# EVALUATION OF ANURAN PERSISTENCE IN AN URBANIZED DROUGHT-AFFECTED SETTING IN THE SOUTHERN HIGH PLAINS

By

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# CHAPTER I

## INTRODUCTION AND JUSTIFICATION

### URBANIZATION AND AMPHIBIANS

The extent of urbanization has increased dramatically over the past century, causing significant alterations to existing physicochemical and ecological conditions of the landscape. This has induced alarming declines of amphibians (Beebee and Griffiths 2005; Cushman 2006; Hamer and McDonnell 2008; Harper et al. 2008; Southerland and Stranko 2008; Bury 2008; Mitchell and Brown 2008), which currently register the highest rates of disappearance among all vertebrate groups; over a third of amphibian species worldwide are threatened by extinction (Stuart et al. 2004; Hamer and McDonnell 2008). This decline has been attributed to a variety of factors such as limited vagility of amphibians, road mortality, habitat loss, degradation and fragmentation, stocking of fish, spread of invasive species, climate change, and diseases (Sinsch 1990; Blaustein et al. 1994; Pearl et al. 2005; Mitchell and Brown 2008), all of which are synergistic with urbanization. In light of the fact that urban habitats are the fastest growing habitat type globally, mitigating urban amphibian declines is crucial to amphibian conservation efforts. However amphibians represent the least studied taxa in urbanized settings (Pickett et al. 2008).

Pond-breeding amphibians are particularly vulnerable to urban development because of their reliance on both aquatic as well as terrestrial resources during different stages of their biphasic life-cycle. At larger scales, roads and impervious surfaces decrease amphibian population viability and persistence in four ways: (1) habitat loss (2) increased mortality (3) resource inaccessibility, and (4) population subdivision (Jaeger et al. 2005). At smaller scales, factors such as hydroperiod alteration (e.g., wetland excavation to increase stormwater retention), exposure to aquatic contaminants, and presence of fish and introduced species can impact juvenile recruitment and breeding site-suitability which may result in local extinctions and altered habitat distribution patterns (Wellborn et al. 1996; Hecnar and M'Closkey 1997; Skelly 2001; Babbitt et al. 2003; Van Buskirk 2003; Doubledee et al. 2003; Taylor et al. 2005; Snodgrass et al. 2008a, 2008b).

Hence, amphibians respond to habitat alterations on multiple scales. An improved understanding of amphibian populations at various scales (.i.e., lake and landscape scales) in response to anthropogenic disturbances is thus critical to the establishment of effective conservation strategies within the realm of continuously expanding urban landscapes.

## JUSTIFICATION FOR STUDY

Highly ephemeral 'playa' wetlands constitute the central hydrological feature in the Southern High Plains with nearly 20,000 wetlands in west Texas alone (Haukos and Smith 1994; Johnson et al. 2011). Playas are shallow, circular depressions fed mainly through precipitation and surface runoff. Although constituting less than 2% of the surface area of the Southern High Plains, playa wetlands serve as unique 'biodiversity centers' on this landscape, and primary breeding habitats for amphibians (Haukos and Smith 1994; Johnson et al. 2011). This region has a semi-arid climate with average annual precipitation ranging from 33 – 45 cm, with the months from April through August receiving the bulk of the rainfall (Bolen et al. 1989; Gustavson et al. 1995). However rainfall occurs as isolated thunderstorms causing playas to have unpredictable hydroperiods with extended dry periods (Haukos and Smith 2003).

Thirteen species of amphibians have been reported to occur in the playas of the Southern High Plains (Anderson et al. 1999; Haukos and Smith 2003), and most have specialized adaptations such as rapid breeding and metamorphosis, and explosive breeding behavior (Bragg 1940; Creusere and Whitford 1976; Dimmitt and Ruibal 1980). Pelobatids (*Spea bombifrons, Spea multiplicata, Scaphiopus couchii*) have very short larval periods and can increase/decrease the rate of metamorphosis in response to resource availability, length of hydroperiod and other environmental stressors (minimum reported larval period is a week for *Scaphiopus couchii* (Morey 2005a) and 12 – 13 days for *Spea spp.* (Farrar and Hey 2005; Morey 2005b)), and can better cope with desiccation caused by drying of breeding pools. Bufonids such as Great Plains Toads *(Anaxyrus cognatus)* and Texas Toads *(Anaxryrus speciosus)* have a larval period anywhere between 17 – 60 days (Graves and Krupa 2005; Dayton and Painter 2005), while other species such as Spotted Chorus Frogs *(Pseudacris clarkii),* Great Plains Narrow-Mouthed Toads *(Gastrophyne olivacea)*, and Plains Leopard Frogs *(Lithobates blairi)* have longer larval periods ≥ 28 days (Wright and Wright 1949; Smith 1956; Degenhardt et al. 1996; Sredl 2005; Sredl and Field 2005; Smith and Keinath 2005). Presence of water, at least long enough to complete metamorphosis is crucial for juvenile recruitment, as is the presence of wetlands with a range of hydroperiods to accommodate amphibian biodiversity in the region.

The Southern High Plains has a history of anthropogenic disturbance; herbivory from the 1800s combined with decades of agriculture have transformed extensive shortgrass prairie grasslands into one of the most intensively cultivated zones of the western hemisphere (Bolen et al. 1989). Cities emerged with agriculture-generated income and subsequent industrialization, the largest of which is Lubbock, Texas (Graves 1961). Playa wetlands in Lubbock have been extensively modified for agriculture, stormwater drainage, and construction of roads, buildings, neighborhoods, and recreational spaces. Since they no longer function as ephemeral playa wetlands and I will refer to them as 'lakes' hereafter in this chapter.

Amphibian studies in cropland and grassland systems of the Southern High Plains have evaluated some aspects of habitat use and observed positive correlations between amphibian occurrences and emergent vegetation cover and water quality (Anderson et al. 1999), hydroperiod (Venne et al. 2012), vulnerability to effects of agricultural chemicals (Venne et al. 2008; Dinehart et al. 2010) and impacts of grassland cultivation on post-metamorphic size, frequency of occurrence, and community richness of amphibians in playa wetlands (Gray et al. 2004; Gray and Smith 2005; Venne et al. 2012). However, several gaps in knowledge still exist regarding amphibian life-histories and habitat use in this area. In particular, amphibian ecology in urban centers of the Southern High Plains has never been addressed. No prior data exist regarding amphibian use of Lubbock's lakes and little concern has been demonstrated for amphibians in the layout and design of urban environments. However, long term residents of the city have noticed declines in amphibian populations, in the absence of characteristic breeding calls following thunderstorms. Amphibian conservation strategies in the city can only be implemented with baseline inventory of amphibians utilizing city wetlands and understanding factors driving observed distribution patterns as a function of various landscape and wetland metrics.

Lubbock, a mid-size size city of 233, 740 (U.S. Census Bureau 2011) is representative of a majority of cities in North America where urban ecology studies are still a rarity, as opposed to large metropolitan centers which are typically the focus of urban ecology research (Parris 2006; Simon et al. 2009; Smallbone et al. 2011). Hence, I aimed at addressing the lack of knowledge in urban amphibian ecology in Lubbock with the following objectives;

- Gather fundamental baseline data including preliminary inventory of amphibian species occurrences and community assemblages using Lubbock's lakes
- Evaluate site-specific and landscape-scale factors that affect amphibian distribution patterns in the city

The Southern High Plains experienced one of its worst droughts in the year 2011, breaking several records for extreme heat and low precipitation (NOAA 2011, 2012a, 2012b). Annual rainfall for Lubbock city in 2011 was 14.9 cm - approximately three times short of its 30-year normal of 48.6 cm (NOAA 2011). Although droughts are a recurring characteristic of the climate of the Plains (Woodhouse and Brown 2001), very few studies have reviewed the impacts of drought on xeric amphibian distribution, much less in an urban setting.

With this unique opportunity to monitor amphibian use of Lubbock's lakes during drought conditions, I refined the study objectives to take the drought into perspective as follows:

(1) Gather fundamental baseline regarding amphibian species occurrences and community assemblages occurring in Lubbock's lakes during the drought year of 2011 (Chapters II and III) and the relatively wetter year of 2012 (Chapter III)

(2) Evaluate the importance of site-specific variables such as water quality, and landscape-scale variables such as road density on amphibian species occurrences over the two years (Chapters II and III)

(3) Formulate easily interpretable models that best explain urban amphibian distribution and drought recovery in Lubbock city (Chapter III)

Chapters II and III have been organized as stand-alone chapters for ease of publication purposes, while Chapter IV provides the overall summary, management recommendations and conclusions.

# CHAPTER II

## URBAN AMPHIBIANS OF THE TEXAS PANHANDLE: BASELINE INVENTORY AND HABITAT ASSOCIATIONS IN A DROUGHT YEAR

### ABSTRACT

Habitat loss, degradation and fragmentation due to urbanization are implicated in amphibian declines worldwide. Conservation efforts require sufficient information on resident species and their habitat interactions, but amphibian ecology is previously unstudied in urban centers of the Southern High Plains. Here, I gathered baseline data on amphibian presence, species richness, and habitat preferences at site-specific and landscape scales during a severe drought year in the city of Lubbock, in northwest Texas. Ephemeral playa wetlands are characteristic of this landscape. During urbanization, these have been extensively modified for stormwater drainage, agriculture, and construction of roads, buildings, and neighborhoods. A semi-arid climate with frequent droughts, together with urbanization, could have an adverse effect on resident amphibians. In 2011, I sampled 23 of these urban lakes for amphibian presence, using a combination of audio, visual, and larval surveys. I detected five amphibian species at seven lakes; Texas Toads (*Anaxyrus speciosus)* and Spotted Chorus Frogs (*Pseudacris clarkii)* being most common. I found significant negative effects of road density near the lake on amphibian species presence and richness. I also detected significant negative effects of high values of basic pH on amphibian species richness. These data can be used for prioritizing lakes

for amphibian conservation strategies, to monitor ecosystem function in urban wetlands, and to guide future development and restoration efforts.

## INTRODUCTION

Globally, amphibians are one of the fastest disappearing vertebrate groups as a result of factors such as limited vagility, road mortality, habitat loss, degradation and fragmentation, stocking of fish, spread of invasive species, climate change, and diseases (Sinsch 1990; Blaustein et al. 1994; Stuart et al. 2004; Pearl et al. 2005; Cushman 2006). As urban boundaries continue to expand, habitats suitable for amphibian breeding and survival decline. Urban ecology studies and efforts to mitigate urban amphibian declines are usually concentrated in metropolitan centers while smaller urban settlements are largely unstudied (Parris 2006; Smallbone et al. 2011). The impacts of urbanization are less obvious in these areas, and hence they provide a source of valuable information (Smallbone et al. 2011) and management strategies can be implemented earlier in city planning.

In urbanized landscapes, amphibians use stormwater ponds and other constructed waters for breeding (Ostergaard et al. 2008; Mason 2008; Simon et al. 2009). By virtue of their biphasic life-cycle, pond-breeding amphibians are sensitive to alterations in their aquatic breeding habitat as well as surrounding terrestrial uplands. Degradation of urban aquatic habitat by accumulated pollutants is a major concern (Taylor et al. 2005; Snodgrass et al. 2008a, 2008b). Impeded landscape connectivity in the form of roads and neighborhoods pose physical barriers to dispersal, increases risk of mortality, reduces choice of suitable breeding/terrestrial habitat, and thus have negative effects on resident amphibian populations (Fahrig et al. 1995; Lehtinen et al.1999; Carr and Fahrig 2001; Parris 2006; Andrews et al. 2008).

The Southern High Plains region of the United States has undergone vast anthropogenic modification. Decades of herbivory and agriculture have transformed extensive shortgrass prairies into one of the most intensively cultivated zones of the western hemisphere (Bolen et al. 1989). Urbanization started in the late 1800s, and presently Lubbock, in northwest Texas, is the largest city in this region (population of 233, 740: U.S. Census Bureau 2011). Numerous ephemeral ‘playa’ wetlands are the predominant hydrological feature and are centers of biodiversity in this otherwise arid landscape (Haukos and Smith 1994; Johnson et al. 2011). These wetlands, fed by precipitation and surface run-off, also serve as primary breeding habitat for amphibians (Haukos and Smith 1994). Unfortunately, playa wetlands of the Southern High Plains face numerous threats from ongoing landscape modification – altered hydrology, altered floral and faunal communities and increased sedimentation being some of the large concerns (Luo et al. 1997; Haukos and Smith 2003).

Most playa wetlands in Lubbock have been excavated to increase stormwater retention capacity; approximately 30 playas are part of the city’s pipe-connected stormwater drainage networks. Playa wetlands have also been modified to make way for urban development such as roads, building, neighborhoods, and recreational parks. Amphibian ecology has never been studied in urban centers of the Southern High Plains, and little concern has been demonstrated for amphibians in the layout and design of urban environments. Amphibian monitoring in regional urban centers such as Lubbock is virtually nonexistent.

Thirteen species of amphibians occur in the Southern High Plains (Anderson et al. 1999; Gray et al. 2004). They are xeric-adapted and lead highly terrestrial lives returning to aquatic habitats only to breed. Many of these species, have short hatching times and rapid metamorphosis, enabling juvenile recruitment in highly ephemeral lakes (Degenhardt et al. 1996; Warburg 1997; Denver et al. 1998). The climate of the Southern High Plains is semi-arid and droughts are a recurring feature (Woodhouse and Brown 2001). Climate models predict a global increase in extreme weather conditions such as drought which may severely cripple persistence of amphibian populations in the Southern High Plains (Boone et al. 2003; Seager et al. 2007; Lawler et al. 2009; Seager and Vecchi 2010). In 2011, Lubbock experienced the worst one-year drought documented in its history, having received only 14.9 cm of rainfall – less than a third of its 30-year normal which stands at 48.6 cm (NOAA 2011, 2012a, 2012b).

The objectives of my study were, thus, two-fold: (a) lay the foundation for amphibian monitoring within the city of Lubbock by gathering information on urban amphibian composition in a drought year, and (b) evaluate the influence of site-specific characters and landscape-scale characters on amphibian presence and species richness under these extreme conditions.

# MATERIALS AND METHODS

## *Study Sites*

Since most of the playas within Lubbock city have been modified in some capacity, I will hereafter refer to them simply as 'lakes'. I chose 26 lakes for amphibian surveys (within Lubbock's city limits) from the city's stormwater maps. However, only 23 lakes were inundated at least once in 2011 and these were surveyed for amphibians from March to October 2011 (Fig. 2.1). All lakes within the study set are impacted by surface water runoff from surrounding roads, neighborhoods and establishments.

## *Amphibian sampling*

I assessed amphibian community composition using a combination of call surveys, visual encounter surveys and larval dipnet surveys (Fellers et al. 1994; Houlahan and Findlay 2003) to increase detection probability. Beginning the night after rain events, starting half an hour after sunset, I performed 5 min call surveys at each lake and identified species calling (Weir and Mossman 2005). The survey was preceded by a null period of 1 min to allow any disturbance caused by my approach to die down. Call surveys were terminated by 0300 hours (Anderson et al. 1999). Following the call survey, I performed a 20 min visual encounter survey (Knutson et al. 2004) by walking along the lake shoreline and wading in the shallow regions, and identifying amphibians seen. For two consecutive weeks following call surveys, I sampled lakes for amphibian larvae using a pipe-sampler (a modified bottomless trash-can), and a dipnet. Tadpoles have been observed to occur mostly along depths of 1 m or less (O'Hara and Blaustein 1981).

Hence, the sampling locations were < 1 m deep and the number of samples was standardized based on lake area. I took eight samples for lake areas < 5 acres, 12 samples for lakes between 5 - 10 acres, and 16 samples for lakes > 10 acres. Samples were at least 5 m apart to ensure independence (Korfel et al. 2009). At a sample site, the pipe sampler was thrown in the littoral zone of the lake, and pushed into the substrate to seal the sample space. A dipnet was then used to sweep tadpoles from the sample space. Sweeps were carried out until I had ten consecutive null sweeps (sweeps without catching any tadpoles) (Werner et al. 2007). All tadpoles caught were identified (Altig 1970) and released. The combination of three survey types increased detection probability and confirmed amphibian species presence.

*Measurement of site-specific variables*

I recorded water quality variables – pH and conductivity (Sadinski and Dunson 1992; Fominykh 2008) – during call surveys and tadpole surveys using a hand-held YSI 63 meter (YSI Inc., Yellow Springs, Ohio, USA). Water quality was measured 1 m from the shoreline at a depth of 5 cm (Pilliod and Peterson 2001) – at one location for call surveys and at two diametrically opposite locations during tadpole surveys. I also recorded emergent vegetation cover, fish presence, hydroperiod and lake area (Hecnar and M'Closkey 1997; Anderson et al. 1999; Burne and Griffin 2005; Hartel et al. 2011). Percent cover of within-lake emergent vegetation was visually estimated, as was presence of fish (presence/absence). Hydroperiod was quantified on an ordinal scale based on repeated lake visits:'1' if lakes dried up within 2 weeks, '2' if lakes dried up within 3 months (but retained water longer than 2 weeks), and ‘3’ if lakes held water for more than 3 months. Lake areas were estimated from the website of the U.S. Fish and Wildlife Service’s National Wetland Inventory (NWI) and ground-truthed during field surveys. Because connectivity may allow fish and amphibians to migrate between lakes, I also categorized lakes as ‘1’ if they were connected via a pipeline as part of the city’s storm water drainage projects, and ‘0’ for those lakes that were not part of the drainage projects.

