Odonate Assemblages of Four Wetland Types Varying in Salinity, Hydroperiod, and Water Chemistry in the Texas Panhandle

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Danielle Husband, B.A.

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Approved

Dr. Nancy McIntyre
Chair of Committee

Dr. Tigga Kingston

Dr. Kerry Griffis-Kyle

Mark Sheridan
Dean of the Graduate School

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I will draft this section later.

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ABSTRACT

Odonates (members of the insect order Odonata: dragonflies and damselflies) are predatory, amphibious invertebrates reliant on freshwater sources for reproduction and development. As a result, these insects are considered an important group of environmental indicators. However, the freshwater sources they rely on are limited in the semi-arid Texas Panhandle, being primarily represented in four forms. Rainfall-filled depressional wetlands called playas are the most numerous, with >20,000 playa basins forming a habitat network in Texas. Less-numerous are spring-fed lakes known as salinas. Urban growth and agricultural land use have effectively created two new wetland types: urban “playa lakes” and salinas with severed spring connections that now function hydrologically like playas (although it is unknown if they function ecologically more like playas or like intact salinas). A comprehensive inventory of odonate assemblages across these four wetland types has never been conducted, which constrains our knowledge about these important insects and whether their presence and abundance reflect gradients of hydroperiod, salinity, and anthropogenic inputs to water chemistry.

I conducted field surveys in the summers of 2020 and 2021 at 58 wetlands (21 playas, 4 salinas, 26 urban playa lakes, and 7 former salinas). I assessed adult species presence, conduced vegetation surveys, and collected water samples for analysis back in the lab. Water chemistry variables analyzed included conductivity [µS], water salinity [ppm], pH, total dissolved solids [TDS ppm], Nitrate (mg/L NO_3), nitrite (mg/L NO_2-N), sulfate (mg/L SO_4), ammonia (mg/L NH_3-N), and phosphate (mg/L PO_4^{3-}). Best subset of environmental variables with maximum (rank) correlation with community dissimilarities (BIOENV) was applied to normalized abiotic variables (each year) to determine which variables produced the highest correlation with assemblage data, thus providing support as to which measured variables structure the assemblage. A multivariate analysis of variance (MANOVA) was used to determine which abiotic variables best explained measured variance across wetland types each summer. Significant MANOVA models were followed by analyses of variance (ANOVA) and Tukey's tests to determine significant differences in abiotic variables among wetland types. Non-metric multidimensional scaling (NMDS) was used to investigate the influence of wetland type on the distribution of species across sites. I computed several metrics to characterize each wetland type's odonate assemblage, including Jaccard and Sørensen indices of beta diversity, Jackknife 1 and 2, Chao, and Bootstrap incidence-based richness estimation methods, and species accumulation curves for each wetland type. Finally, an analysis of similarity (ANOSIM) was used to determine if each wetland type's odonate assemblages significantly differed by wetland type.

Thirty-seven odonate species were detected, with the highest species richness at salinas (N = 31 species), followed by urban playas (23), former salinas (17), and playas (9). NMDS revealed that the odonate assemblage across all four wetland types was nested, with species composition that did not differ greatly across wetland types. The most speciose odonate assemblages occurred at wetlands with longer hydroperiods. I identified several species (N = 12) that may be tolerant of saline environments and merit further study. Salinas and urban playas hosted the most odonate species and the most unique species, yet these wetlands have few protections; recognizing odonate diversity may elicit greater interest in the protection of regionally unique wetlands.

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CHAPTER I

INTRODUCTION

Wetlands are permanently or seasonally inundated areas with hydric soil, an upper layer of anerobic topsoil formed through frequent water saturation (Ramsar Convention on Wetlands, 2018; NRCS, 2022). There are six types of ecosystems classified as wetlands, including riverine (i.e., rivers, creeks), palustrine (i.e. swamps and floodplain marshes), lacustrine (i.e., floodplain lakes, playas), estuarine (i.e., intertidal marshes, mangrove swamps), groundwater-dependent (i.e., rock pools, freshwater springs), and human-made wetlands (i.e. reservoirs, impoundment ponds; Kingsford et al., 2016).

Globally, wetlands are declining three times faster than are forest ecosystems (Ramsar Convention on Wetlands, 2018). It has been estimated that >70% of all wetlands are destroyed or impaired (Kingsford et al., 2016) and that ~87% of wetland areas have been lost entirely (Davidson, 2014). Wetland losses are heavily associated with human land uses (e.g. agriculture, industry, development) and freshwater extraction (Ramsar Convention on Wetlands, 2018). Humans living within wetland areas number in the billions, adding nutrient inputs and altering runoff to wetlands (Strayer and Dudgeon, 2010). Wetland elimination or transformation has spanned centuries, as wetlands have been economically exploited for game, freshwater, and transportation through history (Strayer and Dudgeon, 2010). Remaining wetland areas are altered from the creation of drinking water reservoirs, capture of rainwater runoff, and flood prevention/control that further severs historic ecological function (Strayer and Dudgeon, 2010).

Freshwater wetlands are of particular conservation concern, as they cover less than 1% of the Earth’s surface but provide habitat for ~10% of all known species (Strayer and Dudgeon, 2010). The International Union for the Conservation of Nature estimates that over 140,000 described species rely on freshwater habitat for survival

(Strayer and Dudgeon, 2010; IUCN, 2022). There is no estimate for the percent freshwater wetland loss or total wetland area loss globally for a variety of reasons. For example, for global surface water area there are estimates we have lost ~35% of wetlands between 1970 and 2015, yet we have substituted many of these losses with human-made wetlands like rice paddies and reservoirs (Ramsar Convention on Wetlands, 2018). Although these additions do not compensate for natural wetland loss, these human-made wetlands comprise 12% of wetlands today (Ramsar Convention on Wetlands, 2018). Regardless, 81% of inland wetland species populations have declined since 1970 (Ramsar Convention on Wetlands, 2018).

One region that has historically and presently been impacted by wetland loss is the Texas Panhandle (a region broadly defined here as the area north of 32° N latitude, between 100-103° W longitude, to the northernmost border of the state; Figure 1.1). This region is classified as semi-arid, with an annual average precipitation of less than 60 cm (Sabin and Holliday, 1995). Historically, the Texas Panhandle was short-grass prairie (Smith, 2003). Today, the landscape is a mosaic of towns and cities, cropland, grazed pastures, and former croplands restored to grassland under the USDA's Conservation Reserve Program (Johnson et al., 2012). Because of its relative aridity, the Texas Panhandle would initially appear to be a region depauperate in freshwater availability. However, this region contains thousands of shallow, ephemeral wetlands known regionally as playas that provide habitat resources for wildlife (Bolen et al., 1989). Playas are depressional areas with hydric soil basins (primarily Randall clay and other clay series) (Smith, 2003). These precipitation-fed wetlands are the primary sources of groundwater recharge to the Ogallala Aquifer (Smith, 2003). Playas are dry more frequently than they are wet (Johnson et al. 2011; McIntyre et al., 2018); playa hydroperiods (length of time a wetland holds water) range from 18-453 days depending on seasonal rainfall, groundwater infiltration, and surrounding land use that facilitates or impedes runoff (Tsai and Venne, 2007; Tsai et al., 2010; Collins et al., 2014).

Past and current land-use practices and other human activities are altering the availability and functionality of these wetlands (Bolen et al., 1989; Smith et al., 2011; Anderson et al., 2013; McIntyre et al., 2018). For example, cultivation can break up a playa's clay basin, thereby disrupting its ability to hold water (Smith et al., 2011). Some playas have been dredged and pitted to collect and hold water for livestock; others are pumped or diked during wet seasons and the water is stored for later use (Smith, 2003). Because playas are natural depressions in the landscape, they act as sinks for rainwater runoff and erosional sediments, agrochemicals, and other inputs along the way (Osterkamp and Wood, 1987; Smith, 2003; Anderson et al., 2013; O'Connell et al., 2016). Land conversion for agriculture creates runoff that alters playa water quality (Osterkamp and Wood, 1987; Smith, 2003; O'Connell et al., 2016), and erosional sediments from tillage can fill in playa basins (Smith et al., 2011; Starr et al., 2016; Tang et al. 2018). In fact, one study has indicated that by 2100, projected accumulation of erosional sediments will result in the loss of functional playas throughout the Great Plains (Burris and Skagen, 2013). Modifications as a result of human activities are so prevalent that currently only an estimated 4.7% of playas have no modifications in their basin or watershed (Johnson et al., 2012).

Urban development can likewise affect playa hydroperiod and water quality (Heintzman et al., 2015). The combination of basin modification and presence of impervious surfaces can increase runoff and prolong playa hydroperiod (Collins et al., 2014), so much so that urban playas are often called "playa lakes" (Smith, 2003). Many of the larger cities in the Texas Panhandle (Amarillo, Lubbock) and smaller towns (Abernathy, Plainview, etc.) use playa lakes as drainage systems to mitigate urban flooding. In Lubbock, more than 120 urban playas lakes are incorporated into the stormwater drainage system spanning the city (City of Lubbock, 2010). Playa lakes in these urban areas act as a catchment and water transfer system; as one urban playa reaches water capacity, it drains to the next nearest urban playa basin and so on (City of Lubbock, 2010). Garbage blown in by wind, caught in runoff, or deposited on site is often found at urban playa lakes. Local urban playa lakes have been studied in terms of their coliform bacteria (Moorhead et al., 1998; Durham et al., 2016; Kimbrough et al., 2016), general water quality (Westerfield, 1996; Wolf, 1996), and diseases harmful to wildlife (Barnes et al., 2020).

