertebrate communities of constructed or restored wetlands to resemble those of natural wetlands (Marchetti et al. 2010; Ruhí et al. 2012). For this reason, it would be imperative to sample at the 10-year mark (2025) to assess the community composition of the wetlands in reference to similar natural sites as a benchmark for restoration success. Since there was a temporal correlation with our NMDS scores and collector-gatherer abundances, it is suggested that samples be taken at least monthly throughout the growing season to ensure a more accurate representation of the macroinvertebrate community. Additionally, it is suggested that monitoring continue to identify any changes in assemblages and water quality with the addition of agricultural irrigation water inflow to wetlands. Though the macroinvertebrate community of this and other created sites, may never resemble that of a natural site due to the use of effluent water, these systems provide much needed habitat for aquatic flora and fauna within the desert landscape.

CHAPTER 3: EFFECTS OF A VIRTUAL CURE ON NON-SCIENCE MAJORS AT A COMMUNITY COLLEGE IN THE TIME OF COVID.

3.1 Introduction

Over the years, undergraduate research experiences (UREs) have been shown to increase science literacy, improve students' science identity and bridge the gap between research degrees and underrepresented populations (Hernandez et al. 2018; Olson 2012; P. Sadler and Sonnert 2016; T. D. Sadler et al. 2010; Vora et al. 2020). While having all students conduct research at some point in their academic careers is ideal, it is not often feasible (Desai et al. 2008). With limited and competitive URE positions available, the vast majority of students at four-year universities will not be able to obtain a research position; this number is even less at the community college level (Auchincloss et al. 2014; Bangera and Brownell 2014; Kloser et al. 2013; Weaver, Russell, and Wink 2008).

While we know CUREs allow educators to reach more students and provide them with opportunities that they would otherwise not have access to (Weaver, Russell, and Wink 2008) there may, however, still be a bias as to which students are able to participate. A growing number of universities and colleges have been implementing CUREs as a means of improving student engagement and ownership within science courses; however, this is mostly seen in courses offered to science majors (Ballen et al. 2017; Brownell et al. 2015; Glynn et al. 2011). Because of this, there has been a recent push to develop CUREs specifically for non-majors and their needs in an effort to improve the science literacy of all students in higher education (Ballen et al. 2017).

As previous work has demonstrated, there are significant differences between majors and non-majors regarding motivation and goals (Cook and Mulvihill 2008; Cotner, Thompson, and

Wright 2017; Knight and Smith 2010); understanding these differences is integral when developing a CURE specifically for non-majors. While non-science majors may not benefit as much from learning lab techniques, they may benefit from the improvement of science literacy skills and motivation that comes from conducting research in the classroom (Ballen et al. 2017; Dolan 2016). As science majors may be inclined to have more intrinsic motivation to perform well in science courses, non-majors, who are often required to take a science course as part of their degree plan, may not experience that same level of motivation. A CURE may be the solution as it uses active learning to engage, motivate and increase student performance in the classroom (Ballen et al. 2017; Bangera and Brownell 2014; Rodenbusch et al. 2016; Weaver, Russell, and Wink 2008). While CUREs for non-majors have been thought to be distinct from majors CUREs - in terms of learning objectives and goals – there is very little literature on the experiences of non-majors in CURE settings (Ballen et al. 2017). Therefore, it would be valuable to document non-majors' experiences to determine which features of the CURE benefitted them the most.

Another barrier facing underrepresented students is the lack of opportunity to develop their science identity. Science identity development occurs when students are able to align their perception of a STEM career with their own personal identity; this increases as students see others like themselves in these positions (P. Sadler and Sonnert 2016). The more students build on their science identities, the more comfortable they become in learning science concepts. A key component in developing a strong science identity is a sense of community and affiliation within the scientific community (Vincent-Ruz and Schunn 2008). This is where a CURE versus a research internship can help breakdown the sense of division between the students and the science community. Upon registration students are placed in the course where most everyone is new to research. There is less pressure placed on the students than if they were to conduct research in laboratory setting (Ballen et al. 2017; Bangera and Brownell 2014; Rodenbusch et al. 2016; Slovacek et al. 2012). This allows the students to grow and develop their science identity together. For many at the community college level, this may be the only opportunity they have to develop their science identity. In a study conducted by (Chemers et al. 2011), research experience promoted strong science identity which influenced students' decisions to remain in a STEM field.

CUREs and COVID

In 2020, the COVID-19 pandemic forced many college courses to discontinue in-person learning, in order to curb the spread of the virus. Institutions then had to turn to online and virtual learning to continue the courses already in progress. Online courses, which were usually reserved for remote learners, individuals with accessibility issues or those with very complicated schedules (Roddy et al. 2017), were now the norm for everyone. Even courses that were meant for students to engaged in hands on research activities were now forced into an online setting. This major change, however, challenged educators to develop unique and innovated virtual CURES (vCURE) that still engaged students and allowed for hands on activities in a safe manner (Corson et al. 2021; Majka, Raimondi, and Guenther 2020).

When adapting a CURE for a virtual setting, there are factors that must be taken into consideration that lead to differing applications than that of a non-CURE science course. A common method that was used with vCUREs was the implementation of synchronous meeting times with mentors and teams (Ashkanani et al. 2022; Bennett et al. 2021; Majka, Guenther, and Raimondi 2021; Martín et al. 2021), as one of the unforeseen struggles with vCUREs was the need for increased mentorship from the instructors (Majka, Raimondi, and Guenther 2020). However, (Fey, Theus, and Ramirez 2020), highlights this as a learning opportunity as many ecology and environmental research projects are conducted by remotely distributed teams.

With the current move to deliver more classroom content virtually, it is imperative that we also move to create more vCUREs for students as well. This will not only keep these opportunities available to students who must work virtually but will also give educators the chance to reach more students than they may not have otherwise.

Here we report the implementation and effects of a vCURE in an introduction to biology lab at El Paso Community College (El Paso, TX) with non-science majors during the Fall 2020 and Spring 2021 semesters. Usually, during in person biology or ecology courses, students have the opportunity to leave the classroom and engage in activities outside, many times this what students remember the most from these courses. For this reason, we wanted students to have the opportunity to be able to safely conduct research outside and thus we created an Urban Ecology themed CURE.

Course and Research Design

The Biology 1108 lab is a full semester lab course for non-science major students. During the Fall 2020 and Spring 2021 semesters we had a total of 48 students enrolled in the CURE course. The courses were taught at El Paso Community College (EPCC) following a 100% virtual curriculum.

Throughout the semester, the CURE course followed the Team Based Learning (TBL) model, with the students placed in permanent teams, completing lectures asynchronously, and any synchronous time spent on group activities (Michaelsen and Sweet 2008). The synchronous group meetings occurred once a week and lasted anywhere from 45 to 90 minutes. During these meetings groups would meet with and receive directions from the course instructors. Students were then left to work on group assignments together with Peer Leaders to assist them. Peer Leaders were undergraduate researchers, majoring in biology from UTEP, who were hired to assist with the course.