### *Measurement of landscape-scale variables*

In my study, I defined ‘landscape’ scale as the region within 500 m radius of the lake and these regions were non-overlapping. Terrestrial amphibian core habitat typically ranges from 159 - 290 m from the edge of the aquatic site (Semlitsch and Bodie 2003). Spadefoots showed greater levels of nestedness in anthropogenically disturbed (cropland) playa wetlands (Gray et al. 2004). Being highly terrestrial, Bufonids have been documented making extensive migrations (Sinsch 1987; Miaud et al. 2000). However since amphibian movements are regulated by rainfall, availability of moisture and refugia, danger of desiccation and predator avoidance, very few individuals of the population actually travel such extreme distances (Forester et al. 2006; Semlitsch 2008) and most activities occur within 200 m of the lake edge (Forester et al. 2006). Mean migration distance was reported to be 334 m for Ranids, 279 m for Bufonids and 237 m for Hylids (Lemckert 2004). Hence, I assumed that landscape characteristics at spatial scale of 500 m would be adequate to predict amphibian occurrences.

I obtained land cover information from the Texas Natural Resources Information System (TNRIS) - 2010 NAIP 1 m resolution aerial imagery for Lubbock quadrangles; and the Lubbock city website - GIS layers for roads, lakes and city limits. I used ArcGIS10 (ESRI, Redlands, California, USA) to calculate road density and nearest wetland distance. Since area of the 'landscape' – area inside 500 m of the lake's maximum extent – differed among lakes, road density was expressed as length of road by landscape area to standardize the variable. Nearest wetland distance was the edge-to-edge distance from the focal lake to the next nearest lake. Percent cover of impervious area was computed using remote sensing software ERDAS IMAGINE (Intergraph Corporation part of Hexagon Group), and was calculated as the number of pixels occupied by paved surface divided by total number of pixels occupied by the lake and buffer. Age of development surrounding the lake was estimated on an ordinal scale in 10 year increments; '1' being the oldest development (developed in the 1950s) and '6' being the most recently developed (developed in the 2000s); using information from the site of the Lubbock Central Appraisal District (Lubbock Central Appraisal District 2012). Most of Lubbock's 'urban city' roughly falls in the semi-circular area – referred as the 'loop' hereafter – bounded by the loop 289, and I-27 north (Fig. 2.1). This region forms the urban core and has been developed much earlier in the city's history, while regions outside the loop have recently undergone/are currently undergoing urban development. Hence, lakes within the loop have felt the effects of fragmentation and urbanization for a longer time period and were designated a location class '1' and those outside, '0'.

### *Statistical analysis*

I compared abiotic attributes (both site-specific and landscape-scale) with non-parametric Wilcoxon-Mann-Whitney tests of lake categories based on general city location (in/out of the loop), amphibian presence (presence/absence) and connectivity to drainage project (connected/unconnected). Since I had a small sample size (n = 23), of which amphibians occurred in very few lakes, I used basic univariate logistic regressions to test predictive ability of each abiotic variable for both amphibian presence, species richness, and species-specific presence. All analyses were performed using SAS Ver. 9.3 (SAS Institute Inc., Cary, North Carolina, USA).

## RESULTS

Of the 13 species present in this region, I detected five species of anurans during surveys in 2011 and at only seven lakes of the 23 (Fig. 2.2; Table 2.1). Spotted Chorus Frogs (*Pseudacris clarkii)* and Texas Toads (*Anaxyrus speciosus)* were detected most often - at three and five sites respectively. I also detected Great Plains Narrow-Mouthed Toads (*Gastrophryne olivacea*), Plains Spadefoots (*Spea bombifrons)* and American Bullfrogs (*Lithobates catesbeiana),* a species non-native to the region, at one site each. Over five rain events during the study period, I observed very few breeding aggregations - most of it occurred following a heavy thunderstorm in August with rainfall highs of 7.2 cm in certain areas of the city. Larval amphibians occurred only in four of the lakes. See Table 2.2 for observed species assemblages.

Lakes with amphibian detections were surrounded by significantly lower road densities and impervious area covers when compared to lakes without amphibian detections (Fig. 2.3; $P \leq 0.05$, Wilcoxon-Mann-Whitney test). Of the seven lakes with amphibian detections, only two lay within the loop, while the rest were located outside the loop (Fig. 2.2). Road densities and percent covers of impervious surface were greater near lakes located within the loop (Fig. 2.4; $P \leq 0.05$, Wilcoxon-Mann-Whitney test). Lakes within the loop were also characterized by less emergent vegetation ($P = 0.01$, Wilcoxon-Mann-Whitney test) and were surrounded by older developments ($P < 0.001$; Wilcoxon-Mann-Whitney test).

Logistic regressions further confirmed the above results. Road density and impervious area surface had significant negative effects on amphibian presence (Fig. 2.5; $P \leq 0.05$, Logistic regression). Road density also had a significant negative effect on amphibian species richness ($P = 0.05$; Ordered logistic regression). pH values tending towards neutral values strongly favored amphibian species richness ($P = 0.05$; Logistic regression). See Tables 2.3 and 2.4 for further information on predictor values.

Great Plains Narrow-Mouthed Toads and Spotted Chorus Frogs were both observed calling from fishless lakes with greater cover of thin-stemmed vegetation and breeding displays occurred while holding on to thin stems or from atop wooden debris. These lakes were characterized by relatively neutral pH ranging from 7.0 – 7.3, conductivities ranging from 182.89 – 392.48 µS, and were located in areas developed recently (from the 1980s onward) outside the urban loop. They were also surrounded by lower road densities ranging from 3652.17 to 6644.74 m/km$^2$ (Table 2.4). A lone Plains Spadefoot was also heard vocalizing at one of these lakes. Texas Toads were observed in both fish-filled (two lakes) as well as fishless lakes (three lakes) both within and outside the urban loop. Larger lakes supported Texas Toad presence ($P = 0.05$, Logistic regression). Non-native American Bullfrogs were recorded at one within-loop fish-filled lake characterized by high cover of cattails.

## Discussion

The spring and summer of 2011 were the driest and hottest ever documented in Lubbock, with record number of days over 37.8 °C (NOAA 2011, 2012a, 2012b). As the drought progressed, water levels fell drastically in all the lakes, with many drying entirely even before the start of summer. While stormwater run-off quickly elevated water levels in the lakes following even small rains, these did not last long under persistent high temperatures.

In my study, amphibians were detected only at 30.4% of the lakes, most of which were characterized by lower road densities and impervious area covers, and lower values of basic pH (Fig. 2.2, 2.3, 2.4 and 2.5). Five of the seven lakes in which I detected amphibians were located outside the loop, where development is relatively recent and lakes are less modified compared to the majority of lakes within the loop. Seventy-five percent of lakes within the loop are connected to the stormwater drainage system, and have increased predator presence (fish) and very basic pH (average values ranging from 7.52 to 8.63). Fringing vegetation is constantly uprooted and surrounding lawns are mowed regularly by local authorities as part of lake maintenance procedures. While the occurrence of drought itself may have been responsible for fewer amphibian observations, the data also indicates a negative role of breeding habitat isolation and possibly, degraded water quality.

This study suggests that landscape-scale features, which are indicators of wetland isolation, are as important as site-specific characteristics which define health of breeding habitat. Spotted Chorus Frogs were found in three lakes all located at the city edges, at one of which Great Plains Narrow-Mouthed Toads were also observed. Here average pH levels approached near neutrality and had higher cover of emergent vegetation, as observed previously by Anderson et al. (1999) (Table 2.4). Both species are very small (1.8 - 4.1 cm SVL) (Wright and Wright 1949; Clarke 1984; Stebbins 2003) and have thinner skin than other species in this region (Anderson et al. 1999). They also have prolonged breeding and larval periods relative to Bufonids and Spadefoots (Table 2.5). Hence these species might be more vulnerable to the threats of dehydration and degraded water quality (Stebbins and Cohen 1995; Wellborn et al. 1996; Anderson et al. 1999; Babbitt and Tanner 2000). These species are also highly vulnerable to fish predation (Voris and Bacon, Jr. 1966; Hecnar and M'Closkey 1997; Baber and Babbitt 2003).

Bufonids have larger body size when compared to Hylids or Pelobatids and consequently, greater vagility (Texas Toads: 5.1 – 8.9 cm SVL (Stebbins 2003)). The relative unpalatability/distastefulness of their tadpoles to fish (although not completely unpalatable) has been well documented (Kats et al. 1988; Kurzava and Morin 1998;

Baber and Babbitt 2003). These reasons could be attributed to the wider occurrence of Texas Toads, in both fish-filled and fishless lakes. Bullfrogs, which are unpalatable to most fish (Kats et al. 1988; Hecnar and M'Closkey 1997), were detected at one lake which was characterized by thick cattails and heavy fringing vegetation, thus making it suitable for Bullfrog breeding (Pope 1964). Although we expected to find Bullfrogs at more permanently inundated lakes in the city, the low numbers of detections could be a result of low water levels and a lack of thick emergent/fringing vegetation in the other lakes.

An urban matrix is highly complex with its myriad dispersal obstructions such as roads, paved surfaces, buildings, neighborhoods, and various other anthropogenic components. Low vagilities and dispersal abilities as a result of small body size (Peters 1983) can negatively influence amphibian distribution in complex terrestrial habitat. Small-bodied species of this region such as the spadefoots (Plains Spadefoot: 3.2 – 6.3 cm SVL (Stebbins 2003)), Spotted Chorus Frogs and Great Plains Narrow-Mouthed Toads may display difficulty in moving across urban habitats and show increased nestedness at localized sites (Gray et al. 2004). Road mortalities could decimate native populations when amphibians emerge for breeding, a crucial period of large-scale migration to breeding pools (Fahrig et al. 1995; Forman and Alexander 1998; Carr and Fahrig 2001). In the event of local extinctions, recolonization might be hindered by anthropogenic barriers to amphibian movement.

In my study, impervious area cover within 500 m from the lake was positively correlated with pH ($r_s = 0.50$, $P = 0.02$). Many of the lakes are directly linked to surrounding roads and impervious surfaces by storm drains, likely causing accumulation of heavy metal pollutants and salts from roads and paved surfaces, pesticides and fertilizers for the upkeep of manicured lawns, and other chemical compounds (Hatt et al. 2004; Croteau et al. 2008; Snodgrass et al. 2008a, 2008b). Stormwater ponds have been feared to function as ecological traps because amphibians use them as breeding habitat (Brand and Snodgrass 2009), and this could be especially true in drought years when most other lakes have dried up. Since juvenile recruitment is essential for supporting amphibian source populations in urban areas, there is a need to improve urban breeding habitat quality. Establishing suitable native vegetation communities in and around urban lakes might help offset pollutant accumulation and enhance water quality, while providing suitable sites for oviposition and increase concealment from predators (Castelle et al. 1994; Hecnar and M'Closkey 1997).

Prolonged drought periods may have profound effects on amphibian communities (Babbitt and Tanner 2000). Elevated temperatures for long durations can increase metabolic rates of aestivating amphibians, cause loss of coordination, paralysis or even death (Stebbins and Cohen 1995; Grundy and Storey 1998; Young et al. 2011). In the absence of rain, species that rely on large storms to cue reproduction do not initiate breeding, or entire larval cohorts are lost when ponds dry before the completion of metamorphosis (Warburg 1997). Climate projections predict higher temperatures, faster evaporation rates and more sustained droughts for the Southern High Plains region

(United States Global Change Research Program 2009). Since information about species-specific distribution, dispersal, and persistence is scarce, continuous monitoring of city wetlands are required to understand amphibian population distribution and dynamics. In lieu of the time lag of several decades that exist between changes in urbanization and response in amphibian species occurrence (Lofvenhaft et al. 2004), conservation measures need to be implemented at the earliest to mitigate urban amphibian declines in this region.

**TABLE 2.1.** Amphibians of the Southern High Plains (13 species), and species detected during surveys (in bold) from March to October 2011 in urban lakes in the city of Lubbock, Texas. Amphibians were detected in seven lakes in 2011 (n = 23).

| Species detected | Number of sites found in 2011 |
|---|---|
| **Texas Toad *(Anaxyrus speciosus)*** | **5** |
| Great Plains Toad *(Anaxyrus cognatus)* | 0 |
| Woodhouse's Toad *(Anaxyrus woodhousii)* | 0 |
| Green Toad *(Anaxyrus debilis)* | 0 |
| **Spotted Chorus Frog *(Pseudacris clarkii)*** | **3** |
| **Great Plains Narrow-Mouthed Toad *(Gastrophryne olivacea)*** | **1** |
| **Plains Spadefoot *(Spea bombifrons)*** | **1** |
| New Mexico Spadefoot *(Spea multiplicata)* | 0 |
| Couch's Spadefoot *(Scaphiopus couchii)* | 0 |
| **American Bullfrog *(Lithobates catesbeiana)*** | **1** |
| Plains Leopard Frog *(Lithobates blairi)* | 0 |
| Northern Cricket Frog *(Acris crepitans)* | 0 |
| Barred Tiger Salamander *(Ambystoma tigrinum mavortium)* | 0 |

**TABLE 2.2.** Species assemblages observed during amphibian surveys conducted from March to October 2011 in urban lakes in the city of Lubbock, Texas.

| +Lake | *A. speciosus* | *P. clarkii* | *G. olivacea* | *S. bombifrons* | *L. catesbeiana* |
|---|---|---|---|---|---|
| 13 | | + | *+ | + | |
| 85 | + | + | | | |
| 93 | *+ | | | | |
| 94A | + | | | | |
| 27 | *+ | | | | |
| 21 | *+ | | | | + |
| 132 | | + | | | |

+ Lake ID referred from the city of Lubbock's stormwater maps

* Tadpoles present

**TABLE 2.3.** Variables used to describe anuran breeding site and landscape characteristics for 23 urban lakes in Lubbock, Texas, in 2011.

| Variable | Mean | Min | Max | SE |
|---|---|---|---|---|
| **Site-specific variables** | | | | |
| Mean pH | 7.84 | 7.03 | 8.63 | 0.09 |
| Mean conductivity (µS) | 316.60 | 182.89 | 509.27 | 18.13 |
| Lake area (acres) | 4.97 | 0.07 | 14.26 | 0.96 |
| Emergent vegetation cover (%) | 19.14 | 0.00 | 100.00 | 5.56 |
| *Hydroperiod | - | 1 | 3 | - |
| *Fish presence | - | 0 | 1 | - |
| *Presence of connection (storm water connection) | - | 0 | 1 | - |
| **Landscape-scale variables within 500m from the lake edge** | | | | |
| Road density (m/km²) | 10130.16 | 3652.17 | 17078.03 | 677.35 |
| Impervious area cover (%) | 38.33 | 14.06 | 69.25 | 2.86 |
| Nearest wetland distance (m) | 668.91 | 149.29 | 1506.37 | 61.20 |
| *Location | - | 0 | 1 | - |
| *Age of neighborhood | - | 1 | 6 | - |

* Categorical or ordinal variables which have no mean or standard deviation.

