WATER QUALITY CAN IMPACT WATER RESOURCE USE BY BATS IN URBAN AREAS

by

I'YANNA SCOTT

Bachelor of Science, 2021
Purdue University
West Lafayette, Indiana

Submitted to the Graduate Faculty of the
College of Science and Engineering
Texas Christian University
in partial fulfillment of the requirements
for the degree of

Master of Science

December 2023

Acknowledgements

I would like to thanks to City of Fort Worth Parks and Recreation, Army Cory of Engineers, Texas Christian university for access to site locations Thank you to my technicians and TCU Bat Lab for helping conduct surveys and with call analysis, including E. Agpalo, I. Acheson, I. Schmeder, J. McGee, C. Price, M. Washington, M. Coleman, T. Craig, G. Serrano, H. Eastburn, D. Scott, M. Oyarzabel, A. Welch, A. Appel, D. McClanahan K. Roussel, L. Foster, and F. Mabry. This research could not have been conducted without funding received from the College of Science and Engineering Graduate SERC Grant Award, and Department of Environmental Sciences Graduate Research Grant Award.

Next, I want to thank Dr. Victoria Bennett for the mentorship and guidance over the 2 years on this research. She helped me to build up my more confidence in fieldwork and working with a team. I would also like to thank my committee members Dr. G. Kharel and Dr. B. Lavy for the helpful feedback and advisement through the production of the thesis.

Lastly, I would like to thank my family and friends for being so supportive of me throughout my project with a huge thank you to my sister D’lisha Scott who became my traveling cheerleader. When there were times that the stress of the program and project would get to me, they were always understanding and helped to motivate me towards the finish line. Without their support I would not have continued to strive for this degree.

Table of Contents

List of Figures

List of Tables

Introduction

Since the 1950s, urbanization and urban sprawl have led and continues to lead to the loss, degradation (the deterioration of resources and habitat; Yakovlev et al. 2015), and fragmentation (the process by which larger contiguous habitat and resources are divided into smaller, isolated patches; Dennis et al. 2013) of natural habitats, such as grasslands, forests, and wetlands, globally (McCauley et al. 2013, Lui & Coomes 2016, Zhang et al. 2021). This continued habitat conversion has impacted wildlife, resulting in reduced resource availability, hindered movement patterns, and altered species distribution and abundance (Pitts et al. 2017, Okamiya & Kusano 2018). Many studies have shown that within urban areas, the diversity of native species is lower compared to more natural habitats (Avolio et al. 2020, Chang et al. 2021), with only common generalist species able to adapt and habituate to these environments (Concepcion et al. 2016, Avolio et al. 2020). For example, the crested pigeon (*Ocyphaps lophote*), southern comma (*Polygonia egea*), and raccoon (*Procyon lotor*) are currently thriving in urban environments (Sol et al. 2017, Daniels et al. 2019, Callaghan et al. 2021), because they are able, as generalists, to persist under a broad range of environmental conditions and take advantage of a variety of resources (Roberts & Stewart 2018). In contrast, the sage skipper (*Muschampia proto*) and tropical kingbird (*Tyrannus melancholicus*) have been declining in urban areas, as they are unable to adapt to the environmental conditions associated with these areas (Campos-Silva & Piratelli 2021). Moreover, like many specialists, they require specific resources that do not tend to be available in urban landscapes (Bourg et al. 2016, Garnick et al. 2016, Ainsworth & Drake 2020). One consequence of species loss is that the regulatory ecosystem services provided by these species (i.e., the ecosystem processes that help to maintain environmental health and stability; (Badola et al. 2015, Velamazan et al. 2020) are lost with them, such as pollination, seed dispersal, and pest control (Sabatier et al. 2013, Ladouceur et al. 2020, Theodorou et al. 2020). In many instances, these services continue to be essential in urban environments and the loss of service providers has wider implications for community health and wellbeing, as well as the local and regional economy (Sandifer et al. 2015, Oosterbroek et al. 2016, Araujo et al. 2021). The pollination of natural habitats, including crops, by native species has been hindered and even prevented by urban development (Hadley et al. 2014, Volpe et al. 2016, Hamblin et al. 2018). Such species are often unable to disperse or migrate through urban areas, which offer little vegetation and often impose harsher conditions than more natural habitats (Banaszak-Cibicka 2014, Yow & Ruggless 2019). Thus, to ensure the aforementioned regulatory ecosystem services continue, it is critical that urban environments are managed to encourage an abundance and diversity of species.

The notion that wildlife in urban areas can be beneficial has led to many researchers and practitioners to determine how such areas can be designed, modified, and managed for wildlife (Plummer et al. 2020, Bassett et al. 2022). Subsequently, an increasing number of studies have demonstrated that urban areas can support a diversity of healthy stable wildlife populations (Magle et al. 2016, Aronson et al. 2017, Gallo et al. 2017). The majority of these studies suggest that the presence of green spaces and green belts provide suitable habitat for wildlife. These undeveloped areas comprising grass, trees, shrubs, and other vegetation that have recreational and aesthetic value, include parks, community gardens, and cemeteries (Jim 2004, Morrison et al. 2016, Wang et al. 2017). However, golf courses (Wurth et al. 2020), brown field sites (Bonthoux et al. 2014), and vineyards (Huysman & Johnson 2021) have also been shown to provide equivalent habitat refuges/oases and resources. The latter broadly refers to food, water, shelter, mating opportunities, and movement corridors (Zhang et al. 2017, Chamberlain et al. 2020, Hansen et al. 2020, Maclagan et al. 2020, Murray et al. 2021). Studies have also shown that increasing and maintaining all these types of resources in green spaces and other equivalent areas can to some extent enhance urban areas for wildlife (Aronson et al. 2017, Santini et al. 2019), while other studies suggest that resource availability throughout the urban landscape (including the surrounding neighborhoods and developed areas) would more effectively enable species to persist (Kuvlesky et al. 2013, Freitas et al. 2020, Shutt & Lees 2021, Teitelbaum et al. 2022). Thus, for urban areas to be suitable for a diversity of species, they need to offer an abundance and variety of resources throughout.

Resource selection is based on the idea that animals rely on a set of criteria, either innate or learned, that help them identify habitat and resource quality (De Labra-Hernandez & Renton 2019, Frommhold et al. 2019, Araujo et al. 2021). Subsequently, we assume that wildlife would select traditional or classic resources (defined here as preferred or widely acknowledged as used in a natural habitat) over anthropogenic equivalents in an urban environment. For example, the crowned eagle (*Stephanoaetus coronatus*) requires shelter in the form of nesting sites typically created in trees within forests and woodland habitats (McPherson et al. 2016a, b). Yet, in an urban environment many of these traditional resources are often removed and those that remain are scarce (Zhang et al. 2018, Harlan et al. 2021). In this environment, the crowned eagle uses anthropogenic alternatives that fulfill their resource selection criteria, such as ledges and sills on buildings with two or more stories (McPherson et al. 2016a, b).

Among the resources used by wildlife, water represents the most homologous resource. In urban areas, traditional water resources would include naturally-formed lakes, creeks, streams, and rivers (Ducey et al. 2018). As part of the urban infrastructure, there are likely to be more semi-natural water sources (i.e., formed by natural processes that are modified or altered by humans), such as drainage ditches (Shaw et al. 2015, Fidino et al. 2016), retention ponds (Haider et al. 2019, Lehrer et al. 2021), and reservoirs (Weinberger et al. 2016, Ogie et al. 2018). Moreover, studies have shown that anthropogenic water features (i.e., created to benefit humans and/or improve the aesthetics of an area; Duke & Soulsbury 2021) can be utilized as alternative water sources by wildlife, such as ornamental ponds and lakes (Li et al. 2013, Wang et al. 2020, Thomaz 2021), water fountains (Covaciu-Marcov & Cicort-Lucaciu 2009, Holzer et al. 2017), drainage gutters (Hoelzinger 2014, Gbogbo et al. 2016), and swimming pools (Kloepper et al. 2019, Nystrom & Bennett 2019). These studies also confirmed that despite these anthropogenic water sources being available nearly all year, wildlife preferentially selected not to exploit them when natural or semi-natural water sources were available and accessible (Yuan et al. 2013, Weinberger et al. 2016, Sievers et al. 2018).

Generally, water availability and accessibility dictate the use of water sources by wildlife. Availability is associated with the abundance of water sources in an area, but it is more complex than this (Magle et al. 2014, Bastille-Rousseau & Wittemyer 2021). Hall and Bennett (2021) found that some water sources are only available to wildlife temporally, because they are ephemeral. The seasonality of water sources is, therefore, an important consideration when estimating resource abundance (Hingrat et al. 2007, Hajek & Knapp 2022). Furthermore, for water to be readily available for wildlife, sources need to be distributed amply across the landscape (Schulz & Ioris 2017, Rich et al. 2019) and studies have shown that biodiversity in urban environments can increase when they are (Bogan et al. 2020, Hansen et al. 2020, Hyseni et al. 2021). This distribution of water is also associated with accessibility, which we define here as the ability of the wildlife to utilize an available water source (Ogutu et al. 2014, Ahlers et al. 2016, Young et al. 2019). Factors that influence accessibility, include structural and functional connectivity. Interconnected tree canopies, shrubs, grassland, and even water surface area can improve structural connectivity (Barbaree et al. 2018, Freitas et al. 2020). Essentially, these features form pathways or movement corridors (such as commuting and migration routes), along which wildlife can travel to access resources (Bennett & Zurcher 2013, McIntyre et al. 2016).

Hedgerows, for example, provide the hazel dormice (*Muscardinus avellanarius*) with cover and shelter from predators as they move from one woodland habitat patch to another (Dondina et al. 2016). Functional connectivity then refers to the specific characteristics of those corridors that allow a species to traverse across them, such as gap distance between tree canopies (Hofman et al. 2018, Perkl et al. 2018). For instance, one study revealed that squirrel gliders (*Petaurus norfolcensis*) cross roads via tree-canopies in an urban setting if the gaps between the canopies are <15 m (van der Ree et al. 2010).

Other characteristics impeding water accessibility, include the size of the water source, water quality, and level of clutter (Hall et al. 2016b, Selebatso et al. 2018, Shute et al. 2021). First, certain species may be restricted from using a water source depending on its size (Reyna-Hurtado et al. 2009, Straka et al. 2016, Roug et al. 2020). Species that drink on the wing (i.e., flying low over water and taking a mouthful from the surface; (Godfrey 1943, Tuttle et al. 2006, McAlexander 2013)), for instance, can be hindered from approaching smaller water sources as they do not have enough room to maneuver effectively (Wester 2014, Segre et al. 2015, Mero et al. 2020). More specifically, a study conducted in the Mojave and Great Desert Basin demonstrated that the spotted bat (*Euderma maculatum*), a species known to be less maneuverable (i.e., had a wing morphology with higher aspect ratio and wing loading; Altringham 2011) drank ~70% less often at water sources <2 m in length in comparison to sources >2 m (Hall et al. 2016a). Moreover, the size of natural and semi-natural water sources can change substantially across seasons, with the amount of surface area decreasing when conditions are hot and dry, and increasing when rainfall is more prevalent (De Jong et al. 2015, Sanchez-Montoya et al. 2017, Hall & Bennett 2021). Such seasonal variations in surface area will affect the ability of specific species to access water (Nyberg & Lerner 2000, Naidoo et al. 2020).

Similarly, the quality of the water could dictate whether wildlife utilize a source or not (Huntsinger et al. 2017, Ovalle-Rivera et al. 2020). Broadly, water quality represents the amount of contamination within a waterbody and is defined by the Environmental Protection Agency (EPA) as the pollution of water by biological, chemical and radiological substances that make the water unsuitable for drinking and other activities (EPA 2021). Common substances that pollute water, include oils, heavy metals, fertilizers, pesticides and other chemicals, organic and inorganic materials, and litter (Rosevelt et al. 2013, Reid et al. 2018, Pamuru et al. 2022). In an urban environment, local water sources are readily exposed to all these types of contaminants (Saifur & Gardner 2021). Moreover, urban areas are subject to higher levels of contamination, as activities conducted in these higher populous areas generate more of these contaminants that in turn collect and concentrate on the impervious surfaces. It is the impervious surfaces that alter the water cycle by reducing infiltration (i.e., the movement of water into the ground from the surface) and creating more runoff (i.e., the movement of water over the surface of the land; Fig. 1; Sambito et al. 2021). Subsequently, during rain events the accumulated pollutants are transported via runoff directly into local surface water sources, such as lakes, streams, rivers, and wetlands (known as nonpoint source pollution; Becker & Pinheiro 2019, Koutnik et al. 2021, Xue et al. 2022). Studies have shown that local surface water sources, such as retention ponds and drainage ditches, in cities can have concentrations of heavy metals, phosphorus, and other nutrients above levels that are considered safe for humans as a result of nonpoint source pollution (Litke 1999, Neumann et al. 2021, EPA 2022a, 2023).

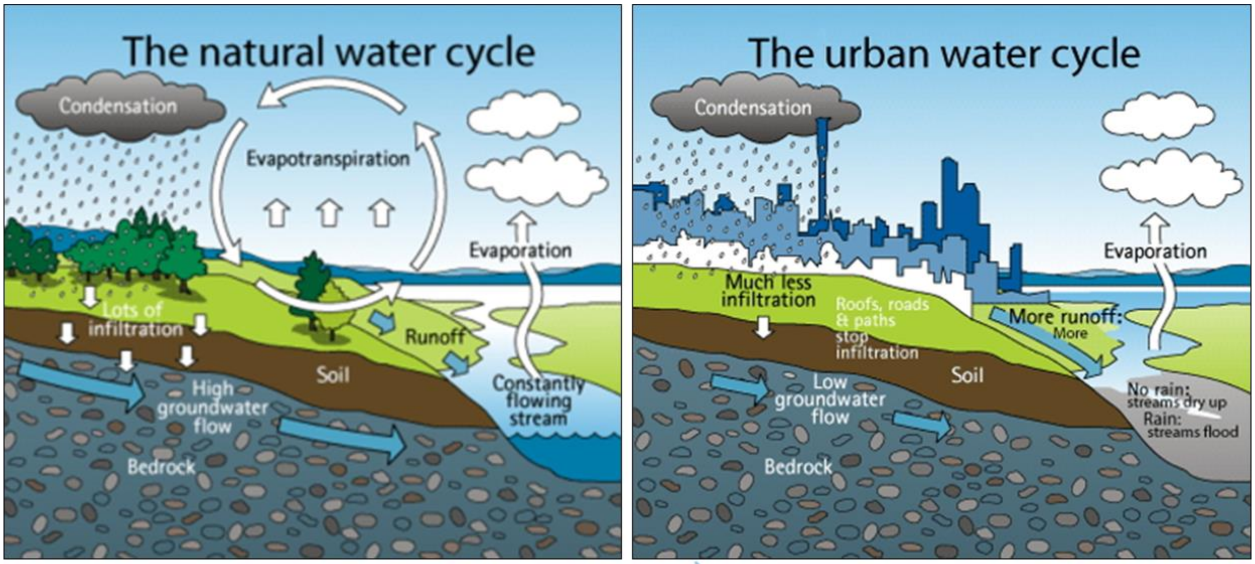


Figure 1: Illustration comparing natural and urban water cycles. Adapted from greaterauckland.org.nz.

Thus, if water sources in urban areas are deemed unsafe for humans (see standards set out by the EPA (2023)) then these polluted water sources have the potential to negatively impact wildlife as well (Burgmeier et al. 2011, Shaw et al. 2015). Research has revealed that the presence of contaminants in water, and therefore water quality, can influence the abundance and distribution, breeding success, and survival of wildlife (Bishop et al. 2000, Croft-White et al. 2017, Clevenot et al. 2018). The abundance and species diversity of macroinvertebrates, for example, decreased in wetlands with higher levels of contamination in urban areas in southern California (Brown et al. 2010). Nevertheless, the majority of studies that explore the impacts of water quality on wildlife focus on species that have life history stages or conduct certain activities exclusively in, on, or in close proximity to water, such as amphibians, fish, aquatic invertebrates, and waterfowl (Pettitt et al. 2018, Perron & Pick 2020, Wilson et al. 2021). In urban environments, most studies tend to investigate the impact of water quality on waterfowl (Schoch et al. 2014, Schulwitz et al. 2015, Murray et al. 2021). Furthermore, to date few studies consider species that only access water resources as part of daily or nightly activity routines, such as drinking and foraging (Tigas et al. 2002, Peterson et al. 2005, Cotner & Schooley 2011). Among those studies that have explored the consequences of urban water pollution on other taxa, these have included terrestrial reptiles, white-tailed deer (*Odocoileus virginianus*), and elephants (*Loxodonta africana*; Berry et al. 2013, Bastille-Rousseau & Wittemyer 2021). In all these studies, the surveys undertaken explore the impact of water quality on the presence/absence of species and/or decrease in abundance (Ovalle-Rivera et al. 2020, Tang et al. 2021). For example, in the river Yamuna in Delhi waterfowl diversity decreased with the deterioration of water quality (Rehman et al. 2021). However, these surveys do not directly demonstrate resource use (Li & Kalcounis-Rueppell 2018, Sievers et al. 2018, Rehman et al. 2021). Thus, the objective of the following study was to assess whether water quality influences the direct use of water sources by terrestrial wildlife in an urban environment.

One taxonomic group that can be used to determine environmental health is bats (Korine et al. 2015, Rachwald 2019). Studies have shown that they are a good indicator of resource availability, biodiversity, environmental disturbance, and water quality (Cunto & Bernard 2012, Lopez-Baucells et al. 2017, Li & Kalcounis-Rueppell 2018, Tuneu-Corral et al. 2020). We, therefore, hypothesize that water sources with higher water quality will have an abundant and diverse community of bats using them (i.e., foraging and drinking), while lower quality water sources will have little to no bat activity and lower species diversity. To explore this hypothesis, we conducted a study using thermal cameras and acoustic monitoring to determine whether water quality has discernible influences for water resource use by bats (i.e., foraging and drinking activities) at water sources in six urban parks and greenspace in Fort Worth, Texas.

Understanding how the water quality of urban sources impacts bats, may not only be used as an indicator of water availability for other wildlife species in urban areas, but also provide insights into the environmental health of local parks and surrounding neighborhoods. Such insights can in turn help identify and prioritize areas of concern in Fort Worth and other similar urban areas across the United States.

Methods

Study Sites

For this study, we selected six water sources in local parks and greenspaces within the Lower West Fork Trinity watershed (HUC 12030102) located in Tarrant, Johnson, Dallas, and Parker Counties, north central Texas (Fig. 2). We chose our water sources (i.e., the study sites) based on 1) known water quality and 2) their suitability for bats. Within the Lower West Fork Trinity watershed, we selected sites within two different subwatersheds that are known to vary in water quality, based on the EPA's water quality listings (EPA 2022b). The first was the Rock Creek subwatershed (HUC 120301020304; hereafter referred to as RCSW) located in Johnson and Tarrant Counties. RCSW covers an area of 97.61 km^2 with a primary stream, Rocky Creek, which flows northeast into Benbrook Lake (a 15.26 km^2 impounded reservoir). The watershed area was predominately rural, comprising broad rangeland areas interspersed with arable land and urban development in the form of townships (TCEQ 2022). From the EPA's water quality listings, RCSW has no water quality impairments for either humans or wildlife (EPA 2022g).

Our second subwatershed was the Lake Como-Clear Fork Trinity River subwatershed (HUC 120301020307; hereafter referred to as TRSW), located in Tarrant County. TRSW covered an area of 101.43 km^2, including the Trinity River Basin and Lake Como (USGS 2022). The general flow of the watershed heads southeast and continues towards Trinity Bay, which drains into the Gulf of Mexico. The Trinity River represents a perennial freshwater stream that flows through the southwest corner of city of Fort Worth, ranked the 13th largest and one of the fastest-growing cities in the U.S. (www.fortworthtexas.gov).

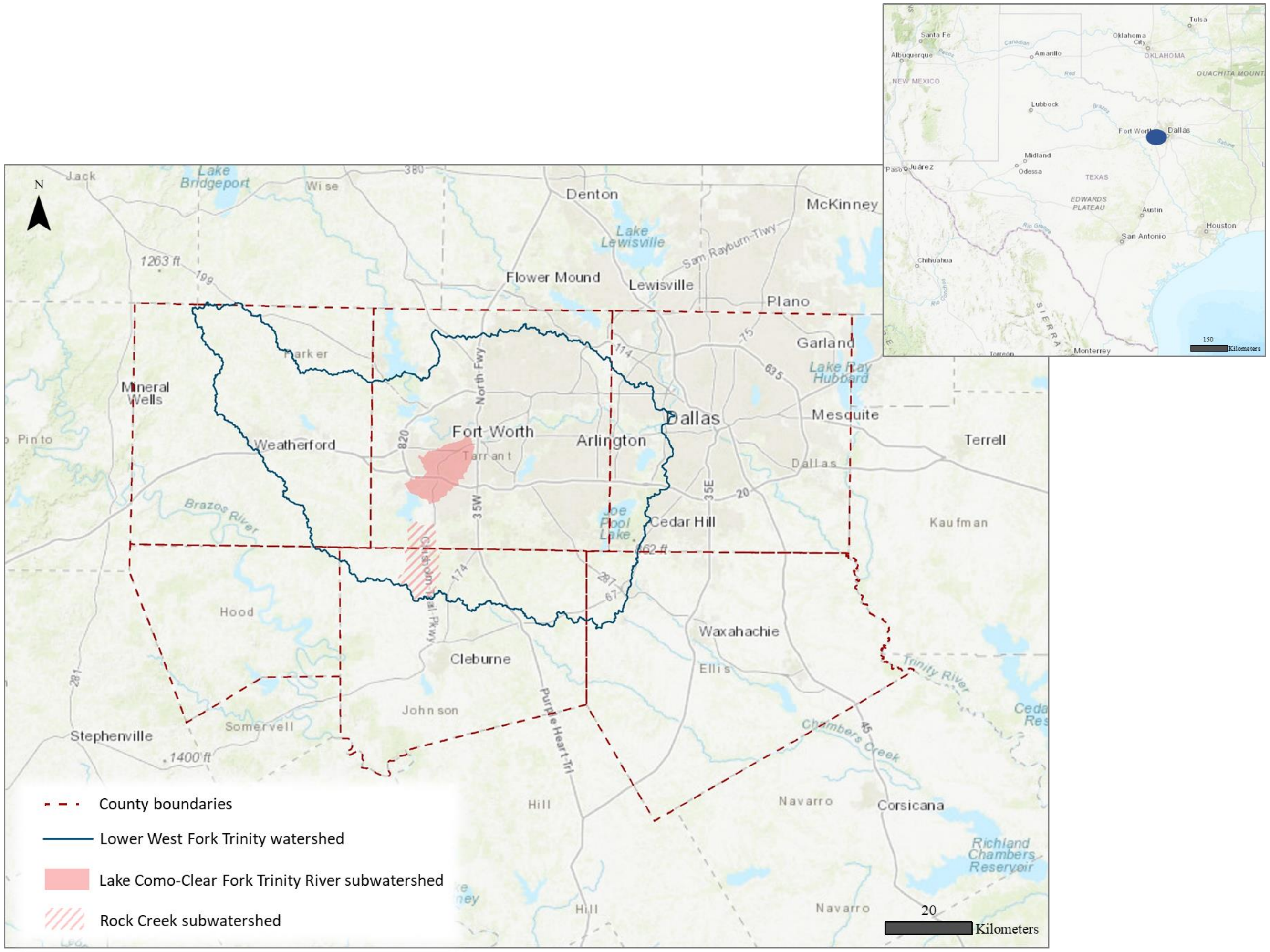


Figure 2: Map of the Lower West Fork Trinity watershed and associated counties in Texas, U.S.A. Sites used in this study were located in the Rock Creek and Lake Como-Clear Fork Trinity River subwatersheds

We selected six survey sites at water sources in areas where bats were known to be active (unpublished data; (Bienz 2016, Smith 2019); Fig. 3). We then ranked these sources based on their runoff and infiltration potentials leading to different pollution contributions. To determine runoff and infiltration potential for each water source, we created a watershed map that depicted the drainage to each source within a 2 km radius. For this, we first stitched available Digital Elevation Models (DEMs) for our entire study area using Mosaic-to-New-Raster in the *Raster Toolset* in the Data Management Toolbox in ArcMap version 10.6 (ESRI Inc., Redlands, CA). We then extracted a 2 km area around each survey site location (Appendix A, Fig. A1). Using the *Hydrology Toolset* in the Spatial Analyst Toolbox, we inserted each 2 km area into the Fill Tool, which clean imperfections in the raster. We then used the Flow Direction Tool to create a raster of flow direction for each 2 km area (Appendix A, Fig. A2). Applying this raster, we used the Flow Accumulation Tool to create a second raster of accumulated flow for each 2 km area and selected the flow accumulation line closest to each water source to identify the outlet point for the catchment (Appendix A, Fig. A3). These outlet points along with the flow direction raster were then input into the Watershed Tool to create a watershed raster for each 2 km area (Appendix A, Fig. A4). We then conducted a spatial analysis using three geospatial datasets that could be used to identify and categorize runoff and infiltration potential; 1) DEMs for the areas surrounding the water sources to determine slope which dictates the direction and flow of runoff, 2) National Land Cover Database classes to categorize landuse surrounding the water sources to identify potential levels of runoff and infiltration, and 3) Soil Survey Geographic Database (SSURGO) to classify and map the soil characteristics related to water infiltration within the area surrounding each water source. Using the DEM raster created above, we used the Slope Tool in the *Surface Toolset* in the Spatial Analyst Toolbox to create a raster displaying variations in slope (gradient) across the study area. We then reclassified this raster in the *Reclassify Toolset* to represent five ranked categories delineating slope; 1) 0% to ≤1%, 2) >1% to ≤2%, 3) >2% to ≤3%, 4) >3% to ≤5%, and 5) >5% (hereafter referred to as the Slope raster; Appendix A, Fig. A5). Next, we reclassified the land cover data to display eight categories; 1) forested areas, 2) wetlands, 3) grasslands, 4) agriculture, 5) barren land/developed open spaces, 6) developed low intensity, 7) developed medium intensity, and 8) developed high intensity (hereafter referred to as the Landuse raster; Appendix A, Fig. A6). For the last geospatial dataset, we converted the SURRGO data into a raster using the polygon-to-raster tool in the *From Raster Toolset* in the Conversion Toolbox. We then reclassified this raster to display four categories based on hydrological soil groups; 1) A, 2) B, 3) C, and 4) D (hereafter referred to as the Soil raster; Appendix A, Fig. A7).

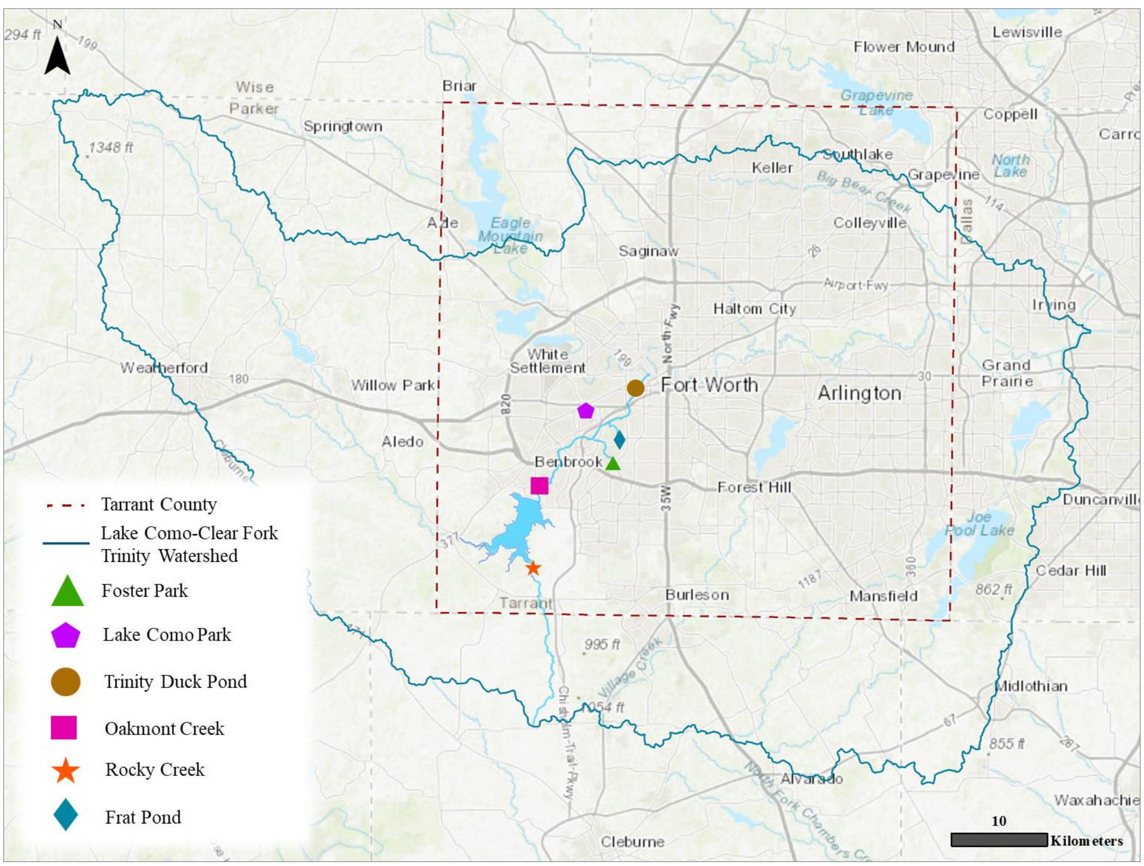


Figure 3: Map of the six survey site locations in Tarrant County, Texas, U.S.A. surveyed from March to September 2021 to 2022.

Lastly, we used extracted-by-mask tool to extract the watershed for each survey site from the Slope, Landuse, and Soil rasters. We then performed a weighted overlay in Weighted Overlay Tool in the *Overlay Toolset* in the Spatial Analyst Toolbox, in which we assigned Slope and Landuse with an influence of 40% each and Soil 20%. Using the ranks for each raster and weighted influence, this spatial analysis tool combined and reclassified these inputs into a single output raster with six categories, ranging from lowest pollution potential (i.e., lowest runoff and highest infiltration), low, moderately low, moderately high, high and highest (i.e., highest runoff and lowest infiltration; hereafter referred to as the Suitability raster; Appendix A, Fig. A8). Using the Suitability raster, we summed the total area (m^2) for each category within each watershed and calculated the percentage each category represented. We then summed the percentage of the high and highest categories (indicating the greatest pollution potential) and ranked the survey sites based on this percentage with the highest percentage potentially representing the site with the most polluted water source and the lowest representing the least polluted water source (Table 1). Each study site is, thus, described below in order of pollution potential (best to worst).

Table 1: Survey sites ranked by their pollution potential (best to worst). Included are the Area and Percent coverage for each category given by the Suitability raster. The high and highest categories are shown in grey.

	Trinity Duck Pond		Rocky Creek		Oakmont Creek		Foster Park Pond		Frat Pond		Lake Como	
	Area (m^2)	Percent (%)	Area (m^2)	Percent (%)	Area (m^2)	Percent (%)	Area (m^2)	Percent (%)	Area (m^2)	Percent (%)	Area (m^2)	Percent (%)
Lowest	0	0	0	0	0	0	0	0	0	0	0	0
Low	0	0	211727	5.7	11201	0.9	0.0	0.0	1140	0.04	0	0
Moderately Low	0	0	2950391	79.3	86615	6.7	0.0	0.0	12389	0.5	0	0
Moderately High	0	0	476651	12.8	830364	64.5	249308	56.1	1236218	47.1	96684	21.9
High	0	0	80731	2.2	358293	27.8	191036	43.0	1321511	50.4	306053	69.3
Highest	0	0	541	0.01	1311	0.1	4198	0.9	51836	2.0	38881	8.8
Pollution Potential	**0**		**2.2**		**27.9**		**43.9**		**52.4**		**78.1**	

Site 1

Our first study site was located in Trinity Park (32°44'53.8"N 97°21'02.4"W; Fig. 4A). The 1.01 km^2 park comprises an urban green space in downtown Fort Worth, which was owned and operated by City of Fort Worth Park and Recreation Department (COFW 2022d). The park, centered around the Trinity River and associated riparian habitat, was intended for community recreation with a network of biking, running, and walking trails and designated fishing areas. Subsequently, the landscape represents heavily manicured grassland interspersed with mature trees, including cedar elm (*Ulmus crassifolia*), bur oak (*Quercus macrocarpa*), Texas live oak (*Quercus fusiformis*), American sycamore (*Platanus occidentalis*), and pecan (*Carya illinoinensis*), and a few small patches of maintained shrubs (Nesom 2013). Toward the north end of the park there is a 20 m^2 artificial pond, known as the Trinity Duck Pond (Fig. 4B). This pond, which acts as a recreational duck pond for the surrounding highly developed downtown area, however the pond is a closed system and run-off from the surrounding area does not drain into the pond. The site received annual maintenance in which the pond is drained and refilled with the city water supply. The pond was used by migratory and resident waterfowl species, including American shoveler (*Spatula clypeata),* American wigeon (*Mareca americana*), mallard (*Anas platyrhynchos*), and lesser scaup (*Aythya affinis*). Previous mist netting surveys and acoustic monitoring conducted in the park and surrounding area from 2015 to 2018 confirmed the presence of all six local species (unpublished data;(Smith 2019).

