Do Open Spaces Provide Refuge for Anurans Within an Urban Matrix?

David Ruple Hutto

Clemson University, drhutto88@gmail.com

Follow this and additional works at: https://tigerprints.clemson.edu/all_theses

Recommended Citation
https://tigerprints.clemson.edu/all_theses/2989

This Thesis is brought to you for free and open access by the Theses at TigerPrints. It has been accepted for inclusion in All Theses by an authorized administrator of TigerPrints. For more information, please contact kokeefe@clemson.edu.
DO OPEN SPACES PROVIDE REFUGE FOR ANURANS WITHIN AN URBAN MATRIX?

A Thesis
Presented to
the Graduate School of
Clemson University

In Partial Fulfillment
of the Requirements for the Degree
Master of Science
Wildlife and Fisheries Biology

by
David Ruple Hutto, Jr.
December 2018

Accepted by:
Dr. Kyle Barrett, Committee Chair
Dr. Catherine Jachowski
Dr. Rob Baldwin
ABSTRACT

Urbanization is among the largest threats to amphibian populations through habitat fragmentation, isolation, and outright destruction. Urban open spaces, such as parks and golf courses, have the potential to provide amphibians with suitable habitat within an urbanized matrix. During the spring and summer of 2018, I conducted dip net surveys and active call surveys to determine the presence and abundance of anurans at 51 wetland sites within the Piedmont ecoregion of South Carolina. Nearly one-third of these wetlands were located within urban open spaces, while the others were situated along a rural – urban gradient. Impervious surface and road density surrounding the wetlands were measured at a core habitat scale (300 m) and average maximum migration scale (750 m). Urban Open Space wetlands were found to have levels of impervious surface similar to High Urbanization wetlands at the larger scale and were intermediate between Low and High Urbanization wetlands at the smaller scale. Road densities were higher at Urban Open Space wetlands at both scales compared to Low and High Urbanization sites. Species richness decreased as impervious surface and road density increased among all wetlands. Wetland category was not a significant driver explaining species richness, but β-diversity was higher among Urban Open Space wetlands than either Low or High Urbanization wetlands. I also evaluated species-specific relative abundances as a function of wetland type (Urban Open Space, High Urbanization, or Low Urbanization), several within-wetland variables and two landscape-scale covariates. No species were influenced by wetland type, however impervious surface had a negative influence on the abundance of one species whereas road density negatively affected three species of anurans. Urban
Open Space wetlands did not appear to increase suitability for anurans relative to urban wetlands, instead showing higher variability in species composition; perhaps attributable to the diversity among sites represented in the Urban Open Space category. Conservation efforts conducted within open spaces should attempt to focus on issues not only at an individual wetland scale, but also at larger scales surrounding the open spaces themselves. Understanding how urbanization at various spatial scales effects anuran species can bolster amphibian conservation efforts in urban matrices.
DEDICATION

I dedicate this thesis to my parents, David and Ann Hutto, my sisters Suzann Weathers and Ashley Wolfe, my nieces Harley and Josie Weathers, and my nephew James Wolfe. The groundwork for this thesis would not have been realized without the teachings of my dad, who gave me some of my first experiences with reptiles and amphibians and taught me to appreciate all that nature holds, and my mom who taught me to see the beauty in everything and cherish every moment. Their and my sisters’ constant support (and light teasing) kept me driven to accomplish the work that I set out to. To my nieces and nephew, I hope that my passion for the natural world continues to encourage them and that they will grow up with a drive to make the world a better place.
ACKNOWLEDGMENTS

The project was funded by Clemson University, South Carolina Wildlife Federation, the Riverbanks Conservation Support Fund, the D. L. Scurry Foundation, and the Margaret H. Lloyd-SmartState South Carolina Endowment for Urban Ecology and Restoration.

