EFFECTS OF A HIGHLY MODIFIED LANDSCAPE ON DIVERSITY OF ANURAN COMMUNITIES IN NORTHWESTERN OHIO

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ABSTRACT

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As human-modified landscape and climate changes proliferate, maintaining biodiversity and understanding the function and quality of available habitat is imperative. Anurans (frogs/toads) can be indicator species of habitat quality and ecosystem productivity, due to their permeable skin, small body size and ectothermy. We explored the relationship between Anurans and habitat quality by assessing the effects of spatial and temporal heterogeneity on the presence of Anurans. Across the Toledo Metropolitan Area (TMA), including the biodiversity hotspot Oak Openings Region (OOR), we surveyed across three years, 67 different wetland sites (N=1800). There was a difference in community assemblage between rural and suburban/urban habitats driven by factors related to human-modification (impervious surface), composition (landcover type) and productivity (e.g., NDVI). Areas with more impervious surface, lower amounts of swamp forest, and lower NDVI had fewer species. The differences in spatial structure but lack of differences in temporal variables among sites suggest spatial factors dominated. We also developed spatial models for predicting species richness across the region to evaluate spatial variables driving community composition and ecosystem productivity. The amount of cropland best predicted species richness, followed by amount of swamp forest. Among individual species, the most important variables differed; cropland (*Acris blanchardi, Lithobates catesbeianus, Anaxyrus americanus, Anaxyrus fowleri* and *Hyla versicolor*), floodplain forest (*Lithobates clamitans*), wet prairie (*Lithobates pipiens*), and swamp forest (*Pseudacris crucifer, Pseudacris triseriata, Lithobates sylvaticus*) were leading influences. Finally, we surveyed 304 local residents to assess their views on topics from support of new parks/preserves to fees to utilize parks, before a 25-minute presentation on Anurans, and resurveying them. There was strong support for many conservation-oriented questions, but lowest support for those that involved money. The presentation significantly increased support for most conservation-oriented questions. This survey can serve managers exploring the expansion of protected areas and public funding. Our research demonstrates the value of non-invasive frog call surveys to assess ecosystem productivity and species richness, while also evaluating potential of expanding local conservation. This approach can be applied anywhere with sufficient environmental data/willing respondents to address questions of ecological interest and cover a wide swath of approaches to improving conservation efforts.

Eleven years ago, I wasn’t even sure college was for me. Today, I can say I’ve gone farther than I ever expected.

I dedicate this to every person in my life who has pushed me to grow more, work harder, and challenge myself more than I ever thought I was capable of.

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INTRODUCTION

As human-mediated land use change has proliferated, studies have shown that these habitat alterations are the greatest threat to wildlife biodiversity in the 21st century (Davison et al., 2021; Segan et al., 2016; Wilson, 1991). These habitat changes, often associated with heavy landscape alterations and changes in landscape function, can not only cause outright mortality, but also can make habitat unsuitable for breeding and population persistence (Almeida-Gomes & Rocha, 2015; Cushman, 2006). Lower genetic diversity, lower species diversity, and introduced species are also all potential effects of habitat alteration (Johnson et al., 2011; Mantyka-Pringle et al., 2011; Segan et al., 2016). Species such as Anurans (frogs and toads), however, have limited dispersal abilities, with many species moving only a few hundred meters over the course of their entire lifecycle (Hamer & McDonnell, 2008; Pittman et al., 2014).

Combined with this limited dispersal, the amphibian taxonomic group (including Anurans) has been classified as the most endangered vertebrate taxon today (Catenazzi, 2015; da Silva et al., 2020). This classification is a result of their sensitivity to habitat change, water and air quality, climate, and disease, as well as limited dispersal ability (Catenazzi, 2015; Johnson et al. 2011). Factors such as loss of habitat cover, change in hydrology, aquatic nutrient and sediment loads, and complete habitat loss have all been found to negatively impact Anuran biodiversity (Gillespie, 2002; Rowland et al. 2006, VanAcker et al., 2019). These species are also chronically underrepresented in scientific research, despite being valuable indicator species for ecologists to understand a study system (Waddle, 2006; Willson & Dorcas, 2003). Rarely are amphibian-oriented studies conducted across a full geographical region, with consistent sampling effort, while also including the entire community of Anuran species considered in the study design (Johnson et al., 2011; Kiviat & Nagy, 2011; Sievers et al., 2019; Stevens et al., 2002;

Stevens & Paszkowski, 2004; Takahara et al., 2020). To address this deficiency, this study sought to evaluate the connection between Anuran species diversity and landscape habitat factors, and the potential influences of habitat restoration in our study area. This work was performed to assess wetland productivity via biotic processes in the Oak Openings Region and Toledo Metropolitan Area in northwestern Ohio, in order to inform both local and worldwide conservation of Anurans.

Urban ecology is a field that has come of age in the 20th and 21st centuries, growing as the global decline in biodiversity continues primarily through the avenues of landscape modification and global climate change (Davison et al., 2021; Mantyka-Pringle et al., 2012; Segan et al., 2016). The continued expansion of human influence results in land use change and landscape alteration, with the potential for declines in water and air quality (Deng & Mendelsohn, 2021; Hall et al., 1999; Qiu et al., 2019). Urban areas also expose remaining species to higher levels of ambient light and noise, as well as exposure to threats these species are unaccustomed to handling, such as cars, roads, higher nutrient loads/poor nutrient cycling, alterations in natural cover, and poor air quality (Gillespie, 2002; Schoeman, 2016; Sievers et al., 2019). Roadways that accompany anthropogenic land change also can serve as a barrier to dispersal for terrestrial species (Forman & Alexander, 1998; Garriga et al., 2012). The light, noise, and threat of mortality can hinder movement out of the city or towards resources, habitat, and breeding opportunities (Forman & Alexander, 1998; Garriga et al., 2012). Work by Trombulak & Frissell (2000) and Long et al. (2010) has shown that roads increase species mortality rates, while also lowering dispersal rates (Forman & Alexander, 1998). Road construction also creates habitat edge, which allows for changes in microclimate that may not be conducive to species found in that area (Forman & Alexander, 1998; Santana Marques, 2020). This can allow for the community to be dominated by disturbance-tolerant or invasive species (Forman & Alexander, 1998; Santana Marques, 2020). These disturbance tolerant species, such as the American bullfrog (*L. catesbeianus*), are often regarded as nuisance species, have been expanding their range, and even predate upon other amphibian species as part of their diet (Kats & Ferrer, 2003). This range expansion and increased predation opportunity also harbors the potential for abetting population decline or community imbalance.

The proliferation of nuisance species stemming from the increase in edge and marginal-quality wetlands, can result in the alteration of the community dynamic and cause species to come into contact that otherwise may not. (Rowe et al., 2019; Strauss et al., 2006). These wetlands can be of marginal quality for a variety of reasons, including systems being degraded by external pollutants and species removal, altered by change in hydrology or climate, or they can be lost altogether (Bedford, 1999; Lougheed et al., 2008; Meng et al., 2017). This effect on amphibian communities was emphasized in Calderon et al. (2019), where the authors found that, in relation to habitat degradation, general amphibian richness was negatively correlated with phosphate/nitrate concentrations, as well as total coliforms in the system, all of which can be related to highly modified landscapes. Furthermore, elevational depression allowing for the creation of forested wetlands can serve as sinks for sediments, nutrients, and metals (Faulkner, 2004). Marginal quality wetlands may also affect Anurans by forcing breeding adults to disperse, which may increase their energetic requirements or expose them to the previously mentioned threats of dispersal (Baguette et al., 2013; Cayuela et al., 2020; Tsianou & Kallimanis, 2020). Alternatively, if they stay in the marginal quality wetlands, breeding activities or tadpole growth/success may be adversely affected by water quality (Cayuela et al., 2020; Hailey et al., 2006; Wood & Richardson, 2009).

Landscape modification does have the potential to allow for species to improve access to resources due to the loss or removal of less disturbance-tolerant competitors, such as among frogs like the Northern leopard frog (*L. pipiens*) (Start et al., 2020). Roads can also offer a secondary heat source for ectothermic species that can be utilized during cooler nighttime periods (Trombulak & Frissell, 2000). However, due to the relatively short distances Anurans move between habitats, their small size, slow movement, and moisture concerns, Anuran species are unlikely to be the beneficiaries of these effects from landscape and roadway development. Vos & Chardon (1998) showed that Moor frog (*Rana arvalis*) occupancy is negatively affected by an increase in traffic volume, and Fahrig et al. (1995) showed that local abundance of frog and toads is inversely related to high-traffic roads, and those same roads exhibited higher incidence of roadkill. Factors related to this development, such as introduced species, amount of vegetation, aquatic conductivity, road density, and surrounding wetland area, were also found by Johnson et al. (2013) to contribute to a 60% lower richness of amphibians in urban wetlands as compared to agricultural or grassland area wetlands.

The combination of habitat loss and use change, pollution with pesticides/heavy metals, alteration of vegetation, development of waterways, combined with human management for fish species, has contributed to the degradation of wetlands across the globe (Davis & Froend, 1999; Gebresllassie et al., 2014; Kingsford et al., 2016; Pope 2008). Approximately 90% of Ohio wetlands have been altered or outright removed since permanent European settlement, and the heavy manufacturing, shipping, and farming presence in the Toledo Metropolitan Area have made this problem especially acute in the study area (Berube & Murray, 2018; Root & Martin, 2017; USDA, 2017). As much as 46.6% of land in the study area is urban and residential/mixed use landcover, and a further 29.9% compromises cropland (Berube & Murray, 2018; Root &

Martin, 2017; USDA, 2017). Moreover, wetland areas has been shown to make up 55% of new development areas in developing countries (Pauchard et al., 2005). Destruction of wetland sites in developed countries may have slowed, but not ended completely, as Kentula et al. (2004) found 6% of historical wetland sites were destroyed between 1982 and 1998 in residential Portland, Oregon. With the incredible loss of wetlands since European settlement, 6% of loss over 16 years is likely still high. Remaining wetlands also are becoming more degraded, through introduction of toxic substances, invasive species, increasing fragmentation; Anuran species have been found to decline with that degradation (Bedford, 1999; Lougheed et al., 2008; Meng et al., 2017).

Zhang et al. (2016) also found that increasing urbanization decreased the microclimate regulation function in wetlands, a factor that is of particular importance to ectothermic taxa, e.g., Anurans. Additionally, Hamer & McDonnell (2008) identified nine key factors (including amount/type of vegetation in aquatic/terrestrial habitat, hydroperiod, exotic predators/competitors, water quality, human disturbance, habitat loss, fragmentation, and creation) that influence amphibian population dynamics, such as the wood frog (*L. sylvaticus*) under highly modified land use conditions. The loss of isolated populations also exacerbates the fragmentation of populations found in areas undergoing land use change, and research by Braaker et al. (2014) has found that connectivity of habitat patches was essential to terrestrial species movement.

Island biogeography theory, developed in 1967 by Macarthur and Wilson, also suggest that as terrestrial habitats become isolated, or ‘islands,’ that fragmentation will continue to make habitat patches smaller, which will limit species dispersal and cause species loss in those areas. While the habitat-amount hypothesis (Fahrig, 2013) also posits that sheer volume of suitable habitat within a buffer is more important than the configuration of habitat, Suara (2021) theorizes that the misinterpretation of the habitat-amount hypothesis has led to dismissal of the importance of habitat configuration within a matrix. It is also possible that for some Anurans, with their limited dispersal ability, this matrix quality hypothesis may be more accurate than the habitat amount hypothesis, making the configuration, quality, and type of habitat matrix, vital for these species, while maintenance of 'island' populations (using the matrix quality hypothesis) is critical to facilitate species movement (Braaker et al., 2014; Pulsford et al., 2017).

As the interaction between a species and their habitat is a critical underpinning of conservation ecology, understanding species habitat and resource availability/distribution, is essential to studying delicate, habitat-specific species such as Anurans (Faulkner, 2004; Hamer & McDonnell, 2008; Johnson et al., 2013). Furthermore, by evaluating the presence and absence of species within a community often used as indicator species, inferences about the quality of habitat can be drawn. As research by Tilman et al. (1996) has shown, a direct correlation can be found between functionality of a system (abiotic/biotic processes) and the number of species within it. Ecologists have also shown a relationship between the productivity (biotic growth) of an ecosystem and the size of the food web, which can also be considered one measure of ecosystem function (Naeem & Li, 1997; Naeem, 2008). Local landscape variables such as vegetation structure and landscape matrix affect how wildlife species use the habitat, further allowing a relationship to be established between habitat quality and community diversity within a habitat. Additionally, Anuran species, which typically have a biphasic life history, often utilize both upland and wetland habitats. Studying amphibians can therefore provide a window into the quality of both habitats. Finally, overall biodiversity has been found to be higher within protected areas (Naughton-Treves et al., 2005). This is likely due to an increase in resource availability within a patch, but also ensures that any fragmentation in or around a protected area will result in increased chance of isolation in amphibian populations, as well as lowering the quality of the area due to edge effect (Santiago-Ramos & Feria-Toribio, 2021). Zhang et al. (2015) also described the importance of urban wetlands as cold islands, to counter the effect of urban heat islands. These numerous factors demonstrate a need to understand how urban factors affect a community of amphibians across the region, and how protected areas can utilize these species to best managed for community and ecosystem diversity.

However, industry knowledge and the resulting public works may not be sufficient to solve the biodiversity crisis. Increasingly, it is becoming more necessary to involve the public in the development of plans for land and water, most clearly on the use of public land that is owned by the government on behalf of the people, and that their tax dollars may contribute to. Additionally, involvement of private landowners in landscape modification of their own volition or in exchange for financial incentive has proven to be an increasingly effective way to improve habitat quality and support for local wildlife conservation efforts (Dayer et al., 2018; Lute et al., 2018; Sorice et al., 2011). To that end, it is essential to understand the values and views of the local population as much as possible while land use change planning is underway. These views may not be similar in different areas, among different cultures, or even within demographic groups, and so it is crucial to seek out the opinions and outlooks on potential avenues for wildlife conservation in any local area before the action is undertaken (Gangaas et al., 2015; Oh & Ditton, 2008; Thompson et al., 2015; van Eeden et al., 2020). Information from surveys of this nature may make the proposed actions more likely to be supported politically and socially, and thus may be the difference between success and failure. Some unsupported but successful projects may also lead to political/social backlash that could prevent similar, necessary actions in the future (Brooks et al., 2012; Brooks et al., 2013; Catalano et al., 2019).

This research focuses on the relationship between amphibian biodiversity and wetland productivity, as well as the relationship between amphibian community diversity and specific landscape factors, and how that relationship can be utilized to better protect wetland habitat. By studying this taxonomic community on a larger spatial and temporal scale than is typical, and utilizing several survey methodologies that are rarely combined, this work provides novelty in understanding relationships between systematically understudied taxa (Anurans) and their environment.

We used Anuran calling surveys and the environmental context for survey locations, in addition to Anuran trapping surveys, to understand the connections between land use change, Anuran community diversity, and productivity of wetland ecosystems. Community diversity of these taxonomic groups, as well as plant density, serve as a correlate for wetland productivity (biotic growth), as species such as the Northern spring peeper (*P. crucifer*) have been found to correlate with ecosystem productivity in the form of this biotic growth (Tilman et al., 1996).

We also utilized modeling techniques to identify the factors most conducive to maximizing biodiversity and system productivity at local, landscape, and regional scales, as well as different temporal scales. Finally, we surveyed local stakeholders to address the views of locals regarding land use changes necessary to protect biodiversity.

Study Area

All portions of this study took place in and around the Oak Openings Region (OOR) in Northwest Ohio, including areas of the Toledo Metropolitan Area (TMA) (Figure 0.1). 67 sites were chosen from 49 protected areas across the area, covering approximately 1000km^2 in the region, and are managed by 19 different organizations (Table 0.1). These areas comprise the largest protected areas in the region, as well as smaller managed lands, golf courses, cemeteries, and community centers, for a more realistic distribution of protected lands and wetland habitat across a gradient of minimal land modification to extensive land modification. Maps of all sites can be found in Figure 0.2.

The OOR, a biodiversity hotspot, contains more threatened species than any other area in Ohio, as well as six globally endangered plant ecosystems, making the region a complex and dynamic area for wildlife (ODNR, 2016). OOR is considered a mixed disturbance landscape, because of the effects of urbanization from the city of Toledo, Ohio, which has a population of 270,000, and a strong manufacturing presence. This is in addition to a large agricultural presence, with 67% of land built-up for 'cultural' purposes and just 33% in natural/semi-natural configurations (Martin & Root, 2020). Conversion of wetland areas to municipal, residential, and agricultural areas has led to the loss of over 90% of Ohio's wetlands. Only ~5% of land in the study region is protected land, including wetlands (ODNR, 2016). This conversion has only continued over the last several decades, as natural areas have been altered to early successional habitat that has reduced woodland cover in the region while increasing impervious surface (Root & Martin, 2017). The increase in highly modified landscapes that stems from these conversion efforts can also stunt the movement of species across the region.

Research Objectives

The objective of this research was to determine the relationship between human modified landscapes and Anuran community diversity and associate that community diversity with the productivity of wetland ecosystems in the study area. The larger goal for the study was to facilitate greater understanding of wetland ecosystems and the species within them, while assisting in land and water conservation in the study region. We addressed several questions related to (1) urban ecology, (2) land use management, (3) species conservation, and (4) environmental sociology. Studying an entire community of amphibian species, which are low to mid-trophic level species that are sensitive to their environment, and a larger spatial scale than most other studies, allows for greater representation of wetland productivity than can be afforded by other measures. Studying these organisms as indicators of wetland productivity through abiotic and biotic processes can help demonstrate that sensitive vertebrate species often need more land and water protected than has currently been afforded to them. We detail the methods undertaken and the principal results to address these questions within the following dissertation sections co-authored with Dr. Karen Root:

Chapter I: Effect of variation in landscape variables on presence/absence of Anuran species:

- Examined how changing variables related to landcover type may affect the community composition of Anuran species, and how that shift in community can be related to wetland ecosystem productivity.

Chapter II: Effect of variation in temporal variables on presence/absence of Anuran species:

- Utilized temporal variables to analyze the effect of weather on Anuran species presence, and if those variables differ between areas on the urbanization gradient.

Chapter III: Modeling predicted species occurrence and relationship with conservation activity:

- Developed spatial models designed to predict the distribution of Anuran species in the region and assess the factors that are most likely to contribute to greater species richness, and the presence of specific species.

Chapter IV: Understanding views of local residents on wildlife conservation and environment:

- Evaluated support among adults local to Northwest Ohio for wildlife conservation utilizing Likert scale surveys, which addressed their relationship to nature, their support for wildlife/environmental causes, and their personal willingness to commit to change on behalf of those causes.

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CHAPTER I: EFFECT OF VARIATION IN LANDSCAPE VARIABLES ON PRESENCE/ABSENCE OF ANURAN SPECIES

Introduction

Biotic and abiotic variation of spatial characteristics in an ecosystem help shape the assemblages of the species found in that same ecosystem (Faulkner, 2004; Hubbell, 2001; Seifan et al., 2012). These communities can likewise be altered by the change in habitat, or microclimate that is facilitated by land use change, with species potentially losing habitat or finding it less suitable for occupancy and breeding (Price et al., 2011; Sievers et al., 2019). Previous research has shown that amphibians specifically are especially vulnerable to this type of threat, with declines in community diversity found in correlation with land use change activity (Clipp & Anderson, 2014; Collins & Storfer, 2003). Research has also demonstrated that Anurans have varying responses to changes in landcover type, altering their behavior in response to these pressures (Parris et al., 2009**;** Sievers et al., 2019). These factors, as well as sensitivity to water quality and temperature, have led Anurans to be described as indicator taxa: species that can be used to indicate the quality of their environment based on their presence, absence, body condition, or population trends (Guzy et al., 2012; Welsh & Olliver, 1998). As Anurans are indicator species, the study of their community provides an opportunity to view a suite of sensitive species of ecological import in a unique, mixed-land use region, to better understand the relationship between wetlands and spatially diverse landscape modification.

Several components of highly modified landscapes, such as increase in road traffic and noise, impervious surfaces, habitat fragmentation, changes in water and air quality, and heat island effects, are all often related to the matrix of landscape, abiotic, and biotic factors that affect the quality of a wetland, including through sedimentation and hydrology changes

(Faulkner, 2004; Hogan & Walbridge, 2007; Lee et al., 2006,). These factors may also relate to the quality of a wetland habitat patch. Hall et al. (1997) defined this quality as the conditions and resources that produce occupancy, survival, and reproduction by an organism and population, and it is often measured by the amount of resources available in a particular habitat (Johnson et al., 2013). For the purposes of this study, wetland quality was defined as wetlands containing higher values of NDVI (Normalized Difference Vegetation Index, which may be a good predictor of carbon or resource inputs), and lower amounts of non-natural ground cover (impervious surface), which have been used previously to indicate trends in biologic activity (Pettorelli et al., 2005; Wiegand et al., 2008).

As the mixture of these landscape-based abiotic and biotic factors within a habitat patch plays a vital role in the functional nature of the patch, these factors can be related to the degree of effective isolation of a population, a key concern for Anurans, whose dual life phases and limited dispersal ability may affect population viability (Ricketts, 2001). Differing spatial scales, e.g., local, landscape, and regional levels can also influence the evaluation of these factors within a system (Knutson et al., 2000; Price et al., 2007). Understanding the effects of habitat at different spatial scales has been shown to be critical for understanding community requirements across multiple taxonomic groups, but multiple scales are not often utilized together when studying Anurans (Gould et al., 2012; Johnson et al., 2008; Knutson et al., 2000; Swanson et al., 2018). These researchers have helped connect the gaps to show the different interactions between habitat variables at multiple scales that affect wildlife, and furthermore, illustrate that landcover changes caused by highly modified landscapes threaten the long-term viability of natural areas and the Anuran species that depend upon them.

Additionally, the ongoing extinction crisis within the *Amphibia* class, limited studies conducted on them, and difficulties establishing population densities among Anurans using calling intensity warrant further study in the best evaluation methods for Anuran populations (Campbell Grant et al., 2023; Lawler et al., 2006). Further, many practitioners and species managers may lack the resources to effectively survey Anurans on the scale necessary to assess population density and viability (Svendsen & Campbell, 2008). It is essential to explore potential alternative methods to address Anuran population densities more comprehensively and simply. These several gaps in knowledge provide an opportunity to apply novel research to a field that needs new data to address the issues it faces.

Frog call surveys have been utilized for decades to take advantage of organisms that are often small and cryptic but rely on outward vocal displays to attract mates (Peterson, 1950). They are regarded as reliable, but flawed, depending on the question, because of the aforementioned inconsistency between calling intensity and population density (Čeirāns et al., 2020; Lawrence, 2017; Pesarakloo et al., 2019; Royle, 2004; Royle & Link, 2005). The efficacy of calling surveys, as well as the quantification of survey effort, has also been well-established in the literature (Bishop, 1997; Crouch & Paton, 2002; Pierce & Gutzwiller, 2004). In the Northwest Ohio area, however, studies have traditionally been limited to single species (Rickard, 2006), in relation to specific variables (Furlong, 2016) or require updating in the face of new land use changes (Baczynski, 2013).

To address these gaps, we designed a study based around the following research questions: 1. What are the effects of local and landscape variables associated with landscape modification on community composition of Anurans? 2. How does an expected change in community composition in more heavily modified systems correlate to measurements of wetland quality (i.e., productivity)? 3. Do some species within the community predict wetlands of a higher quality? 4. Which local and landscape variables are most highly correlated with presence?

Using the 11 Anuran species (Table 1.1) found in the Northwest Ohio area, and the land use change pattern around the metropolitan area of Toledo, Ohio, we utilized presence/absence surveys over three years (2021-2023) to evaluate both local and landscape traits that vary spatially. We designed our study to utilize the most comprehensive and extensive set of calling surveys possible, while retaining practicality and feasibility. We sought to determine how local and landscape traits relate to Anuran community diversity, and how those spatial differences relate to wetland quality. Here, we utilize Anuran species richness in a dynamic urban to rural gradient to assess the efficacy of using Anurans as an indicator of wetland productivity, which is one measure of ecosystem functionality and quality.

Methods

Calling Surveys

Survey sites were identified as under a rural, suburban, or urban classification, determined by the population density within one square mile of the survey point (<500 persons, 500-1999, ≥2000, respectively; data from Meridian Econometrics). These parameters were based on assessments of the same classifications from the U.S. Census Bureau, modified to fit the local area. All sites were chosen based on this density, proximity to wetland habitat, accessibility for surveyors, and being situated within the Toledo Metropolitan Area. Approximately 50 sites were chosen per year with roughly 24 rural, 13 suburban and 13 urban sites selected for each field season. Rural sites received more survey points annually due to the larger size of preserves/parks falling under this classification. Sites were grouped by survey night to include at least one from each urbanization class during each surveying night. The order sites were surveyed (within a night) was flipped every other week, to control the time of surveying. Sites had one additional survey point added if the protected area was larger than 500 acres, thus the higher number of rural sites. Further survey points (only necessary in Oak Openings Metropark, see Table 0.1 in Introduction section) were added to ensure sampling effort commensurate with the size of the protected area.

Prior to surveys beginning, surveyors were trained for 4-6 weeks (~30 minutes per day) to identify local species by call. Eleven species occur in our study area (Table 1.1). We considered three of these species ‘disturbance tolerant’, meaning that their inherent tolerance of the factors associated with anthropogenic land change would make them more resilient to urban environments (Callaghan et al., 2019; Gibbs et al., 2005; Pereyra et al., 2021). These three species (of 10) were *L. catesbeianus*, *L. clamitans*, and *A. americanus*. Because of the similarities in call, and physically identical appearance of *H. versicolor* and *H. chrysoscelis*, we lumped the occurrence of both species as solely *H. versicolor* complex.

Beginning 1 March each year, urban, suburban, and rural sites were surveyed, utilizing Anuran breeding call surveys taking place nightly from 1 March to 31 July (covering all endemic species' seasonal activity periods; Pierce & Gutzwiller, 2004). Seven survey nights per week were conducted by multiple teams for 22 weeks in 2021 and 2022, with six survey nights per week in 2023. Each team conducted between one and five survey nights per week. Each site was visited once per week, on the same night per week, for the full 22 weeks of the sampling period. Sites were not sampled if the daily temperature peak did not reach at least 7.2°C, wind speed was above 20mph, or there was heavy rain or snow, as these variables have been shown to markedly decrease Anuran activity (Villa et al., 2019).

Surveys began no earlier than 30 minutes after sunset. Upon arriving at the survey point, surveyors collected temporal data most commonly shown to affect frog calling (Villa et al., 2019). These included measurements such as air and water temperature (degrees Celsius), along with barometric pressure (inHg) and wind speed (mph), using commercially available apps. Ambient noise (dBs) was also recorded with a handheld microphone (RØDE VideoMic). Wind code (0-5 scale, characterized by visible movement of objects), sky code (0-8 scale, characterized by cloud/precipitation status) and noise index (0-4 scale, characterized by frequency of auditory disruptions), all adapted from FrogWatch USA's survey protocol, were recorded based on surveyor observation (FrogWatch, 2020). Finally, presence of any water body within 50m was identified by surveyor observation at each survey to assist in understanding local hydroperiod.

After collecting temporal data, surveyors remained silent for two minutes to allow for Anuran species acclimation (Frogwatch, 2020). A ten-minute surveying period then began after the acclimation period, which has been shown to be a suitable survey period to maximize species identification (Pierce & Gutzwiller, 2004). Species heard and the intensity of their call were logged by surveyors over the ten-minute survey. A calling intensity index, rated from one to three, was assigned to each species heard during each survey, based on the frequency of calls, with 'one' representing few calls with gaps between each individual, 'two' representing some overlap of calls, and 'three' representing a chorus of calls with constant overlap (Frogwatch, 2020). In 2023, exceptionally large choruses were graded as a four or five, as we believed these additional index values (which we termed a large chorus and a superchorus, respectively) could provide additional information pertaining to species density and population viability. Whenever a frog/toad was heard, random two-minute sections of each survey were recorded with a handheld microphone and were analyzed for potential corrections later. Notes on visual observations, weather and habitat conditions, and specific locations of call (direction, distance, etc.) were also recorded when relevant to assist in species and habitat assessment. Each surveyor was trained prior to beginning surveys, for 4-6 weeks using USGS' Public Quiz for Ohio frog calls and had reference calls downloaded from the same source available to them after each survey.

Local Scale Habitat Data Collection

Twice during the season (March and July), local scale habitat data were collected at each site. Measurements such as ground cover (percentage of litter/grass/bare/other), grass height (cm), leaf litter depth (cm), and number of coarse and fine woody debris were collected at 4-meter (m) increments on a 50-meter transect at each site and averaged (Turner & Root, 2018). Each 50-meter transect was parallel to the wetland and ran through the exact survey point. Other measurements, such as percentage of canopy cover (Figure 1.2), number of snags/shrubs/trees, were collected within a 10-meter radius of the survey point (Jáuregui et al., 2019). Finally, the percentage of emergent and floating vegetation covering the water body was estimated by surveyors (see Table 1.2). See Figure 1.1 for explanation of scale measurements.

Spatial Landscape Factors

Large-scale (250m and 1 kilometer (km)) habitat data was collected using satellite data available to the public through federal, state, and local governments. Using satellite database tools, variables such as distance from road, and size of habitat patch, traffic volume (AADT), and percentage of impervious surface were collected once per season (see Table 1.3). Landcover type was also collected via a 15-class landcover map for the Northwest Ohio area from Martin & Root (2020). These variables can be found in Table 1.3. Data for these variables were collected at both the 250m and 1 kilometer scale to evaluate the effect of these factors when in both a likely home range (250m) and likely upper limit (1km) of travel for all 11 study species (Baldwin et al., 2006; Heemeyer & Lannoo, 2012; Humphries & Sisson, 2012; Kovar et al., 2009). Temporal weather data from each field season including daily precipitation, days under drought, daily wind speed, snowfall, and hourly temperatures were collected via publicly available data via National Oceanic and Atmospheric Administration (NOAA), United States Geological Survey (USGS), or Environmental Protection Agency (EPA), but will be addressed in Chapter 2.

Data Analysis

Total species richness was evaluated for relationships with collected variables within 250m radius (from fixed sampling point at each site), an area based on estimations of habitat buffer size associated with amphibian presence (Harper et al., 2008; Zheng & Natuhara, 2020). Additionally, a 1km buffer from the sampling point was also utilized to examine the effects of a larger scale on the same measurements. Spearman's correlation matrices were utilized for initial assessment of correlations between diversity measurements and spatial characteristics, as well as correlation between different spatial characteristics. Factors that had a notable correlation (>0.30 or <-0.20, based on natural break points in the correlation matrix) were selected for further investigation via Generalized Linear Mixed Modeling (GLMMs) (variables of interest can be found in Tables 1.2 and 1.3). Spatial variable data was aggregated without regard to urbanization class or scale and evaluated against on another, and separately, against richness. Data within a site was collapsed to one measure per year, in an effort to elucidate spatial trends instead of temporal trends. Variables that were removed in this step due to correlation with other variables included grass height, number of fine woody debris, and amount of floating vegetation (Table 1.4). GLMMs were chosen for analysis because our data was time series (3 years) and utilized an unbalanced survey design (more survey sites in Rural areas than in Suburban or Urban). We utilized additive GLMMs following Beresford et al., (2018); Martin et al., (2021); Oliver et al., (2016) based on the additive nature of most of our variables.

Trends and related significance between spatial/structural features and richness within the 250m or 1km buffered areas were investigated using these GLMMs with time using R (version 4.3.2). We utilized a Poisson or gamma distribution with a log link function using the glmmTMB package. Further analysis utilizing the same methods also involved evaluating average number of species per survey, while also evaluating if the presence of 'disturbance tolerant' species altered these results. We also calculated a species' 'perceived' relative abundance, based on the frequency of interaction and calling intensity. We sought to identify spatial factors that were more likely to predict greatest species richness, while also identifying other spatial factors that were likely to predict ecosystem productivity. Further, we sought to utilize Anuran species/community richness to predict ecosystem productivity.

Principal Component Analysis

To further reduce the dimensionality of our data, we utilized Principal Component Analysis (PCA). Variables for PCA were normalized prior to analysis. Example PCA plot of 250m spatial variables can be seen in Figure 1.3. Following this analysis, 6pm to 6am traffic count, raw traffic count, shrub cover, number of snags, and other variables were removed because of close correlation with other variables, namely the variables gathered from remote sensing. Both the 250m and the 1km datasets far surpassed the minimal PCs goal (15 and 16 PCs, respectively), and even when highly correlated variables were removed and a reduced PCA was run, the 250m and 1km sets still featured 8 and 7 PCs, respectively. Variables removed after PCA are included in Table 1.4.

Model Construction

To quantify the impact on our measures of interest, we first utilized single variable GLMMs to assess significance and support for relationships between richness and habitat. We did not conduct tests on variables that were shown to be correlated via Spearman's test or PCA. Variables were selected for further analysis by significant *p*-value. In almost all cases, these were also the model variables that were supported by the best model fit. Model fit was evaluated with Akaike Information Criterion, or AIC. The variables removed varied depending on the response variable (species richness, average species per survey, or class). Variables removed as a result of Spearman's correlation, PCA, or after single variable GLMMs can be found in Table 1.4. In initial analysis, all variables were removed due to correlation with other variables or non-significant model results. However, we felt that it was prudent to attempt to assess which of the local scale variables (e.g., one meter ground cover, grass height, canopy cover, etc.) that we collected was the most informative, even if not inherently more informative than data from the 250m or 1km scale.

We were also interested in defining the spatial differences between each of the three classes along the urbanization gradient of modified landscape (rural, suburban, and urban). This was partially to determine if classifying these areas in this manner would be useful for future surveys, based on their comparisons to other results. Using additive GLMMs, we analyzed the variables that were most likely to contribute to our urbanization class on three different scales; 250m, 1km, and mixed scale (both 250m and 1km variables included).

Results

Calling Surveys – Summary and General Trends

At the conclusion of our three field seasons of sampling, we conducted 1800 frog call surveys over 67 sites. We retained at least 85% of sites between years, but sites were added and removed each year to expand potential generalizability of the dataset. From 1800 surveys, we recorded 2174 individual calling interactions for an average of 1.21 (SEM = 0.002, indicating greater precision). However, this differed among urbanization class (Class), with rural sites averaging 1.67 species per survey (SEM = 0.004), suburban averaging 0.94 species per survey (SEM = 0.004), and urban sites averaging 0.68 species per survey (SEM = 0.004). This divide became starker when considering the species that we considered "disturbance tolerant," with these three species representing 42.18% of all records. With these species removed, the average species per survey in rural areas was 1.16 (SEM = 0.005), suburban areas were 0.41 (SEM = 0.004), and urban areas were 0.21 (SEM = 0.003). Total number of records, and total number of records without disturbance tolerant species can be found in Figure 1.4. Species identified at each site, as well as their perceived relative abundance, can be found in Table 1.5.