.

Figure 4: Trinity Park in Tarrant County, Texas. **A** Shows the map of the park and surrounding areas and **B** shows a picture of the study site at Trinity Duck Pond.

Site 2

Our second site was located in Rocky Creek Park (32°35'38.2"N 97°27'14.1"W; Fig. 5A). The 1.45 km^2 park was one of seven that surrounded Benbrook Lake reservoir, which were owned and operated by the U.S. Army Corps of Engineers (USCOE 2022). While the park was created and maintained for recreation, primarily fishing, camping, and equestrian trail riding, the area immediately surrounding the park comprised 0.95 km^2 of managed prairie intended to promote the conservation of native species (GPRC 2013). Within the park, we selected a site along the Rocky Creek tributary at the southern tip of the park (Fig. 5B). According to the EPA (2022d), there were no water quality impairments associated with this tributary. Moreover, both banks of the stream channel had contiguous lines of mature trees, composed of eastern cottonwood (*Populus deltoides)*, Texas ash (*Fraxinus texensis*), American elm (*Ulmus americana*), chinkapin oak (*Quercus muehlenbergii*), and cedar elm, providing roosting opportunities and connectivity for local bats. Previous mist netting surveys and acoustic monitoring conducted at the site from 2013 to 2018 confirmed the presence of all six species known to be in the area: evening (*Nycticeius humeralis*), eastern red (*Lasiurus borealis*), hoary (*Aeorestes cinereus*), Mexican free-tailed (*Tadarida brasiliensis mexicana*), silver-haired (*Lasionycteris noctivagans*), and tri-colored bats (*Perimyotis subflavus*) (unpublished data; Ammerman et al. 2012).

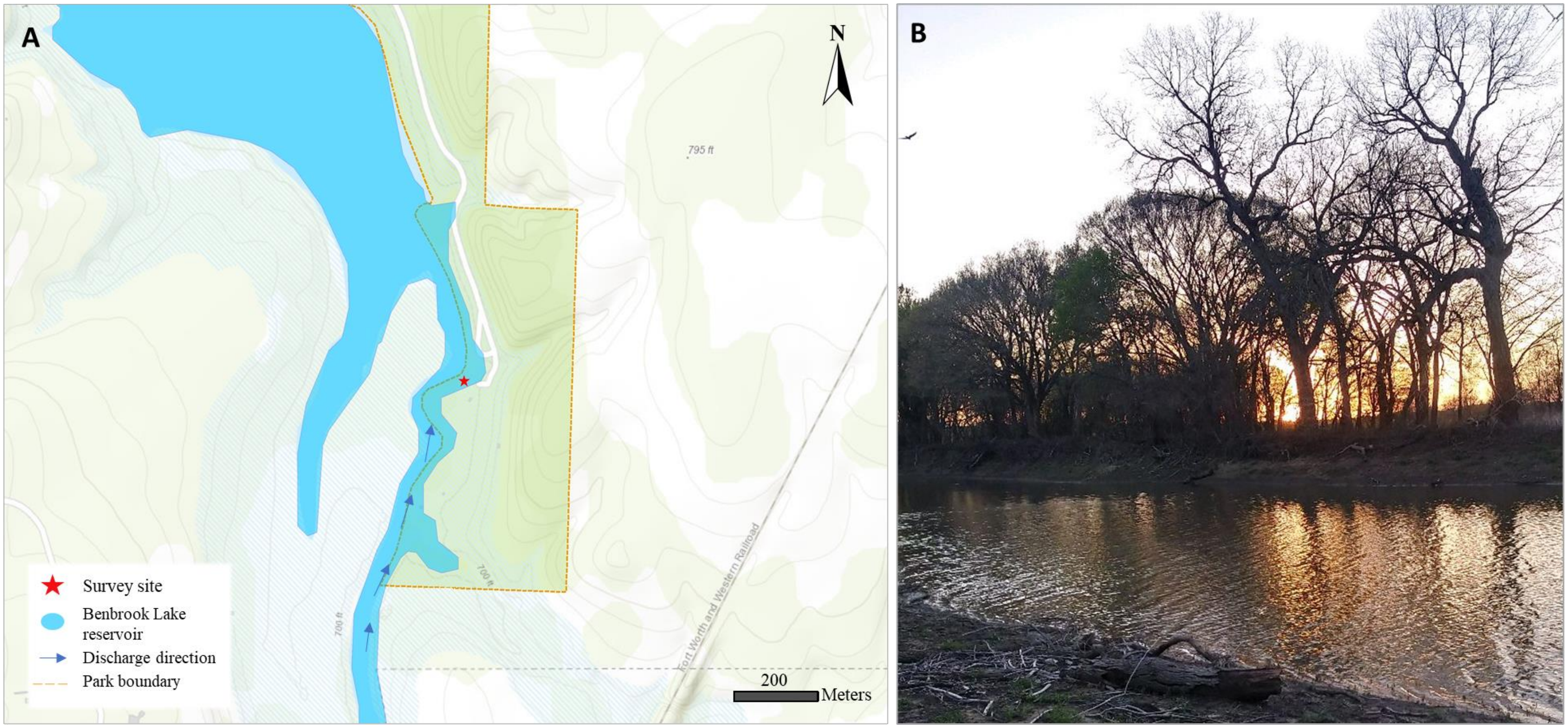


Figure 5: Rocky Creek Park in Tarrant County, Texas. **A** Shows the map of the park and surrounding areas and **B** shows a picture of the study site at the Rocky Creek tributary.

Site 3

Our third study site was at Oakmont Park (32°40'12.5"N 97°25'53.6"W), which was owned and operated by City of Fort Worth Park and Recreation Department (COFW 2022c); Fig. 6A). The 0.51 km^2 park stretches along the Trinity River, 1.5 km from where it flows from the dam on the northeastern edge of Benbrook Lake. The area surrounding includes the Pecan Valley golf course to the southwest and subdivisions of the city of Benbrook encompassing the rest of the park.

Oakmont provides recreational opportunities for this surrounding community in the form of playgrounds, walking, running, and bike trails, and fishing areas that have been intermittently stocked by Texas Parks and Wildlife Department (TPWD) with various fish species since the mid-1980s, including rainbow trout (*Oncorhynchus mykiss*; TPWD 2022). Subsequently, the park is heavily manicured grassland and interspersed by mature trees surrounding the playgrounds, and while understory vegetation and mature trees created riparian habitat along the river edge, including American elm, roughleaf dogwood (*Cornus drummondii*), eastern redbud (*Cercis canadensis*), Texas live oak, and American sycamore (Barnett et al. 2016). Within Oakmont Park, we selected a section of the river at the Art Cowsen Trailhead to the western edge of the park as the closest location to the Benbrook dam (Fig. 6B). At this location the EPA (2022e) designated the river to be impaired with dioxins and PCBs, both registering above regulated MCL. Previous mist netting surveys and acoustic monitoring conducted at the park from 2017 through to 2019 confirmed the presence of all six species known to be in the area (unpublished data;(Smith 2019).

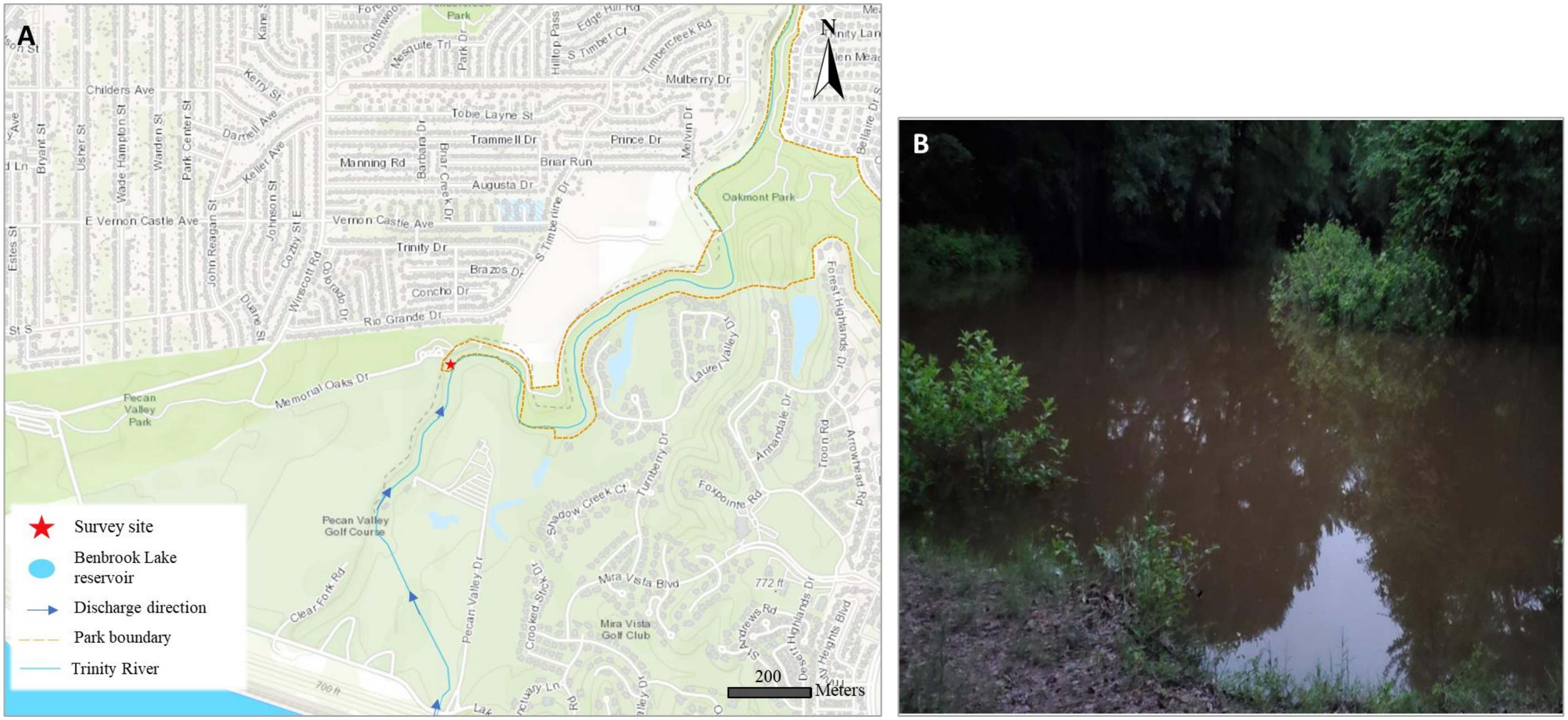


Figure 6: Art Cowsen Trailhead at Oakmont Park in Tarrant County, Texas. **A** shows the map of the park and surrounding areas and **B** shows a picture of the study site along the Clear Fork Trinity River.

Site 4

Our fourth study site was at Foster Park (32°41'02.9"N 97°22'26.6"W), which was owned and operated by City of Fort Worth Park and Recreation Department (Fig. 7A; COFW 2022a). The park covered a 0.48 km^2 linear tree-lined green space centered around a drainage ditch and a ~18 m by 6 m retention pond when at capacity. It is part of a park system (Kellis, Foster and Overton Parks) that provides drainage for the neighborhood, a suburban subdivision of ranch-style single-story housing (common to the region) built in the 1950s along with a walking trail (FWPR 2014). This linear greenbelt comprised heavily manicured grassland interspersed with mature trees (>6 m in height) extending from the drainage ditch, while understory vegetation and mature trees created riparian habitat along the ditch Tree species include bur oak, cedar elm, American elm, common hackberry (*Celtis occidetalis*), and Texas ash. Collectively, these trees offer roosting opportunities for bats (Hall 2020). Within Foster Park, we selected the 108 m^2 retention pond for our third study site (Fig. 7B). While there are no water quality measures available for the pond, its drainage system discharges directly into the Trinity River at the northwestern edge of the neighborhood, where the EPA (2022c)designated this portion of the river to be impaired with Escherichia coli, dioxin herbicides, and polychlorinated biphenyls (PCB) above maximum contaminant levels (MCL; i.e., levels of contamination exceeding 0.05 mg/l, 3 x 108 mg/l, and 5 x 10-4 ppm, respectively). Previous mist netting surveys and acoustic monitoring conducted at the park from 2013 through to 2022 confirmed the presence of all six species known to be in the area, as well as the canyon (*Parastrellus hesperus*), a species thought to have recently expanded its range (Bienz 2016, Agpalo 2020, Hall 2020). More specifically, Nystrom and Bennett (2019) recorded all seven species actively flying over the retention pond in acoustic monitoring surveys and evening, eastern red, hoary and silver-haired bats were observed regularly drinking from the retention pond.

Figure 7: Foster Park in Tarrant County, Texas. **A** Shows the map of the park and surrounding areas and **B** shows a picture of the study site at the Foster Retention Pond.

Site 5

Our Site 5 was on Texas Christian University campus (32°42'14.9"N 97°22'04.0"W; Fig. 8A) by their sports field. This large retention pond acts as drainage for the surrounding subdivisions and the campus community, including the Greek Village, soccer pitch, tennis courts, and athletics field. Tree-lined drainage ditches enter from the subdivision to the south and extend north from the pond to the Trinity River and contained mature American Sycamore, Texas live oak, chinkapin oak, and pecan (TCU 2015). In contrast, the area immediately surrounding the pond comprised manicured grass with a few trees. Walls and fencing in the immediate vicinity provided equivalent commuting routes for bats potentially allowing them access the water source (Entwistle et al. 1996, Foxley et al. 2022). At the pond, we selected a study site at its northern end where the pond flows into the tree-lined drainage drain under a bridge and into a culvert, which also can be used by bats to access the water source (Fig. 8B; Grindal 1999). While there were no water quality measures available for the pond, it was part of a drainage system that discharged into the Trinity River. This portion of the river was designated as impaired with MCLs above *E. coli*, dioxin, and PCBs EPA (2022c). Acoustic monitoring conducted in the area adjacent to the pond from 2014-2017 confirmed an abundance of bat activity, including evening, eastern red, hoary, Mexican free-tailed, silver-haired, and tri-colored bats, while preliminary surveys undertaken in 2020 revealed little bat activity over the pond itself (unpublished data).

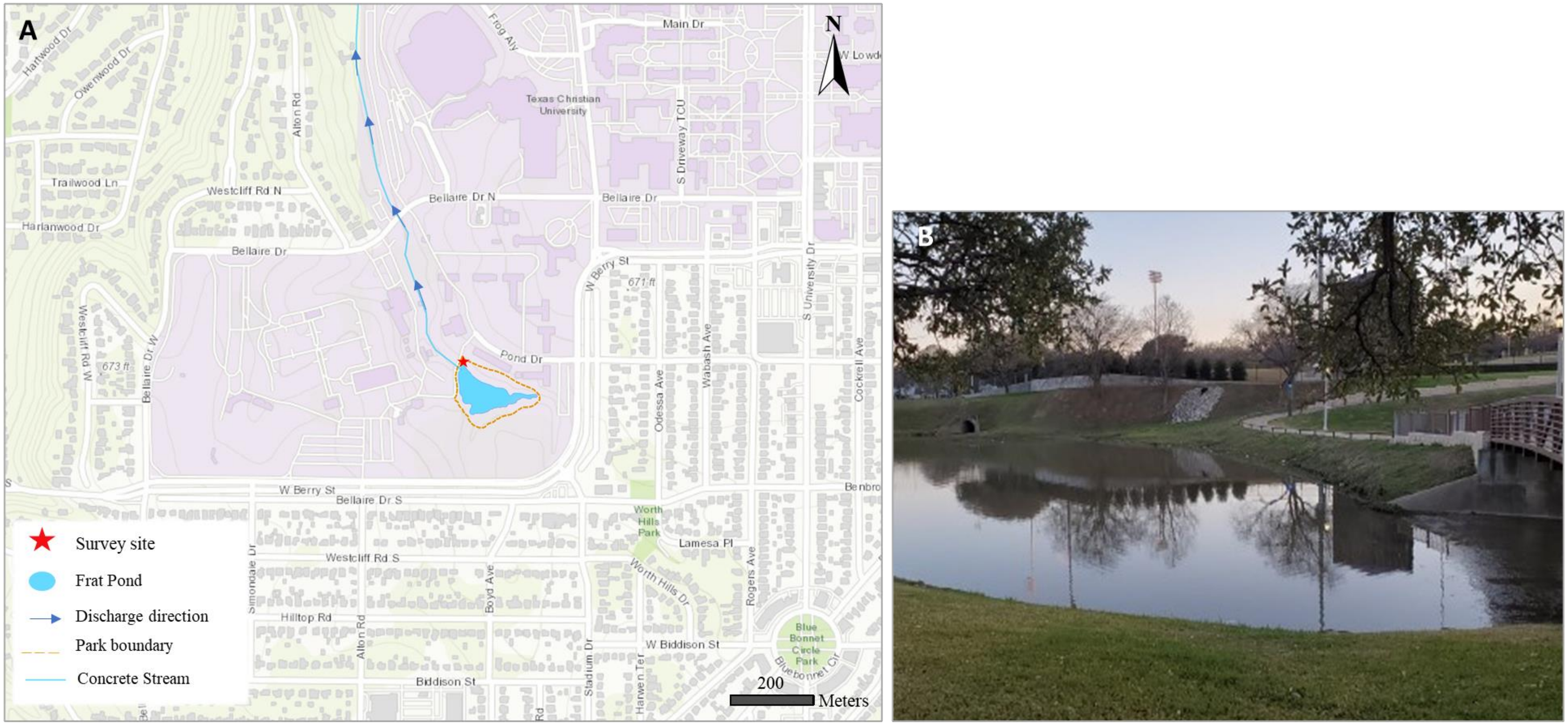


Figure 8: Texas Christian University campus in Tarrant County, Texas. **A** Shows the map of the area and **B** shows a picture of the study site at the Frat retention pond.

Site 6

Our sixth study site was at Lake Como Park (32°43'45.5"N 97°24'06.6"W), which was owned and operated by City of Fort Worth Park and Recreation Department (COFW 2022b); Fig. 9A). The 0.24 km^2 park was built in 1952 as a recreation resort for the surrounding subdivisions; Como, Sunset Heights South, West Beyer and East Libbey. It is located a few blocks south of Interstate 30 and west of Hulen St, which has the communities water drain towards it. Recreational activity in the area includes fishing, biking, walking, and running trails. TPWD stocked the lake with channel catfish (*Ictalurus punctatus)* and bluegills (*Lepomis macrochirus*) in 2022 (TPW 2022). As part of the Dallas-Fort Worth Metroplex, sources of pollution include domestic wastewater and stormwater discharge, industrial wastewater and stormwater discharge, and animal feeding operations (TCEQ 2013). However, the EPA (2022f) reported the water at this site to be impaired with dioxins, PCBs and dieldrin (a chemical pesticide that was banned in 1987 (EPA 2003); MCL; i.e., levels of contamination exceeding ~3 x 10^{-5} ppb). Based on this reporting, the EPA determined that fish and shellfish in local water sources are unfit for consumption due to the bioaccumulation of these toxins. The tree species in the area include American sycamore, American elm, Texas live oak, cedar elm, and Texas ash (Inaturalist 2022). The park managed vegetation through mowing and tree trimming. Within the park, we selected our site to be at the northwest end of the lake because the proximity of the trees facilitated the accessibility to the pond for bats (Fig. 9B). Note that no previous surveys for bats had been conducted in this area prior to our study.

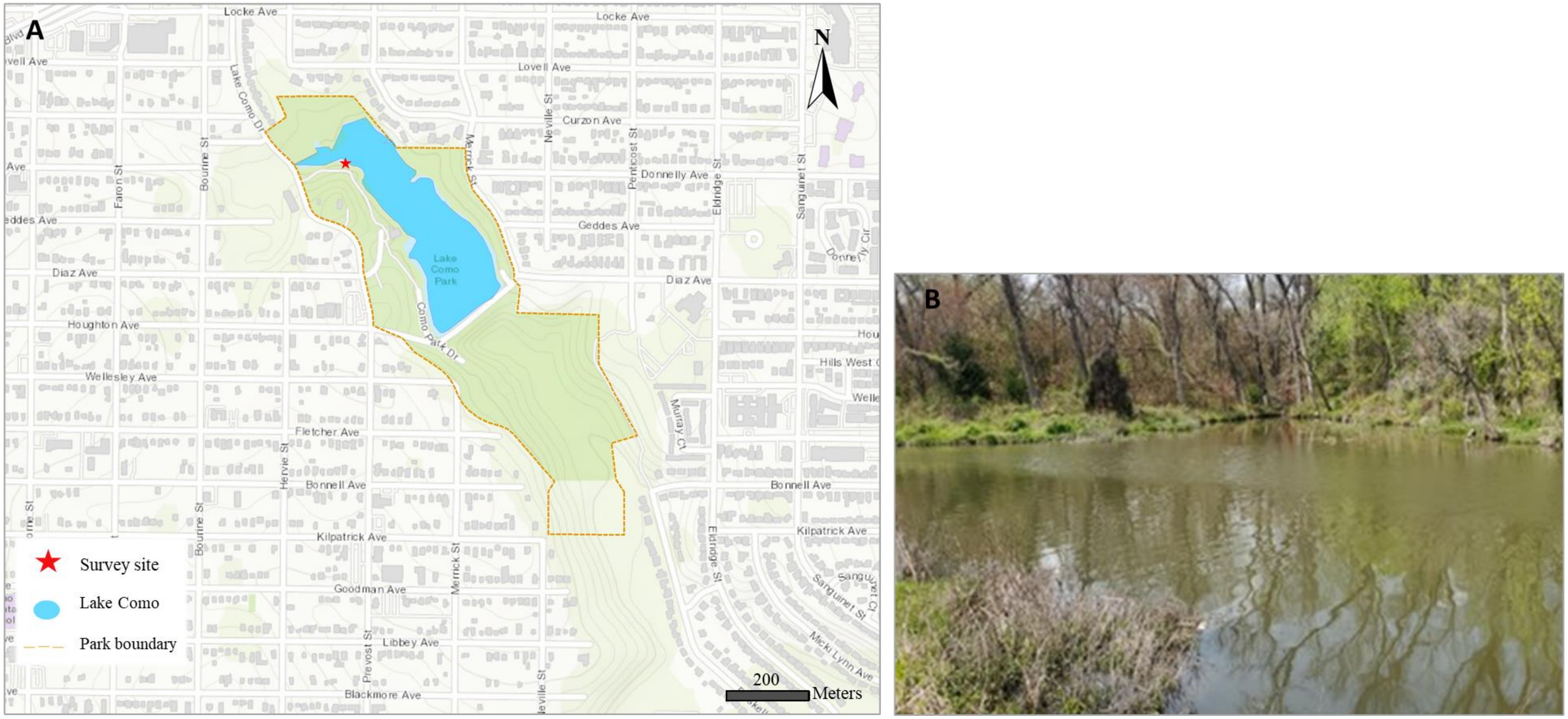


Figure 9: Lake Como Park in Tarrant County, Texas. **A)** Shows the map of the park and surrounding areas and **B)** shows a picture of the study site at Lake Como.

Behavioral Observation Surveys and Acoustic Monitoring

We conducted behavioral observation and acoustic surveys at the six aforementioned study sites (Fig. 3) from March to September 2021 and 2022 to encompass the bat summer activity period for the study area (Fern et al. 2018, Hall & Bennett 2021). As environmental conditions can strongly influence daily bat activity, we performed a paired study in which we surveyed two study sites simultaneously. Furthermore, as certain weather conditions were known to inhibit bat activity, surveys were not conducted when temperatures were <5℃, it was raining, or wind speeds >24 km/hr (Bienz 2016). On these occasions, we rescheduled for the next available night to ensure that all ponds were surveyed consistently throughout the season. Thus, we surveyed each study site once every two weeks with one to two technicians at each site.

At least 30 mins before starting surveys, we set up the following equipment at the two study sites (Fig. 10). At each site, we used one Axis Q1942-E 19mm ThermNetCam 30 FPS (Axis Communications, Lund, Sweden) surveillance camera, configured within the ~9,000-14,000 micrometers infrared spectrum as recommended in Huzzen et al. (2020). The thermals were set to the "Ice-and-Fire" false-color scheme setting, a 640 by 480 pixel resolution, and a 30 frames/sec sampling rate. We placed the thermals on top of a 15-gal tote container and secured it using bungee cords and beanbags, if necessary. Finally, to operate the thermal cameras and record bat activity, we used a HP Compaq 8510w laptop computer installed with a video recording software; Axis Companion (ver. 3.72, Axis Communications AB, Sweden). A 300 W lithium battery was used to power the laptop and thermal camera through a Netgear ProSAFE 8-Port Fast Ethernet PoE Switch and 10 ft and 25 ft ethernet cables, respectively. Note we did not require any supplemental lighting.

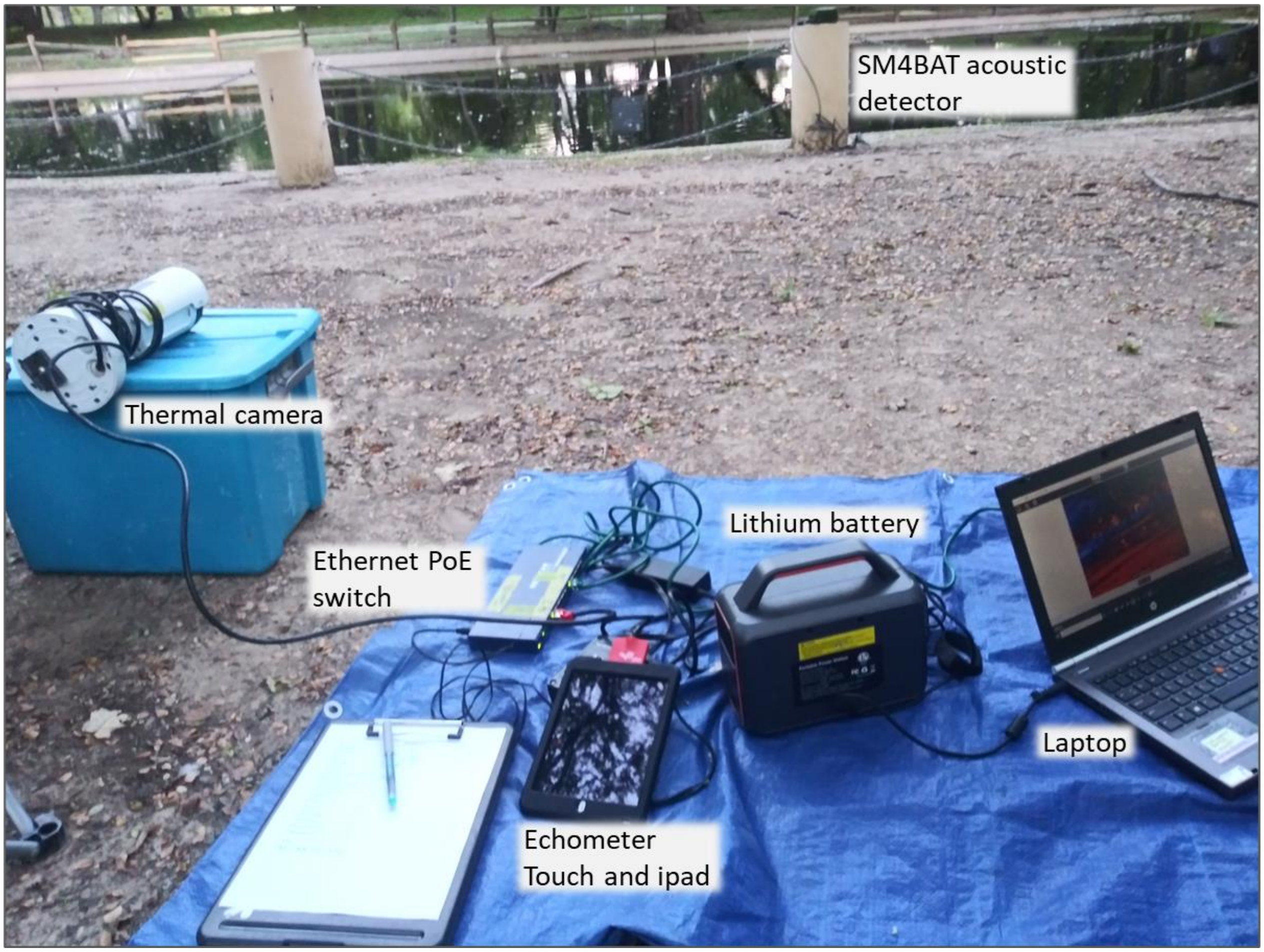
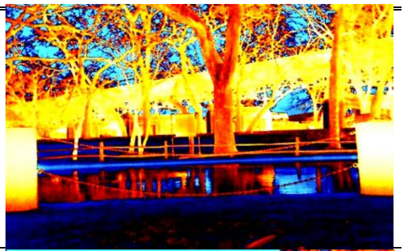
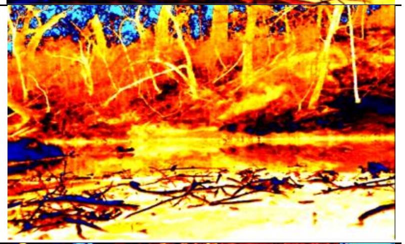
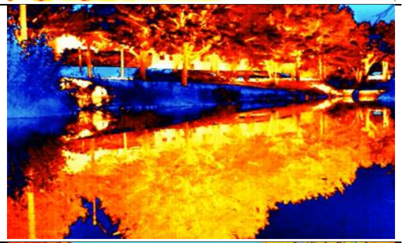


Figure 10: Thermal and acoustic equipment set up.

Each field-of-view represented a view of the pond ~10 m away from the edge of the water. We kept the camera placement consistent between all study sites (independent of pond size) to minimize any bias in data collection caused by variations in scale (Table 2).

Table 2: Thermal camera field-of-views for each of the six study sites.

Study Site	Field-of-view
Site 1: Trinity Duck Pond	
Site 2: Rocky Creek	
Site 3: Oakmont Creek	
Site 4: Foster Park Pond	
Site 5: Frat Pond	
Site 6: Lake Como	

We also used a Song Meter SM4BAT acoustic detector with an external U2 ultrasonic microphone from Wildlife Acoustics (Maynard, Massachusetts) to record bat echolocation calls at the study sites during the surveys. We placed the detector setup at the edge of the pond with its microphone angled toward the surface of the water (Fig. 9) and programmed it to record bats concurrently with the behavioral observation surveys. For this, we synchronized the internal clock on the detector with the time shown on Axis Companion prior to the start of the surveys. In addition, the SM4BAT was set to trigger at the frequencies between 16 kHz and 192 kHz with a 3-sec delay between recordings. We selected this frequency range to encompass the echolocation frequencies of known bat species within Fort Worth (Nystrom & Bennett 2019, Krejsa et al. 2020). Gain threshold was set at 12.0 dB with a trigger volume of 12.0 dB and sound files were recorded in 4-sec standard wav files (.wav). We saved all files created onto 32 GB SD cards with the sample rate at 256 kHz and D-batteries were used to power the detector. Note that although the microphone was directional, it could detect bat acoustic calls from areas outside the field of view of the thermal cameras. Finally, we used an iPad with an Echometer Touch ultrasonic microphone module from Wildlife Acoustics (hereafter referred to as an Echometer) to aid thermal footage processing (see below) and as a backup to the SM4BAT. This hand-held acoustic detector was used by the technicians during the surveys to observe and record real-time observations of bat activity both acoustic via the Echometer Touch app, visually in the thermal camera field-of-view, and in some instances with the naked eye.

We began surveys 20 min after sunset (i.e., dusk) and continued for ~1 hr to incorporate the primary activity period when local bats search for and drink water (i.e., soon after the bats emerged from their roosts (McAlexander 2013). During the 1-hr surveys, we recorded bat activity at the survey sites in six 10-min sessions, primarily to aid with the viewing and processing of the video footage post-survey, however, it also allowed technicians to adjust and accommodate any equipment or recording malfunctions that might have occurred between sessions. In each session, one technician noted whether bat echolocation calls were observed on the Echometer Touch app, while the other technician noted the times bats were observed in the field-of-view and specifically observed drinking in the Axis Companion viewer on the laptop.

Note we also recorded the following data at the start of each survey using WeatherBug and Lunar Phase applications: temperature (^{o}C), average wind speed (km/h), gust speed (km/h), wind direction (cardinal), humidity (%), dew point (^{o}C), barometric pressure (mb), cloud cover (full, partial, or clear), moon phase, moon illumination (%), and whether the moon was visible or not. These variables were recorded as studies have shown that bat activity can be influenced by such environmental conditions and, therefore, have the potential to impact our results (Shute et al. 2021).