I would like to thank my advisor, Dr. Kyle Barrett, for all of he has done for me. His unwavering support, patience, and encouragement allowed me to realize my actual potential within the scientific community. He is an incredible advisor who takes a genuine care in his students, willing to lend advice and assistance no matter the issue. Thank you, Dr. Barrett, for everything. I would also like to thank my committee members, Dr. Rob Baldwin and Dr. Cathy Jachowski for contributing new insights and perspectives to my project. I am also grateful to my lab mates who have supported myself and one another throughout our time at Clemson. Special thanks to my field tech, Addie Carter, for all of her hard work and dedication to my project even in the most challenging environments in the field. To all of the volunteers who donated their time to my project, a huge thank you. I am also indebted to all of those who supported my project by providing access to their property – managers and superintendents of the South Carolina Botanical Gardens, The Walker Golf Course, Boscobel Golf Course, and private landowners. Last, but not least, thank you to my family for supporting my love of wildlife and nature and to Mollye MacNaughton for keeping me sane these last few years.
TABLE OF CONTENTS

<table>
<thead>
<tr>
<th>Section</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>TITLE PAGE</td>
<td>i</td>
</tr>
<tr>
<td>ABSTRACT</td>
<td>ii</td>
</tr>
<tr>
<td>DEDICATION</td>
<td>iv</td>
</tr>
<tr>
<td>ACKNOWLEDGMENTS</td>
<td>v</td>
</tr>
<tr>
<td>LIST OF TABLES</td>
<td>vi</td>
</tr>
<tr>
<td>LIST OF FIGURES</td>
<td>viii</td>
</tr>
<tr>
<td>CHAPTER</td>
<td></td>
</tr>
<tr>
<td>I. DO URBAN OPEN SPACES SUPPORT ANURAN DIVERSITY IN AN INCREASINGLY URBANIZED WORLD?</td>
<td>1</td>
</tr>
<tr>
<td>Abstract</td>
<td>1</td>
</tr>
<tr>
<td>Introduction</td>
<td>3</td>
</tr>
<tr>
<td>Methods and Materials</td>
<td>8</td>
</tr>
<tr>
<td>Results</td>
<td>16</td>
</tr>
<tr>
<td>Discussion</td>
<td>20</td>
</tr>
<tr>
<td>Conclusion</td>
<td>23</td>
</tr>
<tr>
<td>References</td>
<td>35</td>
</tr>
<tr>
<td>II. DO OPEN SPACES WITHIN AN URBAN MATRIX INCREASE ANURAN ABUNDANCE PROBABILITY?</td>
<td>41</td>
</tr>
<tr>
<td>Abstract</td>
<td>41</td>
</tr>
<tr>
<td>Introduction</td>
<td>42</td>
</tr>
<tr>
<td>Methods and Materials</td>
<td>45</td>
</tr>
<tr>
<td>Results</td>
<td>52</td>
</tr>
<tr>
<td>Discussion</td>
<td>54</td>
</tr>
<tr>
<td>Conclusion</td>
<td>57</td>
</tr>
<tr>
<td>References</td>
<td>63</td>
</tr>
</tbody>
</table>
# LIST OF TABLES

<table>
<thead>
<tr>
<th>Table</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.1</td>
<td>Proportion of site categories occupied by species</td>
</tr>
<tr>
<td>2.1</td>
<td>Site and landscape scale call index covariates used in N-Mixture models</td>
</tr>
<tr>
<td>2.2</td>
<td>Top candidate anuran abundance models</td>
</tr>
</tbody>
</table>
LIST OF FIGURES

<table>
<thead>
<tr>
<th>Figure</th>
<th>Description</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.1</td>
<td>Map of study area and wetland sites</td>
<td>28</td>
</tr>
<tr>
<td>1.2</td>
<td>Box and whisker plots of percent impervious surface within 750 m and 300 m of land surrounding wetlands in three categories</td>
<td>29</td>
</tr>
<tr>
<td>1.3</td>
<td>Box and whisker plots of road density in meters within 750 m and 300 m of land surrounding wetlands in three categories</td>
<td>30</td>
</tr>
<tr>
<td>1.4</td>
<td>Chao estimated species richness across three wetland types</td>
<td>31</td>
</tr>
<tr>
<td>1.5</td>
<td>Observed anuran species richness by percentage of impervious surface within 750 m and 300 m of land surrounding wetlands in three categories</td>
<td>32</td>
</tr>
<tr>
<td>1.6</td>
<td>Observed anuran species richness by road density in Meters within 750 m and 300 m of land Surrounding wetlands in three categories</td>
<td>33</td>
</tr>
<tr>
<td>1.7</td>
<td>Box and whisker plot of anuran β-diversity among wetlands in three categories</td>
<td>34</td>
</tr>
<tr>
<td>2.1</td>
<td>Expected call index for four anurans in response to road density/impervious surface within 300 m of land surrounding wetlands</td>
<td>62</td>
</tr>
</tbody>
</table>
CHAPTER ONE

Do urban open spaces support anuran diversity in an increasingly urbanized world?