The 15-week semester was broken up into three phases of the project: the "Crash Course", the "Warm-Up" and the "CURE Project" (Table 3.1). The goal of this set-up was to have the students practice and build on the skills they learn in "Crash Course" multiple times throughout the semester to encourage skill retention. The break-up of the semester also allowed students to practice the project development system before having them develop their main CURE project. This was important for the non-science major students of this course who, previously, may not have had the opportunity to develop projects in the past.

While working on data collection for the "CURE Project", safety was at the forefront when developing possible projects for the students to work on. Since students were able to collect this data on their own, this eliminated the need for students to interact with anyone else face to face. This also served as a lesson in remote research collaboration which is becoming increasingly common in most sciences and especially in biological sciences (Hampton and Parker 2011). We also wanted to remain cognizant of student accessibility and comfort levels with public spaces, so students also had the option to collect air quality data from regional online databases. However, the group as a whole had to agree on one project for the semester (i.e., individuals from one group could not conduct differing projects). Each week group members worked on a specific part of the project (i.e., Introduction, methods, results etc.) then presented their completed projects as their final assignment for the semester.

Table 3.1: Course phase, time period during the 15-week semester and activity performed but students during that phase.

Phase	Week	Activity	Target Skills
"Crash Course"	Weeks 1-5	Students individually watch lecture videos covering a new science skill weekly. Students meet synchronously with group members to complete an assignment related to the weekly lecture.	• Scientific Method • Searching and reading primary literature • Statistical inference • Graphical inference • Data interpretation
"Warm-Up"	Weeks 6-7	Groups are given a data set and must develop an overall question and hypothesis, search for related literature, and choose best data analysis to answer their question. They then present their projects to audience of students, instructors, and peer leaders.	• Implementing science • Teamwork • Statistical inference • Graphical inference • Data interpretation • Connecting observations with questions and hypotheses • Science communication
"CURE Course"	Weeks 8-15	Student groups develop an Urban Ecology related project. They must come up with an overall question and hypothesis, develop methods for data collection, search for related literature, and choose best data analysis to answer their question. They then present their projects to audience of students, instructors and peer leaders.	• Implementing science • Teamwork • Reading primary literature • Methodology • Data Collection • Data interpretation • Statistical inference • Making natural history observations • Graphical inference • Connecting observations with questions and hypotheses • Science communication

3.2 Methods

The instruments used to measure the gains and outcomes from each course were the Test of Science Literacy Skills (TOSLS) (Gormally, Brickman, and Lut 2012) and Undergraduate Research Student Self-Assessment (URSSA) (Hunter et al. 2009). Questions regarding online and virtual learning were added to the end of the URSSA survey. Both surveys were conducted virtually using the QuestionPro online platform (QuestionPro Inc., 2014) and results were only accessible to the researchers. Student responses to the survey were anonymous to the researchers.

The URSSA survey was used to measure changes in students' attitudes and perceived gains within the science community. Because these are self-reported gains, the URSSA survey was not used as a direct assessment of individual ability (Hunter et al. 2009). The original survey asks students to rate their perceived gains in the following categories or units: (3) Thinking and Working Like a Scientist, (4) Personal Gains Related to Research Work, Skills, and (6) Attitudes and Behaviors (Weston and Laursen 2015) (Table 3.4). Additional Units (1,2,5,7 and 8) regarding perceived gains in scientific method, descriptive stats, experimental design, and scientific skills were added to the survey to collect data on more specific skills that were incorporated into the course. Questions regarding online learning were also added to the post-course surveys as way of gauging student attitudes towards online science courses. Mean, standard deviation and gains were calculated and compared for each URSSA Unit and individual question. The Wilcox One Sample t-test was used to compare the post-course survey scores to the means of the pre-course survey scores (York 2016). To measure the standardized effect size of the pre and post surveys, Hedge's g was calculated and reported in conjunction with the Wilcox One Sample t-test (Delacre et al. 2021). For this study, Hedge's g was chosen over Cohen's d due to the relatively small sample size of completed surveys (Korpershoek et al. 2016; Turan 2021). Hedge's g interpretation of effect size is as follows: small effect (<0.5), medium effect (0.5-0.8), large effect (>0.8)(Rosnow and Rosenthal 1991).

The TOSLS survey was used as was to measure gains in scientific skills consisting of 28 multiple choice questions testing 9 different skills related to major aspects of science literacy (Table 3.2)(Gormally, Brickman, and Lut 2012). The raw scores and percentage of correct answers were calculated by skill and combined as an overall score. Mean, standard deviation and gains were calculated and compared for each TOSLS Skill and the overall combined scores. Wilcox One Sample t-test used to compare the scores of the post-course responses to the pre-course survey means. Hedge's g was also used to measure the standardized effect size of the overall pre and post survey scores as well as by Skill.

Table 3.2: Skill number and description of tested skills on Test of Science Literacy Skills. (Gormally, Brickman, and Lut 2012)

#	Skill	Description
1	Identify a valid scientific argument	Recognize what qualifies as scientific evidence and when scientific evidence supports a hypothesis
2	Evaluate the validity of sources	Distinguish between types of sources; identify bias, authority, and reliability
3	Evaluate the use and misuse of scientific information	Recognize a valid and ethical scientific course of action and identify appropriate use of science by government, industry, and media that is free of bias and economic, and political pressure to make societal decisions
4	Understand elements of research design and how they impact scientific findings/conclusion	Identify strengths and weaknesses in research design related to bias, sample size, randomization, and experimental control
5	Create graphical representations of data	Identify the appropriate format for the graphical representation of data given particular type of data
6	Read and interpret graphical representations of data	Interpret data presented graphically to make a conclusion about study finding
7	Solve problems using quantitative skills, including probability and statistics	Calculate probabilities, percentages, and frequencies to draw a conclusion
8	Understand and interpret basic statistics	Understand the need for statistics to quantify uncertainty in data
9	Justify inferences, predictions, and conclusions based on quantitative data	Interpret data and critique experimental designs to evaluate hypotheses and recognize flaws in arguments

3.3 Results

Undergraduate Research Student Self-Assessment

In total, 73 URSSA responses were collected over the two semesters. This number is made up of 42 pre-course and 31 post-course surveys. Student groups during both semesters were made up of predominantly self-identifying Hispanic female students of varying majors and classifications (Table 3.3).

When broken down by category, the students recorded significant gains and medium to large effect sizes in all 7 units that were on the pre- and post-course survey. This excludes Unit 8 as it was only listed on the post-course URSSA survey. The unit that recorded to greatest gains and large effect size was Unit 2, which measured the student's confidence in experimental design skills (+1.1 scale points, Hedge's g = 1.1) (Table 3.4; Figure 3.1). This translates to 16.36% of students reporting their confidence in their abilities to conduct experimental design activities as "Very" or "Extremely" in the pre-course survey increasing to 48.63% in the post-course survey. The greatest gains in this unit were related to conducting statistical analyses such as regression analyses and paired t-tests (Table 5). We also saw large gains with medium effect size reported in Unit 7 (+0.9 scale points, Hedge's g = 0.7); which measured their confidence in general research related skills. Within this unit, students reported significant gains in their ability to plan data collection, interpret results, use scientific literature as well as develop theories ($p < 0.01$).