**TABLE 2.4.** Range of variable values for individual anuran species measured during amphibian surveys from March to October 2011 urban lakes in the city of Lubbock, Texas.

| **Variable** | ***A. speciosus*** | ***P. clarkii*** | ***G. olivacea*** | ***S. bombifrons*** | ***L. catesbeiana*** | ***No detection*** |
|---|---|---|---|---|---|---|
| **No. of sites** | **5** | **3** | **1** | **1** | **1** | **16** |
| Mean pH | 7.25 - 8.22 | 7.03 - 7.25 | 7.12 | 7.12 | 7.52 | 7.30 - 8.63 |
| Mean cond[×] (µS) | 241.08 - 392.48 | 182.89 - 392.48 | 182.89 | 182.89 | 251.92 | 203.60 - 509.27 |
| Lake area (acres) | 2.15 – 17.18 | 0.07 - 3.11 | 0.59 | 0.59 | 17.18 | 0.07 - 11.91 |
| Emveg[××] (%) | 0.88 - 50 | 20 - 50 | 45 | 45 | 0.88 | 0.00 - 100.00 |
| Hp[φ] | 2.00 - 3.00 | 1.00 - 2.00 | 2.00 | 2.00 | 2.00 | 1.00 - 3.00 |
| Fish presence | 0.00 - 1.00 | 0.00 - 0.00 | 0.00 | 0.00 | 1.00 | 0.00 -1.00 |
| Connec[φφ] | 0.00 - 1.00 | 0.00 - 1.00 | 0.00 | 0.00 | 1.00 | 0.00 -1.00 |
| Road ($m/km^2$)[*] | 6644.74 - 12464.04 | 3652.17 - 6644.74 | 6036.75 | 6036.75 | 12464.04 | 6787.70 - 17078.03 |
| Imp[1] (%) | 19.17 - 42.06 | 14.06 - 31.26 | 29.49 | 29.49 | 32.63 | 21.58 - 69.25 |
| Wetdist[11] (m) | 324.25 - 1506.37 | 459.88 - 868.91 | 459.88 | 459.88 | 1506.37 | 149.29 - 1185.83 |
| Loop | 0.00 - 1.00 | 0.00 - 0.00 | 0.00 | 0.00 | 1.00 | 0.00 - 1.00 |
| Age | 2.00 - 6.00 | 4.00 - 6.00 | 4.00 | 4.00 | 2.00 | 1.00 - 6.00 |

[×]conductivity; [××]emergent vegetation cover; [φ]hydroperiod; [φφ]presence of storm water connection; [*]road density; [1]impervious area cover; [11]nearest wetland distance

**TABLE 2.5.** Larval periods for amphibian species of the Southern High Plains.

| **Amphibian species** | **Larval period** |
|---|---|
| Couch's Spadefoot *(Scaphiopus couchii)* | 7 – 16 d[1] |
| Plains Spadefoot *(Spea bombifrons)* | 13 – 20 d[2] |
| New Mexico Spadefoot *(Spea multiplicata)* | 12 – 19 d[3] |
| Great Plains Toad *(Anaxyrus cognatus)* | 17 – 45 d[4] |
| Texas Toad *(Anaxyrus speciosus)* | 18 – 60 d[5] |
| Spotted Chorus Frog *(Pseudacris clarkii)* | 30 – 45 d[6] |
| Plains Leopard Frog *(Lithobates blairi)* | ~ 60 d, overwinter[8] |
| Woodhouse's Toad *(Anaxyrus woodhousii)* | 35 – 49 d[9] |
| Great Plains Narrow-Mouthed Toad *(Gastrophryne olivacea)* | 24 – 50 d[10] |
| American Bullfrog *(Lithobates catesbeiana)* | > 90 d, overwinter[11] |
| Green Toad *(Anaxyrus debilis)* | 21 – 25 d[12] |
| Northern Cricket frog *(Acris crepitans)* | 35 – 90 d[13] |

[1]Morey 2005a; [2]Farrar and Hey 2005; [3]Morey 2005b; [4]Graves and Krupa 2005; [5]Dayton and Painter 2005; [6]Sredl 2005; [8]Smith and Keinath 2005; [9]Sullivan 2005; [10]Sredl and Field 2005; [11]Degenhardt et al. 1996; [12]Painter 2005; [13]Gray et al. 2005; Venne et al. 2012.

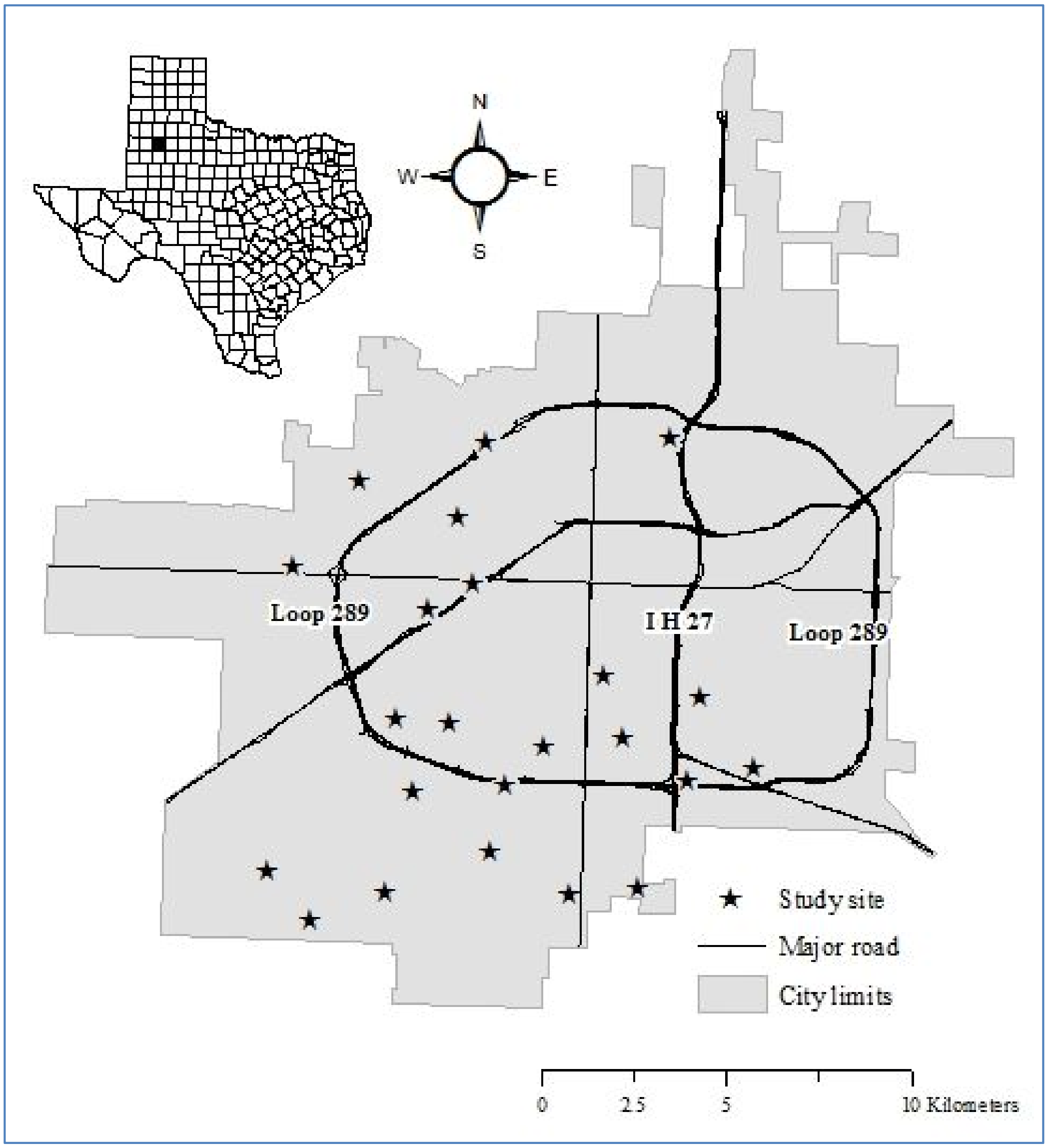


**FIGURE 2.1.** Map showing the 23 lakes surveyed for amphibian species presence from March to October 2011 in the city of Lubbock, Texas. All lakes were within Lubbock's city limits and impacted by stormwater runoff.

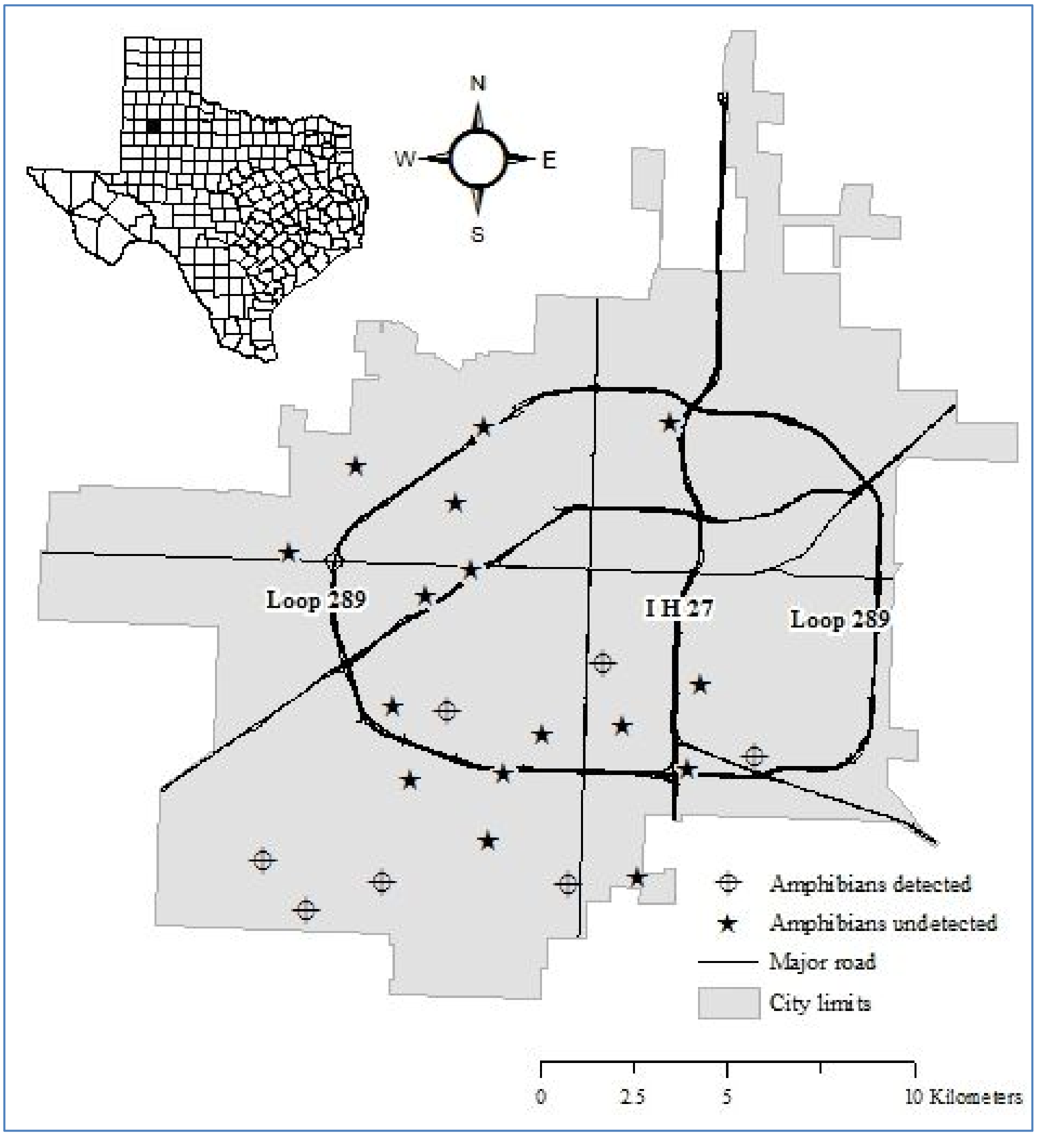


**FIGURE 2.2.** Map showing lakes with and without amphibian detection during surveys from March to October 2011 in the city of Lubbock, Texas. We observed amphibian occurrence at seven lakes during the sampling period.

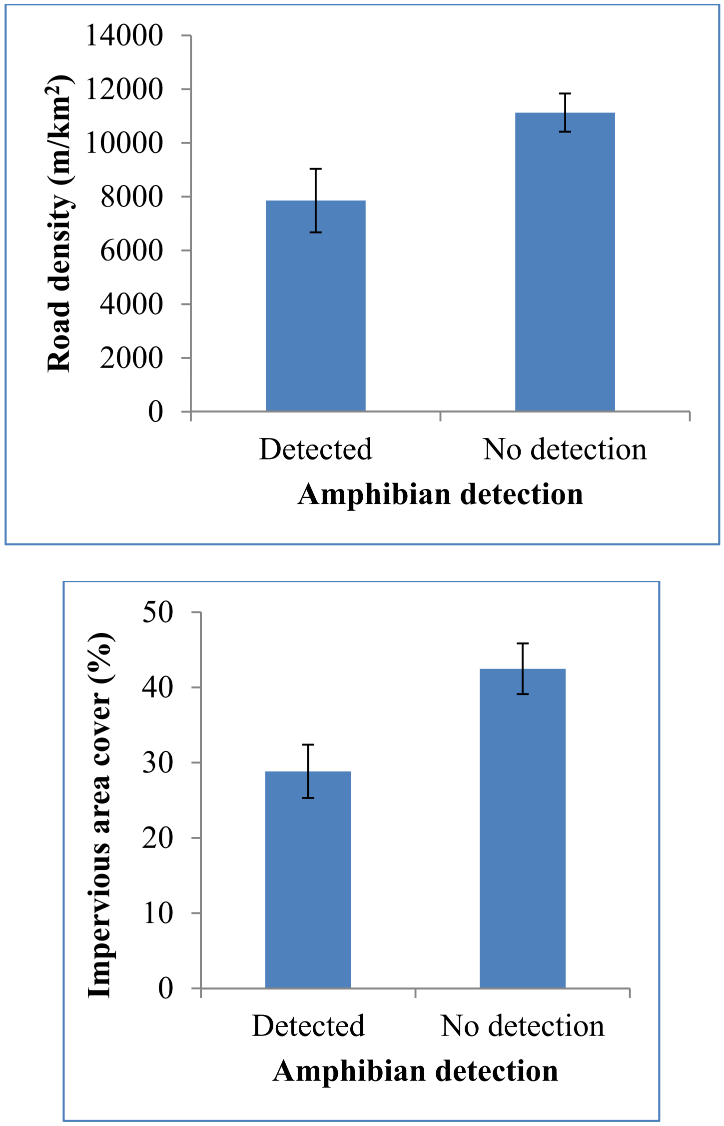


**FIGURE 2.3.** Comparison of landscape-scale factors (within 500 m radius) - road density and impervious area cover - for urban lakes in the city of Lubbock based on amphibian presence (n = 23). Road density and impervious area cover was significantly lower ($P \leq 0.05$) for lakes where amphibians were detected by Wilcoxon-Mann-Whitney non-parametric tests. *Error bars* ± SE.

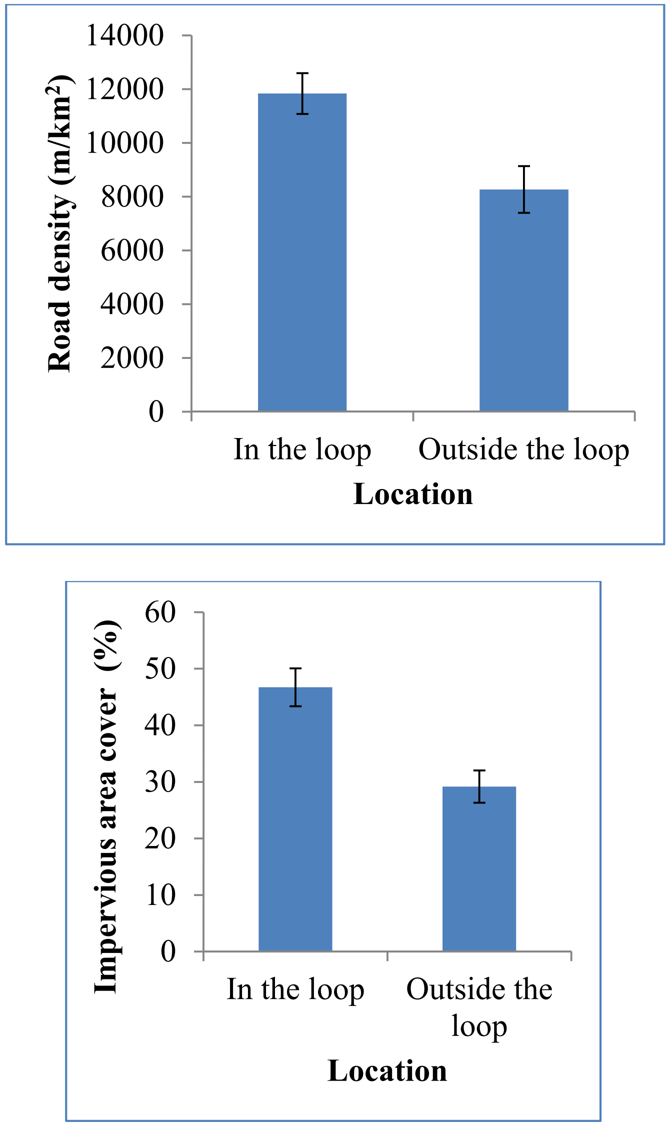


**FIGURE 2.4.** Comparison of landscape-scale factors (within 500 m radius) - road density and impervious area cover - for urban lakes in the city of Lubbock based on lake location within or outside the city loop (n = 23). Road density and impervious area cover was significantly higher ($P \leq 0.05$) for lakes within the loop by Wilcoxon-Mann-Whitney non-parametric tests. *Error bars* ± SE.