The most common species encountered was *L. clamitans*, with 408 records over three years, followed by *P. crucifer* with 375 records. The least common species was *A. fowleri* with 33 records and *L. sylvaticus* with 34 records, though those numbers are likely conservative due to lower detectability. *P. crucifer* was also the most intense calling species, with an average calling intensity (CI) value of 2.26 (SEM = 0.044), while the weakest calling species was *A. fowleri* with a CI of 1.01 (SEM = 0.068).

Local Scale Habitat Data

Among variables that were not removed due to correlation, change in litter depth was the factor that best fit an association with richness (SEM = 0.015, p = 0.421) and average species per survey (SEM = 0.016, p = 0.821) (both negative relationships, lower annual decline in litter depth was found in sites with lower richness). This was also the local scale variable that best explained the difference between urbanization classes, exhibiting a positive relationship (suburban areas had less change in litter depth than rural, and urban less than suburban). All models included very low coefficient estimates and Z-scores, which likely indicates that despite it being the best of the local scale variables to explain richness, the magnitude of the relationship is not strong.

Spatial Assemblage of Urbanization Gradient Classes

Using Generalized Linear Mixed Modeling, the variables that formed the best model to explain the differences between urbanization classes were the percentage of impervious surface (-, or negative relationship) and percentage of residential/mixed landcover (-), both at the 1km scale (SEM = 0.546, p = 0.014; SEM=0.419, p = 0.0.31, respectively, Table 1.8). At 0.9 ΔAIC, those same variables, as well as 250m percentage of floodplain forest (+) (SEM = 0.453, p = 0.267) formed the next supported model. The only other model that was well-supported (within two ΔAIC), contained those same three parameters, while adding percentage of wet prairie at the 1km scale (+) (SEM = 1.92, p = 0.354). Model details can be found in Table 1.8, while results from additional model sets containing only 250m or 1km variables can be found in Tables S1.7-S1.9.

Richness and Survey Yield

Similarly to the tests to define the structure of the urbanization gradient classes, we evaluated which habitat variables would best predict greater species richness, utilizing a combination of 250m and 1km scale variables. The best supported model contained percentage of urban at the 1km scale (-), and percentage of residential/mixed use land, also at the 1km scale (-) (SEM = 0.396, $p = 0.029$; SEM = 0.198, $p < 0.001$, respectively). Six total models were well-supported, with the top five featuring only variables from the 1km set. All six models contained percentage of residential/mixed used landcover, while four models contained either percentage of swamp forest (+) or a measure of NDVI (+). These results can be found with other model details in Table 1.6. Results from model sets containing only 250m or 1km variables in each model can be found in Table S1.1-S1.3.

We also used GLMMs to evaluate the effect of habitat variation on the average species detected per survey, as a different measure to quantify community composition and relative density. The best supported model contained urbanization class (-, SEM = 0.150, $p = 0.038$) and percentage of swamp forest at 1km (+, SEM = 1.334, $p = 0.154$). However, urbanization class alone (-) formed the next best supported model, at 0.1 ΔAIC (SEM = 0.117, $p < 0.001$), indicating that the urbanization class (and by extension, human population density) likely had an effect on the number of species we observed during our surveying. Many models in this set were well-fitting, with six at <1 ΔAIC alone (Table 1.7), which may indicate a complicated system of variables that all influence, to some extent, average species per survey.

Once again, almost all of these best fitting models featured at least one variable on the 1km scale. Among these six best fitting models, the only variables that had positive effects on average species per survey, were again percentage of swamp forest and higher values of NDVI.

Among all well-fitting models, almost all (94%) contained a measure of urbanization (class, impervious surface, or urban/residential landcover), which all had negative effects on average species per survey. Variables that had positive effects included various measures of NDVI, percentage of swamp forest, upland savanna, upland prairie, or upland deciduous forest. The results from model sets containing only 250m or 1km variables in each model can be found in Table S1.4-S1.6.

Wetland Productivity

Utilizing the single variable GLMMs, we also analyzed the predictive ability of productivity of an ecosystem for species richness, using NDVI as a proxy variable (Berveglieri et al., 2021; Paruelo et al., 2001; Wiegand et al., 2008). We found that average NDVI best predicted both species richness and average species per survey. In both cases, an increase in NDVI at the 1km scale best predicted both measures (SEM = 0.005, $p < 0.001$; SEM = 0.018, $p = 0.001$, respectively).

Among individual species, higher amounts of most NDVI values best predicted the relative abundance (number of records/number of surveys * average calling intensity, henceforth RA) of *P. crucifer*. However, greater average NDVI at 250m best predicted greater *L. sylvaticus* RA. Few other species predictive models of an NDVI measure came within 2 ΔAIC of the *P. crucifer* models. These included higher *P. triseriata* RA predicted by greater average NDVI at 1km (1.5 ΔAIC), higher *L. sylvaticus* RA predicted by higher average NDVI at 1km (0.4 ΔAIC), and higher *H. versicolor* RA predicted by higher early NDVI at 1km (1.3 ΔAIC). When comparing models utilizing only *P. crucifer* RA against all species richness or average species per survey, the *P. crucifer* models were the response variable best supported at 250m scale. However, in the models for 1 kilometer data, the results were mixed; the best model for *P.*

crucifer relative abundance was predicted by early NDVI; average species per survey by late NDVI, and species richness by average NDVI. All were well supported, however, with ΔAIC between 0.8 and 3.4. In all cases, higher NDVI values predicted greater relative abundance of Anurans, though the scale spatial scale of interest caused variation.

Discussion

We utilized calling surveys and mixed modeling to assess the impact of various habitat measures on Anuran communities in Northwest Ohio and relate that Anuran community composition to ecosystem productivity. We found that landscape structure repeatedly formed the best models to predict Anuran species richness. Specifically, models including anthropogenic cover classes (urban, residential/mixed, impervious surface), were consistently among the best supported. In each case, those models indicated that the anthropogenic cover classes have a negative impact on species richness and average species per survey. Our urbanization classes were also well defined by these cover types, with the best supported models indicating that an increase in at least one of the anthropogenic cover classes indicated a site was suburban or urban.

We found that across three years of surveying, *L. clamitans* and *P. crucifer* were the most commonly recorded species, while *P. crucifer* was also the loudest on average, theoretically indicating higher densities (as greater calling intensity does not necessarily indicate greater densities in all species). We also found that *L. sylvaticus* and *A. fowleri* were the most rarely encountered species in our study. This is not surprising, as *L. sylvaticus* is an explosive breeder with a limited breeding period (Lambert et al., 2017). Still, the limited number of areas in which we detected those species even once, and the limited number of detections overall, indicates that they should be evaluated further for potential regional decline, or loss of breeding habitat.

Among our Generalized Linear Mixed Models, we found significant evidence for the negative impact of anthropogenic cover classes on Anuran species richness. This is similar to a wealth of previous literature, which finds a connection between the effects of land use change and biodiversity decline (Hamer & McDonnell, 2008; McKinney, 2006; Piano et al., 2020). We also identified that in the best supported models, variables positively related to species richness or average species per survey were almost always percentage of swamp forest, or a measure of NDVI. Swamp forest was the positive variable most commonly included in the best models**.** While the mechanism for the effect of increasing NDVI on Anuran density/richness is not clear (Oldekop et al., 2012; Pilliod et al., 2021; Rowe et al., 2024; Vasconcelos et al. 2019) these results were similar to previous literature in regard to the importance of swamp forest to biodiversity (Hörnberg et al., 1998; Pearlstine et al., 2002). The designation of swamp forest is not used in all studies that utilize landcover classes as explanatory variables; swamp forest as a cover class includes seasonally inundated freshwater forests, which are ideal for the breeding needs of many species (Pfingsten et al., 2013). It is also worth noting that other water-holding classes (wet prairie, perennial ponds, wet shrubland, floodplain forest) had varying effects on models. Perennial ponds did not appear to have any significant effects on our response variables even within single variable models, while floodplain forest, despite showing a significance during variable selection, was included in only a few of the top models for all analyses. Wet prairie was also included in some of the more complex models. Wet shrubland was a unique case, occurring in extremely low number of cells across our entire study area. However, all of these cells occurred in an area of high richness, and so the value of wet shrubland was considered highly significant, but with unreliable estimates. We do not report models that include wet shrubland here, but we suggest that the value of wet shrubland be further evaluated by future studies.

The value of multiple scales in our analysis cannot be understated. Though Anurans have limited dispersal ability compared to other vertebrates, the 250m limit we set likely does not capture all the potential influence of surrounding environments. To better capture this influence, we also modeled the value of many variables within 1km. We found that in most models when both scales were considered, at least one measure at 1km was included, and occasionally both scales were included, e.g., one of the well-fitting models for species richness included 250m percentages of both swamp forest and residential/mixed landcover, as well as late season NDVI at the 1km scale. We offer models here at the 250m or 1km scale solely for specificity if the level of interest for practitioners is at that scale. However, most frequently, the best fitting models contained only variables at the 1km scale. Considering the frequency of inclusion of variables that measured urbanization in those models, this likely indicates the influence of a wider landscape matrix on Anuran richness and density, well-beyond the scale of their typical breeding pools. In that vein, it is important to acknowledge that we had a large number of models within 2 ΔAIC of the best model, specifically in the combined scale groups. Even at this conservative choice of ΔAIC, this means many models could be considered potentially equally valid, as models within two ΔAIC can often be considered as having high support (Burnham & Anderson, 2004). For this reason, there were a high number of variables that can be considered as important to both species richness and average species per survey among Anurans in our study area. Urbanization class, average NDVI, late season NDVI, early season NDVI, percentage of impervious surface, and percentages of residential, upland savanna, swamp forest, urban, upland prairie, upland deciduous forest, and wet prairie were all included on at least one scale in models for species richness or average species per survey. This illustrates the complexity of the system, as previously evidenced from the results of our PCA, but also the difficulty in ascertaining even two or three specific variables that should be used as a "silver bullet" to indicate greater species richness.

Despite these caveats, the best supported models consistently showed that the percentage of impervious surface and percentage of residential/mixed use negatively, and the percentage of swamp forest positively, influenced both species richness and average species per survey. Finally, the coefficient scores and *Z*-scores of most of our models were lower than expected in most cases, which may further indicate the complicated effect of many of these variables when viewed in isolation.

Frequently, models identified at least one measure of NDVI as a useful parameter in predicting species richness. NDVI has often been used as a measure of ecosystem functionality, and may be an indicator of available carbon, or resources that support early life stages (Pettorelli et al., 2005; Wiegand et al., 2008). We were able to identify through modeling that NDVI, specifically average NDVI, predicted greater species richness. Furthermore, we were also able to use models to show that higher levels of NDVI can predict greater relative abundance of *P. crucifer*. This supports the previous findings of Knutson et al. (2000) and Price et al. (2005) that found *P. crucifer* presence predicts higher ecosystem productivity. Utilizing these modeled relationships, we find that greater species richness and abundance of *P. crucifer* can both be used effectively to predict NDVI, and subsequently, ecosystem productivity, at multiple spatial scales. Our results support the body of literature that species richness, especially of indicator taxa, can be used to predict ecosystem productivity, and potentially functionality. Furthermore, our model results indicated that in many cases, especially at smaller scales, higher NDVI values can better predict *P. crucifer* relative abundance than predict species richness. However, extremely high NDVI values may mean no standing water for Anurans to breed. While we do find support for both of these hypotheses, we suggest further research evaluate the maximum NDVI level that provides suitable habitat for a wide suit of Anuran species, while also incorporating the use of Normalized Difference Water Index (NDWI). That level could then be correlated with ecosystem productivity, and Anuran species used appropriately as indicator species.

Our models did not include multiplicative effects, because the variables included were additive. We concede, however, that there may be multiplicative effects on richness based on the combination of variables that cannot be addressed here in our models.

When our findings are viewed in total, we find that there was a negative effect on species richness and the average number of Anuran species observed per survey, when compared to the amount of human-modification on the landscape. This finding holds largely regardless of metric, be it our urbanization classes, percent urban landcover, or percent residential/mixed use landcover. Swamp forest, and to a lesser extent, wet prairie, are likely the landcover variables that have the strongest positive relationship with higher species richness/average species per survey. Our study is the first in this region to conduct such a wide-ranging study on the full suite of Anuran species at multiple scales, providing novel benefits to the field of both urban ecology and landscape ecology. As our results highlight, what drives the spatial dynamics of Anuran communities is complex, especially in human-modified landscapes.

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CHAPTER II: EFFECT OF VARIATION IN TEMPORAL VARIABLES ON PRESENCE/ABSENCE OF ANURAN SPECIES

Introduction

Like most organisms, Anurans utilize timed annual periods to focus their breeding efforts. These periods can range throughout the year depending on the species and geographic location (Green, 2017; Pfingsten et al., 2013). Breeding periods can be conducted by two individuals or, more typically, by several individuals, congregating near available bodies of water that facilitate growth of their aquatic eggs and larvae (Pfingsten et al., 2013). Because of this aquatic substrate requirement, Anuran emergence and breeding periods are often tied to seasonal precipitation patterns (Dervo et al., 2016; Pfingsten et al., 2013; Saenz et al. 2006). Similarly, the ectothermic life history of Anurans also ties their annual cycle to temperature and weather patterns (Fukuyama & Kusano, 1992; Pfingsten et al., 2013). Under the multitude of threats of global climate change (GCC), temporal patterns in precipitation, weather patterns, and atmospheric heat are expected to change regionally over the next several decades (Root et al., 2003). These temporal changes can affect Anurans through altering their body condition, prey density, breeding season time/length, as well as potential for larval development time (Klaus & Lougheed, 2013; Li et al., 2013). Some populations or species may even face extirpation or extinction stemming from these major temporal alterations (Klaus & Lougheed, 2013; Parmesan, 2006; Root et al., 2003). For these reasons, it is fundamental that any study of Anurans consider the effects of temporal changes on amphibians, through the study of individual species or communities (Campbell Grant, 2023).

Through temporal niche partitioning with their taxonomic relatives, as well as thermal activity requirements, many Anuran species have well-established breeding time periods for different areas in their range, which are well documented in the literature, including in the Ohio region (Pfingsten et al., 2013). This predictability makes the study of Anuran community changes feasible, as it relates to temporal changes. Researchers can understand the potential annual changes of several species as a response to temporal changes by utilizing species breeding calls, which are a well-established method of identifying Anuran species (Crouch & Paton, 2002, Dorcas et al., 2009). Understanding the effects of these temporal trends yields insights that could be used to address species declines, potential for extinction, and range shifts that are expected under GCC (Li et al., 2013; Root et al., 2003). Additionally, GCC is expected to alter community composition, with widely distributed generalists predicted to be the most positively affected (Davey et al., 2013; Wassens et al., 2011). This may create an opportunity in our study region for the proliferation of species that we have termed “disturbance tolerant;” species that have been identified as generalists and tolerate landscape modification. Such species are *L. catesbeianus*, *L. clamitans* and *A. americanus*.

Anurans alter their call behavior and emergence time due to temporal changes in factors such as: temperature and precipitation (Dervo et al., 2016), winter season severity (Arietta et al., 2020; McCaffery & Maxell, 2010) and light/noise pollution (Higham et al., 2021; Luscier et al., 2023). There is also evidence for success in utilizing species distribution and density based on similar temporal changes to draw conclusions about the impact of these changes (Higham et al., 2021; Klaus & Lougheed, 2013; Luscier et al., 2023).

Further, the documented effects of temporal shifts in temperature (e.g., the heat island effect) in cities, combined with the potential for GCC changes in the timing, amount, and mechanism of precipitation mean that studies are urgently needed to address the potential effects on species that rely on both temperature and precipitation to maintain populations (Benard, 2015;

Deilami et al., 2018). We expected that our study area would see similar trends, with urban environments featuring higher air and water temperatures, as well as louder ambient noise. We further expected that as a result of these disturbances, the community composition of urban sites would be simpler (e.g., fewer species), and more consistently comprised of disturbance-tolerant species.

Little is known about the general population trends for the native Anuran species in our study area, the Oak Openings Region of Ohio/Toledo Metro Area. It is believed that *A. blanchardi* is declining in many portions of Ohio (Lehtinen & Skinner, 2006), while *P. triseriata* is declining as a species (IUCN, 2023), and *L. pipiens* is declining in western portions of its range (Johnson, et al. 2011). Population density trends are not available for other native species, however. We expected that due to active land use change (towards anthropogenic-use classes) and the effects of GCC, more specialized species would be declining in density, and disturbance-tolerant species would be growing in density and range. This would support previous literature on the subject (Johovic et al., 2020; Nori et al., 2011; Wassens et al., 2011). We further expected to collect extensive data on the temporal variables that most affect Anuran presence, as well as species community composition, and average species per survey. Overall, we sought to identify the temporally shifting factors that may affect Anuran species richness in Northwest Ohio.

Methods

Calling surveys

Survey sites were identified as under a rural, suburban, or urban classification, determined by the population density within one square mile of the survey point (<500 persons, 500-1999, ≥2000, respectively; data from Meridian Econometrics). These parameters are based on assessments of the same classifications from the U.S. Census Bureau, modified to fit the local area. All sites were chosen based on this density, proximity to wetland habitat, accessibility for surveyors, and being situated within the Toledo Metropolitan Area. Approximately 50 sites were chosen per year with roughly 24 rural, 13 suburban and 13 urban sites selected for each field season. Rural sites received more survey points annually due to the larger size of preserves/parks falling under this classification. Sites were grouped by survey night to include at least one from each population density classification during each surveying night. The direction of each survey night (6-9 sites surveyed in order during the same night each week) was flipped every other week, to control the time of surveying. Sites had one additional survey point added if the protected area was larger than 500 acres (hence the higher number of rural sites). Further survey points (only necessary in Oak Openings Metropark) were added to ensure sampling effort commensurate with the size of the protected area.

To collect the data necessary to complete our project goals, beginning 1 March each year, urban, suburban, and rural sites were surveyed, utilizing Anuran breeding call surveys taking place nightly (beginning at least 30 minutes after sunset) through 31 July (covering all endemic species' seasonal activity periods; Pierce & Gutzwiller, 2004). Seven survey nights per week were conducted for 22 weeks in 2021 and 2022, with six survey nights per week in 2023. Each site was to be visited once per week, on the same night per week, for the full 22 weeks of the sampling period. Sites were not to be sampled if the daily temperature peak did not reach at least 7.2°C, wind speed was above 20mph, or there was heavy rain or snow, as these variables have been shown to markedly decrease Anuran activity (Villa et al., 2019).

Upon entering each site for a survey, surveyors collected weather data such as temperature and water temperature (degrees Celsius), along with barometric pressure (inHg) and wind speed (mph), using commercially available apps. Ambient noise (dBs) was also recorded with a handheld microphone (RØDE VideoMic). Wind code (0-5 scale, characterized by visible movement of objects), sky code (0-8 scale, characterized by cloud/precipitation status) and noise index (0-4 scale, characterized by frequency of auditory disruptions), all adapted from FrogWatch USA's survey protocol, were recorded based on surveyor observation (FrogWatch, 2020). These weather data were selected based on literature review of factors most commonly shown to affect frog calling (Villa et al., 2019). Finally, presence of any water body within 50m was identified by surveyor observation at each survey to assist in understanding local hydroperiod.

After collecting weather data, surveyors remained silent for two minutes to allow for Anuran species acclimation (Frogwatch, 2020). A ten-minute surveying period then began after the acclimation period, which has been shown to be a suitable survey period to maximize species identification (Pierce & Gutzwiller, 2004). Species heard, and the intensity of their call were recorded by surveyors over the ten-minute survey. The calling intensity index, rated from one to three, was assigned to each species heard during each survey, based on the frequency of calls, with 'one' representing few calls with gaps between each individual, 'two' representing some overlap of calls, and 'three' representing a chorus of calls with constant overlap (Frogwatch, 2020). In 2023, exceptionally large choruses were graded as a four or five, as we believed these additional index values (which we termed a large chorus and a superchorus, respectively) could provide additional information pertaining to species density and population viability. Random two-minute sections of each site were recorded with a handheld microphone when a frog was heard, to be analyzed for potential corrections later. Notes on visual observations, weather and habitat conditions, and specific locations of call (direction, distance, etc.) were also recorded when relevant to assist in species and habitat assessment. Each surveyor was trained for 4-6 weeks using USGS' Public Quiz for Ohio frog calls and had reference calls downloaded from the same source available to them during and after each survey.

Temporal and Weather Variable Factors

Large-scale habitat data was collected using satellite data available to the public through federal, state, and local governments. Using satellite database tools, weather data from each field season including average daily precipitation, total season precipitation, average/minimum/maximum temperatures, number of days with precipitation, days under drought, and daily wind speed were collected via publicly available data via National Oceanic and Atmospheric Administration (NOAA), United States Geological Survey (USGS), or Environmental Protection Agency (EPA). These data were collected and aggregated separately for three time periods relative to each survey season: the previous breeding season (defined as March through August, the previous calendar year), the previous winter (September through February immediately preceding the survey season) and in-season (March through August during the survey season). These three aggregations were chosen to separately analyze the potential effects of all three different time periods on the number of species identified, calling intensity, and relative abundance, as previous research has shown population fluctuations in Anurans when exposed to starkly different weather conditions during the winter (Bradford, 1983; McCaffery & Maxwell, 2010; Pilliod et al., 2022).

Data Analysis

Total diversity (richness) and species per survey were evaluated for relationships with collected local and regional weather variables. Initial relationships were evaluated with Spearman's correlation matrices, and afterwards, significance between temporal features and urbanization gradient class was determined using ANOVA or Kruskal-Wallis tests (depending on assumption checks) as was species richness. We also calculated detectability using Presence in the single-species modeling platform to determine which temporal factors would most influence the presence of species and assess if detectability would significantly alter our results between site classes (Mackenzie, 2012; Mackenzie et al. 2017).

Results

Calling Surveys – Summary and General Trends

After three field seasons of sampling, we conducted 1800 frog call surveys over 67 sites. We retained at least 85% of sites between years, but sites were added and removed each year to expand potential generalizability of the dataset. Over 1800 surveys, we recorded 2174 individual calling interactions for an average of 1.207 (SEM = 0.002). However, this fluctuated between urbanization gradient class (Class), with rural sites averaging 1.674 species per survey (SEM = 0.0041), suburban averaging 0.936 species per survey (SEM = 0.0041), and urban sites averaging 0.681 species per survey (SEM = 0.0036). This divide became starker when considering the species that we considered “disturbance tolerant,” meaning that their inherent tolerance of the factors associated with anthropogenic land change would make them more resilient to urban environments (Callaghan et al., 2019; Gibbs et al., 2005; Pereyra et al., 2021). These three species (of 10) were *L. catesbeianus*, *L. clamitans*, and *A. americanus*. These three species represented 42.18% of all records, and when not included the average species per survey in rural areas was 1.158 (SEM = 0.0048), suburban areas was 0.406 (SEM = 0.0039), and urban areas were 0.206 (SEM = 0.0029). Total number of records, and total number of records without disturbance tolerant species can be found in Figure 1.4.

The most common species encountered was *L. clamitans*, with 408 records over three years, followed by *P. crucifer* with 375 records. The least common species was *A. fowleri* with 33 records and *L. sylvaticus* with 34 records, though those numbers are likely conservative due to lower detectability. *P. crucifer* was also the most intense calling species, with an average calling intensity (CI) value of 2.26 (SEM = 0.044), while the weakest calling species was *A. fowleri* with a CI of 1.01 (SEM = 0.068).

We found that species generally adhered to their perceived breeding periods and environmental tolerances (Pfingsten et al., 2013). However, we observed several instances of species calling well outside their documented breeding period. The most striking example was *P. triseriata,* which has a documented calling period of early March to very early May in our region, calling occasionally in June and July on cooler nights (Pfingsten et al., 2013; Whitaker, 1971).

Detectability

We modeled detectability of each species over all three field seasons using Presence (version 6.1) and found that detectability did not vary significantly across years within a species but varied between species. As expected, those species that we perceived to be rarer had much lower detectability rates than those we detected frequently. Detection models consistently identified Julian day and water temperature as the factors most likely to impact detection, with air temperature and water presence occasionally identified in select models. However, regardless of the model parameters, estimated detection rates within species did not consistently vary. The only species that had variable detection rates vary was *L. sylvaticus* (0.15 up to 0.52 depending on model parameters).

Temporal Characteristics of Urbanization Gradient Classes

Weather data taken at the time of surveying was aggregated across years for analysis. Between urbanization classes, only three of the 14 measures were significantly different between classes (Table 2.1). Only wind code (p = 0.038) and noise index (p < 0.0001) were significantly different between classes (Figure 2.1a). However, these measures were qualitatively based on surveyor perception, and quantitative measures of the same data did not demonstrate a significant difference between urbanization classes (wind speed, p = 0.168, ambient noise in decibels, p = 0.193). The only other test that was significant was barometric pressure (Hg) between urbanization classes, using Kruskal-Wallis tests (Figure 2.1b).

Additionally, because other studies have previously demonstrated that amphibian populations fluctuate year to year based on previous year's weather (McCaffery & Maxwell, 2010; Pilliod et al., 2022), we utilized weather station data to examine if these affected our findings. We found that again, there were no significant differences between site urbanization classes in any temporal measure (see Table 2.2 for variable list, Figure 2.2). We conducted these tests utilizing the unaggregated data while blocking for individual years.

Temporal Changes and Relationship with Species Richness

We observed that, when measuring species richness, most average temporal measures did not affect how many species were observed in that site each year. The average measure of wind code and noise index again were the only measures that were significantly influential on the number of species observed per year (especially among non-disturbance tolerant, or 'DT' species), with sites that had higher wind codes more likely to have only one species (p = 0.0226) and the same trend in noise index (p = 0.0012) (Figure 2.3). In all other measures, including average time, noise (dB), wind speed, Julian date, air pressure, etc., there was no significant influence on recording higher species richness.

Furthermore, utilizing previous season data (see Table 2.2), we found no significant differences between temporal data measures across a season and greater species richness (*p* range = 0.1207-0.8469). No measures approached significance (*p* = 0.05-0.10) when utilizing Kruskal-Wallis tests after failing assumption checks.

Discussion

From our survey results, we were not able to detect temporal differences driving the observed differences in Auran community assemblage and perceived population size differences. We did not find a difference between any temporal measures we took at the time of each survey, save for three. Wind code and noise index, both qualitative measures based on the perception of the surveyor, were two of the three significantly different between urbanization classes, with rural sites being perceived as less windy and less noisy by surveyors. However, this may have been a function of surveyor bias, at least in regard to noise. Surveyors were aware of the urbanization class designation of a site, and so could inflate the value if they unwittingly believed an urban environment would be noisier. Both of these differences could also have been because of increased vegetation density and difference in structure at even our urban survey sites (Ow & Ghosh, 2017). This demonstrates the volatility of qualitative metrics, especially for the assessment of species presence. It is also possible that the structure of the landscape (e.g., buildings and impervious surfaces compared to forest and water) played a role in both the sound intensity perception and perceived wind intensity. Both of these measures were, however, repudiated by quantitative measures of the same weather data. Wind speed and noise in decibels were not significantly different between classes, despite the conflicting results involving surveyor observation on a qualitative scale. As a result, we are comfortable concluding that the only temporal difference between our site classes was in barometric pressure (Hg), which was slightly lower in rural areas. However, we do not feel that barometric pressure alone explained the variation in richness and average species per survey, as well as differences in calling intensity, which were observed and detailed in Chapter 1.

Following other studies that found differences in amphibian survival from year to year based on past weather, we examined the effect of other time periods on richness and the average species (McCaffery & Maxwell, 2010; Pilliod et al., 2022). The combination of ectothermy and thus temperature sensitivity, and shifting patterns in precipitation under climate change, makes Anurans uniquely sensitive to past precipitation and temperature patterns (Dervo et al., 2016; Green, 2017; Li et al., 2013). We did not find strong or consistent evidence of the effect of differences in average weather data from either the previous winter (September through February) or the previous breeding season (March through August). However, of the significant factors we did identify that were based in previous seasons, both were related to the amount of precipitation received during the previous winter. Sites with only one species identified were likely to receive less precipitation on average than sites where we identified between two and seven species. These results were very general, as these data were averaged over a season, and we did not classify snow and rain separately, which may be an important distinction, especially for early breeders (*P. triseriata, P. crucifer*, *L. sylvaticus*) (Arietta et al., 2020; Muths et al., 2020; Pilliod et al., 2022).

Some species, such as *A. blanchardi,* have a single year life cycle, while others, such as *L. catesbeianus*, can have a one plus year tadpole stage alone (Lehtinen & MacDonald, 2011; Pfingsten et al., 2013). These differences in species' life history can mean differing responses to changes in weather conditions over the course of their (or their parent's) life cycles. As a result, it is likely necessary for future studies to examine the responses of specific species to previous season weather patterns in our study area, similar to Pellet et al. (2006), Saenz et al. (2006), and McCaffery & Maxell (2010).

Because of the consistent frequency of our surveys and multi-year design of our study leading to many site visits during species breeding periods and combining that with the limited variability between model parameter estimates, we feel confident that our surveys accurately reflected the distribution of species and their presence at select sites. This is in addition to the biological relevance of the identification of factors most likely to influence detection (air/water temperatures, Julian date) and their general relationship to time of year.

Our results here run contrary to some other temporal based studies that examine temporal shifts or changes in cities (Deilami et al., 2018; Higham et al., 2021). We did not observe a significantly warmer environment, nor a significantly louder environment, between urban and non-urban environments (Table 2.1). Additionally, artificial light at night (ALAN) has also been shown to disrupt Anuran population persistence and health (Luscier et al., 2023), which we were unable to quantify in our study, and could have differed significantly between our urbanization classes. Our temporal analysis level was coarse (e.g., annual scale), designed to serve as a companion to the analysis in Chapter 1, to identify potential sources of other variation (that were not spatial factors) that could explain local differences in species richness, community composition, and perceived relative density.

It is possible that more detailed analysis could identify specific variables of interest for continued study. Further, our collection of these data, especially the previous season data, relied solely on publicly available government data taken at a coarse temporal scale. Utilizing different methodologies focused on temporal changes, such as more frequent monitoring or use of climate change data, a la Higham et al. (2021), Pilliod et al. (2022) and Luscier et al. (2023), may provide more specific results, especially when examining specific species.

However, our research interest was not in the fine scale phenological changes associated with temporal measures and amphibian activity; instead, we sought to focus on the drivers of Anuran richness and distribution across the landscape. This is largely because there are countless temporal variables that can be investigated on differing time scales, including time of day, day to day, weeks, months, year, previous years, and extrapolating to future years. As our research questions are based on human impacts, we did not feel all of these time scales were relevant. These studies are highly valuable to the field, but we felt it more appropriate to focus on a set of immediately tangible variables that can be addressed by local land managers. It is effectively impossible for a single land manager to address the causes and outcomes of climate change; but it is reasonable and within reach of a manager to address the causes of Anuran community decline in a local area. Anuran behavior and activity have been shown to be affected by temperature, precipitation, and ambient noise, so we felt it prudent to address these questions on a coarse scale.

We are confident based on our statistics and detection probabilities that the few significant results we report here represent biologically relevant and significant trends, but they are not the driving force behind the difference in species richness, community composition, and our perceived relative density. These results can serve as a stepping stone to understanding the effects of temporal changes on Anuran communities, while also helping understand the true effects of spatial variation on the landscape.

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CHAPTER III: USING MAXENT TO EVALUATE RELATIONSHIPS BETWEEN ANURAN OCCURRENCE AND THE LANDSCAPE IN NORTHWEST OHIO

Introduction

One of the primary concerns in ecosystem and community management, and subsequently ecosystem restoration, is understanding the distribution of species on the landscape (Araújo & Guisan, 2006; Tremlová & Münzbergová, 2007). Poor understanding of the location of populations, especially when managing species that have limited dispersal ability (i.e., Anurans), could lead to mismanagement and negative consequences for that species. This issue is compounded when managing an ecosystem, in which organisms have different fundamental and realized niches, respond to external pressures differently, and have differing life histories (Burgman et al., 2005). Recent research has identified the study of communities of amphibians (as opposed to single species) and effects of land use change as a research priority (Campbell Grant, 2023). Further, management organizations with limited resources, including the financial means or personnel to attempt wide-ranging surveys for species, may have difficulty obtaining a data set that is sufficient to address their management needs; these needs may require considerable data regarding habitat suitability and species habitat preference (Field et al., 2005, Moilanen et al., 2005). The collective work regarding this field utilizes Ecological Niche Modeling (ENM), otherwise known as Species Distribution Modeling (SDM), the most reliable form of which has been shown to be Maximum Entropy Modeling (Maxent) (Ahmadi et al., 2023; Elith et al. 2011, Merow et al., 2013).

Maxent can utilize data sets incorporating numerous spatial variables of interest and presence data and use it to predict species occurrence on the landscape (Elith et al., 2011; Warren & Seifer, 2011). This prediction is made utilizing data points where a species was positively identified (occurrence) and relating it to the values of spatial variables at that location and extrapolating those values elsewhere within the landscape map (Elith et al., 2011). Maxent utilizes relative entropy between the density probability of two factors (environmental and occurrence data) to achieve these goals (Chang et al., 2022). The principle of maximum entropy posits that the actual state of knowledge described by the best probability distribution is the one of maximum entropy, where entropy is understood as a measure of uncertainty (Jaynes, 1957). The use of Maxent modeling has become extremely popular particularly because of the high accuracy of Maxent predictive models compared to other models (Elith et al., 2011; Ahmadi et al., 2023) as well as Maxent's robustness to small sample size (Pearson et al., 2007), providing a powerful tool to model species distribution.

Using this method, studies that have collected presence-only data and spatial data at each site across a landscape can use Maxent to predict the occurrence of the same species at other locations within their study area, which can then be used to assess habitat suitability for the species in the study area (Elith et al., 2011). In addition, these maps can be used to provide evidence to managers as places to search for additional populations/sub-populations of their study species (Warren & Seifert, 2011). Maxent can therefore provide an opportunity to both analyze datasets of species occurrence and predict habitat suitability, allowing managers to better understand species' needs on the landscape, while also helping search for additional populations, which may assist in understanding population dynamics and change in distribution (Elith et al., 2011).

As several Anuran species within the Northwest Ohio area are declining at some point across their range, and a comprehensive evaluation of populations for each species within the area has not been established, the Anuran community of this area is likely to benefit from

Maxent modeling. Recent Anuran-oriented studies in this region are few (Furlong, 2016; Martin, 2015). Combined with the comprehensive Anuran calling surveys conducted (see Chapters 1 and 2), and the numerous spatial data collected on the ground and available through remote sensing, Maxent can provide a great deal of additional information regarding the Anuran community in our study area, while controlling for bias in occurrence data. Thus, the goal of this chapter was to utilize these models to generate habitat suitability maps to predict occurrence of species of interest and nuisance species in the area and identify the variables that are most important to predicting species occurrence. Further, we sought to indicate the potential location of additional populations, which managers can use to address their conservation needs (Giovannini et al., 2014; Smith et al., 2012; Warren & Seifert, 2011).