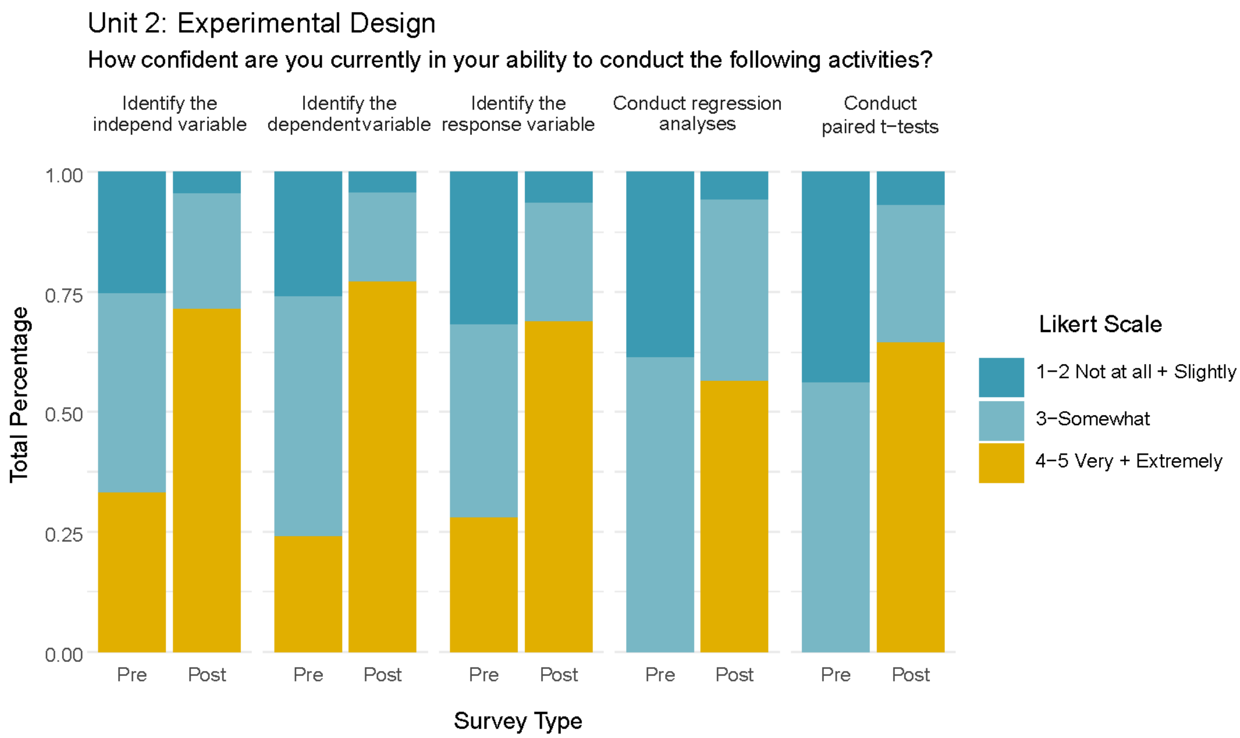

Figure 3.1: Likert scores of pre- and post-course URSSA survey questions for Unit 2: Experimental Design. Responses from the Fall 2020 and Spring 2021 semesters have been combined for an average of 1.15-point gain overall for this Unit. This unit displayed the greatest gains between both semesters of all units.

Of the others, Units 1, 3 and 5 made moderate overall gains in comparison (+ 0.5-0.7 points). While Units 1 and 5 recorded medium effect sizes between pre- and post-Likert scores, Unit 3 record a large effect. Within these units, the areas that students reported the most gains were in formulating and identifying limitations in research questions, preparing and giving scientific presentations and defending an argument when asked research-related questions (all gained 0.8-0.9 points, $p < 0.01$). The areas that students reported the least number of gains was in their confidence in working with computers (+ 0.3 points) and their time management (+ 0.4).

The Unit that showed the least amount of gain was Unit 4, in which students reported changes their own abilities in the science and research communities (Table 3.4). Within this Unit, the students reported an average of 0.5 points gain, though this still indicated a medium effect size

(Hedge's $g = 0.5$). This translates to a 30.48% of students reporting their confidence levels in their participation and personal abilities in science as "Very" or "Extremely" high in the pre survey and 57.60% in the post survey. Within this unit we saw the greatest gains in students reporting their comfort in discussing scientific concepts with others and their ability to do well in future science courses (+ 0.7 points, $p < 0.01$). Students did not report any significant gains in the ability to work independently (+0.1) and in developing patience with the slow pace of research (+0.2). We also saw little gain in Unit 6, in which students reported their attitudes and behaviors related to science and research (Table 3.4). This shows that only 7.95% of students "Agreed" or "Strongly Agreed" with statements related to how they see themselves in the scientific community; this percentage increased to 26.87% post-course. Students did however make significant increases in two of the four individual statements within this unit: "I have a strong sense of belonging to the community of scientists" and "I feel like I belong in the field of science" ($p < 0.05$).

Unit 8, which contained questions regarding the online format of the course, was only measured on the post-course surveys. The average score for this unit indicated that most students "Agreed" with the statements regarding the course in this unit (Table 3.5). The students "Agreed" that this course allowed them to interact with their peers (4.1 ± 0.9), however, they missed having in-person interactions with them (4.0 ± 1.2). Most students found the course to have the appropriate proportion of synchronous and asynchronous teaching (4.0 ± 1.0), clear online evaluation mechanisms (4.2 ± 1.0) and adequate preparation for evaluations (4.1 ± 0.9). Regarding the instructors of the course, most students found they made themselves available via email or other virtual mechanisms (4.2 ± 1.0). Overall, most students agreed that the course was well organized and easy to navigate (4.0 ± 1.1).

Table 3.4: Pre and Post-course URSSA Likert means, standard deviations, gain scores and effect size for the Fall and Spring CURE courses, grouped by question unit. Wilcox One Sample t-test significant differences between post course surveys scores and pre-course survey means are indicated by ***p<0.0001, **p<0.01, *p<0.05, + <0.10; without asterisks indicate non- significance. Hedge's g reported with 95% lower and upper 95% confidence intervals.