ABSTRACT

Urbanization is among the largest threats to amphibian populations through habitat fragmentation, isolation, and outright destruction. Urban open spaces, such as parks and golf courses, have the potential to provide amphibians with suitable habitat within an urbanized matrix. During the spring and summer of 2018, I conducted dip net surveys and active call surveys to determine the presence and abundance of anurans at 51 wetland sites within the Piedmont ecoregion of South Carolina. Nearly one-third of these wetlands were located within urban open spaces, one-third in low development areas, and the last one-third in highly developed areas. Impervious surface and road density surrounding the wetlands were measured at a core habitat scale (300 m) and average maximum migration scale (750 m). Urban Open Space wetlands were found to have levels of impervious surface similar to High Urbanization wetlands at the larger scale and were intermediate between Low and High Urbanization wetlands at the smaller scale. Road densities were higher at Urban Open Space wetlands at both scales compared to Low and High Urbanization sites. Species richness decreased as impervious surface and road density increased among all wetlands. Wetland category was not a significant driver explaining species richness, but β-diversity was more variable among Urban Open Space wetlands than either Low or High Urbanization wetlands. Urban Open Space wetlands did not appear to increase suitability for anurans relative to urban wetlands, instead
showing higher variability in species composition; perhaps attributable to the diversity among sites represented in the Urban Open Space category.
INTRODUCTION

Undeveloped habitat is rapidly being replaced with urban infrastructure such as roads, buildings, houses, large paved areas, and other impervious surfaces (McDonnell & Pickett 1990; Hamer & McDonnell 2010). An increase in housing and building density has a negative effect on native species (Hansen et al. 2005) through habitat loss (Rubbo & Kiesecker 2005; Hamer & McDonnell 2010) and reduction in habitat quality (Pope et al. 2000; McKinney 2002). Decreases in native bird (Germaine et al. 1998), arthropod (Denys & Schmidt 1998; Miyashita et al. 1998), rodent (Bock et al. 2002), and amphibian (Lehtinen et al. 1999) species richness have been noted in correlation with urban density increases. The long-term consequences of urbanization can be difficult to accurately predict, as negative responses of species to development may intensify over decades after initial development (Donnelly 2002; Hansen et al. 2005).

As the size and density of urban development continues to increase, it is important to find areas that can serve as conservation refuges for species that otherwise would be negatively affected by urbanization. Urban open spaces (also referred to as green spaces) may offer areas of biodiversity conservation for native species (Goddard et al. 2009; Nielsen et al. 2014) within a matrix of unsuitable habitat. Urban open spaces are defined within this study as publicly accessible, managed outdoor spaces that are partly or completely covered by significant amounts of vegetation that exist primarily as semi-natural areas within an urban environment (Jim and Chen 2003; Kong et al. 2009; United States Environmental Protection Agency 2017). These areas may be public parks, community gardens, sports recreation zones (e.g. golf courses), or cemeteries. The habitat
fragmentation, destruction, and isolation that occurs due to urbanization are all threats to biodiversity that urban open spaces can help mitigate (Kong et al. 2009). Not only can urban open spaces help to preserve the biodiversity of an area, their presence and the presence of the wide range of plant and animal species they can contain also have positive psychological benefits to the people who utilize them (Fuller et al. 2007; Ambrey & Fleming 2014). This presents an even stronger argument for successfully managing these areas and striving to increase and/or maintain their biological complexity.