Numerous studies have utilized Maxent to explore species distribution modeling since the development of the program. Previous studies have included those that seek to provide data regarding the distribution of endangered plants (Smith et al., 2012); assess avian habitat decline (Wang et al., 2020); and evaluate the distribution of rare mammals (Perkins-Taylor & Frey, 2020). Other studies have examined amphibians for Maxent studies, utilizing the tool to identify priority conservation areas (Giovannini et al., 2014), locate future survey areas (Groff et al., 2014) and address niche suitability across a large portion of an entire country (Rais et al., 2023). To our knowledge, no study as of this writing has utilized Maxent to assess Anuran occurrence in our study area (Northwest Ohio) or the Southeast Michigan area that is linked to our study area via the Oak Openings Region.

Though the part of the study contained in Chapter 3 incorporates variables that predict Anuran occurrence as a tool to understand ecosystem productivity (the rate of biomass generation), it is distinctly different. While Chapters 1 and 2 focused on the local and regional effects of landcover type and related variables on community assemblage, utilization of Maxent modeling allows us to extrapolate those results to a regional scale. Maxent modeling allows for the further examination and prediction of species presence (and thus community composition) in areas that were not surveyed (Warren & Seifert, 2011). As it is impossible for ecological researchers to be everywhere at once, this can provide an extremely valuable tool to focus future research and conservation efforts. The consistent habitat changes over the Oak Openings Region (OOR) and the expected intent to expand investment in the Toledo Metropolitan Area (TMA) over the coming decades help highlight the importance of creating these models, and how these coming changes may affect Anuran species, and thus, ecosystem productivity (Barkholz, 2024). The goal of this chapter was to build these models to predict areas of high species richness and extrapolate those results to other areas that may be suitable for greater richness. Further, we sought to predict the occurrence of individual species, so that managers have the necessary spatial context for management of those species in the region.

Methods

Data collected for the previous chapters were extrapolated across the study area in the larger Oak Openings Region. Also included were continuous biotic and abiotic variables that were associated with habitat suitability and breeding of Anuran based on previous analyses (chapters 1 and 2) to explain variability in species presence and community richness. These extrapolated variables were used to create models of current high occupancy probability areas through Maximum Entropy Species Distribution Modeling (Maxent, version 3.4.4). These models were developed for individual species of interest, as well as to explain the variability in site species richness.

Continuous variables including landcover class percentage and type, normalized difference vegetation index (NDVI, average early season and average late season) and percentage of impervious surface were among the variables utilized to develop models. Landcover type was identified based on the Oak Openings Region landcover map developed in Martin & Root (2020). The 15 landcover classes included: turf/pasture, wet prairie, residential/mixed use, perennial ponds, upland savanna, wet shrubland, swamp forest, upland coniferous forest, upland deciduous forest, floodplain forest, sand barrens, Eurasian meadow, upland prairie, urban, and cropland.

NDVI indices were calculated in ArcGIS (10.8.2) from Landsat-8 imagery for the study area (found in Path 20, Row 31), from the U.S. Geological Survey. Imagery was collected for three time periods per survey year, spring (March/April), summer (June/July) and fall (October/November) and combined across all three study years (2021-2023) as averages using the composite band tool. 'Early' and 'Late' NDVI measures also were combined using the raster calculator to create an average NDVI measure of the March/April and June/July NDVI layers, respectively. We also combined the early and late measurements within years to obtain an average for 2021, 2022 and 2023. NDVI was calculated in ESRI ArcGIS version 10.8.2.The full list of variables can be found in Table 3.1.

All environmental layers featured a 30x30m resolution and were clipped to an expanded study area (encompassing much of southeast Michigan, where the Oak Openings Region continues) in ArcGIS and converted to ASCII files prior Maxent analysis (Merow et al., 2013).

Model Parameters

To assess correlation among variables, we utilized a Spearman correlation matrix with our 21 environmental variables, as including correlated model parameters can lead to overfitting

(Elith et al., 2011; Low et al., 2021; Zhu & Qiao, 2016). Our cutoff for Pearson correlation was 0.70 to prevent over removal of variables, as previous analysis (Chapter 1) has shown the study system is complex, and we wanted to retain as many variables as was feasible. This led to the removal of average annual NDVI in all three years, for a total of 18 remaining variables.

Model Construction

Maxent contains many selectable options for the user to specify model parameters, which can heavily influence the outcome of the models (Lissovsky & Dudov 2021; Warren & Seifert 2011). With this in mind, we utilized ENMevaluate based on Low et al. (2021), Muscarella et al. (2014), Rais et al. (2023), and Sorbe et al. (2023), with a particular emphasis allowing the model to assess the system in the most biologically relevant way to the study species, while avoiding overfitting. Though Maxent is a robust tool, these methods can help prevent sampling bias and spatial autocorrelation.

We utilized ENMevaluate (v. 2.0.4) in R (v. 3.3.3) to assess the Maxent parameters best suited for model tune up. ENMevaluate found the model with the lowest ΔAIC value to be the model that utilized only linear features, with a regularization multiplier of 5. We utilized bootstrapped replication with 10 replicates and a cloglog output format, with 10 replicates and a 30% random test percentage. We constructed models with both cross-validated and bootstrapped replications to test for model fit, but only cross-validated models are reported here. The maximum number of background points was set to 10,000 because of the large number of points we had (>9 million) in the study region. We utilized random seeding and did not have Maxent remove duplicate records, to intentionally give weight to sites that repeatedly had higher diversity/more frequent species encounters, and vice versa for sites with less frequent records. We also constructed a bias file combining occurrence data and environmental data to help

Maxent control for sample site selection, which allowed the program to adjust for the assumption that occurrences were obtained from sites easier to sample than points where they did not occur (Rais et al., 2023; Sorbe et al., 2023). Finally, we had Maxent create response curves based on the models and use jackknifing to measure variable importance.

We developed models for each of the ten individual species, as well as total species richness, models for species richness excluding Disturbance Tolerant species, and models for richness of non-Disturbance Tolerant species (excluding *L. catesbeianus*, *A. americanus*, *L. clamitans*).

Model Evaluation

We checked model fit by using Area Under the Curve (AUC, value of the area underneath the receiver operator curve, generated by Maxent), and comparing bootstrapped and cross-validated models. Per Hanley & McNeil (1982) and Carter et al. (2016), we utilized the following criteria for AUC: models containing values greater than 0.9 indicated an excellent mode, 0.8-0.9 indicated satisfactory, 0.7-0.8 indicated average, 0.6-0.7 was insufficient, and below 0.6 was poor (Araújo & Guisan, 2006). Cross-validated models that differed more than 0.1 AUC from their bootstrapped counterparts were discarded for failure of consistent fit, as were models that did not contain AUC values greater than 0.6 (Araújo & Guisan, 2006; Carter et al., 2016; Hanley & McNeil, 1982). We assessed variable importance to individual models, as permutation importance, which explicitly evaluates the importance on the final model regardless of when it is entered (Phillips, 2017).

Results

Total Richness

Our primary goal in utilizing Maxent was to assess the importance of specific environmental parameters to higher species richness at our sites. We also sought to locate other areas in the study region that may be already suitable, or less suitable than expected, for diverse Anuran communities, to focus conservation efforts. To those ends, we developed a Maxent model with the aforementioned environmental variables, using them to predict occurrence of greater richness (1-8 species, no one site had nine or 10). The model containing the highest potential richness value (eight species) (AUC = 0.821, +/- 0.014) included cropland (negative association with higher richness), percentage of impervious surface (negative), average late season NDVI (positive), Eurasian meadow (positive) and Urban landcover (negative). The importance permutation of each of the predictor variables is shown in Table 3.2. Impervious surface and cropland were the main contributing variables to the model, with these two variables contributing 80.7% and 13.0% of the value to the model, respectively. Each of the remaining variables contributed less than 6%. The probability of greater species richness occurrence was predicted to decrease with the proximity to anthropogenic cover class areas (Figures 3.1 through 3.4). Response curves showed no other parameters had a strong negative or positive effect on the occurrence of eight species.

The model predicting the occurrence of seven species, however, had better fit (AUC = 0.895, +/- 0.031, Table 3.3) and a seemingly more informative map and parameter set. This map (Figure 4.4) contained 11 parameters, with four contributing more than 6% to the model (impervious surface, 51.6%; cropland, 17.7%; urban, 13.5%; upland prairie, 6.2%). Upland prairie was the only one of the four that increased the probability of occurrence with higher value, possibly due to its association with nearby water holding classes. Floodplain forest and swamp forest both increased the probability of predicting the occurrence of seven species and were the only variables in the model with that association, despite their lower contribution to the model. As in the model for eight species, the model predicting the occurrence of any seven species indicated a general decrease in the probability of predicting high richness, with closer proximity to the TMA urban area.

Specialist Species Richness

When modeling species richness explicitly excluding the three species we classified as 'disturbance tolerant,' (DT) model performance generally improved and became more precise. Though a select few sites had six non-disturbance tolerant species identified, the lack of training data led Maxent to limit analysis to one through five species. In the model for five species (i.e., the greatest richness of specialist species), the model (AUC = 0.894, +/- 0.044, Table 3.4) featured higher predictive occurrence values than the models including DT species. This model contained 10 of the potential 18 models parameters, though only impervious surface (60.0%) cropland (15.9%) and swamp forest (11.0%) contributed more than 5% to the model. Permutation importance for this model can be found in Table 3.4 and model map can be found in Figure 3.5. Swamp Forest and late season NDVI were the only parameters that increased the probability of predicting five species, with their increase. These results mirror several of our results from Chapter 1, regarding the value of swamp forest and greater NDVI. All other parameters had no effect or had a negative influence.

Similar to the relationship between the models for both eight and seven species (when including DT), the model for any four specialist species was better fit and more specific than the model for any five species. The model for four species (AUC = 0.917, +/- 0.050, Table 3.5) included 13 of the potential 18 variables, however once again, the contribution was dominated by anthropogenic land use cover classes, with only residential (49.8%), cropland (21.5%) and urban (8.6%) contributing more than 6% to the model. Maps and permutation importance for the four species model can be found in Table 3.5 and Figure 3.6. Once again, floodplain forest and swamp forest were the only parameters that, when increased predicted a higher probability of the occurrence of four species.

Species of Concern

We also sought to identify habitat suitability for specific species in our study area. We were primarily interested in our three species of interest (*L. sylvaticus, A. blanchardi, L. pipiens*) as well as *P. crucifer*, a species that has been established as a specific indicator of habitat quality (Price et al., 2007). Secondarily, we were interested in *L. catesbeianus*, which in many areas is regarded as a nuisance species (Brys et al., 2023; Kats & Ferrar, 2003). These five species are the results we report here in the interest of brevity, though we did obtain model results for the remaining five species.

The best model for *A. blanchardi* (AUC = 0.862, +/- 0.020, Table 3.6) included 13 of the potential 18 variables, with cropland (24.5%), late season NDVI (17.8%), upland prairie (17.1%), urban (15.3%), impervious surface (7.8%) and Eurasian meadow (7.5%) all contributing more than 5% towards the model. The map and permutation importance for *A. blanchardi* occurrence can be found in Table 3.6 and Figure 3.7 (upper). Regardless of model importance, amount of urban, and upland prairie were the factors that predicted *A. blanchardi* presence. Occurrence was negatively associated with cropland, Eurasian meadow, floodplain forest, sand barrens, swamp forest, upland deciduous forest, upland savanna, and wet prairie.

The best model for *L. pipiens* (AUC = 0.877 +/- 0.027, Table 3.7) included 16 of 18 potential variables, with five contributing more than 5% to the model. These were wet prairie (30.3%), residential (19.2%), upland prairie (18.4%), cropland (12.7%), and impervious surface (5.4%), and can be seen in Table 3.7 and Figure 3.7 (mid). The probability of *L. pipiens* occurrence decreased when any of cropland, Eurasian meadow, perennial ponds, residential, sand barrens, upland coniferous forest, upland deciduous, upland savanna, urban, or impervious surface increased. The probability of occurrence increased when wet prairie, upland prairie, or late season NDVI increased.

In the best model for *P. triseriata* (AUC = 0.887, +/- 0.015, Table 3.8), 16 of 18 possible variables were included in the model, again with five parameters contributing more than 5%. These were residential (35.2%), impervious surface (16.9%), cropland (15.5%), swamp forest (8.3%) and Eurasian meadow (5.4%) and can be found in Table 3.8 and Figure 3.7 (lower). Occurrence of *P. triseriata* decreased in likelihood with the increase of cropland, Eurasian meadow, residential, sand barrens, upland coniferous, upland savanna, and impervious surface. The occurrence increased in likelihood when late season NDVI, floodplain forest, swamp forest, upland prairie, or wet prairie increased.

In the model for *P. crucifer*, which is regarded as an indicator of habitat quality, the best model (AUC = 0.890, +/- 0.014, Table 3.9), featured 15 of 18 variables, with five contributing 5% or more. Impervious surface (27.5%), residential (25.4%), cropland (16.7%), urban (8.0%) and Eurasian meadow (5.0%) all factored into the top model for the species, and permutation importance can be found in Table 3.9 and Figure 3.8 (lower). Response curves indicated that the species occurrence responded negatively to an increase in all five parameters. Similar response curves indicated that *P. crucifer* occurrence responded positively to wet prairie, upland prairie, upland deciduous forest, swamp forest, floodplain forest, and late season NDVI. However, only one of these had a permutation importance over 5% (late NDVI, 13.2%).

The model for *L. catesbeianus*, a nuisance and even invasive species in some areas (AUC = 0.818, +/- 0.022, Table 3.10, Figure 3.8 (upper)), showed even marginal suitability in an overwhelming majority of the study area. The best model contained 15 of the possible 18 parameters, with seven contributing more than 5% to the model, including upland prairie (40.8%), Eurasian meadow (11.2%), floodplain forest (11.0%), wet prairie (7.4%), cropland (6.8%), late season NDVI (6.7%) and residential (5.6%). Response curves for these parameters indicated a negative response of occurrence to increasing cropland, Eurasian meadow, residential, and urban area, and a positive response in occurrence to wet prairie, floodplain forest, and late season NDVI.

Discussion

Species richness, as well as the occurrence of specific species, were well predicted by landcover type. Repeatedly, both among individual species and the whole community models, anthropogenic landcover types (residential, urban, cropland) were the variables that most consistently influenced the probability of presence, almost universally negatively. This was not surprising, as previous research has shown negative impact on amphibian habitat when creating cropland (Gustafson & Newman, 2016). Agricultural efforts have also been shown to alter the community composition of Anurans (Hromada et al., 2021). Furthermore, residential, and urban areas have repeatedly been shown to influence amphibian communities and alter habitat function in numerous ways that affect Anuran persistence (Forman & Alexander, 1998; Hamer & McDonnell, 2008; Sievers et al., 2019). While cover classes such as swamp forest and floodplain forest repeatedly were included in models, their model contributions were low compared to the anthropogenic landcover types. Maps for each of the models show the highest probability of occurrence (for both species and communities) are in currently protected areas that feature one or more of the water-holding cover classes (floodplain forest, swamp forest, perennial ponds, wet prairie, wet shrubland). These areas are also primarily outside the Toledo city center and have a lower amount of impervious surface.

Furthermore, all models identified a strong demarcation in suitable habitat between the Ohio/Michigan border, which the city of Toledo abuts. On the Michigan side of the border, our models identified a great deal of suitable habitat just north of Toledo. As species do not follow anthropogenic cultural demarcations, we suggest managers work closely with those from other states to ensure the maintenance of conservation strongholds to prevent urban sprawl from eliminating habitats in adjacent areas. From these models, we can suggest that anthropogenic alteration of landcover type towards cropland and impervious surface-based cover classes lead to a decrease in Anuran species richness. These results are in alignment with our results from Chapter 1, which found that anthropogenic landcover classes featured in lower species richness environments, while greater NDVI and amounts of swamp forest featured in environments with greater species richness.

In the models predicting species richness, anthropogenic land classes made up 92.0% and 77.5% of permutation importance in the models for any eight and any seven species, respectively. These results were similar to the models that specifically modeled species richness of specialist species. Cropland, residential, and urban landcover areas made up 84.3% of the contribution to the best model for the model of any four specialist species, while cropland, impervious surface, and swamp forest comprised 90.8% of the model contribution for the five specialist species model. Swamp forest and floodplain forest were the only cover class types that appeared to have a positive effect on the probability of predicting higher species richness, in the most cases, which has been shown in previous research (Hörnberg et al., 1998; Pearlstine et al., 2002). Wet prairie additionally had the same effect, albeit in more limited cases. These models provide evidence that the quality of habitat in our study area, as well as the manipulation of that habitat for human needs, has a direct impact on the species richness of Anurans, a well-known indicator taxon (Price et al., 2007; Waddle, 2006). The consistent and overwhelming effects of the anthropogenic cultural classes on these models lend more weight to the growing body of literature that suggest human destruction of habitat is causing extirpation and extinction of many species (Banks-Leite et al., 2020; Jantz et al., 2015; Liu et al., 2016; Powers & Jetz, 2019; Segan et al., 2016).

The stronger model fit and occurrence maps exhibiting higher suitability among the models for seven species (full community models) and four species (specialists only) than the eight and five models, respectively, demonstrate the difficulty in modeling for a full community. This may be because of the reduction in species, which allowed the model to take fewer environmental parameters into consideration. Though there is considerable niche overlap between many of the species in the area, the requirements for sustaining eight or more species in a small area may be too great to expect in more than a few isolated patches throughout the study area. This is not a surprise, as some species (*A. blanchardi*) were already believed to be only inhabiting sites adjacent to the largest rivers, and other occurrence maps (*L. sylvaticus*) show limited areas of high suitability in the region, compared to other species. It is also possible that the matrix and arrangement of available habitat may prove beneficial when planning Anuran conservation in the area. Though our study area was too large to investigate the cell-by-cell matrix of each site, future projects should examine the value of a diverse matrix of habitats within various groupings. Some of the most diverse habitats in our study area contained wet prairie, swamp forest, and wet shrubland within close proximity to each other. When exploring habitat restoration and creation of preserves, land managers should explore a diverse matrix of habitats that can sustain a diverse group of species of Anurans, as opposed to one singular habitat type. In other words, to conserve Anurans attention should be paid to preserving a variety of habitats, especially in landscapes with a lot of human modified habitats. Additionally, our models identified a string of the most suitable habitat in the Oak Openings Region of Ohio and Michigan. Especially because of the limited dispersal ability of amphibians, it is essential that managers maintain these strongholds of restoration that can be used as habitat steppingstones by populations. The loss of sufficient connectivity between these suitable habitats may result in population extirpation or genetic consequences.

Within specific species models, as expected, because of niche differences, models differed in what variables predicted individual species presence. Cropland, Eurasian meadow, urban and residential landcover types had a negative effect on the presence of four of the five species we examined closely, including *L. catesbeianus*. Even within a species regarded as nuisance because of its tolerance for human development, it appears anthropogenic landcover type can detrimentally affect presence of a species, which has not always been shown in the literature (Hromada et al., 2021; Porej & Hetherington, 2005). Among the five individual species modeled, the negative effects of anthropogenic and non-water holding classes appeared to be the most pronounced on *P. crucifer*. This was expected, because of the species' previous status as an indicator of wetland quality and functionality (Knutson et al., 2000; Price et al., 2007). We find support for this classification for *P. crucifer* as an indicator species, based on the perceived acute response to habitats with lower productivity and suitability.

If the Oak Openings Region continues to undergo landcover changes as those found between Schetter and Root (2011) and Martin and Root (2020) analyses (map years 2006-2016), our models suggest that significant species richness will be lost, and with it, the more highly functional habitat that may support such species. Residential area increased approximately 5% over the 10 years between studies (Martin & Root, 2020), and this class was a frequent contributor to an estimated lack of occurrences in certain areas, particularly in the city suburbs and commercial areas. We found that both an increase in NDVI and swamp forest can be valuable to harboring greater species richness of Anurans, and we suggest that managers take these findings into consideration when preparing to restore wetland areas.

Several studies have utilized Maxent for broad suites of taxa, and several have utilized the method to model one species or over expansive regions, such as countries or large states (Kidov & Litvinchuk, 2021; Pesarakloo et al., 2020; Westwood et al., 2020). Our study provides evidence of the potential for regionally based SDMs utilizing locally relevant data and can provide direction to future surveys for relevant species, as in Groff et al. (2014). Further, because of the recent trend towards including suites of species in special distribution modeling (Bellamy et al., 2013; Rais et al., 2023), the interaction between species can be considered a factor; predation and competition have been identified as factors that can alter distribution models (Tompkins & Veltman, 2006; Trainor et al., 2014). Added benefits of this modeling approach are the flexibility to rebuild the models with updated data, to examine potential impacts of future conditions (e.g., climate change), and adjust to the particular species and/or target of interest. As a result of utilizing essentially linear approximate distributions, Maxent is able to simplify training, and thus improve efficiency (Phillips et al., 2006; Phillips & Dudik, 2008). Further, as Maxent does not rely on true absence data, it can be utilized with a wide variety of data sets. This method can provide extraordinary value to local land managers seeking to direct their resources more effectively, especially when targeting the effects of land use change, which has been considered a more urgent threat than global climate change (Dale, 1997).

Our study does not come without caveats. Because we are attempting to predict suitable habitat, we are making a clear assumption that presence implies suitability, and greater quality habitat. However, given the ecologically delicate taxa that is the focus of our study, we feel comfortable with this assumption. We are also comfortable assuming that fewer encounters (with which we weighted models via duplicate records), and ecologically inappropriate habitat do indeed exhibit differences in habitat quality. We also did not have the resources available to separately subsample areas Maxent deemed as potential for high occupancy (i.e., field evaluate) following this analysis to provide an independent test of the model, which would have ultimately strengthened our results. Future efforts should focus on both closing this gap and addressing the potential matrix configuration (and distance) insofar as richness and species presence is concerned. Despite the limitations, this study provides a flexible tool to address conservation questions about any number of taxa and locations, at almost any scale. When modeling species presence, the use of remote sensing data, especially, can provide valuable insight that is difficult to capture in other measures of data on the ground. These methods serve as a productive means for assessing species presence, inferring habitat suitability, and focusing conservation efforts and future surveys.

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CHAPTER IV: UNDERSTANDING THE VIEWS OF LOCAL RESIDENTS ON WILDLIFE CONSERVATION AND THE ENVIRONMENT

Introduction

Global climate change and habitat loss have been described as the greatest threats facing biodiversity over the 21st century (Davison et al., 2021; Segan et al., 2016; Wilson, 1991). Extinction rates already exceed expected background rates, and the continued growth of the human population may only fuel worsening climate change and habitat loss (De Vos et al., 2015; Ray & Ray, 2011). The increasing demand of modern infrastructure and crop growth under this population increase, including in the United States, may bring habitat protection to a crisis. As a result of these multiple stressors, it is essential for land and wildlife managers to not only understand the needs of their study organisms, but also the attitudes of their local communities towards wildlife and land conservation if they are to effectively execute necessary conservation projects.

Just 16.64% of land is protected globally, with just 12% of land in the United States protected (UNEP-WCMC & IUCN, 2020; USGS, 2021). Globally, these protected areas also have been shown to harbor greater biodiversity, with richness and abundance 10.6% and 14.5% higher inside protected areas, than outside them, respectively (Gray et al., 2016). As species become threatened, and population numbers decline, protecting areas where species can thrive is the most effective strategy to conserve them (Asaad et al., 2017, Rodrigues et al., 2004).

Additionally, these protected areas, such as parks, preserves, and other green spaces, and wildlife, can provide numerous benefits to local human communities (Daily, 1997; Kilpatrick et al., 2017). Depending on their specific environment, parks and preserves can contribute to water filtration, improvements in air quality, flood prevention, soil erosion prevention, and help modulate temperature (Daily, 1997; Elmqvist et al., 2015). These parks and preserves can also provide mental benefits (forest bathing, etc.) as well as economic benefits (tourism, recreation, jobs) (Elmqvist et al., 2015; Laterra et al., 2019; Wen et al., 2019). Previous research has demonstrated that a large portion of local communities are not aware of the value or potential benefits of these ecosystem services (Norgaard, 2010; Thompson et al., 2016). To that end, it is necessary to ensure local communities are educated on the potential local issues that stem from anthropogenic land change. It is equally important that, if these effects are to be mitigated for both humans and wildlife, that managers understand the levels of local support their constituents have for steps necessary to improve these parks/preserves. While many studies have evaluated a Willingness To Pay (WTP) of park/preserve patrons in numerous places across the world, few studies to our knowledge have evaluated the upshot of these efforts, rather than the potential increase of existing fees (Baral et al., 2008; Ressurreição et al., 2012; Thur, 2010; Uyarra et al., 2010; Wang & Jia, 2012). Previous studies also have primarily focused on either a specific park preserve or an entire national parks system (Zou, 2020), and there is a need for this information on a more manageable, local, or regional scale (Baral et al., 2008; Ressurreição et al., 2012; Thur, 2010; Uyarra et al., 2010; Wang & Jia, 2012).

In the Northwest Ohio, Toledo is home to a declining urban manufacturing area, but the metropolitan area is still home to over 600,000 people, multiple universities, and several farming communities (U.S. Census Bureau, 2020a; U.S. Census Bureau, 2020b). The metropolitan area is also home to a large part of the Oak Openings Region, a unique complex of oak savanna and related ecosystems, which are home to the highest number of threatened species in the state of Ohio (Weber et al., 2016). The conflict between anthropogenic land change and natural ecosystems has led to a loss of 90% of wetlands in Ohio since European settlement, and the extinction or extirpation of 46 species, with 125 currently endangered and 52 currently threatened (Ohio Department of Natural Resources, 2022). As of this writing, approximately only 10% of land in Northwest Ohio is protected as a park or preserve (Martin & Root, 2020). The combination of these factors requires concerted effort to expand high quality habitat for native species (e.g., more parks/preserves/green spaces) or improve the habitat quality of existing network of parks or preserves. These efforts can also improve human health, both mentally and physically, which has declined in numerous measures over the last several decades (Elmqvist et al., 2015, Muennig et al., 2018).

However, especially in a decentralized political system like the United States, these habitat expansion and restoration efforts must come from the local level, and with the support of local communities. Land and species conservation projects without local community support may also be more likely to fail (Brooks et al., 2012; Brooks et al., 2013; Catalano et al., 2019). A large body of literature has provided evidence of the importance of integrating local communities into control of conservation action because of their knowledge of local resources and conditions, as well as an increased incentive in sustainability (Brooks et al., 2013). Additionally, several studies in differing geographical locations have provided evidence that education on environmental topics can improve knowledge and attitudes on wildlife and habitat (Freund et al., 2019; Sousa et al., 2016).

Accordingly, land and species managers must learn to work effectively with their local communities to ensure understanding of the importance of the project, understand the level of local support for their proposed projects, and most importantly, secure that support, if possible, while integrating the community in control over the project.

We sought to measure the level of local support for similar projects in Northwest Ohio, specifically around the Toledo metropolitan area, to provide information on local support to land managers working in the area and develop a template for addressing these questions at the local scale in other areas. We created a survey based around five key lines of inquiry: 1. How do adult Northwest Ohioans feel about conservation efforts of wildlife, land, and water? 2. What percentage of respondents support alteration of preserves/parks for conservation purposes? 3. Do respondents support actions that require personal sacrifice for these issues? (e.g., raise taxes, volunteer, vote, donate) 4. Do certain demographics support these potential changes more than others? And finally, 5. Does exposure to a short presentation on the importance of Anurans and preserves/parks alter responses to any of these questions?

Utilizing these surveys, we addressed specific questions about public support for conservation projects on a local scale, and the actions they would be willing to take to help these projects. We also assessed if an informative educational presentation increased support for these questions. We expected to see limited support for actions that required money or time from respondents, while seeing overall support for the conservation of land, water, and wildlife. We also expected that the informative presentation would increase support for most questions, but that questions involving fishing/boating, the need to retain land for human use, or questions asking about money would remain the same across surveys.

Methods

Study Population

Because our research questions focused on local adults and their support for local environmental action, respondents were required to reside in our study area, comprised of the city of Toledo (Lucas County) and the four immediate adjacent counties (Fulton, Henry, Ottawa, Wood). Per the 2020 U.S. Census, the five study counties have a combined population of 668,916, with 63.8% of that population residing in Lucas County. The five counties are variable in racial, educational, and political (based on 2020 election results per each county's Board of Elections) composition, as was the amount of land used for parks/preserves within each county (U.S. Census Bureau, 2020a; U.S. Census Bureau, 2020b). Table 1 shows the comparison of the study area counties in these variable demographics, compared to our representative sample demographics (detailed below).

Survey Development

Following a design similar to that found in the literature of surveying local stakeholders (Farmer et al., 2011; Greiner et al., 2015; Karanth et al., 2008; Nepal & Spiteri, 2011; Sousa et al., 2016; Tisdell et al., 2007), we wrote a 30-question Likert scale survey (with seven additional demographic questions and two open-ended questions) based around five key lines of inquiry. Respondents reflected on how they felt about the outdoors/wildlife, how they felt about the environmental movement, if they felt greater protections of wildlife/parks were warranted, and if they supported specific actions proposed in the survey to address these issues. Respondents were asked to answer if they agreed with the statement before them, on a scale from Strongly Disagree to Strongly Agree (1 to 5). These Likert scale questions were followed by seven demographic questions about their background and identity. Two open ended questions were added as attention checks and protection against robotic automated responses ("bots"). The survey was designed with paired positive and negative questions to control for potential contradictory answers to ensure respondents were reading and answering the questions honestly (henceforth 'positively phrased' or 'negatively phrased'). The complete survey can be found in Table S4.2.

To evaluate whether exposure to a short presentation on the importance of Anurans and preserves/parks altered responses, we recorded a 25-minute presentation on the value of Anurans to the local ecosystem and importance of parks/preserves to both humans and wildlife. The intent was to test the effect of conservation education on survey responses (Freund et al., 2019; Sousa et al., 2016).

Survey Deployment and Presentation

Surveys were administered digitally via Qualtrics or in-person at a local presentation in May 2022. We advertised surveys via social media (Facebook, Twitter, Instagram, and Nextdoor), where the first 155 respondents would receive a $10 Amazon gift card. Respondents were able to follow a link on these advertisements, where they were prompted with a consent form (requiring respondents to be 18 or older), followed by a question of which county they reside in, to ensure they were compliant with the terms of the survey. Those that were compliant with those qualifications then answered the 30-question Likert scale surveys, followed by two open-ended questions, and six more demographic questions. This preliminary survey constituted the “before presentation” or “before” survey.

Following the initial Likert-scale survey, respondents watched a 25-minute presentation about the importance of Anurans to our local ecosystems (e.g., indicator taxa), the researchers’ personal work with these organisms, and the value of preserves/parks to humans. Immediately following the presentation, the respondents then completed the same survey, with questions in a new order. Respondents created a 5-digit code to facilitate linking pre and post responses for analysis. These constituted the “after presentation” or “after” surveys. We sent eligible respondents their compensation for the study in July 2022 via email, but respondents were otherwise anonymous. We gave in-person respondents (less than 2% of all responses) an identical presentation in May 2022, which only differed in that both before and after surveys were submitted on paper.

'Bot' deterrent/removal procedures

We ran a 16-point check to ensure responses were human and from the study area, rather than robotic responses programmed to claim compensation for completing the survey. These checks included those such as impossibly quick completion of the survey (finishing in less than one minute despite a 25-minute video), completing the survey with an improbable number of clicks on the page (e.g., clicking as few as zero times on the page), nonsensical answers in any of the open-ended questions, and failure to select the proper answer in attention check questions. The full list of 16-point checks can be found in Table S4.1. Surveys were disqualified as bots if they received five or more points of the 16-point check, did not receive compensation, and were not included in data analysis.

Data Analysis

We divided Likert-scale questions from the survey into values-based questions (abstractions that would be rooted in the feelings of an individual, 16 of 30 total questions), and action-based questions (direct actions, or what they would do to improve the environment, 13 of 30). We designed this division to assess local support for actionable steps to restore habitat, but also to gauge the relationship between values and action-based questions, as well as with similar questions from the same category. One question was regarded as knowledge-based and was not grouped with the others for analysis. A question was considered supported by a respondent if they answered a 4 (agree) or 5 (strongly agree), which were averaged across the survey respondents to gauge total support for a question. Questions were phrased either positively (meaning we expected respondents to agree or strongly agree) or negatively (we expected respondents to disagree or strongly disagree). Examples for each type of question can be found in Table 2. We also grouped data to analyze the differences in response between different demographic or cultural groups.

Demographic questions were also coded numerically, to be treated as either discrete or categorical data, depending on the question (questions such as the respondent's gender were coded categorically, while age and educational background were coded as discrete data). We reverse-coded negative questions (i.e., strongly agree on a negative question received a 1, and vice versa) and summed scores for one respondent on action-based questions, for a scale of how amenable a participant was to conservation action as a whole. We referred to this summed score as a respondent's Conservation Action Score (CAS). This allowed us to compare differences between demographic groups across all action questions. Higher total summed scores would indicate more overall support for conservation action.

Initially, we utilized cross-correlation matrices with Pearson correlation coefficients to assess what relationship, if any, responses to certain questions had to one another. Following Levene's tests to confirm normal variance, we ran ANOVAs using JMP statistical software (v. 17) to test for significant differences between demographic groups and question feedback. Additionally, we ran Paired T-tests (JMP v. 17) to test for significant differences between responses before and after the presentation. These tests were also used to assess significant differences in the CAS between groups as well as before and after the presentation. We utilized Cronbach's α to test for consistency and reliability between survey results before and after the presentation (Cronbach & Meehl, 1955).

Results

Demographic description

At the completion of the survey period and bot removal, we received 300 digital surveys and 4 in-person surveys for a total of 304 surveys completed. Of our 304 survey respondents, 56.1% self-identified as women, with 41.2% identifying as men, though 13.5% of total (41 respondents) declined to identify their gender in the survey. Just one individual (0.3%) identified as non-binary or another gender. Approximately 39.5% of respondents were in the age 18-29 group, with a large 48.0% in the 30-44 age group. A small minority, 10.2%, were in the 45-64 group, with a very small number (~2.0%) in the 65+ age group.

Respondents were disproportionately female (56.1%), and disproportionately younger (87.5% under age 45). Respondents were also overwhelmingly likely to have at least one post-secondary degree, with 70.1% having at least a bachelor's degree, despite the same group comprising only 27.5% of the potential study population area. Breakdown of our demographic groups, and their comparison to the study area population, can be found in Figure 1.