Although most playas are inundated exclusively from precipitation and runoff, a few dozen have direct connections to groundwater sources via springs (Osterkamp and Wood, 1987; Smith, 2003). Because groundwater contains dissolved salts that remain behind after the water evaporates at the surface, these playas gradually become saline over time and support a regionally unique halophytic biota; these wetlands are known as salt playas or salinas (Heintzman et al., 2017). True playas are ephemeral wetlands, fully drying out for months or years, whereas flowing springs at salinas provide a more consistent source of surface water (Heintzman et al., 2017). If the water table remains at the level consistent with springs, water continues to flow directly from aquifers such as the Ogallala Aquifer year-round. However, as agriculture and development dominate this region, the water table is falling below spring level at many salinas (Heintzman et al., 2017). The saturated thickness of the Ogallala Aquifer is declining rapidly in this region with estimates of 0.3-0.6 meters lost per year and a saturated thickness of less than 15 meters (Ashworth, 2006). As springs cease to flow, salinas transition to “former” salinas that resemble playas more than salinas, hydrologically speaking. There is no current estimate as to how many salinas versus former salinas exist today in the Texas Panhandle, but an assessment of remotely sensed imagery from 1986-2013 for 39 salinas showed that two never held water during that span, and a third dried out but regained water at least once (Heintzman et al., 2017). There is great uncertainty as to how salinas will fare as water demands continue in the region where agriculture dominates and urban areas continue to grow.

Seasonal (playas, former salinas) and more persistent (urban playas, salinas) regional wetlands may differ in their habitat value for biodiversity. As water availability is variable at seasonal wetlands, one would consider those with persistent water to have greater habitat value due to greater potential for survival. This would be especially the case for organisms reliant on water for a part or entirety of their life cycle like aquatic plants and invertebrates. Odonates, or members of the insect order Odonata (dragonflies and damselflies), are biologically reliant on freshwater sources for reproduction, being amphibious invertebrates that spend most of their lives in or near water. As nymphs, odonates are entirely aquatic. Although length of time spent as a nymph can vary, once emerged as adults, water sources remain important sites for mating/reproduction and foraging (Paulson, 2009). Odonates are predators as both nymphs and adults. Their amphibious life history as well as their high trophic position have made odonates a “flagship” group of environmental indicators (Hornung and Rice, 2003; Paulson, 2009). Texas Panhandle odonates have not been extensively studied, as this region is predominately (>95% surface area) privately owned (Texas A&M, 2014). This lack of access has resulted in a critical knowledge gap in understanding regional odonate diversity and the habitats they occupy. No studies have compared odonate assemblages across all four regional wetland types, a pivotal step in understanding their diversity and distribution in this region.

There are ~105 odonate species recorded across the Texas Panhandle (Abbott, 2022). The first published odonate field study for the Texas Panhandle region occurred in 2003 (Abbott et al., 2003). In that study, 14 Panhandle counties were surveyed for the first time. The Panhandle’s odonate assemblage was found to feature a mix of eastern forest and western grassland species (Abbott et al., 2003). Although this study did not solely focus on playa wetlands (also including human-made water sources), 35 species were recorded, resulting in 73 new county records and 4 new state records

(Abbott et al., 2003). This study established that this region was undersampled and data-deficient in terms of odonate biodiversity. Later playa-specific surveys in 16 counties resulted in 25 new species records for the region (Reece and McIntyre, 2009). Another odonate diversity resource is The Dragonfly Society of the Americas' online OdonataCentral (OC) database (odonatacentral.org). OC users can submit observation records, which are vetted by reviewers (Abbott, 2022). Prior to this thesis, ~4400 vetted odonate records (out of over 250,000 records overall) from the Texas Panhandle were reported on OC; roughly 1700 of these records were from playas and salinas. The distribution of these records in the Panhandle reflects land access and frequency of surveys (Figure 1.1).

Odonates may be sensitive to salinity, hydroperiod, and water chemistry. Odonate nymphs have shown to have reduced immune responses in high saline environments, which could negatively impact their survival (Mangahas et al., 2019). Some species, families, or suborders may exhibit higher sensitivity to salinity, such as members of the family Gomphidae in the dragonfly suborder, which are less prevalent in higher-salinity waters compared to families in the damselfly suborder (Zia et al., 2018). Thus, there is a possibility that certain odonate species in this region may be limited by physiological tolerances to salinity. In terms of hydroperiod, higher odonate diversity has been observed at urban playas than rural playas surrounded by cropland or grassland (Husband and McIntyre, 2021). Urban playas are consistently fed by urban runoff and hold water longer, thus these wetlands serve as critical odonate refugia even during periods of sustained drought (Husband and McIntyre, 2021). There is also ample evidence supporting odonate sensitivity to changes in water chemistry from dissolved oxygen to ammonia and phosphate (Beketov, 2002; Chang et al., 2007; Al-Shami et al., 2014; Mangahas et al., 2019). Low pH and high ammonia levels are known to cause mortality in odonate nymphs in the Coenagrionidae, Lestidae, and Libellulidae (Beketov, 2002).

In this thesis, I determined which of the Texas Panhandle four wetland types (playa, urban playa, salina, former salina) support the greatest odonate species richness, compared community similarity, and identified differences in salinity, hydroperiod, and water chemistry that may be driving regional species diversity. This information is needed to form a better understanding of regional odonate diversity at these different wetland types. This information is useful for any future regional odonate studies, as my determinations across these wetland types may lead to further questions at smaller site or within wetland type scales. Moreover, because odonates are wetland indicators (Hornung and Rice, 2003; Catling, 2005; Osborn, 2005; Chang et al., 2007; Al-Shami et al., 2014; Abdul et al., 2017; Perron et al., 2021), their presence and diversity may be used to represent responses in other taxa, such as amphibians. I will submit all species photo vouchers to OdonataCentral to update Texas Panhandle county species lists. The findings are presented in Chapter II; Chapter II as written has been submitted for publication (Husband and McIntyre, in review) and uses plural pronouns. Chapter III consists of concluding remarks.

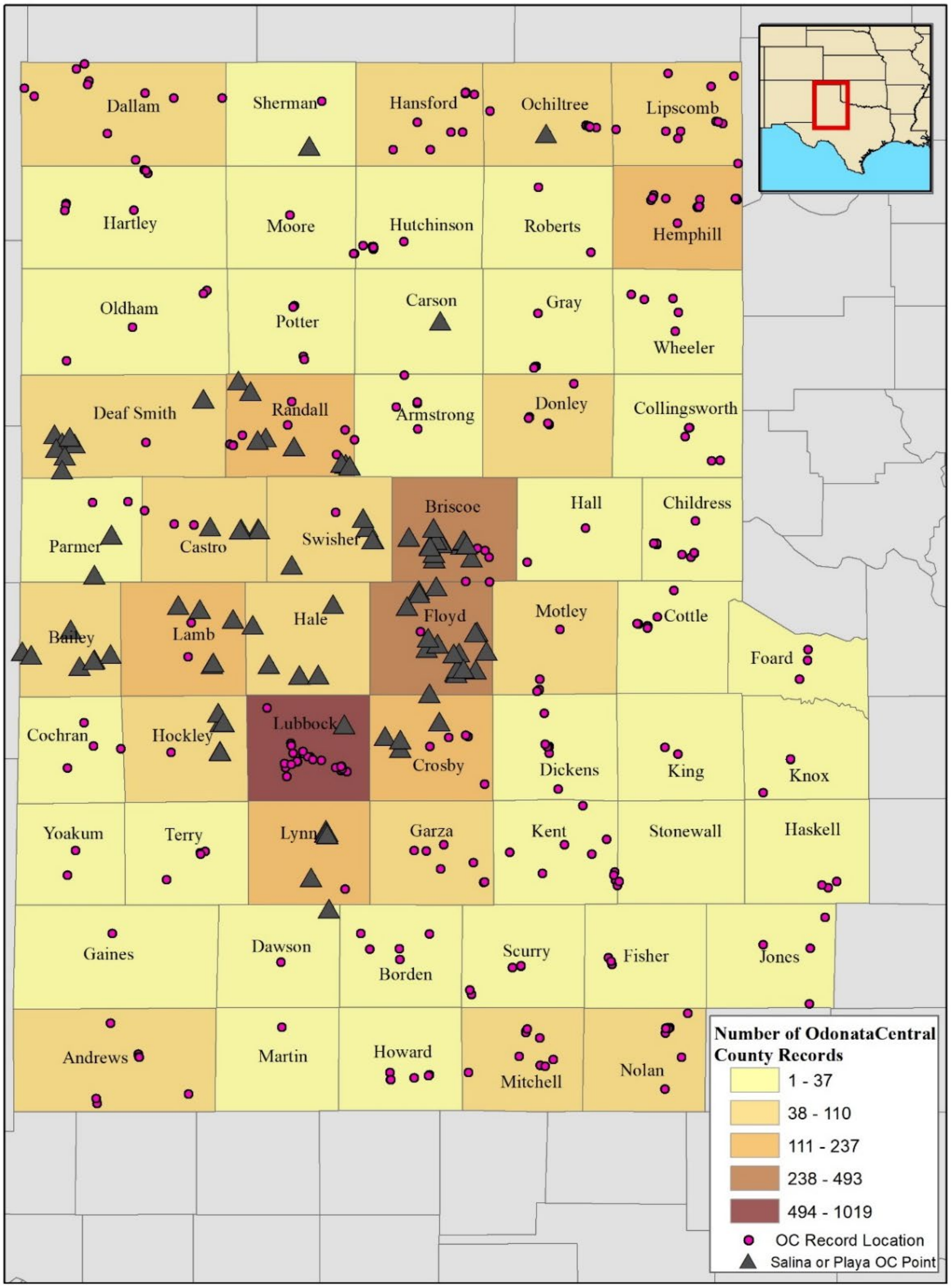

Figure 1.1: OdonataCentral odonate records from salinas and playas (296 sites representing 1738 records; gray triangles) for the Texas Panhandle (4457 odonate records overall prior to this thesis; red point) by county (N = 58). Counties with darker shading indicate more records. Layer credit: ArcGIS online (states, county layer), OdonataCentral (records).