Following each survey, we downloaded and converted all footage to .mp4 files using HandBrake Software (Version 1.5.1, Handbrake Team, GitHub), which we then uploaded into Vosaic video analysis software (Version 1.1.3686, Studiocode Business Group, Lincoln, Nebraska). We used this online application to determine and detail the extent to which bats utilized each pond as a foraging and drinking resource (see Fig. 11). First, we recorded the total time (secs) bats were present in the field-of-view during the 1-hr surveys. Note that when multiple bats were observed in the field-of-view, we recorded the amount of time each individual bat was present and then summed the total time per hour all bats were present.

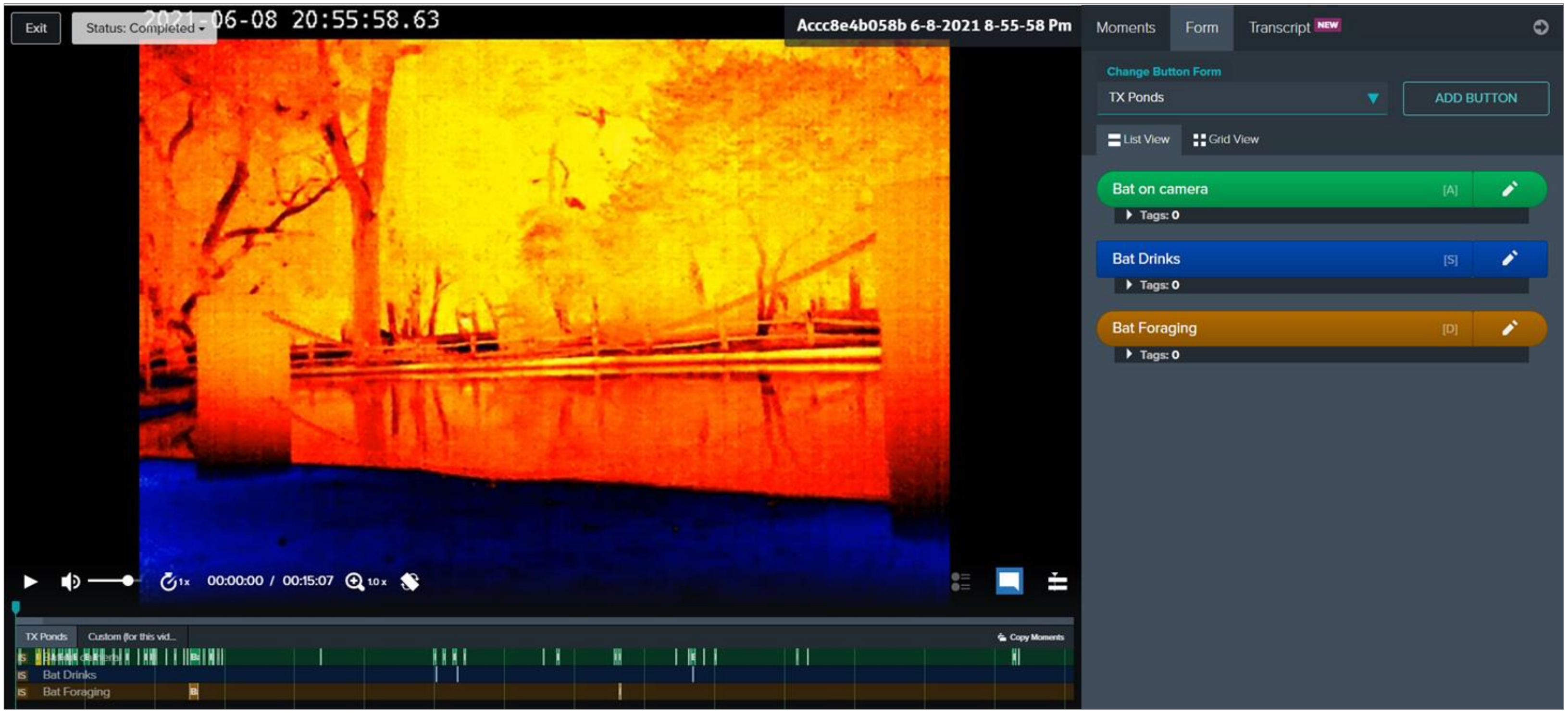


Figure 11: Screenshot of Vosiac Video Analysis Software viewing window with timeline shown beneath the footage and a custom-built activity form on the right. To aid in the analysis of the videos, we created buttons in the activity form to identify 1) bats present in the field-of-view, 2) bats drinking, and 3) bats foraging. Thus, when a bat appeared in the field-of-view, the *Bat on camera* button was toggled on and when the bat left the field-of-view, this activity was toggled off. This resulted in a *moment* being created along the timeline, which identified the start and end time of the activity selected. Once complete, video timelines were exported to Excel spreadsheets and combined to calculate the following variables: 1) total time (secs) bats were present in the field-of-view, 2) total time spent foraging, and 3) total number of drinking events recorded during each 1-hr survey.

Second, we recorded the total time (secs) bats were observed foraging in the field-of-view during the 1-hr surveys. Based on Huzzen et al. (2020), we defined foraging activity as a characteristic zig-zagging flight in which a bat made ≥3 turns in pursuit of prey. Finally, we identified the total number of drinking events to occur in the field-of-view during the 1-hr surveys. For this, we defined a drinking event as a bat swooping down to the surface of the water with its body angled head-first towards the water and making contact with the surface ≥1 times as it passed over the water (Fig. 12A; Tuttle et al. 2006, McAlexander 2013, Nystrom & Bennett 2019). This behavior often created ripples or a splash at the point of contact with the water to help with identification (Fig. 12B; Kloepper et al. 2019, Agpalo 2020).

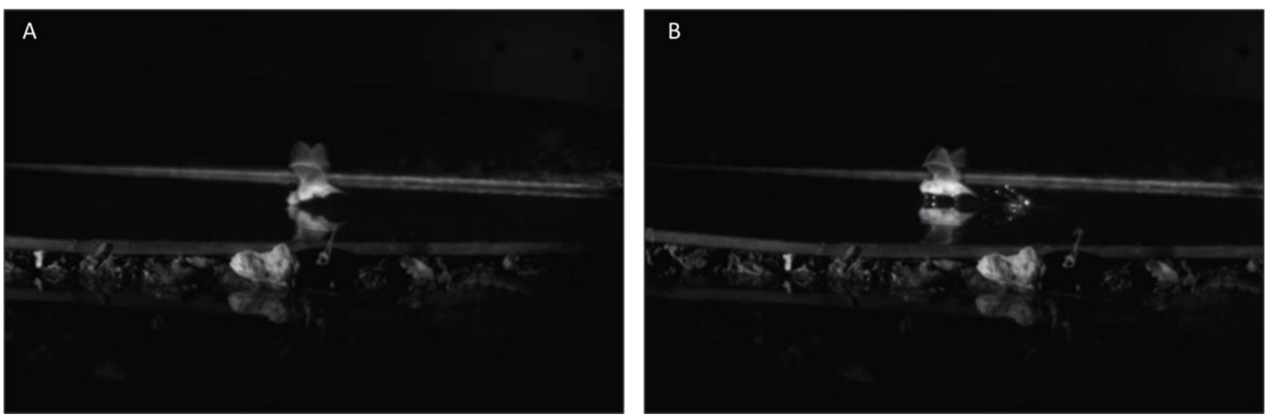


Figure 12: High speed camera image of eastern red bat (*Lasiurus borealis*) drinking on the wing. A) demonstrates posture assumed by a drinking bat and B) shows the splash created at the point of contact with the water. Images taken from (Bienz 2016).

We acknowledge that an observation-bias could have occurred when using one individual to view all the footage. Thus, to assess whether the majority of bat activity in the footage had been identified, we enlisted an additional viewer to examine and mark-up 10% of the videos. This viewer reviewed the footage blind (i.e., they were not provided with any prior information from data collected during surveys, such as the real-time observations of bat activity recorded, or the acoustic data recorded on either the Echometer or SM4BAT acoustic detectors during the surveys). We then compared mark-ups to determine if bat activity had been underestimated. This comparison revealed that both viewers recorded 464 of the 1606 observations, 573 by the primary viewer only, and 10 by the additional viewer only. Thus, we proceeded with the data analysis acknowledging that the primary viewer was effectively recording 98% of the bats in the footage.

For the sound files recorded on the acoustic detectors, we first downloaded the files into SonoBat Scrubber software (Version 4, Arcata, California) which filtered out a majority of the acoustic files containing noise; such as wind, rain, and stridulating insects. We then used SonoBat bat call analysis software (Version 3.04, Arcata, California) to identify bat echolocation calls visually and audibly among the remaining files and where possible, manually identified calls to species. Thus, for each call file, we recorded, 1) site, 2) date, 3) time, 4) number of bats, 5) species of each bat present (if possible), and 6) activity exhibited by each bat. For the latter, we defined 4 acoustically distinct activities (Nystrom & Bennett 2019, Agpalo 2020): 1) *commuting* (e.g., bats that were moving through the area) – consecutive, equally-spaced calls (i.e., individual chirps), which were constant, steadily decreasing and/or steadily increasing in call strength. In addition, any sound file with <2 calls were be categorized as commuting; 2) *searching* – consecutive, equally spaced calls that varied in strength due to the bat turning its head from side to side (Fig. 13); 3) *approach*, which demonstrated either foraging activity or that a bat was approaching the surface of the water – call intervals varied, trending toward becoming shorter, while call strength varied between constant, steadily decreasing, or steadily increasing (Fig. 13; Lewanzik et al. 2019, Stidsholt et al. 2020); and 4) *terminal buzz*, which are commonly associated with bats catching prey items, landing, and coming into drink. Thus, we used *feeding buzzes* and *drinking buzzes* to ascertain whether bats were using the ponds as a resource for either foraging and/or drinking (Hulgard & Ratcliffe 2016, Lewanzik et al. 2019). In a foraging buzz, the interval between successive calls decreased rapidly and the frequency of these calls shifted higher or lower (depending on species; Fig. 13), while in a drinking buzz, the interval between successive calls decreased rapidly and the frequency of these calls remained constant. In addition, an audible splash may be heard after the drinking buzz (species dependent; Kloepper et al. 2019, McGee 2022), and used to confirm drinking.

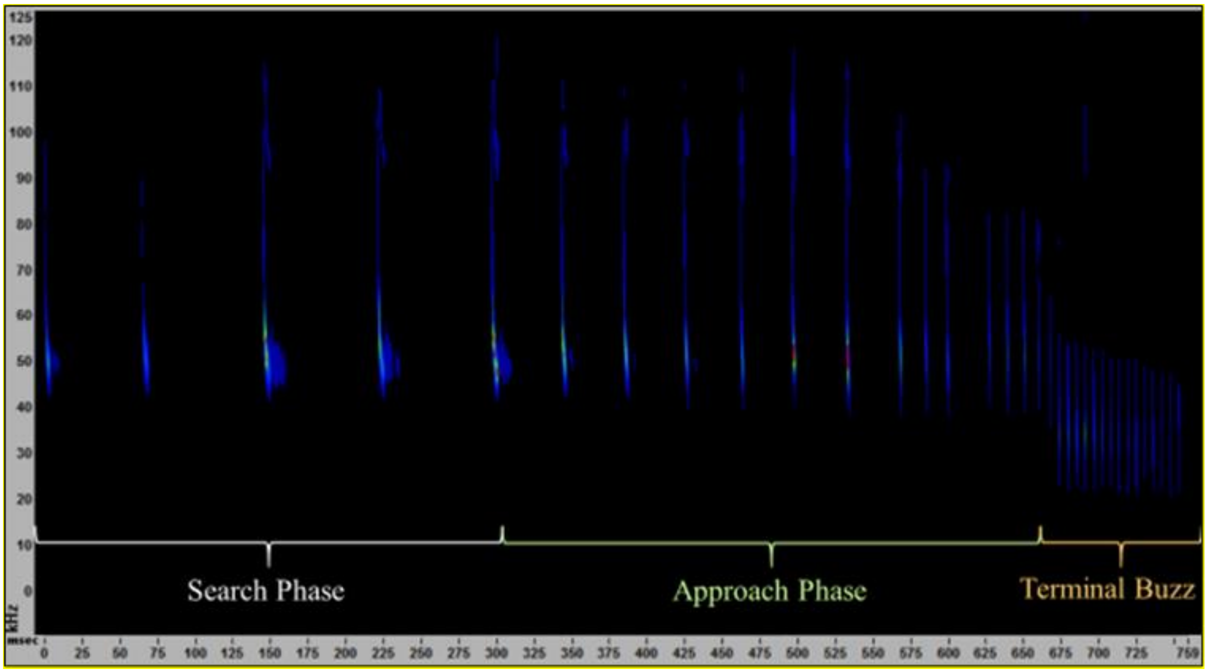


Figure 13: Spectrograph delineating the different ‘search’, ‘approach’ and ‘terminal buzz’ phases of activity observed in a bat echolocation call (taken from McGee 2022).

Once processed, we had a total of eight dependent variables from the behavioral observation and acoustic surveys. These included, observationally, 1) the average time bats were present at each site (secs), 2) the average time spent foraging per study site per night (secs), 3) average drinking rate per study site per night, and acoustically the average number of 4) bat calls, 5) resource-related calls (i.e., approach calls and terminal buzzes combined), 6) foraging buzzes and 7) drinking buzzes recorded throughout the 1-hr survey per study site per night. In addition, we also explored species-specific activity at the study sites creating a further variable related to species diversity: 8) number of species.

Analysis

To explore whether known water quality had discernible consequences for water resource use by bats, we first tested our eight dependent variables for normalcy. This preliminary analysis of the data revealed that the majority of the data were not normally distributed (Appendix B). Thus, we opted to proceed using a non-parametric Kruskal-Wallis with a post hoc Tukey test to determine if each of our dependent variables varied with ranked pollution potential. For the latter, we performed Bonferroni correction to provide more reliable and conservative estimates of significance and therefore increase confidence in our findings. We conducted all analysis in IBM SPSS Statistics (ver. 25, Armonk, NY) and where $\alpha=0.05$.

Results

We surveyed all six study sites from 23 March to 27 September 2021 and 2 March to 21 September 2022, for a total of 86 survey nights (41 in 2021 and 45 in 2022). We recorded bats to be present at all study sites, although we only observed bats on 73 survey nights. Among these surveys, we observed a total of 5,299 occurrences in which bats were in the field of view. The total time bats were present in the field-of-view per pond per night averaged 230 ±351.19 SD sec (ranging from 1 sec to 34 min 10 secs). We also observed bats foraging on 178 occasions at five of the six study sites on 40 survey nights. On these nights, the total time spent foraging per pond per night averaged 14 sec ±42.6 SD (ranging from 1 to 313 sec). Lastly, we observed 1,704 drinking events at all six study sites on 67 survey nights. For nights drinking occurred, the average drinking rate per pond per night was 21 ±25.2 SD (ranging from 1 to 107).

From the acoustic surveys, we recorded 25,443 bat passes in proximity to the study sites on 85 survey nights. We recorded an average of 118.1 ±197.6 SD bat passes per study sites per night (ranging from 0 to 1269) and identified all six local bat species, although not all species were recorded at every pond surveyed. Among these bat passes, 11,239 were identified as commuting and searching for an average of 130.69 ±132.58 SD passes per study sites per night (ranging from 0 to 651). We identified a total of 7,554 approach phase calls. This acoustic activity was recorded at all six study sites with an average of 87.84 ±89.79 SD approach phase calls per pond per night (ranging from 0 to 359). Furthermore, we identified a total of 6550 terminal buzzes across all six study sites, for an average of 76.26 ±108.54 SD buzzes per pond per night (ranging from 0 to 441). Among these terminal buzzes, 3703 were identified as feeding buzzes and 2527 drinking buzzes, which average of 43.07 ±78.86 SD feeding buzzes per pond per night (ranging from 0 to 341) and 29.44 ±39.26 SD drinking buzzes per pond per night (ranging from 0 to 167).

From the calls manually identified to species, we recorded six local bat species emitting resource-related calls, feeding buzzes, and drinking buzzes in proximity to the study sites. The only species not recorded was the canyon bat. The average time spent foraging per pond per night (secs) and the average number of drinking events recorded for each species is provided in Table 3. As tri-colored, Mexican free-tailed, and hoary bats had <100 resource-related bat passes recorded we did not consider them in the species-specific analysis below.

Table 3: Average number of bat passes identified to species with approach phase calls and terminal, feeding and drinking buzzes recorded in 1-hr acoustic monitoring surveys conducted at the six study sites. Includes ±SD and the maximum and minimum number of bat passes, along with the total number of bat passes recorded with resource-related calls in them (i.e., approach phase and terminal buzzes).

Species	Approach phase	Terminal Buzz	Feeding Buzz	Drinking buzz	Total # bat passes with resource use
Eastern red (*Lasiurus borealis*)	18.78 ±20.8 (0 to 115)	8.72±10.02 (0 to 53)	2.67 ±5.22 (0 to 39)	4.64±6.76 (0 to 29)	2338
Hoary (*Lasiurus cinereus*)	0.21 ±0.65 (0 to 4)	0.09±0.36 (0 to 2)	0.03±0.18 (0 to 1)	0.02±0.15 (0 to 1)	80
Silver-haired (*Lasionycteris noctivagans*)	4.94 ±6.0 (0 to 31)	2.09±3.87 (0 to 23)	1.29±2.92 (0 to 20)	0.36±1.09 (0 to 8)	2330
Evening (*Nycticeius humeralis*)	61.23 ±75.1 (0 to 336)	65.63 ±103.55 (0 to 422)	38.48±75.7 (0 to 326)	24.0±36.11 (0 to 163)	9040
Tri-colored (*Perimyotis subflavus*)	1.44 ±2.76 (0 to 15)	0.87 ±1.91 (0 to 9)	0.36±1.23 (0 to 9)	0.34±1.09 (0 to 8)	164
Mexican free-tailed (*Tadarida brasiliensis*)	1.55 ±5.19 (0 to 34)	0.27 ±1.68 (0 to 15)	0.16±1.30 (0 to 12)	0.02±0.22 (0 to 2)	152

For bat presence, we observed the average time (secs) bats spent at study sites during the 1-hr surveys varied between sites (Fig. 14) and determined there to be a significant difference between sites (N=168, K=102.86, df=5, p<0.001). This difference was driven by Sites 1and 4, which were significantly higher than the other four study sites, with Site 1 having significantly higher bat presence than Site 4 (see Appendix C Table C1).

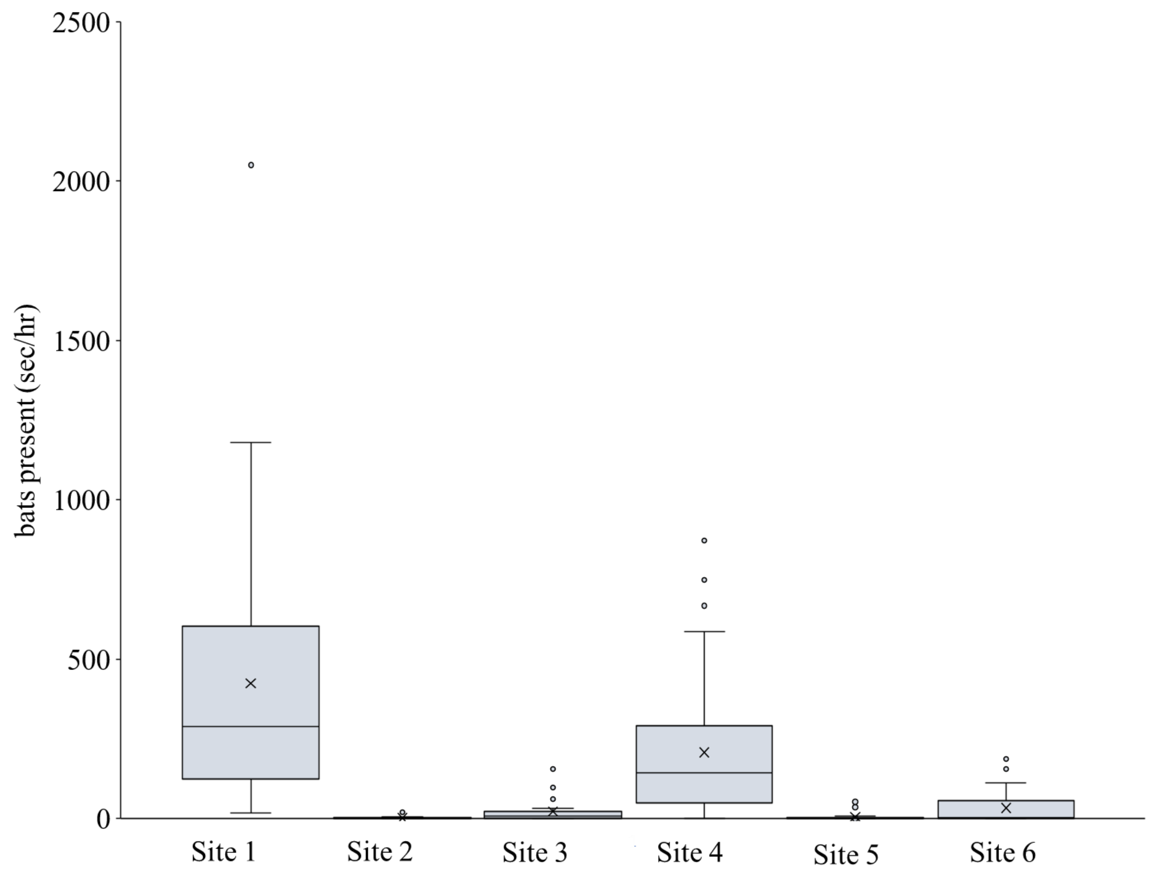


Figure 14: Average time bats were observed in the field-of-view in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For foraging, we observed that across the study sites the average time (secs) bats foraged at a pond during the 1-hr surveys varied between sites (Fig. 15) and determined this difference to be significant (N=168, K=46.67, df=5, p<0.001). This difference was driven by Sites 1 and 4, where Site 1 had significantly higher foraging activity than Sites 2, 3, 5 and 6, while Site 4 had significantly higher foraging activity than Sites 2 and 5 (Appendix C Table C2).

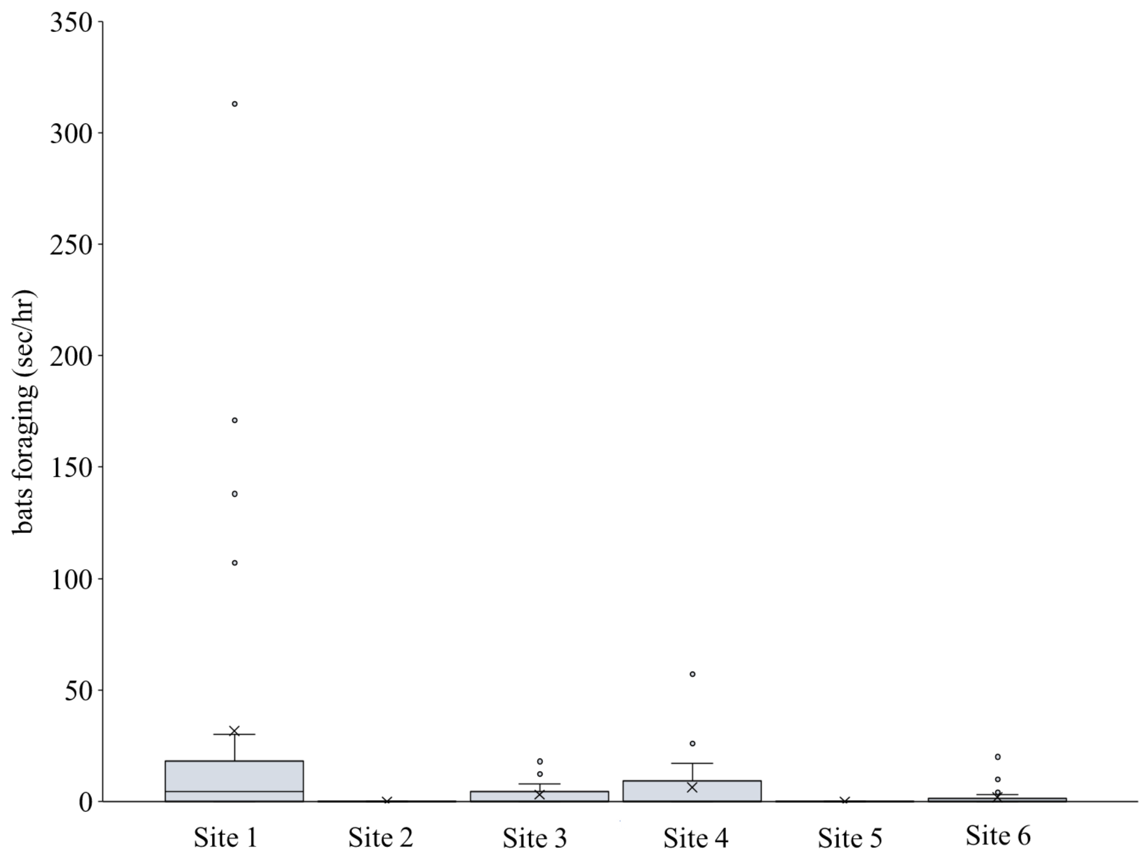


Figure 15: Average time bats were observed foraging in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For drinking events, we found that the average number of drinking events observed during the 1-hr surveys varied between sites (Fig. 16) and determined that drinking activity was significantly different (N=168, K=112.69, df=5, p<0.001). This difference was driven by Sites 1 and 4, which both had significantly higher drinking activity than the other four study sites. Moreover, we confirmed that drinking activity at Sites 1 and 4 was similar (Appendix C Table C3).

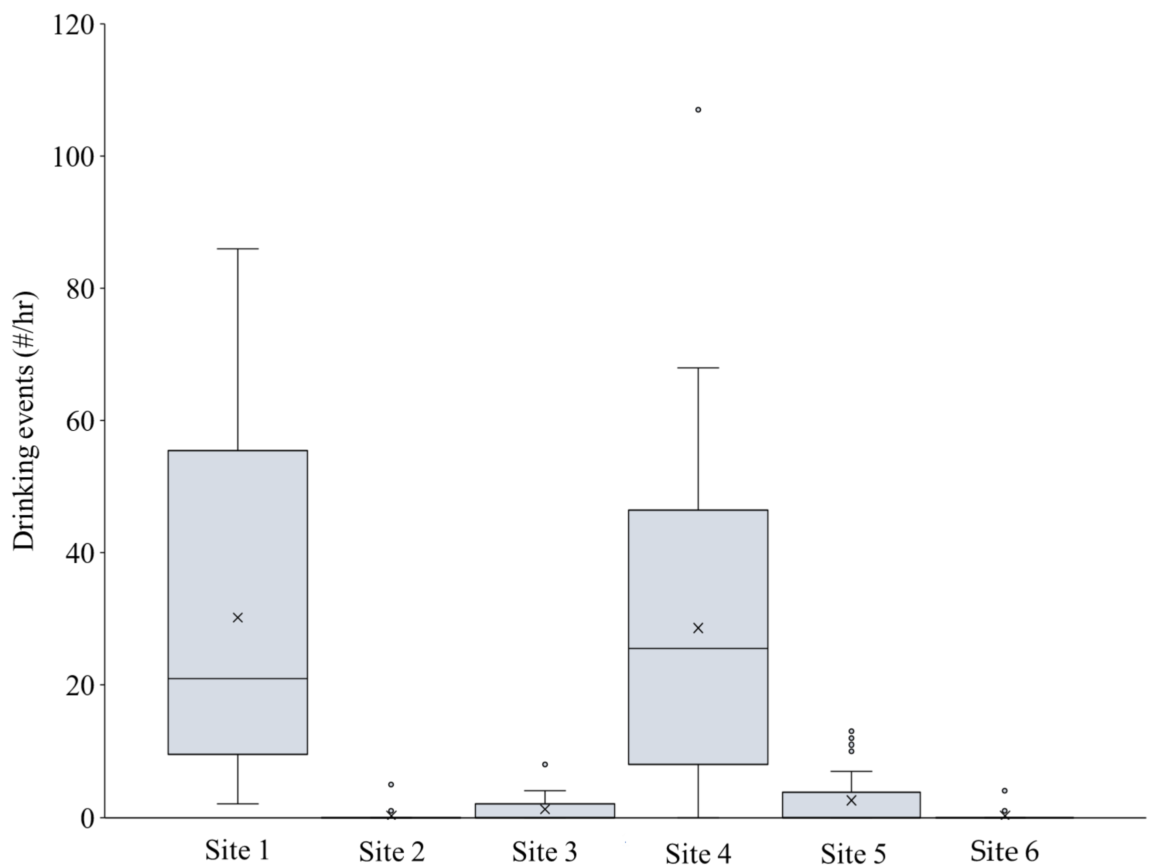


Figure 16: Average time bats were observed drinking in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For acoustic activity, we found that the average number of bat calls recorded during the 1-hr surveys varied between sites (Fig. 17) and confirmed that acoustic activity was significantly different (N=166, K=91.66, df=5, p<0.001). This difference was driven by Sites 1 and 4, in which Site 1 had significantly more acoustic activity than all the other five sites, and Site 4 had significantly more acoustic activity than Sites 2 and 5 (Appendix C Table C4).

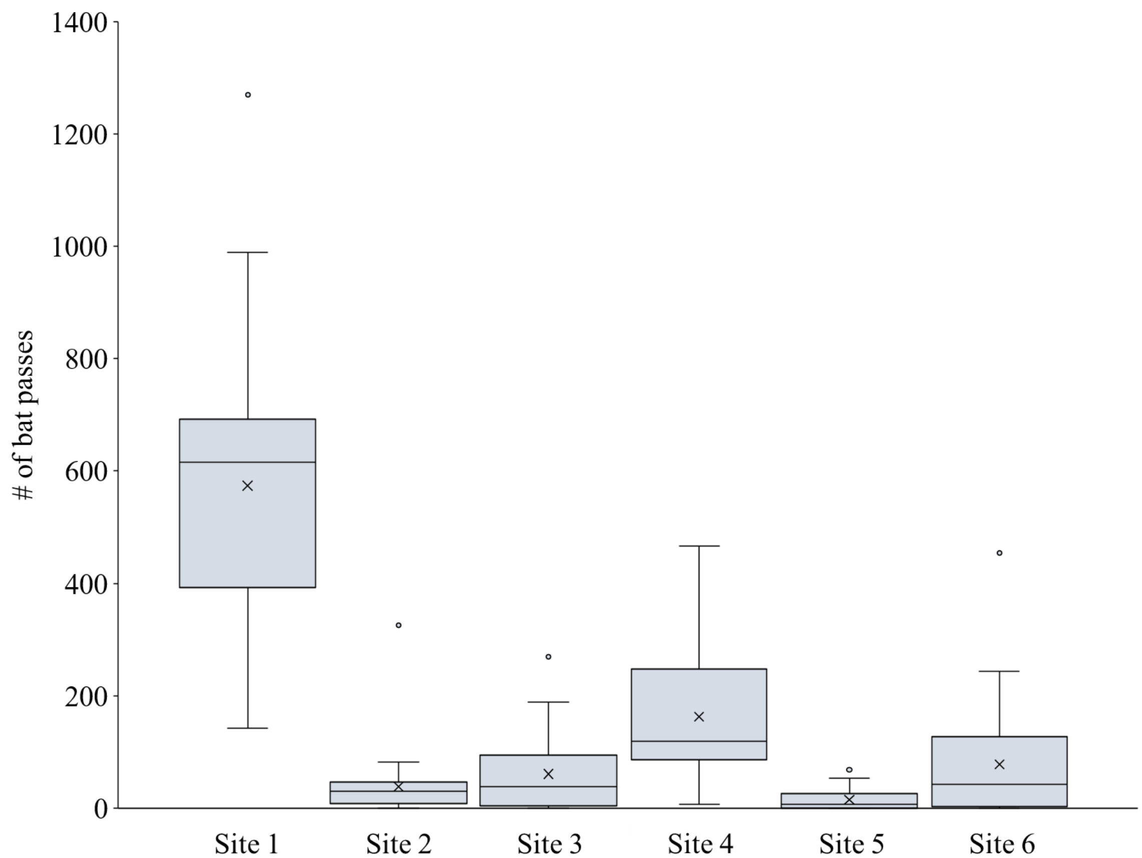


Figure 17: Average number of bat passes recorded in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For resource-related acoustic activity, we observed that the average number of resource-related calls recorded during the 1-hr surveys varied between sites (Fig. 18) and determined that this activity was significantly different (N=166, K=101.39, df=5, p<0.001). This difference was driven by Sites 1 and 4, in which Site 1 had significantly more resource-related acoustic activity than all the other five sites, and Site 4 had significantly more resource-related activity than Sites 2 and 5 (Appendix C Table C5).