Consistency of Results

The Coefficient α within the 'before' surveys was 0.926, which showed evidence of high internal consistency reliability (α=0.926).Value questions displayed a similar reliability (α = 0.895) relative to the action questions (α = 0.838). The correlation between "Value" and "Action" was significant (r=0.787, p <0.001). The correlations show similar items had stronger correlations than dissimilar items. For example, in the 'before' survey the strongest correlations were between questions 3 (I support creating more nature preserves) and 21 (I support new parks, designed with wildlife in mind) r=0.493; while in the 'after survey' the strongest correlation was between 3 and 12 (We should improve our parks to make them better for wildlife) *r*=0.562. Concepts measuring similar traits have been shown to have stronger correlations (Cronbach & Meehl, 1955). The two weakest correlations were between questions 2 (I would rather spend time outside than inside) and 6 (I do not feel comfortable in the outdoors) *r*=0.003, which were two dissimilar constructs, thus providing evidence of theoretical construct validity, and validating our overall results.

Full Survey Breakdown – High Levels of Overall Support

Across the full suite of respondents, 19 of 21 positively phrased questions (which denote either positive attitudes or actions towards the environment) were supported by a majority of respondents (50.1% or higher) prior to the presentation (ranging from 47.4 to 72.4% before, 46.44 to 79.39% after). The exceptions to this support for positively phrased questions were those that asserted fishing/boating access was important in a park, and support for a significant fee (entry fee, tax, levy, etc.) to use parks. Alternatively, seven of eight negatively phrased questions (which denote less favorable attitudes/actions towards the environment) were supported by a minority of respondents (49.9% or lower, data ranging from 27.0 to 36.8% before, 26.34 to 39.53% after). The lone exception was that a majority of respondents felt they hear too much about the environment. After the presentation, these results largely held, with the fishing/boating question the only positively phrased question without majority support. Both before and after the presentation, negative questions almost exclusively did not have a majority of support, most with just 30-40% support. The negative question with the greatest percentage of respondents agreeing was from those who believed they heard too much about the environment (58.9%, 64.5% after).

Demographic Differences

Responses to 11 of 13 action-based questions showed a significant relationship between a supportive response and respondents that most recently visited a park in the before survey (See Figure 2). Despite the highly significant results for many of these questions, low R^2 values suggest that being a more recent visitor to a park/reserve alone does not explain the variability in our results. There were not clear trends in support among other demographic groups. Different groups of age, county of residence, and educational demographic categories showed a significant relationship with just one to four action-based questions (of 13, select tests can be found in Table 4.3), but there was not a discernible trend between the groups or questions. What type of place a person was raised (big city, rural area, etc.) showed no significant relationship with their likelihood to support any action-based questions. In the 'after' surveys, just as in the 'before' surveys, no other group besides the most recent park/reserve visit was likely to predict a positive response in more than two of the 13 action-based questions. There was no notable trend in predicting a positive or negative response between different demographic groups.

Several demographic groups not related to a respondent's last visit to a park had a significant relationship with a positive response to at least one action-based question. However, these significant relationships did not fit a discernable trend or pattern that would demonstrate that a certain demographic group (county, age, gender, etc.) predicted a general willingness to support conservation activity. Further specific results can be seen in Table S3.

Comparing Before and After Surveys

Most positive action-based questions showed between 58-72% of support before the presentation, while most values questions had between 53-70% support. However, after the presentation, 11 of the 13 action-based questions showed significant increase in support from the first survey (see Figure 2 for detailed test results), with increases in support ranging from 1.60% to 11.01% of respondents, and most changing between five and 10 percent. Donating to wildlife organizations and a question about voting patterns were the only action-based questions that did not change significantly after the presentation. We found values-based questions showed less change than action-based questions, with most support changing between two and five percent, and only two questions receive significantly more support after the presentation: need more awareness about environmental issues ($p < 0.001$) and parks provide benefits for humans ($p = 0.004$).

Several questions received more support during the post-presentation survey but did not reach significance with an alpha value of $p<0.05$. These questions showed an increase in the number of respondents who felt they hear too much about environmental causes, and those that believed we need more parks for humans. However, the percentage of participants that said they donate to wildlife organizations (+2.05%) and vote for officials that protect land/water (0.92%) increased, though not significantly. On negatively phrased questions, those that said they would not change their property without a tax break (+0.11%) also increased slightly. In addition, both in surveys before and after the presentation, negative questions of both values and actions tended to only correlate with other negative questions of both types. These questions are important, as they exhibit the type of responses that are likely to be more difficult to change (Trevors et al., 2016).

Before the presentation, most responses to values-based questions that were directly tied to land and species management (for example; 'I am a wildlife lover', 'we should protect as much wildlife as we can,' and 'we need more awareness about environmental issues') were more likely to be correlated with responses only to some action-based questions (such as, 'I would make changes to my property to help wildlife,' 'I support new parks designed with wildlife in mind' and 'I would reduce my pesticide usage'). Those who indicated they were not comfortable outdoors or did not care if wildlife was protected responded negatively to those same action-based questions.

After the presentation, questions involving the direct use of the respondent's money, such as donating to wildlife organizations or approval of a significant fee to help support parks/preserves, still had the fewest correlations among responses with other questions. Correlations between responses to like category questions (values-values or actions-actions) as well as differing category questions (values-actions) were generally higher after the presentation than before.

Conservation Action Scores

Conservation Action Scores across all respondents were significantly higher after the presentation than before ($p < 0.0001$, r^2=0.469, F-ratio=259.76), indicating the efficacy of a short presentation on encouraging support for conservation. The respondent's last visit to a Metropark was the most likely factor to predict a higher Conservation Action Score, with significant differences both overall and between each group, both before and after the surveys (p-values ranged from <0.001 to 0.005 between groups). Additionally, before the survey, 18 to 29-year-old respondents were also significantly more likely to have a higher summed score than other age groups ($p = 0.044$, r^2=0.027). Gender, where a respondent was raised, and county of residence all did not predict a score. Finally, education level did predict a higher score both before ($p = 0.043$) and after ($p = 0.015$) the presentation. However, the differences between each individual group were not significant, though respondents with a post-secondary degree were more likely to have a higher score, on average.

Questions of Interest

We expected to find limited support for a question asking about voting preferences. Contrary to our expectation, before the presentation, this was the action-based question with the most support, with 72.4% of responses indicating positively that they vote for officials that protect land/water/wildlife. This result did not significantly change after the presentation (p = 0.249), with 73.3% of respondents responding in the affirmative, which may indicate that the presentation did not move their opinions on voting priorities. Both before and after the presentation, respondents that had visited a Metropark within the last year were significantly more likely to support this question on voting preference than other groups (p < 0.0001 before, p < 0.0001 after).

Questions regarding potential fees to use a park generally received less support than other action-based questions. 62.5% of total respondents supported a small fee to help their local parks prior to the survey, with 69.4% agreeing with the statement after the presentation. When another question asked about a "significant" fee, however, just 49.01% of respondents agreed with the question prior to the presentation, making this one of the few action-based questions that did not receive a majority of support prior to the presentation. After the presentation, 55.6% of respondents agreed with support for a significant fee.

Furthermore, we found that the action-based questions that were more passive, received on average 5.40% more support before the presentation than questions that asked if respondents would personally take a direct action (p = 0.002). This gap grew to 8.12% after the presentation, as support for passive actions increased from 5.89 to 11.09%, while support for several of the more active questions stagnated, with some increasing as little as 0.92%. After the presentation, the difference between passive and active action questions was also significant, with passive questions receiving significantly more support ($p = 0.002$).

Discussion

Respondents to our survey were disproportionately female and younger, with at least one post-secondary degree. While skewed, our demographics are largely aligned with previous research that show younger generations, women, and the more highly educated are among the groups that most consistently support environmental causes, and thus would be groups that are most likely to respond to the survey (Brochado et al., 2017; Pew Research Center, 2022). While we remain confident our demographic skew did not significantly affect the analysis of the data, our project shows the limits of study designs of this kind. Though conservation managers need information on the feelings of the public to prepare environmental projects, studies that recruit exclusively online, especially on social media, may skew the sample to younger female individuals, which must be considered (Stern et al., 2018). Additionally, not only did offering Amazon gift cards invite a significant number of bot responses, conducting the surveys entirely online ensured that bot responses would become part of our study group, and have to be weeded out (Griffin et al., 2021). These same reasons may also be partly to blame for the skew towards younger individuals in our respondent pool.

We suggest that future studies maximize recruitment efforts to incorporate a representative sample of the study population, while also ensuring continued feasibility of the study. We further suggest that studies with monetary offers retain at least one in-person component, to avoid bot submissions to a survey. It may also be prudent to explore alternative categories to bin and analyze demographic responses, as the method of collection may be a limitation on the applicability of demographic data and warrants further study (Fernandez et al., 2016; Murthy et al., 2016). Similar studies have found a complicated relationship between demographic differences in response to similar questions, and so it is important to recognize that results may be regionally specific or a result of study population selection (Cooper et al., 2015, Randolph & Troy, 2008; Ressurreição et al., 2012). We also were unable to follow up with our survey respondents at a later date, to assess if changes to action question responses were long-term, or merely the result of immediate exposure to the presentation. We suggest that future studies conduct a survey at a later date (e.g., six months after) to help address this gap.

Responses to questions involving less direct action, especially value-based questions, had more correlations to one another than responses to action-based questions. Responses to more direct questions, such as those regarding volunteering for wildlife awareness campaigns or park renovation, and to questions involving the direct payment of money (fees, taxes, or donations) to wildlife causes, had the fewest significant relationships with positive responses for other questions. This lack of relationship was found in responses to both values and action-based questions. We did not find these results particularly surprising, as questions that ask respondents to sacrifice money or time were expected to be those that received the lowest support based on previous studies when compared to other questions (Samnaliev et al., 2006; Uyarra et al., 2010; Wang & Jia, 2008; Zou, 2020). While many studies show an increase in park-goer willingness to pay (WTP), we believed that the conservative political composition, more general survey respondent pool (i.e., not necessarily park-goers) and more vague nature of our survey questions would lead to lower support compared to previous studies (Barral et al., 2008; Uyarra et al., 2010; Wang & Jia, 2008). We believe these gaps can possibly be explained by a willingness for others to undertake action to address conservation issues, but a lower willingness to engage personally with these actions.

Our analysis showed a respondent's relationship to the park was the factor most likely to predict their responses. Though correlative values (R^2) remained low for most questions that had a significant relationship with more recent park visitors, we believe that this group (more frequent/recent visitors) are the group most likely to publicly support conservation projects and should be sought out for engagement when a project is planned. Additionally, the same group was the most likely to strongly agree with the statement that they talk to friends and family about the environment, meaning that this group is potentially the most likely to publicly engage with other, less frequent visitors, and help increase project support.

One of the key takeaways of our study is that exposure to a presentation on the importance of land and water protection to humans/wildlife significantly increased the number of respondents willing to support actions to improve parks/preserves in the local area. Most positive questions had 5-10% more support after the presentation than prior, with the largest increases among questions such as support for more nature preserves (13.99% change) and supporting a significant fee to help parks/preserves (13.42% change). Even before the presentation, support for action-based questions had considerable support, with 10 action-based questions receiving a majority of support, and up to 72.4% of support when asked if they would vote for officials that support protecting wildlife/land/water. These results portray a public that is willing to support changes to improve habitat for wildlife but is even more willing to support these actions if they are made aware of the importance of the actions first. Other studies on gauging public support for conservation action and the value of environmental education have found positive support for related efforts, supporting the strength of our results (van der Ploeg, 2011; Moss et al., 2015; Lute & Attari, 2016). However, local, well-communicated data stemming from public surveys likely provides significant value to land managers and policymakers (Brooks et al., 2012; Brooks et al., 2013; Theobald et al., 2000). Following these results, and stressing the importance of local context, we encourage managers to offer opportunities to engage with, educate, and also importantly, listen to concerns from, the general public.

Questions involving the use of the respondent's money directly had the lowest amount of support, as did questions about volunteering their time and energy. Action-based questions with the highest support primarily were based on the desire for someone else to carry out those actions or changes they could make to their own property/lifestyle. While the majority of support is notable, the 13.8% gap between two of these questions, support for a "small" or "significant" fee, is large enough to be notable to those exploring new taxes, levies, or fees on their parks/preserves. Responses to the two questions were not correlated in the 'before' survey, and while they were correlated in the 'after' survey ($r = 0.383$), there is clearly a disconnect between the willingness to support the two questions. There were no trends in demographic groups seemingly willing to support these fee-based questions, other than those who had visited a park most recently.

Values-based questions were far less likely to receive more support after the presentation than actionable questions were, indicating that while values do inform support for many of the actions proposed in the survey, it is more challenging to change the values of local stakeholders. As such, persuasive campaigns for support should likely focus on the actions, not the underlying values that they may be related to. Furthermore, while the changes were not significant, there was also a notable difference in responses to several questions that suggest for a very small minority, going through the presentation and survey process actually hardened their opinion against these environmental-based actions (e.g., 4.00% and 5.64% more respondents respectively saying that they support more parks for humans, and hear too much about environmental causes).

This small minority should be considered when seeking support for projects that require high amounts of public awareness and may have small margins of support. Even questions that received a negative response showed a large minority that could be mobilized to sway public opinion, most received 40-60% disagreeing with the negative premise.

Positive response to many questions, especially after exposure to the presentation, can provide insight into how much support local stakeholders (i.e., taxpaying adults and residents) have for these action-based questions, specifically. The high level of support for these questions, both before and after the presentation, can and should be used to move forward with necessary and scientifically informed restoration and management activities, where applicable. Our results indicate that dedicated time with our stakeholders, even less than 30 minutes of engagement, and regardless of their background, can increase public support for these efforts, potentially between 5-10%, which could be the difference between a measure/proposed project receiving the funding it needs, or failing.

Finally, a question about voting for officials that support land and water conservation received unexpectedly high support but did not change significantly after the presentation. This lack of significant increase is not surprising, considering the partisan nature of conservation policy and conservative political composition of our study area (Casola et al., 2022; Ehret et al., 2018). However, these percentages are substantially higher than expected, especially in a generally politically conservative area, after previous national surveys of the general voting public have shown approximately 38% of voters, on average, listing climate change as a top voting priority, with just 9% of Republican voters surveyed doing so (Pew Research Group, 2022).

Our study provides a novel insight into the detailed feelings and support for conservation action in a generally conservative area, in a state that has undergone heavy loss of habitat for manufacturing and agriculture. It is our hope that these results can be utilized to encourage local environmental projects. However, we acknowledge that the population sample we surveyed, and the highly local nature of our study may limit the transferability of specific results to other areas.

Regardless, our study can provide significant value on several overall trends. High support overall for many action-based questions, especially among the group that is more connected with parks/preserves, was clear among our survey respondents. Despite high support overall, questions that were more passive received more support than active questions, asking more of the respondents. This trend should be accounted for when both planning surveys of this type and estimating local support in the future. Younger, more-highly educated respondents are slightly more likely to support conservation action overall using the Conservation Action Score, even if these trends did not extend to individual questions. Finally, exposure to a short educational presentation on the value of conservation and local wildlife significantly increased support for both action and values-based questions, and we highly encourage increasing the number of educational opportunities for local stakeholders prior to planning an environmental project, especially ones that require the use of levies and taxes.

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CHAPTER V: CONCLUSIONS AND IMPLICATIONS

This study investigates how Anurans can be used as indicators and conduits for conservation. We also furthered an active understanding of the spatial matrix on Anuran richness. Habitat alteration and reduced quality can potentially arise from increased land use change and exposure to anthropogenic stressors. Low species richness and simplified community structure are possible consequences of habitat alteration and decrease in habitat quality. Low species richness is commonly believed to result in less stable and lower functional habitat (Naeem & Li, 1997; Tilman et al., 2012). Our study also addresses the willingness of local adults to support conservation efforts in the area. It is likely conservation efforts will be less effective across the world without local support. We utilized field surveys, modeling, and sociological surveys to address these questions, in Chapters 1-4.

Chapter 1 assessed the effects of spatial habitat structure on Anuran community composition and illustrated how ecosystem productivity, as one measure of functionality, can indicate both Anuran diversity and the relative abundance of specific species. We found that Anuran diversity dropped in suburban and urban areas, and the frequency of encounters with species that were not Disturbance Tolerant was significantly lower than in rural areas. We determined landcover composition, percentage of impervious surface, and greater values NDVI were the factors that explained the most variability in our Anuran dataset between areas of lower and higher species richness. Our research demonstrates the value in utilizing auditory surveys as measurements for exploring occurrence, and the relationship of species occurrence with spatial structure. Further, the connection between higher richness and NDVI should continue to be studied as a measure of ecosystem functionality, and we suggest that community richness, and specifically the presence of *P. crucifer*, be explored further as proxies for measuring the productivity and functionality of wetland ecosystems.

Chapter 2 examined the temporal effects on community structure and species richness of Anurans, especially important considering the ectothermic nature of Anurans and their dependence on annual cycle. This chapter, as a temporal study, serves as a complement to the spatial of Chapter 1. We found relatively minimal differences between study locations across various temporal measures, including air and water temperature, even ambient noise. However, the detectability of several species was influenced by these variables, and so the lack of difference across our urbanization gradient supports the conclusion that the spatial results of Chapter 1 are primary drivers the differences in community composition.

Chapter 3 mapped the occurrence of select species of interest among the Anuran community and predicts areas that are likely to have higher species richness within the study area. We found a lower percentage of cropland and impervious surface were the factors that best explained an increase in species richness. Lower amounts of cropland and higher amounts of swamp forest were the factors that most explained the occurrence of our prime species of interest, *P. crucifer*, due to its previous association with high quality habitat (Knutson et al., 2000; Price et al., 2007). These models helped identify potential locations to direct future Anuran surveys, and we saw that richness was well predicted by our environmental parameters. These models can assist in assessment of habitat quality via species richness and identify areas that may be suitable for alteration or restoration efforts to encourage recruitment.

Chapter 4 sought to understand the level of support among local adults for conservation activity in the area. We surveyed 304 adults in Northwest Ohio about their attitudes towards conservation and found high levels of support for numerous conservation-based activities, including support for park fees, and voting for officials that support conservation. We also found that exposure to a relatively short educational presentation significantly increased support for most of those statements regarding conservation action. These data, and the template for this survey, can be used to assess support in other local municipalities, while providing valuable information to local land managers.

Trapping Study

As a companion study to Chapters 1 and 2, we attempted trapping surveys to assess population densities of Anuran species at select sites within our study area and relate them to our novel KR index. This index was designed to be more informative than a typical calling survey, while still maintaining ease of use for practitioners without statistical tools or background. The KR Index is formulated as (number of records/number of surveys) * average calling intensity. However, due to resource and time constraints, we were unable to perform the true recapture of individuals, and so our captures were each treated as novel. We had low success capturing more than *P. crucifer* and *L. sylvaticus* in the March trapping session and more than *L. catesbeianus* and *L. clamitans* tadpoles in the May trapping session. For those reasons, we report the results of that study here.

Trapping surveys were conducted for fully metamorphosized frogs from March through July 2023. Each accessible survey location (N=3) had 22 commercially available minnow traps deployed around their perimeter, floating one to three meters from the shoreline. Each trap had a used plastic bottle to ensure floatation, and the traps were tied to the shoreline. Traps varied in composition and size of opening, with 20 traps a light mesh material with a collapsible internal scaffolding (measuring 43.25cm by 25.5cm with openings either 3.25cm or 5cm) (‘mesh’ traps) and 46 traps a firm black metal (measuring 40.75cm by 22.75cm with openings ranging from

3.25cm or 5cm) ('metal' traps). Traps of different sizes and construction were distributed evenly across all three survey sites to avoid trap type bias in capture. Traps were checked by teams of two during daylight hours for any Anuran species individuals, who were identified to species and released. Traps were checked daily for four days, before being removed and stored until the next trapping period. Because of the constraint on resources and time, as well as the extremely low recapture rates documented in both the Anuran mark-recapture literature as well as, in particular, in this study area, each capture was treated as unique throughout each trapping week (McCaffery et al., 2015; Muths et al., 2016). Waders, bins, and any other equipment were sanitized with a 3% bleach solution between sites, to ensure no disease spread (Bletz et al., 2023).

Of our three sampling areas, Site 1 (rural) had an observed community (through frog calling surveys) of seven species, including (*P. triseriata, P. crucifer, L. sylvaticus, A. americanus, H. versicolor, L. catesbeianus* and *L. clamitans*). At Site 2 (rural) we had previously observed 9 species (all except *A. blanchardi*) and at Site 3 (suburban) we had previously observed 4 species (*A. americanus, A. blanchardi, L. clamitans* and *L. catesbeianus*). Site 1 was a series of ephemeral pools within a deciduous and coniferous forest mix, lacking significant understory, and was located in a large local park. Site 2 was approximately 2.6km from Site 1 but was a permanent pond with a more open canopy and in a more consistently trafficked area of the same park; this park was surrounded by deciduous forest, and a more defined understory. Site 3 was a medium-sized lake adjacent to a large river, and near major highway and roads, with a small buffer of forest (<5m) surrounding the lake, and significant understory overgrowth.

In the March sampling period, we captured 70 mature Anurans in 225 trap nights of three species. All 70 were captured at one of the rural sites, with 28 at Site 1 and 42 at Site 2, for a capture rate of 0.33 and 0.56, respectively. At Site 1, seven captures were *P. crucifer*, while 21 captures were *L. sylvaticus*. At Site 2, one capture was *L. clamitans*¸ while 41 captures were *P. crucifer*. Despite having fewer 'mesh' traps overall and thus fewer trap nights, significantly more Anurans were captured in greater numbers using mesh traps over metal traps, with 57% of captures at Site 1 and 81% of captures at Site 2 coming from mesh traps ($p < 0.001$). We note that this trend did not extend to larvae (tadpoles), with ~40% of tadpole captures (in both trapping periods) captured in mesh traps. *P. crucifer* seemed to specifically be the most susceptible to capture in the mesh traps, with 79.2% of captures of that species coming from mesh traps. Despite regularly hearing *P. triseriata* calls while checking traps at Site 1 specifically, we were unable to capture even one adult. As there were no captures at Site 3, we could not compare the difference between capture success between it and Sites 1 and 2. Neither site had significantly higher capture success, either in percentage captured or in number caught, in either adult frogs or tadpoles.

In the May trapping period, capture success of adults plummeted, with just 8 adults captured in 233 trap nights, including five *L. clamitans*, one *L. catesbeianus*, one *L. sylvaticus* and one *H. versicolor*, for a capture rate of 0.03 per trap night. We were enormously successful at capturing tadpoles during this period, with 783 total tadpoles, mostly *L. clamitans* and *L. catesbeianus,* with fewer *P crucifer*. The success rate for capturing tadpoles this period was 3.36 tadpoles per trap night. We did not capture any adults at Site 3 during either trapping period, including *A. blanchardi,* despite registering several adults calling at our trapping site on the lake before, during and after the May trapping session.

We had originally planned a third trapping session, in July, but cancelled it because of low success, as well as the reduced likelihood of improving success so late in the breeding season. We expected even lower success in July because the opening size of our traps likely was not suitable for capturing adult *L. clamitans* and *L. catesbeianus* (and possibly *L. pipiens*) and other summer breeding species (*A. fowleri*, *H. versicolor*) likely would not enter the water/enter traps with enough frequency to improve success. These factors, combined with the conclusion of *P. triseriata*, *P. crucifer*, *L. sylvaticus* and *A. americanus* breeding periods long before the July trapping period, led us to cancel our third effort. After two trapping periods (458 trap nights) we are able to report exceedingly low mortality rates when utilizing minnow traps for Anuran trapping. We recorded just four mortalities, in two events, all of either *L. clamitans* or *L. catesbeianus* tadpoles in mesh traps, potentially in events of cannibalism. Both mortality events were recorded at Site 2, with one having 13 other (living) tadpoles inside and the other having two living. These four mortalities equate to a 0.87% mortality rate per trap night.

We attempted trapping during the 2023 breeding season to quantify abundance and population density of our study species at three of our sites. We found that mesh, rectangular traps caught significantly more individuals than oblong metal traps (p=0.0002). This trend was driven by *P. crucifer*, which were the most commonly captured species. As a result of resource constraints, we were not able to conduct true mark-recapture. For this reason, or capture results should be interpreted with caution. However, we do provide evidence of the value of utilizing mesh traps in such studies. Future studies might dedicate themselves to more concretely quantifying *P. crucifer* populations and relating their density to calling intensity. As *P. crucifer* has been regarded as an indicator of wetland quality, such a study could prove highly beneficial to lowering the resources and time required to survey for wetland quality (Knutson et al., 2000; Price et al., 2007).

We sought to use these data to inform the development of the KR index, however, low trapping success limited our ability to confidently suggest the use of the index with populations in the future. We do believe, however, that the potential for an index similar to the KR would provide significant value to species and land managers who do not have the capability to engage in mark-recapture surveys.

On an additional note, among calling surveys, the intensity of call has long been utilized as a mechanism to gauge a relative density of species when they are heard calling. The standard for calling intensity has been 1-3 for several decades, with 1 being between one and several frogs, but gaps between calls, 2 being several frogs with overlapping calls, and 3 being a full chorus. However, in our extensive frog call surveying, we often experienced species that occur in such high densities in select areas, that it is impossible to ignore the potential quantifiable differences between chorus sizes. For example, populations of *P. crucifer* may call at the consistency required of a chorus with potentially as few as 10-15 calling individuals, but we encountered densities at some sites that we would estimate in the hundreds of calling individuals. As the purpose of auditory surveys is often to collect the most accurate data possible, with less effort than actually capturing species, we believe it is prudent and warranted that calling intensity scale be expanded from the traditional 1-3 to 1-4, and possibly 1-5 if necessary. We have labeled a calling intensity of 4 a 'large chorus' and 5 a 'superchorus.' While these values would only be useful for select species (e.g., *P. crucifer, H. versicolor*) depending on the study area and species density, we believe that increased upper scale limit will allow for a more realistic interpretation of species populations in a study area. This can not only aid our understanding the species of high density but serve as a more accurate scale on which to place species that occur at lower densities (e.g., *L. catesbeianus* in our study) for similar reasons. Unfortunately, we did not have time to critically evaluate the scientific backing behind these claims in our current study, and so they are mentioned here as a recommendation for exploration in the future.

Conclusions

Overall, our research contributes to the understanding and knowledge of spatial dynamics and community structure in ecology. This work contributes to the growing field of urban ecology, as we assessed the effect of urbanization on species richness and community composition. This research identifies the factors that are associated with greater species richness of Anurans and utilized Anuran richness and species presence to predict habitat quality/ecosystem productivity. These findings are applicable to the great many areas across the globe that are affected by human-mediated land use change and habitat degradation. Studies of this nature are extremely valuable towards understanding the effects of habitat loss and degradation can have on wildlife, especially those of limited dispersal ability and complicated life history. It is possible that the records we obtained may not be representative of the Anuran community within the area, however, we feel confident given our several years of data that they are as accurate as possible.

Following the results of Chapter 1 and 3, the development of this work provides tools and recommendations for land managers to manage amphibian populations more effectively under their purview, by focusing on limiting impervious surface, encouraging native plant growth (NDVI), and prioritizing the restoration of swamp forest. This methodology can potentially aid conservation activities to minimize the negative impact on Anurans.

Our results, especially in Chapter 3 (Maxent mapping), have the potential to be viewed as static, but are intended to be fluid over the course of time to address conservation challenges and following new information on the study area or species. The areas that were identified as being below average quality for higher species richness could potentially be improved to encourage species richness and population growth of the existing species. Additionally, the areas that were identified as suitable for higher species richness but are not currently protected can be targeted for acquisition or protection by interested managers when available, allowing the progression of conservation goals into the future.

The methodology used in this research is not limited to the Oak Openings Region or Northwest Ohio. It can be applied anywhere with Anuran species present and the potential availability of environmental data to tackle interesting questions about them. This research also has the ability to be scalable, adaptable, updatable, and targeted to land managers, making it an accessible series of applications to address significant conservation questions.

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APPENDIX A. TABLES

Table 0.1. List of protected areas identified as suitable for survey sites for this study and the number of survey sites for each location, which are scaled to overall size of the protected area.

Protected Area	#Sites	Protected Area	#Sites
Anderson Property (Toledo Metroparks)	1	Ottawa Hills	1
Bay View Park	1	Pearson Metropark	2
Beaver Creek Preserve	1	Providence Metropark	1
Blue Creek Metropark	1	Roth Memorial Cemetery	1
Brandywine Country Club	1	Sawyer Quarry	1
Brookwood Metropark	1	Secor Metropark	2
Camp Miakonda	1	Shoreland Park	1
Cedar Creek Park	1	Side Cut Metropark	1
Collins Park	1	Stone Oak Country Club	1
Duck Creek	1	Swan Creek Metropark	1
Fallen Timbers Fairways	1	Sylvan Prairie Park	1
Farnsworth Metropark	1	Three Meadows Park	1
Howard Marsh Metropark	2	Toledo Botanical Garden	1
International Park	1	Otsego Park	1
Irwin State Nature Preserve	1	Toledo Memorial Cemetery	1
J.C. Reuthinger Memorial Preserve	1	Toledo Metroparks Corridor Properties	2
Jermain Park	1	Toledo Muslim CC	1
Keil Property	1	Van Fleet Ditch	1
Kitty Todd Nature Preserve	4	Walbridge Park	1
Manhattan Marsh Metropark	1	Westwinds Metropark	1
Maumee State Forest	3	Wildwood Metropark	1
Middlegrounds Metropark	1	Winterfield Park	1
Oak Openings Metropark	5	Wiregrass Lake	1
Owens Community College	1	Woodlawn Cemetery	1
Orleans Park	1	W.W. Knight Preserve	1

Table 1.1. Eleven species of Anurans are found in the study area, listed here alphabetically by genus, and their common name.

Scientific Name	Common Name
Acris blanchardi	Blanchard’s cricket frog
Anaxyrus (Bufo) americanus	American toad
Anaxyrus (Bufo) fowleri	Fowler’s toad
Hyla chrysoscelis	Cope’s gray tree frog
Hyla versicolor	Gray tree frog
Lithobates (Rana) catesbeianus	American bull frog
Lithobates (Rana) clamitans	Green frog
Lithobates (Rana) pipiens	Northern leopard frog
Lithobates (Rana) sylvaticus	Wood frog
Pseudacris crucifer	Northern Spring peeper
Pseudacris triseriata	Western chorus frog

Table 1.2. Local-scale habitat data measurements to be collected at the beginning (March) and end (July) of each field season (2021-2023), to compare differences in habitat to differences in diversity and species presence/absence.

Variable Name (unit)	Description	Frequency
Litter depth (cm)	Depth of leaf litter adjacent to 50m transect, alternating sides.	2x per year
Ground cover (%)	Percentage of cover adjacent to 50m transect, alternating sides, classified as grass, litter, bare, or other.	2x per year
Grass height (cm)	Height of grass adjacent to 50m transect, alternating sides.	2x per year
# of coarse woody debris	Number of woody debris over 10cm diameter intersecting with transect.	Once per year
# of fine woody debris	Number of piles of woody debris under 10cm diameter intersecting with transect.	Once per year
# of snags	Number of dead trees standing at >45-degree angle and over 10cm diameter, Within 10m radius of survey point.	Once per year
# of trees	Number of living trees, standing at a >45-degree angle, over 10cm in diameter, Within 10m radius of survey point.	Once per year
# of shrubs	Number of woody plants not classified as trees within 10m radius of survey point.	Once per year
Canopy Cover (%)	Average percentage of cover taken from HabitApp at point and 10m from point in each cardinal direction.	2x per year
Emergent vegetation (%)	Percentage of water body covered by new growth, non-woody vegetation growing above the edge/surface of the water.	2x per year
Floating vegetation (%)	Percentage of water body covered by living, non-woody vegetation growing atop the surface of the water.	2x per year

Table 1.3. Large-scale habitat data measurements to be collected via independent site tools maintained by federal, state, and local governments to measure habitat differences between sites and compare levels of diversity. All measurements below will be collected once per year.

Variable Name (unit)	Description
Landcover	Determined and classified by Root and Martin 2017 map within 250m 1km buffer of location. 15 total cover classes: turf/pasture, wet prairie, residential/mixed, perennial pond, upland savanna, wet shrubland, swamp forest, upland coniferous forest, upland deciduous forest, floodplain forest, sand barrens, Eurasian meadow, upland prairie, urban, and cropland
Elevation (m)	Elevation in meters from sea level
Normalized Difference Vegetation Index (NDVI)	Vegetation quantification based on remote sensing of infrared and near infrared satellite imagery of study locations. Utilized for spring (March/April), summer (June/July) and fall (October/November)
Impervious Surface	Percentage of impervious surface within a 250m or 1km buffer, utilizing ArcGIS layers from the National Landcover Database (NLCD)
Annual Average Traffic Volume (AADT)	Annual average daily traffic obtained through the Ohio Department of Transportation
Raw Traffic Count	Pure count of number of vehicles passing the closest road to a study wetland over a 24-hour period during the study season, as measured by the Ohio Department of Transportation
Traffic Count 6-6	Count of number of vehicles passing the closest road to a study wetland during 6pm to 6am hours, the time period Anurans are most likely to call or travel

Table 1.4. Variables removed during analysis using Spearman's correlation, Principal Component Analysis, or GLMMs. All removed for specific analysis were removed during GLMMs.

Variable	Stage of Removal
Removed for all analysis:	
Fine woody debris	Spearman's correlation
6pm to 6am traffic count	Spearman's correlation
Raw traffic count	Spearman's correlation
Change in litter cover	PCA
Change in emergent vegetation	PCA
Change in floating vegetation	Spearman's correlation
Change in grass cover	PCA
Change in grass height	Spearman's correlation
Shrub cover	PCA
Canopy cover	PCA
Number of snags	PCA
Number of trees	PCA
Removed for Richness analysis:	
Change in litter depth	
AADT	
Early season NDVI (both)	
% Turf (both)	
% Perennial ponds (both)	
% Upland coniferous forest (250m)	
% Floodplain forest (250m)	
% Sand barrens (both)	
% Eurasian meadow (both)	
% Upland prairie (both)	
% Cropland (250m)	
% Wet shrubland (1km)	
Removed for urbanization class analysis:	
AADT	
Early season NDVI (both)	
% Turf (both)	
% Wet prairie (both)	
% Perennial ponds (both)	
% Wet shrubland (both)	
% Upland coniferous (250m)	
% Floodplain forest (250m)	
% Sand barrens (250m)	
% Eurasian meadow (both)	
% Upland prairie (both	
% Urban (250m)	

Variable	Stage of Removal
% Cropland (250m)	
Removed for analysis of average species per survey:	
AADT	
Average NDVI (250m)	
Early season NDVI (both)	
% Turf (both)	
% Wet prairie (both)	
% Perennial ponds (both)	
% Upland savanna (250m)	
% Wet shrubland (both)	
% Upland coniferous (both)	
% Upland deciduous (250m)	
% Floodplain forest (both)	
% Sand barrens (both)	
% Eurasian meadow (both)	
% Upland prairie (both)	
% Urban (250m)	
% Cropland (both)	

Table 1.5. Species identified at each site and relative abundance. Species included if they were recorded at least once in any of the three survey years (2021-2023). Relative abundance calculated by multiplying percentage of encounters by average calling intensity (ranked one to five). Sites and species listed alphabetically. Sites that are omitted did not have any Anurans identified. R=Rural site, S=Suburban site, U=Urban site.