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CHAPTER II

ODONATE ASSEMBLAGES OF FOUR WETLAND TYPES VARYING IN SALINITY, HYDROPERIOD, AND WATER CHEMISTRY

Introduction

Freshwater wetlands cover ~1% of the earth's surface yet provide habitat to >40% of the world's biodiversity (Mitra et al., 2003). Globally, as many as 87% of wetlands are estimated to have been modified or lost entirely since the beginning of the industrial age (Davidson, 2014; Hu et al., 2017). This has contributed to the decline of aquatic invertebrates worldwide (Collen et al., 2014; Collier et al., 2016; Reid et al., 2019). As wetland alterations and losses are projected to continue in the foreseeable future, documenting the aquatic biodiversity at wetlands is needed to provide a benchmark in understanding what could be lost, particularly in regions and among wetlands that are not traditionally considered to be of great conservation importance. One such region is the North American Great Plains. Although the Great Plains are semi-arid with episodic droughts and seasonal precipitation (April-September; Shafer et al., 2014), it is also characterized by several types of wetlands that differ in origin, hydrology, and abundance.

The most numerous of these are the ~80,000 ephemeral wetlands called playas (Smith, 2003). Playas are shallow, rainwater- and runoff-fed wetlands that are more often dry than wet (Bolen et al., 1989; Smith et al., 2011; Daniel et al., 2014; McIntyre et al., 2018). They are the primary sources of recharge to the Ogallala Aquifer, a 450,000 km² groundwater repository spanning from South Dakota to Texas (Osterkamp and Wood, 1987; Rosenberg et al., 1999; Smith, 2003), and are regional foci for biodiversity, including insects (Haukos and Smith, 1994; Smith, 2003). Hydroperiods range from 18-453 days depending on seasonal rainfall, groundwater infiltration, and surrounding land use that facilitates or impedes runoff (Tsai et al., 2007; Collins et al., 2014). The highest density of playas occurs in the Southern High

Plains (SHP) of New Mexico and Texas, the southernmost portion of the North American Great Plains (Fish et al., 1998). Historically, these playas occurred within a shortgrass prairie matrix that has been since modified for agriculture in the form of livestock grazing (~49% land area) and tillage cultivation (>24% land area; Starr and McIntyre, 2020). Erosional sediments from tillage can lead to basin infill (Skagen et al., 2008). Consequently, it has been estimated that as many as 85% of playas in the SHP have lost their ability to retain water (Daniel et al., 2011; Collins et al., 2014). Additionally, agriculture surrounding playas is the source of various chemicals (e.g. nitrite, nitrate, ammonia, phosphate, and other components of livestock waste, fertilizers, herbicides, or pesticides, as well as total dissolved solids from erosional sediments) that wind up in runoff and thence in playas (Hornung and Rice, 2002; Smith, 2003; Anderson et al., 2013).

The SHP also contains three other wetland types in lower abundances: urban playas, salinas, and former salinas. Most playas in urban areas of the SHP have been modified for stormwater management and/or recreation (Collins et al., 2014), primarily by deepening their basins and constructing channels for entrance of runoff. These alterations, in addition to the presence of supplementary water sources (i.e., urban irrigation runoff), lengthens the hydroperiod of urban playas relative to non-urban ones (Collins et al., 2014). These prolonged hydroperiods have led to their colloquial name of “urban playa lakes.” Anthropogenic water inputs also mean that urban playas can differ in water chemistry (e.g. in sulfate, polycyclic aromatic hydrocarbons, and other components of automotive exhaust) compared to the other types of regional wetlands (City of Lubbock, 2010; Heintzman et al., 2015).

Salinas (also called salt playas or saline lakes) are spring-fed wetlands that are far less numerous (<50 in the SHP) yet larger than playas (Heintzman et al., 2017). Groundwater mineral content accrues within salina basins as evaporation occurs, creating a saline environment (Smith, 2003; Rosen et al., 2013). Increased groundwater use for agriculture (starting ca. 1940) and diminished recharge of the

Ogallala Aquifer has lowered the water table below spring depths in many areas, halting water flow at many salinas (Rosenberg et al., 1999). From 1986 to 2013, the saturated thickness of the Ogallala Aquifer declined by 18.6%; this depletion effectively altered spring flows from year-round to seasonal or nonexistent (Heintzman et al., 2017). Today, there are estimates of as few as 10 salinas with functioning springs, and only ~60% that hold water year-round (Rosen et al., 2013; Heintzman et al., 2017). Salinas that now lack consistent spring flow (herein called former salinas) hydrologically behave like playas, receiving water only through rainfall and being dry for long periods of time, yet with saline basins.

Collectively, these four wetland types—playas, urban playas, salinas, and former salinas—provide the primary forms of aquatic habitat in the SHP. Their differences likely support different biota, but it is unknown as to whether hydroperiod, salinity, or water chemistry is the most influential in structuring assemblages of most organisms such as aquatic and semi-aquatic insects. Lack of such baseline knowledge hinders monitoring the effects of environmental change and establishment of realistic conservation goals.

Due to their amphibious life history and high trophic level, odonates (Odonata: dragonflies and damselflies) have been used as indicators of water and wetland quality (Catling, 2005; Osborn, 2005; Chang et al., 2007; Al-Shami et al., 2014; Abdul et al., 2017; Perron et al., 2021). In a region where water is inherently scarce, odonates of the SHP are restricted to these four wetland types; the differences in hydroperiod, salinity, and water chemistry among the four types may filter the regional odonate assemblage, resulting in distinctions by wetland type. For example, various water chemistry parameters (e.g. dissolved oxygen, ammonia, phosphate) are known to affect odonate development (Beketov, 2002; Chang et al., 2007; Al-Shami et al., 2014; Mangahas et al., 2019), and low pH and high ammonia levels are known to cause mortality in odonate nymphs in the Coenagrionidae, Lestidae, and Libellulidae (Beketov, 2002). High salinity can cause physiological stress to many freshwater organisms; for example, odonate nymphs in high saline environments exhibit diminished immune responses (Mangahas et al., 2019), and the dragonfly families Gomphidae and are less prevalent in higher-salinity waters compared to damselfly families (Zia et al., 2018). We thus expected odonate richness and species composition to differ among the four wetland types of the SHP. Specifically, urban water bodies and rural ones influenced by agriculture (grazing and cultivation) should be lower in dissolved oxygen due to excess nutrient loads from runoff, which has been shown to reduce Zygopteran diversity (Osborn, 2005) and overall odonate diversity (Ishak et al., 2021). Most species in our focal region have not been examined in terms of their responses to anthropogenic chemicals, although we expect differences in the odonate assemblage given the above-mentioned studies. In the absence of specific information on species in this region, assemblage patterns in this study may uncover potential water chemistry and salinity responses that can guide future research.

Our understanding of SHP odonate diversity at these wetland types has been limited due to restricted access to private property and little knowledge about the quality or condition of local freshwater sources. Initial surveys focused on roadside ditches and public urban parks (Abbott, 2001). Further studies extended regional odonate knowledge to playas, finding similar odonate assemblages at playas surrounded by cropland and grassland (Hernandez et al., 2006; Craig et al., 2008; Reece and McIntyre, 2009a, 2009b). However, there has been no study conducted to date that has assessed odonate diversity beyond playas and urban areas. Herein, we address these deficiencies in knowledge of regional odonate diversity at all four regional wetland types (playa, urban playa, salina, former salina) to describe how odonate assemblages are structured along axes of hydroperiod, salinity, and water chemistry (Figure 2.1). We expected that salinity of salinas and former salinas will support species able to endure halophytic environments relative to the more freshwater basins of playas and urban playas (*salinity hypothesis*). Because the hydroperiod of each of the four wetland types can be designated as either short (playas, former salinas) or long (urban playas, salinas) based on the presence of water inputs other than precipitation, we expected larger species with longer nymphal development times will only appear at wetlands with longer hydroperiods (salinas and urban playas) compared to variable rainfall-driven hydroperiods at playas and former salinas, which may not hold water for the necessary length of time for development of some univoltine species (*hydroperiod hypothesis*). In addition to these inherent differences among wetland types, surrounding land use may also present different water chemistry barriers to survival. Thus, we expect playas surrounded by agriculture and urbanization to have water conditions more indicative of anthropogenic land use compared to salinas and former salinas (*water chemistry hypothesis*).

Materials and Methods

Study sites

We surveyed 26 urban playas (Bailey, Hale, Lubbock, Randall counties), 4 salinas (Gaines and Lynn counties), 7 former salinas (Andrews, Bailey, Terry counties), and 21 playas (Castro, Floyd, Lubbock, Swisher counties) in Texas (Figure 2). Add more details here about each type of wetland.

The majority of wetlands on the SHP are on private property (Smith, 2003). Urban playas, however, are publically accessible and thus represent a large proportion of our study sites. One site, Muleshoe National Wildlife Refuge, is a federal property for which we applied for a Research and Monitoring Special Use Permit to access three salinas. The remaining field sites were privately owned. Only sites that held water were surveyed for odonates; this differed between years. In summer 2020 (relatively dry), we sampled 24 urban playas, 2 salinas, 1 former salina, and 0 playas (because all playas were dry); in summer 2021 (wetter than 2020), we sampled 26 urban playas, 5 salinas, 4 former salinas, and 13 playas (Table 1).

Odonate data

Surveys were conducted between 1 June and 30 September 2020 and 2021 on days above 23°C with clear to moderately clear skies and low wind (<35 km/hr) between 0900 and 1400. Surveys were performed using an Esri Survey123 application designed by D.M. Husband (https://github.com/D-Husband/Texas-panhandle-odonates). All adult odonates were visually identified at three survey points (at least 50 m apart) per wetland per visit. Adults were the focus due to greater feasibility of rapid and accurate identification compared to other life stages. Adequately representing adult odonate richness at a given site entails balancing sampling duration with repeated site visits (Bried et al., 2011) as well as time needed to visit sites that are 100s of km apart. Our surveys lasted no more than 15 minutes (5 min at each of three points). Each wetland was visited and surveyed twice each summer, once during early summer and once during late summer, meaning that each wetland was surveyed for 30 min in total per year. Twenty-one sites were surveyed only in either 2020 or 2021 (rather than in both years) due to land access permission or lack of water. Thus, the number of sampling visits per site over the duration of our study ranged from 2-4. Voucher photos were taken to document species occurrences; a few voucher specimens were collected for further identification. These specimens are housed in the Department of Biological Sciences at Texas Tech University.