Unit 1: Scientific Method and Descriptive Stats	Pre-course	2.6 ± 1.2
	Post-course	3.4 ± 1.0***
	Gain	0.8
	Effect Size	0.7 [0.4, 1.0]
Unit 2: Experimental Design	Pre-course	2.4 ± 1.0
	Post-course	3.5 ± 1.0***
	Gain	1.1
	Effect Size	1.1 [0.9, 1.4]
Unit 3: Thinking and Working Like a Scientist	Pre-course	2.9 ± 1.0
	Post-course	3.7 ± 0.9***
	Gain	0.8
	Effect Size	0.8 [0.7, 1.0]
Unit 4: Personal Gains	Pre-course	3.0 ± 1.0
	Post-course	3.5 ± 1.0***
	Gain	0.5
	Effect Size	0.5 [0.4, 0.7]
Unit 5: Skills	Pre-course	3.0 ± 1.0
	Post-course	3.7 ± 1.0***
	Gain	0.7
	Effect Size	0.6 [0.5, 0.7]
Unit 6: Attitudes and Behaviors	Pre-course	2.3 ± 1.0
	Post-course	2.8 ± 1.2**
	Gain	0.5
	Effect Size	0.5 [0.3, 0.7]
Unit 7: Research Skills etc.	Pre-course	2.5 ± 0.9
	Post-course	3.4 ± 1.0***
	Gain	0.9
	Effect Size	0.8 [0.6, 0.8]
Unit 8: Post Online Course Ratings	Post-course	4.0 ± 1.0

Test on Science Literacy

A total of 71 TOSLS surveys were completed, including pre and post surveys from CURE courses for both semesters. Of the 71, 39 were completed pre-course surveys, while the other 32 were post course survey. When comparing the overall TOSLS scores, there was a significant difference and a medium effect size in the mean between the pre (43%) and post-course (52%) scores, increasing 9 percentage points ($p = 0.008$, Hedge's $g = 0.53$) (Figure 3.2). When broken down by Skill, however, the gains were highly variable, with only one skill making significant gains (Figure 3.3).

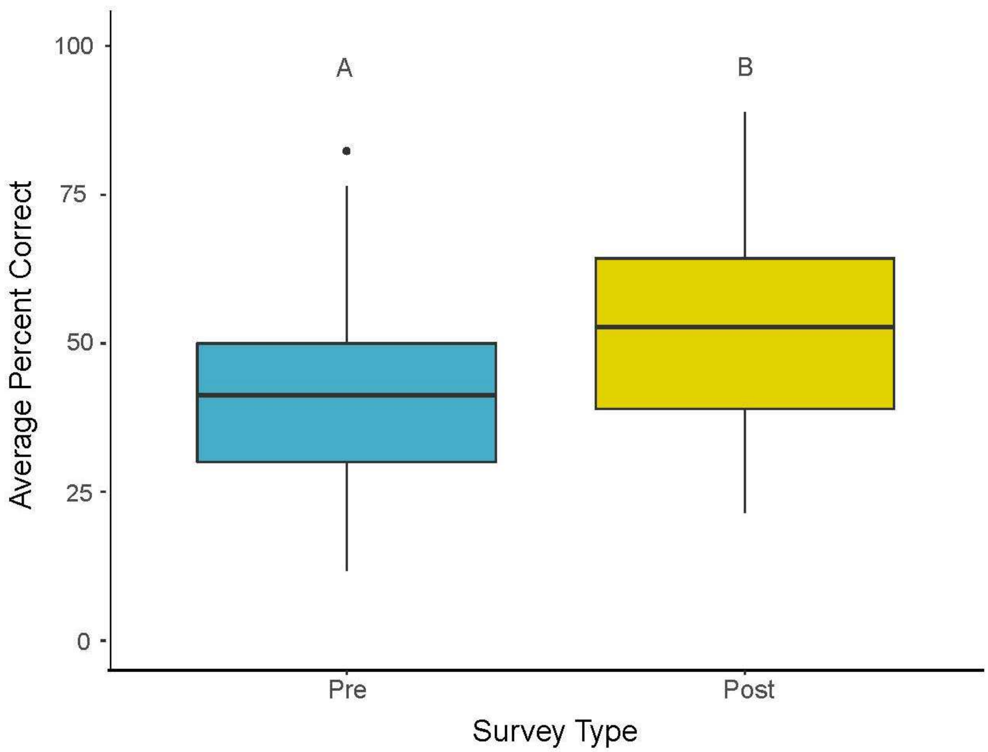

Figure 3.2: Combined average percent correct TOSLS scores of the Fall 2020 and Spring 2021 CURE courses at EPCC. Scores are grouped by pre- and post-course surveys. Letters indicate statistical differences between pre- and post-course TOSLS scores ($p=0.001$). Hedge's $g = 0.54$, 95% CI [0.15, 0.92]

Of the nine Skills on the TOSLS, Skill 2 was the only one in which students made significant gains ($p = 0.04$) (Table 3.6). This skill tested students' ability to distinguish between types of sources of information and identify bias and was comprised of 3 questions on the TOSLS. Within this Skill, we recorded a large effect size in which students increased the average scores by 14 percentage points between the pre- and post-course surveys (Hedge's g = 0.93) (Table 3.6).

Though not significant, other Skills had moderate gains, including Skill 3, 4, and 8, all of which increased their averages by 11 percentage points. While Skills 3 and 8 reported medium effect sizes, Skill 4 saw a large effect size between pre- and post-course scores (Table 3.6). Skill 3 was comprised of three questions on the survey and assessed students' ability to recognize valid and ethical course of action by various entities (e.g., government, industry, media). Skill 4 was made up of three questions and assessed students' ability to identify strengths and weaknesses in research design scenarios (e.g., bias, sample size, randomization, and experimental control). Skill 8 was comprised of 3 questions and assessed the students understanding of the need for statistics in the sciences.