Site	Class	Species	Relative Abundance
Anderson Property (Toledo Metroparks)	U	*A. americanus*	0.078
		L. catesbeianus	0.308
		L. clamitans	0.308
		L. pipiens	0.078
Bay View Park	S	*A. americanus*	0.325
		H. versicolor	0.080
		L. pipiens	0.063
		P. triseriata	0.188
Beaver Creek Preserve	R	*A. americanus*	0.161
		A. blanchardi	0.790
		A. fowleri	0.097
		H. versicolor	0.420
		L. catesbeianus	0.307
		L. clamitans	0.323
		P. crucifer	0.129
		P. triseriata	0.194
Blue Creek Metropark	R	*A. americanus*	0.135
		A. fowleri	0.081
		H. versicolor	0.570
		L. catesbeianus	0.054
		L. clamitans	0.189
		L. pipiens	0.297
		P. crucifer	1.162
		P. triseriata	0.216
Brandywine Country Club	S	*L. catesbeianus*	0.250
Brookwood Metropark	U	*A. americanus*	0.417
		L. catesbeianus	0.167
		L. clamitans	0.250

Site	Class	Species	Relative Abundance
Camp Miakonda	S	*A. americanus*	0.139
		L. catesbeianus	0.500
		L. clamitans	0.917
		P. crucifer	0.056
Cedar Creek Park	S	*A. americanus*	0.400
		P. crucifer	0.067
Collins Park	S	*A. americanus*	0.300
Duck Creek	U	*A. americanus*	0.408
		A. blanchardi	0.026
		H. versicolor	0.030
		L. catesbeianus	0.026
		L. clamitans	0.158
		L. pipiens	0.026
		P. crucifer	0.026
		P. triseriata	0.184
Fallen Timbers Fairways	S	*H. versicolor*	0.130
		L. catesbeianus	0.375
		L. clamitans	0.125
Farnsworth Metropark	R	*A. americanus*	0.111
		A. blanchardi	0.389
		A. fowleri	0.074
		H. versicolor	0.350
		L. catesbeianus	0.037
Howard Marsh Metropark	R	*A. americanus*	0.143
		L. catesbeianus	0.839
		L. clamitans	0.500
		L. pipiens	0.143
Irwin State Nature Preserve	R	*A. americanus*	0.384
		A. fowleri	0.116
		H. versicolor	1.520
		L. catesbeianus	0.186
		L. clamitans	0.570
		L. pipiens	0.256
		P. crucifer	1.326
		P. triseriata	1.000

Site	Class	Species	Relative Abundance
J.C. Reuthinger	S		
Memorial Preserve		*A. americanus*	0.256
		A. blanchardi	0.026
		A. fowleri	0.077
		H. versicolor	0.080
		P. triseriata	0.590
Jermain Park	U	*A. americanus*	0.357
		L. pipiens	0.024
Keil Property	U	*H. versicolor*	0.170
Kitty Todd Nature Preserve	R	*A. americanus*	0.175
		A. blanchardi	0.018
		A. fowleri	0.026
		H. versicolor	0.910
		L. catesbeianus	0.110
		L. clamitans	0.377
		L. pipiens	0.079
		L. sylvaticus	0.018
		P. crucifer	1.290
		P. triseriata	0.632
Manhattan Marsh Metropark	U	*A. americanus*	0.268
		L. catesbeianus	0.342
		L. clamitans	0.317
		L. pipiens	0.220
		L. sylvaticus	0.024
		P. crucifer	0.244
		P. triseriata	0.183
Maumee State Forest	R	*A. americanus*	0.063
		A. fowleri	0.019
		H. versicolor	0.420
		L. catesbeianus	0.131
		L. clamitans	0.187
		L. pipiens	0.065
		L. sylvaticus	0.168
		P. crucifer	1.266
		P. triseriata	0.631
Middlegrounds Metropark	U	*A. blanchardi*	3.864
		A. americanus	1.500
		L. catesbeianus	2.227
		L. clamitans	0.317

Site	Class	Species	Relative Abundance
Oak Openings Metropark	R	*A. americanus*	0.278
		A. fowleri	0.023
		H. versicolor	0.590
		L. catesbeianus	0.282
		L. clamitans	0.658
		L. pipiens	0.060
		L. sylvaticus	0.075
		P. crucifer	1.086
		P. triseriata	0.248
Owens Community College	S	*A. americanus*	0.077
		H. versicolor	0.310
		L. catesbeianus	0.500
		P. triseriata	0.462
Orleans Park	S	*A. americanus*	0.256
		A. blanchardi	0.930
		H. versicolor	0.070
		L. catesbeianus	0.302
		L. clamitans	0.535
		L. pipiens	0.047
		P. crucifer	0.047
Otsego Park	R	*A. blanchardi*	0.333
Ottawa Hills	U	*A. americanus*	0.273
		L. catesbeianus	0.061
		L. clamitans	0.030
Pearson Metropark	S	*A. americanus*	0.287
		A. fowleri	0.013
		L. catesbeianus	0.227
		L. clamitans	0.380
		L. pipiens	0.127
		P. crucifer	0.027
		P. triseriata	0.313
Providence Metropark	R	*A. americanus*	0.175
		A. blanchardi	0.050
		A. fowleri	0.100
		H. versicolor	0.600
		L. catesbeianus	0.150
		L. clamitans	0.150
		L. pipiens	0.050
		P. crucifer	0.900

Site	Class	Species	Relative Abundance
Roth Memorial Cemetery	S	*A. americanus*	0.295
		H. versicolor	0.090
		L. catesbeianus	0.273
		L. clamitans	0.568
Sawyer Quarry	R	*A. americanus*	0.250
		H. versicolor	0.170
		L. catesbeianus	0.119
		L. pipiens	0.071
		P. crucifer	0.024
Secor Metropark	R	*A. americanus*	1.063
		A. fowleri	0.063
		H. versicolor	3.810
		L. catesbeianus	0.688
		L. clamitans	1.969
		L. pipiens	0.750
		L. sylvaticus	0.063
		P. crucifer	4.313
		P. triseriata	3.938
Shoreland Park	U	*A. americanus*	0.050
		L. catesbeianus	0.338
		L. clamitans	0.250
		L. pipiens	0.100
		P. crucifer	0.075
Side Cut Metropark	S	*A. americanus*	0.114
		A. blanchardi	1.443
		L. catesbeianus	0.400
		L. clamitans	0.057
Stone Oak Country Club	S	*A. americanus*	0.222
		L. catesbeianus	0.444
		L. clamitans	0.333
		P. crucifer	0.333
		P. triseriata	0.222
Swan Creek Metropark	U	*A. americanus*	0.250
		L. catesbeianus	0.111
		L. clamitans	0.444
		L. pipiens	0.083
		P. crucifer	0.083
Sylvan Prairie Park	S	*A. americanus*	0.500

Site	Class	Species	Relative Abundance
		A. fowleri	0.048
		H. versicolor	1.170
		L. catesbeianus	0.095
		L. clamitans	0.476
		L. pipiens	0.333
		P. crucifer	0.786
		P. triseriata	0.024
Toledo Botanical Garden	U	*A. americanus*	0.378
		H. versicolor	0.180
		L. catesbeianus	0.743
		L. clamitans	0.581
		P. crucifer	0.081
Toledo Memorial Cemetery	S	*A. americanus*	0.182
		H. versicolor	0.480
		L. catesbeianus	0.546
		L. clamitans	0.750
		P. crucifer	0.046
Toledo Metroparks	R	*A. americanus*	0.259
Corridor Properties		*A. fowleri*	0.024
		H. versicolor	0.440
		L. catesbeianus	0.047
		L. clamitans	0.200
		L. pipiens	0.059
		L. sylvaticus	0.271
		P. crucifer	1.124
		P. triseriata	0.482
Toledo Muslim CC	U	*A. americanus*	0.338
		H. versicolor	0.640
		L. clamitans	0.075
		L. pipiens	0.225
		P. crucifer	0.075
Van Fleet Ditch	R	*A. americanus*	0.200
		A. fowleri	0.200
		H. versicolor	0.800
		L. catesbeianus	0.067
		L. clamitans	0.133
		P. crucifer	0.667
		P. triseriata	0.067
Westwinds Metropark	R	*A. americanus*	0.409

Site	Class	Species	Relative Abundance
		A. fowleri	0.045
		H. versicolor	0.430
		L. clamitans	0.045
		L. pipiens	0.182
		P. crucifer	0.955
		P. triseriata	0.636
Wildwood Metropark	S	*A. americanus*	0.244
		H. versicolor	0.220
		L. clamitans	0.366
		L. pipiens	0.024
Winterfield Park	U	*A. americanus*	0.220
		H. versicolor	0.120
		L. clamitans	0.040
Wiregrass Lake	R	*A. americanus*	0.159
		A. fowleri	0.057
		H. versicolor	1.230
		L. catesbeianus	0.511
		L. clamitans	0.386
		L. pipiens	0.045
		P. crucifer	1.080
		P. triseriata	0.580
Woodlawn Cemetery	U	*A. americanus*	0.250
		A. blanchardi	0.050
		L. catesbeianus	0.425
		L. clamitans	0.475
W.W. Knight Preserve	S	*A. americanus*	0.227
		A. blanchardi	1.534
		L. catesbeianus	0.671
		L. clamitans	0.114

Table 1.6. Generalized Linear Mixed Models for combinations of variables at any scale, to predict species richness in Northwest Ohio study area. Included are models containing variables at either 250m, 1km, or both. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC. Class refers to urbanization gradient classes (urban, suburban, or rural).

Variable(s)	Estimate	Std. Error	*Z*	*P*	ΔAIC
% Residential-Mixed 1km + %Urban 1km:					
(Intercept)	1.97	0.07	27.95	<0.001***	**+0**
%Residential 1km	-1.03	0.20	-5.22	<0.001***	
%Urban 1km	-0.86	0.40	-2.18	0.029*	
Average NDVI 1km + %Residential-Mixed 1km:					
(Intercept)	1.36	0.28	4.84	<0.001***	**+0.1**
Avg NDVI 1km	2.22	1.02	2.19	0.029*	
%Residential 1km	-1.01	0.20	-5.00	<0.001***	
%Residential-Mixed 1km + %Swamp Forest 1km:					
(Intercept)	1.78	0.11	16.28	<0.001***	**+0.5**
%Residential 1km	-0.98	0.22	-4.47	<0.001***	
%Swamp 1km	-0.98	0.46	2.13	0.032*	
Late NDVI 1km + %Residential-Mixed 1km:					
(Intercept)	1.37	0.31	4.47	<0.001***	**+0.9**
Late NDVI 1km	1.41	0.71	1.98	0.048*	
%Residential 1km	-0.92	0.24	-3.83	<0.001***	
Class + %Residential-Mixed 250m + %Swamp Forest 250m + Late NDVI 1km:					
(Intercept)	1.37	0.33	4.17	<0.001***	**+2.0**
Class	-0.12	0.07	-1.63	0.103	
%Res. 250m	-0.37	0.20	-1.85	0.064.	
%Swamp 250m	0.46	0.25	1.83	0.067.	
Late NDVI 1km	1.15	0.74	1.57	0.118	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table 1.7. Generalized Linear Mixed Models for combinations of variables at any scale, to predict average species per survey in Northwest Ohio study area. Included are models containing variables at either 250m, 1km, or both. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC. Class refers to urbanization gradient classes (urban, suburban, or rural).

Variable(s)	Estimate	Std. Error	*Z*	*P*	ΔAIC
Class + %Swamp Forest 1km:					
(Intercept)	0.53	0.35	1.55	0.121	**+0**
Class	-0.31	0.15	-2.08	0.038*	
%Swamp 1km	1.90	1.33	1.42	0.154	
Class:					
(Intercept)	0.92	0.23	4.00	<0.001***	**+0.1**
Class	-0.45	0.12	-3.86	<0.001***	
%Impervious 1km + %Swamp Forest 1km:					
(Intercept)	0.23	0.21	1.07	0.286	**+0.2**
%Impervious 1km	-1.43	0.70	-2.06	0.040*	
%Swamp 1km	2.02	1.30	1.55	0.121	
Class + Late NDVI 1km:					
(Intercept)	-0.01	0.77	-0.01	0.992	**+0.6**
Late NDVI 1km	2.11	1.68	1.25	0.210	
Class	-0.32	0.16	-2.01	0.045*	
Class +%Urban 1km:					
(Intercept)	-0.89	0.23	3.82	<0.001***	**+0.7**
Class	-0.37	0.14	-2.68	<0.001***	
%Urban 1km	-1.13	0.93	-1.22	0.224	
% Impervious 1km:					
(Intercept)	-0.50	0.14	3.58	<0.001***	**+0.8**
%Impervious 1km	-2.11	0.55	-3.84	<0.001***	
Late NDVI 250m + Class:					
(Intercept)	0.24	0.66	0.37	0.714	**+0.9**
Class	-0.38	0.13	-2.82	0.005**	
Late NDVI 250m	1.57	1.43	1.10	0.271	
%Residential-Mixed 1km + %Swamp Forest 1km:					
(Intercept)	0.35	0.29	1.19	0.232	**+1.0**

Variable(s)	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Residential 1km	-0.92	0.53	-1.81	0.071.	
%Swamp 1km	2.09	1.34	1.56	0.118	
Class + %Swamp Forest 250m:					
(Intercept)	0.72	0.31	2.33	0.020*	**+1.1**
Class	-0.38	0.14	-2.74	0.006**	
%Swamp 250m	0.69	0.71	0.97	0.332	
Average NDVI 1km + %Residential-Mixed 1km:					
(Intercept)	-0.27	0.66	-0.41	0.690	**+1.3**
Avg NDVI 1km	3.74	2.43	1.54	0.124	
%Residential 1km	-1.17	0.48	1.54	0.014*	
%Residential-Mixed 1km + %Urban 1km:					
(Intercept)	0.74	0.20	3.78	<0.001***	**+1.3**
%Residential 1km	-1.21	0.47	-2.59	0.010**	
%Urban 1km	-1.39	0.89	-1.57	0.116	
Class + %Residential-Mixed 250m:					
(Intercept)	0.89	0.23	3.79	<0.001***	**+1.4**
Class	-0.36	0.16	-2.31	0.021*	
%Residential 250m	-0.40	0.47	-0.85	0.397	
Late NDVI 1km + %Residential-Mixed 1km:					
(Intercept)	-0.28	0.68	-0.41	0.682	**+1.4**
Late NDVI 1km	2.42	1.60	1.52	0.130	
%Residential 1km	-1.00	0.54	-1.85	0.064.	
%Swamp Forest 250m + %Impervious 1km:					
(Intercept)	0.35	0.19	1.83	0.067.	**+1.4**
%Swamp 250m	0.79	0.69	1.14	0.254	
%Impervious 1km	-1.74	0.63	-2.74	0.006**	
(Intercept)	0.71	0.19	3.74	<0.001***	**+1.6**
%Residential 1km	-1.49	0.41	-3.61	<0.001***	
Late NDVI 1km + %Impervious 1km:					
(Intercept)	-0.28	0.71	-0.40	0.690	**+1.6**
Late NDVI 1km	2.02	1.82	1.11	0.267	
%Impervious 1km	-1.43	0.82	-1.75	0.081.	
Late NDVI 1km + %Swamp Forest 1km:					
(Intercept)	-0.86	0.46	-1.85	0.064.	**+1.6**
Late NDVI 1km	2.52	1.54	1.65	0.101	

Variable(s)	Estimate	Std. Error	*Z*	*P*	ΔAIC
Late NDVI 1km	2.25	1.33	1.69	0.091.	
Class + %Upland Savanna 1km:					
(Intercept)	0.80	0.29	2.75	0.006**	**+1.6**
Class	-0.41	0.13	-3.05	0.002**	
%Up. Sav. 1km	1.54	2.36	0.65	0.514	
Class + %Residential-Mixed 1km:					
(Intercept)	0.80	0.29	2.75	0.006**	**+1.6**
Class	-0.32	0.24	-1.37	0.171	
%Residential 1km	-0.52	0.83	-0.62	0.533	
Class + Average NDVI 250m + Late NDVI 250m:					
(Intercept)	0.19	0.64	0.30	0.760	**+1.7**
Class	-0.33	0.14	-2.34	0.019*	
Avg NDVI 250m	-5.38	4.89	-1.10	0.271	
Late NDVI 250m	5.09	3.44	1.48	0.140	
%Upland Prairie 250m + %Impervious 1km:					
(Intercept)	0.61	0.18	3.38	<0.001***	**+1.7**
%Up. Prair. 250m	-0.65	0.62	-1.05	0.293	
%Impervious 1km	-2.15	0.56	-3.86	<0.001***	
%Impervious 250m + %Swamp 1km:					
(Intercept)	0.11	0.19	0.56	0.570	**+1.8**
%Impervious 250m	-1.41	0.89	-1.60	0.111	
%Swamp 1km	2.51	1.27	1.99	0.047*	
Class + %Impervious 1km + %Swamp Forest 1km:					
(Intercept)	0.45	0.40	1.12	0.264	**+1.8**
Class	-0.20	0.30	-0.65	0.514	
%Impervious 1km	-0.62	1.44	-0.43	0.668	
%Swamp 1km	1.85	1.33	1.38	0.166	
%Impervious 1km + %Residential-Mixed 1km:					
(Intercept)	0.63	0.20	3.18	0.001**	**+1.8**
%Impervious 1km	-1.35	0.99	-1.37	0.172	
%Residential 1km	-0.70	0.72	-0.96	0.337	
Class + Average NDVI 250m:					
(Intercept)	0.63	0.60	1.04	0.298	**+1.8**
Class	-0.43	0.12	-3.46	<0.001***	
Avg NDVI 250m	1.09	2.03	0.53	0.595	

Variable(s)	Estimate	Std. Error	*Z*	*P*	ΔAIC
Class + Early NDVI 1km:					
(Intercept)	0.63	0.58	1.09	0.275	**+1.8**
Class	-0.43	0.12	-3.49	<0.001***	
Early NDVI 1km	1.74	3.21	0.54	0.587	
Class + %Impervious 1km:					
(Intercept)	0.79	0.33	2.39	0.017*	**+1.8**
Class	-0.30	0.30	-0.98	0.325	
%Impervious 1km	-0.80	1.45	-0.55	0.582	
%Residential-Mixed 250m + %Impervious 1km:					
(Intercept)	0.55	0.15	3.66	<0.001***	**+1.8**
%Res. 250m	-0.45	0.46	-0.98	0.326	
%Impervious 1km	-1.66	0.73	-2.26	0.024*	
%Residential-Mixed 250m+ %Swamp Forest 1km:					
(Intercept)	0.14	0.21	0.67	0.505	**+1.9**
%Res. 250m	-0.64	0.41	-1.55	0.121	
%Swamp 1km	2.59	1.26	2.06	0.039*	
%Swamp Forest 1km + %Impervious 1km + Late NDVI 1km:					
(Intercept)	-0.19	0.72	-0.26	0.795	**+1.9**
%Swamp 1km	1.76	1.37	1.29	0.199	
%Impervious 1km	-1.13	0.85	-1.33	0.185	
Late NDVI 1km	1.16	1.93	0.60	0.549	
Class + %Impervious 250m:					
(Intercept)	0.88	0.26	3.34	<0.001***	**+2.0**
Class	-0.40	0.19	-2.12	0.034*	
%Imp. 250m	-0.41	1.21	-0.34	0.732	
% Swamp Forest 1km + %Upland Deciduous Forest 1km:					
(Intercept)	-0.18	0.13	-1.40	0.162	**+2.0**
%Swamp 1km	2.84	1.15	2.47	0.014*	
%Up. Dec. 1km	2.03	1.41	1.44	0.150	
Late NDVI 250m + %Residential-Mixed 250m:					
(Intercept)	-0.01	0.59	-0.02	0.984	**+2.0**
Late NDVI 250m	1.78	1.38	1.29	0.197	
%Residential 1km	-1.24	0.46	-2.68	0.007**	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table 1.8. Generalized Linear Mixed Models for combinations of variables at any scale, to predict urbanization gradient classes in Northwest Ohio study area. Classes were numerically ranked (Rural = 1, Suburban = 2, Urban = 3), and as a result, positive results indicate an increase in the variable as urban areas are approached. Included are models containing variables at either 250m, 1km, or both. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC.

Variable(s)	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Impervious 1km + %Residential-Mixed 1km:					
(Intercept)	-0.10	0.15	-0.70	0.485	**+0**
%Impervious 1km	1.34	0.55	2.45	0.014*	
%Residential 1km	0.90	0.42	2.16	0.031*	
%Impervious 1km + %Flood Forest 250m + %Residential-Mixed 1km:					
(Intercept)	-0.16	0.16	-1.02	0.308	**+0.9**
%Impervious 1km	1.38	0.55	2.51	0.012*	
%Fld. For. 250m	0.50	0.45	1.11	0.267	
%Residential 1km	0.93	0.42	2.21	0.027*	
%Impervious 1km + %Flood Forest 250m + Residential-Mixed 1km + %Wet prairie 1km:					
(Intercept)	-0.20	0.20	-1.35	0.178	**+2.0**
%Impervious 1km	1.43	0.56	2.57	0.010*	
%Fld. For. 250m	0.59	0.46	1.27	0.206	
%Residential 1km	1.04	0.44	2.36	0.018*	
%Wet Prairie	1.78	1.92	0.93	0.354	

* Significant at $p < 0.05$

Table 2.1. Relationships between urbanization gradient class and various temporal measures, taken during Anuran auditory surveys from 2021-2023. Evaluated in JMP v.11 using Kruskal-Wallis tests. *F*-ratio and R^2 values are included only for significant tests and stem from ANOVA tests on the same variables.

Variable Name (units)	*F*-ratio	R^2	*P*
Air pressure (Hg)	---	---	0.043*
Air temperature (°C)	---	---	0.254
% time under D0: “Abnormally dry”	---	---	0.902
% time under D1: “Moderate drought”	---	---	0.888
Julian day of survey	---	---	0.524
Lunar Illumination %	---	---	0.403
Lunar phase by stage of cycle	---	---	0.686
Noise (dB)	---	---	0.192
Noise index	0.36	17.82	<0.001***
Sky code	---	---	0.320
Time of survey	---	---	0.851
Water present	---	---	0.415
Water temperature	---	---	0.392
Wind code	0.10	3.44	<0.001***
Wind speed	---	---	0.136

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$

Table 2.2. Temporal variables during important life stages for Anurans, evaluated for influence on species richness. Evaluated in JMP v.11 using ANOVA.

Variable Name (units)
Previous Year, March through August
Average precipitation (cm)
Number of days with precipitation
Total precipitation (cm)
Average temperature (°C)
Average minimum temperature (°C)
Average maximum temperature (°C)
Average wind speed (mph)
Previous Winter, September through February
Average precipitation (cm)
Number of days with precipitation
Total precipitation (cm)
Average temperature (°C)
Average minimum temperature (°C)
Average maximum temperature (°C)
Average wind speed (mph)
In year, March through August
Average precipitation (cm)
Number of days with precipitation
Total precipitation (cm)
Average temperature (°C)
Average minimum temperature (°C)
Average maximum temperature (°C)
Average wind speed (mph)

Table 3.1. Variables utilized for Maxent study with name and units, description of the variable, and the frequency of measurement.

Variable Name (units)	Description Measurement	Frequency
Landcover	Classification of 30 by 30m square within one of 15 landcover classes Landcover classification created by Martin and Root (2020) Classes include: Turf/pasture Wet prairie Residential/mixed use Perennial ponds Upland savanna Wet shrubland Swamp forest Upland coniferous forest Upland deciduous forest Floodplain forest Sand barrens Eurasian meadow Upland prairie Urban Cropland	Once
NDVI	Normalized Difference Vegetation Index derived from LIDAR data, ratio of Near-Infrared and Red from satellite data that measures biomass of vegetation	Twice per year (March/April, June/July)
Impervious surface	Percentage of impervious surface, derived from the National Landcover Database via U.S. Geological Survey	Once (2021)

Table 3.2. Environmental variables contributing to the best Maxent models for any eight species (AUC = 0.821). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.1.

Variable	Percent contribution	Permutation importance
Impervious surface	80.7	35.8
Cropland	13.0	51.9
Late season NDVI	4.4	5.4
Urban	1.6	4.3
Eurasian meadow	0.3	2.6
Upland savanna	0	0
Upland deciduous	0	0
Perennial ponds	0	0
Floodplain forest	0	0
Wet shrubland	0	0
Wet prairie	0	0
Upland prairie	0	0
Upland coniferous	0	0
Turf/pasture	0	0
Swamp forest	0	0
Sand barrens	0	0
Early season NDVI	0	0

Table 3.3. Environmental variables contributing to the best Maxent models for any seven species (AUC=0.895). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.4.

Variable	Percent contribution	Permutation importance
Impervious surface	51.6	4.2
Cropland	17.7	59.6
Urban	13.5	13.7
Upland prairie	6.2	4.2
Floodplain forest	3.1	2.3
Eurasian meadow	3.0	8.7
Swamp forest	2.3	2.1
Early season NDVI	0.9	2.3
Late season NDVI	0.8	1.9
Wet prairie	0.7	0.8
Upland savanna	0.1	0.3
Upland deciduous	0	0
Upland coniferous	0	0
Sand barrens	0	0
Perennial ponds	0	0
Wet shrubland	0	0
Turf pasture	0	0

Table 3.4. Environmental variables contributing to the best Maxent models for any five specialist species (AUC = 0.894). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.5.

Variable	Percent contribution	Permutation importance
Impervious surface	60.0	30.8
Cropland	15.9	55.1
Swamp forest	11.0	6.8
Residential	5.0	2.1
Urban	4.3	2.8
Eurasian meadow	2.0	1.4
Upland prairie	1.4	0.6
Upland deciduous	0.3	0.2
Sand barrens	0	0.1
Floodplain forest	0	0.1
Wet shrubland	0	0
Wet prairie	0	0
Upland savanna	0	0
Upland coniferous	0	0
Turf/pasture	0	0
Perennial ponds	0	0
Late season NDVI	0	0
Early season NDVI	0	0

Table 3.5. Environmental variables contributing to the best Maxent models for any four specialist species (AUC = 0.917). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.6.

Variable	Percent contribution	Permutation importance
Residential/mixed use	49.8	29.5
Cropland	21.5	49.1
Urban	8.6	3.6
Swamp forest	5.3	2.0
Impervious surface	5.2	2.1
Floodplain forest	3.9	1.9
Eurasian meadow	3.1	4.1
Sand barrens	1.1	2.3
Upland deciduous	1.1	1.4
Late season NDVI	0.2	2.7
Upland savanna	0.2	0.6
Early season NDVI	0.1	0.5
Wet prairie	0	0.2
Perennial ponds	0	0
Turf/pasture	0	0
Upland coniferous	0	0
Upland prairie	0	0
Wet shrubland	0	0

Table 3.6. Environmental variables contributing to the best Maxent model for predicting the occurrence of *Acris blanchardi* (AUC = 0.862). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.7.

Variable	Percent contribution	Permutation importance
Cropland	24.5	37.7
Late season NDVI	17.8	18.7
Upland prairie	17.1	1.8
Urban	15.3	7.2
Impervious surface	7.8	15.4
Eurasian meadow	7.5	8.3
Upland deciduous	2.8	0.5
Early season NDVI	1.9	5.8
Floodplain forest	1.6	0.4
Upland savanna	1.2	1.2
Swamp forest	1.1	1.1
Sand barrens	0.7	0.3
Wet prairie	0.5	1.4
Residential/mixed use	0.3	0
Upland coniferous	0	0
Perennial ponds	0	0
Turf pasture	0	0
Wet shrubland	0	0

Table 3.7. Environmental variables contributing to the best Maxent model for predicting the occurrence of *Lithobates pipiens* (AUC = 0.877). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.7.

Variable	Percent contribution	Permutation importance
Wet prairie	30.3	6.9
Residential/mixed use	19.2	14.1
Upland prairie	18.4	2.7
Cropland	12.7	50.9
Impervious surface	5.4	0.2
Floodplain forest	3.9	1.1
Upland deciduous	2.8	0.7
Eurasian meadow	1.8	2.8
Urban	1.7	9.5
Upland savanna	1.4	1.8
Sand barrens	1.1	5.8
Early season NDVI	0.8	2.1
Late season NDVI	0.5	1.1
Upland coniferous	0.1	0
Perennial ponds	0	0.2
Swamp forest	0	0.2
Turf/pasture	0	0
Wet shrubland	0	0

Table 3.8. Environmental variables contributing to the best Maxent model for predicting the occurrence of *Pseudacris triseriata* (AUC = 0.887). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.7.

Variable	Percent contribution	Permutation importance
Residential/mixed use	35.2	17.5
Impervious surface	16.9	2.6
Cropland	15.5	47.9
Swamp forest	8.3	0.8
Eurasian meadow	5.4	6.1
Urban	3.9	5.2
Late season NDVI	3.3	10.0
Upland savanna	3.2	3.8
Sand barrens	2.7	2.8
Upland prairie	1.6	0
Wet prairie	1.5	0.8
Upland coniferous	0.9	0.3
Early season NDVI	0.6	1.9
Floodplain forest	0.5	0
Upland deciduous	0.4	0.2
Perennial ponds	0.2	0.1
Turf/pasture	0	0
Wet shrubland	0	0

Table 3.9. Environmental variables contributing to the best Maxent model for predicting the occurrence of *Pseudacris crucifer* (AUC = 0.897). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.8.

Variable	Percent contribution	Permutation importance
Impervious surface	28.6	9.7
Residential/mixed use	22.0	9.2
Cropland	17.7	45.7
Eurasian meadow	6.1	6.4
Urban	5.9	6.6
Swamp forest	4.4	0.2
Upland savanna	3.5	2.3
Late season NDVI	3.1	13.5
Sand barrens	2.8	1.9
Early season NDVI	1.8	3.9
Upland prairie	1.4	0
Upland coniferous	1.0	0
Floodplain forest	0.9	0
Upland deciduous	0.4	0.2
Perennial pond	0.4	0.2
Wet prairie	0.1	0.1
Turf/pasture	0	0
Wet shrubland	0	0

Table 3.10. Environmental variables contributing to the best Maxent model for predicting the occurrence of *Lithobates catesbeianus* (AUC = 0.818). The percentage contribution evaluates how much the variable adds to the Maxent model based on the order the variables, while permutation importance evaluates the importance of the variable based on the final model. Map results can be found in Figure 3.8.

Variable	Percent contribution	Permutation importance
Upland prairie	40.8	1.4
Eurasian meadow	11.2	16.0
Floodplain forest	11.0	1.1
Wet prairie	7.4	0.9
Cropland	6.8	34.4
Late season NDVI	6.7	12.5
Residential/mixed use	5.6	8.1
Upland savanna	2.7	0.7
Sand barrens	2.2	4.4
Urban	1.4	7.8
Impervious surface	1.3	9.4
Swamp forest	1.0	0
Early season NDVI	0.8	2.7
Upland deciduous	0.8	0.6
Upland coniferous	0.4	0.1
Perennial ponds	0	0
Turf/pasture	0	0
Wet shrubland	0	0

Table 4.1. Demographic breakdown of the adults in the five counties used in our study area (Lucas, Ottawa, Wood, Henry, Fulton). Data from the U.S. Census Bureau, 2020. Votes for the 2020 Democratic candidate only listed for the Presidential election.

County	Population Size	% Under 45	% Female	% White	% Votes 2020 - Democratic	% Bachelor's Deg. or Higher
Lucas	426,643	34.82%	52.19%	73.40%	57.04%	20.99%
Ottawa	40,367	26.34%	49.90%	90.30%	37.46%	19.99%
Wood	131,592	38.76%	50.72%	85.10%	45.29%	27.22%
Henry	27,601	31.36%	50.52%	91.18%	27.47%	16.80%
Fulton	42,713	31.54%	50.88%	86.50%	29.22%	13.12%
Total	668,916	44.75%	51.61%	73.42%	42.47%	27.48%

Table 4.2. Matrix of example questions from the Likert-scale survey, demonstrating positively/negatively phrased or action-based/values-based questions. Negative questions are those that we expected a negative (i.e., disagree or strongly disagree) response on the survey. Each question on the survey was deemed either action or values-based, while also being deemed positively or negatively phrased.

Question Type:	Action-based	Values-Based
Positively Phrased	I would reduce my pesticide use to help protect wildlife.	Seeing wildlife is important to me when I visit a park.
Negatively Phrased	I would not change my property for wildlife without a tax break.	I do not care if our parks protect wildlife.

Table 4.3. Select action-based questions that showed a significant difference in support between at least two groups of any demographic, besides time since last park visit. After ANOVA tests, Tukey's post-hoc test was run between all groups to determine a significant difference between groups within one demographic question. The higher support group in all cases demonstrated more support for a question than the lower support group.