Environmental Data and Analyses

Survey locations were situated on the perimeter of each wetland basin (littoral zone), as this is where emergent vegetation used for emergence and perching occurs (Corbet, 1999); percentage of shoreline with vegetation was estimated visually during each sampling visit (0-100% in increments of 1%). Thirteen water variables known to be present in agricultural or urban runoff into aquatic system were analyzed; these variables have been included in other odonate studies, making our work more directly comparable (Osborn, 2005; Al-Shami *et al.,* 2014; Holtmann *et al*., 2018; Perron *et* *al.,* 2020; Ishak *et al.,* 2021). Only basins with water were sampled. Some water chemistry variables (conductivity [µS], water temperature [°C], salinity [ppm]), pH, and total dissolved solids [TDS ppm]) were measured on site with a Waterproof ExStik® II (Extech Instruments, Boston, Massachusetts, USA); other variables were measured in the lab. Two 50 mL water samples were collected at sites that held water (meaning that for some sites, like playas, only one sample was collected, as they were only inundated in summer 2021 but not in 2020). For salinas, water samples were collected from the area near the spring(s) rather than within the saline basin for two main reasons: (1) at all salinas, water flow downstream from streams dried before reaching the basin proper in both 2020 and 2021, and (2) odonate activity was concentrated at the springs and nearly absent from the saline basins. For sites with multiple water sources, e.g. salinas with multiple springs, two water samples were collected from each water source and values were averaged for analysis. Samples were transported to the Department of Biological Sciences, Texas Tech University, for same-day processing. Nitrate (mg/L NO_3), nitrite (mg/L NO_2^{-}—N), sulfate (mg/L SO_4), ammonia (mg/L NH_3-N), and phosphate (mg/L PO_4^{3-}) were measured using a Hach Drel 2400 (Hach, Loveland, Colorado, USA) complete water quality kit. Turbidity (NTU) was evaluated using Hach 2100P Turbidimeter.

Water chemistry values were analyzed separately by year due to interannual variation in which types of wetlands could be surveyed (were wet). Prior to running statistical tests, abiotic data were normalized (non-percent values by Log_{10}, percent values by arcsine square root), and variables that were highly correlated ($r > 0.8$) were removed from all analyses (2020: conductivity removed, $r = 0.98$ with salinity; total dissolved solids removed, $r = 1$ with salinity; NH_3 removed, $r = 0.82$ with NO_2; 2021: NH_3 removed, $r = 0.82$ with turbidity; total dissolved solids removed, $r = 1$ with salinity). Best subset of environmental variables with maximum (rank) correlation with community dissimilarities (BIOENV) was applied to the remaining normalized abiotic variables each year to determine which variables produced the highest correlation with assemblage data, thus providing support as to which measured variables structure the assemblage. A multivariate analysis of variance (MANOVA) was used to determine which abiotic variables best explained measured variance across wetland types each summer. Significant MANOVA models were followed by analyses of variance (ANOVA) and Tukey's post-hoc tests to determine significant differences in abiotic variables among wetland types. Finally, an analysis of similarity (ANOSIM) was used to determine if each wetland type's odonate assemblages significantly differed by wetland type. All analyses were conducted in RStudio 1.4.1103 (R Core Team, 2021) .

Odonate assemblage characterization

Using the R package vegan (Oksanen et al., 2020), we computed several metrics to characterize each wetland type's odonate assemblage. Species accumulation curves determined if the odonate assemblages at each wetland type were adequately sampled. Because richness estimations can determine degrees of species nestedness and turnover, which are important indicators of assemblage stability (Craig et al., 2008), alpha, beta, and gamma diversity were calculated for each site, for the different types of wetland, and for the entire dataset as a whole, respectively (Magurran, 1988). Jaccard and Sørensen indices of beta diversity for species presence/absence data were calculated and compared across wetland types (Magurran, 1988). Jackknife 1 and 2, Chao, and Bootstrap incidence-based richness estimation methods were generated to extrapolate odonate richness at each wetland type. These estimates were compared to determine the best-fit model (i.e., closest 95% confidence interval to actual measured site richness). Finally, a non-metric multidimensional scaling (NMDS) ordination was constructed with a Bray-Curtis dissimilarity matrix, using the 'metaMDS' function from vegan; stress plots were used to determine the optimal number of axes to reduce overfitting (in our case, optimal $k = 2$). We examined the ordination axes as dependent variables to investigate the influence of wetland type on the distribution of species across sites.

Results

In summer 2020 we sampled 24 urban playas, 2 salinas, and 1 former salina. Twenty-seven species were recorded that summer compared to 35 in summer 2021, when 26 urban playas, 5 salinas, 4 former salinas, and 13 playas were sampled (Table 2.1).

Over both summers, 37 odonate species were detected. The highest species richness was found at salinas (31 species), followed by urban playas (23), former salinas (17), and playas (9) (Table 2.2, Figure 2.3). Eight species were observed across all wetland types; the remaining species were found at more than one wetland type (Table 2.1). Both Jaccard and Sørensen indices were consistent in the order of most similar and least similar odonate assemblages by wetland types (Table 2.3). Odonate assemblages were most similar at salinas and playas and dissimilar at urban playas and salinas (Figure 2.4, Table 2.3).

Only odonate sampling at playas adequately represented regional richness, indicated by an asymptotic species accumulation curve (Figure 2.3). The species accumulation curves at the other wetland types indicate that further sampling is needed to adequately estimate odonate richness. Estimated odonate richness at salinas, urban playas, and former salinas showed the greatest disparities from their actual measured richness values (Table 2.4). In contrast, richness estimates for playas were very close to their measured values, with low beta diversity (Table 2.4). Urban playas and salinas showed the greatest turnover, as extrapolated richness values deviated widely from observed odonate assemblage richness (Table 2.4).

The NMDS plot at $k = 2$ had a stress value of 0.187, indicating a reasonable ability to condense multivariate data into a two-dimensional ordination (McCune and Grace, 2002). The NMDS revealed that the odonate assemblage across all four wetland types was nested, evidenced by high overlap in 95% confidence interval ellipses (Figure 2.5). Former salinas were the most nested, with none of its ellipse extending beyond the other three, indicating that all of the species encountered at former salinas were also observed at other wetland types. Salinas, playas, and urban playas, on the other hand, all had some extension beyond their ellipses, indicating species not encountered elsewhere (Figure 2.5). The odonate assemblage was statistically different between salinas and urban playas in summer 2020 (the only two wetland types that held water in 2020) (ANOSIM: $R = 0.344$; $p = 0.0325$), but during summer 2021 when there was much more water available on the landscape, odonate assemblages did not significantly differ among wetland types (ANOSIM: $R = 0.0825$; $p = 0.1319$).

In 2020, site richness data at sampled salinas and urban playas were highly correlated with salinity and percent vegetation surrounding the basin (BIOENV: $R = 0.34$). In 2021, site richness data at sampled urban playas, playas, salinas, and former salinas were highly correlated with nitrate, sulfate, percent basin fill, percent vegetation surrounding the basin, and salinity (BIOENV: $R = 0.24$). There were differences in water availability, water chemistry, and availability of edge vegetation by wetland type, with differences between years. Among salinas ($N = 2$) and urban playas ($N = 24$) in 2020, there was a significant difference in sulfate, percent basin fill, percent edge vegetation, and salinity (Table 2.6). Among sampled salinas ($N = 7$), playas ($N = 13$), urban playas ($N = 26$), and former salinas ($N = 6$) in 2021, there was a significant difference in sulfate, phosphate, nitrate, percent basin fill, percent edge vegetation, salinity (Table 2.6). For 2021, there likewise were significant differences in these variables with respect to wetland type (Table 2.6).

Contrary to our expectations, salinity was not arranged along a gradient from salinas and former salinas (expected to have high salinity) to playas and urban playas (low salinity). Instead, the highest aqueous salt concentrations were found at urban playas (Tables 2.5-2.7), meaning that urban species must exhibit some degree of salt tolerance. Similarly, water chemistry did not fall along a gradient from urban playas and playas (high concentrations of chemicals of anthropogenic origin) to salinas and former salinas (low expected concentrations) (Tables 2.5-2.7). Instead, different variables were associated with each of the wetland types; the odonates unique to each wetland type thus may be ones tolerant of high values of those variables. Hydroperiod did behave as expected, with salinas and urban playas having water present on nearly 100% of the sampling visits, and playas and former salinas with water present 0% of the time in 2020 and ~50% and 62% of the time in 2021, respectively (Table 2.1). The largest-bodied univoltine species (with body size being positively associated with length of nymphal development) (*Rhionaeschna multicolor*, *Anax junius*, *Phanogomphus militaris*) were indeed found primarily or exclusively at the wetlands with the longest hydroperiods (Table 2.2).

Discussion

Hydroperiod hypothesis

Eight species were recorded at all wetland types and represented a large proportion of the species observed at wetlands with relatively short hydroperiods (i.e., playas and former salinas). Consistent with the hydroperiod hypothesis, we recorded more species overall and more species that were unique to both urban playas and salinas compared to the other types of wetlands. This finding supports the idea that wetlands with relatively longer hydroperiods have odonate assemblages that differ from those at wetlands with shorter hydroperiods (i.e., playas and former salinas). Indeed, the assemblages at playas and former salinas consisted mainly of ubiquitous odonate species. This finding suggests that longer hydroperiods are essential for many odonate species in this dry region, lending to the understood importance of freshwater for invertebrates in semi-arid climates.

Hydroperiod is known to structure biotic communities through a variety of proximate mechanisms (Wellborn et al., 1996). For example, semi-permanent to permanent wetlands can support more complex food webs (Wellborn et al., 1996). In general, the presence of fish negatively affects odonates, selecting for species with physical or behavioral adaptations against predation (McPeek, 1990). Although some urban parks are stocked with fish (City of Lubbock, 2021), the wetlands of our focal area are naturally fishless (Smith, 2003). Fish may not be a proximal driver of odonate diversity in this study system. However, we believe the subject may need further investigation at urban playa lakes. We anecdotally noted fewer odonates at sites with abundant stocked fish, but this was not a universal narrative at all parks.