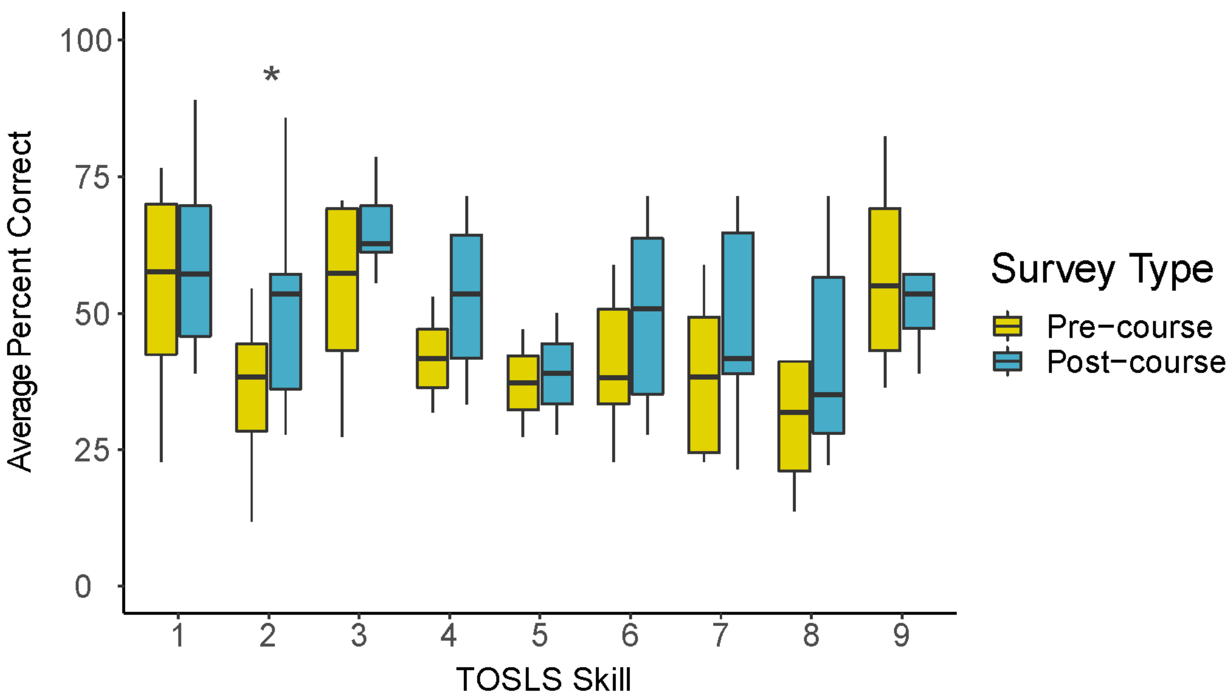

Figure 3.3: Average percent correct TOSLS scores of the Fall 2020 and Spring 2021 CURE courses at EPCC grouped by Skill. Wilcoxon rank sum significant difference between pre and post course surveys *p<0.05, without asterisks indicate non- significance

Table 3.6: Pre and Post-course TOSLS average scores out of 100 points ± standard deviation, gain and Hedge's g effect size for the Fall and Spring CURE courses, grouped by question unit. Wilcox One Sample t-test significant difference between post course surveys and pre-course survey means are indicated by *$p<0.05$; without asterisks indicate non- significance. Hedge's g reported with 95% lower and upper 95% confidence intervals.

TOSLS Skill		
Skill 1: Identify a valid scientific argument	Pre-course	54 ± 21
	Post-course	59 ± 19
	Gain	5
	Effect Size	0.25 [-0.9, 1.4]
Skill 2 - Evaluate the validity of sources	Pre-course	36 ± 12
	Post-course	50 ± 17 *
	Gain	14
	Effect Size	0.93 [-1.9, 0.02]
Skill 3 - Evaluate the use and misuse of scientific information	Pre-course	54 ± 18
	Post-course	65 ± 8
	Gain	11
	Effect Size	0.76 [-1.9, 0.5]
Skill 4 - Understand elements of research design and how they impact scientific findings/conclusion	Pre-course	42 ± 8
	Post-course	53 ± 15
	Gain	11
	Effect Size	0.83 [-2.1, 0.4]
Skill 5 - Create graphical representations of data	Pre-course	37 ± 14
	Post-course	39 ± 16
	Gain	2
	Effect Size	0.06 [-2.5, 2.3]
Skill 6 - Read and interpret graphical representations of data	Pre-course	40 ± 13
	Post-course	50 ± 17
	Gain	10
	Effect Size	0.62 [-1.7, 0.42]
Skill 7 - Solve problems using quantitative skills, including probability and statistics	Pre-course	38 ± 15
	Post-course	47 ± 20
	Gain	9
	Effect Size	0.48 [-1.7, 0.72]
Skill 8 - Understand and interpret basic statistics	Pre-course	30 ± 12
	Post-course	41 ± 21
	Gain	11
	Effect Size	0.56 [-1.8, 0.65]

Skill 9 - Justify inferences, predictions, and conclusions based on quantitative data	Pre-course	57 ± 20
	Post-course	50 ± 09
	Gain	-7
	Effect Size	- 0.3 [-1.2, 1.9]

The Skill that students gained the least in was Skill 1, which was comprised of three questions on the survey and tested students' ability to recognize valid scientific evidence and whether it supports a hypothesis. Students only gained 5 percentage points and had negligible effect size, starting at 54% and scoring 59% correct on the post-course survey (Hedge's g = 0.06).

There was one Skill that did not make any gains, but lost points between the pre and post course surveys. Skill 9 assessed the students' abilities to critique experimental designs and hypotheses as well as recognize flaws in an argument. It was comprised of 2 questions on the survey. Though the difference was not significant and had a small effect size, student scores decreased from 57% correct on the pre-course survey to 50% correct on the post-course survey (Hedge's g = -0.3).

3.4 DISCUSSION

During the COVID-19 pandemic, the worldwide switch to virtual learning led to an almost cease of hands-on activities and interactive CURE courses. This meant a decrease in peer-to-peer interaction, and meaningful application of research skills for students in science courses. By incorporating Team-based Learning and multiple opportunities for practice of research-based skills into our vCURE, students were able to significantly improve on their overall TOSLS scores and increase their confidence levels in several general science and research related activities. This remains consistent with other studies that have found early undergraduate research experiences to be beneficial to science majors and non-majors alike (Stanford et al. 2017).

For this group, it seems the repeated practice of activities related to experimental design had the biggest impact on students perceived abilities. After introducing these lessons during the "Crash Course", students were then able to apply these concepts (e.g., identifying variables, understanding statistical analysis) during the "Warm-Up" phase and again while working on their own projects during the "CURE Course". This remains consistent with the results from other studies evaluating the effects of ecology-based CURES on undergraduates (Kloser et al. 2013). By continually scaffolding these lessons through the semester, students are able to solidify their understanding and comfort level with these practices (Lin et al. 2012). Student also greatly increased their perceived abilities in performing other research related activities such as using scientific literature to guide research, data collection and explaining the results of a study; all of which are included in the "Crash Course" phase of the course. Data analysis was also a topic that was introduced during the "Crash Course" and practiced multiple times throughout the semester which is likely why students experienced increased confidence in this area.

At the end of the "Warm-Up" and "CURE Course" modules, student groups were required to prepare oral presentations over their practice and CURE projects. The groups then presented to the instructor, peer leaders and other students enrolled in the course. After the "Warm-Up" presentation, students received feedback from the audience members which was then used to improve on their final "CURE Course" presentation. We believe this feedback was integral to the students increased confidence in this area. Often student presentations are left to the end of a semester with little constructive feedback and sometimes no chance of using the feedback to improve. This is a key part of learning (Hattie and Timperley 2007; Wisniewski, Zierer, and Hattie 2020) that curriculum in higher education is often criticized for neglecting (Fielding, Dunleavy, and Langan 2010).

By implementing weekly synchronous virtual meetings, students were able to collaborate on weekly assignments and projects throughout the semester, which students seemed to appreciate. We believe this contributed to the improvement of their applied skills as the TBL model has been show to increase student performance (Anwar et al. 2012) as well as long term retention and critical thinking (McInerney and Fink 2003). This continuous teamwork, however, seemed to contribute to the small gains in students' ability to work independently (+ 0.1 points). As educators, we found the use of both synchronous and asynchronous methods allowed us to have direct interaction with the students in small group settings which is something that could have been lost in a virtual setting. Other studies have also indicated that students found a mix of synchronous and asynchronous methods to be the most effective when it came to distance learning during the pandemic (Chen, Kaczmarek, and Ohyama 2021).