One group of animals that may particularly benefit from proper management of urban open spaces is amphibians. Their relatively small body sizes and small home ranges make them ideal candidates for studies focusing on localized effects of urbanization. While many studies have sought to evaluate amphibian responses to urbanization, few have evaluated how this group responds to small-scale habitat protection within a developed matrix (Hamer and McDonnell 2008; Scheffers and Paszkowski 2012). The question still remains as to the relative influence of local environmental variables versus landscape scale elements as driving factors structuring amphibian communities (Villaseñor et al. 2017). Amphibian species are at a higher risk of extinction than any other taxon (Wake & Vredenburg, 2008; Hamer & McDonnell 2008) with nearly one-third (32%) of the world’s amphibians listed as threatened and 43% with declining populations as of 2004 (Baillie et al. 2004; Villaseñor et al. 2017). These declines are noted worldwide, with North America being no exception (Stuart et al. 2004; Miller et al. 2018). Along with urbanization, other drivers of amphibian decline include habitat destruction via deforestation, predator and exotic species introduction,
increased levels of pollution, logging and mining practices, human consumption, climate change, and disease (Fahrig et al. 1995; Hamer & McDonnell 2010; Hossack et al. 2012). Anurans in particular are largely wetland-associated amphibian species requiring adequate space within and around aquatic and terrestrial habitats to carry out essential life-history processes such as foraging, sheltering, and reproduction. These terrestrial and wetland habitats typically need to be physically linked in order to allow for dispersal and migration between populations (Hamer & Parris 2010). The modification and outright destruction of wetland habitats has been one of the most widespread forms of habitat degradation throughout the last several centuries (Urban & Roehm 2018). While wetland destruction was historically largely caused by agriculture, urbanization increasingly eclipses agriculture in its threats to wetland health (Gutzwiller & Flather 2011; Urban & Roehm 2018).

Urbanization and associated road densities can influence the movement of amphibians between suitable habitats and can increase the exposure of a wetland to contaminants such as pesticides, herbicides, and other chemicals (Rubbo & Kiesecker 2005; Hamer & McDonnell 2008; Sievers et al. 2018). Additionally, urban development can increase eutrophication through lawn fertilizer and agricultural chemical runoff (Knutson et al. 1999; Rubbo & Kiesecker 2005), and alter wetland hydrology (Smallbone et al. 2011; Sievers et al. 2018). Impervious surfaces near wetlands also increase habitat loss and fragmentation leading to a separation of breeding wetlands from important natural habitats (Smallbone et al. 2011, Rytwinski & Fahrig 2015). Wetlands that are isolated by urban infrastructure may not be easily accessible and may be altered (or
removed altogether) from their natural state, therefore decreasing the likelihood that a certain species will occur in an urban area (Semlitsch 2000; Hamer & Parris 2011). Not only do roads act as barriers between otherwise contiguous habitats, traffic along these roads has a direct negative effect on anuran population (Fahrig et al. 1995; Marsh et al. 2017). Frogs and toads may struggle to navigate even short distances (in relation to their overall dispersal abilities) in urbanized areas (Hamer & Parris 2011). Factors such as vehicle collisions, exposure to runoff (salt, oil, etc.), noise, exhaust emissions, and vibrations can all affect anuran populations through either direct mortality or by interrupting the behavior of frogs and toads (Buchanan 1993; Marsh et al. 2017).

If amphibian declines are to be curtailed, existing habitat must be protected and areas that humans utilize should be managed in a way that minimizes negative impacts toward wildlife (Puglis & Boone 2012). Urban open spaces can provide important habitat and habitat connectivity for anuran species, particularly those spaces containing wetlands or ponds (Hamer & Parris 2011; Puglis & Boone 2012). Not only could anurans benefit from these open spaces, the open spaces could benefit from them. Tadpoles feed on algae and juvenile and adult anurans feed on insects which could reduce the need to utilize pesticides and herbicides, or to stock fish as an insect control method (Puglis & Boone 2012; Knutson et al. 1999). Jackson et al. (2011) suggested guidelines for incorporating amphibian habitat into golf courses in Canada. Their approach was one that took playability into account and the needs of breeding anurans to develop a series of design guidelines that would aid landscapers and architects in providing quality habitat for
amphibians. Studies such as these suggest that the goals of wetland wildlife conservation and human quality of life are not necessarily at odds.