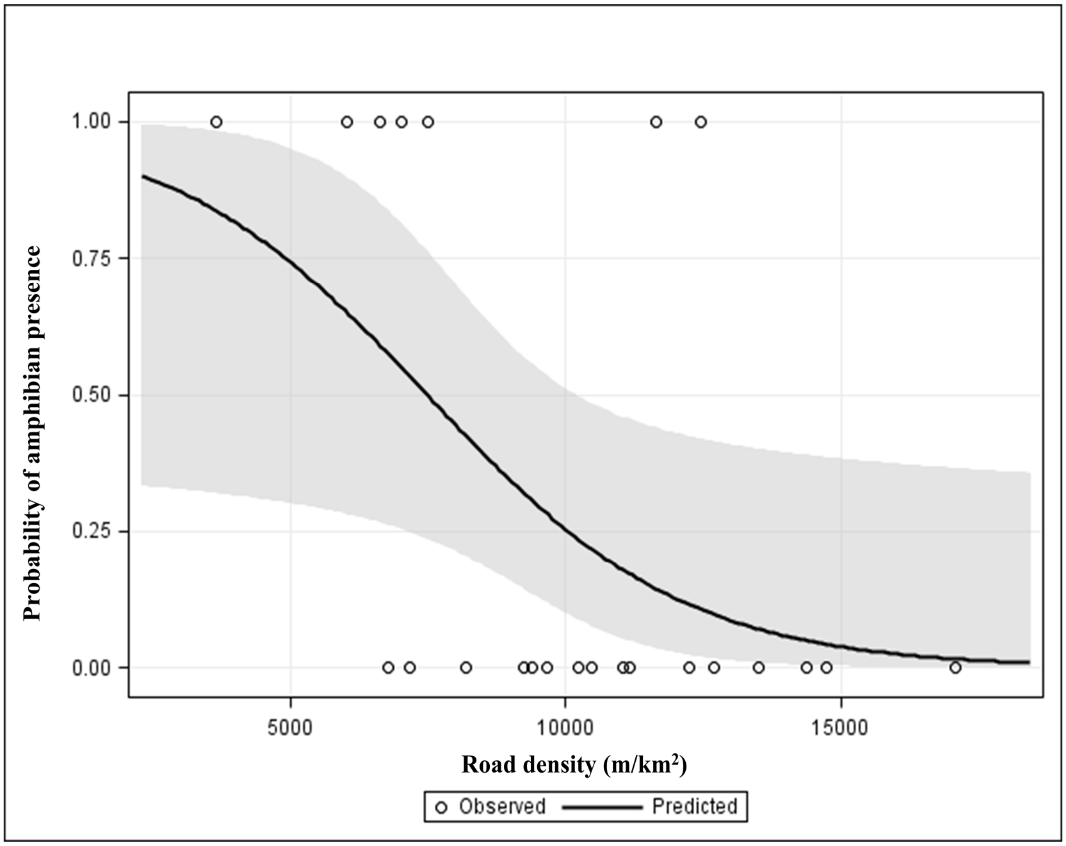


**FIGURE 2.5.** Probability of amphibian occurrence in urban lakes of Lubbock as a function of road density within 500 m of the lake's maximum aquatic extent in 2011.

# CHAPTER III

## USING DROUGHT DATA TO SET MANAGEMENT PRIORITIES FOR URBAN ANURAN MANAGEMENT

### ABSTRACT

Urbanization, due to associated habitat degradation and fragmentation, is threatening amphibian survival worldwide. Mitigating urban amphibian declines is critical for amphibian conservation and requires understanding of amphibian life-histories and their use of urban landscapes. Since amphibian monitoring is non-existent in urban centers of the Southern High Plains, I conducted amphibian surveys in 2011 and 2012 in the city of Lubbock, west Texas, to establish fundamental baseline data regarding amphibian species occurring within the city, and evaluated site-suitability at site-specific and landscape scales at 23 urban lakes. While droughts are a recurring phenomenon here, the year 2011 broke past records for drought intensity and severity. I observed greater species richness and incidence of amphibian occurrence in 2012 which was relatively wetter; Bufonids (*Anaxyrus speciosus* and *A. cognatus*) were the most widespread. Using data from the drought, I attempted to establish a simple method to base management recommendations in the event of data scarcities associated with natural climatic extremes. This was used to create a preliminary grouping of lakes in order of amphibian management priority and level of management effort, thus emphasizing the importance of data gathered under drought conditions towards amphibian management efforts in the region.

# INTRODUCTION

Globally, urban development has been associated with amphibian population declines because of consequent loss, degradation and fragmentation of aquatic and terrestrial habitats (Beebee and Griffiths 2005; Cushman 2006; Hamer and McDonnell 2008; Harper et al. 2008; Southerland and Stranko 2008; Bury 2008; Mitchell and Brown 2008). Amphibians sometimes use stormwater ponds and other constructed waters for breeding in urbanized landscapes (Ostergaard et al. 2008; Simon et al. 2009; Birx-Raybuck et al. 2010; McCarthy and Lathrop 2011) which exposes them to potential risks to their population viability in the form of accumulated pollutants (fertilizers, heavy metals, road oil and other undesirable elements), modified hydrological regimes, and stocked fish (Taylor et al. 2005; Croteau et al. 2008; Snodgrass et al. 2008a, 2008b). Impeded landscape connectivity in the form of roads and neighborhoods, and associated risk of mortalities prevent effective juvenile dispersal, reduces choice of suitable breeding/terrestrial habitat and may have negative effects on size of resident amphibian populations (Fahrig et al. 1995; Lehtinen et al. 1999; Carr and Fahrig 2001; Parris 2006).

Amphibians are seriously affected by prolonged drought conditions because of their dependence on precipitation to initiate behaviors such as emergence and reproduction (Boone 2003; Carey and Alexander 2003). Moreover, amphibian population distributions fluctuate between years in response to environmental variations and changes occurring to their breeding and terrestrial habitats (Skelly 2001; Guerry and Hunter 2002). Hence, improving our understanding of amphibian populations at various scales (.i.e., site and landscape scales) in response to anthropogenic disturbances is critical to the establishment of effective conservation strategies.

Urban development in the Southern High Plains began in the late 1800s (Graves 1961); Lubbock, in northwest Texas, emerged as the largest city in the area (population of 233, 740; U.S. Census Bureau 2011). Historically covered by extensive shortgrass prairie grasslands, this region is currently one of the most agriculturally impacted in the western hemisphere (Bolen et al. 1989). Ephemeral 'playa' wetlands – the predominant hydrological feature and primary breeding habitat for amphibians in this otherwise arid landscape – have also been negatively impacted by ongoing landscape modifications (Haukos and Smith 1994; Johnson et al. 2011). Roughly 135 playa wetlands exist within Lubbock's city limits and most have been excavated to increase stormwater retention in the absence of conventional methods for stormwater drainage. Two major stormwater drainage systems run west-east engaging approximately 30 pipe-connected playa wetlands. Playa wetlands have also been modified to make way for urban development such as roads, buildings, neighborhoods, and recreational parks.

Lubbock suffered an unusually severe dry spell in 2011; one of the worst documented in its history with record numbers of days over 37.8°C in the months of June and August (NOAA 2012a, 2012b). Annual rainfall accumulation was 14.9 cm – over a fourth of which occurred in December – falling approximately three times short of its 30-year normal of 48.6 cm (NOAA 2011). While droughts are a recurring feature in this semi-arid region (Anderson et al. 1999; Woodhouse and Brown 2001), the drought of 2011 brought home the threat of increased frequency and severity of droughts as predicted by several climate models (Boone et al. 2003; Seager et al. 2007; Lawler et al. 2009; Seager and Vecchi 2010).

Thirteen species of amphibians occur in the Southern High Plains – Texas Toads *(Anaxyrus speciosus)*, Great Plains Toads *(Anaxyrus cognatus)*, Woodhouse's Toads *(Anaxyrus woodhousii)*, Green Toads *(Anaxyrus debilis)*, New Mexican Spadefoots *(Spea multiplicata)*, Plains Spadefoots *(Spea bombifrons)*, Couch's Spadefoots *(Scaphiopus couchii)*, Northern Cricket Frogs *(Acris crepitans)*, Plains Leopard Frogs *(Lithobates blairi)*, Spotted Chorus Frogs *(Pseudacris clarkii)*, Great Plains Narrow-Mouthed Toads *(Gastrophryne olivacea)*, American Bullfrogs *(Lithobates catesbeiana)* and Barred Tiger Salamanders *(Ambystoma tigrinum mavortium)* (Anderson et al. 1999; Gray et al. 2004; Lannoo 2005). Amphibians of this region are xeric-adapted and lead highly terrestrial lives, returning to the aquatic habitats only to breed. Many of these species have short hatching times and rapid metamorphosis, enabling juvenile recruitment in highly ephemeral lakes (Degenhardt et al. 1996; Warburg 1997; Denver et al. 1998). In North America, amphibian breeding has been strongly correlated with ambient air temperature and overall precipitation (Owen 1989; Degenhardt et al. 1996; Johnson and Batie 2001). Isolated thunderstorms in the spring and summer initiate breeding activity in the Southern High Plains, with full choruses occurring at rain-filled pools and playas within few hours of rainfall (Bragg 1940; Creusere and Whitford 1976; Dimitt and Ruibal 1980). Presence of water, at least long enough to complete metamorphosis is crucial, as is the presence of wetlands with a range of hydroperiods to accommodate amphibian biodiversity in the region (Warburg 1997). However, extreme droughts limit water availability thus posing problems of heat stress, desiccation, and inhibition of metamorphosis in rapidly drying pools (Jaeger 1980; Warburg 1997). Even though xeric amphibians are speculated to be long-lived, and have behavioral adaptations to avoid desiccation such as burrowing (Warbur 1997), prolonged droughts may severely cripple persistence of amphibian populations in this region.

Amphibian monitoring in urban centers of the Southern High Plains such as Lubbock is virtually nonexistent, and little concern has been demonstrated for amphibians in the layout and design of urban environments. The cumulative effects of drought conditions and urbanization may cause local extinctions in the absence of suitable management strategies. Hence there is a need for knowledge of existing urban amphibian composition, the sites they occupy, and a baseline to begin management efforts.

My objectives for the study were to:

(1) Document the presence and species richness of amphibians occurring in lakes within Lubbock's city limits during the drought year of 2011 and the relatively wetter year of 2012 to gather fundamental baseline data regarding amphibians currently existing within this urban landscape

(2) Evaluate the importance of site-specific variables and landscape-scale variables on amphibian species occurrences

(3) Use data from drought to prioritize lakes for amphibian management

I hypothesized that lakes surrounded by greater amount of urban development in the form of roads and impervious surfaces would have lower incidences of amphibian occurrence. I expected to find wide distribution of endemic Bufonids and the more aquatic-habitat preferring American Bullfrogs *(Lithobates catesbeiana)* due their relative unpalatability by fish (Kats et al. 1988) and greater vagilities associated with their larger body size (Peter 1983). I also expected to see fewer species with longer larval period species such as the Plains Leopard Frogs (*Lithobates blairi)* and Bullfrogs in 2011 than in 2012. Based on the results of this study, I will highlight the importance of drought data for developing a relatively simple method for prioritizing urban lakes for amphibian management.

## MATERIALS AND METHODS

### *Study Sites*

I surveyed 23 modified playa wetlands within Lubbock's city limits (referred to as 'lakes' hereafter) for amphibians from March to October 2011 and March to June in 2012 (Fig. 3.1). Lakes for sampling were chosen from the city's stormwater maps. All lakes within the study set are impacted by surface water runoff from surrounding roads, neighborhoods and establishments. Necessary permits were obtained from landowners before conducting surveys at any of the sites.

### *Data collection*

I used a combination of amphibian call surveys, visual encounter surveys and larval surveys to assess amphibian species occurrence (Fellers et al. 1994; Houlahan and Findlay 2003). Beginning the night after rain events, starting half an hour after sunset, I performed 5 min call surveys at each lake and identified species calling (Weir and Mossman 2005). Following the call survey, I conducted a 20 min visual encounter survey (Knutson et al. 2004) by walking along the lake shoreline/wading in the shallow regions, and identifying amphibians seen. For two consecutive weeks following call surveys, I sampled lakes for amphibian larvae once a week using a pipe-sampler and a dipnet. The number of samples was standardized based on lake area and efforts were made to sample all microhabitat types. All tadpoles caught were identified and then released (Altig 1970). The combination of three survey types increased detection probability and confirmed amphibian species presence.

I recorded water quality variables – pH and conductivity – approximately 1 - 2 m from the shoreline at a depth of 5 cm (Pilliod and Peterson 2001), during call surveys (one location) and tadpole surveys (two or three equally spaced locations) using a hand-held YSI 63 meter (YSI Inc., Yellow Springs, Ohio, USA). I visually estimated percent cover of within-lake emergent vegetation and presence/absence of fish. Hydroperiod was quantified on an ordinal scale based on repeated lake visits; in 2011: '1' if lakes dried up within 2 weeks, '2' if lakes dried up within 3 months (but retained water longer than 2 weeks), and '3' if lakes held water for more than 3 months; in

2012: ‘1’ if lakes were dry until rain events, ‘2’ if lakes had very little water (as puddles to start with) but quickly filled up with rain events , and ‘3’ if lakes held considerable water even before rain events. Ordinal categorization of lake hydroperiod in 2012 differed from 2011 since 2012 was relatively wetter. Lake areas were estimated from the website of the U.S. Fish and Wildlife Service’s National Wetland Inventory (NWI) and ground-truthed during field surveys. Because connectivity may allow fish and amphibians to migrate between lakes, I also categorized lakes as ‘1’ if they were connected via a pipeline as part of the city’s storm water drainage projects, and ‘0’ for those lakes that were not part of the drainage projects.

I obtained land cover information from the Texas Natural Resources Information System (TNRIS) - 2010 NAIP 1 m resolution aerial imagery for Lubbock quadrangles; and the Lubbock city website - GIS layers for roads, lakes and city limits. I used ArcGIS10 (ESRI, Redlands, California, USA) and ERDAS (Intergraph Corporation part of Hexagon Group) to calculate road density and percent cover of impervious area respectively, within 500 m of the lake’s maximum extent. Since this area differed among lakes, road density was expressed as length of road by landscape area to standardize the variable. I also recorded nearest wetland distance and age of development surrounding the lake - estimated on an ordinal scale in 10 year increments; ‘1’ being the oldest development (developed in the 1950s) and ‘6’ being the most recently developed (developed in the 2000s); using information from the site of the Lubbock Central Appraisal District (Lubbock Central Appraisal District 2012). Most of Lubbock’s ‘urban’ area roughly falls in the semi-circular region – referred as the ‘loop’ hereafter – bounded by the loop 289, and I-27 north (Fig. 3.1) and was developed much earlier in the city’s history. Hence, lakes within the loop have felt the effects of fragmentation and urbanization for a longer time period and were designated a location class ‘1’ and those outside, ‘0’.

Since amphibian movements are influenced by rainfall, availability of moisture and refugia, danger of desiccation and predator avoidance, very few individuals of the population travel extreme distances (Forester et al. 2006; Semlitsch 2008) and most activities occur within 300 m of the lake edge (Semlitsch and Bodie 2003; Forester et al. 2006). Mean migration distance was reported to be 334 m for Ranids, 279 m for Bufonids and 237 m for Hylids (Lemckert 2004). Hence, I assumed that landscape characteristics at spatial scale of 500 m would be adequate to predict amphibian occurrences.

### *Statistical analysis*

Because of small sample size (n = 23) and the unusual severity of the drought in 2011, I had an overdispersion of sampling zeros – too many lakes with zero detection/observation - to allow for standard statistical and modeling procedures. Therefore I used a combination of many analytical methods to make inferences about the data. For the same reason, I assessed variables and models separately for 2011 and 2012.

I compared abiotic attributes (both site-specific and landscape-scale) with Wilcoxon-Mann-Whitney non-parametric tests of lake categories based on general city location, amphibian presence and presence of pipe-connection to stormwater drainage systems. I also used basic univariate logistic regressions to test the effect of each abiotic variable on amphibian presence, species richness, and species-specific presence, during 2011 and 2012. Predictor variables were not transformed since logistic regression makes no assumptions regarding their distributions.