While the course improved on students' confidence to "do" science, there was less of an effect on students' perceptions about themselves within the scientific community. Students made the least number of gains in Units related to how they can contribute to or view themselves in the overall scientific community. Being non-science majors, this is not entirely surprising as it has been documented that often times they are less likely to see science as personally relevant (Cotner, Thompson, and Wright 2017; Rannikmäe, Rannikmäe, and Holbrook 2006).

Of the TOSLS Skills, students greatly improved on their abilities to evaluate validity and distinguish between difference types of sources of information. As part of the "Crash Course", students learned about research articles, how they differ from other news outlets, and why other resources may not be valid sources of information. This proves to be a beneficial lesson as other studies have shown that, comparted to science majors, non-science undergraduates are less likely to be able to engage and critique news reports they read from various sources (Lin, 2014). Recent events, such as the COVID-19 pandemic, have highlighted the importance of information literacy to such an extent that courses have since been created and implemented to address this issue in undergraduate students (Scheibenzuber, Hofer, and Nistor 2021). This part of the course has shown to be effective and useful as information literacy has quickly become a common learning outcome for undergraduate students of all disciplines (Fosnacht 2020).

Students also greatly improved on their overall ability to understand and identify strengths and weaknesses in different components of research design related to bias, sample size and experimental control. By working with students in small group settings we were able guide them through their experimental design and teach these concepts as they related to their own projects. The increased sense of project ownership, which is a staple of CUREs, has also shown the to increase student understanding of these types of concepts (Cooper et al. 2019).

While student improved in 8 of the 9 skills on the TOSLS, there was decreased overall mean scores in Skill 9. This Skill tested the student's ability to evaluate hypotheses and recognize flaws based on graphed data, which was a component that was introduced during the "Crash Course" and students were allowed to practice this skill throughout the semester, just like the others. Like this Skill, students made negligible gains on their ability identify the appropriate format for the graphical representation of data (Skill 5). While students scored low on this skill, they did, however, gain 0.9 points in their self-reported ability to analyze data for patterns. They also experienced gains in their ability to read and interpret graphed data (Skill 6). While it is difficult to explain these conflicting results, it is important to note that reading and interpreting graphs is a complex activity which can lead to many cognitive errors (Glazer 2011) and many students struggle with choosing the correct graph to display data along with interpretation (Pérez-Echeverría, Postigo, and Marín 2018). It has also been suggested that the interpretation of different types of graphs needs to be explicitly taught and the skill has to be practiced consistently (Glazer, 2011). While its possible these students required more scaffolding to be able to improve on this skill, it may also be possible that other factors may have affected the averages within these units.

Though improvements were made, we believe the overall stress of the pandemic and lack of motivation due to digital fatigue and burnout may have affected the scores of the students (Meeter et al. 2020; Mheidly, Fares, and Fares 2020). Due to these factors, it is believed that some students may not have put in much effort (i.e., choosing answers at random) into the post-course survey. Students also understood that their grade would not be affected by the scores they received on the post-course TOSLS. While studies have determined that assigning a grade to the completion of the TOSLS does not significantly affect students efforts or scores (Segarra et al. 2018) we believe it might have increased student efforts at the end of the semester when digital fatigue was at its highest.

Overall, we can say that the vCURE had a positive effect on the students' learning outcomes and confidence levels. In open-ended questions, students were asked "What aspects of the course do you feel were the most beneficial to your future career plans?". We received answers such as: "Collaborating and communicating with a team for a project/group assignment.", "The most beneficial thing that I learned in this course was gathering data and working with teammates" and "Communication skills, time management". This shows that the students felt they improved on skills that are beneficial to them in their future, non-STEM, careers such as their communication and collaborative abilities. Though the course was required to take place in a virtual setting due to stay-at-home orders, continuing to offer vCURE courses increases accessibility for non-traditional student populations seeking research experience (Roddy et al. 2017). In the future this curriculum may be adapted to better meet the needs of STEM-majors and increase the availability of, much needed, ecology-based CURE courses.

Table 3.3: Number of students enrolled, completed surveys and demographic information collected from the URSSA pre- and post- course surveys grouped by semester.

Semester	Fall 2020		Spring 2021	
Total Students Enrolled	28		27	
Survey Type	Pre-test	Post-test	Pre-test	Post-test
Total Surveys Completed	22	17	20	14
Declared Major				
Psychology	5	4	1	0
Education	7	4	9	5
Communications	0	0	6	5
Criminal Justice	5	4	1	1
Business/Accounting	2	3	0	0
Dental Assistant	0	0	1	1
Liberal Arts	0	1	2	2
Not Disclosed	3	2	0	0
Declared Gender				
Male	3	4	7	4
Female	18	13	13	10
Not Disclosed	1	0	0	0
Declared Hispanic or Latinx				
Hispanic	22	17	19	13
Non-Hispanic	0	0	1	1
Declared Race				
White Non-Hispanic	0	0	2	1
Black Non-Hispanic	0	0	1	1
Hispanic	17	12	17	12
Asian American	0	0	0	0
Native American	1	1	0	0
Native Hawaiian or Pacific Islander	0	0	0	0
Two or more races	1	2	0	0
Mexican international	3	2	0	0
Other international	0	0	0	0
Unknown	0	0	0	0
Declared Classification				
Freshmen/rising sophomore	0	0	10	4
Sophomore/rising junior	17	12	8	9
Junior/rising senior	4	4	2	1
Senior	1	1	0	0
Graduate Student	0	0	0	0
Other	0	0	0	0

Table 3.5: Pre and Post-course URSSA Likert means, standard deviations for each individual question for the Fall and Spring CURE courses, grouped by question unit. Wilcox One Sample t-test significant difference between post course surveys scores and pre-course survey means are indicated by ***p<0.0001, **p<0.01, *p<0.05; without asterisks indicate non- significance.