While it is well-established that urban development can decrease the diversity and abundance of anurans, the efficacy of small-scale buffers around development to bolster amphibian populations and diversity remains an open question (Calhoun et al. 2005). Reviews of the literature indicate wetland breeding amphibians require anywhere from 300 – 750 m of upland habitat around breeding sites to carry out necessary life-history processes (Semlitsch 2000; Semlitsch & Bodie 2003; Parris 2006; Birx-Raybuck et al. 2009). I identified 51 wetlands along a rural-urban gradient in of the Piedmont of South Carolina that vary in the amount of developed land surrounding the wetland. A subset of these sites lie within open spaces, and the amount of development around these wetlands is relatively high at large buffer sizes (750m); however, these spaces have lower surrounding development at smaller scales (300m). I assessed anuran assemblage of wetlands within three different landscape categories (Low Urbanization, High Urbanization, and Urban Open Space) that were defined by the amount of development with a 750 m buffer. My objective was to evaluate anuran species richness and community composition as a function of within-wetland habitat features and the amount of urbanization within the surrounding landscape. I predicted that species richness would be highest at the “Low Urbanization” sites and that anuran assemblage at “Urban Open Space” sites would more closely resemble that of “Low Urbanization” sites given that urban open spaces may provide a buffer zone from urban influence.
METHODS AND MATERIALS

Study Area and Landscape Characteristics

I considered 73 potential wetland sites for inclusion in the study using Google Earth Pro and a .kmz file of wetlands derived from the National Wetlands Inventory (NWI; https://www.fws.gov/wetlands/Data/State-Downloads.html). This file included information regarding specific wetlands within Anderson, Oconee, and Pickens Counties, SC and I was able to obtain landowner information for several wetlands in the file using county tax assessor records. Throughout the late spring and summer of 2017, I identified 51 wetland sites that I determined were suitable for inclusion in the study (Figure 1.1). I chose these wetlands based on their accessibility and differences in the amount of urbanized space surrounding each wetland. These wetlands are situated within a gradient ranging from rural areas to areas of high urbanization. I specifically targeted wetlands that were located in managed, publicly accessible urban open spaces (golf courses, parks, and gardens) in order to address the study objectives. I used a handheld Garmin GPS unit (GPSmap 62s, Garmin, Ltd.) to acquire coordinates for each site that were loaded into ArcMap 10.4.1. Each wetland was then delineated as a unique polygon in ArcGIS (ESRI, Redlands, CA) using a satellite imagery base map combined with knowledge of the actual wetland boundaries determined by ground truthing. Two buffers were added around each wetland, one at 300 m to represent the average core habitat around the wetland (Semlitsch & Bodie 2003) and one at 750 m to encompass maximum average migration range for the anuran species within the study area. Maximum migration estimates for 95% of individuals within anuran populations average 703 m (Rittenhouse & Semlitsch...
Developed land, impervious surface, and road map layers data were all gathered from the South Carolina Department of Transportation (SCDOT), and I calculated the amount of each within both buffer radii. Developed land and impervious surface were calculated as percent coverage within the buffer, while road density was assessed as total length within the buffer.

I labelled wetlands with impervious surface $\geq 30\%$ within the 750m buffer “High Urbanization”, whereas wetlands with $< 30\%$ impervious surface received the label “Low Urbanization.” Impervious surfaces and other forms of development impacting 25 – 30% of upland habitat surrounding wetlands has been shown to lead to declines in populations of breeding amphibians (Calhoun et al. 2005). I gave the wetlands located in urban open spaces the label “Urban Open Space”. This resulted in a breakdown of 16 Low Urbanization, 16 High Urbanization, and 19 Urban Open Space wetlands. Percentage of impervious surface surrounding wetlands in urban open spaces was not used to label these wetlands, and my Urban Open Space category included sites that would have been placed in either of the other two categories. The distinct management strategies associated with wetlands and adjacent terrestrial habitat in these areas create the impetus for assigning them to their own category. I examined the Pearson correlation coefficient for each pair of landscape variables, and removed percent developed land from all future analyses as it was highly correlated with impervious surface at both the 750m and 300m scale (0.92 and 0.85, respectively).