I was interested in testing the ability of survey data from a severe drought year (2011) in predicting sites likely to be used by amphibians during a relatively wetter year (2012). For this, I created a candidate set of variables significant at $P < 0.25$ for amphibian presence and species richness (Hosmer and Lemeshow 2000). Biologically meaningful models were created from this candidate set and ranked by an information-theoretic approach using Akaike's Information Criterion corrected for small sample sizes ($AIC_c$) (Burnham and Anderson 2002), for amphibian presence, species richness and species-specific presence in 2011 and 2012. The maximum number of variables in a model was limited to three to avoid over-parameterization. The corrected Akaike's Information Criterion for small samples ($AIC_c$), difference in $AIC_c$ value of the best model to model $i$ ($\Delta_i$), and Akaike weight ($w_i$) was used to rank models from the candidate set. Increasing values of $\Delta_i = (AIC_{c\,i} - AIC_{c\,min})$ implies decreasing strength of model $i$, and Akaike weight $w_i$ gives the weight of evidence in favor of a particular model $i$ being the best among models in the candidate set (Burnham and Anderson 2002). For each response variable (amphibian presence, species richness, species-specific presence), models with $\Delta_i \leq 2$ were considered to have a substantial level of empirical support (Burnham and Anderson 2002). Given the collinearity among predictor variables, the well-supported models were intended to provide the best estimate of the dispersion of the response variable.

Well-supported models for species richness in 2011 ($\Delta_i \leq 2$) were used to predict sites with amphibian occurrence in 2012. First, predicted values of species richness for lakes were extracted using regressions from each of the supported models for species richness in 2011 ($\Delta_i \leq 2$). These values were then averaged for each lake. I sorted the lakes in ascending order of the average expected species richness. This list was then directly split into three approximately equal categories:

- Category 1 (bottom seven lakes from the sorted table; they have highest values of expected species richness) – maximum potential for amphibian use,
- Category 2 (middle eight lakes from the table; they have intermediate values of expected species richness) – intermediate potential for amphibian use, and
- Category 3 (top eight lakes from the sorted table; they have lowest values of expected species richness) – least potential for amphibian use.

This prediction was matched with amphibian occurrences in 2012 when precipitation (while still low) was considerably higher, to check the efficacy of the prediction, after which a management priority ranking was attached to each group. All analyses were performed using SAS Ver. 9.3 (SAS Institute Inc., Cary, North Carolina, USA).

## RESULTS

### *Amphibian surveys*

*In 2011 (See Fig. 3.2)***.—**Of the 13 species present in this region, I detected five species during surveys in 2011 and at only seven lakes of the 23 (30.4%) (Table 3.1). Spotted Chorus Frogs (*Pseudacris clarkii)* and Texas Toads (*Anaxyrus speciosus)* were detected most often - at three and five sites respectively. I also detected Great Plains Narrow-Mouthed Toads (*Gastrophryne olivacea*), Plains Spadefoots (*Spea bombifrons)* and American Bullfrogs (*L. catesbeiana),* a species non-native to the region, at one site each. Other species that are widespread in this region such as the Great Plains Toads (*Anaxyrus cognatus*) and the New Mexico Spadefoots (*Spea multiplicata*) (Gray et al. 2004) were not detected in 2011. I observed very few breeding aggregations, most of which occurred following a heavy thunderstorm in August with rainfall highs of 7.2 cm in certain areas of the city. Larval amphibians occurred only in four of the lakes (Table 3.2); Texas Toad tadpoles in three lakes, and Great Plains Narrow-Mouthed Toad tadpoles in one lake.

In 2011, species richness varied from 0 to 3 species; highest species richness was detected at one lake only (Table 3.2). Great Plains Narrow-Mouthed Toads and Spotted Chorus Frogs were both observed calling from fishless lakes with greater cover of thin-stemmed vegetation (Table 3.3) and breeding displays occurred while holding on to thin stems or from atop wooden debris. These lakes were characterized by relatively neutral pH ranging from 7.0 – 7.3, and were located in areas developed recently (from the 1980s onward: categories 4 to 6) outside the urban loop. They were also surrounded by lower road densities ranging from 3652.17 to 6644.74 m/km$^2$. Additionally, a lone Plains Spadefoot was heard vocalizing at one of these lakes. Texas Toads were observed in both fish-filled (two lakes) as well as fishless lakes (three lakes) both within and outside the urban loop. Non-native American Bullfrogs were recorded at one within-loop fish-filled lake characterized by high cover of cattails.

*In 2012 (See Fig. 3.2)***.—**A significantly wetter year, 2012 had a greater incidence of amphibian occurrences in Lubbock. By the end of May 2012, the city had received an average rainfall of 9.53 cm, with averages for May and June together totaling to 6 cm. I observed many more breeding aggregations along with the emergence of three more species - Great Plains Toads (*A. cognatus*), New Mexican Spadefoots (*S. multiplicata*), Plains Leopard Frogs (*L. blairi*) and Barred Tiger Salamanders (*Ambystoma tigrinum mavortium*). Since my sampling techniques were not adequate for salamander detection, I excluded their presence from all analyses and data representations. One lake that was initially part of the inundated sample set in 2011, remained dry during the duration of the study in 2012.

I detected anurans at 11 sites and observed as many as five species chorusing simultaneously at some sites in 2012 (Table 3.2). Once again, Texas Toads were most widespread occurring in nine sites, followed by the Great Plains Toads in five sites. Spotted Chorus Frogs and Plains Leopard Frogs were observed at four sites each, and

Great Plains Narrow-Mouth Toads occurred in two sites. In 2012, however, I experienced difficulties in fine-scale identification of spadefoots by their calls, especially in the presence of huge vocalizing congregations of multiple species. Hence for the sake of analysis, I lumped the species together, i.e., the Plains Spadefoots and the New Mexican Spadefoots, and referred to them collectively as spadefoots, or *Spea spp.*, and I observed them in four sites. Bullfrogs showed no change in their distribution and occurred at the same site they were detected at in 2011. Anuran larvae were observed at four sites following call surveys during March to May 2012 (only call surveys were conducted in June 2012): Spadefoot tadpoles at one lake, Plains Leopard Frog tadpoles in one lake, and Bufonid tadpoles in two lakes (Table 3.2).

Anuran species richness ranged from zero to five species; maximum anuran species richness occurred in three lakes (Table 3.2). Anurans were detected in all five lakes that were completely dry before rain events (hydroperiod class ‘1’). Only three of 13 permanent lakes (hydroperiod ‘3’) and three of four mostly-dry-until-rain lakes (hydroperiod ‘2’) were used by anurans. Amphibians occurred most often in lakes with less modification, i.e., in recently developed neighborhoods outside the loop, with greater covers of emergent and fringing vegetation, and less in lakes surrounded by mowed lawns. Great Plains Narrow-Mouthed Toads were never observed with Bufonids, though they co-occurred with Spotted Chorus Frogs and Plains Leopard Frogs; they occurred in two ephemeral fishless lakes having a narrow range of near-neutral pH (7.5 – 7.6; Table 3.4) with 52.5 – 70% emergent vegetation cover. Only one of the lakes with Spotted Chorus Frogs had fish, and here they called from temporary pools outside the lake at a distance away. Bufonids (Texas Toads and Great Plains Toads), as in 2011, were observed in fish-filled as well as fishless lakes. Site and landscape characteristics varied widely for these species. Spadefoots exclusively occurred in lakes in newly built neighborhoods (built in the 2000's: category 6).

*Responses to site-specific and landscape characteristics*

*In 2011 (See Fig. 3.2)*.**—**Lakes with amphibian detections were surrounded by significantly lower road densities and impervious area covers when compared to lakes without amphibian detections (Fig.3.3; $P \leq 0.05$, Wilcoxon-Mann-Whitney test). Of the seven lakes with amphibian detections, only two lay within the loop, while the rest were located outside the loop. Lakes within the loop were more isolated and were surrounded by higher road densities and percent covers of impervious surface (Fig. 3.4; $P \leq 0.05$, Wilcoxon-Mann-Whitney test). Lakes within the loop were also characterized by less emergent vegetation ($P = 0.01$, Wilcoxon-Mann-Whitney test) and were surrounded by older developments ($P < 0.001$; Wilcoxon-Mann-Whitney test).

Logistic regressions further confirmed the above results. Road density and impervious area surface had significant negative effects on amphibian presence ($P \leq 0.05$, Logistic regression). Road density also had a significant negative effect on amphibian species richness ($P = 0.05$; Ordered logistic regression). pH values tending towards neutral values strongly favored amphibian species richness ($P = 0.05$;

Logistic regression). Larger lakes supported Texas Toad presence (*P* = 0.05, Logistic regression).

*In 2012 (See Fig. 3.2)*.—Lakes where amphibians were detected in 2012 had significantly lower pH (Fig. 3.5; *P* = 0.03, Wilcoxon-Mann-Whitney test), higher cover of emergent vegetation (Fig. 3.6; *P* = 0.00, Wilcoxon-Mann-Whitney test), and lesser cover of impervious surface within 500 m from the lake edge (Fig. 3.3; *P* = 0.01, Wilcoxon-Mann-Whitney test). Seven of these lakes lay outside the loop and were characterized by lower conductivity values (*P* = 0.00, Wilcoxon-Mann-Whitney test) and higher cover of emergent vegetation (*P* = 0.03, Wilcoxon-Mann-Whitney test).

Logistic regressions on amphibian presence as the response variable indicated negative influence of pH, presence of fish, length of hydroperiod, adjacent road density and impervious area cover and positive association with emergent vegetation cover ($P \leq 0.05$, Logistic regression). Occurrence of Spotted Chorus Frogs, Texas Toads and Plains Leopard Frogs were negatively affected by conductivity ($P \leq 0.05$, Logistic regression). Smaller length of hydroperiod seemed to favor Texas Toad occurrence (*P* = 0.05, Logistic regression). All species occurrences were negatively influenced by landscape variables; road density and impervious area surface ($P \leq 0.06$, Logistic regression). Spotted Chorus Frogs and Plains Leopard Frogs also responded negatively to age of surrounding development and were observed more in lakes within recently developed neighborhoods ($P \leq 0.05$, Logistic regression). Ordered logistic regressions fitted to species richness showed negative influence of pH, presence of fish, hydroperiod length, cover of emergent vegetation, road density and impervious area cover ($P \leq 0.05$, Logistic regression).

*Model selection*

As stated earlier, I was interested in the ability of amphibian distribution data collected during severe drought (2011) in predicting sites likely to be used by amphibians the following year. Multiple significant correlations existed among predictor variables (Tables 3.5 and 3.6) which made it difficult to tease apart independent effects of individual predictors, given the few amphibian occurrences observed. Noting these correlations, the goal was to form parsimonious models which ascribe less individual significance to specific independent variables but which construct predictions of factors that might influence the dependent variable – amphibian presence, species richness, and species-specific presence.

We retained variables that were significant at 0.25 level from logistic regressions and combined them into biologically meaningful models avoiding the inclusion of highly correlated variables ($r_s \leq 0.7$) within the same model. Although presence of fish was not significant at the 0.25 level for presence or species richness in 2011, it was retained because of its high relevance in predicting presence of pool-breeding amphibians in literature (Kats et al. 1988; Wellborn et al. 1996; Hartel 2007). We developed seven models (Table 3.7) and these were ranked using the $AIC_c$ criterion for amphibian presence, species richness and species-specific presence in 2011 (Table

3.8) and 2012 (Table 3.9). Among the individual species in 2011, only Texas Toad occurrence was modeled because it was detected most widely in that year (five sites of 23). In 2012, all anuran species occurrences were modeled with the exception of American Bullfrogs (one site) and Narrow-Mouthed Toads (two sites).

*In 2011 (See Table 3.8)*.—Five models were well supported, both for amphibian presence and species richness, in 2011. The best model for predicting amphibian presence was the combination model with ‘pH and road density’ ($w_i$ = 0.23) and that for species richness was the water quality model with ‘pH and conductivity’ ($w_i$ = 0.25). Other well supported models for both these variables included age of neighborhood and nearest wetland density. The only well supported model for Texas Toad presence was the age of neighborhood model ($w_i$ = 0.43). All of these models were negatively associated with amphibian presence, species richness and Texas Toad presence.

*In 2012 (See Table 3.9)*.—Only one model was well supported for amphibian presence in 2012: the ‘hydroperiod’ model ($w_i$ = 0.83). Most lakes in the study area were permanent, i.e., hydroperiod class ‘3’, and this was negatively associated with amphibian presence. The ‘pH, conductivity, and road density’ combination model ranked highest for amphibian species richness ($w_i$ = 0.28), and again, negatively influenced the response. Other supported models for amphibian species richness included the hydroperiod model, the wetland isolation model with ‘road density and nearest wetland distance', and the predator model with 'presence of fish'. Increasing hydroperiod, lake isolation, and fish presence decreased amphibian species richness.

Top model for Spotted Chorus Frogs and Plains Leopard Frogs was the wetland isolation model with 'road density and nearest wetland distance' ($w_i$ = 0.61), 'neighborhood age' model for Great Plains Toads and Spadefoots ($w_i$ = 0.30 and $w_i$ = 0.78 respectively), and 'hydroperiod' model for Texas Toads ($w_i$ = 0.43). All the models were negatively associated with the response.

*Lake prioritization*

Categorization of lakes based on average values of expected species richness were effective in predicting sites with amphibian occurrence in 2012 (Table 3.10); I expected to detect amphibians in categories '1' and '2' but not in category '3'. Amphibians were detected in six out of seven lakes (86%) in Category '1' (high potential for amphibian use), in five out of eight lakes (63%) in Category '2' (medium potential for amphibian use), and in none of the seven lakes in category '3' (least potential for amphibian use).

Lakes in category '1' included fishless lakes surrounded by lower degree of wetland isolation, higher covers of emergent and fringing vegetation, and were located (mostly) outside the loop. These lakes were relatively less modified. Lakes in this category are likely to be very important in sustaining amphibian populations in the area and would require the least amount of management effort. Hence they were assigned a priority ranking of '1'.

Category '2' includes lakes with some favorable characteristics benefitting amphibian presence, such as, higher vegetation cover or lesser isolation. This category has lakes located both inside and outside the loop. Most of these lakes have fish, which could be an important factor influencing amphibian presence here. Therefore these lakes would require a greater degree of management effort to make them completely favorable for amphibian occurrence and were assigned a priority ranking '2'.

Category '3' includes lakes that are highly isolated in terms of high surrounding road densities and impervious area covers, have high density of fish, hardly any emergent or fringing vegetation; a large management effort is required to make these lakes suitable for amphibian breeding and were assigned a priority ranking '3'.

In all, I had an 83% success rate in absolute prediction of sites with likelihood of amphibian presence in 2012.

## DISCUSSION

I detected amphibians at 30.4% of the study lakes in 2011 and 50% of the study lakes in 2012. Most of these lakes were located along the city periphery (outside the loop) characterized by recent development, and lesser degree of lake modification. However, there were still a significant number of lakes (which were inundated) where I did not observe any amphibians both years – 16 lakes in 2011 and 11 lakes in 2012. Most of these lakes were located within the loop, 75% of which are part of the city's stormwater drainage systems, have stocked fish from previous years, higher values of basic pH, higher conductivity, and less or no emergent or fringing vegetation.

In 2011, water levels fell drastically in all the lakes, with many lakes drying entirely even before the start of summer. However, even small rains quickly elevated water levels in the lakes due to storm water run-off. But with most of the summer experiencing elevated temperatures – record 100 straight days with highs over 32°C and 48 days with highs of 37.8°C during the summer months (NOAA 2012b), evapotranspiration rates were likely very high and water levels diminished rapidly. The rain events in 2012 - significantly higher than 2011 - allowed for more breeding events and longer lake hydroperiod which might have benefitted larval development. While the occurrence of the drought itself may have been responsible for few amphibian detections overall, the combined effects of breeding habitat isolation and modified lake characteristics may have prevented effective lake colonization by amphibians.

Aestivating amphibian species in arid and semi-arid conditions have unique metabolic depression strategies that allow them to deal with prolonged periods of starvation and water stresses. It also aids in egg maturation and metabolic depression (preserves locomotor functions) during the dormancy period (Grundy and Storey 1998; James 2010; Young et al. 2011). This permits quick emergence upon favorable conditions, e.g., during thunderstorms, and initiation of breeding and foraging activities. Prolonged droughts can impair organ and tissue functions, locomotion, and muscle contraction abilities because of elevated soil temperatures and reduced soil moisture levels over extended periods (Grundy and Storey 1998; Booth 2006; James 2010; Young et al. 2011). Young et al. (2011) observed increased metabolism in aestivating *C. alboguttata* at elevated temperatures (metabolic rate at 30$^{o}$C was approximately double that at 20$^{o}$C or 24$^{o}$C) which seriously impaired their metabolism depression capacity, and Booth (2006) demonstrated loss of body water from *C. alboguttata* in sandy soil with a water potential of -1000 kPa. Low rainfall events also prevent occurrence of breeding or results in the loss of entire larval cohorts due to rapid drying of pools before the completion of metamorphosis (Warburg 1997). Limited amphibian emergence and low population numbers during droughts create difficulties in assessment of amphibian communities (Dodd 1992; Babbitt and Tanner 2000) and hence very few studies have reviewed the impacts of drought on xeric amphibian distribution, much less in an urban setting. Amphibian surveys and the $AIC_c$ ranked models contributed to the limited knowledge that existed about resident amphibians by providing valuable baseline information regarding amphibian distribution and breeding habitat use in Lubbock city during and after a drought year.