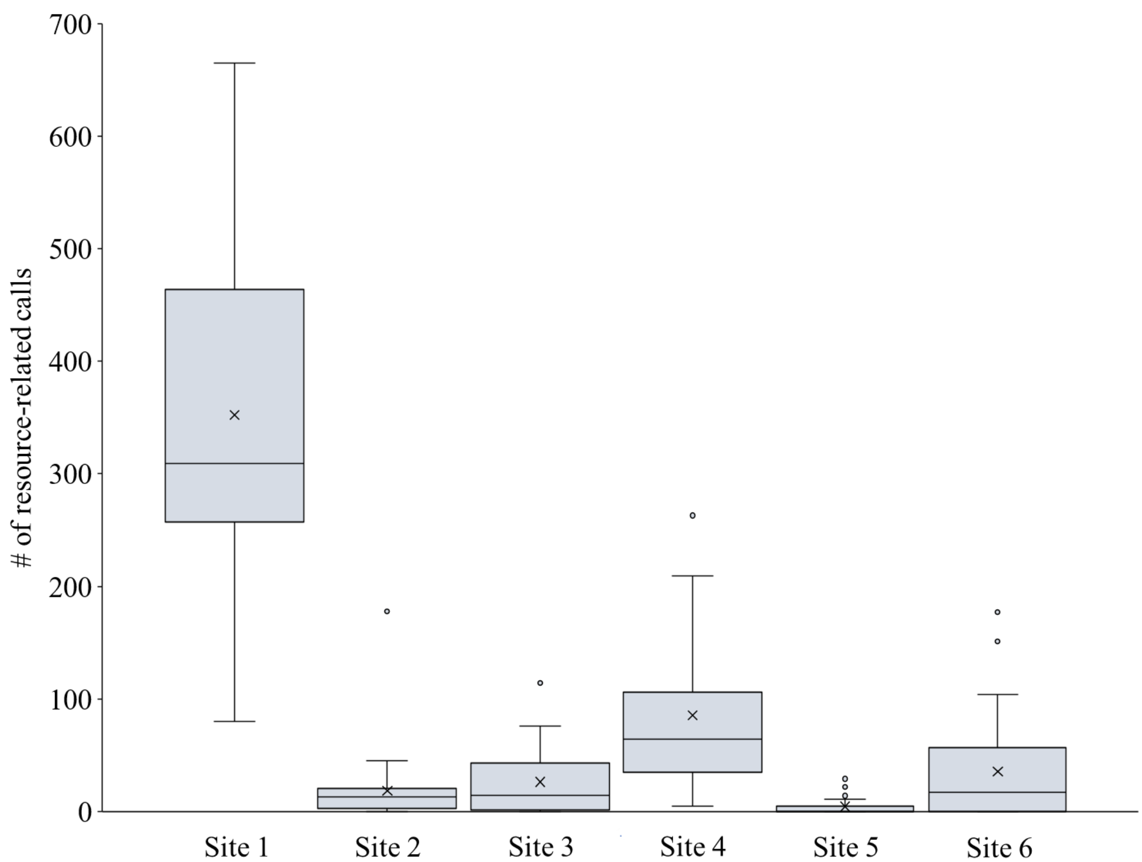


Figure 18: Average number of resource-related calls in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For acoustic foraging activity, we found that the average number of feeding buzzes recorded during the 1-hr surveys varied between sites (Fig. 19) and determined this variation to be significantly different (N=166, K=78.83, df=5, p<0.001). This difference was driven by Sites 1 and 4, in which Site 1 had significantly more feeding buzzes than all the other five sites, and Site 4 had significantly more feeding buzzes than Sites 2 and 5 (Appendix C Table C6).

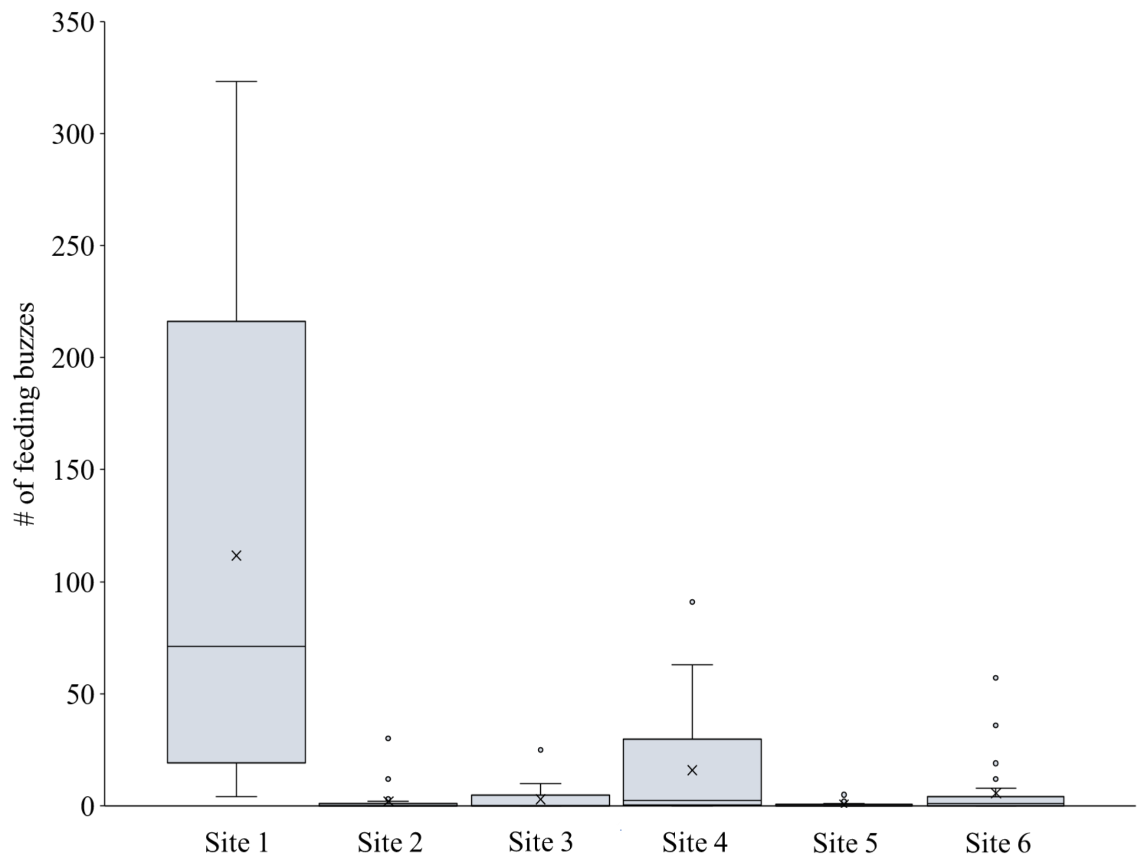


Figure 19: Average number of feeding buzzes recorded in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For acoustic drinking activity, we found that the average number of drinking buzzes recorded during the 1-hr surveys varied between sites (Fig. 20) and determined this variation to be significantly different (N=166, K=115.40, df=5, p<0.001). This difference was driven by Sites 1 and 4, in which Sites 1 and 4 had significantly higher acoustic drinking activity than all the other four sites, but there was no significant difference in drinking between Sites 1 and 4 (Appendix C Table C7).

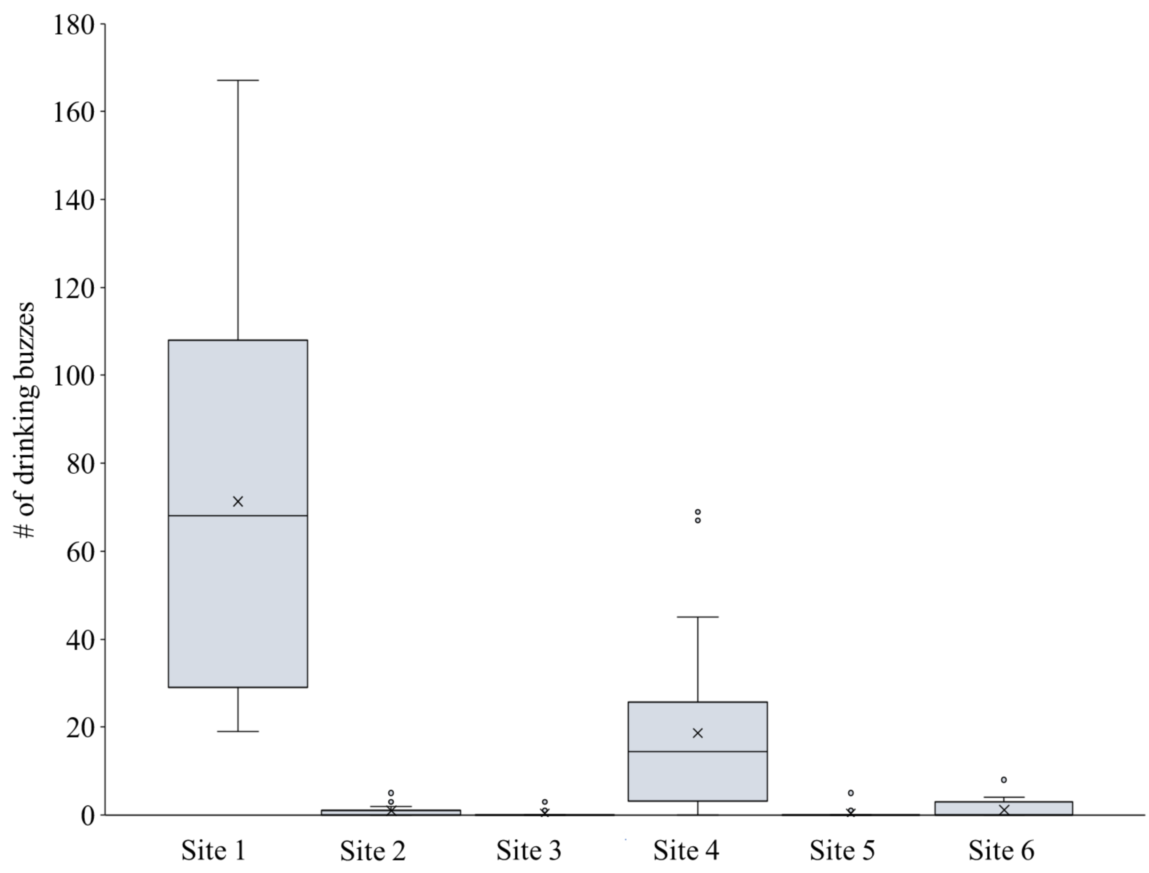


Figure 20: Average number of drinking buzzes recorded in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For diversity at the study sites, we found that the number of species recorded during the 1-hr surveys varied between sites (Fig. 21) and confirmed that this variation was significantly different (N=166, *K*=41.69, *df*=5, *p*<0.001). This difference was driven by Sites 1 and 3, in which Site 1 had significantly more species than the other five sites, while Site 3 had significantly more species than Site 5 (Appendix C Table C8).

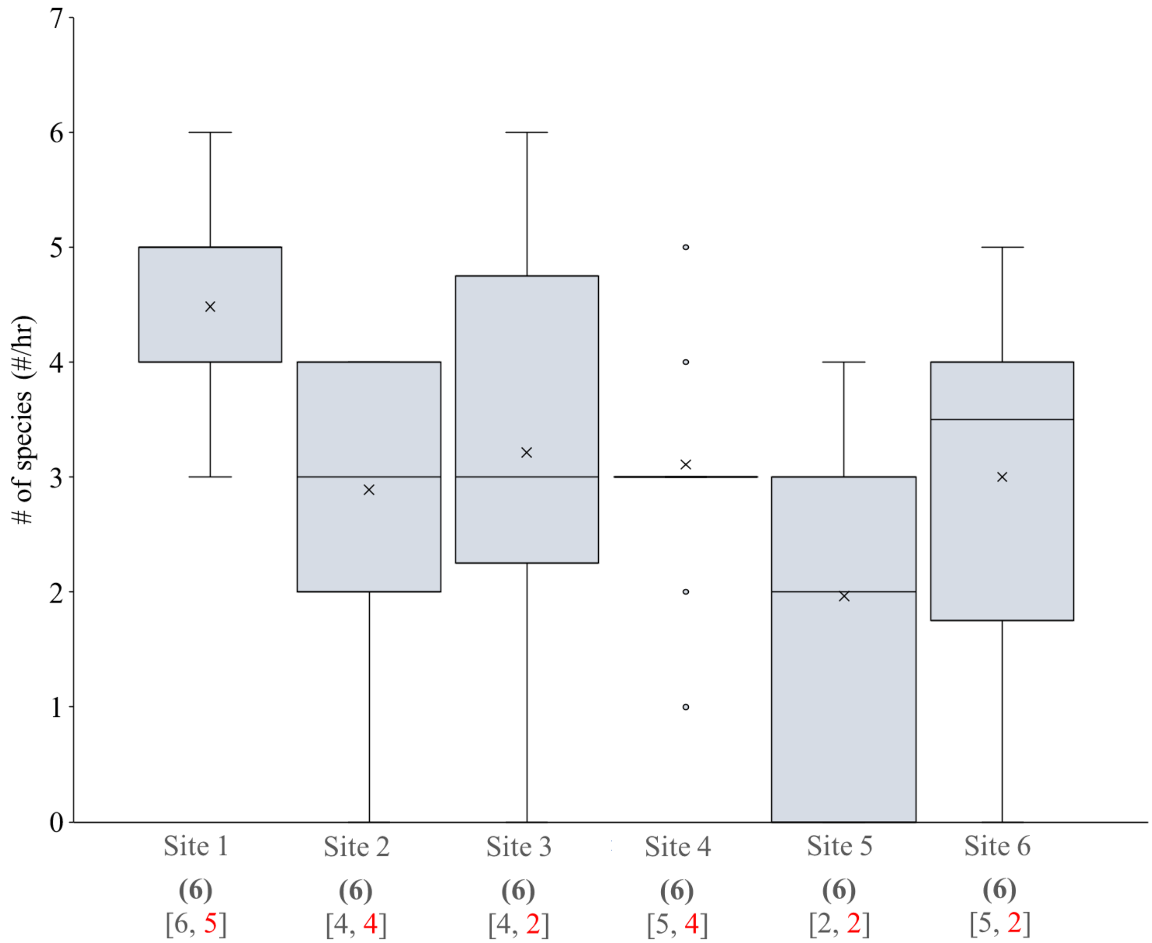


Figure 21: Number of species recorded in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The total number of species recorded at each site are given in bold parentheses and the number of species recorded emitting feeding and drinking buzzes (red), respectively, are given in brackets. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For species-specific resource-related acoustic activity, we observed that the average number of resource-related calls recorded for evening bats during the 1-hr surveys varied between sites (Fig. 22) and determined this variation was significantly different (N=166, K=97.49, df=5, p<0.001). This difference was driven by Sites 1, 4 and 6, where Sites 1 and 4 had significantly higher resource-related activity than the other four sites and Site 6 had significantly higher activity than Site 5 (Appendix C Table C9).

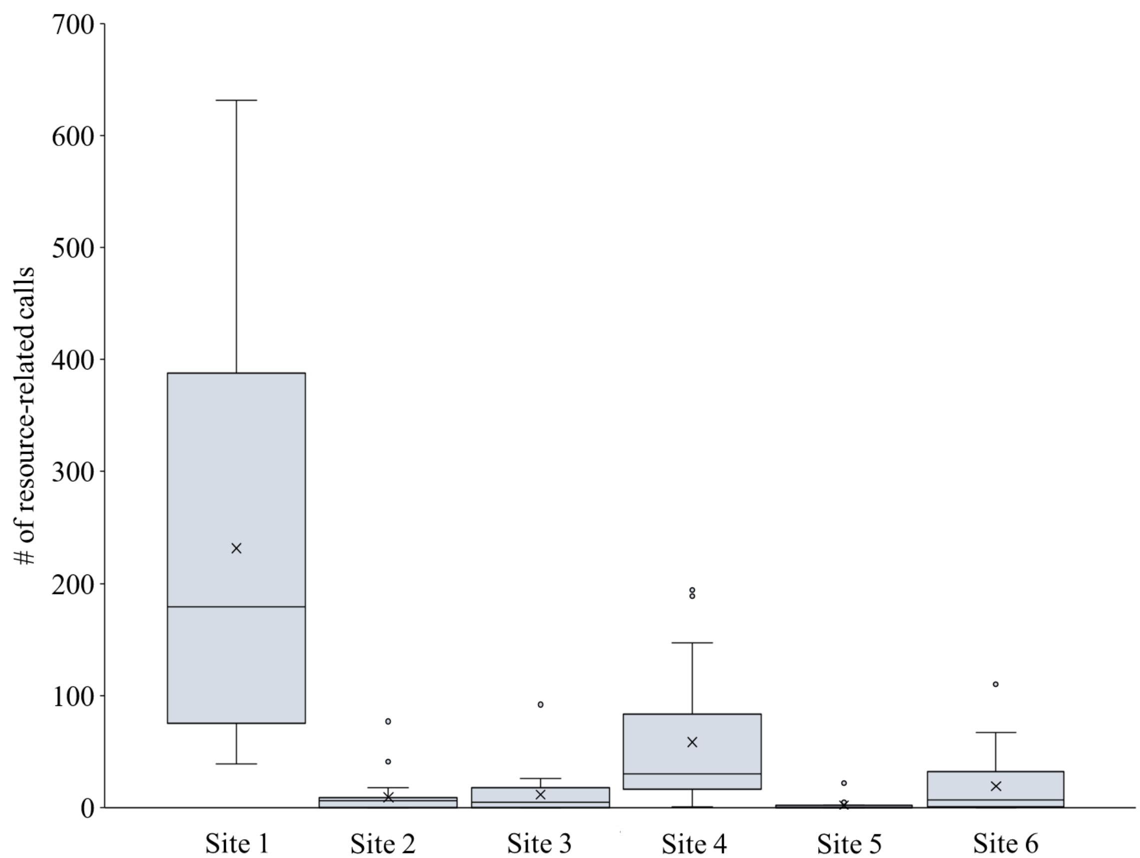


Figure 22: Average number of resource-related calls for evening bats in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For eastern red bats, the average number of resource-related calls recorded during the 1-hr surveys varied between sites (Fig. 23) and determined this variation to be significantly different (N=166, K=53.00, df=5, p<0.001). This difference was driven by Sites 1 and 4, where Site 1 had significantly higher eastern red bat activity compared to Sites 2, 3, 5 and 6, and Site 4 only had significantly higher activity than Site 5 (Appendix C Table C10).

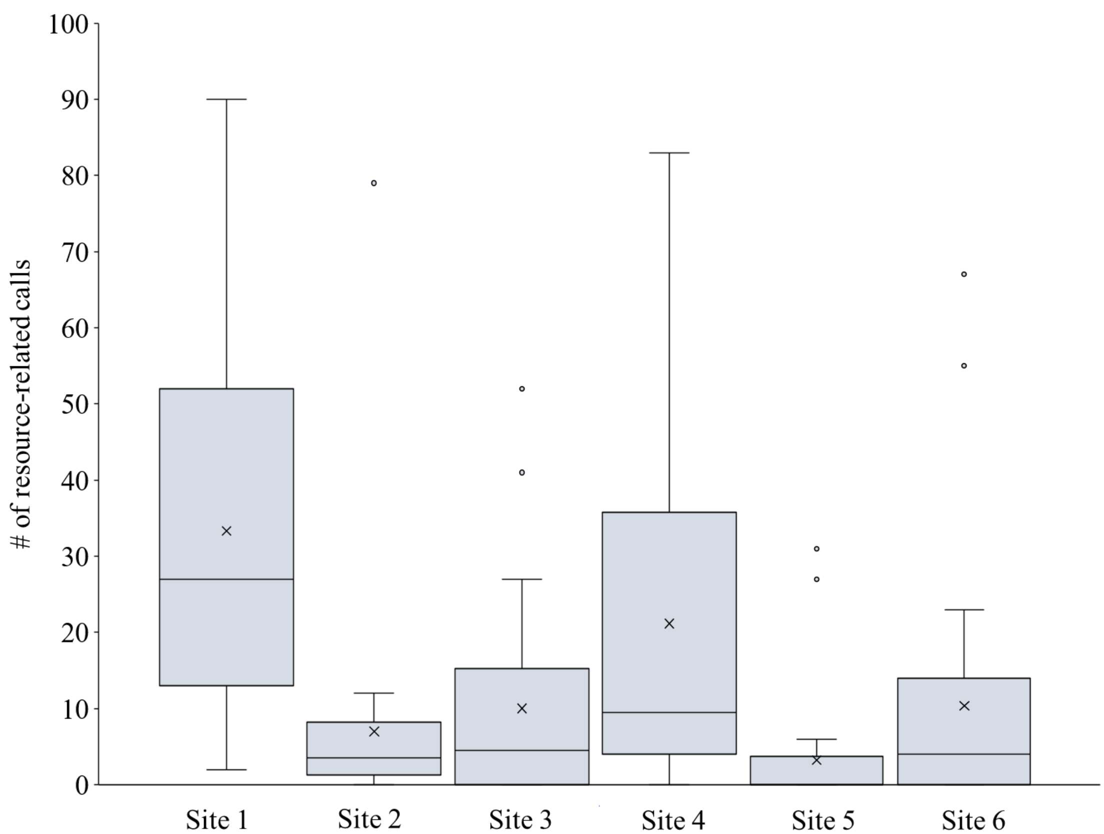


Figure 23: Average number of resource-related calls for eastern red bats in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For silver haired bats, the average number of resource-related calls recorded during the 1-hr surveys varied between sites (Fig. 24) and determined this variation to be significantly different (N=166, K=65.98, df=5, p<0.001). This difference was driven by Site 1, which had significantly higher silver-haired bat activity than the other five study sites (Appendix C Table C11).

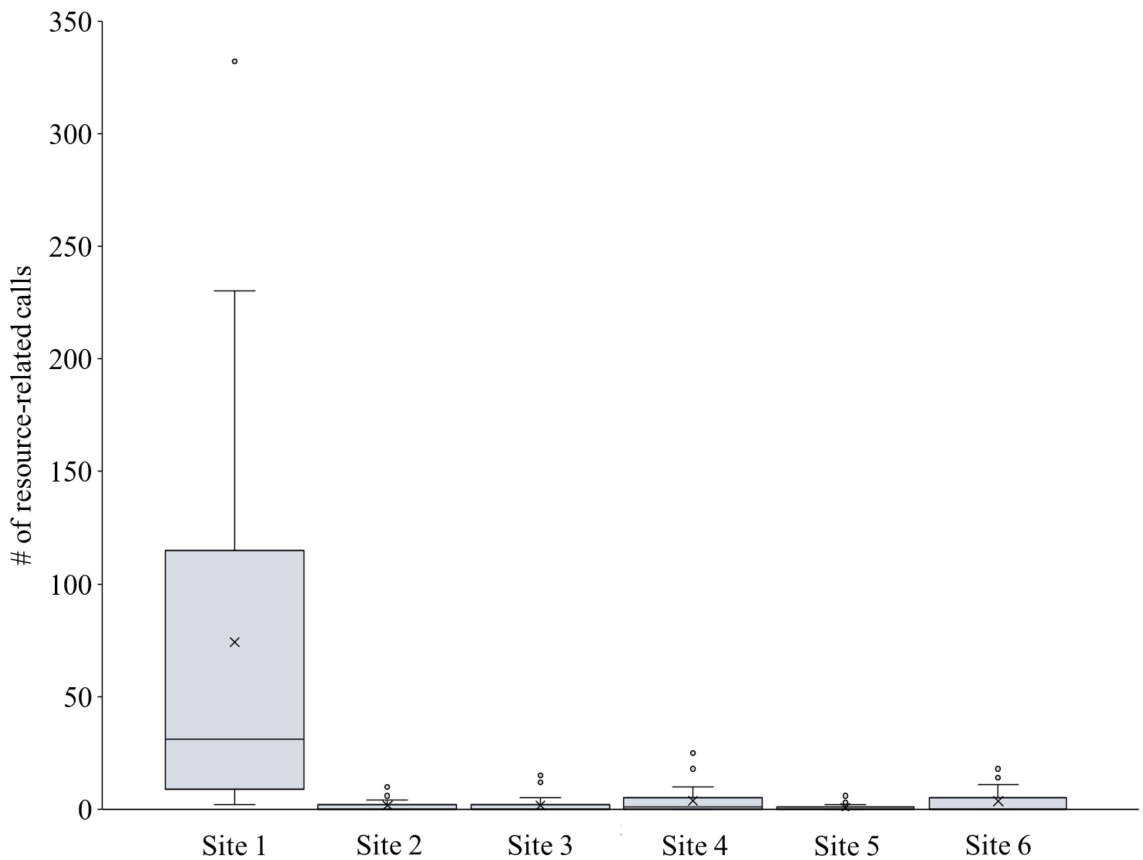


Figure 24: Average number of resource-related calls for silver-haired bats in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For species-specific acoustic foraging activity, we observed that the average number of evening bat feeding buzzes recorded during the 1-hr surveys varied between sites (Fig. 25) and determined this variation was significantly different (N=166, K=78.76, df=5, p<0.001). This difference was driven by Sites 1 and 4, where Site 1 had significantly higher evening bat feeding buzzes compared to all the other sites, and Site 4 only had significantly higher activity than Site 5 (Appendix C Table C12).

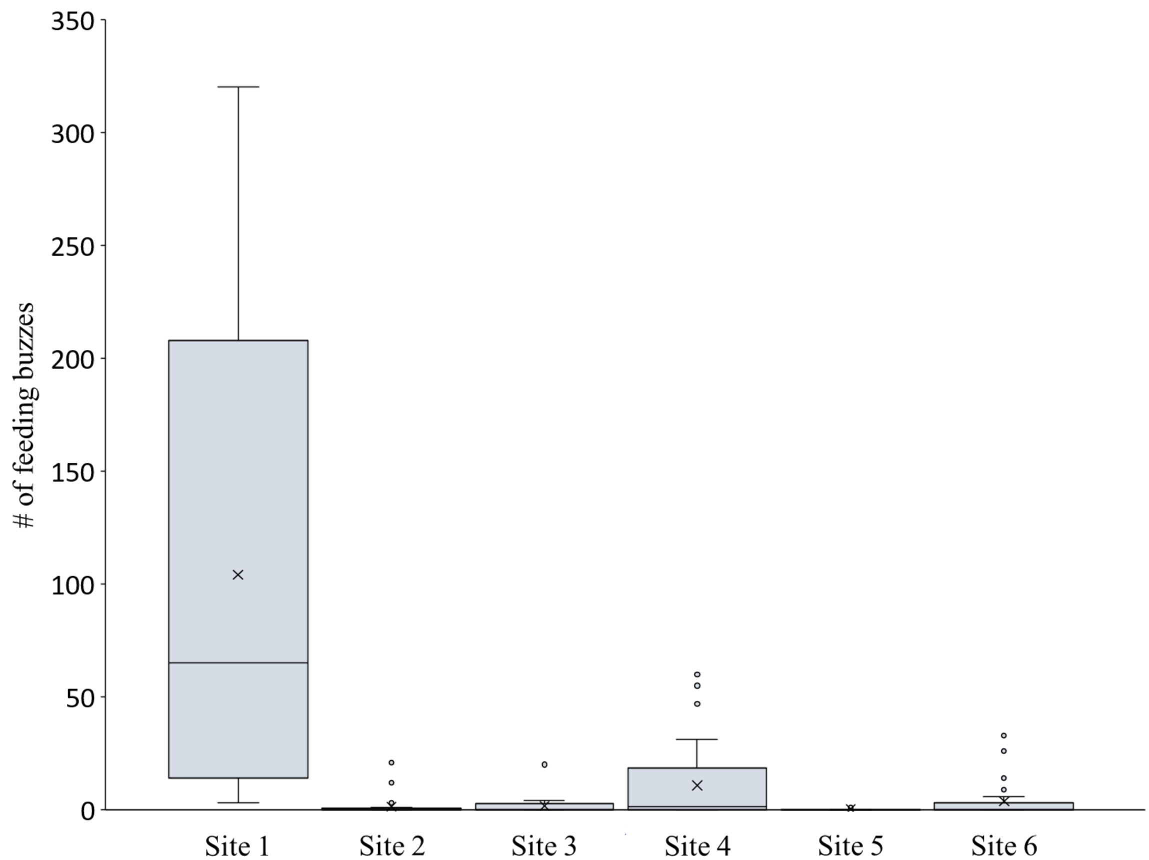


Figure 25: Average number of feeding buzzes for evening bats in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For eastern red bats, the average number of feeding buzzes varied between sites (Fig. 26) and determined this variation was significantly different (N=166, K=50.51, df=5, p<0.001). This difference was driven by Site 1, which had significantly higher eastern red feeding buzzes than the other five study sites (Appendix C Table C13).

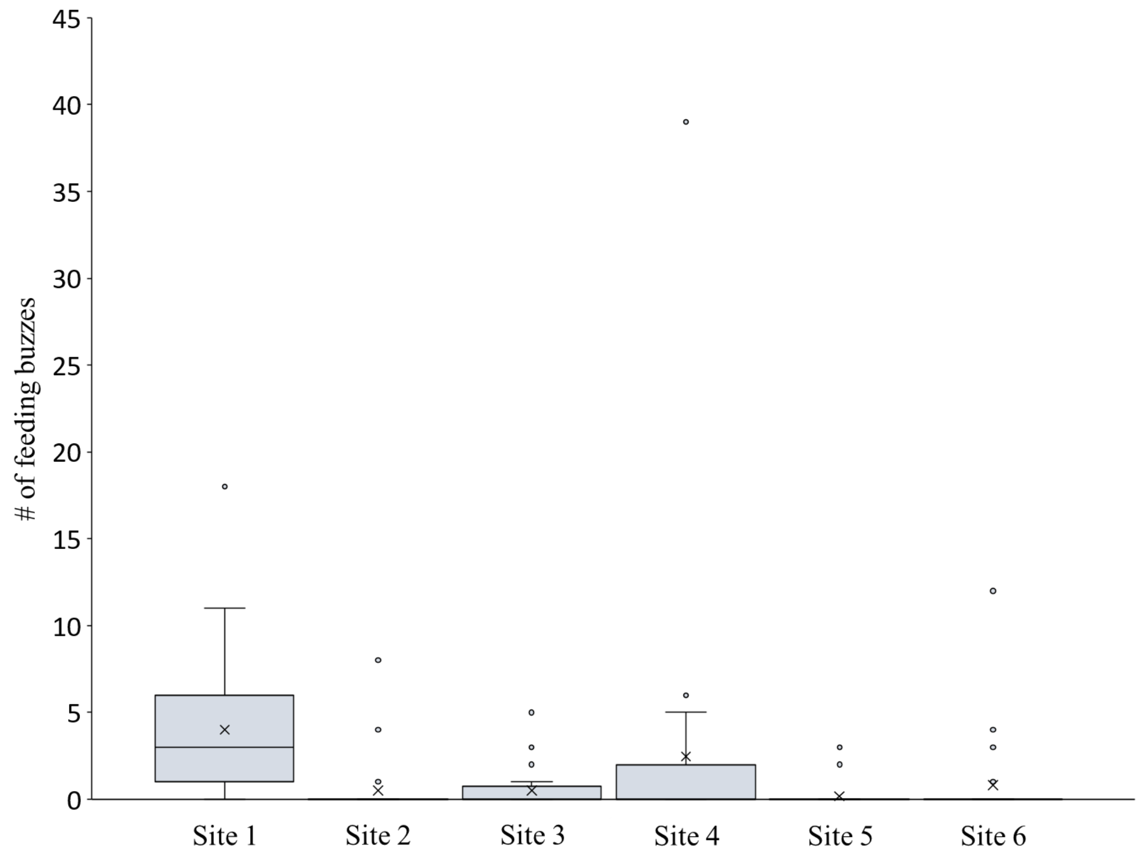


Figure 26: Average number of feeding buzzes for eastern red bats in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For silver haired bats, the average number of feeding buzzes varied between sites (Fig. 27) and determined this variation to significantly different (N=166, K=50.24, df=5, p<0.001). This difference was driven by Site 1, which had significantly higher silver-haired feeding buzzes than the other five study sites (Appendix C Table C14).