The urban playa lakes, being used in stormwater drainage, had relatively long hydroperiods, but the presence of water is not the only factor that could be responsible for the high diversity that we saw at those wetlands (Figure 6). Although urbanization is associated with declines in odonate diversity in some areas (Villalobos-Jiménez et al., 2016), we found the contrary: urban playas were the second-most diverse wetland type. This could be because urban development in this region is associated with creation of greater habitat heterogeneity (seen in Figure 2.1), primarily through afforestation, and habitat heterogeneity is positively associated with greater odonate diversity (Goertzen and Suhling, 2013). For example, Clapp Park in Lubbock, Texas, anecdotally appeared to have greater habitat heterogeneity due to the presence of reeds, open water, upland vegetation, and tree cover to a degree not seen at other sites, and had the greatest odonate diversity as well as several species that were found nowhere else. Differentiating between the influence of habitat heterogeneity and hydroperiod on odonate diversity would require additional data on these variables. Even within a wetland type, sites differed in localized conditions such as coverage of edge vegetation, presence of floating macrophytes, and trees, which contributed to differences in odonate diversity by site (Table 2.8).

Odonate communities at the four wetland types were largely similar, but some differences in environmental conditions were noted by year that would potentially affect the regional species pool. Summer 2020 was the 11th driest on record for the SHP, with only ~62% of the annual precipitation average of 47.14 cm; National Weather Service, 2021); former salinas and playas were completely dry that year, so only salinas and urban playas could be sampled then (and there was a difference in the odonate assemblage noted between these two wetland types that year). There was more rainfall in 2021 (39.52 cm), meaning that all four wetland types could be surveyed that year. However, no significant differences in odonate assemblages across wetland types were found that year. Even though they could only be surveyed in 2021 (being dry in 2020), only playas were determined to be sufficiently sampled via species accumulation curves; playas had the fewest observed species (N = 9) and many species that were found at all wetland types. Because playas are ephemeral and thus more often dry than wet, it is not surprising that their assemblage largely consisted of regionally common species and genera known to be migratory (e.g. *Tramea* and *Sympetrum*).

Salinity hypothesis

We anticipated that salinas and former salinas would support unique (salt-tolerant) odonate assemblages. Salinas did support 12 species found nowhere else, including ones known to be salt-tolerant (e.g. *Libellula composita*; Larsen, 2008). Other species like *Phanogomphus militaris*, *Dythemis fugax*, and several *Ischnura* species may be salt-tolerant, as they were detected only at salinas, but further investigation is needed on possible salt tolerance in these species. Former salinas, on the other hand, did not have any of these species. Additionally, we expected salinity to be much lower at playas and urban playa lakes relative to salinas and former salinas. Surprisingly, however, urban playas had much higher salinity values than did salinas (Figure 2.6). The likely explanation for this is that water samples were taken where odonates were found, which for salinas was not in the basin but instead by springs. These samples were thus lower in salinity than if they had been taken by the basin's edge where odonate habitat was not ideal, being devoid of emergent vegetation (Table 2.5, S7). Salinity is known to cause a decline in the abundance of aquatic invertebrates, impacting their distribution and colonization behaviors (Swanson et al., 1988; Halse et al., 1998; Carver et al., 2009; Preston et al., 2018). Salinity was a significant variable at sampled sites across years and was found to influence community structure both summers. Odonate assemblages at urban playas and former salinas, where salinity was greatest, were highly dissimilar, indicating that factors other than salinity drive odonate diversity in this largely freshwater area. Odonates were overall depauperate at the majority of former salinas where the highest salinity was measured. Similarly low invertebrate diversity has been measured in wetlands of the Prairie Pothole Region where brine inputs from oil and gas industries have increased salinity in otherwise low salinity habitats (Preston et al., 2018). Odonate richness at urban playas did not reflect the same high salinity = low diversity trend, which may suggest another factor may be responsible for odonate survival there.

Water chemistry hypothesis

We hypothesized that water chemistry at playas and urban playas would differ from that at salinas and former salinas due to differences in runoff by surrounding anthropogenic activity and land use. From our study, however, no species was associated in terms of presence with any of the water chemistry variables we measured, possibly because although the wetland types differed in water chemistry, they did not follow the expected gradient by wetland type. For example, sulfate and salinity were two variables that significantly differed in both years among sampled sites, being high at salinas, former salinas, and urban playas and thus not following any of our hypothesized gradients. Sulfate naturally occurs in groundwater, so its higher measured presence at salinas and former salinas is not surprising given their current or former spring connections (Hudak, 2000). Its presence in urban playas may be explained by the fact that this region’s alkaline soils cause iron deficiencies in many plants; treatment occurs in urban and agricultural areas and uses sulfate in its application (Mengel, 1994; Tangen et al., 2021; Utah State University, 2021). Runoff of these applications may thus explain high urban sulfate levels. No studies have investigated the effects of high-sulfate waters for odonate nymphs or adults. Amphibian assemblages at Southern High Plains playas have been examined in respect to chemicals of anthropogenic origin (Venne et al., 2006). Amphibians were not found to bioaccumulate metals used in agriculture (Venne et al., 2006). Similar to our study findings, playa water quality had less of an influence of amphibian species presence than other potential factors like hydroperiod and vegetation (Gray et al., 2004; Venne et al., 2006).

Conclusions

Odonate assemblages differed at four wetland types of the Southern High Plains. Most species occurred at all four types; salinas had the highest species richness and the most species found at no other wetland type. Urban playas were the next most speciose following salinas. Differences in diversity were associated with differences in hydroperiod by wetland type more than salinity or water chemistry.

Because short-term studies are inherently weaker in representing species richness, abundances, and occurrences of specialist species (Dolný et al., 2021), long-term monitoring is imperative to capture interannual differences in odonate assemblages. Due to underestimation of odonate diversity at most of our regional wetland types, we recommend that future studies should be lengthened to account for interannual variation in dry and wet conditions. Replication is inherently difficult in natural studies in arid regions, as weather conditions vary greatly year to year and from location to location within a year. In addition, incorporating nymph sampling would expand our understanding of which odonate species emerge from sites, rather than sampling adults that possibly emerged elsewhere and dispersed to their present sampled location. Although there is some difficulty in identification and application to monitoring programs, nymphs provide direct insight into odonate fitness (Catling, 2005; Jeanmougin et al., 2014; Chen et al., 2020; Perron et al., 2021). Similarly, we did not directly sample vegetation, although given the unique plant life at salinas recorded by Rosen et al., (2013), we recommend more rigorous basin edge and upland sampling be conducted, since there is evidence that certain odonate species rely on specific vegetation structures (Goertzen and Suhling, 2013; Jeanmougin et al., 2014; Perron et al., 2021). Finally, more rigorous, long-term water chemistry monitoring is recommended for future studies. The two water samples we collected from most sites provided only a snapshot of water conditions throughout the summer season and beyond.

There are few federal or state protections in place to protect wetlands of the SHP. Most of three of the four wetland types examined occur on private property, complicating regional-scale conservation efforts (Smith, 2003). Most such efforts work one-on-one with landowners. For example, the U.S. Department of Agriculture's Conservation Reserve Program works with private landowners to reduce upland soil deposition into playa basins through planting of grass buffers (Smith et al., 2011; Belden et al., 2012), and non-profit organizations like the Playa Lakes Joint Venture have provided guidance and resources for playa restoration and management (Smith, 2003). Despite these efforts, playa losses are ongoing. Likewise, salinas are a vanishing wetland type in the region, with estimates of fewer than 10 with functioning springs in the Texas panhandle (Rosen et al., 2013; Heintzman et al., 2017). This is of particular conservation concern, given how many odonate species were found only at salinas. Former salinas are especially shrouded in uncertainty as to how odonates will persist there in the future. Former salinas and playas appeared to provide fine habitat in wet months, but because they dry up frequently and for long durations, they may act as an "ecological trap," providing habitat resources for species but with hidden costs to persistence and fitness in the long term (Dwernychuk and Boag, 1972). Such may also be the case for at least some urban playas. In our study area, most urban parks are landscaped, and vegetation is mowed to the urban playa lake's edge, which can eliminate emergence and perching substrate. There is little evidence that the state or local municipalities test water quality at urban playas, nor any evidence on how poor water quality issues at urban playas are remediated in this region. As urban playas are consistently inundated by anthropogenic sources, odonates could be a model taxon to monitor the health of these sites. Public properties, like urban playas, also present opportunities for public engagement and education on the importance of these organisms and wetlands.

Although protecting playas, salinas, former salinas, and urban playas for conservation is complicated given differences in access, odonates inhabit them all, and deficiencies in our knowledge of their distribution remain a central challenge in enacting protections. As issues of climate change and continued anthropogenic impacts continue, there is uncertainty as to how odonates will fare in years to come. The SHP is semi-arid but as our study revealed, it is not devoid of water. Pockets of wetland habitat exist both year-round and seasonally, offering habitat for odonates. Odonates continue to fascinate the public eye as charismatic insects capable of flight. By bringing attention to odonates in the Texas panhandle, we hope to inspire greater awareness to their protection and conservation.

Table 2.1: Frequency of the occurrence of water in sampled wetlands of the Texas Panhandle (U.S.A.), 2020-2021.

Wetland type	**% Wet**	**% Dry**	**Site Visits**	**N**
Urban playa	100	0	100	26
Playa	50	50	30	21
Salina	100	0	12	4
Former salina	62	38	13	7

Table 2.2: Odonate species detected at each wetland type at a single site (**X**^{1}) or at several sites (x). Dragonflies (suborder Anisoptera) are listed first alphabetically, followed by damselflies (suborder Zygoptera).