This study reiterates the importance of both landscape-scale and site-specific characteristics in understanding amphibian use of urban wetlands. Water quality was an important predictor for both amphibian presence and species richness, with some species being more vulnerable than others. Spotted Chorus Frogs and Great Plains Narrow-Mouthed Toads have thinner, more delicate skin than other species in the region, and also have longer larval periods (Table 3.11). As a result, they have increased susceptibility to degraded water quality of breeding lakes (Anderson et al.

1999). Both these species indicated a preference for more neutral waters with larger cover of thin-stemmed vegetation and woody debris from where they engaged in breeding displays (Bragg 1943; Wright and Wright 1949; Anderson et al. 1999). In most instances Bufonids were observed calling from clumps of vegetation along the shoreline and didn't seem to show any particular preference for specific vegetative structure. Although not evident from my study, Spadefoots have been shown to respond positively with vegetation cover (Torrence 2007). In lakes which have been converted to city parks, vegetation and fringing vegetation was periodically removed which may have changed/affected amphibian use of the lake.

In 2011, nearly 48% of the lakes in the study area had water permanently (all had fish), 43% dried up within 3 months (two lakes had fish) and 9% dried up in two weeks. In 2012, 56.5% were permanently filled and had considerable water even before the first rains, of which 92.3 % had fish - all except one from 2011. As hypothesized, Texas Toads (in 2011 and 2012) and Great Plains Toads (in 2012) were widely distributed within our study area, occurring in both fish-filled and fishless lakes. Bufonid tadpoles maybe distasteful to fish at many larval stages (Kats et al. 1988; Kurzava and Morin 1998; Nystrom and Abjornsson 2000), and this may have improved chances of their occupancy at fish-filled lakes. However, their occurrence was clearly skewed towards fishless lakes. At one instance Texas Toads were observed preferentially calling from a newly formed pool just outside a lake with heavy fish concentration. Both Spotted Chorus Frogs and Great Plains Narrow-Mouthed Toads are also highly vulnerable to fish predation (Voris and Bacon, Jr.

1966; Hecnar and M'Closkey 1997; Baber and Babbitt 2003). At the one fish-filled lake where Spotted Chorus Frogs were observed, like Texas Toads, they were observed calling from temporary puddles outside the fish-filled lake demonstrating their preference for an ephemeral fish-free breeding pool.

As I hypothesized, Plains Leopard Frogs - which have a larval period of ~ 60d (Table 3.11) - were detected only in 2012. Venne (2012) stressed the importance of hydroperiod in shaping playa occupancy by different Plains species, and observed the occurrence of Plains Leopard Frogs in wetlands with hydroperiods > 50d. Among my study sites, most of the lakes which met these hydroperiod requirements in 2011 were usually highly modified lakes stocked with fish from previous years. Vulnerability to fish-predation (Kruse and Francis 1977) may have prevented this species from using these lakes in 2011. Bullfrogs were observed in a semi-permanent lake (both in 2011 and 2012) which is part of the city's stormwater drainage system and characterized by thick cattails and heavy fringing vegetation thus making it suitable for its breeding and existence (Pope 1964). Although I expected to find Bullfrogs at more inundated fish-filled lakes in the city, the low numbers of detection could be a result of lows in water levels in 2011, and a lack of thick emergent/fringing vegetation in the other lakes.

Like previous work, my study indicates negative impacts of increased road density and impervious surfaces in the landscape around the lakes on amphibian occurrence (Gray et al. 2004; Gray and Smith 2005; Parris 2006; Houlahan and Findlay 2003; Gagne and Fahrig 2007; Simon et al. 2009; Smallbone et al. 2011). Many of the lakes are directly linked to surrounding roads and impervious surfaces by storm drains, likely causing accumulation of heavy metal pollutants, road salts, pesticides and fertilizers and other chemical compounds, which are likely to decrease amphibian survival and juvenile recruitment (Hatt et al. 2004; Croteau et al. 2008; Snodgrass et al. 2008a, 2008b). This can be a serious problem in drought conditions when these lakes are the only sources of standing water for breeding. If species occurring in my study area show high nestedness (like that documented for Spadefoots in cropland playas (Gray et al. 2004)) and site fidelity (like that documented for Great Plains Narrow-Mouthed Toads in Texas (Jameson 1956)), then impediments to movement in the urban landscape could have significant impacts on amphibian populations. Road mortalities can be fatal to native populations at the crucial period of large-scale migration to breeding pools following thunderstorms (Fahrig et al. 1995; Carr and Fahrig 2001). Similarly, low vagilities of amphibians, greater degree of habitat isolation, and decreased habitat quality might increase the risk of urban wetlands functioning as sinks. In the event of local extinctions, recolonization might be hindered by anthropogenic barriers to amphibian movement. In a region where water availability is highly unpredictable, drought occurrence could prove fatal to isolated populations under the circumstances.

Presently information about species-specific distribution, dispersal, and persistence in the area is scarce. Since a time lag of several decades exists between changes in urbanization and response in amphibian species occurrence (Lofvenhaft et al. 2004), the situation calls for immediate mitigation efforts to enhance future amphibian persistence in the city. Therefore, I attempted to apply available data right away towards setting management priorities for amphibian conservation in the city. With this in mind, I conducted amphibian surveys in 2011 and 2012, and was extremely interested in evaluating the applicability of drought data towards management goals for the city. As it turns out, the severest one-year drought in the region (2011) yielded a good prediction (83%) for sites used by anuran species in a relatively wetter year (2012).

Lakes in category ‘1’ included fishless lakes, relatively less modified, surrounded by lower degree of wetland isolation, higher covers of emergent and fringing vegetation, and were located (mostly) outside the loop. The lake with no anuran detection in Category ‘1’ remained dry during the entire period of the study in 2012, and hence there is a good chance that anurans might have used that lake later in the season when it did fill up; this lake is surrounded by a relatively lesser road density (7174.75 $m/km^2$) and impervious area cover (28.46%), and has a vast amount of adjacent undeveloped land with good vegetation cover. With minimal restoration efforts, such as maintaining surrounding unused land and its good vegetation cover, along with some regulation of quality of stormwater runoff, these lakes have high potential of supporting anuran populations.

Category ‘2’ includes lakes - both inside and outside the loop - with some favorable characteristics benefitting anuran presence, such as, higher vegetation cover or lesser isolation. Most of these lakes have fish, which could be an important factor influencing anuran presence here. On speaking to some of the residents around the two lakes with no anuran detection in this category, I learnt about previous incidences of anuran occurrence in these lakes in the year before the start of my study. One of these lakes is split into two: a large permanent, fish-filled lake, and another small, highly ephemeral, fishless pool. Residents claimed to have seen anurans in the little pool during the heavy rains of 2010. Hence lakes in Category'2' have a high possibility of having anuran occurrences with the right management effort, such as removal of stocked fish, establishing emergent/fringing vegetation and vegetated buffers (Castelle et al. 1994; Hecnar and M'Closkey 1997) and increasing connectivity through under-road culverts and vegetated pathways (Calhoun et al. 2005; Schmidt and Zumbach 2008). Some of them have storm drains leading to the lake, and thus efforts should be dedicated to improving water quality of runoff.

Category '3' includes lakes that have least suitability of being used by anurans. These lakes are highly isolated in terms of high surrounding road densities and impervious area covers have high fish densities, and hardly any emergent or fringing vegetation; a large management effort is required to make these lakes suitable for anuran breeding. This calls for increased connectivity between sites through under-road culverts and vegetated pathways (Calhoun et al. 2005; Schmidt and Zumbach 2008), and a large effort in fish removal from these lakes, along with establishment of vegetated buffers and emergent vegetation (Castelle et al. 1994; Hecnar and M'Closkey 1997). Since these lakes are surrounded by high levels of urban development, costs incurred in management practices are likely to be the highest.

Hence, I was able to lay the foundation for amphibian management efforts in the city by forming a preliminary grouping of lakes based on potential for anuran use and degree of management effort which can be used by city managers. Climate projections predict higher temperatures, faster evaporation rates and more sustained droughts for the Southern High Plains region (United States Global Change Research Program 2009). No doubt, continuous monitoring of city wetlands is required for better understanding anuran population distributions and dynamics, and in turn, constantly strategizing for increasing efficiency of management methods.

**TABLE 3.1.** Amphibian species detected during surveys from March to October 2011 and March to June 2012 in urban lakes in the city of Lubbock, Texas. Amphibians were detected in seven lakes in 2011 and in 11 lakes in 2012 (n = 23).

| Species detected | Number of sites found in 2011 | Number of sites found in 2012 |
|---|---|---|
| *Anaxyrus speciosus* | 5 | 9 |
| *Anaxyrus cognatus* | 0 | 5 |
| *Pseudacris clarkii* | 3 | 4 |
| *Gastrophryne olivacea* | 1 | 2 |
| *Spea spp.*[*] | 1 | 4 |
| *Lithobates catesbeiana* | 1 | 1 |
| *Lithobates blairi* | 0 | 4 |

[*]Plains spadefoot (*S. bombifrons*) was heard clearly at one site in 2011. In 2012, however, we could not distinguish clearly between Plains and New Mexican spadefoots (*S. multiplicata*), and hence lumped them together as *Spea spp.*

**TABLE 3.2.** Anuran species assemblages observed during amphibian surveys conducted from March to October 2011 and March to June 2012 in urban lakes in the city of Lubbock, Texas.

| **Year** | **+Lake** | ***A. speciosus*** | ***A. cognatus*** | ***P. clarkii*** | ***G. olivacea*** | ***Spea spp.*** | ***L. catesbeiana*** | ***L. blairi*** |
|---|---|---|---|---|---|---|---|---|
| 2011 | 13 | | | + | *+ | + | | |
| | 85 | + | | + | | | | |
| | 93 | *+ | | | | | | |
| | 94A | + | | | | | | |
| | 27 | *+ | | | | | | |
| | 21 | *+ | | | | | + | |
| | 132 | | | + | | | | |
| 2012 | 13 | | | + | + | | | *+ |
| | 85 | | | | | | | |
| | 93 | + | + | + | | + | | + |
| | 94A | + | + | + | | + | | + |
| | 27 | + | + | | | | | |
| | 21 | + | | | | | + | |
| | 132 | + | + | + | | *+ | | + |
| | 56 | | | | + | | | |
| | 48 | *+ | *+ | | | | | |
| | 105 | *+ | | | | | | |
| | 52 | | | | | | | |
| | 42 | + | | | | | | |
| | 84 | + | | | | + | | |

*Tadpoles present; + Lake ID referred from the city of Lubbock's stormwater maps

**TABLE 3.3.** Range of variable values for individual anuran species measured during amphibian surveys from March to October 2011 urban lakes in the city of Lubbock, Texas.

| **Variable** | ***A. speciosus*** | ***P. clarkii*** | ***G. olivacea*** | ***S. bombifrons*** | ***L. catesbeiana*** | ***No detection*** |
|---|---|---|---|---|---|---|
| **No. of sites** | **5** | **3** | **1** | **1** | **1** | **16** |
| Mean pH | 7.25 - 8.22 | 7.03 - 7.25 | 7.12 | 7.12 | 7.52 | 7.30 - 8.63 |
| Mean cond[*] (µS) | 241.08 - 392.48 | 182.89 - 392.48 | 182.89 | 182.89 | 251.92 | 203.60 - 509.27 |
| Lake area (acres) | 2.15 – 17.18 | 0.07 - 3.11 | 0.59 | 0.59 | 17.18 | 0.07 - 11.91 |
| Emveg[**] (%) | 0.88 - 50 | 20 - 50 | 45 | 45 | 0.88 | 0.00 - 100.00 |
| Hp[φ] | 2.00 - 3.00 | 1.00 - 2.00 | 2.00 | 2.00 | 2.00 | 1.00 - 3.00 |
| Fish | 0.00 - 1.00 | 0.00 - 0.00 | 0.00 | 0.00 | 1.00 | 0.00 -1.00 |
| Connec[φφ] | 0.00 - 1.00 | 0.00 - 1.00 | 0.00 | 0.00 | 1.00 | 0.00 -1.00 |
| Road ($m/km^2$) | 6644.74 - 12464.04 | 3652.17 - 6644.74 | 6036.75 | 6036.75 | 12464.04 | 6787.70 - 17078.03 |
| Imp[1] (%) | 19.17 - 42.06 | 14.06 - 31.26 | 29.49 | 29.49 | 32.63 | 21.58 - 69.25 |
| Wetdist[11] (m) | 324.25 - 1506.37 | 459.88 - 868.91 | 459.88 | 459.88 | 1506.37 | 149.29 - 1185.83 |
| Loop | 0.00 - 1.00 | 0.00 - 0.00 | 0.00 | 0.00 | 1.00 | 0.00 -1.00 |
| Age | 2.00 - 6.00 | 4.00 - 6.00 | 4.00 | 4.00 | 2.00 | 1.00 -6.00 |

[*]conductivity; [**]emergent vegetation; [φ]hydroperiod; [φφ]connection to storm water drainage; [1]impervious area cover;

[11]distance to nearest wetland

**TABLE 3.4.** Range of variable values for individual anuran species measured during amphibian surveys from March to June 2012 in urban lakes in the city of Lubbock, Texas.

| **Variable** | ***A. speciosus*** | ***A. cognatus*** | ***P. clarkii*** | ***G. olivacea*** |
|---|---|---|---|---|
| **No. of sites** | **9** | **5** | **4** | **1** |
| Mean pH | 7.56 - 8.25 | 7.56 - 8.25 | 7.47 - 8.18 | 7.47 - 7.60 |
| Mean cond (μS) | 145.74 - 293.18 | 145.74 - 293.18 | 145.74 - 235.12 | 156.35 - 461.35 |
| Lake area (acres) | 1.00 - 17.89 | 1.33 - 17.89 | 0.73 - 17.89 | 0.73 - 1.25 |
| Emveg (%) | 0.88 - 50 | 6.67 - 50.00 | 6.67 - 52.5 | 52.50 - 70.00 |
| Hp | 2.00 - 3.00 | 1.00 - 3.00 | 1.00 - 3.00 | 1.00 - 2.00 |
| Fish | 0.00 - 1.00 | 0.00 - 1.00 | 0.00 - 1.00 | 0.00 - 0.00 |
| Connec | 0.00 - 1.00 | 0.00 - 1.00 | 0.00 - 1.00 | 0.00 - 0.00 |
| Road (m/km²) | 3652.17 - 12464.04 | 3652.17 - 11639.02 | 3652.17 - 7513.90 | 6036.75 - 12698.66 |
| Imp (%) | 14.06 - 43.48 | 16.06 - 42.06 | 14.06 - 33.21 | 29.49 - 39.67 |
| Wetdist (m) | 324.25 - 1506.37 | 324.25 - 868.91 | 324.25 - 868.91 | 459.88 - 445.65 |
| Loop | 0.00 - 1.00 | 0.00 - 1.00 | 0.00 - 0.00 | 0.00 - 1.00 |
| Age | 1.00 - 6.00 | 3.00 - 6.00 | 4.00 - 6.00 | 3.00 - 4.00 |

| **Variable** | ***Spea spp.*** | ***L. catesbeiana*** | ***L. blairi*** | ***No detection*** |
|---|---|---|---|---|
| **No. of sites** | **4** | **1** | **4** | **11** |
| Mean pH | 7.56 - 8.18 | 7.99 | 7.47 - 8.18 | 7.78 - 8.49 |
| Mean cond (μS) | 145.74 - 235.12 | 172.74 | 145.74 - 235.12 | 197.25 - 375.42 |
| Lake area (acres) | 1.24 - 17.89 | 17.18 | 0.73 - 17.89 | 3.11 - 11.91 |
| Emveg (%) | 6.67 - 52.5 | 24.17 | 6.67 - 52.50 | 0.25 - 31.67 |
| Hp | 1.00 - 3.00 | 2.00 | 1.00 - 3.00 | 2.00 - 3.00 |
| Fish | 0.00 - 1.00 | 0.00 | 0.00 - 1.00 | 0.00 - 1.00 |
| Connec | 0.00 - 1.00 | 1.00 | 0.00 - 1.01 | 0.00 - 1.01 |
| Road (m/km²) | 3652.17 - 7513.90 | 12464.04 | 3652.17 - 7513.90 | 6644.74 - 17078.03 |
| Imp (%) | 14.06 - 33.21 | 32.63 | 14.06 - 33.21 | 24.22 - 69.25 |
| Wetdist (m) | 324.25 - 888.92 | 1506.37 | 324.25 - 868.91 | 149.29 - 1185.83 |
| Loop | 0.00 - 0.00 | 1.00 | 0.00 - 0.00 | 0.00 - 1.00 |
| Age | 6.00 - 6.00 | 2.00 | 4.00 - 6.00 | 1.00 - 6.00 |

**TABLE 3.5.** Bivariate Spearman's correlation coefficients for predictors measured during amphibian surveys from March to October 2011 for urban lakes in the city of Lubbock, Texas.

| **Variables** | **Hp** | **pH** | **Cond** | **Emveg** | **Fish** | **Road** | **Wetdist** | **Imp** | **Age** | **Connec** | **Locn** |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Area | 0.658** | 0.627** | 0.143 | -0.543** | 0.635** | 0.442* | 0.105 | 0.516* | -0.199 | 0.643** | 0.361 |
| Hp | | 0.777** | 0.214 | -0.647** | 0.838** | 0.499* | -0.392 | 0.654** | -0.287 | 0.350 | 0.350 |
| pH | | | 0.082 | -0.646** | 0.681** | 0.405 | -0.203 | 0.495* | -0.283 | 0.217 | 0.243 |
| Cond | | | | -0.314 | 0.106 | 0.041 | 0.204 | 0.257 | -0.159 | 0.249 | 0.302 |
| Emveg | | | | | -0.694** | -0.333 | -0.070 | -0.341 | 0.455* | -0.308 | -0.564** |
| Fish | | | | | | 0.635** | -0.212 | 0.688** | -0.542** | 0.313 | 0.565** |
| Road | | | | | | | -0.341 | 0.683** | -0.605** | 0.144 | 0.604** |
| Wetdist | | | | | | | | -0.217 | 0.039 | 0.407 | 0.118 |
| Imp | | | | | | | | | -0.601** | 0.354 | 0.656** |
| Age | | | | | | | | | | -0.188 | -0.773** |
| Connec | | | | | | | | | | | 0.394 |

** Correlation is significant at the 0.01 level (2-tailed).

* Correlation is significant at the 0.05 level (2-tailed).