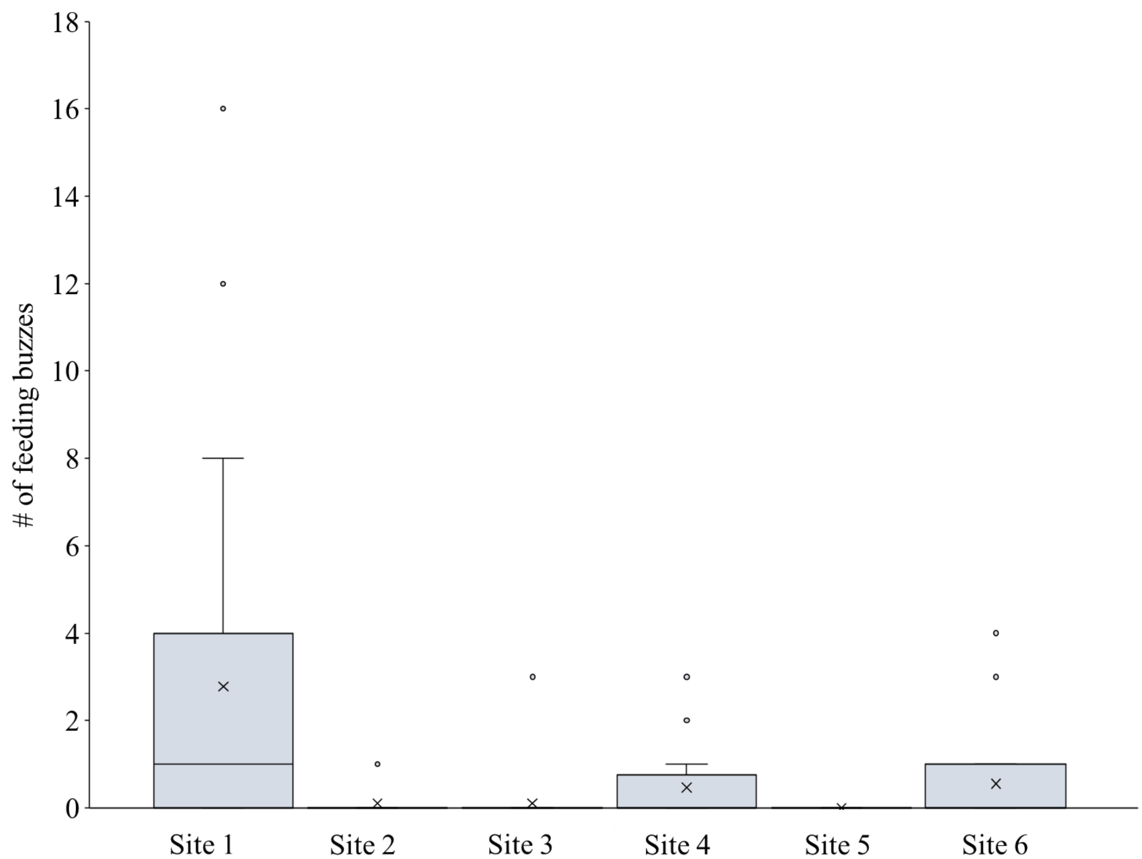


Figure 27: Average number of feeding buzzes for silver-haired bats in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For species-specific acoustic drinking activity, we observed that the average number of evening bat drinking buzzes recorded during the 1-hr surveys varied between sites (Fig. 28) and determined this variation was significantly different (N=166, K=124.03, df=5, p<0.001). This difference was driven by Sites 1 and 4, which both had significantly higher levels of evening bat drinking compared to the other four sites (Appendix C Table C15).

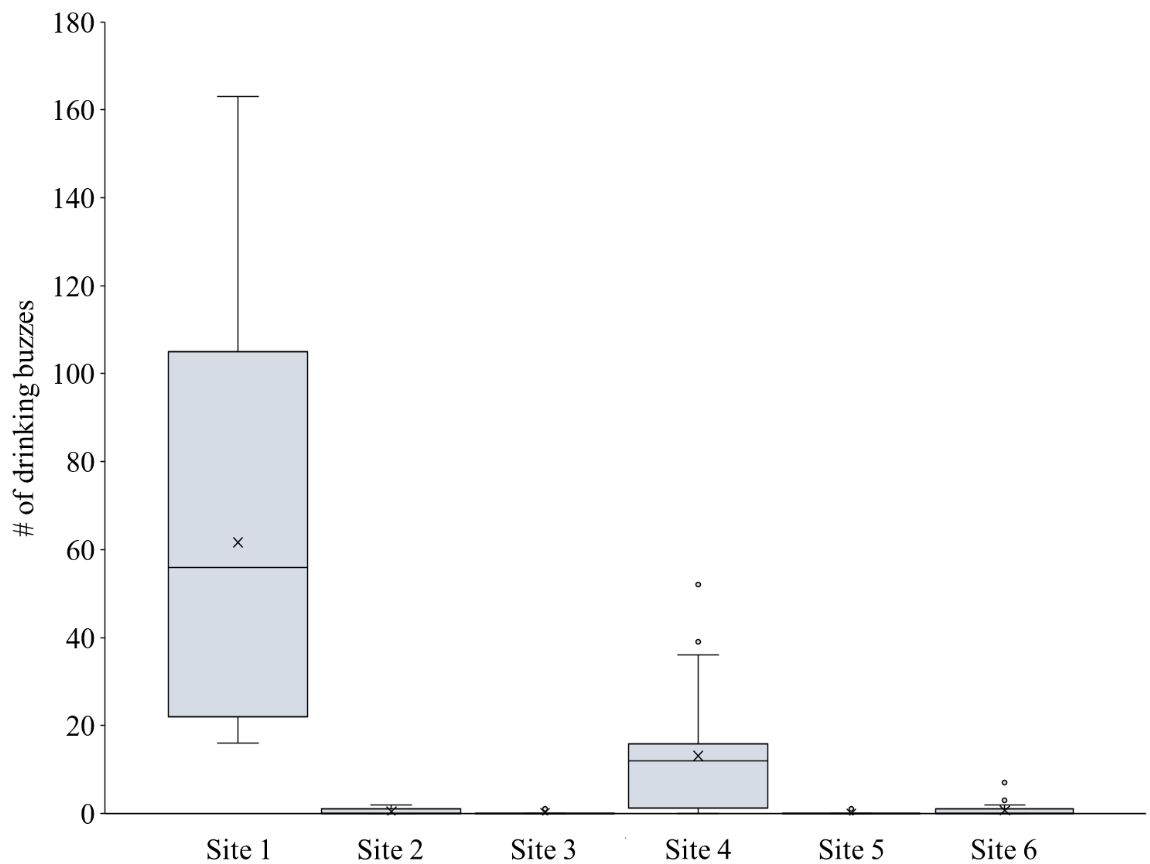


Figure 28: Average number of drinking buzzes for evening bats in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For eastern red bats, the average number of drinking buzzes varied between sites (Fig. 29) and determined this variation to be significantly different (N=166, K=75.93, df=5, p<0.001). This difference was driven by Sites 1 and 4, which both had significantly higher levels of eastern red bat drinking activity compared to the other four sites (Appendix C Table C16).

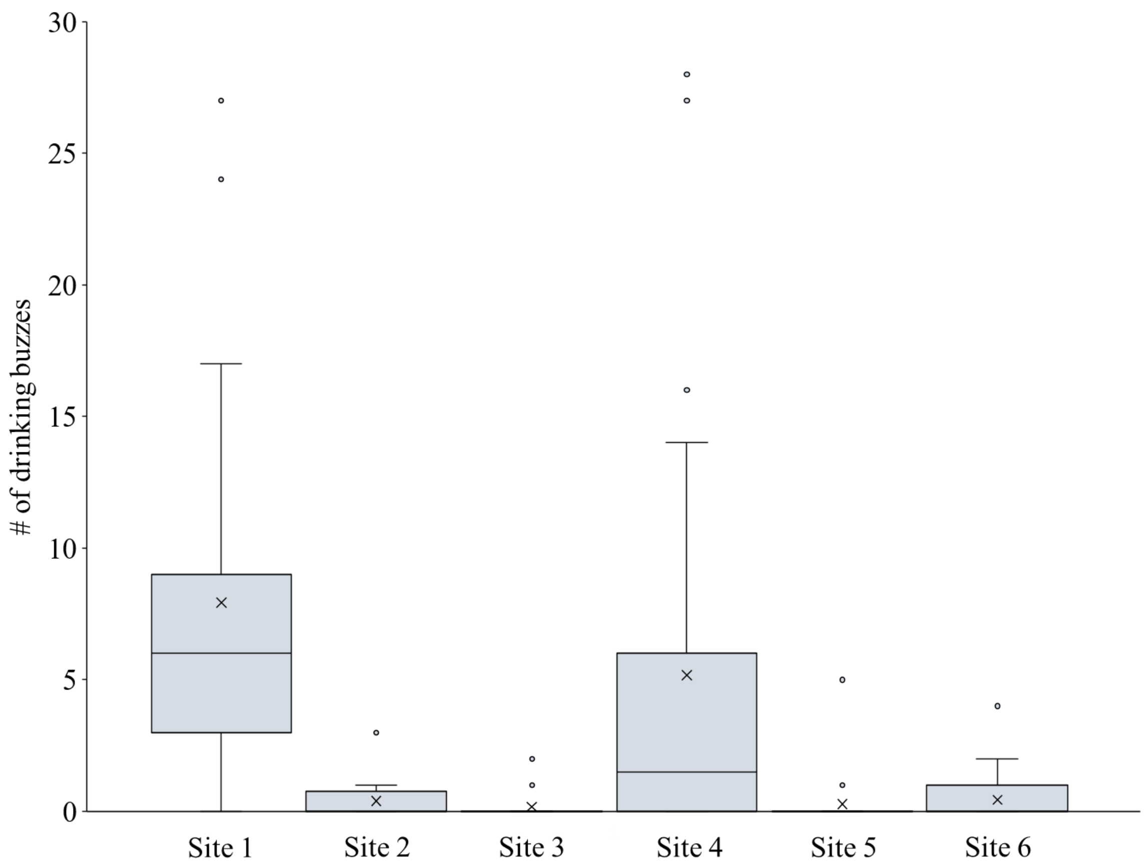


Figure 29: Average number of drinking buzzes for eastern red bats in 1-hr surveys at six study sites in Fort Worth, Texas, USA. The sites are listed by ranked pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond and Lake Como.

For silver haired bats, the only sites where their drinking buzzes were recorded were Sites 1 (n = 22), 2 (n = 3) and 4 (n = 6).

Discussion

Our study revealed that pollution potential and, therefore, water quality influenced the direct use of ponds by bats. As hypothesized, the pond with the highest water quality had a greatest abundance and diversity of bats. Site 1, the Trinity Duck Pond which represented a closed system that was annually emptied and refilled with clean water from the city's water supply represented 60% of all the bat activity observed, 73% of the observed foraging activity, 48% of the observed drinking activity, 67% of the feeding buzzes recorded, 80% of the drinking buzzes recorded, and had the highest number of species foraging (6 of 6) and drinking (5 of 6). However, bat abundance and diversity among our remaining five sites, did not appear to steadily decrease with ranked pollution potential as anticipated. Site 4, the retention pond in Foster Park, represented 30% of all bat activity observed with a pollution potential of 21%, while Sites 2 (a river tributary entering Benbrook Lake) and 3 (a river tributary just after Benbrook Lake) represented 1% and 3% respectively of all bat activity observed. However, the diversity of species that were recorded drinking at Site 2 (4 of 6) was equivalent to Site 4 (4 of 6) compared to Site 3 which only had two species recorded. Moreover, Site 2 was the only site to have the hoary bat drink. This result could be an indicator of water quality, so while activity was much higher at Site 4, the diversity of species drinking at Site 2 was equivalent, albeit infrequent, suggesting that while this site represented a drinking resource for bats, other factors may have been influencing the suitability of the site for bats. We can only speculate that bat activity at Site 2 was lower than we expected, because the water at this survey site was faster flowing than any of our other sites. We know from the literature that the species known to be in the area prefer lentic water sources (Zahn & Maier 1997, Warren et al. 2000) and this may have deterred bats from frequently using Site 2 as a water resource. Moreover, Site 3 was also a flowing water source, but the survey site was located at a section of the Trinity River that pooled, slowing down the flow of water at this site. This difference in flow may explain why more bats were observed at this site compared to Site 2, despite Site 2 having lower pollution potential (i.e., 1% at Site 2 compared to 13% at Site 3). We propose that potentially selecting a survey site along the Benbrook Lake bank instead of a faster flowing tributary would have yielded different results.

Comparing pollution potential to abundance and diversity between Sites 4, 5 and 6, Site 4 not only had higher abundance of observed bat activity, it represented 14% of the observed foraging activity, 45% of the observed drinking activity, 11% of the feeding buzzes recorded, 20% of the drinking buzzes recorded, and had the second highest number of species foraging (5 of 6) and drinking (4 of 6). In contrast, Site 5 with a pollution potential of 26% had <1% of observed bat activity, no observed foraging activity, <1% of the observed drinking activity, <1% of the feeding buzzes recorded, <1% of the drinking buzzes recorded, and had the lowest number of species foraging (2 of 6) and drinking (2 of 6), while with a pollution potential of 38% had 4% of observed bat activity, 4% of observed foraging activity, 4% of the observed drinking activity, 4% of the feeding buzzes recorded, 1% of the drinking buzzes recorded, and had four species foraging and only 2 species recorded drinking. These results indicate that both Sites 5 and 6 are likely to have water quality levels that are unsuitable for bats, in comparison to Site 4. Of all our sites, Site 6 is known have MCL above water quality standards prescribed by the (EPA 2022f) and that it is registered as contaminated above appropriate levels for human consumption. Assuming that bat resource use is an indicator of water quality, specifically as a drinking resource, our findings suggest that Site 5 could also be contaminated above appropriate levels for human consumption. More specifically, these results suggest that while Site 4’s drainage system discharges directly into a portion of the Trinity River registered as impaired, this site is unlikely to be a major point source of contamination.

Our results confirm that while water may be present in the form of retention ponds, drainage ditches, and even intercity rivers in urban areas, these water sources are not necessarily available to bats due to their water quality. Interestingly, the decrease in bat activity we observed differs from previous studies where water quality was found not to influence water resource use (Kalcounis-Rueppell et al. 2007, Razgour et al. 2010, Laverty & Berger 2020). For example, Scott et al. (2010) compared acoustic calls from *Pipistrellus pipistrellus* and *P. pygmaeus* at two riparian habitats that differed in water quality and found no discernable difference in activity recorded. However, none of these studies explored drinking and foraging activity explicitly. It is likely that the sites selected in these studies represented other resources, such as a movement corridor. Riparian habitats, for instance, are known to be important for bats as commuting routes (Furmankiewicz & Kucharska 2009, Cortes & Gillam 2020, Barre et al. 2021). Thus, by not determining specific resource use, it is possible that the abundance of bats at different sites appear to be similar. In contrast, a number of studies support our findings that water quality influences species diversity, not just for bats but other taxa (Burgmeier et al. 2011, Straka et al. 2016, Tiegs 2017, Alfonso et al. 2020). For instance, Laverty and Berger (2020) found bat activity to decrease by an equivalent ~60% at sites with MCL water quality standards above levels deemed harmful for humans.

For resource-use specifically, our results indicate that water quality influences the use of water as both a foraging and drinking resource for bats, but the most evident impact of water quality was on drinking activity. This finding makes sense as the consumption of contaminated water will have a more direct impact on the bats and, therefore, drinking activity. Note that we could not find any other studies that support or oppose our results for the impact of water quality on bat drinking activity, because our study is the first study to explore drinking behavior explicitly. In contrast, we found that foraging was influenced by water quality to a much lesser extent, particularly in our observed foraging activity. This difference in resource use is likely due, firstly, to the thermal camera field-of-view not capturing activity 10 m above the water surface. While there are some bat species known to forage close to or even glean from water surfaces (Ratcliffe et al. 2001, Zsebok et al. 2013, Denzinger et al. 2018), the majority of species in our study area prefer to forage at heights beyond 10 m (Fern et al. 2018, Reimer et al. 2018). As our acoustic detectors can record bat activity up to 35 m away and, therefore, beyond the 10 m limit of the thermal cameras, it is not surprising that we recorded 67% more in acoustic foraging activity. These survey results highlight the benefits of conducting both thermal cameras surveys and acoustic monitoring to impart a more credible representation of bat activity and resource use (Razgour et al. 2010, Buscaino et al. 2012). Thus, considering that the acoustic data will more effectively reflect foraging activity, our acoustic results are supported by a number of studies that recorded a similar impact of water quality on bat foraging (Salvarina 2016). Abbott et al. (2009), for example, found that foraging decreased significantly for the *Myotis daubentonii* and *Pipistrellus pygmaeus* when foraging at sites that were contaminated with sewage effluence downstream.

Finally, our study demonstrated that quality of drinking water negatively impacted species diversity. We recorded a 60% decrease in species among our sites with the highest pollution potential. A number of studies support our findings that water quality impacts species diversity using water resources not just for bats, but for other taxa including birds and ungulates (Larsen-Gray & Loehle 2022, Li et al. 2022). More specifically, at sites above or potentially above MCL hoary, silver-haired, tri-colored, and Mexican free-tailed bats were not recorded. These results suggest that such species may be more sensitive to water quality than other species in our study area, such as evening and eastern red bats, which were recorded at all the sites independent of water quality. Again, other studies support our findings, by showing that less tolerant species are lost when water quality declines (Korine et al. 2015, Perkin & Bonner 2016). While we confirmed at least two species in our study area are drinking from and foraging for prey over impaired water sources, this poses a concern for their health and safety, especially if individuals are frequently drinking and foraging at these sites. Studies have shown that bats and other wildlife are at risk of bioaccumulation of bacteria and other contaminants (i.e., heavy metals and *E. coli*) from both the water source and prey (Gondwe et al. 2021, Oliveira et al. 2021, Cory-Toussaint et al. 2022), thus understanding the health implications for such contamination may highlight the importance of improving urban greenspace water quality for bats.

We acknowledge that water quality is not the only factor impacting water resource use by bats. Studies have shown that factors can include, but are not limited to noise or light pollution in the surrounding area, such as the athletic fields flood lights at Site 5 (Russo et al. 2019, Michie et al. 2020, Domer et al. 2021). While another potential factor is likely to be clutter on the water surface (Broders et al. 2004, Rodriguez & Sanchez 2022). We noted that floating trash and debris appeared to reduce the available surface area, which bats could access to drink effectively at both Sites 5 and 6. Seasonal changes could also have an influence on bat activity within the area, by fluctuating water levels and potentially increasing contamination. For instance, during hot dry conditions in the summer months in north central Texas, many of our sites had limited water availability. Further studies could expand and explore these factors and their impact on resource use.

Overall, we suggest that bat resource usage and diversity in urban parks and greenspaces could potentially increase with the improvement of the water quality. With urban planners and wildlife practitioners working to enhance these study sites, it could impact not only the habitat for wildlife but the surrounding local community. From this study we propose that 1) bats can be used as an indicator of water quality and 2) water sources that are not suitable for human consumption are not suitable for bats, and potentially other wildlife. Furthermore, our study confirms that not all water sources present in an urban environment are suitable and, therefore, available to bats.

Conclusion

Understanding how the water quality of urban environments impacts bats, may not only be used as an indicator of water availability for other wildlife species, but also provide insights into the environmental health of urban areas. We recommend urban landscape practitioners and city park managers implement strategies to improve the water quality of local study sites for bats. Community outreach trash clean-ups schemes could be one way to maintain water quality, as well as promoting awareness for wildlife and engaging surrounding neighborhoods in efforts to improve the health of their local environment.

Appendix A

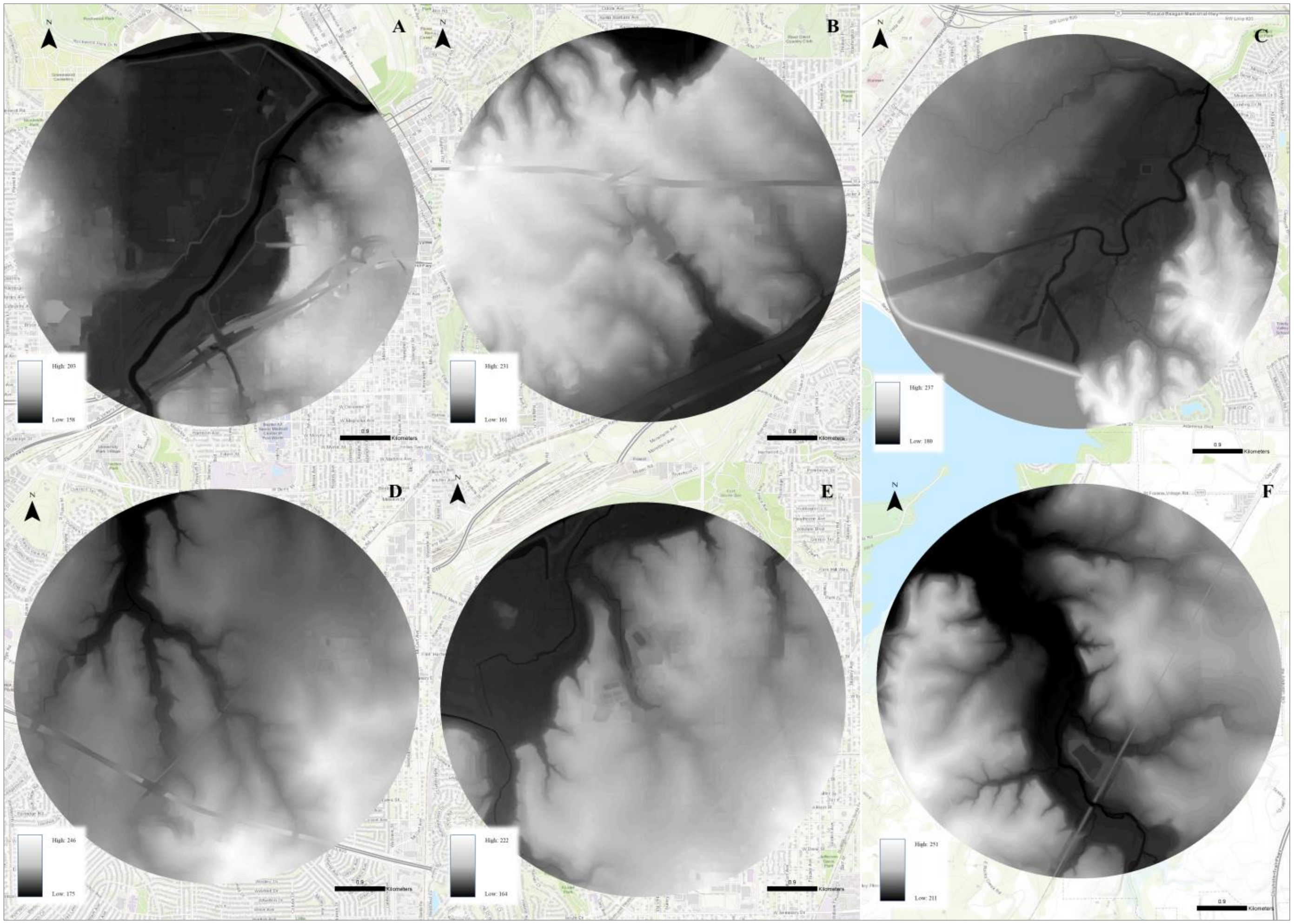


Figure A1: Digital Elevation Models in a 2 km^2 radius for six survey sites. The sites are labeled A) Trinity Duck Pond, B) Lake Como, C) Oakmont Creek, D) Foster Park Pond, E) Frat Pond, and F) Rocky Creek. The cell size for each raster is 1 m by 1 m.

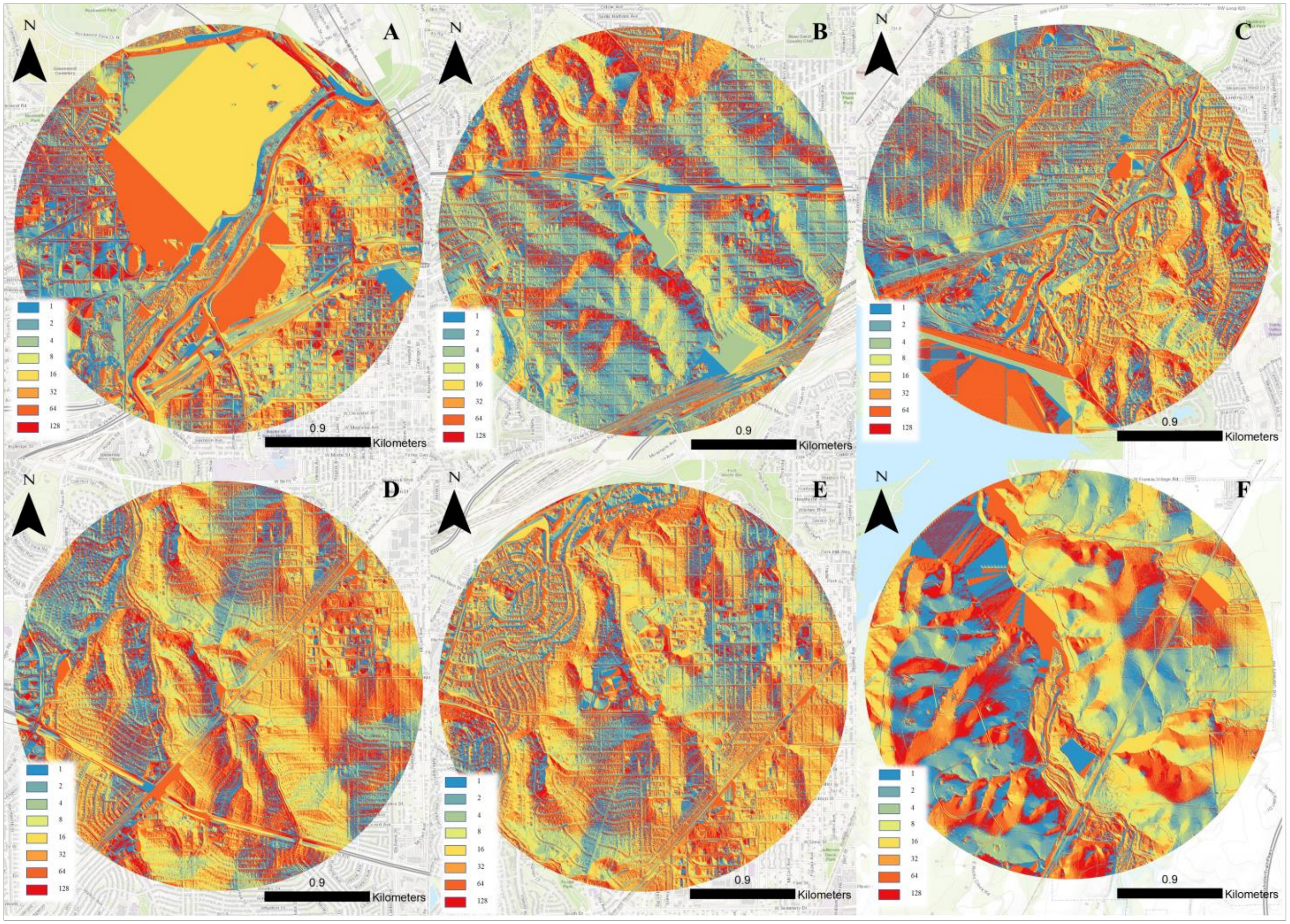


Figure A2: Flow direction maps in a 2 km^2 radius for six survey sites. The sites are labeled A) Trinity Duck Pond, B) Lake Como, C) Oakmont Creek, D) Foster Park Pond, E) Frat Pond, and F) Rocky Creek. The cell size for each raster is 1 m by 1 m.

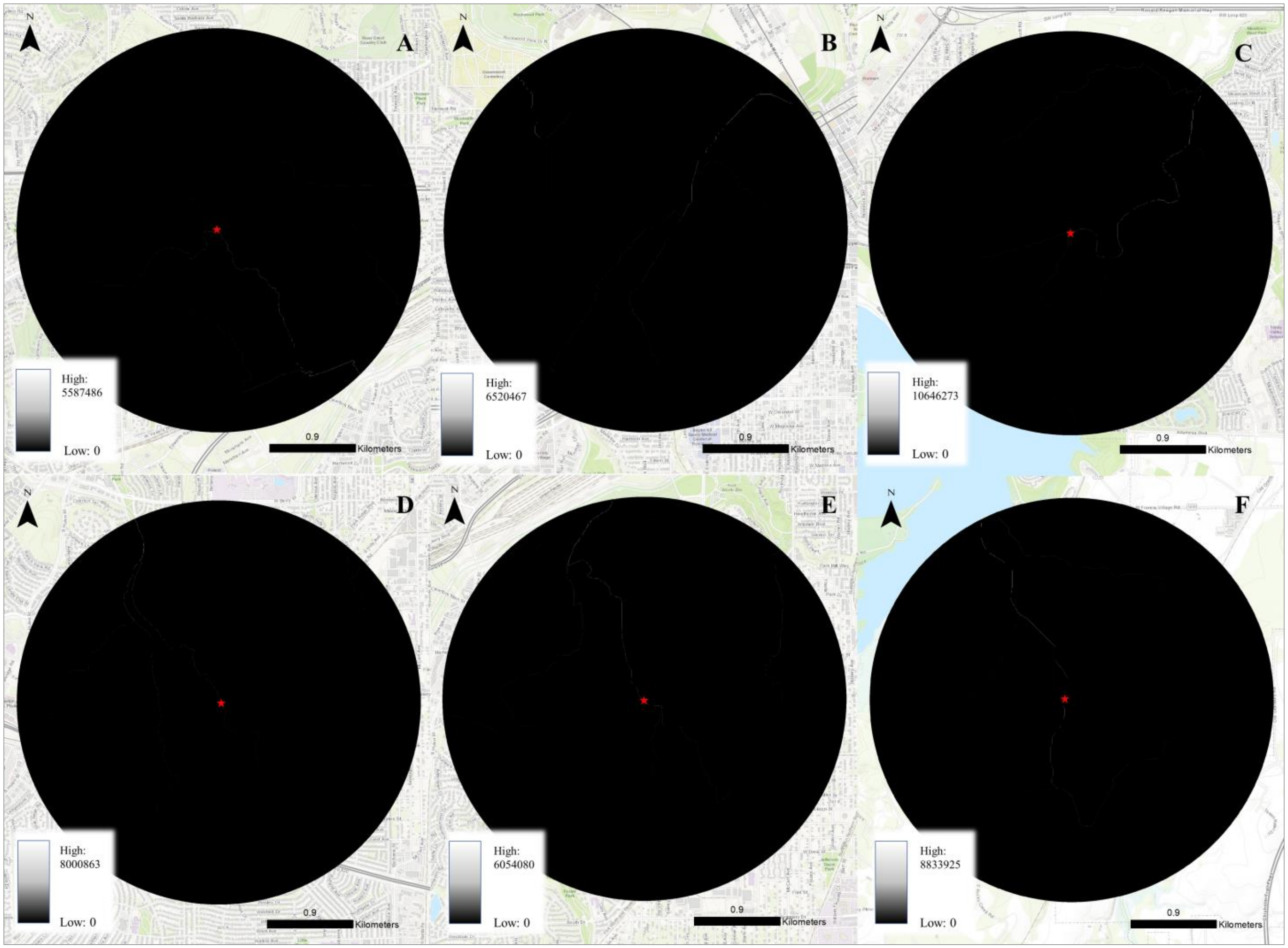


Figure A3: Flow Accumulation rasters in a 2 km^2 radius for six survey sites with the red star representing the outlet. The sites are labeled A) Lake Como B) Trinity Duck Pond, C) Oakmont Creek, D) Foster Park Pond, E) Frat Pond, and F) Rocky Creek. The cell size for each raster is 1 m by 1 m. Trinity is a closed system pond, so it did not have an outlet point.