To determine the distance from each wetland to the nearest body of water, I obtained a secondary layer of wetlands from the NWI that included all wetlands in South
Carolina. I divided polygons within the shapefile into two separate categories (riverine bodies and freshwater wetlands [freshwater emergent, forested/shrub, and ponds]). I calculated the straight-line distances from the edge of each wetland to each of the wetlands in these two categories. The distance between study sites was also measured to account for the possibility that study sites may be each other’s closest neighbor.

**Wetland Site Characteristics**

At each site, I recorded habitat data during each daytime dip net survey performed from March – July 2018 (additional information appears below under “Anuran Dip net Surveys”). I obtained wetland size through wetland delineation in ArcGIS. I measured wetland depth (m) at the deepest point using a meter stick, wetlands deeper than the 1.2-m limit of the depth stick were given a depth of > 1.2 m. I obtained average organic layer depth by measuring the depth (cm) of the submerged organic layer at each dip net stop and then averaging them together for each wetland. After leaf out (July), I measured canopy cover at each wetland by taking photos at each cardinal location using an iPhone 7 (Apple, Cupertino, CA) with a fisheye lens attachment (Amir, Shenzhen, Guangdong, China). For wetlands small enough (<0.01 ha) where multiple photos were not necessary to obtain canopy cover, I only took one photo. Conversely, at wetlands too large to obtain an accurate canopy reading with just four photos, I took a photo at every other dip net site. I then used the Gap Light Analyzer (Cary Institute of Ecosystem Studies, Millbrook, NY) to attain a percentage of canopy cover for each photo. I averaged the canopy cover across all photos for a wetland to produce its canopy cover reading. I characterized aquatic vegetation cover by making visual estimates of the amount of emergent and
submerged aquatic vegetation and placing these estimates into categories (1 = 0-25%, 2 = 26-50%, 3 = 51-75%, 4 = 76-100%), and a wetland was assigned the average value calculated across all dip net sample locations. I noted the presence or absence of a zone of herbaceous terrestrial vegetation within a 1-m buffer around wetlands, and assigned such buffer as present if it was at least 1 m in width and present around at least half of the wetland edge, and absent otherwise. Parris (2006) and Puglis and Boone (2012) suggest that terrestrial vegetation buffer zones around golf course wetlands may provide a more suitable habitat structure for anurans, providing shelter for metamorphs and adults as well as acting as sites for calling and amplexus. These buffer zones may also help mitigate the effects of applied chemicals (Puglis & Boone 2012). Fish presence was visually noted and further evaluated via dip net surveys, wetlands where fish were not observed were assumed to not contain fish. I noted the presence or absence of water during all dip net and call surveys to determine hydroperiod of each wetland during the sampling period (February-July). Wetlands that held water throughout the course of the study were given a value = 1 (permanent), and those that were dry at any point during the study were given a value = 0 (temporary). I measured pH, conductivity, and water temperature using an Oakton PCTSTestr™ (Cole Parmer, Vernon Hills, IL) during each dip net visit. I gathered air temperature, relative humidity, and max wind speed using a Kestrel 2500 unit (Nielsen-Kellerman, Boothwyn, PA) and visually assessed sky conditions (0 = clear, 1 = partly cloudy, 2 = cloudy/overcast, 3 = fog, 4 = drizzle, 5 = showers) during each dip net and call survey.
Anuran Dip Net Surveys

Each wetland was surveyed three times from March – July, with the exception of five wetlands that only received two dip net surveys due to weather and time constraints. Dip net surveys lasted for a minimum of 20 minutes at each site (Kruger et al. 2015). As there was a large variation in wetland size (size range = 0.002 - 8.71 ha), I increased survey time by 10 minutes as wetlands doubled in size, up to a survey time of one hour. Therefore, small wetlands (size range = 0.002 – 0.12 ha), medium wetlands (size range = 0.13 - 0.24 ha), large wetlands (size range = 0.26 – 0.60 ha), and extra-large wetlands (size range = 0.64 – 8.71 ha) were surveyed for 20, 30, 40, and 60 minutes, respectively. As traversing many of those extra-large wetlands was not possible in a 50-60 minute time frame, I employed a sub-sampling technique. Specifically, I dip-netted within the wetland for 60 minutes, at 20 random points along the perimeter (selected within ArcGIS) for 3 minutes each. During this time I used a long handled, D-frame dip net to survey the shallow edge of wetlands and any vegetation present within these edges for larval anurans. Tadpoles and other anuran life stages were identified to species as closely as possible in the field and a small subset of those tadpoles that were unidentifiable were returned to the lab for further identification (IACUC AUP2018-007). I also recorded any visual encounters of adult/juvenile/and metamorphs, and egg masses (as they were identifiable) in order to assess species richness and diversity.