**TABLE 3.6.** Bivariate Spearman's correlation coefficients between observed amphibian presence/species richness/species-specific presence and measured site-specific/landscape-scale predictors for urban lakes in the city of Lubbock, Texas in 2011

| **Variables** | **Area** | **Hp** | **pH** | **Cond** | **Emveg** | **Fish** | **Road** | **Wetdist** | **Imp** | **Age** | **Connec** | **Locn** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Presence | 0.121 | -0.261 | -0.349 | -0.271 | 0.214 | -0.182 | -0.456* | 0.199 | -0.456* | 0.409 | 0.123 | -0.313 |
| Species richness | 0.089 | -0.287 | -0.408 | -0.288 | 0.234 | -0.204 | -0.455* | 0.187 | -0.443* | 0.357 | 0.121 | -0.307 |
| *A. speciosus* | 0.429* | -0.026 | -0.056 | -0.159 | 0.032 | 0.037 | -0.175 | 0.207 | -0.254 | 0.301 | 0.339 | -0.128 |
| *P. clarkii* | -0.360 | -0.422* | -0.584** | -0.019 | 0.428* | -0.442* | -0.584** | 0.097 | -0.389 | 0.359 | -0.112 | -0.405 |
| *L. catesbeiana* | 0.257 | -0.161 | -0.193 | -0.225 | -0.225 | 0.187 | 0.193 | 0.354 | -0.096 | -0.263 | 0.223 | 0.204 |
| *S. bombifrons* | -0.257 | -0.161 | -0.322 | -0.354 | 0.257 | -0.243 | -0.321 | -0.225 | -0.161 | 0.033 | -0.204 | -0.223 |
| *G. olivacea* | -0.257 | -0.161 | -0.322 | -0.354 | 0.257 | -0.243 | -0.321 | -0.225 | -0.161 | 0.033 | -0.204 | -0.223 |

** Correlation is significant at the 0.01 level (2-tailed).

* Correlation is significant at the 0.05 level (2-tailed).

**TABLE 3.7.** Models to be ranked by $AIC_c$ criterion for amphibian presence, species richness and species-specific presence for years 2011 and 2012.

| Model | Variables |
|---|---|
| Water quality model | pH + conductivity |
| Hydroperiod model | hydroperiod |
| Predator model | presence of fish |
| Wetland isolation model | road density + nearest wetland distance |
| Development model | neighborhood age |
| Combination models | pH + road density |
| | pH + conductivity + road density |

**TABLE 3.8.** Well supported models ($\Delta_i \leq 2$) representing relationships between site-specific and landscape-scale variables and amphibian presence, species richness and Texas Toad *(A. speciosus)* presence in 2011. $AIC_c$: Akaike's Information Criterion corrected for small sample sizes; $\Delta_i$ : difference in criterion values of the best ranked model to the model *i*; $w_i$: Akaike weights.

| Models | $AIC_c$ | $\Delta_i$ | $w_i$ |
|---|---|---|---|
| **Amphibian presence** | | | |
| pH + road density | 28.81 | 0.00 | 0.23 |
| road density + nearest wetland distance | 28.88 | 0.07 | 0.22 |
| neighborhood age | 28.91 | 0.10 | 0.22 |
| pH + conductivity + road density | 29.77 | 0.97 | 0.14 |
| pH + conductivity | 30.73 | 1.92 | 0.09 |
| **Species richness** | | | |
| pH + conductivity | 47.96 | 0.00 | 0.25 |
| pH + road density | 48.35 | 0.39 | 0.20 |
| neighborhood age | 49.00 | 1.04 | 0.15 |
| pH + conductivity + road density | 49.00 | 1.05 | 0.15 |
| road density + nearest wetland distance | 49.33 | 1.37 | 0.12 |
| ***A. speciosus* presence** | | | |
| neighborhood age | 26.49 | 0.00 | 0.43 |

**TABLE 3.9.** Well supported models ($\Delta_i \leq 2$) representing relationships between site-specific and landscape-scale variables and amphibian presence, species richness and species-specific presence in 2012. $AIC_c$: Akaike's Information Criterion corrected for small sample sizes; $\Delta_i$ : difference in criterion values of the best ranked model to the model *i*; $w_i$: Akaike weights.

| **Models** | **$AIC_c$** | **$\Delta_i$** | **$w_i$** |
|---|---|---|---|
| **Amphibian presence** | | | |
| hydroperiod | 23.42 | 0.00 | 0.83 |
| **Species richness** | | | |
| pH + conductivity + road density | 65.59 | 0.00 | 0.28 |
| hydroperiod | 66.02 | 0.43 | 0.22 |
| road density + nearest wetland distance | 66.15 | 0.56 | 0.21 |
| presence of fish | 66.40 | 0.81 | 0.18 |
| ***P. clarkii* presence** | | | |
| road density + nearest wetland distance | 13.89 | 0.00 | 0.61 |
| pH + road density | 15.07 | 1.18 | 0.34 |
| ***A. speciosus* presence** | | | |
| hydroperiod | 29.23 | 0.00 | 0.42 |
| road density + nearest wetland distance | 31.19 | 1.96 | 0.16 |
| ***L. blairi* presence** | | | |
| road density + nearest wetland distance | 13.89 | 0.00 | 0.61 |
| pH + road density | 15.07 | 1.18 | 0.34 |
| ***A. cognatus* presence** | | | |
| neighborhood age | 24.96 | 0.00 | 0.30 |
| pH + road density | 25.39 | 0.43 | 0.24 |
| road density + nearest wetland distance | 26.45 | 1.49 | 0.14 |
| ***Spea spp.* presence** | | | |
| neighborhood age | 12.27 | 0.00 | 0.78 |
| pH + road density | 16.14 | 3.87 | 0.11 |

**TABLE 3.10.** Preliminary ranking of lakes for urban amphibian management based on average values of expected species richness using data collected from March to October 2011, a severe drought period, during amphibian surveys in urban lakes in Lubbock, Texas: (1) highest amphibian use potential and highest priority for management, (2) medium amphibian use potential and intermediate priority for management, (3) lowest amphibian use potential and least priority for management. Prediction success is expressed as percentage below each category.

| | **Ranking** | | |
|---|---|---|---|
| | **1** | **2** | **3** |
| **Lake ID from stormwater maps** | 21[*+] | 27[*+] | 20 |
| | 56[+] | 29 | 44 |
| | 105[+] | 51 | 17 |
| | 84[+] | 94A[*+] | 22 |
| | 85[*] | 42[+] | 24 |
| | 13[*+] | 46 | 16 |
| | 132[*+] | 93[*+] | 89 |
| | | 48[+] | 31 |
| **Prediction success** | 86 % | 63 % | 100 % |

[*]Lakes with amphibian detection in 2011.

[+] Lakes with amphibian detections in 2012.

**TABLE 3.11.** Larval periods for anuran species of the Southern High Plains.

| Amphibian species | Larval period |
|---|---|
| Couch's Spadefoot *(Scaphiopus couchii)* | 7 – 16 d[1] |
| Plains Spadefoot *(Spea bombifrons)* | 13 – 20 d[2] |
| New Mexico Spadefoot *(Spea multiplicata)* | 12 – 19 d[3] |
| Great Plains Toad *(Anaxyrus cognatus)* | 17 – 45 d[4] |
| Texas Toad *(Anaxyrus speciosus)* | 18 – 60 d[5] |
| Spotted Chorus Frog *(Pseudacris clarkii)* | 30 – 45 d[6] |
| Plains Leopard Frog *(Lithobates blairi)* | ~ 60 d, overwinter[8] |
| Woodhouse's Toad *(Anaxyrus woodhousii)* | 35 – 49 d[9] |
| Great Plains Narrow-Mouthed Toad *(Gastrophryne olivacea)* | 24 – 50 d[10] |
| American Bullfrog *(Lithobates catesbeiana)* | > 90 d, overwinter[11] |
| Green Toad *(Anaxyrus debilis)* | 21 – 25 d[12] |
| Northern Cricket Frog *(Acris crepitans)* | 35 – 90 d[13] |

[1]Morey 2005a; [2]Farrar and Hey 2005; [3]Morey 2005b; [4]Graves and Krupa 2005; [5]Dayton and Painter 2005; [6]Sredl 2005; [8]Smith and Keinath 2005; [9]Sullivan 2005; [10]Sredl and Field 2005; [11]Degenhardt et al. 1996; [12]Painter 2005; [13]Gray et al. 2005; Venne et al. 2012.

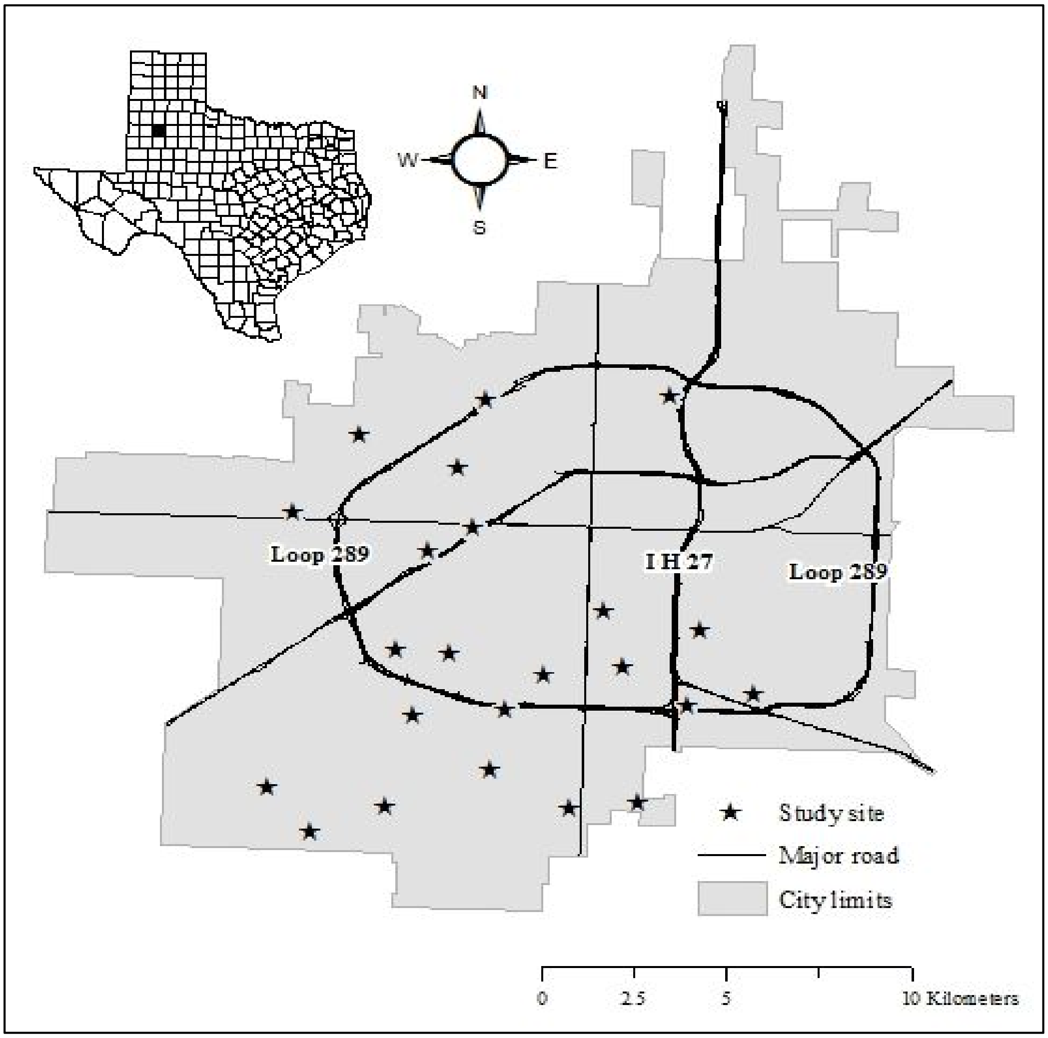


**FIGURE 3.1.** Map showing the 23 lakes surveyed for amphibian species presence from March to October 2011 and March to June 2012. All lakes were within Lubbock's city limits and impacted by stormwater runoff to varying degrees.

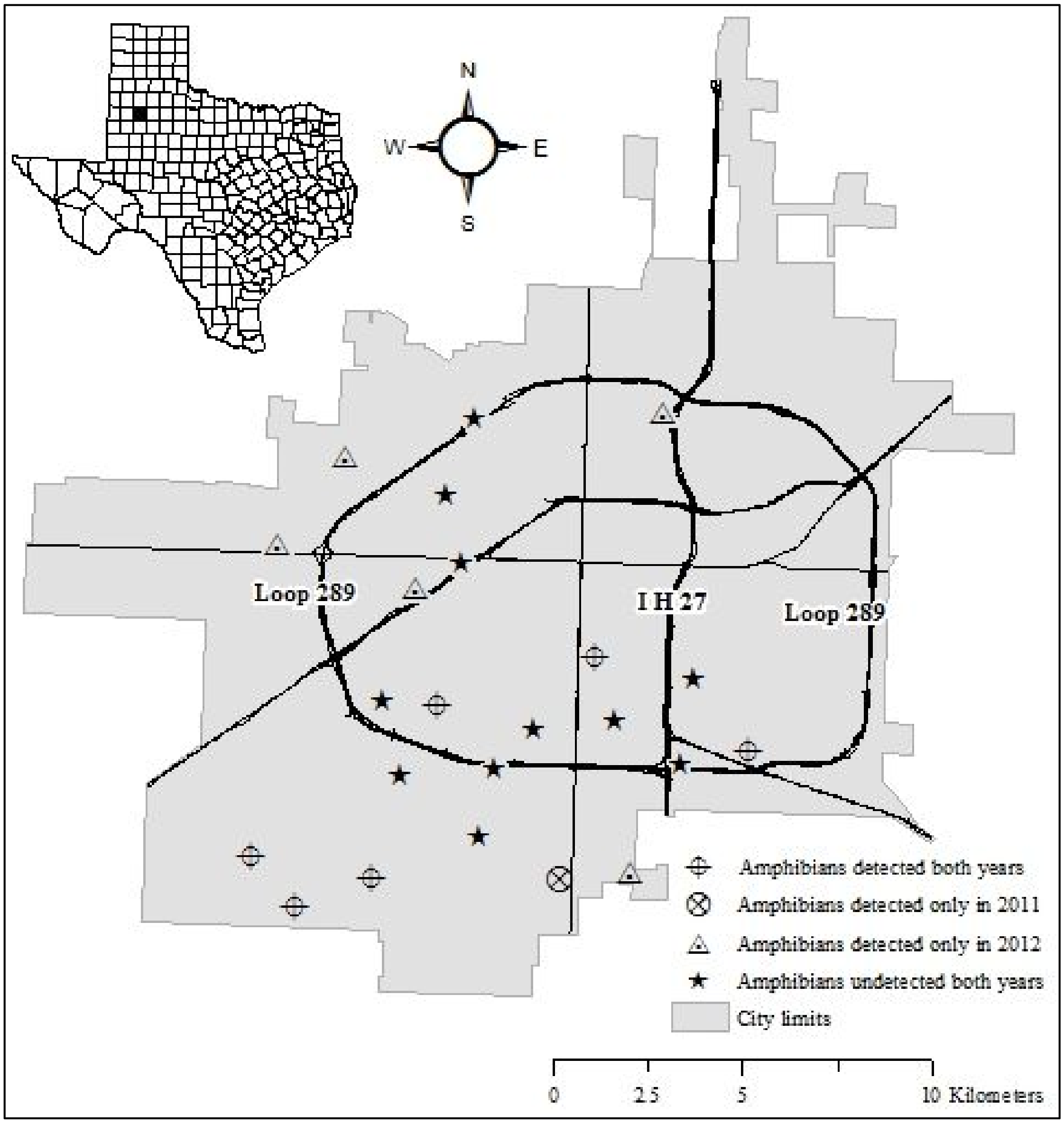


**FIGURE 3.2.** Map showing lakes with and without amphibian detection during surveys from March to October 2011 and March to June 2012. We observed amphibian occurence at seven lakes in 2011 and in 11 lakes in 2012. Since our sampling methods were not adequate for salamanders, they are not represented in this figure.