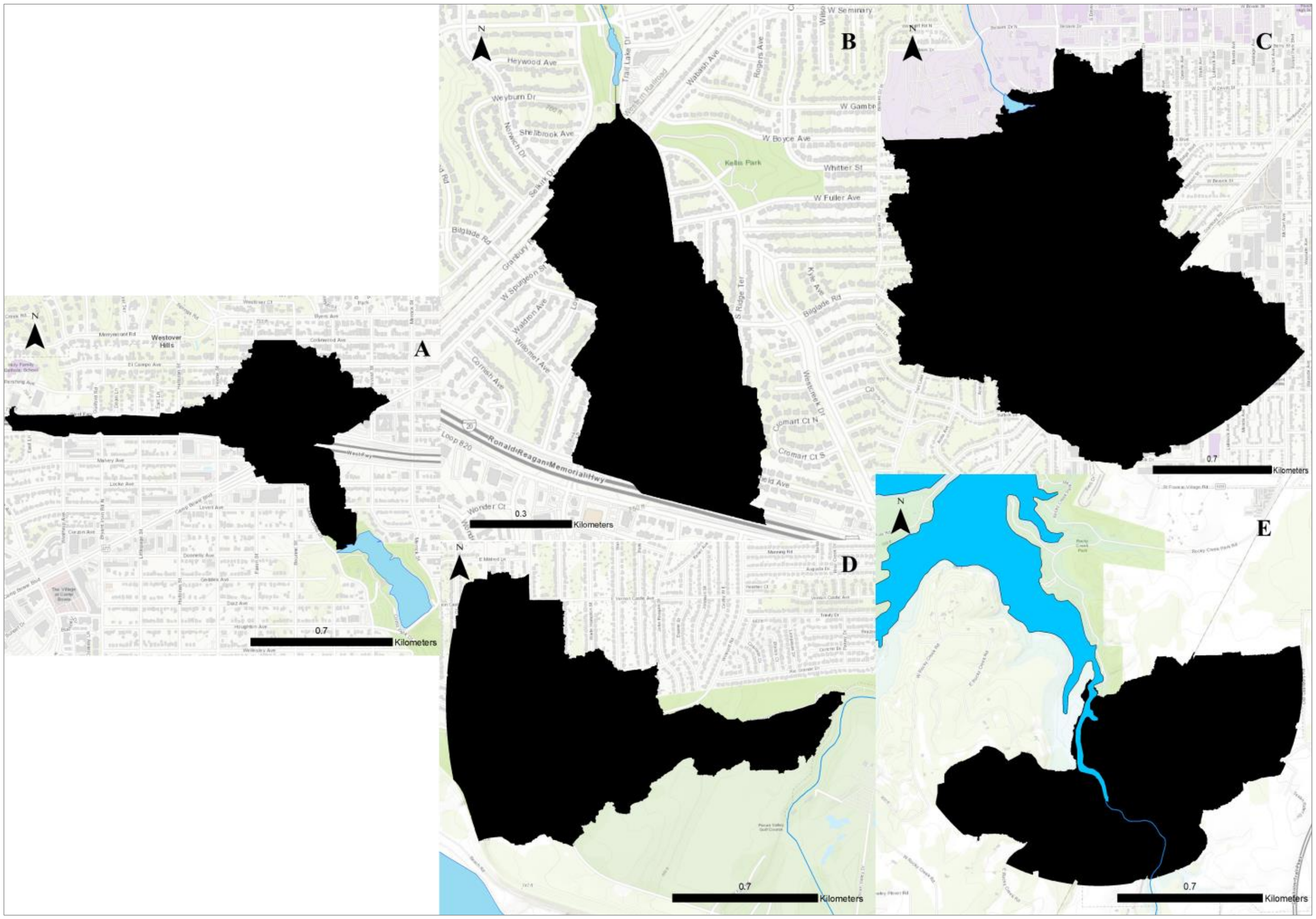


Figure A4: Generated watersheds in a 2 km^2 radius for six survey sites. The sites are labeled A) Lake Como, B) Foster Park Pond, C) Frat Pond, D) Oakmont Creek, and E) Rocky Creek. The cell size for each raster is 1 m by 1 m

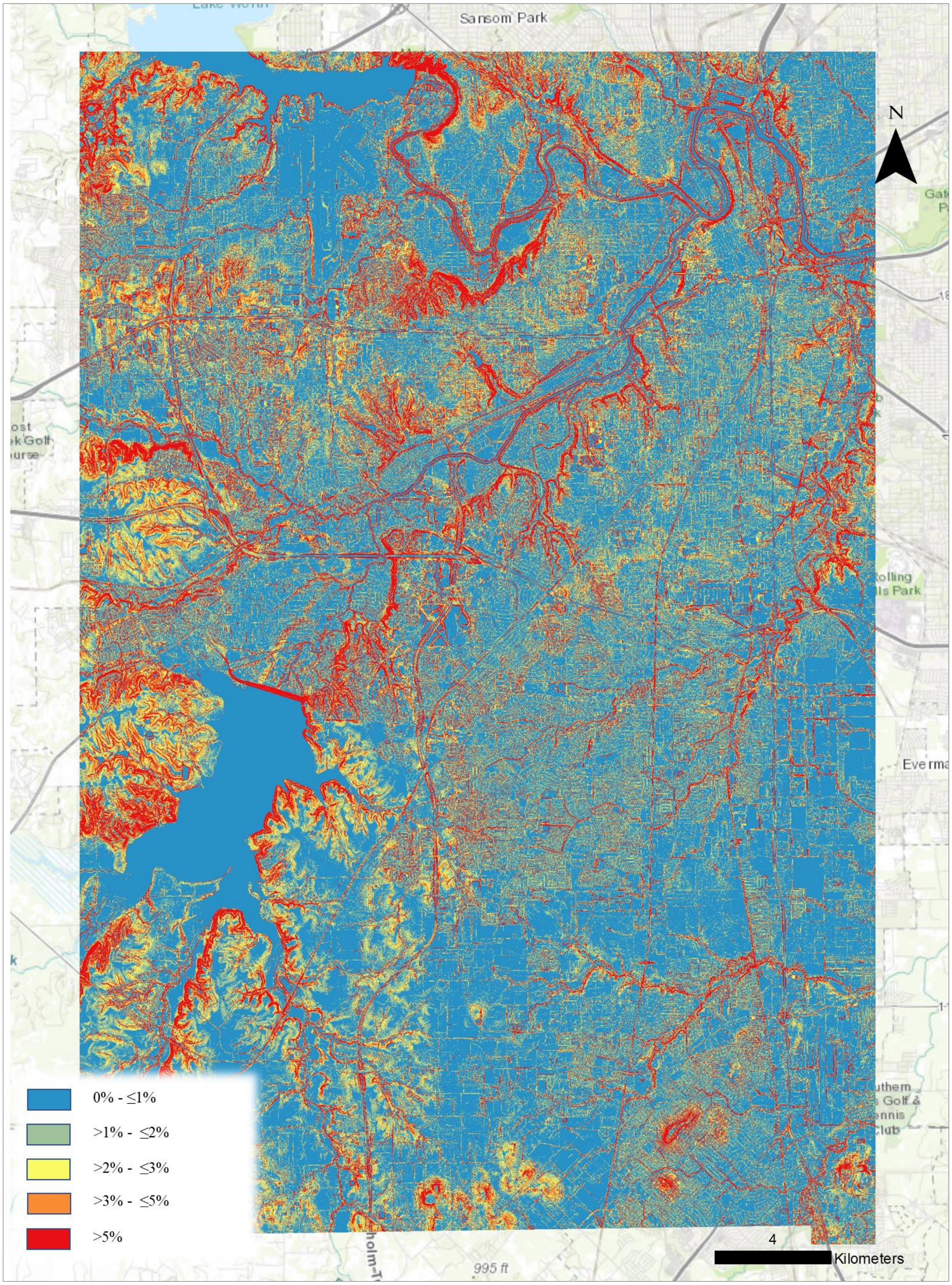


Figure A5: Slope raster created from six stitched together Digital Elevation Models from USGS over the entire study area. We then reclassified into five slope classes which was designated by using a quantile separation. The cell size for the raster is 1 m by 1 m.

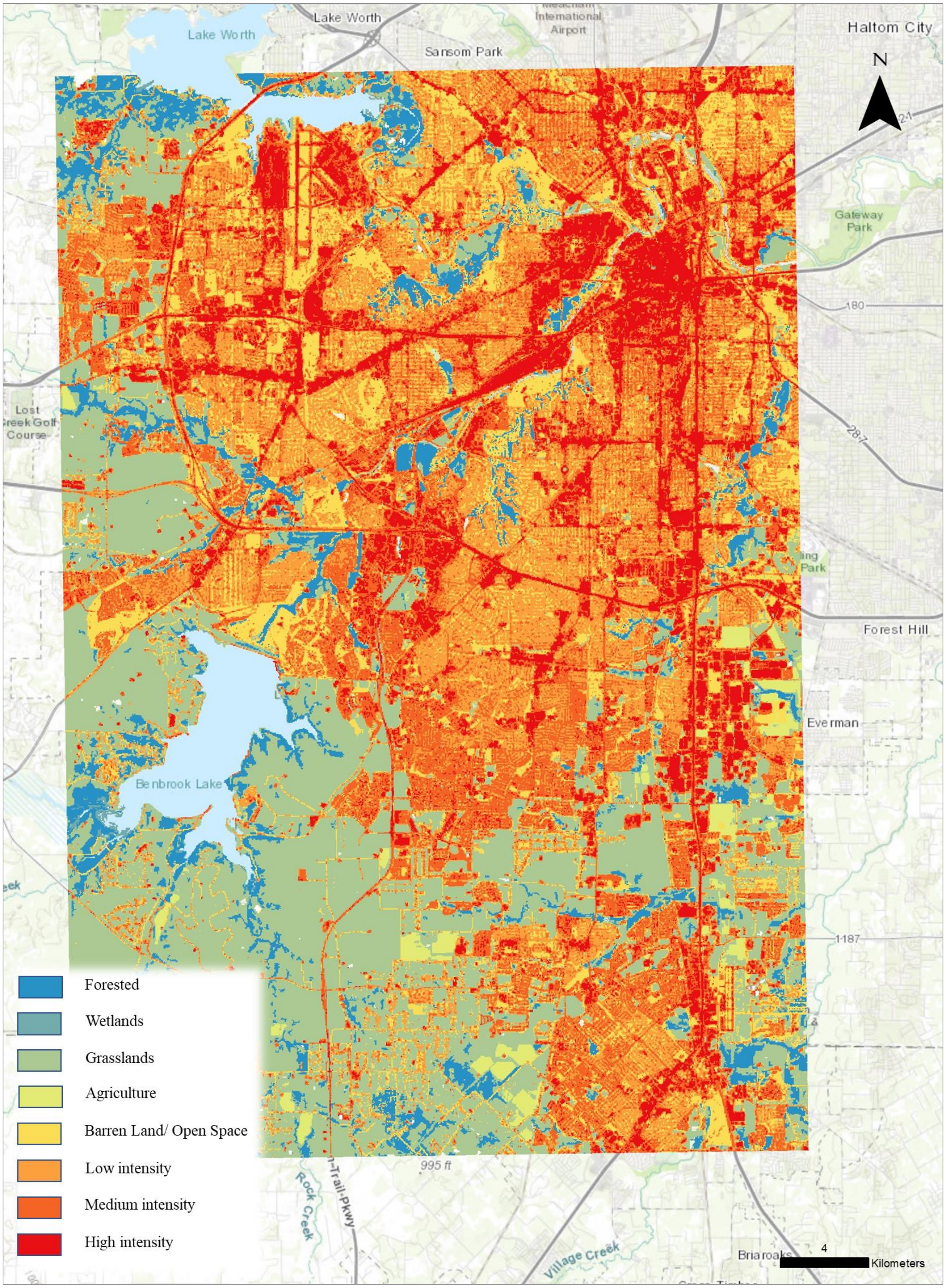


Figure A6: Landuse raster taken from the MRLC clipped to the study area. Then, we reclassified the map into eight classifications for the land cover in the study area. The cell size for the raster is 1 m by 1 m.

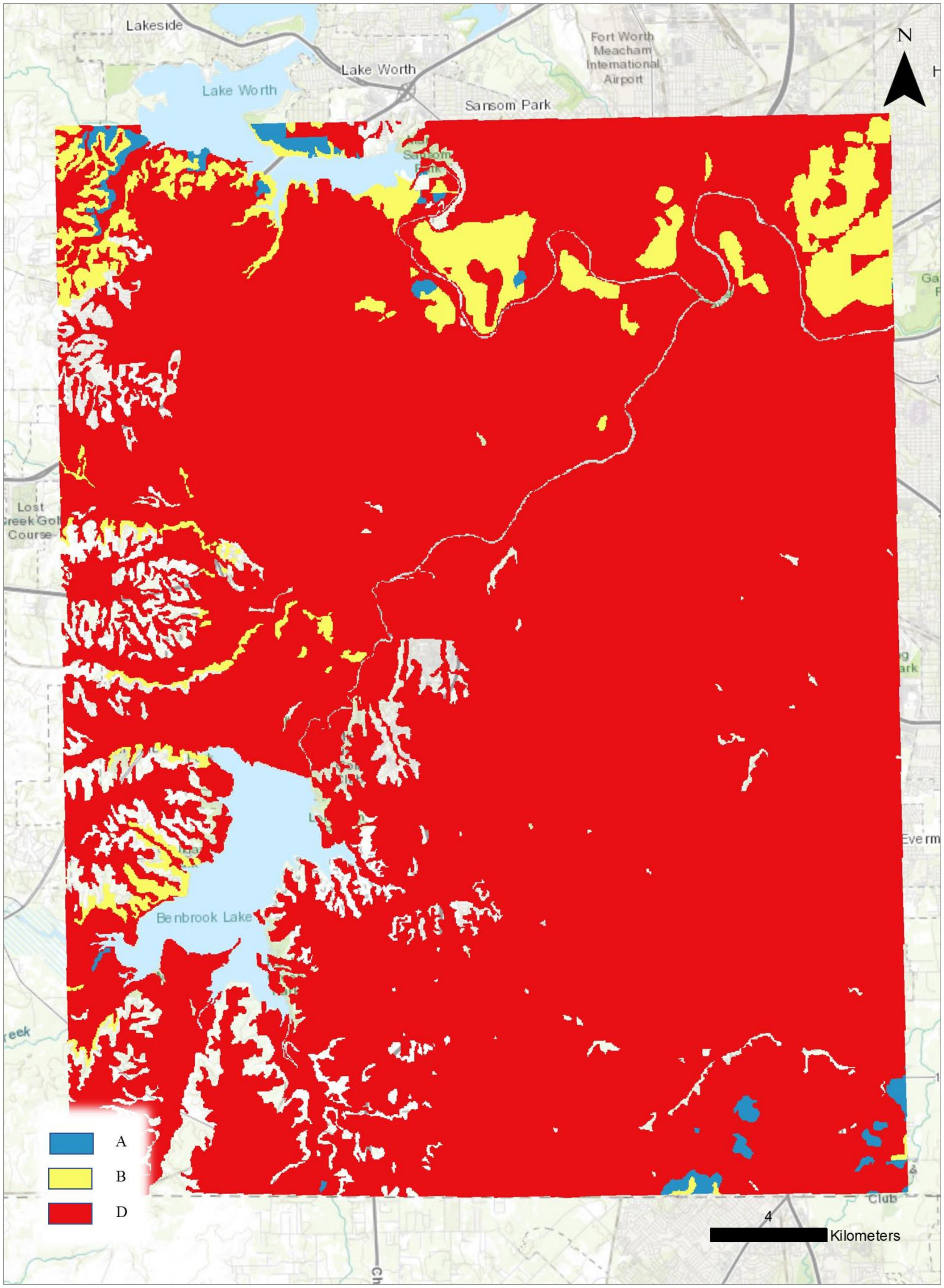


Figure A7: Soil raster of the Study area from the USDA. We clipped it to encompass the Study area and reclassified it into the four Hydrologic soil groups for which we only had three soil types in the area. The cell size for the raster is 1 m by 1 m.

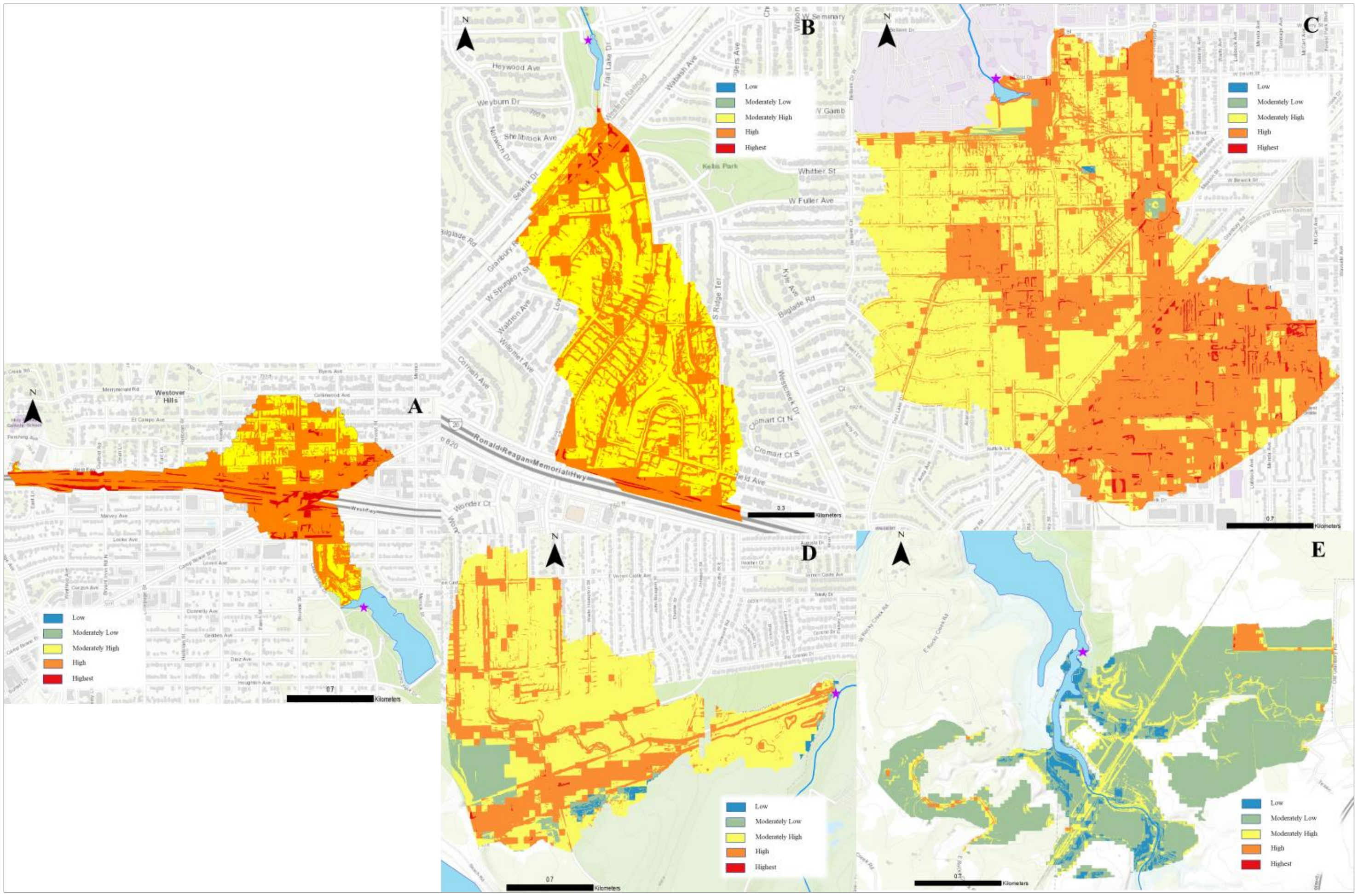


Figure A8: Suitability Maps created using the Slope, Landuse and Soil rasters shaped to the area of the watershed for six survey sites. The sites are labeled A) Lake Como, B) Foster Park Pond, C) Frat Pond, D) Oakmont Creek, and E) Rocky Creek. The cell size for each raster is 1 m by 1 m.

Appendix B

Table B1: Descriptive statistics for time bats observed in the field-of-view in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	424.92	1.68	22.87	214.97	6	34.07
Standard Error	84.80	0.76	7.23	45	2.88	9.75
Median	288	0	7	143	0	3.76
Mode	N/A	0	0	32	0	0
Standard Deviation	448.73	4.01	38.27	238.11	15.26	51.61
Sample Variance	201358.42	16.08	1464.65	56696.63	232.74	2663.94
Kurtosis	5.45	16.96	5.29	1.56	6.44	2.17
Skewness	2.11	3.85	2.35	1.54	2.74	1.67
Range	2033	20	155	872	55	186.49
Minimum	17	0	0	0	0	0
Maximum	2050	20	155	872	55	186.49
Sum	11897.80	47	640.36	6019.09	168	954.02
Count	28	28	28	28	28	28

Table B2: Descriptive statistics for time bats observed foraging in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	31.65	0	3.05	6.32	0	1.96
Standard Error	13.2295	0	1.13	2.27	0	0.86
Median	4.5	0	0	0	0	0
Mode	0	0	0	0	0	0
Standard Deviation	7041	0	5.96	12.02	0	4.54
Sample Variance	4900.57	0	35.55	144.45	0	20.63
Kurtosis	9.77008	.	2.38	11.71	.	9.11
Skewness	3.03706	.	1.87	3.13	.	2.90
Range	313	0	20	57	0	20
Minimum	0	0	0	0	0	0
Maximum	313	0	20	57	0	20
Sum	886.2	0	85.27	177	0	55
Count	28	28	28	28	28	28

Table B3: Descriptive statistics for number of drinking events in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	30.24	0.25	1.25	28.62	0.21	2.57
Standard Error	4.79	0.18	0.36	4.68	0.15	0.78
Median	21	0	0	25.5	0	0
Mode	12	0	0	0	0	0
Standard Deviation	25.35	0.97	1.90	24.77	0.79	4.14
Sample Variance	642.40	0.94	3.62	613.31	0.62	17.12
Kurtosis	-0.79	23.61	4.75	2.24	21.53	1.21
Skewness	0.76	4.75	2.02	1.26	4.50	1.58
Range	84	5	8	107	4	13
Minimum	2	0	0	0	0	0
Maximum	86	5	8	107	4	13
Sum	846.60	7	35.09	801.18	6	71.86
Count	28	28	28	28	28	28

Table B4: Descriptive statistics for number of bat calls in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	573.85	38.61	60.86	162	15.36	77.70
Standard Error	49.39	11.38	13.16	24.32	3.80	20.19
Median	615	30.50	39	118.50	7	43
Mode	627	47	0	140	0	0
Standard Deviation	256.66	60.20	69.63	128.70	20.08	104.89
Sample Variance	65876.36	3624.40	4847.83	16563.85	403.28	11002.14
Kurtosis	0.76	20.72	1.68	-0.01	0.70	5.35
Skewness	0.60	4.28	1.43	1	1.39	2.09
Range	1127	326	269	462	68	454
Minimum	142	0	0	4	0	0
Maximum	1269	326	269	466	68	454
Sum	15494	1081	1704	4536	430	2098
Count	27	28	28	28	28	27

Table B5: Descriptive statistics for number of resource-related calls in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	352.15	18.61	26.32	85.36	4.36	35.63
Standard Error	31.32	6.24	5.57	13	1.39	9.38
Median	309	13	14.50	64	0	17
Mode	.	3	0	39	0	0
Standard Deviation	162.73	33	29.47	68.78	7.38	48.74
Sample Variance	26481.82	1089.06	868.30	4730.31	54.46	2375.78
Kurtosis	-0.64	21.87	1.41	0.31	4.29	2.12
Skewness	0.40	4.46	1.27	1.10	2.12	1.65
Range	585	178	114	258	29	177
Minimum	80	0	0	5	0	0
Maximum	665	178	114	263	29	177
Sum	9508	521	737	2390	122	962
Count	27	28	28	28	28	27

Table B6: Descriptive statistics for number of feeding buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	111.63	2.07	2.93	15.96	0.50	5.70
Standard Error	19.86	1.13	0.98	4.63	0.22	2.48
Median	71	0	0	2.50	0	1
Mode	120	0	0	0	0	0
Standard Deviation	103.19	5.99	5.18	24.48	1.14	12.89
Sample Variance	10648.17	35.92	26.81	599.37	1.30	166.22
Kurtosis	-0.72	18.82	12.19	2.24	9.18	10.33
Skewness	0.83	4.19	3.13	1.70	2.92	3.15
Range	319	30	25	91	5	57
Minimum	4	0	0	0	0	0
Maximum	323	30	25	91	5	57
Sum	3014	58	82	447	14	154
Count	27	28	28	28	28	27

Table B7: Descriptive statistics for number of drinking buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	71.33	1	0.39	18.68	0.32	1.15
Standard Error	7.98	0.25	0.16	3.52	0.19	0.38
Median	68	1	0	14.50	0	0
Mode	27	0	0	1	0	0
Standard Deviation	41.48	1.33	0.83	18.61	0.98	1.97
Sample Variance	1720.62	1.78	0.69	346.15	0.97	3.90
Kurtosis	-0.78	2.26	3.11	1.83	20.40	4.31
Skewness	0.44	1.62	2.03	1.45	4.32	1.98
Range	148	5	3	69	5	8
Minimum	19	0	0	0	0	0
Maximum	167	5	3	69	5	8
Sum	1926	28	11	523	9	31
Count	27	28	28	28	28	27

Table B8: Descriptive statistics for number of species in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	4.48	2.89	3.21	3.11	1.96	2.96
Standard Error	0.17	0.25	0.33	0.14	0.28	0.32
Median	5	3	3	3	2	3
Mode	5	4	3	3	3	4
Standard Deviation	0.89	1.31	1.75	0.74	1.48	1.65
Sample Variance	0.80	1.73	3.06	0.54	2.18	2.73
Kurtosis	-0.60	0.40	-0.56	2.70	-1.43	-0.94
Skewness	-0.12	-1.16	-0.44	-0.17	-0.23	-0.49
Range	3	4	6	4	4	5
Minimum	3	0	0	1	0	0
Maximum	6	4	6	5	4	5
Sum	121	81	90	87	55	80
Count	27	28	28	28	28	27

Table B9: Descriptive statistics for number of evening bats resource-related calls in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	231.56	9	11.50	58.57	2.07	19.11
Standard Error	34.50	2.97	3.43	11.05	0.86	4.97
Median	179	5.50	4.50	30	0	7
Mode	75	0	0	15	0	0
Standard Deviation	179.27	15.73	18.14	58.46	4.57	25.85
Sample Variance	32136.95	247.33	329.22	3417.74	20.88	668.10
Kurtosis	-0.51	13.65	14.69	0.85	13.66	4.95
Skewness	0.76	3.50	3.43	1.34	3.44	2.04
Range	592	77	92	193	22	110
Minimum	39	0	0	1	0	0
Maximum	631	77	92	194	22	110
Sum	6252	252	322	1640	58	516
Count	27	28	28	28	28	27

Table B10: Descriptive statistics for number of eastern red bats resource-related calls in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	33.33	6.96	10.04	21.14	3.21	10.37
Standard Error	4.75	2.76	2.52	4.51	1.43	3.15
Median	27	3.50	4.50	9.50	0	4
Mode	24	2	0	5	0	0
Standard Deviation	24.69	14.62	13.33	23.85	7.58	16.35
Sample Variance	609.77	213.89	177.59	568.87	57.51	267.40
Kurtosis	-0.54	23.89	2.96	0.52	9.48	6.39
Skewness	0.69	4.73	1.75	1.25	3.14	2.50
Range	88	79	52	83	31	67
Minimum	2	0	0	0	0	0
Maximum	90	79	52	83	31	67
Sum	900	195	281	592	90	280
Count	27	28	28	28	28	27

Table B11: Descriptive statistics for number of silver-haired bats resource-related calls in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	74.30	1.75	1.79	3.79	0.75	3.63
Standard Error	16.50	0.60	0.69	1.11	0.28	1.09
Median	31	0	0	1	0	0
Mode	4	0	0	0	0	0
Standard Deviation	85.71	3.16	3.64	5.87	1.46	5.64
Sample Variance	7346.60	9.97	13.29	34.47	2.12	31.86
Kurtosis	1.68	3.14	7.58	6.29	5.98	2.15
Skewness	1.39	2.02	2.76	2.40	2.41	1.74
Range	330	11	15	25	6	19
Minimum	2	0	0	0	0	0
Maximum	332	11	15	25	6	19
Sum	2006	49	50	106	21	98
Count	27	28	28	28	28	27

Table B12: Descriptive statistics for number of evening bats feeding buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	104.07	1.43	1.64	10.86	0.32	3.74
Standard Error	19.71	0.85	0.73	3.34	0.14	1.57
Median	65	0	0	1.50	0	0
Mode	.	0	0	0	0	0
Standard Deviation	102.41	4.48	3.86	17.68	0.72	8.17
Sample Variance	10488.07	20.03	14.90	312.72	0.52	66.81
Kurtosis	-0.57	15	20.33	2.21	6.89	7.41
Skewness	0.91	3.84	4.26	1.78	2.58	2.77
Range	317	21	20	60	3	33
Minimum	3	0	0	0	0	0
Maximum	320	21	20	60	3	33
Sum	2810	40	46	304	9	101
Count	27	28	28	28	28	27

Table B13: Descriptive statistics for number of eastern red bats feeding buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	4	0.50	0.50	2.46	0.18	0.81
Standard Error	0.78	0.31	0.22	1.40	0.13	0.47
Median	3	0	0	0	0	0
Mode	2	0	0	0	0	0
Standard Deviation	4.06	1.67	1.14	7.40	0.67	2.43
Sample Variance	16.46	2.78	1.30	54.70	0.45	5.93
Kurtosis	4.35	16.52	9.18	24.20	13.76	18.41
Skewness	1.80	3.98	2.92	4.79	3.77	4.12
Range	18	8	5	39	3	12
Minimum	0	0	0	0	0	0
Maximum	18	8	5	39	3	12
Sum	108	14	14	69	5	22
Count	27	28	28	28	28	27

Table B14: Descriptive statistics for number of silver-haired bats feeding buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	2.78	0.11	0.11	0.46	0	0.56
Standard Error	0.76	0.06	0.11	0.17	0	0.23
Median	1	0	0	0	0	0
Mode	0	0	0	0	0	0
Standard Deviation	3.96	0.31	0.57	0.92	0	1.19
Sample Variance	15.72	0.10	0.32	0.85	0	1.41
Kurtosis	4.38	5.61	28	2.71	.	4.40
Skewness	2.08	2.69	5.29	1.94	.	2.31
Range	16	1	3	3	0	4
Minimum	0	0	0	0	0	0
Maximum	16	1	3	3	0	4
Sum	75	3	3	13	0	15
Count	27	28	28	28	28	27

Table B15: Descriptive statistics for number of evening bats drinking buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

Descriptive Statistics	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	61.63	0.43	0.21	13.07	0.04	0.70
Standard Error	8.04	0.14	0.11	2.57	0.04	0.30
Median	56	0	0	12	0	0
Mode	16	0	0	1	0	0
Standard Deviation	41.79	0.74	0.57	13.60	0.19	1.56
Sample Variance	1746.17	0.55	0.32	184.88	0.04	2.45
Kurtosis	-0.54	0.53	6.03	1.62	28	9.87
Skewness	0.66	1.44	2.64	1.42	5.29	2.94
Range	147	2	2	52	1	7
Minimum	16	0	0	0	0	0
Maximum	163	2	2	52	1	7
Sum	1664	12	6	366	1	19
Count	27	28	28	28	28	27

Table B16: Descriptive statistics for number of eastern red bats drinking buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The sites ranked by pollution potential from left to right; Trinity Duck Pond, Rocky Creek, Oakmont Creek, Foster Park Pond, Frat Pond, and Lake Como. Shading highlights data that are not normally distributed.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean	7.93	0.39	0.18	5.18	0.29	0.44
Standard Error	1.33	0.16	0.09	1.46	0.18	0.17
Median	6	0	0	1.50	0	0
Mode	3	0	0	0	0	0
Standard Deviation	6.90	0.83	0.48	7.75	0.98	0.89
Sample Variance	47.61	0.69	0.23	60.08	0.95	0.79
Kurtosis	1.54	5.68	7.85	3.45	21.88	9.43
Skewness	1.34	2.45	2.81	1.99	4.52	2.82
Range	27	3	2	28	5	4
Minimum	0	0	0	0	0	0
Maximum	27	3	2	28	5	4
Sum	214	11	5	145	8	12
Count	27	28	28	28	28	27

Appendix C

Table C1: Post Hoc Tukey test for time bats observed in the field-of-view in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA. The six sites

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-97.125	12.652	-7.677	<.001	0.000
Site 1 vs Site 3	-69.536	12.652	-5.496	<.001	0.000
Site 1 vs Site 4	-17.679	12.652	-1.397	0.162	1.000
Site 1 vs Site 5	-95.839	12.652	-7.575	<.001	0.000
Site 1 vs Site 6	-70.179	12.652	-5.547	<.001	0.000
Site 2 vs Site 3	27.589	12.652	2.181	0.029	0.438
Site 2 vs Site 4	79.446	12.652	6.279	<.001	0.000
Site 2 vs Site 5	1.286	12.652	0.102	0.919	1.000
Site 2 vs Site 6	26.946	12.652	2.130	0.033	0.498
Site 3 vs Site 4	51.857	12.652	4.099	<.001	0.001
Site 3 vs Site 5	-26.304	12.652	-2.079	0.038	0.564
Site 3 vs Site 6	-0.643	12.652	-0.051	0.959	1.000
Site 4 vs Site 5	78.161	12.652	6.178	<.001	0.000
Site 4 vs Site 6	-52.500	12.652	-4.150	<.001	0.000
Site 5 vs Site 6	25.661	12.652	2.028	0.043	0.638

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C2: Post Hoc Tukey test for time bats observed foraging in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-56.018	10.133	-5.528	<.001	0.000
Site 1 vs Site 3	-34.482	10.133	-3.403	<.001	0.010
Site 1 vs Site 4	-17.768	10.133	-1.754	0.080	1.000
Site 1 vs Site 5	-56.018	10.133	-5.528	<.001	0.000
Site 1 vs Site 6	-36.821	10.133	-3.634	<.001	0.004
Site 2 vs Site 3	21.536	10.133	2.125	0.034	0.503
Site 2 vs Site 4	38.250	10.133	3.775	<.001	0.002
Site 2 vs Site 5	0.000	10.133	0.000	1.000	1.000
Site 2 vs Site 6	19.196	10.133	1.895	0.058	0.872
Site 3 vs Site 4	16.714	10.133	1.650	0.099	1.000
Site 3 vs Site 5	-21.536	10.133	-2.125	0.034	0.503
Site 3 vs Site 6	-2.339	10.133	-0.231	0.817	1.000
Site 4 vs Site 5	38.250	10.133	3.775	<.001	0.002
Site 4 vs Site 6	-19.054	10.133	-1.880	0.060	0.901
Site 5 vs Site 6	19.196	10.133	1.895	0.058	0.872