Question	Before/After Survey	Demographic Question	Higher Support Group	Lower Support Group	*P*-Value
Donate to wildlife	Before	County	Fulton	Ottawa	0.028*
Significant fee	Before	County	Fulton	Ottawa	0.001**
No new preserves	After	County	Henry	Ottawa	0.022*
				Wood	0.029*
	After	County	Fulton	Lucas	0.043*
				Ottawa	0.006**
				Wood	0.012*
	After	Education	Graduate	Some College	0.007*
Volunteer parks	Before	Education	Undergrad	Graduate	0.035*
Volunteer awareness	Before	Education	Undergrad	Graduate	0.025*
Reduce pesticides	After	Education	Undergrad	Some College	0.030*
Improve parks	After	Education	Graduate	HS or Less	0.005**
				Some College	0.006**

* Significant at $p < 0.05$
** Significant at $p < 0.01$

APPENDIX B. FIGURES

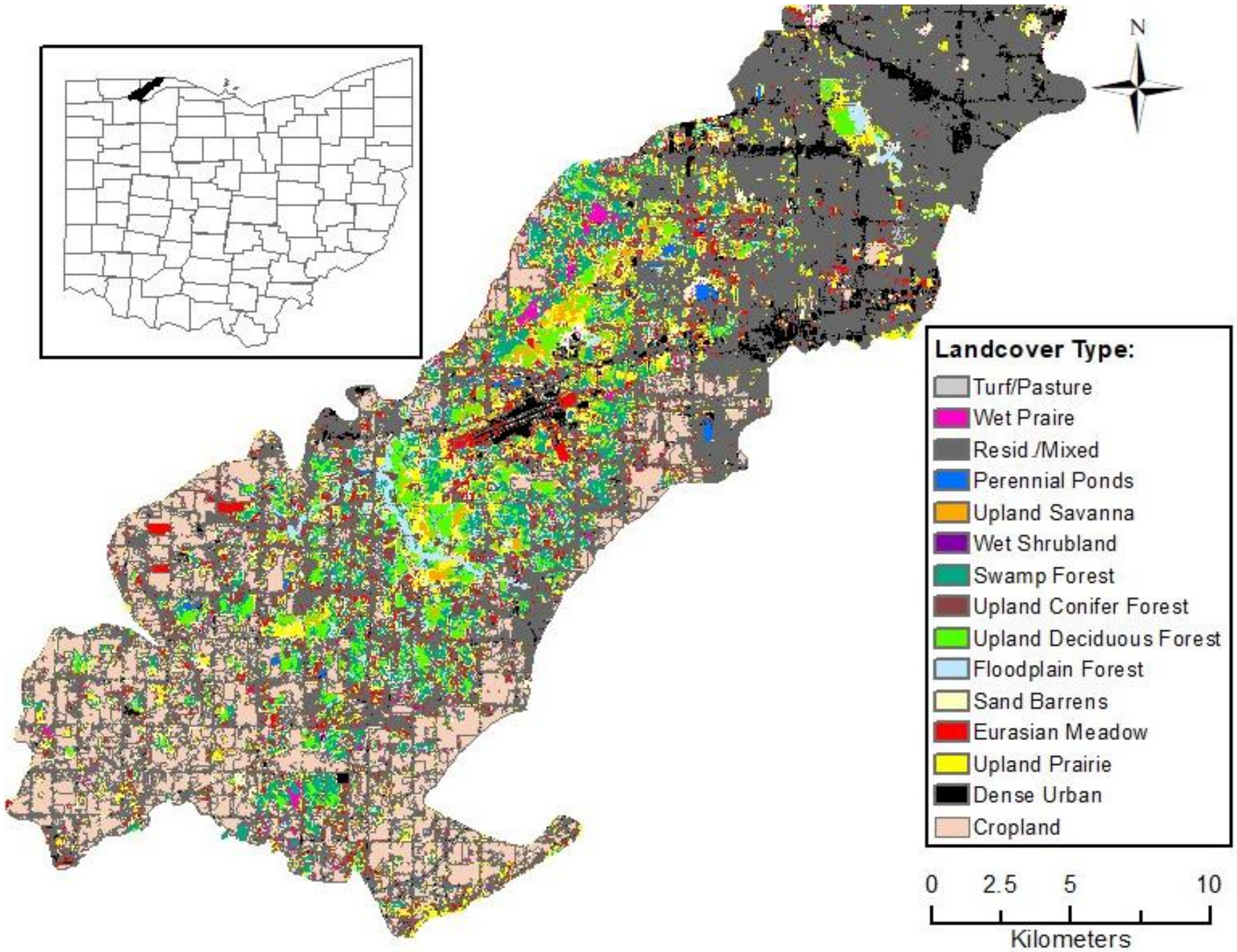


Figure 0.1. The Oak Opening Region in Northwest Ohio, with land use classified under a 15-category landcover map (developed by Martin & Root 2020), and Oak Openings Region location within the state of Ohio (insert).

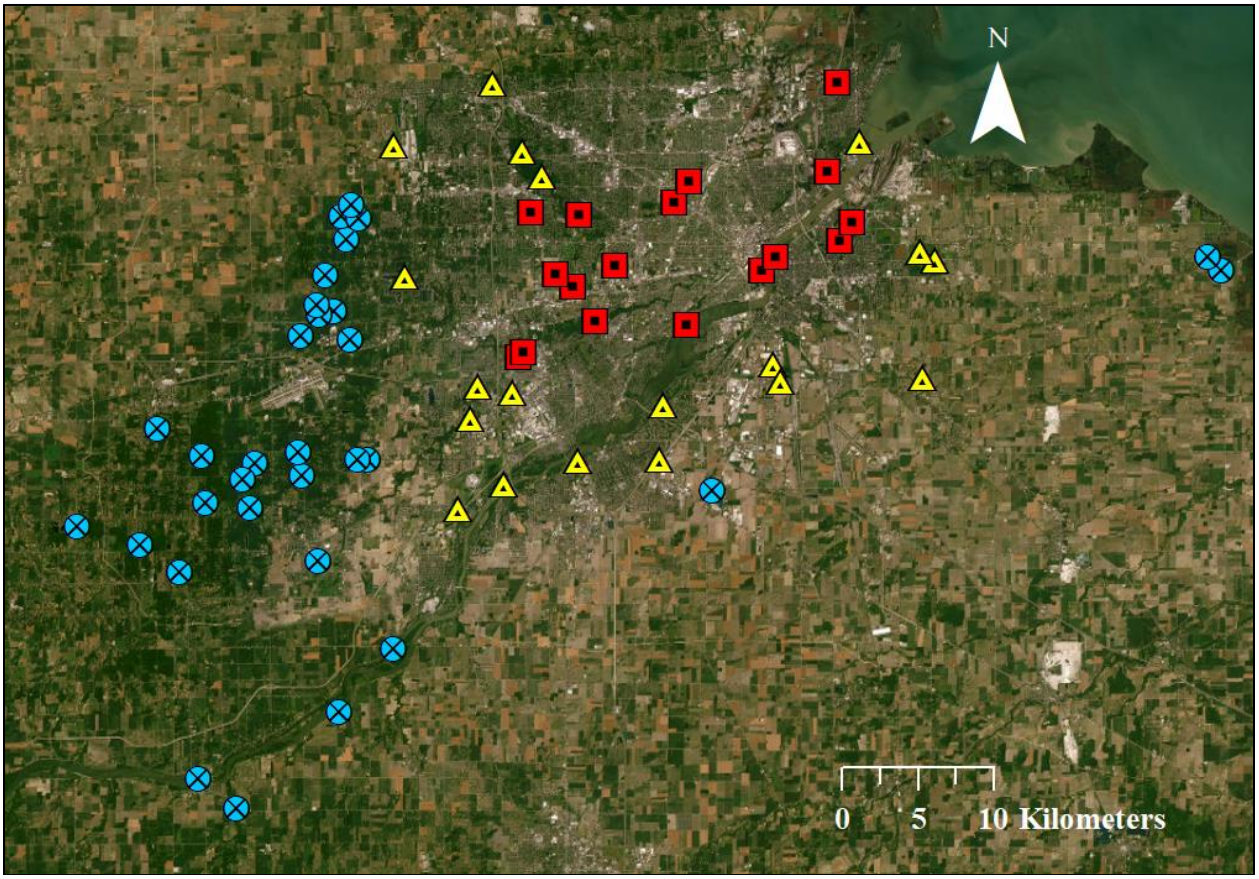


Figure 0.2. Map of sites across the Oak Openings Region and Toledo Metropolitan Area. 67 sites were utilized at least once, with 40-50 sites surveyed per year. Blue circles represent rural sites, yellow triangles represent suburban sites, and red squares represent urban sites.

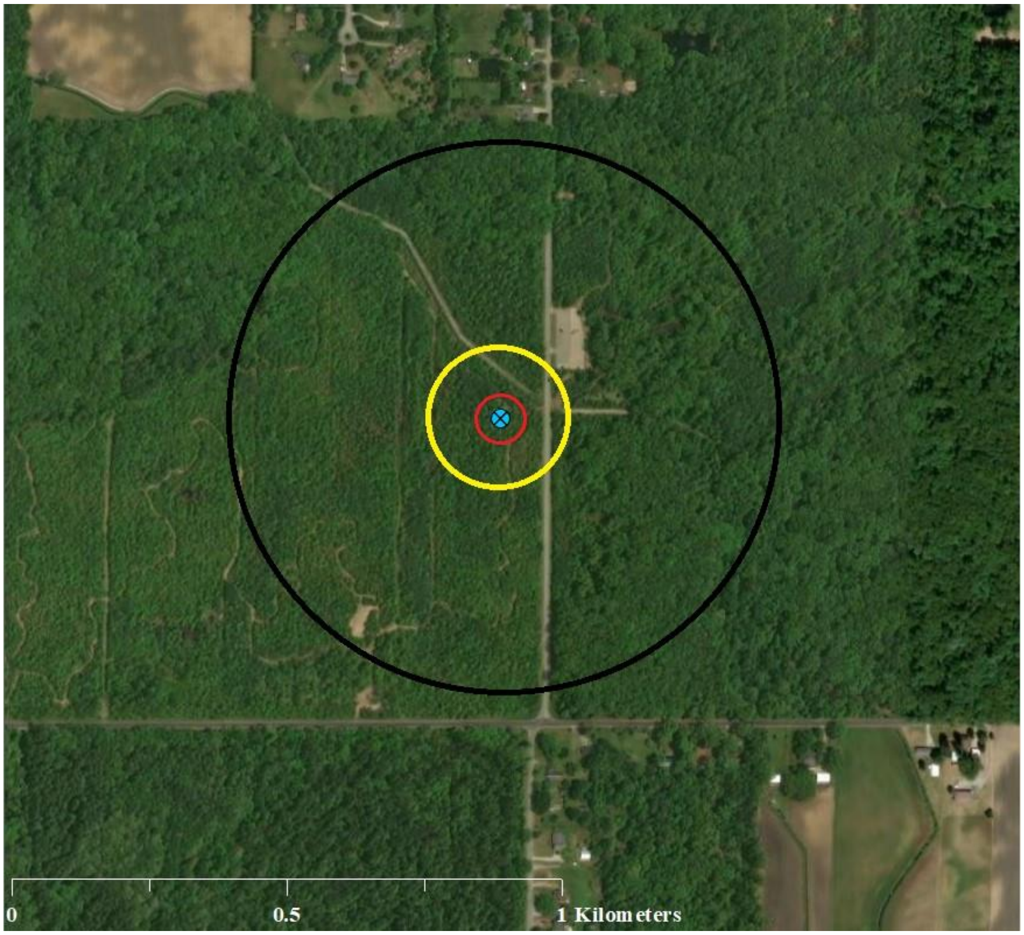


Figure 1.1. Example scale measures at a rural survey site. The red circle represents “local” scale, taken between 10 and 50m of the survey point. Yellow circle represents "landscape" scale or "large scale at 250m, and black circle represents "landscape" scale or "large" scale at 1km.

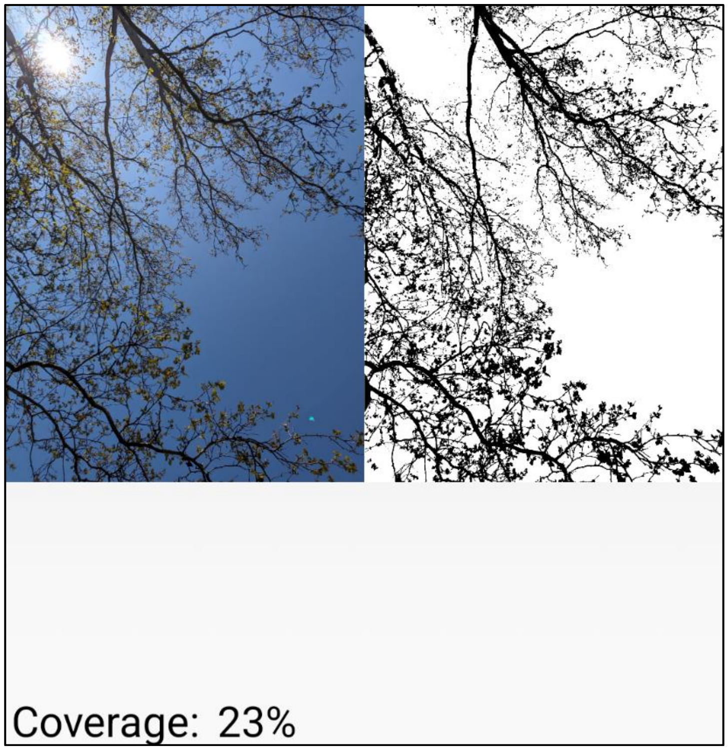


Figure 1.2. Sample of canopy cover measurement taken through HabitApp. The left photo is the color version, and the right is the black and white version with the corresponding amount of canopy cover as a percentage based on black pixels.

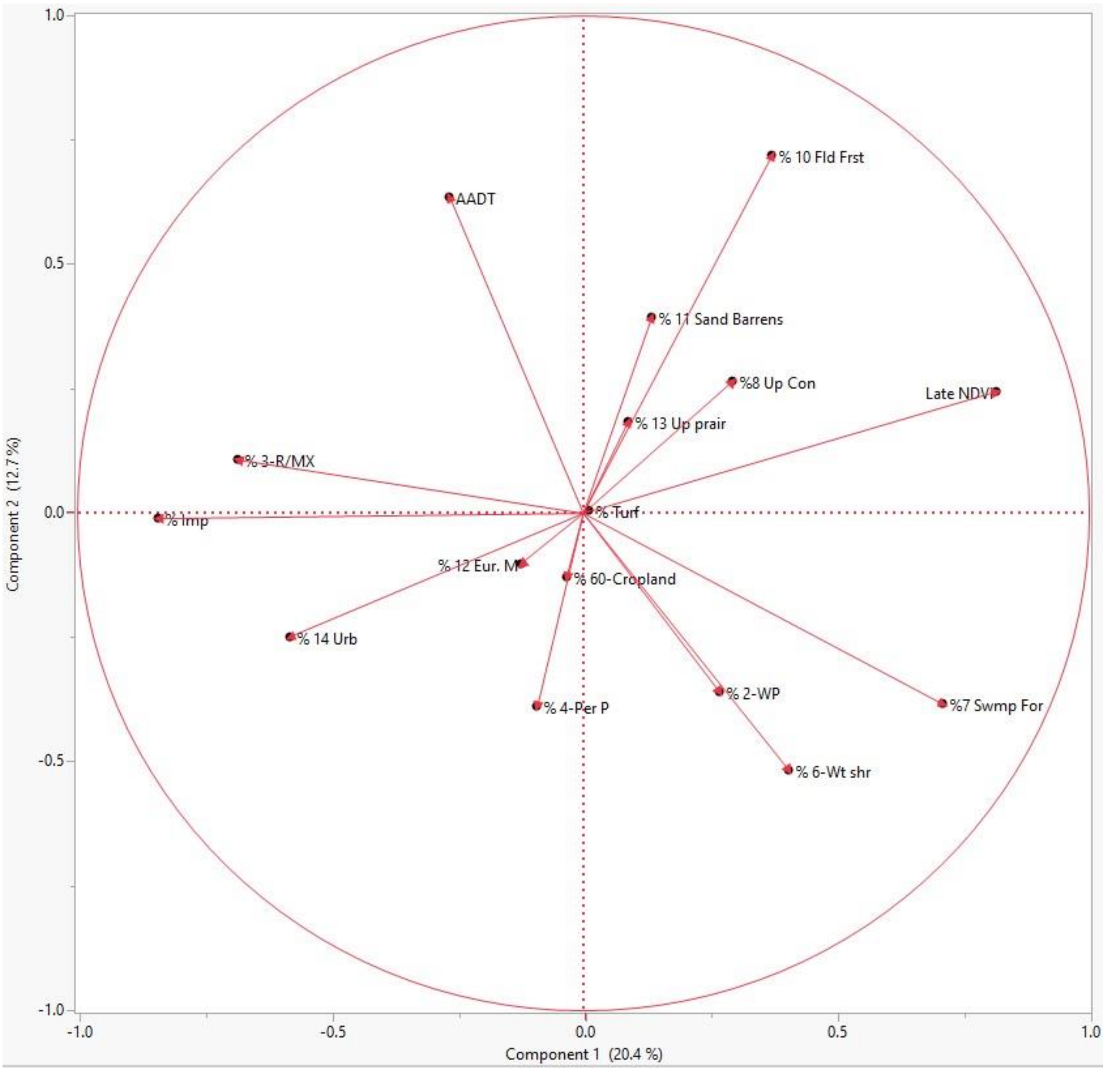


Figure 1.3. Example Principal Component Analysis utilizing landcover, traffic, and NDVI data. Each axis represents one Principal Component, a combination of scaled variables that explain a percentage of variability in the data set (in parentheses). More highly correlated variables form a more acute angle. AADT=Average Annual Daily Traffic, % Imp=Percentage of Impervious Surface, Late NDVI=June/July Normalized Difference Vegetation Index. All other variables are percentages of landcover, including “WP” (wet prairie), “R/Mx” (residential/mixed use), “Per P” (perennial ponds), “Wt Shr” (wet shrubland), “Swmp For” (swamp forest), “Up Con” (upland coniferous forest), “Fld Frst" (floodplain forest), Sand barrens, “Eur. M” (Eurasian meadow), “Up prair” (upland prairie), “Urb” (urban), and turf.

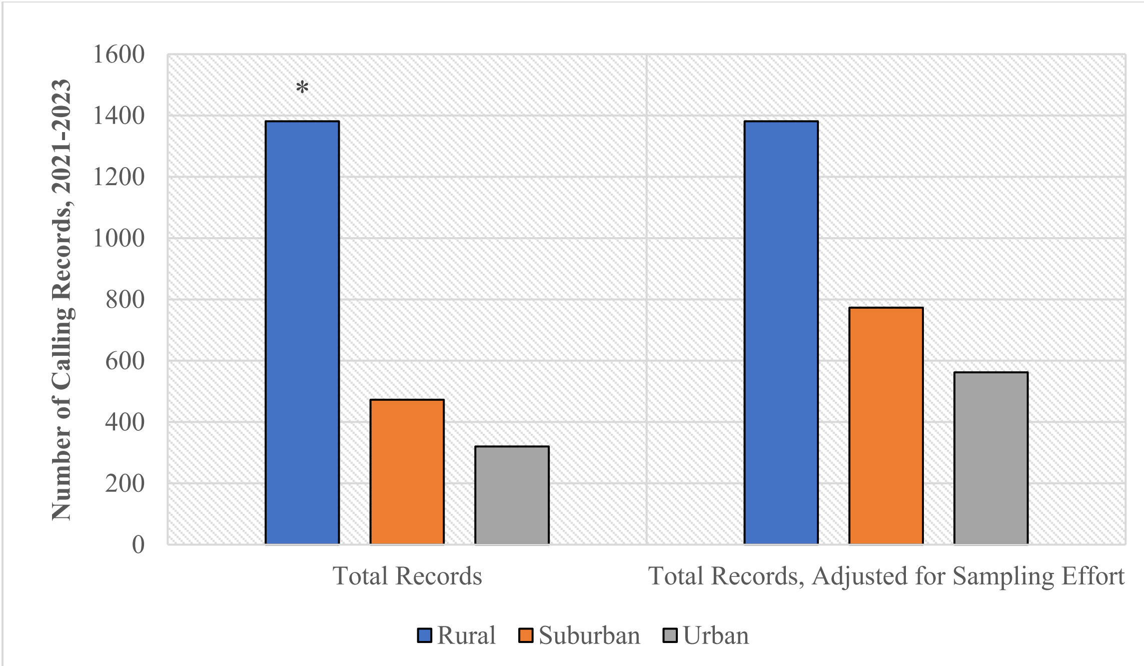
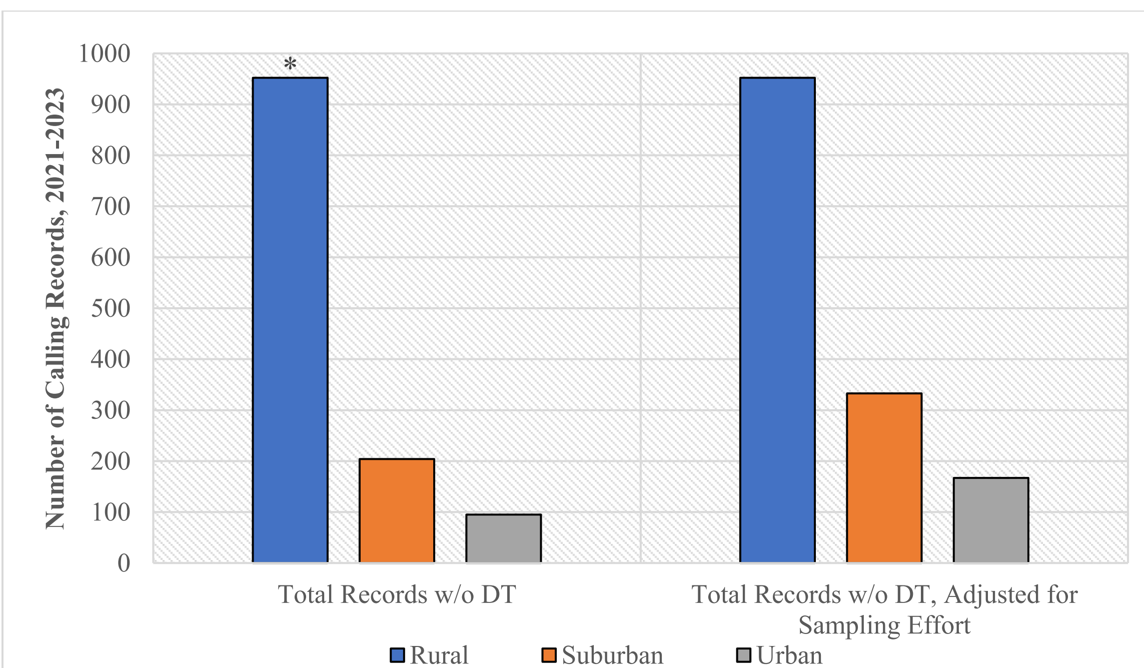


Figure 1.4. (Upper) Total number of Anuran species records across urbanization gradient class (left) and expected number of records across those classes if sampling effort were equal (right). (Lower) Number of Anuran species records across the urbanization gradient class without Disturbance-Tolerant species records (left) and when extrapolated for sampling effort (right). Asterisk represents significant difference at $p = 0.05$. Records adjusted for sampling effort were not tested for significant difference, as they were based on extrapolated data.



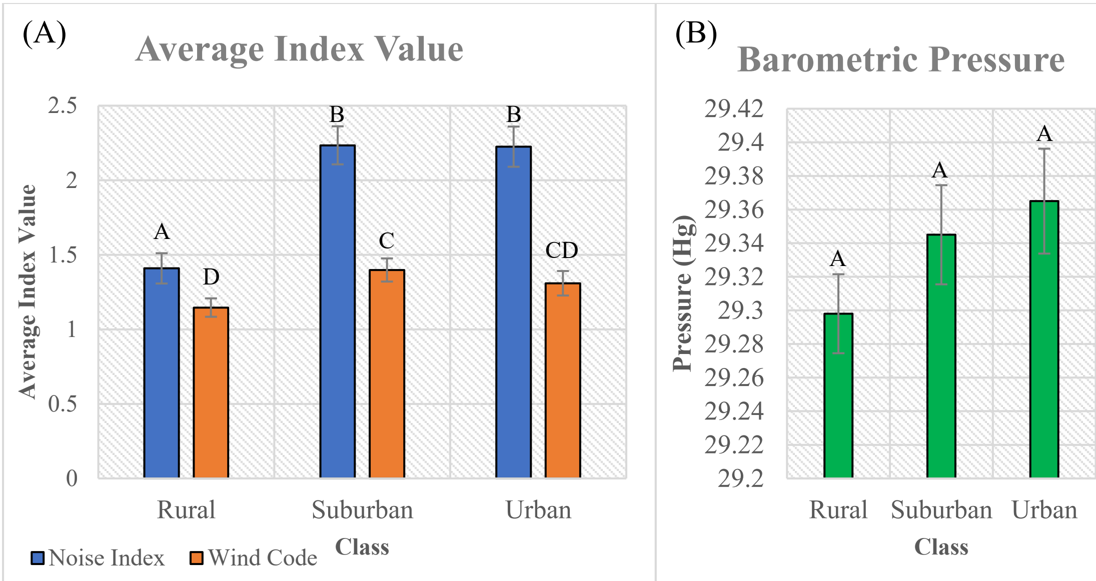


Figure 2.1. Significant differences in index values (A) and barometric pressure (B) between urbanization gradient classes across study period (2021-2023) using ANOVA and Kruskal-Wallis tests. Bars that share letters are not significantly different from one another at p = 0.05. Barometric pressure tests were not signficantly different between classes, but the overall test showed significant differences.

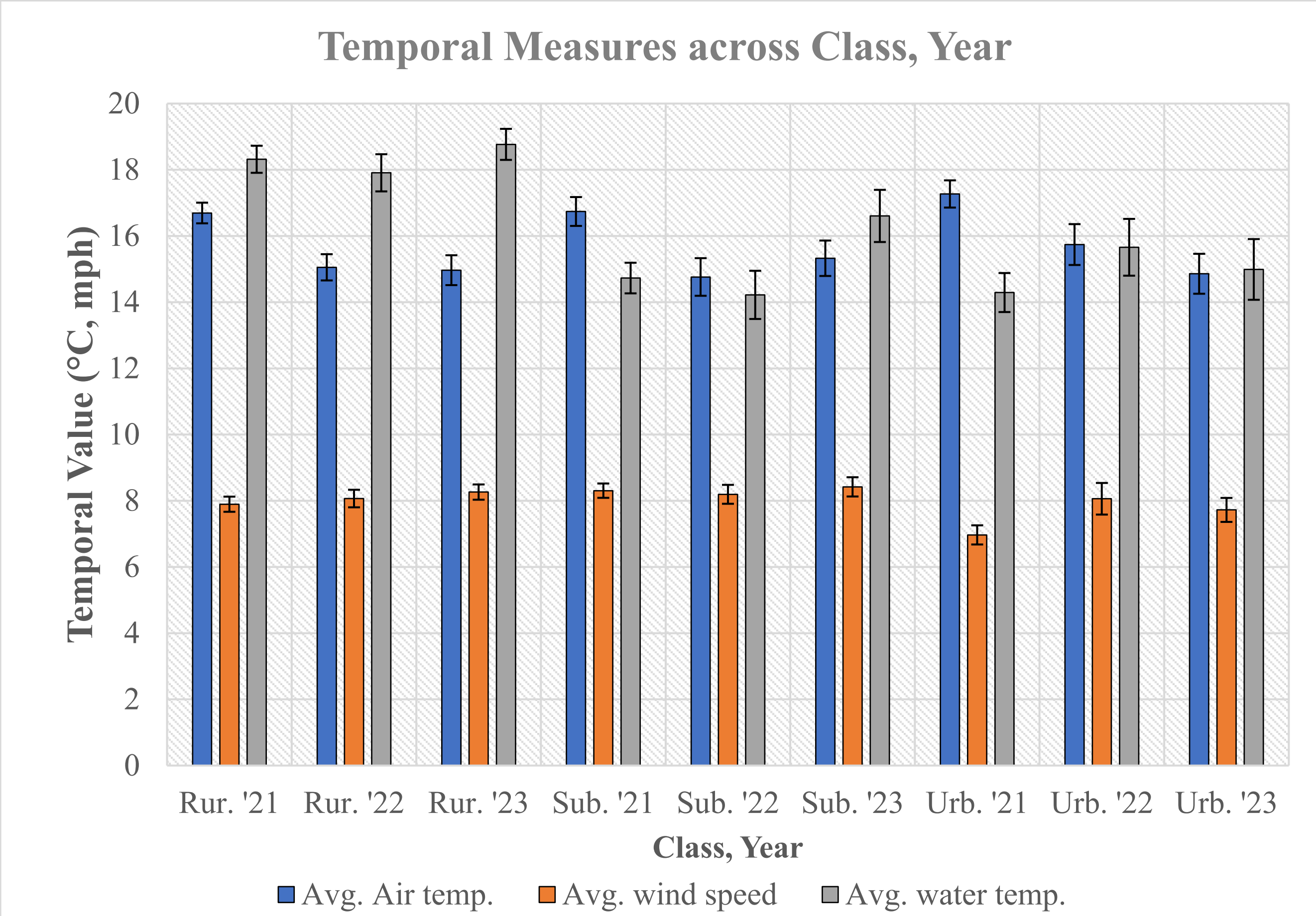


Figure 2.2. Differences between temporal measures across our study time period (2021-2023) during each field season. No measures were found to be significantly different between urbanization classes. Bars represent average of all measures taken within that urbanization class over one field season, with standard error of the mean.

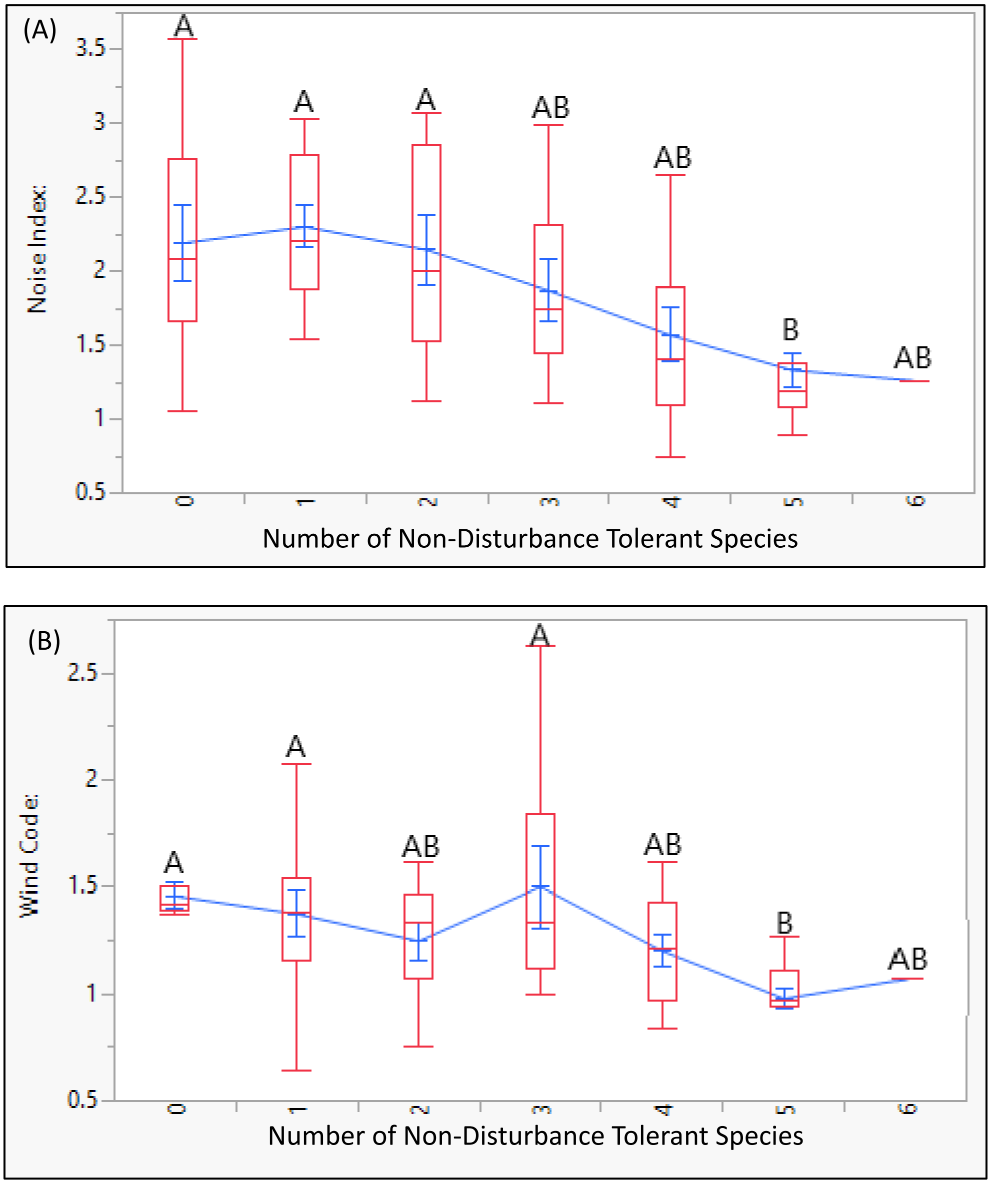
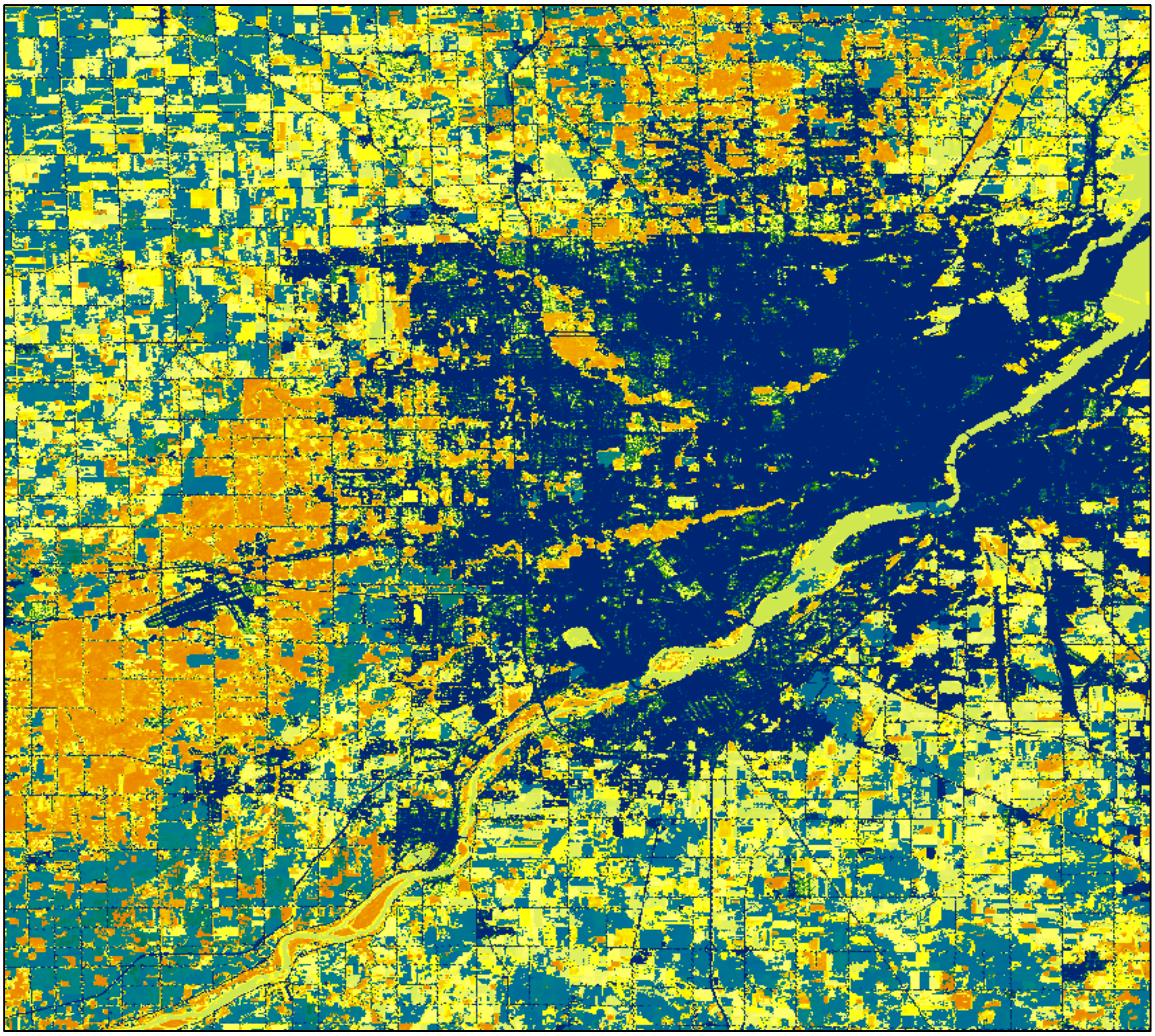


Figure 2.3. Box plots showing the general decline in the number of non-Disturbance Tolerant species as noise index (A) and wind code (B) increase. Plots that share letters are not significantly different at $p = 0.05$. Blue bars represent mean error bars, blue line is mean connection.