Unit	Individual Questions	Pre-course	Post-Course
Unit 1	**How knowledgeable are you currently about the following areas/topics/concepts?**		
	1. The different steps of the Scientific Method (i.e., making observations, asking questions, developing hypothesis, test the hypothesis, etc.)	3.1 ± 1.1	3.7 ± 1.0*
	2. Descriptive statistics (i.e., mean, range, variance, standard deviation)	2.7 ± 1.0	3.2 ± 1.0***
	3. Inferential statistics (i.e., regression, t-test, ANOVA)	1.9 ± 1.2	3.6 ± 0.9***
Unit 2	**How confident are you currently in your ability to conduct the following activities?**		
	4. Identify the independent variable for an experiment	2.7 ± 1.0	3.6 ± 0.9**
	5. Identify the dependent variable for an experiment	2.6 ± 1.0	3.6 ± 0.9***
	6. Identify the response variable for an experiment	2.5 ± 1.0	3.5 ± 0.9**
	7. Conduct regression analyses	2.0 ± 1.0	3.3 ± 0.9***
	8. Conduct paired t-tests	1.8 ± 0.9	3.3 ± 1.0***
Unit 3	**How confident are you currently in your ability to conduct the following general research activities?**		
	9. Analyzing data for patterns	2.8 ± 0.9	3.8 ± 0.9***
	10. Figuring out the next step in a research project	2.9 ± 1.1	3.6 ± 0.9**
	11. Problem solving in general	3.2 ± 1.0	3.7 ± 0.9**
	12. Formulating a research question that could be answered with data	2.9 ± 1.0	3.7 ± 0.9**
	13. Identifying limitations of research methods and designs	2.6 ± 0.9	3.6 ± 0.9***
	14. Understanding the theory and concepts guiding my research project	2.9 ± 0.9	3.6 ± 0.9**
	15. Understanding the connections among scientific disciplines	2.6 ± 1.0	3.5 ± 0.9**
	16. Understanding the relevance of research to my coursework	3.02 ± 1.0	3.6 ± 0.9*
Unit 4	**How confident are you currently in your:**		
	17. Ability to contribute to science	2.6 ± 0.9	3.3 ± 0.9**
	18. Comfort in discussing scientific concepts with others	2.5 ± 0.9	3.3 ± 0.8**
	19. Ability to do well in future science courses.	2.7 ± 0.9	3.5 ± 0.9**
	20. Ability to work independently	3.6 ± 0.9	3.7 ± 0.9
	21. Developing patience with the slow pace of research	3.3 ± 0.9	3.5 ± 1.0
	22. Understanding what everyday research work is like	3.0 ± 0.9	3.7 ± 1.0**
	23. Taking greater care in conducting procedures in the lab or field	3.1 ± 1.0	3.7 ± 1.0*

Unit 5	**How confident are you currently in your ability to conduct the following general research activities?**		
	24. Writing scientific reports or papers	2.7 ± 1.0	3.2 ± 1.0*
	25. Making oral presentations	2.9 ± 1.2	3.7 ± 1.1**
	26. Defending an argument when asked research-related questions	2.7 ± 1.1	3.5 ± 1.1**
	27. Explaining my project to people outside my field	2.9 ± 0.9	3.5 ± 1.0***
	28. Preparing a scientific presentation	2.7 ± 0.9	3.7 ± 1.0**
	29. Keeping a detailed lab notebook.	3.1 ± 1.2	3.7 ± 1.1*
	30. Conducting Observations in the lab or field	2.9 ± 1.1	3.6 ± 1.0**
	31. Using statistics to analyze data	2.7 ± 1.0	3.4 ± 1.0**
	32. Calibrating instruments needed for measurement	2.6 ± 1.0	3.4 ± 1.0**
	33. Working with computers	3.7 ± 1.0	4.0 ± 1.1
	34. Understanding journal articles	2.9 ± 0.9	3.5 ± 1.0**
	35. Conducting database or internet searches	3.1 ± 0.9	3.7 ± 1.0**
	36. Managing my time	3.5 ± 1.0	3.9 ± 1.1*
	37. Critical or creative thinking	3.4 ± 0.9	4.0 ± 0.9**
	38. Working in a team setting	3.3 ± 1.0	4.1 ± 0.8**
Unit 6	**To what extent are the following statements true of you:**		
	39. I have a strong sense of belonging to the community of scientists.	2.2 ± 1.0	2.8 ± 1.1*
	40. I derive great personal satisfaction from working on a team that is doing important research.	3.1 ± 1.0	3.5 ± 1.0
	41. I have come to think of myself as a "scientist".	2.0 ± 0.8	2.4 ± 1.1*
	42. I feel like I belong in the field of science.	2.0 ± 0.8	2.6 ± 1.2*
Unit 7	**Indicate to what extent you are confident that you could complete the following tasks:**		
	43. Use technical science skills (use of tools, instruments, and/or techniques)	2.5 ± 0.8	3.3 ± 1.0**
	44. Generate a research question to answer.	2.7 ± 0.8	3.4 ± 1.1**
	45. Figure out what data/observations to collect and how to collect them.	2.4 ± 0.8	3.5 ± 1.0***
	46. Create explanations for the results of the study.	2.5 ± 0.9	3.5 ± 1.0***
	47. Use scientific literature and/or reports to guide research.	2.4 ± 0.9	3.4 ± 1.0***
	48. Develop theories (integrate and coordinate results from multiple studies).	2.3 ± 0.9	3.4 ± 1.1**
	49. Ask relevant questions.	2.7 ± 1.2	3.5 ± 1.0**
	50. Identify what is known and not known in a problem.	2.7 ± 0.9	3.3 ± 1.0**
	51. Understand scientific concepts.	2.5 ± 0.8	3.4 ± 1.0**
	52. See connections between different areas of science and mathematics.	2.5 ± 0.9	3.4 ± 1.0**
Unit 8	**Rate this online course with respect to the following criteria:**		
	53. This online course allowed me to interact with my peers in an online setting.		4.1 ± 0.9
	54. This online course used different media (e.g., external sites and videos) that enhanced my learning.		4.0 ± 1.0

	55. I missed having in-person interactions with my peers and instructor.	4.0 ± 1.2
	56. If I were given a choice, I would prefer an in-person version of this course.	4.0 ± 1.2
	57. This online course was well organized and easy to navigate.	4.0 ± 1.1
	58. The amount of time I dedicated to this online course was reasonable.	4.1 ± 1.0
	59. The online evaluation mechanisms (e.g., quizzes, assignments, exams) were clear.	4.1 ± 1.1
	60. The content provided adequately prepared me for the evaluations.	4.1 ± 0.9
	61. The instructor(s) made themselves available via email, or other virtual mechanisms.	4.2 ± 1.0
	62. The proportion of synchronous ("live") versus asynchronous (e.g., recorded) teaching was appropriate	4.0 ± 1.0
	63. The computer and internet connection I used to access online resources was reliable.	4.2 ± 0.8

CONCLUSIONS

This research aimed to fill gaps in knowledge regarding wetlands of the desert southwest, identify how changes in water delivery affected and newly restored effluent sourced wetland and investigate the effects of a virtual CURE on non-science major community college students.