**a* Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C3: Post Hoc Tukey test for number of bats observed drinking in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-90.804	12.188	-7.45	<.001	0.000
Site 1 vs Site 3	-70.089	12.188	-5.75	<.001	0.000
Site 1 vs Site 4	-6.036	12.188	-0.495	0.620	1.000
Site 1 vs Site 5	-90.982	12.188	-7.465	<.001	0.000
Site 1 vs Site 6	-67.161	12.188	-5.51	<.001	0.000
Site 2 vs Site 3	20.714	12.188	1.7	0.089	1.000
Site 2 vs Site 4	84.768	12.188	6.955	<.001	0.000
Site 2 vs Site 5	-0.179	12.188	-0.015	0.988	1.000
Site 2 vs Site 6	23.643	12.188	1.94	0.052	0.786
Site 3 vs Site 4	64.054	12.188	5.255	<.001	0.000
Site 3 vs Site 5	-20.893	12.188	-1.714	0.086	1.000
Site 3 vs Site 6	2.929	12.188	0.24	0.810	1.000
Site 4 vs Site 5	84.946	12.188	6.969	<.001	0.000
Site 4 vs Site 6	-61.125	12.188	-5.015	<.001	0.000
Site 5 vs Site 6	23.821	12.188	1.954	0.051	0.76

*****a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C4: Post Hoc Tukey test for number of bat passes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-89.246	12.956	-6.888	<.001	0.000
Site 1 vs Site 3	-79.585	12.956	-6.143	<.001	0.000
Site 1 vs Site 4	-41.96	12.956	-3.239	0.001	0.018
Site 1 vs Site 5	-109.46	12.956	-8.448	<.001	0.000
Site 1 vs Site 6	-79.13	13.074	-6.053	<.001	0.000
Site 2 vs Site 3	9.661	12.838	0.753	0.452	1.000
Site 2 vs Site 4	47.286	12.838	3.683	<.001	0.003
Site 2 vs Site 5	-20.214	12.838	-1.575	0.115	1.000
Site 2 vs Site 6	10.116	12.956	0.781	0.435	1.000
Site 3 vs Site 4	37.625	12.838	2.931	0.003	0.051
Site 3 vs Site 5	-29.875	12.838	-2.327	0.020	0.299
Site 3 vs Site 6	0.456	12.956	0.035	0.972	1.000
Site 4 vs Site 5	67.5	12.838	5.258	<.001	0.000
Site 4 vs Site 6	-37.169	12.956	-2.869	0.004	0.062
Site 5 vs Site 6	30.331	12.956	2.341	0.019	0.288

*****a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C5: Post Hoc Tukey test for number of resource-related calls in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-87.542	12.916	-6.778	<.001	0.000
Site 1 vs Site 3	-80.649	12.916	-6.244	<.001	0.000
Site 1 vs Site 4	-38.738	12.916	-2.999	0.003	0.041
Site 1 vs Site 5	-115.667	12.916	-8.955	<.001	0.000
Site 1 vs Site 6	-81.481	13.033	-6.252	<.001	0.000
Site 2 vs Site 3	6.893	12.798	0.539	0.590	1.000
Site 2 vs Site 4	48.804	12.798	3.813	<.001	0.002
Site 2 vs Site 5	-28.125	12.798	-2.198	0.028	0.420
Site 2 vs Site 6	6.06	12.916	0.469	0.639	1.000
Site 3 vs Site 4	41.911	12.798	3.275	0.001	0.016
Site 3 vs Site 5	-35.018	12.798	-2.736	0.006	0.093
Site 3 vs Site 6	-0.833	12.916	-0.064	0.949	1.000
Site 4 vs Site 5	76.929	12.798	6.011	<.001	0.000
Site 4 vs Site 6	-42.743	12.916	-3.309	<.001	0.014
Site 5 vs Site 6	34.185	12.916	2.647	0.008	0.122

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C6: Post Hoc Tukey test for number of feeding buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-87.265	12.367	-7.056	<.001	0.000
Site 1 vs Site 3	-75.282	12.367	-6.087	<.001	0.000
Site 1 vs Site 4	-50.122	12.367	-4.053	<.001	0.001
Site 1 vs Site 5	-96.193	12.367	-7.778	<.001	0.000
Site 1 vs Site 6	-72.611	12.479	-5.819	<.001	0.000
Site 2 vs Site 3	11.982	12.254	0.978	0.328	1.000
Site 2 vs Site 4	37.143	12.254	3.031	0.002	0.037
Site 2 vs Site 5	-8.929	12.254	-0.729	0.466	1.000
Site 2 vs Site 6	14.653	12.367	1.185	0.236	1.000
Site 3 vs Site 4	25.161	12.254	2.053	0.040	0.601
Site 3 vs Site 5	-20.911	12.254	-1.706	0.088	1.000
Site 3 vs Site 6	2.671	12.367	0.216	0.829	1.000
Site 4 vs Site 5	46.071	12.254	3.76	<.001	0.003
Site 4 vs Site 6	-22.489	12.367	-1.818	0.069	1.000
Site 5 vs Site 6	23.582	12.367	1.907	0.057	0.848

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C7: Post Hoc Tukey test for number of drinking buzzes in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-80.962	12.287	-6.589	<.001	0.000
Site 1 vs Site 3	-98.855	12.287	-8.046	<.001	0.000
Site 1 vs Site 4	-30.052	12.287	-2.446	0.014	0.217
Site 1 vs Site 5	-101.712	12.287	-8.278	<.001	0.000
Site 1 vs Site 6	-88.463	12.398	-7.135	<.001	0.000
Site 2 vs Site 3	-17.893	12.175	-1.47	0.142	1.000
Site 2 vs Site 4	50.911	12.175	4.182	<.001	0.000
Site 2 vs Site 5	-20.75	12.175	-1.704	0.088	1.000
Site 2 vs Site 6	-7.501	12.287	-0.61	0.542	1.000
Site 3 vs Site 4	68.804	12.175	5.651	<.001	0.000
Site 3 vs Site 5	-2.857	12.175	-0.235	0.814	1.000
Site 3 vs Site 6	10.392	12.287	0.846	0.398	1.000
Site 4 vs Site 5	71.661	12.175	5.886	<.001	0.000
Site 4 vs Site 6	-58.411	12.287	-4.754	<.001	0.000
Site 5 vs Site 6	13.249	12.287	1.078	0.281	1.000

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C8: Post Hoc Tukey test for Number of species observed 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-51.946	12.61	-4.119	<.001	0.001
Site 1 vs Site 3	-40.732	12.61	-3.23	0.001	0.019
Site 1 vs Site 4	-52.143	12.61	-4.135	<.001	0.001
Site 1 vs Site 5	-79.357	12.61	-6.293	<.001	0.000
Site 1 vs Site 6	-47.259	12.724	-3.714	<.001	0.003
Site 2 vs Site 3	11.214	12.495	0.898	0.369	1.000
Site 2 vs Site 4	-0.196	12.495	-0.016	0.987	1.000
Site 2 vs Site 5	-27.411	12.495	-2.194	0.028	0.424
Site 2 vs Site 6	4.687	12.61	0.372	0.710	1.000
Site 3 vs Site 4	-11.411	12.495	-0.913	0.361	1.000
Site 3 vs Site 5	-38.625	12.495	-3.091	0.002	0.030
Site 3 vs Site 6	-6.527	12.61	-0.518	0.605	1.000
Site 4 vs Site 5	27.214	12.495	2.178	0.029	0.441
Site 4 vs Site 6	4.884	12.61	0.387	0.699	1.000
Site 5 vs Site 6	32.098	12.61	2.545	0.011	0.164

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C9: Post Hoc Tukey test for number of resource-related calls for evening bats in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	K test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-85.29	12.892	-6.616	<.001	0.000
Site 1 vs Site 3	-81.63	12.892	-6.332	<.001	0.000
Site 1 vs Site 4	-33.04	12.892	-2.563	0.010	0.156
Site 1 vs Site 5	-111.183	12.892	-8.624	<.001	0.000
Site 1 vs Site 6	-71.611	13.009	-5.505	<.001	0.000
Site 2 vs Site 3	3.661	12.774	0.287	0.774	1.000
Site 2 vs Site 4	52.25	12.774	4.09	<.001	0.001
Site 2 vs Site 5	-25.893	12.774	-2.027	0.043	0.640
Site 2 vs Site 6	13.679	12.892	1.061	0.289	1.000
Site 3 vs Site 4	48.589	12.774	3.804	<.001	0.002
Site 3 vs Site 5	-29.554	12.774	-2.314	0.021	0.310
Site 3 vs Site 6	10.019	12.892	0.777	0.437	1.000
Site 4 vs Site 5	78.143	12.774	6.117	<.001	0.000
Site 4 vs Site 6	-38.571	12.892	-2.992	0.003	0.042
Site 5 vs Site 6	39.572	12.892	3.069	0.002	0.032

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C10: Post Hoc Tukey test for Average number of resource-related calls for eastern red bats in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-59.319	12.864	-4.611	<.001	0.000
Site 1 vs Site 3	-53.64	12.864	-4.17	<.001	0.000
Site 1 vs Site 4	-27.051	12.864	-2.103	0.035	0.532
Site 1 vs Site 5	-86.176	12.864	-6.699	<.001	0.000
Site 1 vs Site 6	-53.944	12.98	-4.156	<.001	0.000
Site 2 vs Site 3	5.679	12.747	0.446	0.656	1.000
Site 2 vs Site 4	32.268	12.747	2.532	0.011	0.170
Site 2 vs Site 5	-26.857	12.747	-2.107	0.035	0.527
Site 2 vs Site 6	5.374	12.864	0.418	0.676	1.000
Site 3 vs Site 4	26.589	12.747	2.086	0.037	0.555
Site 3 vs Site 5	-32.536	12.747	-2.553	0.011	0.160
Site 3 vs Site 6	-0.304	12.864	-0.024	0.981	1.000
Site 4 vs Site 5	59.125	12.747	4.639	<.001	0.000
Site 4 vs Site 6	-26.894	12.864	-2.091	0.037	0.548
Site 5 vs Site 6	32.231	12.864	2.506	0.012	0.183

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C11: Post Hoc Tukey test for number of resource-related calls for silver-haired bats in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA.

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-78.04	12.314	-6.338	<.001	0.000
Site 1 vs Site 3	-78.647	12.314	-6.387	<.001	0.000
Site 1 vs Site 4	-57.575	12.314	-4.676	<.001	0.000
Site 1 vs Site 5	-87.183	12.314	-7.08	<.001	0.000
Site 1 vs Site 6	-66.185	12.425	-5.327	<.001	0.000
Site 2 vs Site 3	-0.607	12.201	-0.05	0.960	1.000
Site 2 vs Site 4	20.464	12.201	1.677	0.093	1.000
Site 2 vs Site 5	-9.143	12.201	-0.749	0.454	1.000
Site 2 vs Site 6	11.854	12.314	0.963	0.336	1.000
Site 3 vs Site 4	21.071	12.201	1.727	0.084	1.000
Site 3 vs Site 5	-8.536	12.201	-0.7	0.484	1.000
Site 3 vs Site 6	12.462	12.314	1.012	0.312	1.000
Site 4 vs Site 5	29.607	12.201	2.427	0.015	0.229
Site 4 vs Site 6	-8.61	12.314	-0.699	0.484	1.000
Site 5 vs Site 6	20.997	12.314	1.705	0.088	1.000

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C12: Post Hoc Tukey test for number of feeding buzzes for evening bats in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-86.581	11.918	-7.265	<.001	0.000
Site 1 vs Site 3	-76.206	11.918	-6.394	<.001	0.000
Site 1 vs Site 4	-55.938	11.918	-4.694	<.001	0.000
Site 1 vs Site 5	-90.849	11.918	-7.623	<.001	0.000
Site 1 vs Site 6	-74.037	12.025	-6.157	<.001	0.000
Site 2 vs Site 3	10.375	11.809	0.879	0.38	1.000
Site 2 vs Site 4	30.643	11.809	2.595	0.009	0.142
Site 2 vs Site 5	-4.268	11.809	-0.361	0.718	1.000
Site 2 vs Site 6	12.544	11.918	1.053	0.293	1.000
Site 3 vs Site 4	20.268	11.809	1.716	0.086	1.000
Site 3 vs Site 5	-14.643	11.809	-1.24	0.215	1.000
Site 3 vs Site 6	2.169	11.918	0.182	0.856	1.000
Site 4 vs Site 5	34.911	11.809	2.956	0.003	0.047
Site 4 vs Site 6	-18.099	11.918	-1.519	0.129	1.000
Site 5 vs Site 6	16.812	11.918	1.411	0.158	1.000

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C13: Post Hoc Tukey test for number of feeding buzzes for eastern red bats in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-60.26	10.653	-5.657	<.001	0.000
Site 1 vs Site 3	-53.546	10.653	-5.027	<.001	0.000
Site 1 vs Site 4	-39.242	10.653	-3.684	<.001	0.003
Site 1 vs Site 5	-66.063	10.653	-6.202	<.001	0.000
Site 1 vs Site 6	-54.222	10.749	-5.044	<.001	0.000
Site 2 vs Site 3	6.714	10.555	0.636	0.525	1.000
Site 2 vs Site 4	21.018	10.555	1.991	0.046	0.697
Site 2 vs Site 5	6.038	10.653	0.567	0.571	1.000
Site 2 vs Site 6	-5.804	10.555	-0.55	0.582	1.000
Site 3 vs Site 4	14.304	10.555	1.355	0.175	1.000
Site 3 vs Site 5	-12.518	10.555	-1.186	0.236	1.000
Site 3 vs Site 6	-0.677	10.653	-0.064	0.949	1.000
Site 4 vs Site 5	26.821	10.555	2.541	0.011	0.166
Site 4 vs Site 6	-14.98	10.653	-1.406	0.160	1.000
Site 5 vs Site 6	11.841	10.653	1.112	0.266	1.000

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C14: Post Hoc Tukey test for number of feeding buzzes for silver-haired bats in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig[a]
Site 1 vs Site 2	-49.475	9.337	-5.299	<.001	0.000
Site 1 vs Site 3	-54.118	9.337	-5.796	<.001	0.000
Site 1 vs Site 4	-37.153	9.337	-3.979	<.001	0.001
Site 1 vs Site 5	-57.296	9.337	-6.136	<.001	0.000
Site 1 vs Site 6	-36.222	9.422	-3.845	<.001	0.002
Site 2 vs Site 3	-4.643	9.252	-0.502	0.616	1.000
Site 2 vs Site 4	12.321	9.252	1.332	0.183	1.000
Site 2 vs Site 5	-7.821	9.252	-0.845	0.398	1.000
Site 2 vs Site 6	13.253	9.337	1.419	0.156	1.000
Site 3 vs Site 4	16.964	9.252	1.834	0.067	1.000
Site 3 vs Site 5	-3.179	9.252	-0.344	0.731	1.000
Site 3 vs Site 6	17.896	9.337	1.917	0.055	0.829
Site 4 vs Site 5	20.143	9.252	2.177	0.029	0.442
Site 4 vs Site 6	0.931	9.337	0.1	0.921	1.000
Site 5 vs Site 6	21.074	9.337	2.257	0.024	0.360

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C15: Post Hoc Tukey test for number of drinking buzzes for evening bats in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-86.979	11.764	-7.394	<.001	0.000
Site 1 vs Site 3	-95.443	11.764	-8.113	<.001	0.000
Site 1 vs Site 4	-31.265	11.764	-2.658	0.008	0.118
Site 1 vs Site 5	-101.997	11.764	-8.671	<.001	0.000
Site 1 vs Site 6	-87.056	11.87	-7.334	<.001	0.000
Site 2 vs Site 3	-8.464	11.656	-0.726	0.468	1.000
Site 2 vs Site 4	55.714	11.656	4.78	<.001	0.000
Site 2 vs Site 5	-15.018	11.656	-1.288	0.198	1.000
Site 2 vs Site 6	-0.077	11.764	-0.007	0.995	1.000
Site 3 vs Site 4	64.179	11.656	5.506	<.001	0.000
Site 3 vs Site 5	-6.554	11.656	-0.562	0.574	1.000
Site 3 vs Site 6	8.388	11.764	0.713	0.476	1.000
Site 4 vs Site 5	70.732	11.656	6.068	<.001	0.000
Site 4 vs Site 6	-55.791	11.764	-4.743	<.001	0.000
Site 5 vs Site 6	14.941	11.764	1.27	0.204	1.000

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

Table C16: Post Hoc Tukey test for number of drinking buzzes for evening bats in 1-hr surveys at 6 study sites in Fort Worth, Texas, USA

Treatments	Test statistic	Std. Error	Std. test Statistic	p-value	Adj. sig
Site 1 vs Site 2	-68.929	11.441	-6.024	<.001	0.000
Site 1 vs Site 3	-76.321	11.441	-6.671	<.001	0.000
Site 1 vs Site 4	-26.625	11.441	-2.327	0.020	0.299
Site 1 vs Site 5	-75.714	11.441	-6.618	<.001	0.000
Site 1 vs Site 6	-66.019	11.545	-5.718	<.001	0.000
Site 2 vs Site 3	-7.393	11.337	-0.652	0.514	1.000
Site 2 vs Site 4	42.304	11.337	3.731	<.001	0.003
Site 2 vs Site 5	-6.786	11.337	-0.599	0.549	1.000
Site 2 vs Site 6	2.91	11.441	0.254	0.799	1.000
Site 3 vs Site 4	49.696	11.337	4.384	<.001	0.000
Site 3 vs Site 5	0.607	11.337	0.054	0.957	1.000
Site 3 vs Site 6	10.303	11.441	0.9	0.368	1.000
Site 4 vs Site 5	49.089	11.337	4.33	<.001	0.000
Site 4 vs Site 6	-39.394	11.441	-3.443	<.001	0.009
Site 5 vs Site 6	9.696	11.441	0.847	0.397	1.000

*a Significance values have been adjusted by the Bonferroni correction for multiple tests.

References

ABBOTT, I. M., D. P. SLEEMAN, AND S. HARRISON. 2009. Bat activity affected by sewage effluent in Irish rivers. Biological conservation 142:2904-2914.

AGPALO, E. 2020. Improving Urban Habitats for Bats: What Makes a Bat-Friendly Residential Swimming Pool? , ProQuest Dissertations Publishing.

AHLERS, A. A., E. J. HESKE, AND R. L. SCHOOLEY. 2016. Prey distribution, potential landscape supplementation, and urbanization affect occupancy dynamics of American mink in streams. Landscape Ecology 31:1601-1613.

AINSWORTH, A. AND D. R. DRAKE. 2020. Classifying Hawaiian plant species along a habitat generalist-specialist continuum: Implications for species conservation under climate change. Plos One 15.

ALFONSO, S., B. SADOUL, X. COUSIN, AND M.-L. BEGOUT. 2020. Spatial distribution and activity patterns as welfare indicators in response to water quality changes in European sea bass, *Dicentrarchus labrax*. Applied Animal Behaviour Science 226.

ALTRINGHAM, J. D. 2011. Chapter 2 Flight. Pp. 37–60 in Bats: From Evolution to Conservation.

AMMERMAN, L., C. HICE, AND D. SCHMIDLY. 2012. Bats of Texas, Texas A&M University press.

ARAUJO, A., D. S. DOS SANTOS, F. LINS-DE-BARROS, AND S. D. HACON. 2021. Linking ecosystem services and human health in coastal urban planning by DPSIWR framework. Ocean & Coastal Management 210.

ARONSON, M. F. J., et al. 2017. Biodiversity in the city: key challenges for urban green space management. Frontiers in Ecology and the Environment 15:189-196.

AVOLIO, M., et al. 2020. Urban plant diversity in Los Angeles, California: Species and functional type turnover in cultivated landscapes. Plants People Planet 2:144-156.

BADOLA, R., S. A. HUSSAIN, P. DOBRIYAL, AND S. BARTHWAL. 2015. Assessing the effectiveness of policies in sustaining and promoting ecosystem services in the Indian Himalayas. International Journal of Biodiversity Science Ecosystem Services & Management 11:216-224.

BANASZAK-CIBICKA, W. 2014. Are urban areas suitable for thermophilic and xerothermic bee species (Hymenoptera: Apoidea: Apiformes)? Apidologie 45:145-155.

BARBAREE, B. A., et al. 2018. Dynamic surface water distributions influence wetland connectivity within a highly modified interior landscape. Landscape Ecology 33:829-844.

BARNETT, J. M., P. F. HUDAK, AND G. O. DICK. 2016. Removing Chinese Privet (Ligustrum sinense Lour.) and Establishing Native Plants in a Flooded Riparian Corridor in North-Central Texas, USA. Environmental quality management 26:121-129.

BARRE, K., et al. 2021. Artificial light may change flight patterns of bats near bridges along urban waterways. Animal Conservation 24:259-267.

BASSETT, C. G., R. GILPIN, AND K. DONOHUE. 2022. Lessons Learned from Developing Best Management Practices for Urban Tree Care and Wildlife. Arboriculture & Urban Forestry 48:1-8.

BASTILLE-ROUSSEAU, G. AND G. WITTEMYER. 2021. Characterizing the landscape of movement to identify critical wildlife habitat and corridors. Conservation Biology 35:346-359.

BECKER, N. AND I. G. PINHEIRO. 2019. Potentiality of permeable pavements in the improvement of surface runoff quality: a review. Urbe-Revista Brasileira De Gestao Urbana 11.

BENNETT, V. J. AND A. A. ZURCHER. 2013. When corridors collide: Road-related disturbance in commuting bats. Journal of Wildlife Management 77:93-101.

BERRY, B. S., K. MAGORI, A. C. PEROFSKY, D. E. STALLKNECHT, AND A. W. PARK. 2013. Wetland cover dynamics drive hemorrhagic disease patterns in white-tailed deer in the united states. Journal of Wildlife Diseases 49:501-509.

BIENZ, C. 2016. Surface texture discrimination by bats: Implications for reducing bat mortality at wind turbines. ProQuest Dissertations Publishing.

BISHOP, C. A., J. STRUGER, L. J. SHIROSE, L. DUNN, AND G. D. CAMPBELL. 2000. Contamination and wildlife communities in stormwater detention ponds in Guelph and the Greater Toronto Area, Ontario, 1997 and 1998. Part II - Contamination and biological effects of contamination. Water Quality Research Journal of Canada 35:437-474.

BOGAN, M. T., D. EPPEHIMER, H. HAMDHANI, AND K. HOLLIEN. 2020. If you build it, they will come: rapid colonization by dragonflies in a new effluent-dependent river reach. Peerj 8.

BONTHOUX, S., M. BRUN, F. DI PIETRO, S. GREULICH, AND S. BOUCHE-PILLON. 2014. How can wastelands promote biodiversity in cities? A review. Landscape and Urban Planning 132:79-88.

BOURG, A., F. ESCOBAR, I. MACGREGOR-FORS, AND C. E. MORENO. 2016. Got Dung? Resource Selection by Dung Beetles in Neotropical Forest Fragments and Cattle Pastures. Neotropical Entomology 45:490-498.

BRODERS, H. G., C. S. FINDLAY, AND L. G. ZHENG. 2004. Effects of clutter on echolocation call structure of Myotis septentrionalis and M-lucifugus. Journal of Mammalogy 85:273-281.

BROWN, J. S., M. SUTULA, C. STRANSKY, J. RUDOLPH, AND E. BYRON. 2010. Sediment Contaminant Chemistry and Toxicity of Freshwater Urban Wetlands in Southern California1. Journal of the American Water Resources Association 46:367-384.

BURGMEIER, N. G., S. D. UNGER, J. L. MEYER, T. M. SUTTON, AND R. N. WILLIAMS. 2011. Health and habitat quality assessment for the eastern hellbender (*Cryptobranchus alleganiensis alleganiensis*) in Indiana, USA. Journal of Wildlife Diseases 47:836-848.

BUSCAINO, G., et al. 2012. The underwater acoustic activities of the red swamp crayfish Procambarus clarkii. Journal of the Acoustical Society of America 132:1792-1798.

CALLAGHAN, C. T., D. E. BOWLER, AND H. M. PEREIRA. 2021. Thermal flexibility and a generalist life history promote urban affinity in butterflies. Global Change Biology 27:3532-3546.

CAMPOS-SILVA, L. A. AND A. J. PIRATELLI. 2021. Vegetation structure drives taxonomic diversity and functional traits of birds in urban private native forest fragments. Urban Ecosystems 24:375-390.

CHAMBERLAIN, D., C. REYNOLDS, A. AMAR, D. HENRY, E. CAPRIO, AND P. BATARY. 2020. Wealth, water and wildlife: Landscape aridity intensifies the urban luxury effect. Global Ecology and Biogeography 29:1595-1605.

CHANG, C.-R., M.-C. CHEN, AND M.-H. SU. 2021. Natural versus human drivers of plant diversity in urban parks and the anthropogenic species-area hypotheses. Landscape and urban planning 208.

CLEVENOT, L., C. CARRE, AND P. PECH. 2018. A Review of the Factors That Determine Whether Stormwater Ponds Are Ecological Traps And/or High-Quality Breeding Sites for Amphibians. Frontiers in Ecology and Evolution 6.

COFW. 2022a. Foster Park. https://www.fortworthtexas.gov/departments/parks/parks-and-trails/foster-park. Accessed 8/20/2022.

COFW. 2022b. Lake Como park. https://www.fortworthtexas.gov/departments/parks/parks-and-trails/lake-como. Accessed 8/20/2022.

COFW. 2022c. Oakmont Park. https://www.fortworthtexas.gov/departments/parks/parks-and-trails/oakmont-park. Accessed 8/20/22.

COFW. 2022d. Trinity Park. https://www.fortworthtexas.gov/departments/parks/parks-and-trails/trinity-park. Accessed 08/20/2022.

CONCEPCION, E. D., M. K. OBRIST, M. MORETTI, F. ALTERMATT, B. BAUR, AND M. P. NOBIS. 2016. Impacts of urban sprawl on species richness of plants, butterflies, gastropods and birds: not only built-up area matters. Urban Ecosystems 19:225-242.

CORTES, K. M. AND E. H. GILLAM. 2020. Assessing the use of rivers as migratory corridors for temperate bats. Journal of Mammalogy 101:448-454.

CORY-TOUSSAINT, D., P. J. TAYLOR, AND I. E. J. BARNHOORN. 2022. Non-invasive sampling of bats reflects their potential as ecological indicators of elemental exposure in a diamond mining area, northern Limpopo Province, South Africa. Environmental Science and Pollution Research 29:13647-13660.

COTNER, L. A. AND R. L. SCHOOLEY. 2011. Habitat Occupancy by Riparian Muskrats Reveals Tolerance to Urbanization and Invasive Vegetation. Journal of Wildlife Management 75:1637-1645.

COVACIU-MARCOV, S.-D. AND A.-S. CICORT-LUCACIU. 2009. Big and nonmethamorphic Triturus cristatus larvae from north-western Romania. Biharean Biologist 3:87-89.

CROFT-WHITE, M. V., M. CVETKOVIC, D. ROKITNICKI-WOJCIK, J. D. MIDWOOD, AND G. P. GRABAS. 2017. A shoreline divided: Twelve-year water quality and land cover trends in Lake Ontario coastal wetlands. Journal of Great Lakes Research 43:1005-1015.

CUNTO, G. C. AND E. BERNARD. 2012. Neotropical bats as indicators of environmental disturbance: what is the emerging message? Acta Chiropterologica 14:143-151.

DANIELS, S. E., R. E. FANELLI, A. GILBERT, AND S. BENSON-AMRAM. 2019. Behavioral flexibility of a generalist carnivore. Animal Cognition 22:387-396.

DE JONG, G. D., S. P. CANTON, J. S. LYNCH, AND M. MURPHY. 2015. Aquatic invertebrate and vertebrate communities of ephemeral stream ecosystems in the arid southwestern United States. Southwestern Naturalist 60:349-359.

DE LABRA-HERNANDEZ, M. A. AND K. RENTON. 2019. Learning-by-consequence foraging model of the Northern Mealy Amazon in a modified landscape of tropical moist forest. Journal of Ornithology 160:497-507.

DENNIS, R. L. H., L. DAPPORTO, J. W. DOVER, AND T. G. SHREEVE. 2013. Corridors and barriers in biodiversity conservation: a novel resource-based habitat perspective for butterflies. Biodiversity and Conservation 22:2709-2734.

DENZINGER, A., M. TSCHAPKA, AND H.-U. SCHNITZLER. 2018. The role of echolocation strategies for niche differentiation in bats. Canadian Journal of Zoology 96:171-181.

DOMER, A., et al. 2021. Adverse effects of noise pollution on foraging and drinking behaviour of insectivorous desert bats. Mammalian Biology 101:497-501.

DONDINA, O., L. KATAOKA, V. ORIOLI, AND L. BANI. 2016. How to manage hedgerows as effective ecological corridors for mammals: A two-species approach. Agriculture Ecosystems & Environment 231:283-290.

DUCEY, M. J., K. M. JOHNSON, E. P. BELAIR, AND B. D. COOK. 2018. The Influence of Human Demography on Land Cover Change in the Great Lakes States, USA. Environmental Management 62:1089-1107.

DUKE, L. AND C. D. SOULSBURY. 2021. Human-wildlife interaction networks at urban blue spaces. Area 53:122-133.

ENTWISTLE, A. C., P. A. RACEY, AND J. R. SPEAKMAN. 1996. Habitat exploitation by a gleaning bat, Plecotus auritus. Philosophical Transactions of the Royal Society B-Biological Sciences 351:921-931.

EPA. 2003. Health Effects Support Document for Aldrin/Dieldrin https://www.epa.gov/sites/default/files/2014-09/documents/support_cc1_aldrin-dieldrin_healtheffects.pdf. Accessed 8/20/2022.

EPA. 2021. Types of drinking water contaminats. https://www.epa.gov/ccl/types-drinking-water-contaminants. Accessed 6/20/2022.

EPA. 2022a. Drinking Water Regulations and Contaminants. https://www.epa.gov/sdwa/drinking-water-regulations-and-contaminants. Accessed 8/20/2022.

EPA. 2022b. How's My Waterway? Informing the Conservation about Your Water. https://mywaterway.epa.gov/community/Lake%20Como-Clear%20Fork%20Trinity%20River/overview. Accessed 4/08/2022.

EPA. 2022c. Water Report: Clear Fork Trinity River Below Benbrook Lake. https://mywaterway.epa.gov/waterbody-report/TCEQMAIN/TX-0829_02/2022. Accessed 09/07/2022.

EPA. 2022d. Waterbody Report: Benbrook Lake. https://mywaterway.epa.gov/waterbody-report/TCEQMAIN/TX-0830_05/2022. Accessed 8/20/2022.

EPA. 2022e. Waterbody Report: Clear Fork Trinity River below Benbrook Lake TX-0829_03. https://mywaterway.epa.gov/waterbody-report/TCEQMAIN/TX-0829_03/2022. Accessed 8/20/2022.

EPA. 2022f. Waterbody Report: Lake Como. https://mywaterway.epa.gov/waterbody-report/TCEQMAIN/TX-0829A_01/2022. Accessed 9/07/2022.

EPA. 2022g. Waterbody Report: Rock Creek. https://mywaterway.epa.gov/community/120301020304/overview. Accessed 09/07/2022.

EPA. 2023. National Primary Drinking Water Regulations. https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations. Accessed 08/20/2022.

FERN, R. R., H. T. DAVIS, J. A. BAUMGARDT, M. L. MORRISON, AND T. A. CAMPBELL. 2018. Summer activity patterns of four resident south Texas bat species. Global Ecology and Conservation 16.

FIDINO, M. A., E. W. LEHRER, AND S. B. MAGLE. 2016. Habitat Dynamics of the Virginia Opossum in a Highly Urban Landscape. American Midland Naturalist 175:155-167.

FOXLEY, T., P. LINTOTT, AND E. STONE. 2022. What drives bat activity at field boundaries? Journal of environmental management 329:117029-117029.

FREITAS, S. R., L. R. TAMBOSI, N. P. GHILARDI-LOPES, AND M. D. WERNECK. 2020. Spatial and temporal variation of potential resource availability provided by street trees in southeastern Brazil. Urban Ecosystems 23:1051-1062.

FROMMHOLD, M., et al. 2019. Breeding habitat and nest-site selection by an obligatory "nest-cleptoparasite", the Amur Falcon Falco amurensis. Ecology and Evolution 9:14430-14441.

FURMANKIEWICZ, J. AND M. KUCHARSKA. 2009. Migration of bats along a large river valley in southwestern Poland. Journal of Mammalogy 90:1310-1317.

FWPR. 2014. Overton and Foster Park master plan. Final Draft. City of Fort Worth Parks and Community Services Department Park Master Plans:1-61.

GALLO, T., M. FIDINO, E. W. LEHRER, AND S. B. MAGLE. 2017. Mammal diversity and metacommunity dynamics in urban green spaces: implications for urban wildlife conservation. Ecological Applications 27:2330-2341.

GARNICK, S., J. DI STEFANO, M. A. ELGAR, AND G. COULSON. 2016. Ecological specialisation in habitat selection within a macropodid herbivore guild. Oecologia 180:823-832.

GBOGBO, F., L. H. HOLBECH, AND R. O. ERIOM. 2016. Usage pattern of artificial water sources by birds on a rapidly. expanding university campus in Ghana. Malimbus 38:1-9.

GODFREY, R. 1943. The South African cliff swallow (Petrochelidon spilodera Sund.). Ostrich 13:219-226.