Species	**Urban Playa**	**Playa**	**Salina**	**Former Salina**
Anax junius	x	x		x
Brachymesia gravida			**X**^{1}	
Celithemis eponina	**X**^{1}			
Dythemis fugax			**X**^{1}	
Erythemis simplicicollis	x		x	**X**^{1}
Erythemis vesticulosa	x			
Libellula comanche			**X**^{1}	
Libellula composita			**X**^{1}	
Libellula luctuosa	x		x	x
Libellula pulchella	x	**X**^{1}	x	x
Libellula saturata	x		x	x
Orthemis ferruginea	x		x	x
Pachydiplax longipennis	x		x	
Pantala flavenscens	x	x	x	x
Pantala hymenaea	x	x	x	x
Perithemis tenera	x		x	
Plathemis lydia	x		x	
Phanogomphus militaris			x	
Plathemis subornata			**X**^{1}	
Rhionaeschna multicolor			x	
Sympetrum corruptum	x	x	x	x
Tramea lacerata	x	x	x	x
Tramea onusta	x	x	x	x
Argia alberta			x	
Argia immunda			x	
Argia sedula				**X**^{1}
Enallagma civile	x	x	x	x
Hetaerina americana	**X**^{1}		**X**^{1}	
Ischnura barberi	x		x	**X**^{1}
Ischnura damula			**X**^{1}	
Ischnura demorsa			**X**^{1}	
Ischnura denticollis			x	**X**^{1}
Ischnura hastata	**X**^{1}			
Ischnura ramburii	**X**^{1}			
Ischnura verticalis	**X**^{1}		**X**^{1}	
Lestes alacer			x	
Lestes australis		x		**X**^{1}
Telebasis salva	**X**^{1}		**X**^{1}	**X**^{1}

Table 2.3: Jaccard and Sørensen beta diversity index values.

Jaccard Index	Former Salina	Playa	Salina
Playa	0.529		
Salina	0.412	0.250	
Urban playa	0.538	0.333	0.543
Sørensen Index	Former Salina	Playa	Salina
Playa	0.692		
Salina	0.583	0.400	
Urban playa	0.700	0.500	0.700

Table 2.4: Richness indices for each wetland type.

Type	**N**	**Observed richness**	**Chao (SE)**	**Jack1 (SE)**	**Jack2**	**Bootstrap (SE)**
Former salina	4	17	20.38 (3.72)	21.50 (4.29)	23.17	19.15 (2.75)
Playa	13	9	9.23 (0.69)	9.92 (0.92)	9.22	9.58 (0.88)
Salina	4	31	35.54 (3.75)	39.25 (9.42)	41.42	35.13 (6.48)
Urban playa	26	23	40.31 (22.73)	28.77 (3.07)	33.42	25.4 (1.74)

Table 2.5: Average and standard error (SE) water chemistry values for each wetland type per year.

Type	**Year**	**NO_3 (SE) mg/L**	**SO_2 (SE) mg/L**	**PO_4 (SE) mg/L**	**NO_2 (SE) mg/L**	**NH_3 (SE) mg/L**	**NTU (SE)**	**% Basin Fill (SE)**	**% Edge Veg. (SE)**	**NaCl (SE) ppm**	**Water Temp (SE) °C**	**pH (SE)**	**Cond. (SE) µS/cm**	**TDS (SE) ppm**
Salina*, n = 5	2020	1.88 (0.68)	70 (0)	0.70 (0.43)	0.01 (0)	0.17 (0.07)	136.09 (88.13)	55.20 (15.93)	69.00 (69)	4.23 (1.21)	27.26 (0.54)	7.66 (0.16)	8.59 (2.43)	6.02 (1.71)
Salina*, n = 7	2021	2.76 (0.67)	70 (0)	1.57 (0.38)	0.01 (0)	0.04 (0.01)	7.70 (2.81)	82.14 (7.06)	72.14 (9.47)	142.71 (132.13)	26.27 (0.79)	7.87 (0.13)	1713.43 (264.39)	242.72 (40.1)
Urban playa, n = 24	2020	2.19 (1.20)	26.60 (4.66)	1.27 (0.56)	0.03 (0.02)	0.25 (0.07)	83.46 (39.65)	83.96 (4.26)	13.96 (5.77)	174.97 (28.16)	25.84 (0.38)	8.13 (0.10)	343.77 (57.88)	242.72 (40.10)
Urban playa, n = 26	2021	1.08 (0.18)	15.69 (2.91)	1.64 (0.21)	0.01 (0)	0.09 (0.01)	30.22 (4.18)	95.69 (2.56)	21.73 (5.23)	109.55 (11.31)	26.07 (0.43)	8.11 (0.10)	226.62 (24.63)	156.16 (15.76)
Former salina, n = 5	2021	0.96 (0.21)	59.60 (9.3)	0.44 (0.23)	0.04 (0.01)	0.02 (0.01)	5.70 (1.88)	75 (9.17)	6.20 (3.51)	668.84 (186.93)	22.96 (4.75)	8.27 (0.11)	1341.96 (373.06)	697.24 (178.51)
Playa, n = 13	2021	3.06 (1.09)	10.46 (5.13)	2.15 (0.18)	0.01 (0)	0.20 (0.05)	193.48 (84.56)	93.08 (6.65)	91.92 (3.2)	129.60 (9.94)	25.55 (0.89)	8.07 (0.49)	270.41 (20.58)	186.09 (14.21)

*Note: Water samples collected from salinas were taken from the freshwater spring.

Table 2.6: MANOVA results for variables measured at 24 urban playa and 2 salina sites visited in summer 2020, and 26 urban playa, 7 salinas, 5 former salinas, and 13 playa sites visited in summer 2021; entries in bold are significantly different between the two wetland types (in 2020) and across wetland types (in 2021) at the alpha = 0.05 level. Variables omitted from analyses are noted with '--'.

Variable	**2020**		**2021**	
	F value	**P value**	**F value**	**P value**
NO_3	0.347	0.6	1.325	0.3
SO_2	**6.859**	**0.01**	**19.170**	**0.000**
PO_4	0.234	0.6	**3.348**	**0.03**
NO_2	0.207	0.7	**6.669**	**0.001**
Turbidity	0.137	0.7	--	--
% Basin fill	**6.294**	**0.02**	1.393	0.3
% Edge vegetation	**13.075**	**0.001**	**33.832**	**0.000**
Salinity	**184.650**	**0.000**	**9.224**	**0.000**
Water temperature	2.493	0.1	2.544	0.07
pH	3.586	0.07	0.232	0.9
Conductivity	--	--	0.545	0.7

Table 2.7: Post-hoc Tukey's test results for abiotic variables compared across wetland types in 2021 (fsalina = former salina); comparisons shaded and in bold are significantly different at the alpha = 0.05 level (P = playa, S = salina, FS = former salina, U = urban playa).

Variable	Comparison	p-value	Direction of Difference
Sulfate	**playa-fsalina**	**0.000**	**Neg, P<FS**
	salina-fsalina	0.987	
	urban-fsalina	**0.001**	**Neg, U<FS**
	salina-playa	**0.000**	**Pos, S>P**
	urban-playa	**0.008**	**Pos, U>P**
	urban-salina	**0.000**	**Neg, U<S**
Phosphate	**playa-fsalina**	**0.017**	**Pos, P>FS**
	salina-fsalina	0.451	
	urban-fsalina	0.136	
	salina-playa	0.596	
	urban-playa	0.327	
	urban-salina	0.995	
Nitrite	**playa-fsalina**	**0.001**	**Neg, P<FS**
	salina-fsalina	**0.008**	**Neg, S<FS**
	urban-fsalina	**0.001**	**Neg, U<FS**
	salina-playa	1.000	
	urban-playa	0.994	
	urban-salina	0.998	
% Edge vegetation	**playa-fsalina**	**0.000**	**Pos, P>FS**
	salina-fsalina	**0.001**	**Pos, S>FS**
	urban-fsalina	0.670	
	salina-playa	0.516	
	urban-playa	**0.000**	**Neg, U<P**
	urban-salina	**0.000**	**Neg, U<S**
Salinity	playa-fsalina	0.712	
	salina-fsalina	**0.000**	**Neg, S<FS**
	urban-fsalina	0.459	
	salina-playa	**0.000**	**Neg, S<P**
	urban-playa	0.948	
	urban-salina	**0.000**	**Pos, U>S**

Table 2.8: Localized conditions (average coverage of edge vegetation, presence of floating macrophytes and trees (Y = yes, N = no) at field sites. "--" and shaded cells indicate a site that was never sampled as it never held water during our sampling visits.

Site name	**Average % edge vegetation coverage**	**Macrophyte presence**	**Tree presence**
Abernathy	31.2	N	Y
Buster Long	7.5	N	Y
Clifford Andrews	11.3	N	Y
Caudle	17.5	N	Y
Clapp	83.8	Y	Y
Dupree	13.8	N	Y
E. Givens	31.3	Y	Y
Earl Crow	23.8	N	Y
Charles A. Guy	25	Y	Y
Higginbotham	1.3	N	Y
Hoel	0	N	Y
Jack Stevens	17.5	N	Y
Jan Jennings	35	N	Y
Leftwich	21.3	N	Y
Leroy Elmore	2.5	N	Y
Lobo	0.7	N	Y
Mahon	93.8	N	Y
Maxey	11.3	N	Y
McAlister	37.5	N	Y
McCullough	17.5	Y	N
Muleshoe City Park	11.8	Y	N
OW Ribble	0.5	N	Y
Remington	1.8	N	Y
Travis Trussel	35	Y	Y
SE Amarillo	82.5	N	Y
SE Canyon	55	N	Y
Tahoka	49.5		
Bean	100	Y	Y
Mackenzie	32.5	Y	Y
Mound	65	Y	Y
Coyote	--		
Baileyboro	--		
Goose	10	N	Y
Paul's	0.5	Y	Y
Rich	15	Y	Y

Table 2.8: Continued

Site name	**Average % edge vegetation coverage**	**Macrophyte presence**	**Tree presence**
Shafter	--		
White	0	N	Y
B1	100	N	N
B2	100	N	N
Castro1	100	N	N
Cotton Center	100	N	N
D2C	--		
Dbirk	100	N	N
Dport	--		
Floyd1	100	N	Y
Floyd2	100	N	N
Mil	--		
Mur	65	N	Y
TTU Range Barn	100	N	Y
S26G	--		
S1	80	N	Y
S2	85	N	Y
S3	75	Y	Y
SPart	--		
SW2007_3C	--		
SW2007_4C	--		
Wson	--		
Wten	90	N	Y