Anuran Call Surveys

I conducted call surveys once per month February-June for a total of five call surveys per wetland. These took place during the evenings beginning approximately 30
minutes after sundown and ending no later than 0100 the following morning. Surveys were conducted when temperatures were between 5.6°C and 30°C (NAAMP; Wier & Mossman 2005; Steelman & Dorcas 2010) in order to maximize detection probability. In keeping with the protocol set forth by the North American Amphibian Monitoring Program (NAAMP), I conducted surveys when wind speed was less than or equal to a level 3 (8-12 mph) and did not conduct them during times of heavy rainfall as this may have affected hearing ability. I spent five minutes actively listening at each site and recorded calls as an index of abundance as per NAAMP protocol (i.e. 1 = individuals can be counted; space between calls, 2 = calls of individuals can be distinguished but there is some overlap of calls, 3 = full chorus, calls are constant, continuous and overlapping). I used the Massachusetts noise index in order to account for ambient noise surrounding each wetland where (0 = no effect on sampling, 1 = slight effect on sampling, 2 = moderate effect on sampling, 3 = serious effect on sampling, 4 = profound effect on sampling).

**Treefrog Retreats**

In order to account for adult treefrogs present at wetlands during dip net surveys, I deployed white polyvinyl chloride (PVC) pipe retreats around wetlands. Each wetland received at least two retreats and I scaled up the number of retreats similar to the way I scaled up the time spent dip net surveying at a wetland. Wetlands surveyed for 20 minutes received two, 30 minute wetlands received three, 40 minute wetlands received four, and the large wetlands actively surveyed for 60 minutes received five retreats for a total of 133 PVC retreats. Retreats were constructed similar to Boughton et al. (2000).
retreats measured 61-cm long and were 3.81-cm inside diameter. Each was fitted with a cap on the bottom to allow water to remain inside the retreat. A hole was drilled 25.4 cm from the bottom of the retreat to allow for excess water from rain events to drain out. I hung the retreats approximately 2m above the ground in hardwood trees near the wetland edge using a small carabiner clip attached to a length of paracord that was tied around the trunk or limb of the tree. At wetlands where there were no suitable trees in which to hang retreats, they were secured to metal garden stakes at the ground level near the wetland edge. I deployed PVC retreats after the first round of dip net surveys had been conducted (March) so retreats were only checked twice at each wetland during subsequent dip net surveys. Each time retreats were checked, I identified any treefrog(s) within, cleaned out any debris, and replaced or added water as necessary.

Data Analysis

To evaluate the separation in landscape structure among the assigned land use categories, I used two separate one-way Analysis of Variance (ANOVA) tests followed by Tukey HSD pairwise comparison to assess differences in the amount of impervious surface and road density surrounding wetlands by type (Low, High, and Urban Open Space) at both 750 m and 300 m. I conducted a Multivariate Analysis of Variance (MANOVA) followed by Tukey HSD pairwise comparison to evaluate differences in habitat variables and wetland category. Habitat variables included pH, conductivity, wetland depth, organic layer depth, canopy cover, area, and distances to the nearest river, and freshwater wetland. I used a contingency table analysis followed by Pearson’s chi-
squared test to account for differences in categorical environmental variables among wetland categories.

I estimated anuran species richness using the Chao index for incidence data. The Chao index estimates richness based on the number of rare species detected (i.e., species that were only detected once or twice during sampling). To test the hypothesis that anuran species richness is influenced by the amount of development, regardless of the assigned land use category, I performed linear regression on observed species richness against the percentage of impervious surface and road density at both the 300 m and 750 m scales. I estimated $\beta$-diversity between wetlands using the Bray-Curtis distance metric on all anuran species detected. This metric ranges from 0 (identical species composition) to 1 (no shared species between sites). $\beta$-diversity estimates were subjected to an ANOVA followed by Tukey HSD pairwise comparison to assess differences among wetland categories.