In Chapter 1, we visited various desert wetlands located throughout Texas, New Mexico and Arizona. We were able to collect base line water and macroinvertebrate data for some sites that, to our knowledge, had never been sampled and for others that had not been sampled in years. The results indicated that desert wetlands flooded by various water sources primarily differed along a gradient of salinity. As for macroinvertebrate assemblages, these differences were found to be primarily driven by increased nutrient concentrations from effluent water. Specifically, sites receiving wastewater were found to have lower Simpson Diversity Index scores and more uneven distributions of relative abundances. We also observed lower percentages of metrics related to diversity and environmental sensitivity such as % Ephemeroptera-Odonata-Tricoptera (EOT) within high nutrient sites. Additionally, functional feeding distributions were less even at these sites with filter feeders being the dominant group. Non-wastewater sites had higher Simpson Diversity Index scores and had more even relative abundances of both sensitive and tolerant taxa (e.g., % Coleoptera, % Hemiptera, % Diptera, % EOT) These sites also had higher percentages of functional feeding groups including predators and collector-gatherers. Increased salinity levels were also shown to correlate with lower Simpson Index scores indicating that increased salinity resulted in a decline in macroinvertebrate diversity and evenness. To enhance the water quality and diversity in their sites, it is suggested that managers of these valuable created habitats might try to find less nutrient-rich water sources, or dilute effluent with another water source such as groundwater. Data collected from this project can be used as a baseline to monitor changes due to water availability or climate change.

In Chapter 2, we collected data from a newly restored wetland in El Paso, TX and, tracked how changes in water regimes over time affected water quality and macroinvertebrate assemblages. The increased use of effluent water during the growing season within the Rio Bosque Wetlands created more permanent bodies of water within the ponds and continued flow within the channels. Since the channels continued to receive groundwater, salinity related variables were not significantly different post water increase. In comparison, the ponds were significantly lower in salinity related variables such as conductivity. Due to the use of effluent water, nutrient levels were significantly higher in both the ponds and channels than in previous years. Our results also indicated that macroinvertebrate assemblages were altered in response to the change in water regimes. Due to the difference in water source ratios, there were differences among the ponds and channels even after the change in water regime. The ponds had significantly higher relative abundance of Anisoptera, Zygoptera, Diptera and Ephemeroptera while channels were dominated by salt tolerant taxa such as amphipods. Additionally, there was an increase in the relative abundance passive disperses in the subsequent years. Though specific metrics have not been shown to be reliable indicators of restoration, the increased present of non-insect passive dispersers may indicate advanced stages of succession (Ruhí et al. 2012). Overall, the Rio Bosque Wetlands is displaying succession patterns similar to those of other, more established desert wetlands flooded with effluent water (Chapter 1).

Since many studies indicate it takes 10 years for restored or created wetlands to have similar macroinvertebrate assemblages as natural sites (Marchetti, Garr, and Smith 2010; Ruhí and Batzer 2014), additional sampling of the Rio Bosque Wetlands ponds and channels is recommended till at least 2025. Since there was a temporal correlation with our sampled communities, it is suggested that samples be taken at least monthly throughout the growing season to ensure a more accurate representation of the macroinvertebrate community. As suggested in Chapter 1, the increased mixing of ground and effluent water can be used to further dilute nutrients and salinity within both the ponds and the channels to increase abundances of salinity and nutrient sensitive taxa. Overall, these data can be used to further monitor changes in water quality and macroinvertebrate communities through the restoration process and as a reference site to track the restoration efforts of other wetlands receiving wastewater as a primary water source.

In Chapter 3, we created and implemented a virtual ecology-based CURE at a local community college with non-STEM majors. By incorporating Team-based Learning (TBL) and multiple opportunities for practice of research-based skills into our vCURE, students were able to significantly improve on their overall TOSLS scores and increase their confidence levels in several general science and research related activities. The use of synchronous group meetings gave students the opportunity to work together with their peers and instructors in a virtual setting and is strongly recommended in future implementations of vCUREs. In open ended responses, students felt that the course helped them improved on skills that would be beneficial to them in the future, including their communication, collaborative, and critical thinking skills. This shows that non-science majors can still benefit from CUREs though they do not intend to pursue a research related career. In future iterations of the course, increased measures should be taken to scaffold certain topics such as graph interpretation since this is a skill that students struggled with.

In an effort to increase wetland restoration awareness and give students a more interactive curriculum, a wetland-themed CURE can be implemented to continue macroinvertebrate sampling at the Rio Bosque Wetlands. Students will experience using historical data as well as collecting their own data to track the health of the wetlands. Structures including TBL and the 3 modules used in the vCURE can be adapted and used in an in-person CURE to further enhance course. A similar, in-person, wetland-themed CURE was partially implemented in 2020, prior to stay-at-home orders at El Paso Community College. Generally, the students involved responded well to the course and enjoyed the hands-on experiences at wetlands visited, despite the move to virtual learning prior to the end of the semester.

Overall, we I can say that the vCURE had a positive effect on the students' learning outcomes and confidence levels. As more courses return to in-person learning, this course can be used as model to continue to offer virtual CUREs as a way to increase accessibility to non-traditional student populations seeking research experience (Roddy et al. 2017). In the future this curriculum may be adapted to better meet the needs of STEM-majors and increase the availability of, much needed, ecology-based CURE courses.

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APPENDIX

Table A-1: Orders, families and function feeding groups of macroinvertebrates sampled from Arizona, New Mexico and Texas Wetland sites during the summer months of 2016 and 2020.

Order	Family	Functional Feeding Group
Amphipoda	Hyallelidae	Collector-gatherer
Cladocera	Daphniidae	Filterer
Coleoptera	Dytiscidae	Predator
	Hydrophilidae	Predator
Diptera	Chironomidae	Collector-gatherer
	Culicidae	Collector-gatherer
	Stratiomyidae	Collector-gatherer
	Tabanidae	Collector-gatherer
Ephemeroptera	Baetidae	Collector-gatherer
Gastropoda	Physidae	Scraper
Hemiptera	Belostomatidae	Predator
	Corixidae	Predator
	Naucoridae	Predator
	Notonectidae	Predator
Odonata	Aeshnidae	Predator
	Coenagrionidae	Predator
	Lestidae	Predator
	Libellulidae	Predator
Ostracoda	Cyprididae	Filterers
Tricoptera	Linmephillidae	Collector-gatherer

VITA

Anna Elisa Piña was born and raised in El Paso, Texas. She graduated from Eastwood High School in 2007 and went on to the University of Texas at El Paso to complete her undergraduate education. While pursuing her bachelor's degree, she was Math and Science Teacher's Academy fellow and was part of the Future Educators of Math and Science organization. After she completed her Bachelor's in Science with a minor in Education, Anna became a high school teacher at Coronado High School. During her time as a science teacher, Anna sponsored several clubs including a Hiking Club to teach students about the local desert ecology. Her teaching career took her to several other schools in the El Paso area before bringing her back to UTEP for graduate school.

While Anna worked on her doctorate, she served as a teaching assistant for undergraduate biology courses and graduate level biostatistics. During the summer, she worked with the STEMGrow program, mentoring students from El Paso Community College as they completed their own wetland related research projects. As a Hispanic-Alliance for the Graduate Education and the Professoriate fellow, Anna received invaluable training and mentorship which she applied to her education research. As one of the founding members of The Biology, Environmental, and Engineering Graduate Student Organization, Anna worked alongside her peers to create an environment where students could feel a sense of community, union, and support.

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