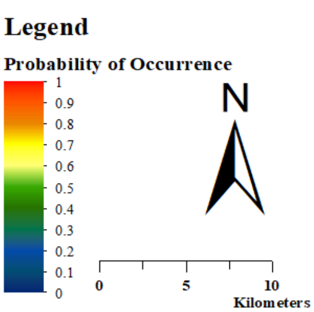


Figure 3.1. Maxent model results mapped as the probability of the occurrence of any eight species in the study area of Northwest Ohio. Map is a zoomed in version of the total study area, to allow for more detailed viewing. Each color shows the probability of occurrences from low probabilities (in blue) to high probabilities (in red), ranging from 0 to 1. The maximum number of species recorded at any site was nine (at one site) and eight (at 11 sites).

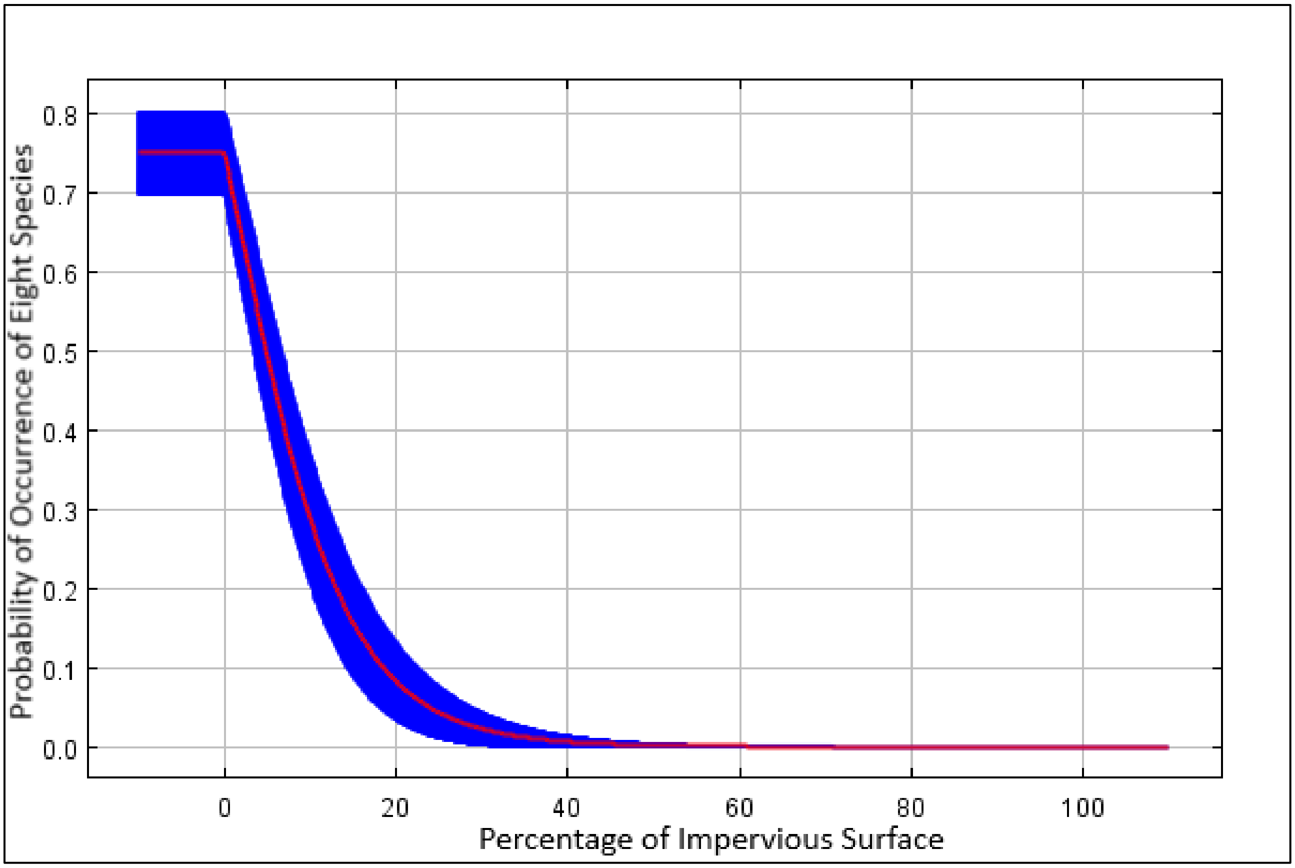


Figure 3.2. Response curve of the predicted probability of occurrence of eight species in the study area to the percentage of impervious surface. The x-axis represents the percentage of impervious surface on the landscape, 0% to 100%. The y-axis represents the probability of the occurrence of any eight species found in the area. The curve shows the mean response of the 10 replicate Maxent runs (red) and the mean +/- one standard deviation (blue).

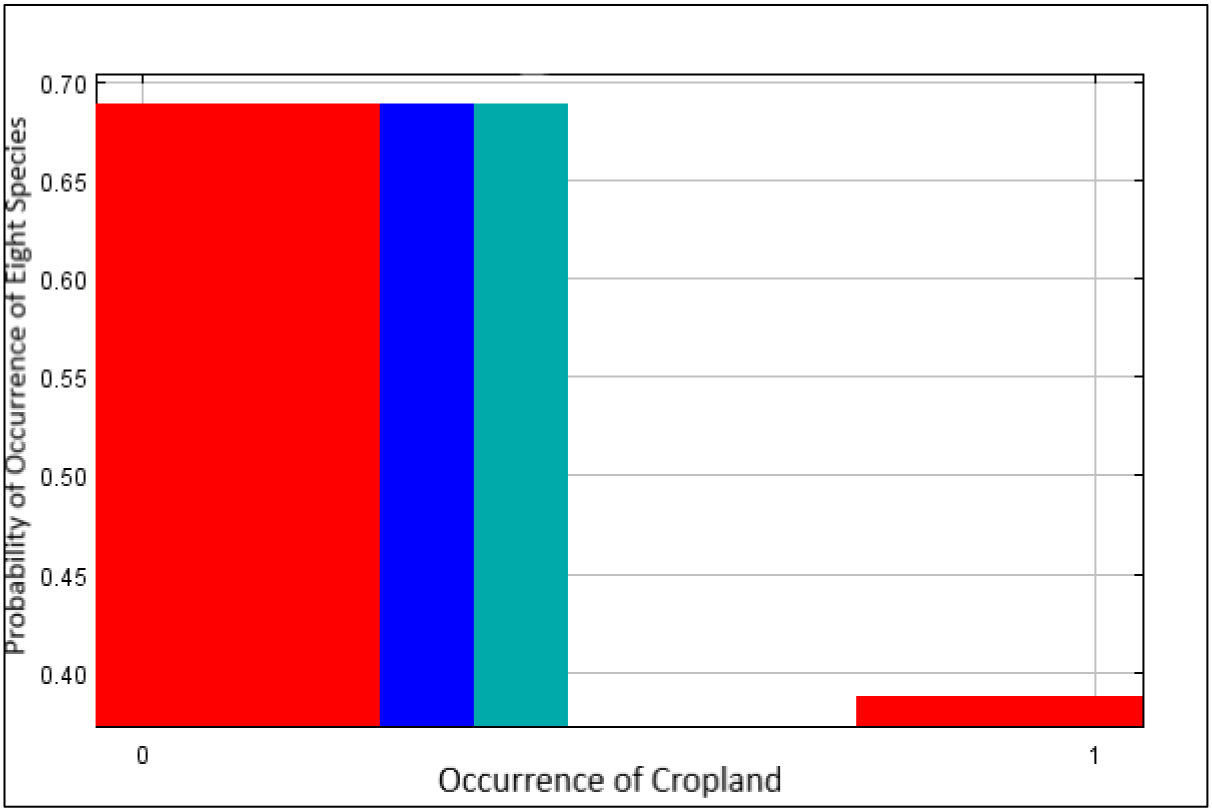
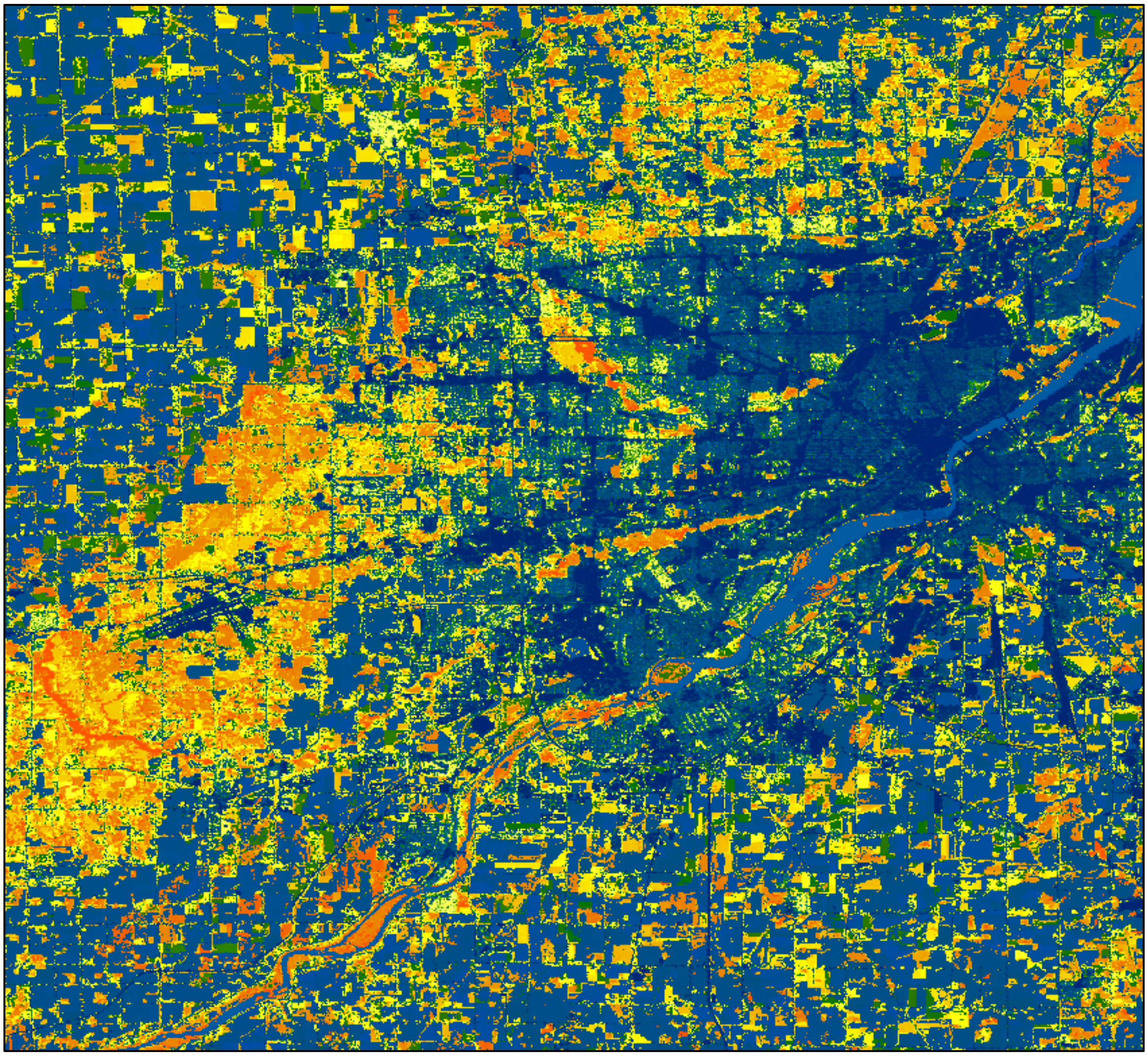


Figure 3.3. Response of the predicted probability of occurrence of eight species in the study area to the occurrence of cropland as a landcover type. The x-axis represents the occurrence of cropland on the landscape. The y-axis represents the probability of the occurrence of any eight species found in the area. The bars show the mean response of the 10 replicate Maxent runs (red) and the mean +/- one standard deviation (blue and green) in response to the presence of cropland.



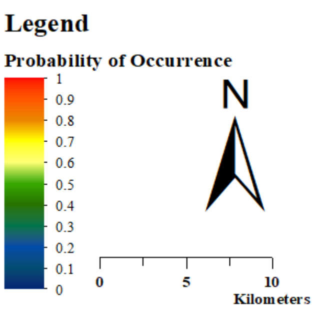
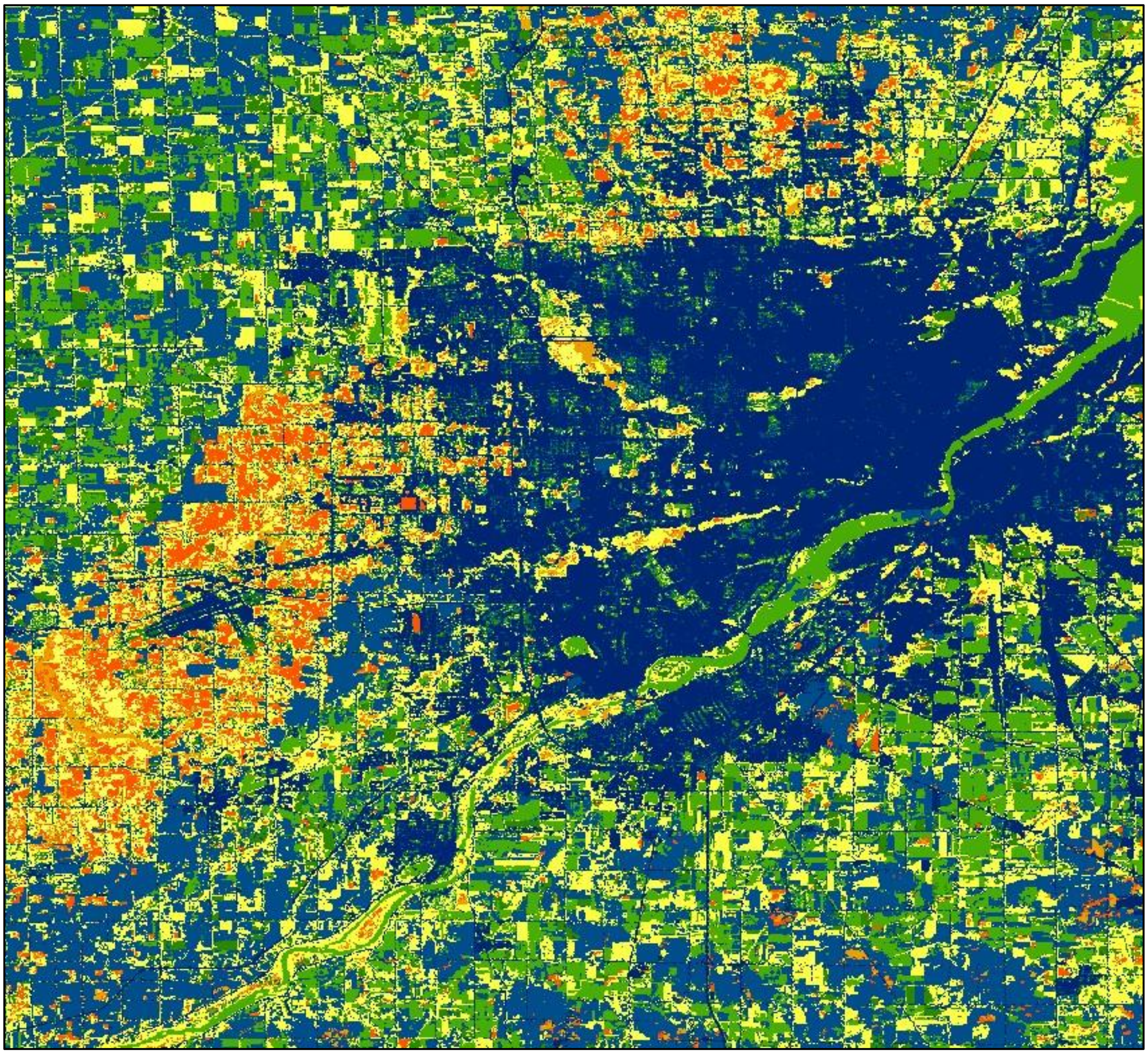


Figure 3.4. Maxent model results mapped as the probability of the occurrence of any seven species in the study area of Northwest Ohio. Map is a zoomed in version of the total study area, to allow for more detailed viewing. Each color shows the probability of occurrences from low probabilities (in blue) to high probabilities (in red), ranging from 0 to 1.



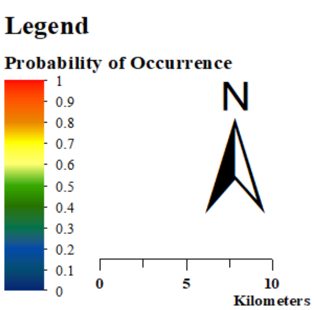
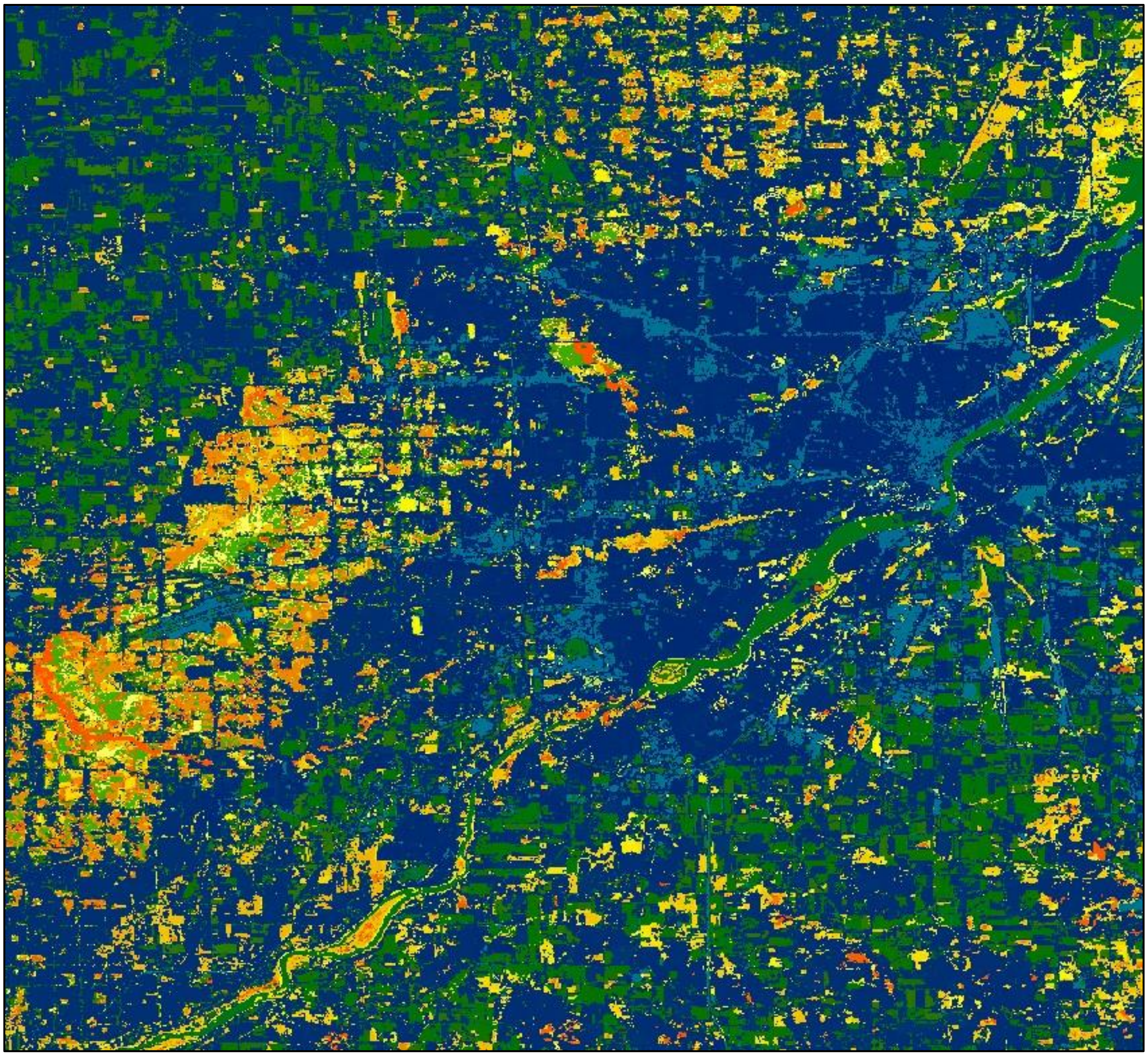


Figure 3.5. Maxent model results mapped as the probability of occurrence of any five specialist species (excluding *Anaxyrus americanus, Lithobates clamitans* and *Lithobates catesbeianus*) in the study area of Northwest Ohio. Map is a zoomed in version of the total study area, to allow for more detailed viewing. Color shows probability of occurrences from low probabilities (blue) to high (red), ranging from 0 to 1. The maximum number of any specialist species observed at a site was six.



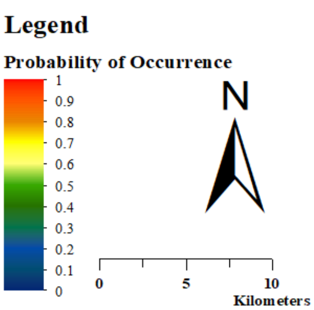
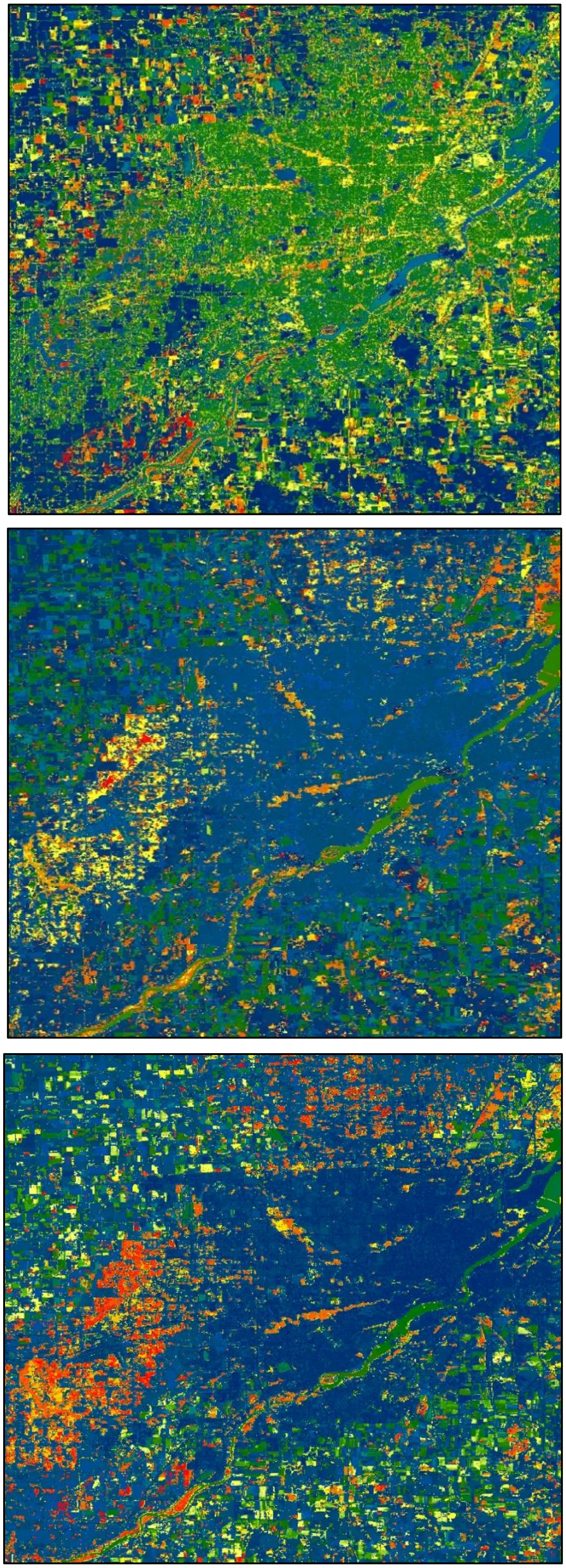


Figure 3.6. Maxent model results mapped as the probability of the occurrence of any four specialist species (excluding *Anaxyrus americanus, Lithobates clamitans* and *Lithobates catesbeianus*) in the study area of Northwest Ohio. Map is a zoomed in version of the total study area, to allow for more detailed viewing. Each color shows the probability of mortality occurrences from low probabilities (in blue) to high probabilities (in red), ranging from 0 to 1.



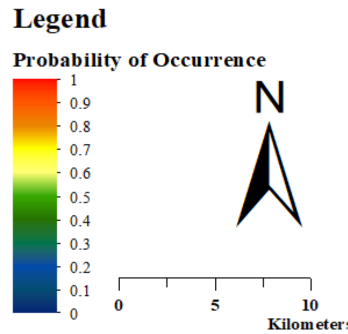
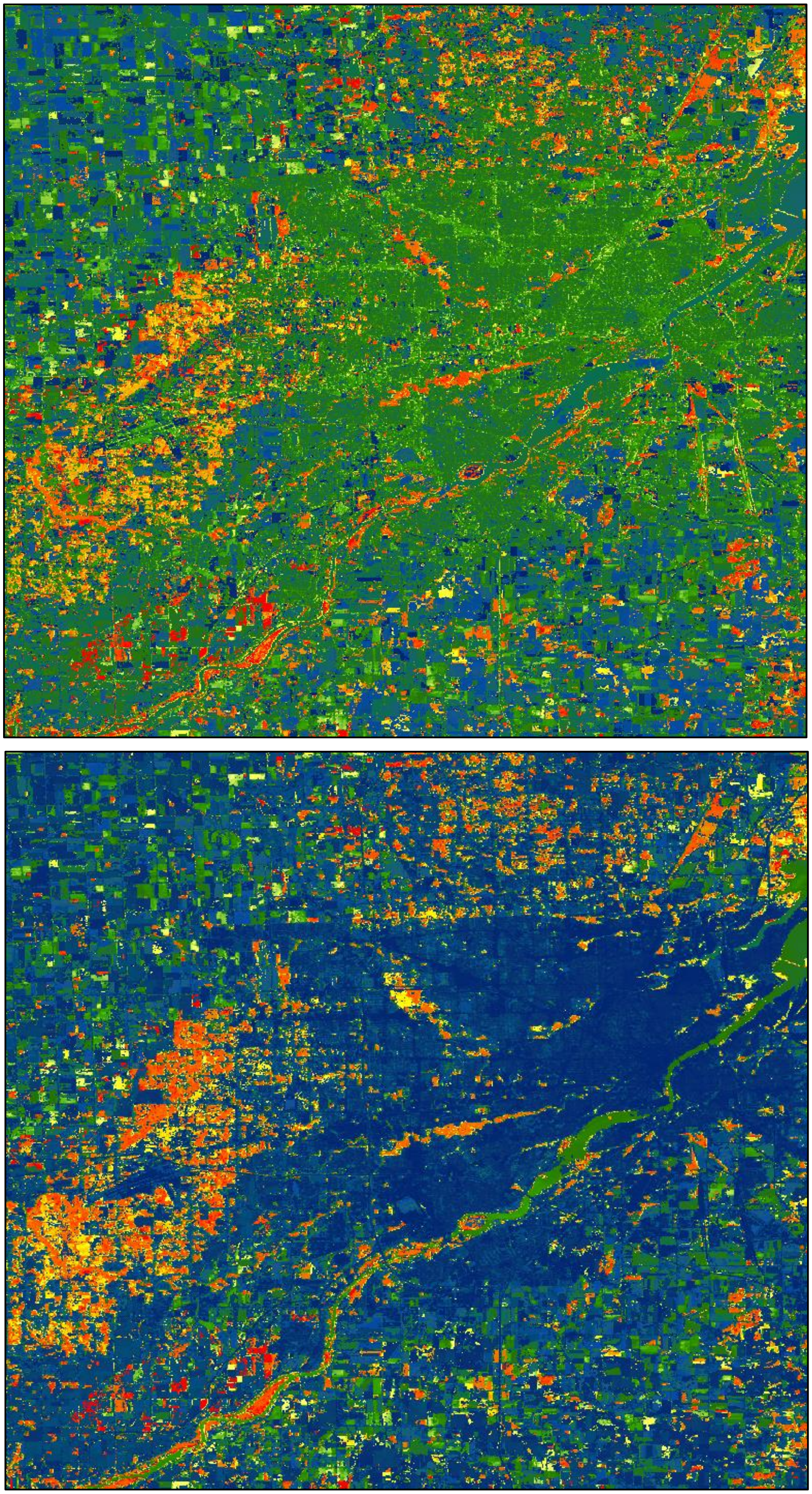


Figure 3.7. Maxent model results mapped as the probability of the occurrence of *A. blanchardi* (top), *L. pipiens* (middle), and *P. triseriata* (bottom) in the study area of Northwest Ohio. Maps are a zoomed in version of the study area, to allow for detailed viewing. Each color shows the probability of mortality occurrences from low probabilities to high.



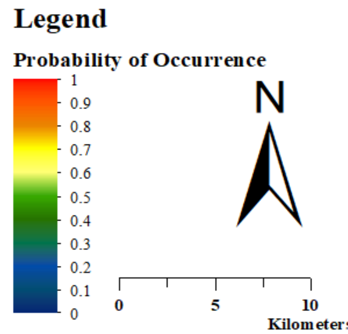


Figure 3.8. Maxent model results mapped as the probability of the occurrence of *L. catesbeianus* (top), and *P. crucifer* (bottom) in the study area of Northwest Ohio. Maps are a zoomed in version of the total study area, to allow for more detailed viewing. Each color shows the probability of mortality occurrences from low probabilities (in blue) to high (in red), ranging from 0 to 1.

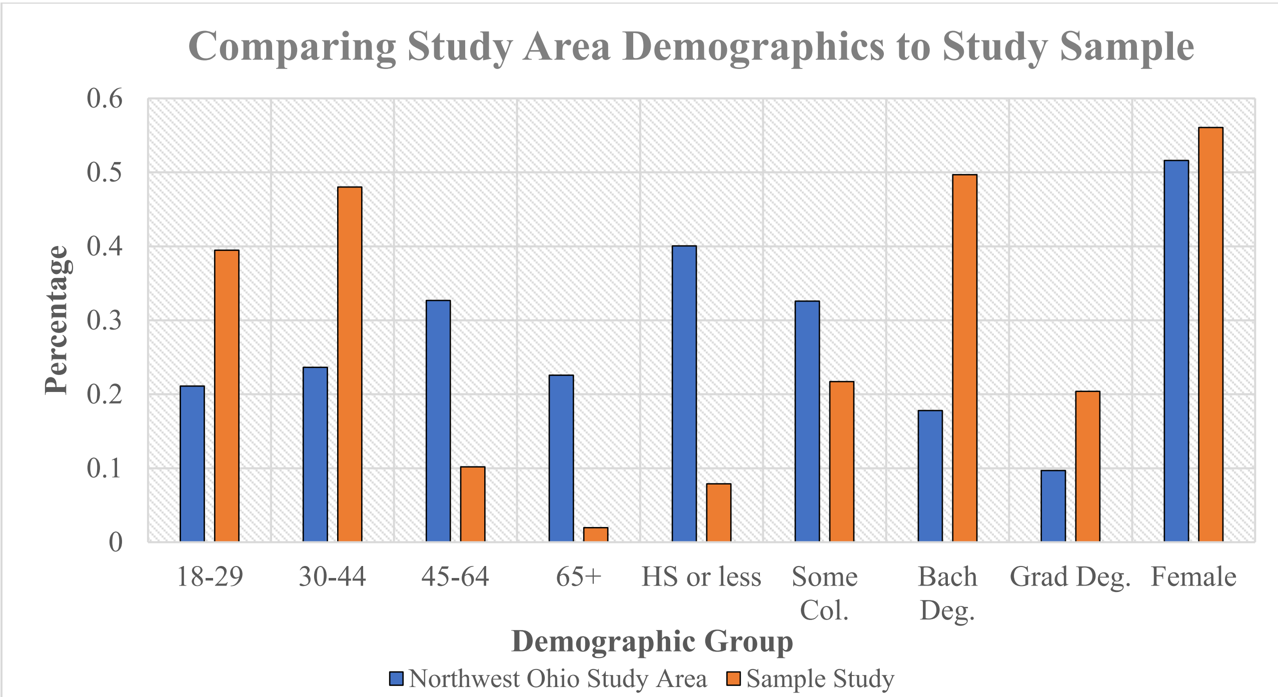


Figure 4.1. Demographic comparison of the Northwest Ohio study area to the demographics of our survey respondents (304 total). Data from the U.S. Census Bureau, 2020.