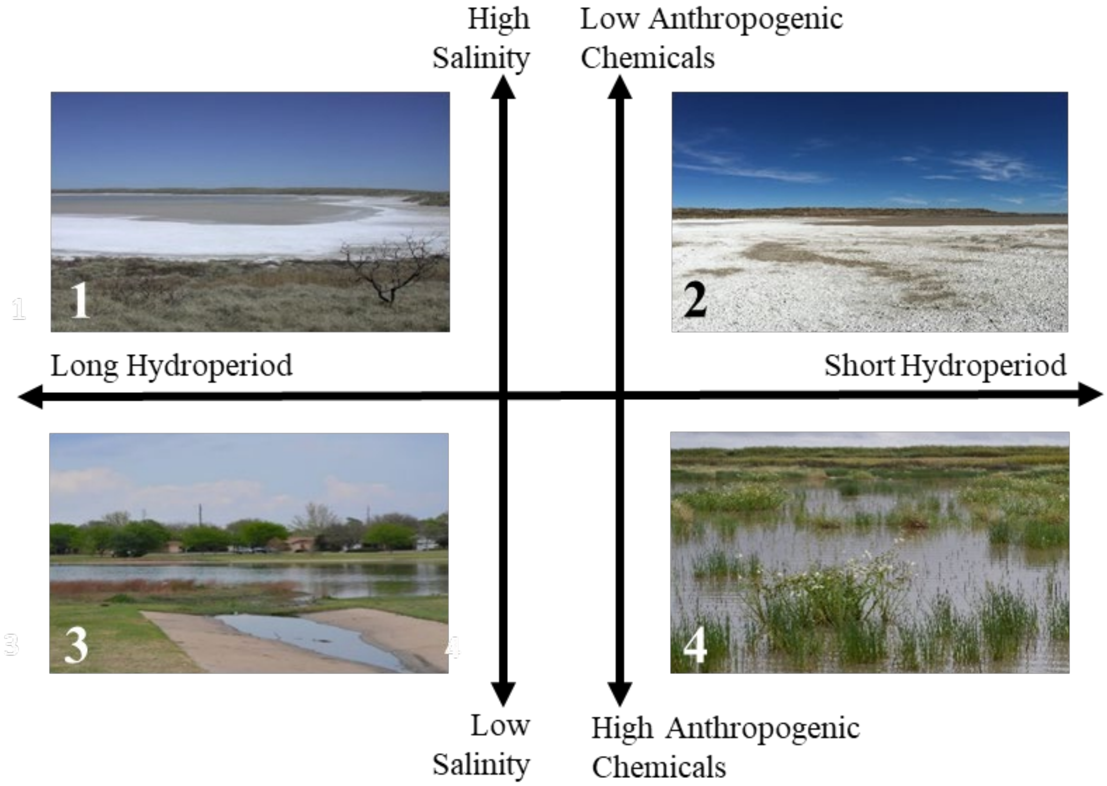

Figure 2.1: We hypothesized that salinas (1), former salinas (2), urban playas (3), and playas (4) will support different odonate assemblages according to gradients in hydroperiod, salinity, and water chemistry. Hydroperiod hypothesis: we expect wetlands with inherently longer hydroperiods (1, 3) to have an assemblage composed of larger-bodied species that require longer hydroperiods for development relative to wetlands with shorter hydroperiods (2, 4). Salinity hypothesis: we expect saline environments (1, 2) to have a unique assemblage composed of salt-tolerant species compared to the assemblages at the more-common freshwater wetlands (3, 4). Water chemistry hypothesis: we expect wetlands surrounded by more anthropogenic influences (3, 4) to feature species more tolerant of chemicals compared to those surrounded by other land uses (1, 2).

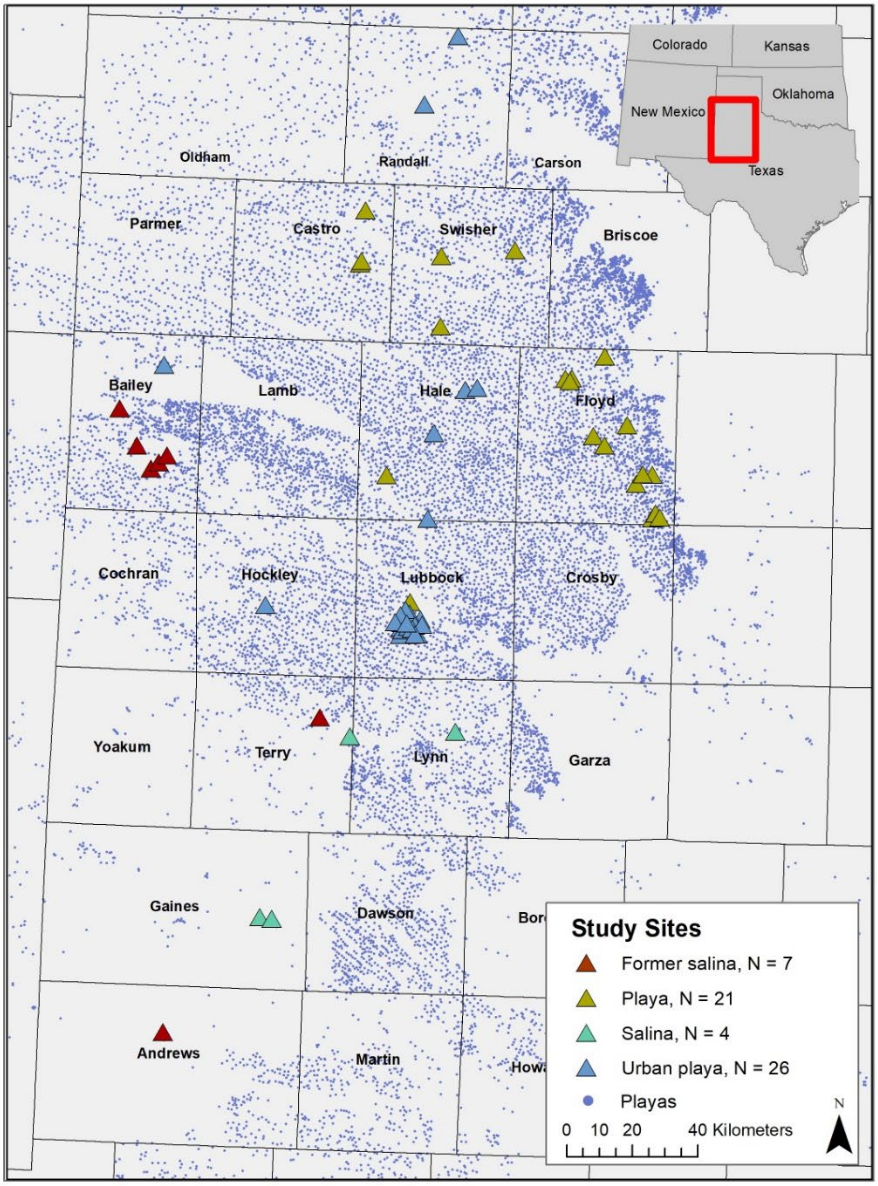

Figure 2.2: Playas are by far the most numerous wetland type in the region (blue points; basemap from Playa Lakes Joint Venture, https://pljv.org/for-habitat-partners/maps-and-data/data-downloads/). Urban playas were the most sampled wetland type (N = 26, blue triangle), followed by playas (N = 21, olive triangle), former salinas (N = 7, burgundy triangle), and salinas (N = 4, sea green triangle). Colors selected from the R package *rcartocolor* (Nowosad, 2018) safe palette for red-green color blindness. Symbols oversized for visualization.

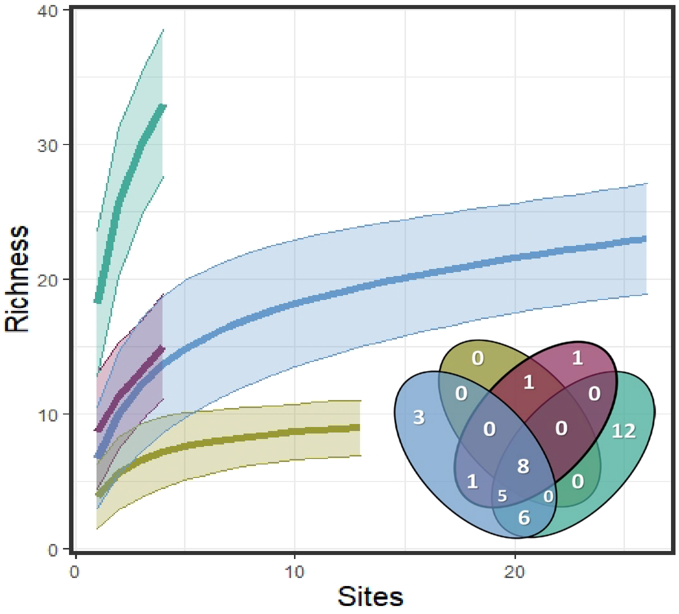

Figure 2.3: Playas (olive) appear to have been adequately sampled to represent regional odonate richness (N = 105; Abbott, 2020) compared to salinas (sea green) and former salinas (burgundy). Eight species were found at all wetland types, 12 were exclusively at salinas, 1 only at former salinas, and 3 only at urban playas. Thirteen species were shared between wetland types. Colors selected from the R package *rcartocolor* (Nowosad, 2018) safe palette for red-green color blindness.

Figure 2.4: Representation of overall results: Odonate assemblages at salinas and urban playas were the most dissimilar in terms of beta diversity whereas salinas and playas had the most similar odonate assemblages.