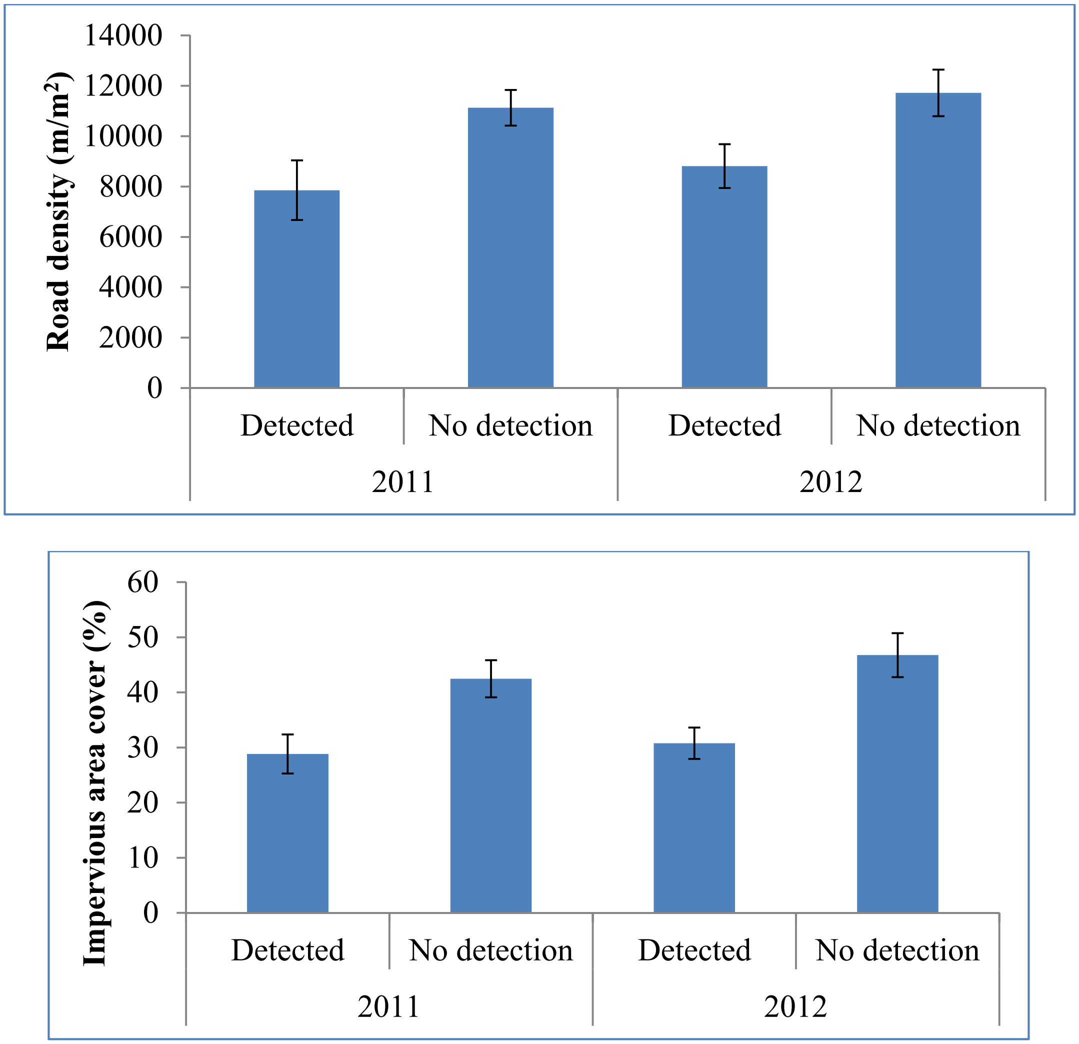


**FIGURE 3.3.** Comparison of landscape-scale factors (within 500 m radius) - road density and impervious area cover - for urban lakes in the city of Lubbock based on amphibian presence (n = 23). Road density and impervious area cover was significantly lower ($P \leq 0.05$) for lakes where amphibians were detected by Wilcoxon-Mann-Whitney non-parametric two-sample tests. *Error bars* ± SE.

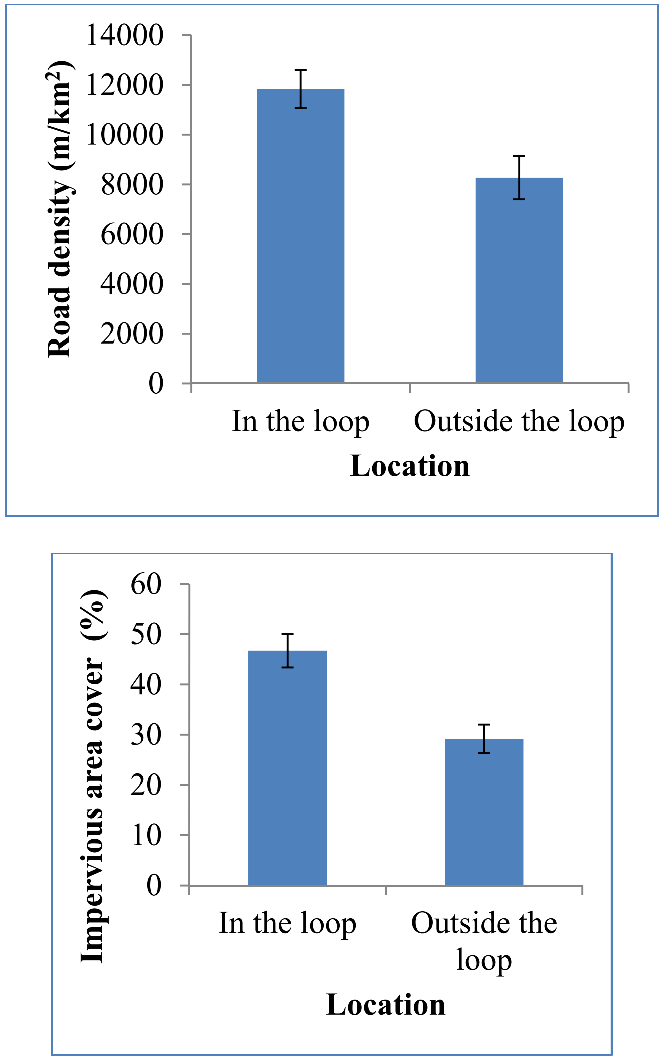


**FIGURE 3.4.** Comparison of landscape-scale factors (within 500 m radius) - road density and impervious area cover - for urban lakes in the city of Lubbock based on lake location within or outside the city loop (n = 23) in 2011. Road density and impervious area cover was significantly higher ($P \leq 0.05$) for lakes within the loop by Wilcoxon-Mann-Whitney non-parametric two-sample tests. *Error bars* ± SE.

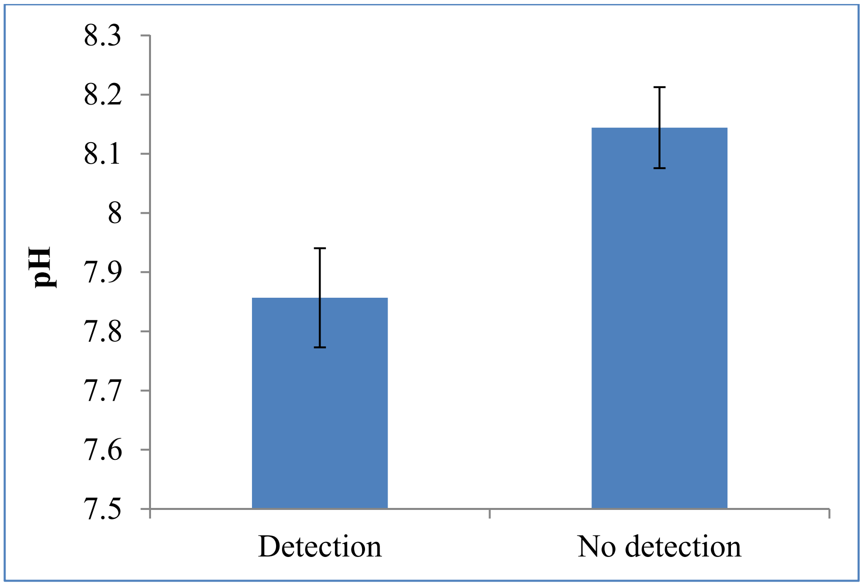


**FIGURE 3.5.** Comparison of pH for urban lakes in the city of Lubbock based on amphibian presence in 2012 (n = 23). pH was significantly lower ($P \leq 0.05$) for lakes with amphibian detection by Wilcoxon-Mann-Whitney non-parametric two-sample tests. *Error bars* ± SE.

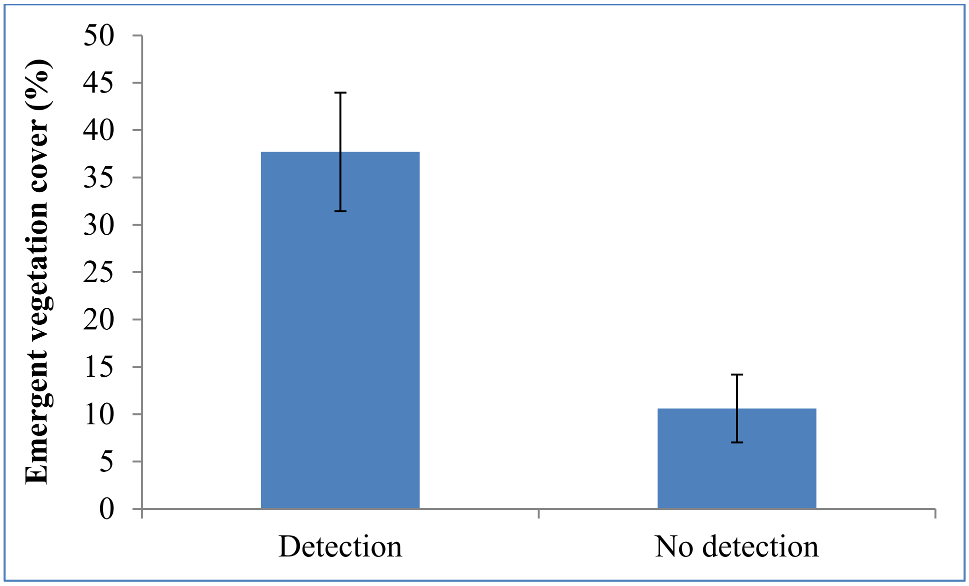


**FIGURE 3.6.** Comparison of percent cover of emergent vegetation for urban lakes in the city of Lubbock based on amphibian presence in 2012 (n = 23). Percent cover of emergent vegetation was significantly higher ($P \leq 0.05$) for lakes with amphibian detection by Wilcoxon-Mann-Whitney non-parametric two-sample tests. *Error bars* ± SE.

# CHAPTER IV

## SUMMARY, MANAGEMENT RECOMMENDATIONS AND CONCLUSIONS

This study is the first attempt at surveying for urban amphibians in the city of Lubbock, the largest city in the Southern High Plains. In an effort to gather fundamental baseline data for the area I tried to answer questions such as '*What amphibian species inhabit the city; which lakes are they using and why?*' Urbanization has contributed to widespread amphibian declines and hence knowledge of urban amphibian distributions is important to mitigate for some of the associated negative effects. Although severe drought conditions prevailed during this time, the situation presented unique opportunities for:

1. Contributing to the limited knowledge regarding amphibian distribution during drought conditions
2. Setting amphibian management priorities for the city of Lubbock

Over the course of my study which spanned two years, I detected seven anuran species; Texas Toads, Spotted Chorus Frogs, Great Plains Narrow-Mouthed Toads, American Bullfrogs, Plains Spadefoots, New Mexico Spadefoots, Great Plains Toads, and Plains Leopard Frogs; I detected the former five species during the severe drought year of 2011, and all seven species during the wetter year of 2012. Like similar research elsewhere, the study clearly indicated negative influences of wetland isolation caused by urban features such as roads and paved surfaces. Lakes with amphibian occurrence were, for the most part, limited to the edges of the city where urban development is recent or currently ongoing. These have also suffered a lesser degree of modification relative to lakes in parts of the city with a longer history of urban development. The importance of site-specific features of the lake such as water quality, hydroperiod, vegetation cover and fish presence in shaping amphibian distribution was also emphasized. Bufonids were the most widespread and occurred in a variety of conditions. While only the Bufonids and American Bullfrogs inhabited lakes in highly urbanized interior parts of the city, other species were restricted to the relatively less urbanized city edges.

One of the main interests of the study was to establish the importance of drought data in setting management goals. So I focused on testing the predictability of data from drought-afflicted 2011 for sites likely to have anurans the following, wetter year of 2012 (Chapter III). I used modeling procedures involving an Information-Theoretic Approach using Akaike's Information Criterion corrected for small sample sizes ($AIC_c$) to derive a better representation of the factors influencing anuran distribution and for deriving the prediction. Lakes were broadly lumped into three groups of varying potential for anuran use, and the prediction was validated using survey data from the following year. This yielded a prediction success of 83%, thus demonstrating the significance of gathering amphibian data during 'non-normal' years. With this established, I assigned each group a management priority ranking to provide city managers an idea of where to concentrate management efforts on (See below).

Group 1 - highest potential for amphibian use, highest management priority, and least management effort required

Group 2 – medium potential for amphibian use, medium management priority, and higher degree of management effort required

Group 3 – least potential for amphibian use, least management priority, and highest management effort required

Establishing vegetative buffers is a common method of protecting wetlands from the effects of surrounding land use and runoff. Many states such as Massachusetts and New Jersey have regulatory buffer zones for wetlands, 30-120 m wide, which greatly improves water quality and significantly reduces nitrogen and phosphorous inputs (Hubbard and Lowrance 1994; Schultz et al. 1995), but are still deemed inadequate for amphibians and reptiles (Semlitsch and Bodie 2003; Richter et al. 2008). Semlitsch and Bodie (2003) proposed a stratified design: an aquatic buffer immediately adjacent to the wetland to protect water resources, a core habitat overlapping with the aquatic buffer and extending out from the wetland edge for about 300 m, and an outer buffer zone extending approximately 50 m to protect core habitat from edge effects of surrounding land. Providing 'stepping-stones' in the form of restored/constructed wetlands with an array of hydroperiods can further mitigate the losses due to stocked predators, or in the events of calamities such as droughts (as experienced in 2011), while also improving habitat connectivity (Semlitsch 2002) by providing chances for opportunistic breeding (Babbitt and Tanner 2000). Establishing culverts through careful planning to facilitate amphibian dispersal and movement in areas of very wide roads is also gaining popularity (Calhoun et al. 2005; Schmidt and Zumbach 2008). The Arizona Game and Fish Department (2006) recommends that box culverts be easily accessible, frequently placed (depending on the species), at least a foot high, have with an open top-fitted grate, and installation of a 1.5 to 2.5 ft preventative fence top to prevent amphibians from jumping/climbing over (Arizona Game and Fish Department, Habitat Branch 2006).

The Southern High Plains region, with its vastly modified landscape, erratic weather patterns, unpredictable water availability and growing urbanization forms a very interesting system for amphibian studies. Implementation of urban management strategies poses a challenge since none are in place, and information about species-specific distribution, dispersal, and persistence is scarce. For a drought afflicted region such as the Southern High Plains, where drought frequency and intensity is expected to increase in future, the value of data generated during these conditions cannot be ignored in management implementation. My study sets a benchmark for anuran distribution and management in the city of Lubbock and a baseline for future research into urban anuran persistence in the area. However, it is to be emphasized that strategizing for conservation is an ever dynamic process involving constant modifications and updates to make the best use of available data. In addition, public education programs highlighting regional amphibian biodiversity and their specific ecosystem roles and services are of utmost importance to gain impetus for amphibian conservation in the Southern High Plains. If basic conservation measures can be implemented at the earliest, we can mitigate urban amphibian declines in this region.

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