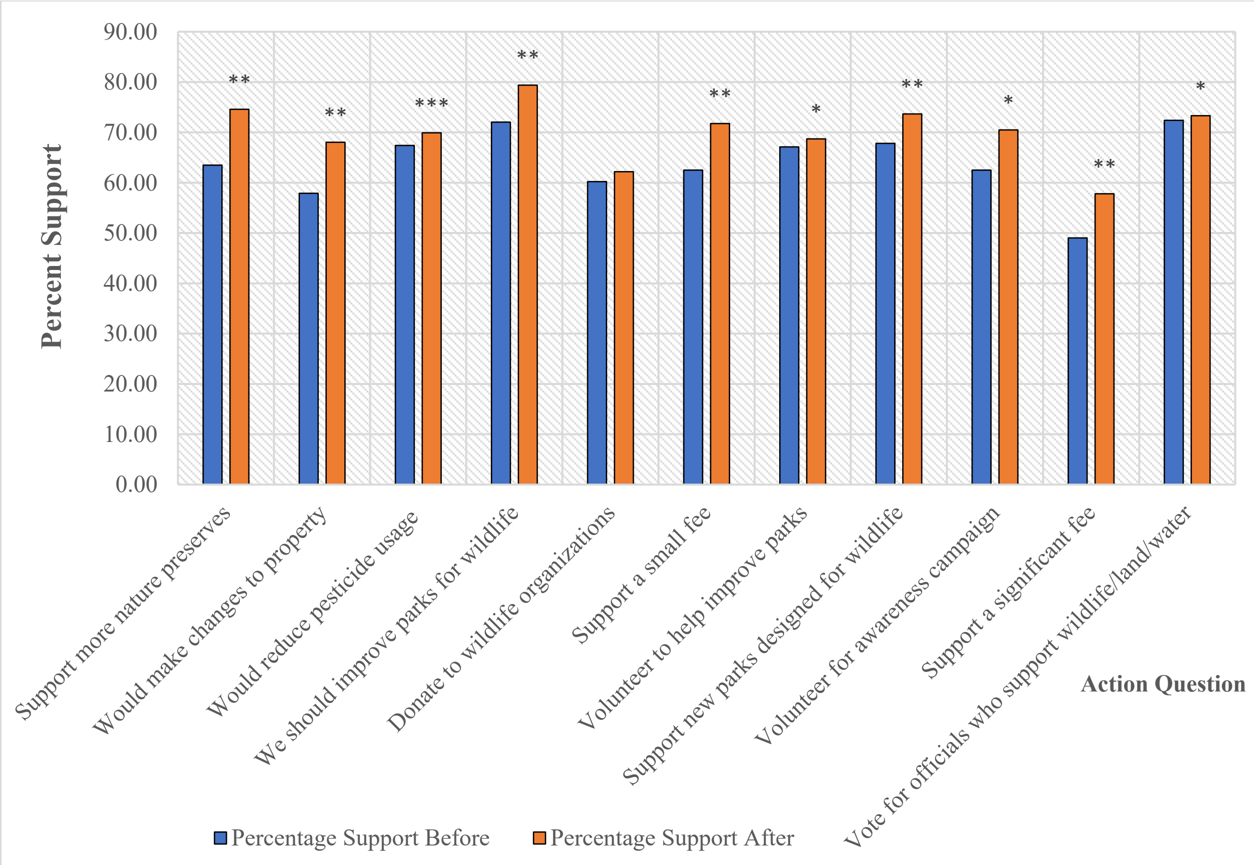


Figure 4.2. Notable results supported by the majority of respondents (agree or strongly agree) across all demographic groups, both before and after the informational presentation, using a Paired T-test. Significant difference between before and after surveys at $p = 0.05$ denoted by asterisk. Dual asterisks indicate significant difference at $p = 0.005$. Three asterisks indicate significant difference in a One-Tailed Paired T-test at $p = 0.05$.

APPENDIX C. SUPPLEMENTAL TABLES

Table S1.1. Generalized Linear Mixed Models for only combinations of variables at 250m scale, for species richness in Northwest Ohio study area. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC. Class refers to urbanization gradient classes (urban, suburban, or rural).

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Residential-Mixed + Class + %Swamp forest:					**+0**
(Intercept)	1.85	0.12	15.19	<0.001***	
%Residential	-0.43	0.20	-2.20	0.028*	
Class	-0.17	0.07	2.57	0.010*	
%Swamp forest	0.55	0.25	2.23	0.026*	
%Impervious + Class + %Swamp forest:					**+0.9**
(Intercept)	1.79	0.13	13.53	<0.001***	
%Impervious	-0.50	0.52	-0.97	0.330	
Class	-0.19	0.08	-2.29	0.022*	
%Swamp forest	0.65	0.25	2.62	0.009**	
%Impervious Surface + %Urban + %Residential-Mixed + Class					**+1.6**
(Intercept)	2.06	0.11	19.40	<0.001***	
%Impervious	0.51	0.61	0.84	0.402	
%Residential	-0.62	0.21	-2.89	0.004**	
%Urban	-0.71	0.33	-2.15	0.031*	
Class	-0.24	0.08	-2.89	0.004**	
Class + %Residential-Mixed + Late season NDVI					**+1.7**
(Intercept)	1.56	0.28	5.51	<0.001***	
Class	-0.19	0.07	-2.86	0.004**	
%Residential	-0.43	0.20	-2.12	0.034*	
Late NDVI	1.07	0.62	1.73	0.084.	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table S1.2. Generalized Linear Mixed Models for combinations of only variables at 1km scale, for richness in Northwest Ohio study area. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC.

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Residential-Mixed + %Urban					**+0**
(Intercept)	1.97	0.07	27.95	<0.001***	
%Residential 1km	-1.03	0.20	-5.22	<0.001***	
%Urban 1km	-0.86	0.40	-2.18	0.029*	
Average NDVI + %Residential-Mixed					**+0.1**
(Intercept)	1.36	0.28	4.84	<0.001***	
Avg NDVI 1km	2.22	1.02	2.19	0.029*	
%Residential 1km	-1.01	0.20	-4.98	<0.001***	
%Residential-Mixed + %Swamp forest					**+0.5**
(Intercept)	1.78	0.11	16.28	<0.001***	
%Residential 1km	-0.98	0.22	-4.47	<0.001***	
%Swamp forest 1km	0.98	0.46	2.13	0.032*	
Late season NDVI + %Residential-Mixed					**+0.9**
(Intercept)	1.37	0.31	4.47	<0.001***	
Late NDVI 1km	1.41	0.71	1.98	0.048*	
%Residential 1km	-0.92	0.24	-3.83	<0.001***	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table S1.3. Generalized Linear Mixed Models for combinations of variables at both 250m and 1km scale, for richness in Northwest Ohio study area. All variables not listed did not produce significant models. Variables without scale listed are taken at 250m. Models listed in order of ΔAIC. Class refers to urbanization gradient classes (urban, suburban, or rural).

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
Class + %Residential-Mixed + %Swamp Forest +Late season NDVI 1km					**+0**
(Intercept)	1.37	0.33	4.17	<0.001***	
Class	-0.12	0.07	-1.63	0.103	
%Residential	-0.37	0.20	-1.85	0.064 .	
%Swamp forest	0.46	0.25	1.83	0.067 .	
Late NDVI 1km	1.15	0.74	1.57	0.118	
Class + %Wet Prairie + %Residential-Mixed + Late season NDVI 1km					**+0.1**
Class	-0.12	0.08	-1.64	0.102	
%Residential	-0.32	0.20	-1.58	0.113	
%Swamp Forest	0.42	0.26	1.66	0.098 .	
%Wet prairie	0.44	0.31	1.42	0.157	
Late NDVI 1km	1.23	0.74	1.65	0.099 .	
Class + %Residential-Mixed + %Swamp Forest					**+0.6**
(Intercept)	1.85	0.12	15.19	<0.001***	
Class	-0.17	0.07	-2.57	0.010*	
%Residential	-0.43	0.19	-2.20	0.028*	
%Swamp forest	0.55	0.25	2.23	0.026*	
%Residential-Mixed + %Swamp Forest + Late season NDVI 1km					**+0.7**
(Intercept)	1.01	0.25	4.04	<0.001***	
%Residential	-0.48	0.19	-2.57	0.010*	
%Swamp Forest	0.54	0.25	2.20	0.028*	
Late NDVI 1km	1.69	0.67	2.51	0.012*	
%Wet Prairie + %Residential-Mixed + %Swamp Forest + Late season NDVI 1km					**+0.8**
(Intercept)	0.95	0.25	3.73	<0.001***	
%Residential	-0.43	0.19	-2.28	0.022*	
%Swamp forest	0.51	0.25	2.02	0.043*	
Late NDVI 1km	1.77	0.68	2.61	0.009**	
%Wet prairie	0.44	0.31	1.41	0.159	
%Swamp forest + % Impervious 1km					**+0.9**

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
(Intercept)	1.63	0.08	20.69	<0.001***	
% Swamp Forest	0.67	0.24	2.80	0.006**	
% Impervious1km	-1.30	0.30	-4.29	<0.001***	
Class + %Wet Prairie + %Residential-Mixed + %Swamp Forest					**+1.0**
(Intercept)	1.83	0.12	14.82	<0.001***	
Class	-0.18	0.07	-2.61	0.009**	
%Residential	-0.39	0.20	-1.96	0.050*	
%Swamp forest	0.52	0.25	2.09	0.037*	
%Wet prairie	0.41	0.31	1.30	0.193	
Class + Late season NDVI + %Residential-Mixed + %Swamp forest + %Wet Prairie					**+1.1**
(Intercept)	1.49	0.28	5.25	<0.001***	
Class	-0.16	0.07	-2.38	0.017*	
Late NDVI	0.87	0.65	1.35	0.180	
%Residential	-0.32	0.20	-1.58	0.114	
%Swamp forest	0.41	0.26	1.57	0.120	
%Wet prairie	0.47	0.32	1.48	0.140	
Class + %Residential-Mixed + %Wet Prairie + %Swamp forest + Late season NDVI 1km + %Impervious 1km					**+1.9**
(Intercept)	1.34	0.33	4.01	<0.001***	
Class	-0.08	0.12	-0.66	0.510	
%Residential	-0.30	0.20	-1.48	0.139	
%Swamp forest	0.43	0.26	1.69	0.091 .	
%Wet prairie	0.42	0.31	1.35	0.177	
Late NDVI 1km	1.07	0.80	1.34	0.181	
%Impervious 1km	-0.33	0.65	-0.50	0.617	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table S1.4. Generalized Linear Mixed Models for combinations of only variables at 250m scale, for average species per survey in Northwest Ohio study area. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC. Class refers to urbanization gradient classes (urban, suburban, or rural).

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
Class					
(Intercept)	0.92	0.23	4.00	<0.001***	**+0**
Class	-0.45	0.12	-3.86	<0.001***	
Class + Late season NDVI					**+0.8**
(Intercept)	0.24	0.66	0.37	0.714	
Class	-0.38	0.13	-2.82	0.005**	
Late season NDVI	1.57	1.43	1.10	0.271	
Class + %Residential-Mixed					**+1.3**
(Intercept)	0.89	0.23	3.79	<0.001**	
Class	-0.36	0.16	-2.31	0.021*	
%Residential	-0.40	0.47	-0.85	0.397	
Class + %Impervious					**+1.9**
(Intercept)	0.88	0.26	3.34	<0.001***	
Class	-0.40	0.19	-2.12	0.034 *	
%Impervious	-0.42	1.21	-0.34	0.732	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table S1.5. Generalized Linear Mixed Models for combinations of only variables at 1km scale, for average species per survey in Northwest Ohio study area. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC.

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Impervious + %Swamp forest					**+0**
(Intercept)	0.23	0.21	1.07	0.286	
%Impervious 1km	-1.43	0.70	-2.06	0.039*	
%Swamp Forest 1km	2.02	1.30	1.55	0.120	
%Residential-Mixed + %Swamp forest					**+0.8**
(Intercept)	0.35	0.29	1.20	0.230	
%Residential 1km	-0.95	0.53	-1.81	0.071 .	
%Swamp Forest 1km	2.09	1.34	1.56	0.118	
Average NDVI + %Residential					**+1.1**
(Intercept)	-0.27	0.66	-0.41	0.686	
Avg NDVI 1km	3.74	2.43	1.54	0.124	
%Residential 1km	-1.17	0.48	-2.45	0.014*	
%Residential-Mixed + %Urban					**+1.1**
(Intercept)	0.74	0.20	3.78	<0.001***	
%Residential 1km	-1.21	0.47	-2.59	0.010**	
%Urban 1km	-1.39	0.89	-1.57	0.116	
Late season NDVI + %Residential-Mixed					**+1.2**
(Intercept)	-0.28	0.68	-0.41	0.682	
Late NDVI 1km	2.42	1.60	1.52	0.130	
%Residential 1km	-1.00	0.54	-1.85	0.064 .	
Late season NDVI + %Swamp forest					**+1.4**
(Intercept)	-0.86	0.46	-1.85	0.064 .	
Late NDVI 1km	2.52	1.54	1.64	0.101	
%Swamp Forest 1km	2.25	1.33	1.69	0.091 .	
Late season NDVI + %Impervious					**+1.4**
(Intercept)	-0.29	0.71	-0.40	0.690	
Late NDVI 1km	2.02	1.82	1.11	0.267	
%Impervious 1km	-1.43	0.82	-1.75	0.081 .	
%Impervious + %Residential-Mixed					**+1.6**
(Intercept)	0.63	0.20	3.18	0.001**	

Variable	Estimate	Std. Error	Z	P	ΔAIC
%Impervious 1km	-1.35	1.00	-1.37	0.172	
%Residential 1km	-0.69	0.72	-0.96	0.337	
%Swamp forest + %Impervious + Late season NDVI					**+1.7**
(Intercept)	-0.19	0.72	-0.26	0.795	
%Swamp Forest 1km	1.76	1.37	1.29	0.199	
%Impervious 1km	-1.13	0.85	-1.33	0.185	
Late NDVI 1km	1.16	1.93	0.60	0.548	
%Swamp forest + %Upland deciduous forest					**+1.8**
(Intercept)	-0.18	0.13	-1.40	0.162	
%Swamp forest 1km	2.84	1.15	2.47	0.014*	
%Upland decid. 1km	2.03	1.41	1.44	0.150	
Average NDVI + %Swamp forest					**+1.9**
(Intercept)	-0.85	0.54	-1.58	0.115	
Avg NDVI 1km	3.51	2.53	1.39	0.165	
%Swamp Forest 1km	2.59	1.27	2.04	0.042*	
Average NDVI + %Impervious					**+1.9**
(Intercept)	-0.12	0.76	-0.15	0.878	
Avg NDVI 1km	2.37	2.87	0.83	0.410	
%Impervious 1km	-1.68	0.76	-2.21	0.027*	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table S1.6. Generalized Linear Mixed Models for combinations of variables at both 250m and 1km scale, for average species per survey in Northwest Ohio study area. Models only included with a ΔAIC ≤2. Variables with scale unlabeled are taken at 250m scale. Models listed in order of ΔAIC. Class refers to urbanization gradient classes (urban, suburban, or rural).

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
Class + %Swamp forest 1km					**+0**
(Intercept)	0.54	0.35	1.55	0.121	
Class	-0.31	0.15	-2.08	0.038*	
%Swamp forest 1km	1.90	1.33	1.42	0.154	
%Swamp Forest 1km + %Impervious 1km					**+0.2**
(Intercept)	0.23	0.21	1.07	0.286	
%Impervious 1km	-1.43	0.70	-2.06	0.040*	
%Swamp forest 1km	2.02	1.30	1.55	0.121	
Class + %Urban 1km					**+0.7**
(Intercept)	0.89	0.23	3.82	<0.001***	
Class	-0.37	0.14	-2.68	0.007**	
%Urban 1km	-1.13	0.93	-1.22	0.224	
Class + Late season NDVI					**+0.9**
(Intercept)	0.24	0.66	0.37	0.713	
Class	-0.38	0.13	-2.82	0.005**	
Late season NDVI	1.57	1.43	1.10	0.271	
Class + Late season NDVI 1km					**+0.9**
(Intercept)	0.24	0.66	0.37	0.714	
Class	-0.38	0.13	-2.82	0.005**	
Late NDVI	1.57	1.43	1.10	0.271	
%Residential-Mixed 1km + %Swamp Forest 1km					**+1.0**
(Intercept)	0.35	0.29	1.20	0.232	
%Residential 1km	-0.95	0.53	-1.81	0.071 .	
%Swamp forest 1km	2.09	1.34	1.56	0.118	
Class + %Swamp forest					**+1.1**
(Intercept)	0.72	0.31	2.33	0.020*	
Class	-0.38	0.14	-2.74	0.006**	
%Swamp forest	0.69	0.71	0.97	0.332	
%Residential-Mixed 1km + %Urban 1km					**+1.3**
(Intercept)	0.74	0.20	3.78	<0.001***	

Variable(s)	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Residential 1km	-1.21	0.47	-2.59	0.010**	
%Urban 1km	-1.39	0.89	-1.57	0.116	
Class + %Residential-Mixed					**+1.4**
(Intercept)	0.89	0.23	3.79	<0.001***	
Class	-0.36	0.16	-2.31	0.021*	
%Residential	-0.40	0.47	-0.85	0.397	
%Swamp Forest + %Impervious					**+1.4**
(Intercept)	0.35	0.19	1.83	0.067 .	
%Swamp forest	0.79	0.69	1.14	0.254	
%Impervious 1km	-1.74	0.63	-2.74	0.006**	
%Residential-Mixed + Late season NDVI 1km					**+1.4**
(Intercept)	-0.28	0.68	-0.41	0.682	
Late NDVI 1km	2.42	1.60	1.52	0.130	
%Residential 1km	-1.00	0.53	-1.85	0.064 .	
%Impervious 1km + Late season NDVI 1km					**+1.6**
(Intercept)	-0.28	0.71	-0.40	0.690	
Late NDVI 1km	2.02	1.82	1.11	0.267	
%Impervious 1km	-1.43	0.82	-1.75	0.081 .	
%Swamp Forest 1km + Late season NDVI 1km					**+1.6**
(Intercept)	-0.86	0.46	-1.85	0.064 .	
Late NDVI 1km	2.52	1.54	1.64	0.101	
%Swamp Forest 1km	2.25	1.33	1.69	0.091 .	
Class + %Upland savanna 1km					**+1.6**
(Intercept)	0.80	0.29	2.75	0.006**	
Class	-0.41	0.13	-3.05	0.002**	
%Upland sav. 1km	1.54	2.35	0.65	0.514	
Class + %Residential-Mixed 1km					**+1.7**
(Intercept)	0.90	0.23	3.83	<0.001***	
Class	-0.32	0.24	-1.37	0.171	
%Residential 1km	-0.52	0.83	-0.62	0.533	
%Upland Prairie + %Impervious 1km					**+1.7**
(Intercept)	0.61	0.18	3.38	<0.001***	
Upland Prairie	-0.65	0.62	-1.05	0.293	
%Impervious 1km	-2.15	0.56	-3.86	<0.001***	
Class + Average NDVI + Late season NDVI					**+1.7**
(Intercept)	0.19	0.64	0.30	0.765	
Class	-0.33	0.14	-2.34	0.019*	

Variable(s)	Estimate	Std. Error	Z	P	ΔAIC
Avg NDVI	-5.38	4.89	-1.10	0.271	
Late NDVI	5.09	3.44	1.48	0.140	
Class + Average NDVI					**+1.8**
(Intercept)	0.63	0.60	1.04	0.298	
Class	-0.43	0.12	-3.46	<0.001***	
Average NDVI	1.08	2.03	0.53	0.595	
Class + %Swamp Forest 1km + %Impervious 1km					**+1.8**
(Intercept)	0.45	0.40	1.12	0.264	
Class	-0.20	0.30	-0.65	0.514	
%Impervious 1km	-0.62	1.44	-0.43	0.668	
%Swamp for. 1km	1.84	1.33	1.38	0.166	
%Impervious + %Swamp forest 1km					**+1.8**
(Intercept)	0.11	0.19	0.56	0.574	
%Impervious	-1.41	0.89	-1.60	0.111	
%Swamp for. 1km	2.51	1.27	1.99	0.047*	
%Residential-Mixed 1km + %Impervious 1km					**+1.8**
(Intercept)	0.63	0.20	3.18	0.002**	
%Impervious 1km	-1.35	0.99	-1.37	0.172	
%Residential 1km	-0.70	0.72	-0.96	0.337	
%Residential-Mixed + %Impervious 1km					**+1.8**
(Intercept)	0.55	0.15	3.66	<0.001***	
%Residential 1km	-0.45	0.46	-0.98	0.326	
%Impervious 1km	-1.66	0.73	-2.26	0.024*	
Class + Early season NDVI 1km					**+1.8**
(Intercept)	0.63	0.58	1.09	0.275	
Class	-0.43	0.12	-3.49	<0.001***	
Early NDVI 1km	1.74	3.21	0.54	0.587	
Class + %Impervious 1km					**+1.8**
(Intercept)	0.79	0.33	2.39	0.017*	
Class	-0.29	0.30	-0.98	0.325	
%Impervious 1km	-0.80	1.45	-0.55	0.582	
%Residential-Mixed + %Swamp Forest 1k					**+1.9**
(Intercept)	0.14	0.21	0.67	0.505	
%Residential	-0.64	0.41	-1.55	0.121	
%Swamp Forest 1km	2.59	1.26	2.06	0.039*	
%Swamp Forest 1km + %Impervious 1km + Late season NDVI 1km					**+1.9**
(Intercept)	-0.19	0.72	-0.26	0.795	

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
Late NDVI 1km	1.16	1.93	0.60	0.548	
%Impervious 1km	-1.13	0.85	-1.33	0.185	
%Swamp forest 1km	1.76	1.37	1.29	0.199	
Late season NDVI + %Residential-Mixed1km					**+2.0**
(Intercept)	-0.01	0.59	-0.02	0.984	
Late NDVI	1.78	1.38	1.29	0.197	
%Residential 1km	-1.24	0.46	-2.68	0.007**	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table S1.7. Generalized Linear Mixed Models for combinations of only variables at 250m scale, for urbanization class (rural, suburban, urban) in Northwest Ohio study area. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC.

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Impervious + %Swamp forest					**+0**
(Intercept)	0.43	0.11	3.93	<0.001***	
%Swamp forest	-1.01	0.52	-1.95	0.051 .	
%Impervious	1.72	0.44	3.86	<0.001***	
%Impervious + %Residential-Mixed					**+1.3**
(Intercept)	0.20	0.11	1.94	0.053 .	
%Residential	0.43	0.26	1.69	0.091 .	
%Impervious	1.65	0.50	3.34	<0.001***	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$

Table S1.8. Generalized Linear Mixed Models for combinations of only variables at 1km scale, for urbanization class in Northwest Ohio study area. All variables not listed did not produce significant models. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC.

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Impervious + %Residential					**+0**
(Intercept)	-0.10	0.15	-0.70	0.485	
%Impervious 1km	1.34	0.55	2.45	0.014*	
%Residential1km	0.90	0.42	2.16	0.031*	
Late season NDVI + %Residential					**+2.8**
(Intercept)	0.38	0.40	0.95	0.342	
Late NDVI 1km	-1.39	0.97	-1.42	0.155	
%Residential 1km	1.44	0.31	4.62	<0.001***	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
NDVI = normalized difference vegetation index

Table S1.9. Generalized Linear Mixed Models for combinations of variables at both 250m and 1km scale, for urbanization class in Northwest Ohio study area. Models only included with a ΔAIC ≤2. Models listed in order of ΔAIC.

Variable	Estimate	Std. Error	*Z*	*P*	ΔAIC
%Impervious 1km + %Residential-Mixed 1km + Percent Flood forest 250m					**+0**
(Intercept)	-0.16	0.16	-1.02	0.308	
%Impervious 1km	1.38	0.55	2.51	0.012*	
%Floodplain forest	0.50	0.45	1.11	0.267	
%Residential 1km	0.93	0.42	2.21	0.027*	
%Impervious 1km + %Wet prairie 1km + %Residential-Mixed 1km + Percent Floodplain forest 250m					**+1.1**
(Intercept)	-0.27	0.20	-1.35	0.178	
%Impervious 1km	1.43	0.56	2.57	0.010*	
%Floodplain forest	0.59	0.46	1.27	0.206	
%Residential 1km	1.04	0.44	2.36	0.018*	
%Wet prairie 1km	1.78	1.92	0.93	0.354	

* Significant at $p < 0.05$
** Significant at $p < 0.01$
*** Significant at $p < 0.001$
. Near significance at $0.05 < p < 0.10$
NDVI = normalized difference vegetation index

Table S4.1**:** 16-point checklist to ensure survey responses were not from automated robotic programs. Each survey could fail a total of 4 checks before being disqualified as a robotic response.

#	**Check:**
1	Improbable number of clicks on informational video page
2	Allowing more than 25 minutes to pass before clicking on the video page
3	Nonsensical answers to either open-ended question
4	Suspicious email address
5	Suspicious similarities in 5-digit self-assigned codes in surveys conducted on the same day
6	Impossibly fast completion time
7	Suspicious similarities in responses to multiple demographic questions in surveys conducted on the same day
8, 9	Failure to confirm they lived in one of the five study counties both before and after the survey (2)
10	Inconsistent answers on county of residence between before and after survey
11, 12	Failure to provide requested answer during attention check question (2)
13	Surveys suspiciously started at exactly the same time
14	Surveys suspiciously ended at exactly the same time
15	Starting and completing the survey between 1am and 5am
16	Suspicious writing in open-ended question for “current profession” demographic question

Table S4.2: Full copy of the survey questionnaire, including demographic questions.

General Conservation Survey Questionnaire **Participant Number: _____**

Please place a check or X in the box that most closely aligns with how you feel about the numbered statement to the left. Please write the participant number you created in your initial survey in the top right corner.

Demographic question 1: What is your current county of residence?

		Strongly Disagree	Disagree	Neutral	Agree	Strongly Agree
1	I would rather spend time outside than inside					
2	I support creating more nature preserves					
3	I would make changes to my property to help protect wildlife					
4	I am an environmentalist/wildlife lover					
5	Nature preserves are not worth the money nor space					
6	Humans should be free to build wherever we want					
7	I prefer to buy from companies that are environmentally friendly					
8	We need more parks for human activities, not wildlife					
9	I would reduce my pesticide use to help protect wildlife					
10	We should improve our parks to make them better for wildlife					
11	I do not care if our parks protect wildlife					
12	It is important to protect as many animals and plants as we can					
13	I donate to wildlife organizations					
14	I support a small fee if it means helping local parks and nature preserves					
15	I would volunteer to help improve parks for wildlife					
16	I would not change my property for wildlife without a tax break					
17	We need more awareness about environmental issues					
18	I support new parks, designed with wildlife in mind					
19	I hear too much about the environment/wildlife					
20	I talk to my friends and family about the environment					
21	Open space for human activity is important in a park					

22	If people are not allowed in them, I do not support creating nature preserves					
23	Preserves and parks provide benefits to humans					
24	I would volunteer to help in an awareness campaign for environmental issues					
25	I support a significant fee if it means helping local parks and nature preserves					
26	I do not feel comfortable in the outdoors					
27	Seeing wildlife is important to me when I visit a park					
28	Having access to fishing/boating in parks is important to me					
29	For the sake of wildlife, humans should stop developing land					
30	I vote for officials who support wildlife and protect our land/water					

Open Ended 1: In one sentence, what you would use to describe a park that you believe would be good for both humans and wildlife?

Open Ended 2: In one sentence, what is your primary reason for supporting or not supporting wildlife conservation in the area?

Demographic Information:

1. What is your current age?

    18-29 30-44 45-64 65+

2. How do you identify your gender?
3. Which is closest to your educational background?

    A. High School Diploma or Less

    B. Some College

    C. Undergraduate Degree

    D. Graduate Degree

4. How would you describe your current profession?

5. When did you last visit a Metropark or Zoo?

    A. Within the Last Year

    B. Within the Last 3 Years

    C. Within the Last 5 years

    D. Not within the last 5 years

6. Were you raised in a big city, small city/town, suburbs, or rural area?

    Big City Small City Suburbs Rural Area

7. What county do you currently live in?

APPENDIX D: INSTITUTIONAL ANIMAL CARE AND USE COMMITTEE APPROVAL

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Office of Research Compliance

DATE:	February 18, 2023
TO:	Karen Root, PhD
FROM:	Bowling Green State University Institutional Animal Care and Use Committee
PROJECT TITLE:	[1861200-2] Assessing the Effects of Urbanization on Wetland Ecosystem Function in An Ohio Hotspot
IACUC REFERENCE #:	
SUBMISSION TYPE:	Amendment/Modification
ACTION:	APPROVED
APPROVAL DATE:	February 18, 2023
EXPIRATION DATE:	March 2, 2025
REVIEW TYPE:	Designated Member Review

Thank you for your submission of Amendment/Modification materials for the above referenced research project. The Bowling Green State University Institutional Animal Care and Use Committee has APPROVED your submission. All research must be conducted in accordance with this approved submission. Please make sure that all members of your research team read the approved version of the protocol.

The following modifications have been approved:

- Amendment/Modification - Addendum_05-18 KronRoot 2022.doc (UPDATED: 01/20/2023)
- Other - Orosz Communication 2023.png (UPDATED: 01/20/2023)

If you have any questions, please contact the IACUC Administrator at 419-372-8753 or iacuc@bgsu.edu. Please include your project title and reference number in all correspondence with this committee.

This letter has been electronically signed in accordance with all applicable regulations, and a copy is retained within Bowling Green State University Institutional Animal Care and Use Committee's records.

APPENDIX E: INSTITUTIONAL REVIEW BOARD APPROVAL

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Office of Research Compliance

Institutional Review Board

DATE:	April 28, 2022
TO:	Brian Kron
FROM:	Bowling Green State University Institutional Review Board
PROJECT TITLE:	[1892279-2] Understanding the Views of Local Residents on Wildlife Conservation and the Environment
SUBMISSION TYPE:	Revision
ACTION:	APPROVED
APPROVAL DATE:	April 28, 2022
EXPIRATION DATE:	April 18, 2023
REVIEW TYPE:	Expedited Review
REVIEW CATEGORY:	Expedited review category # 7

Thank you for your submission of Revision materials for this project. The Bowling Green State University Institutional Review Board has APPROVED your submission. This approval is based on an appropriate risk/benefit ratio and a project design wherein the risks have been minimized. All research must be conducted in accordance with this approved submission.

The final approved version of the consent document(s) is available as a published Board Document in the Review Details page. You must use the approved version of the consent document when obtaining consent from participants. Informed consent must continue throughout the project via a dialogue between the researcher and research participant. Federal regulations require that each participant receives a copy of the consent document.

Please note that you are responsible to conduct the study as approved by the IRB. If you seek to make any changes in your project activities or procedures, those modifications must be approved by this committee prior to initiation. Please use the modification request form for this procedure.

All UNANTICIPATED PROBLEMS involving risks to subjects or others and SERIOUS and UNEXPECTED adverse events must be reported promptly to this office. All NON-COMPLIANCE issues or COMPLAINTS regarding this project must also be reported promptly to this office.

This approval expires on April 18, 2023. You will receive a continuing review notice before your project expires. If you wish to continue your work after the expiration date, your documentation for continuing review must be received with sufficient time for review and continued approval before the expiration date.

If you have any questions, please contact the Institutional Review Board at 419-372-7716 or irb@bgsu.edu. Please include your project title and reference number in all correspondence with this committee.

BOWLING GREEN STATE UNIVERSITY

Office of Research Compliance

Institutional Review Board

DATE:	May 12, 2022
TO:	Brian Kron
FROM:	Bowling Green State University Institutional Review Board
PROJECT TITLE:	[1892279-3] Understanding the Views of Local Residents on Wildlife Conservation and the Environment
SUBMISSION TYPE:	Amendment/Modification
ACTION:	APPROVED
APPROVAL DATE:	May 11, 2022
EXPIRATION DATE:	April 18, 2023
REVIEW TYPE:	Expedited Review
REVIEW CATEGORY:	Expedited review category # 7

Thank you for your submission of Amendment/Modification materials for this project. The Bowling Green State University Institutional Review Board has APPROVED your submission. This approval is based on an appropriate risk/benefit ratio and a project design wherein the risks have been minimized. All research must be conducted in accordance with this approved submission.

1. Adjust consent forms to reflect that studies deemed to be ineligible or fraudulent will be disqualified from receiving gift card incentive. Also add in distinct languange about the number of respondants who will receive surveys.

2. Add captcha question to Qualtrics online survey.

3. Add more strict attention checks to survey questionnaire and demographic questions to help parse out fraudulent responses. Found in questions 10, 33, 34, and demographic question #7

The final approved version of the consent document(s) is available as a published Board Document in the Review Details page. You must use the approved version of the consent document when obtaining consent from participants. Informed consent must continue throughout the project via a dialogue between the researcher and research participant. Federal regulations require that each participant receives a copy of the consent document.

Please note that you are responsible to conduct the study as approved by the IRB. If you seek to make any changes in your project activities or procedures, those modifications must be approved by this committee prior to initiation. Please use the modification request form for this procedure.

All UNANTICIPATED PROBLEMS involving risks to subjects or others and SERIOUS and UNEXPECTED adverse events must be reported promptly to this office. All NON-COMPLIANCE issues or COMPLAINTS regarding this project must also be reported promptly to this office.

This approval expires on April 18, 2023. You will receive a continuing review notice before your project expires. If you wish to continue your work after the expiration date, your documentation for continuing review must be received with sufficient time for review and continued approval before the expiration date.

If you have any questions, please contact the Institutional Review Board at 419-372-7716 or irb@bgsu.edu. Please include your project title and reference number in all correspondence with this committee.

This letter has been electronically signed in accordance with all applicable regulations, and a copy is retained within Bowling Green State University Institutional Review Board's records.

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Office of Research Compliance

Institutional Review Board

DATE:	May 31, 2023
TO:	Brian Kron
FROM:	Bowling Green State University Institutional Review Board
PROJECT TITLE:	[1892279-5] Understanding the Views of Local Residents on Wildlife Conservation and the Environment
SUBMISSION TYPE:	Revision
ACTION:	APPROVED
APPROVAL DATE:	May 27, 2023
EXPIRATION DATE:	May 26, 2024
REVIEW TYPE:	Expedited Review
REVIEW CATEGORY:	Expedited review category # 7

Thank you for your submission of Revision materials for this project. The Bowling Green State University Institutional Review Board has APPROVED your submission. This approval is based on an appropriate risk/benefit ratio and a project design wherein the risks have been minimized. All research must be conducted in accordance with this approved submission.

The final approved version of the consent document(s) is available as a published Board Document in the Review Details page. You must use the approved version of the consent document when obtaining consent from participants. Informed consent must continue throughout the project via a dialogue between the researcher and research participant. Federal regulations require that each participant receives a copy of the consent document.

Please note that you are responsible to conduct the study as approved by the IRB. If you seek to make any changes in your project activities or procedures, those modifications must be approved by this committee prior to initiation. Please use the modification request form for this procedure.

All UNANTICIPATED PROBLEMS involving risks to subjects or others and SERIOUS and UNEXPECTED adverse events must be reported promptly to this office. All NON-COMPLIANCE issues or COMPLAINTS regarding this project must also be reported promptly to this office.

This approval expires on May 26, 2024. You will receive a continuing review notice before your project expires. If you wish to continue your work after the expiration date, your documentation for continuing review must be received with sufficient time for review and continued approval before the expiration date.

If you have any questions, please contact the Institutional Review Board at 419-372-7716 or irb@bgsu.edu. Please include your project title and reference number in all correspondence with this committee.