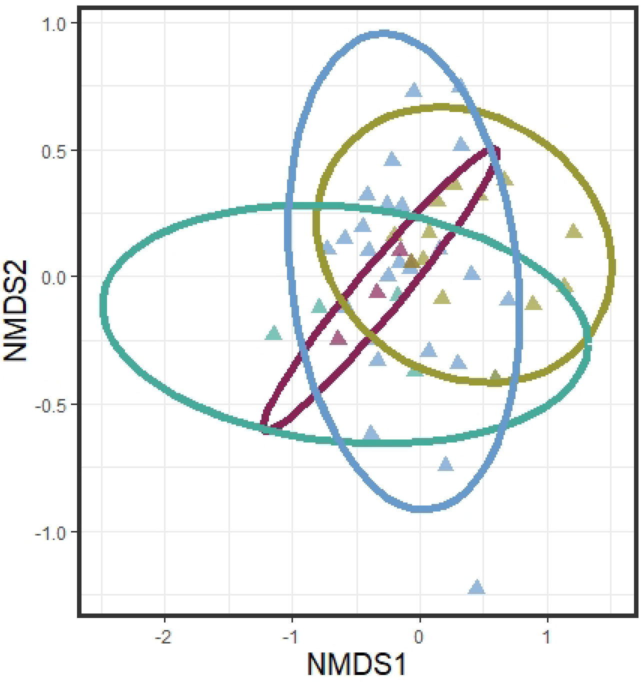

Figure 2.5: Non-metric multidimensional scaling plot with 95% confidence ellipses of the centroids in ordination space of odonate species occurrences at four regional types of wetlands (blue = urban playas; olive = playas; sea green = salinas; burgundy = former salinas). Colors selected from the R package *rcartocolor* (Nowosad, 2018) safe palette for red-green color blindness.

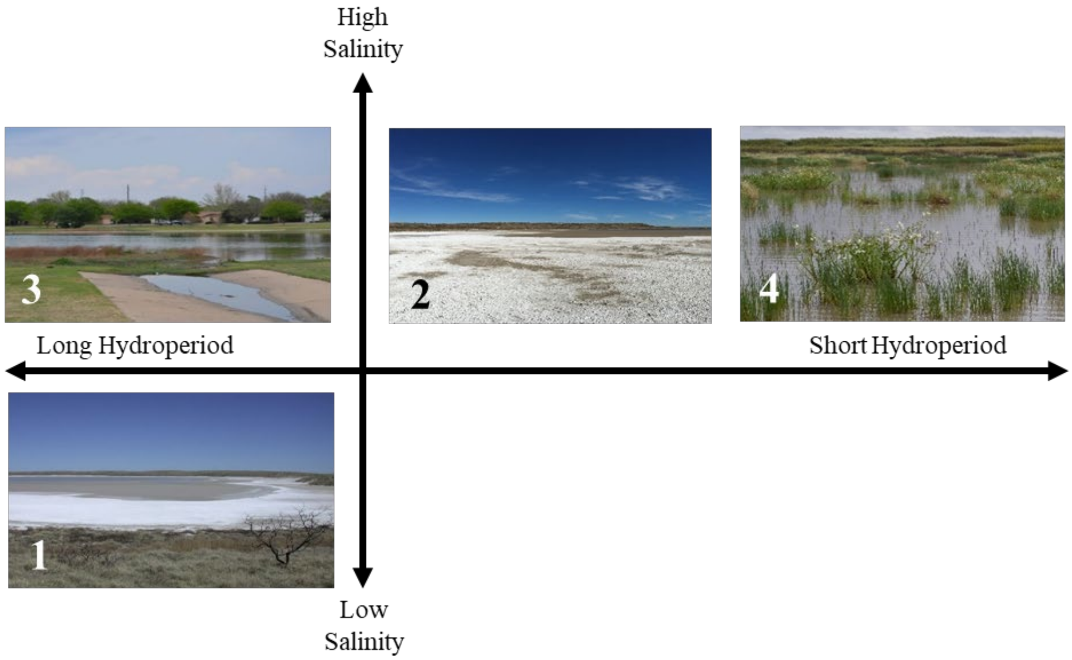

Figure 2.6: Salinity values were contrary to our salinity hypothesis, as salinas (1) had the lowest salinity values compared to urban playas (3), playas (4), and former salinas (2). Consistent with our hydroperiod hypothesis, urban playas (3) and salinas (1) were consistently inundated compared to playas (4) and former salinas (2). Water chemistry variables did not exhibit any patterns by wetland type.

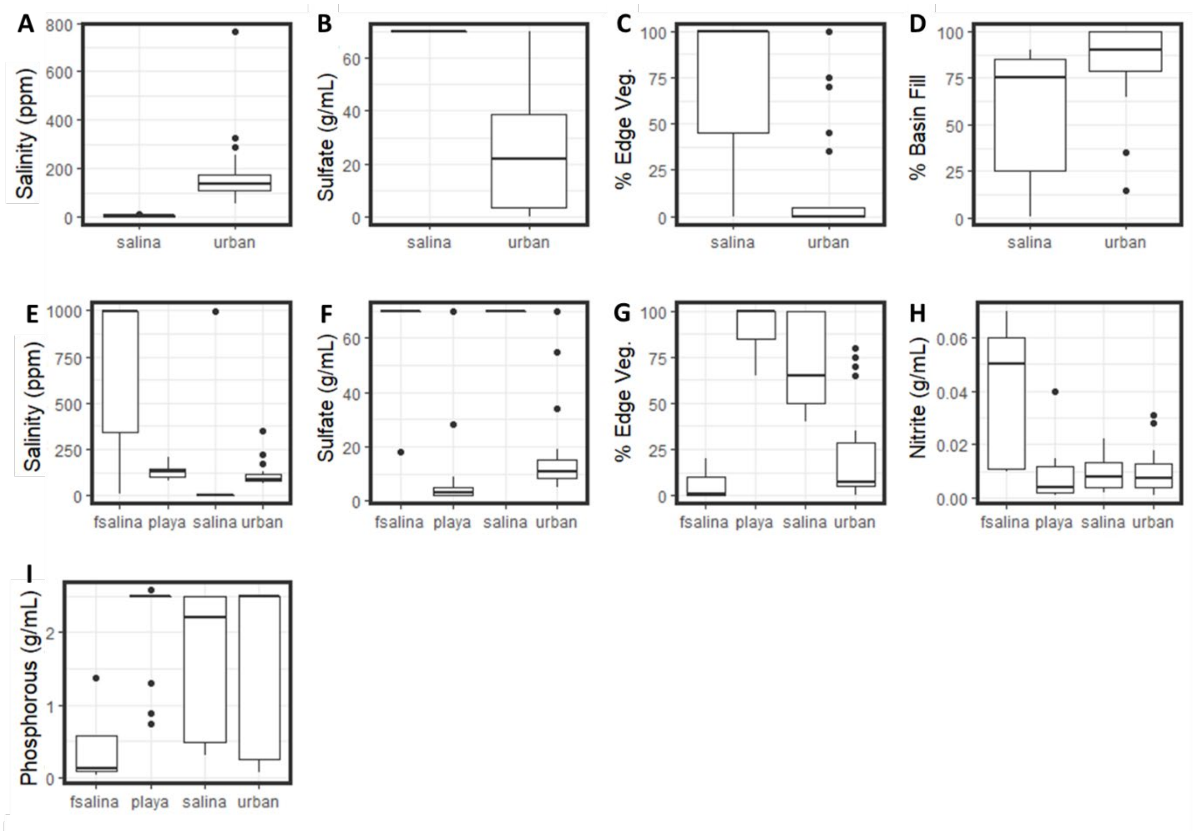

Figure 2.7: Among salinas and urban playas sampled in 2020, salinity, sulfate, % edge vegetation, and % basin fill significantly differed between sites (A-D). Among all wetland types sampled in 2021, salinity, sulfate, % edge vegetation, nitrite, and phosphorous significantly differed between sites (E-I).

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CHAPTER III

CONCLUSION

Odonates are charismatic insects that inspire many with their vast array of colors and impressive flight ability. Their transformational metamorphosis has led to various interpretations of their symbology through the ages (Worldwide Dragonfly Association, 2022). Although they are often regarded highly among nature enthusiasts, odonate conservation can be fraught with difficulty. Documenting odonate diversity to forecast changes in their abundance and distribution, such as on platforms like OdonataCentral and other nonprofit monitoring projects like Dragonfly Pond Watch (Abbott, 2020; Pond Watch, 2022), have been successful at compiling and collecting odonate data nationally and globally, but limits exist on private property where access is not readily available, as I found out firsthand with my research. Protecting odonates means protecting freshwater resources, as they are intrinsically connected to freshwater for all stages of their life cycle. Restoring emergent vegetation for perching and eclosion has been found to be an effective addition to wetlands and ponds to assist odonates (Goertzen and Suhling, 2013; Perron et al., 2021). Monitoring and measuring the human impact to local water bodies from fertilizer and petrochemical runoff over time may help forecast changes in the odonate community (Chang et al., 2007). These are larger issues that must be thought of when considering odonate conservation.

As my research has focused on an area of Texas dominated by agriculture and other human-modified landscapes, I reflect on my personal recommendations for protecting odonates in this region. The Texas Panhandle is overwhelmingly composed of private property with a lack of wetland protections outside of voluntary playa protections such as the CRP Wetlands Restoration Initiative (U.S. Department of Agriculture, 2004). Unlike regional charismatic fauna of interest to Texas Parks and Wildlife and other entities, such as bison and pronghorn, odonates and other invertebrates largely go unnoticed (Texas Parks and Wildlife Department, 2022). The ephemeral nature of most wetlands in this region concerns those interested in migratory animals like waterfowl, excluding freshwater macroinvertebrates like odonates.

Unlike vertebrates, particularly those protected under targeted acts and legislation, odonates are smaller-bodied, with populations that are not commonly monitored. My thesis findings have indicated that several species are found in this region only at salinas. Although I cannot claim that their infrequent detection warrants immediate protection, it is worrisome given the expected climate changes and potential water/wetland human modifications to come. As freshwater availability defines sites where odonates are attracted, particularly in semi-arid areas, this knowledge deficit is troublesome. There is still much to discover, as there were numerous salinas, former salinas, playas, and even urban playas I was unable to survey. My research provides foundational information on how salinity, hydroperiod, and water chemistry affect the odonate assemblages across four types of wetlands in this region.

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