GONDWE, M. J., M. MURRAY-HUDSON, N. M. MAZRUI, O. MOSES, E. MOSIMANYANA, AND O. MOGOBE. 2021. A review of the limnology of the Okavango Delta, Botswana. African Journal of Aquatic Science 46:251-273.

GPRC. 2013. Fort Worth Prairie Park & Rock Creek/Lake Benbrook Complex: A Regional Grassland Park for all of North Texas; A National Epicenter for Ecological Health. https://gprc.org/wp-content/uploads/2013/05/White-Paper-FWPP-2013-Email.pdf. Accessed 09/07 2022.

GRINDAL, S. D. 1999. Habitat use by bats, Myotis spp., in western Newfoundland. Canadian Field-Naturalist 113:258-263.

HADLEY, A. S., S. J. K. FREY, W. D. ROBINSON, W. J. KRESS, AND M. G. BETTS. 2014. Tropical forest fragmentation limits pollination of a keystone under story herb. Ecology 95:2202-2212.

HAIDER, H., A. R. GHUMMAN, I. S. AL-SALAMAH, Y. GHAZAW, AND R. H. ABDEL-MAGUID. 2019. Sustainability Evaluation of Rainwater Harvesting-Based Flood Risk Management Strategies: A Multilevel Decision-Making Framework for Arid Environments. Arabian Journal for Science and Engineering 44:8465-8488.

HAJEK, O. L. AND A. K. KNAPP. 2022. Shifting seasonal patterns of water availability: ecosystem responses to an unappreciated dimension of climate change. New Phytologist 233:119-125.

HALL, E. M. 2020. Home range expansion by evening bats (Nycticeius humeralis) in an urban environment. Texas Christian University, Fort Worth, Texas.

HALL, E. M. AND V. J. BENNETT. 2021. Seasonal variation in home range size of evening bats (Nycticeius humeralis) in an urban environment. Journal of Mammalogy 102:1497-1506.

HALL, L. K., C. T. LAMBERT, R. T. LARSEN, R. N. KNIGHT, AND B. R. MCMILLAN. 2016a. Will climate change leave some desert bat species thirstier than others? Biological Conservation 201:284-292.

HALL, L. K., R. T. LARSEN, M. D. WESTOVER, C. C. DAY, R. N. KNIGHT, AND B. R. MCMILLAN. 2016b. Influence of exotic horses on the use of water by communities of native wildlife in a semi-arid environment. Journal of Arid Environments 127:100-105.

HAMBLIN, A. L., E. YOUNGSTEADT, AND S. D. FRANK. 2018. Wild bee abundance declines with urban warming, regardless of floral density. Urban Ecosystems 21:419-428.

HANSEN, C. P., A. W. PARSONS, R. KAYS, AND J. J. MILLSPAUGH. 2020. Does Use of Backyard Resources Explain the Abundance of Urban Wildlife? Frontiers in Ecology and Evolution 8.

HARLAN, T., R. XU, AND J. HE. 2021. Is small hydropower beautiful? Social impacts of river fragmentation in China's Red River Basin. Ambio 50:436-447.

HINGRAT, Y., F. YSNEL, M. SAINT JALME, J. LE CUZIAT, P. M. BERANGER, AND F. LACROIX. 2007. Assessing habitat and resource availability for an endangered desert bird species in eastern Morocco: the Houbara Bustard. Biodiversity and Conservation 16:597-620.

HOELZINGER, J. 2014. Collared Dove Streptopelia decaocto brooding in the rain gutter of a house in Neckargroningen (Rural District of Ludwigsburg) - observations at an extreme nesting site. Ornithologische Jahreshefte fuer Baden-Wuerttemberg 30:123-124.

HOFMAN, M. P. G., M. W. HAYWARD, M. J. KELLY, AND N. BALKENHOL. 2018. Enhancing conservation network design with graph-theory and a measure of protected area effectiveness: Refining wildlife corridors in Belize, Central America. Landscape and Urban Planning 178:51-59.

HOLZER, K. A., R. P. BAYERS, N. THIEN TAO, AND S. P. LAWLER. 2017. Habitat value of cities and rice paddies for amphibians in rapidly urbanizing Vietnam. Journal of Urban Ecology 3:juw007-Article No.: juw007.

HULGARD, K. AND J. M. RATCLIFFE. 2016. Sonar sound groups and increased terminal buzz duration reflect task complexity in hunting bats. Scientific Reports 6.

HUNTSINGER, L., et al. 2017. Save water or save wildlife? Water use and conservation in the central Sierran foothill oak woodlands of California, USA. Ecology and Society 22.

HUYSMAN, A. E. AND M. D. JOHNSON. 2021. Habitat selection by a predator of rodent pests is resilient to wildfire in a vineyard agroecosystem. Ecology and Evolution 11:18216-18228.

HUZZEN, B. E., A. M. HALE, AND V. J. BENNETT. 2020. An effective survey method for studying volant species activity and behavior at tall structures. Peerj 8.

HYSENI, C., J. HEINO, L. M. BINI, U. BJELKE, AND F. JOHANSSON. 2021. The importance of blue and green landscape connectivity for biodiversity in urban ponds. Basic and Applied Ecology 57:129-145.

INATURALIST. 2022. Lake Como Park, Fort Worth, Texas US, Tx open Space. https://www.inaturalist.org/places/lake-como-park. Accessed 10/22/2022.

JIM, C. Y. 2004. Green-space preservation and allocation for sustainable greening of compact cities. Cities 21:311-320.

KALCOUNIS-RUEPPELL, M. C., V. H. PAYNE, S. R. HUFF, AND A. L. BOYKO. 2007. Effects of wastewater treatment plant effluent on bat foraging ecology in an urban stream system. Biological Conservation 138:120-130.

KLOEPPER, L. N., A. M. SIMMONS, AND J. A. SIMMONS. 2019. Echolocation while drinking: Pulse-timing strategies by high- and low-frequency FM bats. Plos One 14.

KORINE, C., A. M. ADAMS, U. SHAMIR, AND A. GROSS. 2015. Effect of water quality on species richness and activity of desert-dwelling bats. Mammalian Biology 80:185-190.

KOUTNIK, V. S., et al. 2021. Distribution of microplastics in soil and freshwater environments: Global analysis and framework for transport modeling. Environmental Pollution 274.

KREJSA, D. M., S. K. DECKER, AND L. K. AMMERMAN. 2020. Noteworthy records of 14 bat species in Texas including the first record of *Leptonycteris yerbabuenae* and the second record of *Myotis occultus*. Occasional Papers Museum of Texas Tech University 368:1-10.

KUVLESKY, W. P. B., L. A. , B. M. C. BALLARD, T. A. HEWITT, D. G. DEYOUNG, C. A. HENKE, S. E. HERNADEZ, F. , AND F. C. BRYANT. 2013. Managing Populations. Pp. 299-320 in Wildlife Management and Conservation: Contemporary Principles and Practices (III PRKaJWC ed.), The John Hopkins University Press.

LADOUCEUR, E., et al. 2020. Reducing dispersal limitation via seed addition increases species richness but not above-ground biomass. Ecology Letters 23:1442-1450.

LARSEN-GRAY, A. L. AND C. LOEHLE. 2022. Relationship Between Riparian Buffers and Terrestrial Wildlife in the Eastern United States. Journal of Forestry 120:336-357.

LAVERTY, T. M. AND J. BERGER. 2020. Do bats seek clean water? A perspective on biodiversity from the Namib Desert. Biological Conservation 248.

LEHRER, E. W., T. GALLO, M. FIDINO, R. J. KILGOUR, P. J. WOLFF, AND S. B. MAGLE. 2021. Urban bat occupancy is highly influenced by noise and the location of water: Considerations for nature-based urban planning. Landscape and Urban Planning 210.

LEWANZIK, D., A. K. SUNDARAMURTHY, AND H. R. GOERLITZ. 2019. Insectivorous bats integrate social information about species identity, conspecific activity and prey abundance to estimate cost-benefit ratio of interactions. Journal of Animal Ecology 88:1462-1473.

LI, D., S. CHEN, H. LLOYD, S. ZHU, K. SHAN, AND Z. ZHANG. 2013. The importance of artificial habitats to migratory waterbirds within a natural/artificial wetland mosaic, Yellow River Delta, China. Bird Conservation International 23:184-198.

LI, D., et al. 2022. The Influence of Ecological Engineering on Waterbird Diversity in Different Habitats within the Xianghai Nature Reserve. Diversity-Basel 14.

LI, H. AND M. KALCOUNIS-RUEPPELL. 2018. Separating the effects of water quality and urbanization on temperate insectivorous bats at the landscape scale. Ecology and Evolution 8:667-678.

LITKE, D. W. 1999. Review of Phosphorus Control Measures in the United States and Their Effects on Water Quality.

LOPEZ-BAUCELLS, A., L. CASANOVA, X. PUIG-MONTSERRAT, A. ESPINAL, F. PARAMO, AND C. FLAQUER. 2017. Evaluating the use of Myotis daubentonii as an ecological indicator in Mediterranean riparian habitats. Ecological Indicators 74:19-27.

LUI, G. V. AND D. A. COOMES. 2016. Tropical nature reserves are losing their buffer zones, but leakage is not to blame. Environmental Research 147:580-589.

MACLAGAN, S. J., T. COATES, B. A. HRADSKY, R. BUTRYN, AND E. G. RITCHIE. 2020. Life in linear habitats: the movement ecology of an endangered mammal in a peri-urban landscape. Animal Conservation 23:260-272.

MAGLE, S. B., E. W. LEHRER, AND M. FIDINO. 2016. Urban mesopredator distribution: examining the relative effects of landscape and socioeconomic factors. Animal Conservation 19:163-175.

MAGLE, S. B., L. S. SIMONI, E. W. LEHRER, AND J. S. BROWN. 2014. Urban predator–prey association: coyote and deer distributions in the Chicago metropolitan area. Urban ecosystems 17:875-891.

MCALEXANDER, A. 2013. Evidence that bats perceive wind turbine surfaces to be water. ProQuest Dissertations Publishing.

MCCAULEY, L. A., D. G. JENKINS, AND P. F. QUINTANA-ASCENCIO. 2013. Isolated Wetland Loss and Degradation Over Two Decades in an Increasingly Urbanized Landscape. Wetlands 33:117-127.

MCGEE, J. 2022. The Use of Swimming Pools by Bats in a Game Reserve in South Africa. ProQuest Dissertations Publishing.

MCINTYRE, N. E., J. C. DRAKE, AND K. L. GRIFFIS-KYLE. 2016. A connectivity and wildlife management conflict in isolated desert waters. Journal of Wildlife Management 80:655-666.

MCPHERSON, S. C., M. BROWN, AND C. T. DOWNS. 2016a. Crowned eagle nest sites in an urban landscape: Requirements of a large eagle in the Durban Metropolitan Open Space System. Landscape and Urban Planning 146:43-50.

McPherson, S. C., M. Brown, and C. T. Downs. 2016b. Diet of the crowned eagle (Stephanoaetus coronatus) in an urban landscape: potential for human-wildlife conflict? Urban Ecosystems 19:383-396.

Mero, T. O., A. Zuljevic, K. Varga, and S. Lengyel. 2020. Wing-size related habitat choice of Great Reed Warbler females: the role of habitat quality and management. Avian Conservation and Ecology 15.

Michie, L. E., J. D. Thiem, J. A. Facey, C. A. Boys, D. A. Crook, and S. M. Mitrovic. 2020. Effects of suboptimal temperatures on larval and juvenile development and otolith morphology in three freshwater fishes: implications for cold water pollution in rivers. Environmental Biology of Fishes 103:1527-1540.

Morrison, J. L., I. G. W. Gottlieb, and K. E. Pias. 2016. Spatial distribution and the value of green spaces for urban red-tailed hawks. Urban Ecosystems 19:1373-1388.

Murray, M. H., et al. 2021. Site Fidelity is Associated with Food Provisioning and Salmonella in an Urban Wading Bird. Ecohealth 18:345-358.

Naidoo, R., et al. 2020. Mapping and assessing the impact of small-scale ephemeral water sources on wildlife in an African seasonal savannah. Ecological Applications 30.

Nesom, G. 2013. Trees and Shrubs of Trinity Park, Fort Worth. http://www.guynesom.com/TrinityParkPlantsweb.htm. Accessed 4/07/2022.

Neumann, A., et al. 2021. A review of the current state of process-based and data-driven modelling: guidelines for Lake Erie managers and watershed modellers. Environmental Reviews 29:443-490.

Nyberg, D. and I. Lerner. 2000. Revitalization of ephemeral pools as frog breeding habitat in an Illinois forest preserve. Journal of the Iowa Academy of Science 107:187-190.

NYSTROM, G. S. AND V. J. BENNETT. 2019. The importance of residential swimming pools as an urban water source for bats. Journal of Mammalogy 100:394-400.

OGIE, R. I., P. PEREZ, K. T. WIN, AND K. MICHAEL. 2018. Managing hydrological infrastructure assets for improved flood control in coastal mega-cities of developing nations. Urban Climate 24:763-777.

OGUTU, J. O., et al. 2014. Large herbivore responses to surface water and land use in an East African savanna: implications for conservation and human-wildlife conflicts. Biodiversity and Conservation 23:573-596.

OKAMIYA, H. AND T. KUSANO. 2018. Evaluating Movement Patterns and Microhabitat Selection of the Japanese Common Toad (Bufo japonicus formosus) Using Fluorescent Powder Tracking. Zoological Science 35:153-160.

OLIVEIRA, J. M., A. L. F. DESTRO, M. B. FREITAS, AND L. L. OLIVEIRA. 2021. How do pesticides affect bats? - A brief review of recent publications. Brazilian journal of biology = Revista brasleira de biologia 81:499-507.

OOSTERBROEK, B., J. DE KRAKER, M. HUYNEN, AND P. MARTENS. 2016. Assessing ecosystem impacts on health: A tool review. Ecosystem Services 17:237-254.

OVALLE-RIVERA, N. A., et al. 2020. Chatacteristics of Artifical Water Sources and Its Relationship with the Use By Wildlife in Coahuila, Mexico. Agrociencia 54:825-842.

PAMURU, S. T., E. FORGIONE, K. CROFT, B. V. KJELLERUP, AND A. P. DAVIS. 2022. Chemical characterization of urban stormwater: Traditional and emerging contaminants. Science of the Total Environment 813.

PERKIN, J. S. AND T. H. BONNER. 2016. Historical Changes in Fish Assemblage Composition Following Water Quality Improvement in the Mainstem Trinity River of Texas. River Research and Applications 32:85-99.

PERKL, R., L. M. NORMAN, D. MITCHELL, M. FELLER, G. SMITH, AND N. R. WILSON. 2018. Urban growth and landscape connectivity threats assessment at Saguaro National Park, Arizona, USA. Journal of Land Use Science 13:102-117.

PERRON, M. A. C. AND F. R. PICK. 2020. Water quality effects on dragonfly and damselfly nymph communities: A comparison of urban and natural ponds. Environmental Pollution 263.

PETERSON, M. N., R. R. LOPEZ, E. J. LAURENT, P. A. FRANK, N. J. SILVY, AND J. G. LIU. 2005. Wildlife loss through domestication: The case of endangered key deer. Conservation Biology 19:939-944.

PETTITT, B. A., G. R. BOURNE, AND M. A. BEE. 2018. Predictors and benefits of microhabitat selection for offspring deposition in golden rocket frogs. Biotropica 50:919-928.

PITTS, S. L., B. D. HUGHES, AND I. MALI. 2017. Rattlesnake nuisance removals and urban expansion in Phoenix, Arizona. Western North American Naturalist 77:309-316.

PLUMMER, K. E., S. GILLINGS, AND G. M. SIRIWARDENA. 2020. Evaluating the potential for bird-habitat models to support biodiversity-friendly urban planning. Journal of Applied Ecology 57:1902-1914.

RACHWALD, A. 2019. Bats as indicators of the condition of the forest environment. Sylwan 163:228-236.

RATCLIFFE, J. M., J. W. DAWSON, M. B. FENTON, AND J. H. FULLARD. 2001. To glean or not to glean? Behavioural plasticity in two species of Myotis. Bat Research News 42:176-176.

RAZGOUR, O., C. KORINE, AND D. SALTZ. 2010. Pond characteristics as determinants of species diversity and community composition in desert bats. Animal Conservation 13:505-513.

REHMAN, S., N. K. TIWARY, AND A. J. URFI. 2021. Conservation monitoring of a polluted urban river: an occupancy modeling study of birds in the Yamuna of Delhi. Urban Ecosystems 24:1399-1411.

REID, K., K. SCHNEIDER, AND B. MCCONKEY. 2018. Components of Phosphorus Loss From Agricultural Landscapes, and How to Incorporate Them Into Risk Assessment Tools. Frontiers in Earth Science 6.

REIMER, J. P., E. F. BAERWALD, AND R. M. R. BARCLAY. 2018. Echolocation activity of migratory bats at a wind energy facility: testing the feeding-attraction hypothesis to explain fatalities. Journal of Mammalogy 99:1472-1477.

REYNA-HURTADO, R., E. ROJAS-FLORES, AND G. W. TANNER. 2009. Home range and habitat preferences of white-lipped peccaries (*Tayassu pecari*) in Calakmul, Campeche, Mexico. Journal of Mammalogy 90:1199-1209.

RICH, L. N., S. R. BEISSINGER, J. S. BRASHARES, AND B. J. FURNAS. 2019. Artificial water catchments influence wildlife distribution in the Mojave Desert. Journal of Wildlife Management 83:855-865.

ROBERTS, P. AND B. A. STEWART. 2018. Defining the 'generalist specialist' niche for Pleistocene Homo sapiens. Nature Human Behaviour 2:542-550.

RODRIGUEZ, J. D. AND F. SANCHEZ. 2022. Physical clutter affects the use of artificial ponds by the Lesser Bulldog Bat Noctilio albiventris (Chiroptera: Noctilionidae). Papeis Avulsos de Zoologia (Sao Paulo) 62:e202262019-Article No.: e202262019.

ROSEVELT, C., M. LOS HUERTOS, C. GARZA, AND H. M. NEVINS. 2013. Marine debris in central California: Quantifying type and abundance of beach litter in Monterey Bay, CA. Marine Pollution Bulletin 71:299-306.

ROUG, A., et al. 2020. Seasonal movements and habitat use of African buffalo in Ruaha National Park, Tanzania. Bmc Ecology 20.

RUSSO, D., et al. 2019. Effects of artificial illumination on drinking bats: a field test in forest and desert habitats. Animal Conservation 22:124-133.

Sabatier, R., K. Meyer, K. Wiegand, and Y. Clough. 2013. Non-linear effects of pesticide application on biodiversity-driven ecosystem services and disservices in a cacao agroecosystem: A modeling study. Basic and Applied Ecology 14:115-125.

Saifur, S. and C. M. Gardner. 2021. Loading, transport, and treatment of emerging chemical and biological contaminants of concern in stormwater. Water Science and Technology 83:2863-2885.

Salvarina, I. 2016. Bats and aquatic habitats: a review of habitat use and anthropogenic impacts. Mammal Review 46:131-143.

Sambito, M., A. Severino, G. Freni, and L. Neduzha. 2021. A Systematic Review of the Hydrological, Environmental and Durability Performance of Permeable Pavement Systems. Sustainability 13.

Sanchez-Montoya, M. M., M. Moleon, J. A. Sanchez-Zapata, and D. Escoriza. 2017. The biota of intermittent and ephemeral rivers: amphibians, reptiles, birds, and mammals 299-322.

Sandifer, P. A., A. E. Sutton-Grier, and B. P. Ward. 2015. Exploring connections among nature, biodiversity, ecosystem services, and human health and well-being: Opportunities to enhance health and biodiversity conservation. Ecosystem Services 12:1-15.

Santini, L., M. Gonzalez-Suarez, D. Russo, A. Gonzalez-Voyer, A. von Hardenberg, and L. Ancillotto. 2019. One strategy does not fit all: determinants of urban adaptation in mammals. Ecology Letters 22:365-376.

Schoch, N., et al. 2014. Wildlife Criterion Value for the Common Loon (Gavia immer) in the Adirondack Park, New York, USA. Waterbirds 37:76-84.

Schulwitz, S. E., M. M. Chumchal, and J. A. Johnson. 2015. Mercury Concentrations in Birds from Two Atmospherically Contaminated Sites in North Texas, USA. Archives of Environmental Contamination and Toxicology 69:390-398.

Schulz, C. and A. A. R. Ioris. 2017. The Paradox of Water Abundance in Mato Grosso, Brazil. Sustainability 9.

Scott, S. J., G. McLaren, G. Jones, and S. Harris. 2010. The impact of riparian habitat quality on the foraging and activity of pipistrelle bats (<i>Pipistrellus</i> spp.). Journal of Zoology 280:371-378.

Segre, P. S., R. Dakin, V. B. Zordan, M. H. Dickinson, A. D. Straw, and D. L. Altshuler. 2015. Burst muscle performance predicts the speed, acceleration, and turning performance of Anna's hummingbirds. Elife 4.

Selebatso, M., G. Maude, and R. W. S. Fynn. 2018. Assessment of quality of water provided for wildlife in the Central Kalahari Game Reserve, Botswana. Physics and Chemistry of the Earth 105:191-195.

Shaw, R. F., P. J. Johnson, D. W. Macdonald, and R. E. Feber. 2015. Enhancing the Biodiversity of Ditches in Intensively Managed UK Farmland. Plos One 10.

Shute, K. E., S. C. Loeb, and D. S. Jachowski. 2021. Seasonal Shifts in Nocturnal Habitat Use by Coastal Bat Species. Journal of Wildlife Management 85:964-978.

Shutt, J. D. and A. C. Lees. 2021. Killing with kindness: Does widespread generalised provisioning of wildlife help or hinder biodiversity conservation efforts? Biological Conservation 261.

Sievers, M., K. M. Parris, S. E. Swearer, and R. Hale. 2018. Stormwater wetlands can function as ecological traps for urban frogs. Ecological Applications 28:1106-1115.

SMITH, K. 2019. Assessing the Potential Impacts of Radio Transmitters on Bat Flight and Behavior in a Controlled Environment. ProQuest Dissertations Publishing.

SOL, D., I. BARTOMEUS, C. GONZALEZ-LAGOS, AND S. PAVOINE. 2017. Urbanisation and the loss of phylogenetic diversity in birds. Ecology Letters 20:721-729.

STIDSHOLT, L., R. MUELLER, K. BEEDHOLM, H. MA, M. JOHNSON, AND P. T. MADSEN. 2020. Energy compensation and received echo level dynamics in constant-frequency bats during active target approaches. Journal of Experimental Biology 223.

STRAKA, T. M., P. E. LENTINI, L. F. LUMSDEN, B. A. WINTLE, AND R. VAN DER REE. 2016. Urban bat communities are affected by wetland size, quality, and pollution levels. Ecology and Evolution 6:4761-4774.

TANG, W., A. BIGLARI, R. EBARB, T. PICKETT, S. SMALLIDGE, AND M. WARD. 2021. A Smart Sensing System of Water Quality and Intake Monitoring for Livestock and Wild Animals. Sensors 21.

TCEQ. 2013. Thirteen total maximum daily loads for indicator bacteria in the Lower West Fork Trinity River Watershed. www.tceq.texas.gov/downloads/water-quality/tmdl/greater-trinity-recreational-66/66d-lower-west-fork-trinity-river-tmdl-adopted.pdf. Accessed 9/21/2022.

TCEQ. 2022. Rocky Creek and Salado Creek: Investigating Aquatic Life Conditions www.tceq.texas.gov/waterquality/tmdl/33-middlebrazos.html. Accessed 09/21/2022.

TCU. 2015. Future for pond in Greek is uncertain. https://tcu360.com/2015/03/06/33890future-pond-greek-uncertain/. Accessed 10/25/2022.

TEITELBAUM, C. S., S. ALTIZER, AND R. J. HALL. 2022. Habitat Specialization by Wildlife Reduces Pathogen Spread in Urbanizing Landscapes. American Naturalist.

THEODOROU, P., et al. 2020. Urban areas as hotspots for bees and pollination but not a panacea for all insects. Nature Communications 11.

THOMAZ, S. M. 2021. Ecosystem services provided by freshwater macrophytes. Hydrobiologia 850:2757-2777.

TIEGS, L. A. 2017. Wintering whooping crane behavior and habitat quality at the Aransas National Wildlife Refuge on the Texas Gulf Coast.

TIGAS, L. A., D. H. VAN VUREN, AND R. M. SAUVAJOT. 2002. Behavioral responses of bobcats and coyotes to habitat fragmentation and corridors in an urban environment. Biological Conservation 108:299-306.

TPW. 2022. Stocking History for Como (Fort Worth). https://tpwd.texas.gov/fishboat/fish/action/stock_bywater.php?WB_code=0188. Accessed 10/25/2022 2022.

TPWD. 2022. Stocking History for trinity River. https://tpwd.texas.gov/fishboat/fish/action/stock_bywater.php?WB_code=1592. Accessed 10/23/2022 2022.

TUNEU-CORRAL, C., X. PUIG-MONTSERRAT, C. FLAQUER, M. MAS, I. BUDINSKI, AND A. LOPEZ-BAUCELLS. 2020. Ecological indices in long-term acoustic bat surveys for assessing and monitoring bats' responses to climatic and land-cover changes. Ecological Indicators 110.

TUTTLE, S. R., C. L. CHAMBERS, AND T. C. THEIMER. 2006. Potential Effects of Livestock Water-Trough Modifications on Bats in Northern Arizona. Wildlife Society bulletin 34:602-608.

USCOE. 2022. Corps of Engineers Parks. https://www.swf-wc.usace.army.mil/benbrook/Recreation/Parks/Corpsparks.shtml. Accessed 10/27/2022.

USGS. 2022. Science in your watershed. https://water.usgs.gov/wsc/a_api/wbd/subwatershed12/120301020307.html. Accessed 09/04/2022.

van der Ree, R., S. Cesarini, P. Sunnucks, J. L. Moore, and A. Taylor. 2010. Large Gaps in Canopy Reduce Road Crossing by a Gliding Mammal. Ecology and Society 15.

Velamazan, M., R. Perea, and M. N. Bugalho. 2020. Ungulates and ecosystem services in Mediterranean woody systems: A semi-quantitative review. Journal for Nature Conservation 55.

Volpe, N. L., W. D. Robinson, S. J. K. Frey, A. S. Hadley, and M. G. Betts. 2016. Tropical Forest Fragmentation Limits Movements, but Not Occurrence of a Generalist Pollinator Species. Plos One 11.

Wang, J. W., C. H. Poh, C. Y. T. Tan, V. N. Lee, A. Jain, and E. L. Webb. 2017. Building biodiversity: drivers of bird and butterfly diversity on tropical urban roof gardens. Ecosphere 8.

Wang, Y.-K., C.-C. Li, K.-Y. Huang, and C.-C. Tsai. 2020. Can Aquaculture Ponds Be Managed as Foraging Habitats for Overwintering Water Birds? An Experimental Approach. Sustainability 12.

Warren, R. D., D. A. Waters, J. D. Altringham, and D. J. Bullock. 2000. The distribution of Daubenton's bats (Myotis daubentonii) and pipistrelle bats (Pipistrellus pipistrellus) (Vespertilionidae) in relation to small-scale variation in riverine habitat. Biological conservation 92:85-91.

Weinberger, I. C., S. Muff, A. de Jongh, A. Kranz, and F. Bontadina. 2016. Flexible habitat selection paves the way for a recovery of otter populations in the European Alps. Biological Conservation 199:88-95.

WESTER, P. 2014. Feeding on the wing: hovering in nectar-drinking Old World birds - more common than expected. Emu-Austral Ornithology 114:171-183.

WILSON, W. A., M. WIPFLER, AND J. STEVENS. 2021. How Surface Water Management Can Benefit Fish Conservation in Urban Streams. Journal of Fish and Wildlife Management 12:383-394.

WURTH, A. M., E. H. ELLINGTON, AND S. D. GEHRT. 2020. Golf Courses as Potential Habitat for Urban Coyotes. Wildlife Society Bulletin 44:333-341.

XUE, J. Y., Q. R. WANG, AND M. H. ZHANG. 2022. A review of non-point source water pollution modeling for the urban-rural transitional areas of China: Research status and prospect. Science of the Total Environment 826.

YAKOVLEV, A. S., et al. 2015. Regulatory and legislative aspects of the ecological evaluation and control of soil degradation in Russia on the basis of the assessment of soil ecological functions. Eurasian Soil Science 48:991-996.

YOUNG, J. K., J. GOLLA, J. P. DRAPER, D. BROMAN, T. BLANKENSHIP, AND R. HEILBRUN. 2019. Space Use and Movement of Urban Bobcats. Animals 9.

YOW, D. M. AND M. RUGGLESS. 2019. The Urban Heat Island at a Regional University in Kentucky, USA. Journal of the Kentucky Academy of Science 80:27-37.

YUAN, B. D., C. B. TANG, Z. WANG, C. H. LU, AND Y. ZOU. 2013. Density, behavior and habitat selection of red-bellied squirrels (Callosciurus erythraeus castaneoventris) in Longjiang riverside of Yizhou, Guangxi, China. Asia Life Sciences 22:549-564.

ZAHN, A. AND S. MAIER. 1997. Hunting activity of bats at streams and ponds. Zeitschrift für Säugetierkunde 62:1-11.

ZHANG, J. D., et al. 2017. Modeling activity patterns of wildlife using time-series analysis. Ecology and Evolution 7:2575-2584.

ZHANG, M. M., S. WANG, B. J. FU, G. Y. GAO, AND Q. SHEN. 2018. Ecological effects and potential risks of the water diversion project in the Heihe River Basin. Science of the Total Environment 619:794-803.

ZHANG, X., T. J. LARK, C. M. CLARK, Y. YUAN, AND S. D. LEDUC. 2021. Grassland-to-cropland conversion increased soil, nutrient, and carbon losses in the US Midwest between 2008 and 2016. Environmental Research Letters 16.

ZSEBOK, S., F. KROLL, M. HEINRICH, D. GENZEL, B. M. SIEMERS, AND L. WIEGREBE. 2013. Trawling bats exploit an echo-acoustic ground effect. Frontiers in Physiology 4.

Vita

Personal Background

I'Yanna Scott
Crown Point, Indiana
Daughter of Dillard Scott and Trinette Scott-Jennings

Education

2017	Diploma, Merrillville High School, Merrillville, IN
2021	Bachelor of Science, Wildlife & Fisheries and Aquatic Science, Minor Statistics, Purdue University, West Lafayette, IN

Experience

2021-Present	Teaching Assistantship, Texas Christian University
2020 (May-Jun)	Research Technician, Purdue University
2018-2019 (Aug-Jul)	Statistic LLC, Purdue University

Presentations

Scott, I. and V. Bennett 2023. Enhancing urban areas for bat communities: Water quality influences water availability. The Wildlife Society. Louisville, KY.

Scott, I. and V. Bennett. 2023. Enhancing urban areas for bat communities: Water quality influences water availability. Michael and Sally McCracken Student Research Symposium. Texas Christian University. Fort Worth, TX.

Taua, N., I. Scott, L. D'Acunto, P. Zollner, and C. Gerdes. 2019. Evaluating the contribution acoustic monitors have in predicting bat mist-netting success. Joint Statistical Meeting. Denver, CO.

Taua, N., I. Scott, L. D'Acunto, P. Zollner, and C. Gerdes. 2019. Evaluating the contribution acoustic monitors have in predicting bat mist-netting success. 11th Midwest Bat Working Group. Chicago, IL.

Awards

2023	\$500 – TWS student Travel grant award to present at the 30th The Wildlife Society's conference, The Wildlife Society
2022	\$400 – Graduate student travel grant award to present at the 30th The Wildlife Society's conference, Texas Christian University
2022	\$1,972 – SERC graduate research grant award, College of Science and Engineering, Texas Christian University
2022	\$3,000 – ENSC graduate research grant award, Department of Environmental Sciences, Texas Christian University

Abstract

RESTORING WATER QUALITY TO IMPROVE URBAN PARKS AND ENHANCE BAT COMMUNITIES

By I'Yanna Scott, M.S., 2023
School of Science and Engineering
Texas Christian University

Thesis Advisor: Dr. Victoria J. Bennett, Associate Professor and Graduate Director

Committee Members: Dr. Gehendra Kharel, Assistant Professor

Dr. Brendan Lavy, Assistant Professor

For urban areas to be suitable for wildlife, water resources need to be available and accessible and water quality can dictate whether wildlife utilizes a resource or not. In urban environments, water has higher levels of contamination that can negatively impact wildlife. Moreover, the impact of water quality on species' presence has been explored, the direct responses of wildlife to water quality have not. To assess whether water quality influences the direct use of water by bats in an urban environment in Texas, we used thermal cameras and acoustic monitoring to determine whether water quality had a discernible influence on bat foraging and drinking activities. We found that contaminated water sources had >80% less bat activity, less foraging, and less drinking, as well as a 60% reduction in species diversity. Our study confirms that not all water sources present in an urban environment are suitable and, therefore, available to bats.