I used Redundancy Analysis (RDA) to test for changes in community composition across wetland categories. RDA is a direct gradient analysis which summarizes linear relationships between components of response variables that are explained by a set of explanatory variables. Species data were summed across all dip net visits and standardized on a scale of 0 – 10 with the maximum count of each species across all sites given a value of 10. All other abundance values for each species were then standardized as a function of the highest abundance recorded for that species. The RDA was constrained by wetland type. I used Indicator Species Analysis to determine species associated with human land use as a function of wetland category. The indicator value
index defines sets of species that distinguish predefined habitats (Dufrene and Legendre 2009). All statistical analysis were performed in Program R Version 3.4.1 (R Core Team 2017).

RESULTS

Environmental and landscape variation among wetland categories

Landscape measures revealed the greatest distinction among wetland categories. There were significant differences among values of impervious surface surrounding wetlands within each category at both the 750 m ($F_{2,48} = 27.67, P < 0.0008$; Fig. 1.2A) and 300 m scale ($F_{2,48} = 19.08, P < 0.0001$; Fig. 1.2B). At the 750 m scale percent impervious surface was significantly higher in High Urbanization sites relative to Low Urbanization wetlands (Mean = 57.06 and 15.75, respectively; $P < 0.00001$) and in Urban Open Space wetlands relative to Low Urbanization sites (Mean = 45.63 and 15.75, respectively; $P < 0.00001$). There was no significant difference in impervious surface between High Urbanization and Urban Open Space wetlands at the 750 m scale (Mean = 57.06 and 45.63, respectively; $P = 0.11$). At the 300 m scale the percentage of impervious surface was significantly higher at High Urbanization wetlands relative to both Urban Open Space (Mean = 57.56 and 34.79, respectively; $P = 0.002$) and Low Urbanization wetlands (Mean = 16.94; $P < 0.001$). Impervious surface was also significantly higher around Urban Open Space wetlands at the 300 m scale relative to Low Urbanization wetlands (Mean = 34.79 and 16.94, respectively; $P = 0.02$). Difference in road length surrounding all wetland types was also significantly different at both the 750 m scale
(\(F_{2,48} = 40.88, P < 0.00001;\) Fig. 1.3A) and the 300 m scale (\(F_{2,48} = 28.27, P < 0.00001;\) Fig. 1.3B). At the 750 m scale road densities were significantly higher at Urban Open Space wetlands relative to High Urbanization sites (Mean = 25,091 and 14,460, respectively; \(P < 0.001\)) and at Urban Open Space wetlands relative to Low Urbanization sites (Mean = 25,091 and 8,197, respectively; \(P < 0.001\)). Road densities at the 750 m scale were also significantly higher at High Urbanization wetlands than Low Urbanization sites (Mean = 14,460 and 8,197, respectively; \(P = 0.008\)). Road density at the 300 m scale was higher around Urban Open Space wetlands than both High (Mean = 5,336 and 2,429, respectively; \(P < 0.001\)) and Low Urbanization wetlands (Mean = 1,665; \(P < 0.001\)). There was no significant difference in road density between High and Low Urbanization wetlands at the 300 m scale (Mean = 2,429 and 1,665, respectively; \(P = 0.34\)).

Local measures of habitat were largely similar among wetlands, with the exception of canopy cover and terrestrial buffer immediately surrounding the wetland. Canopy cover differed among all three wetland categories (\(F_{2,48} = 7.09, P = 0.002\)). Low Urbanization wetlands had the highest mean canopy cover (51.1\%) significantly higher than High Urbanization wetlands and Urban Open Space wetlands (\(F_{1,30} = 12.97, P = 0.001\) and \(F_{1,33} = 9.83, P = 0.003\), respectively). Urban Open Space wetlands had a mean canopy cover (26\%) just greater than High Urbanization wetlands (24.3\%), and the two showed no significant difference in percentage of canopy cover (\(F_{1,33} = 0.04, P = 0.83\)). Contingency table analysis and Pearson’s chi-squared test revealed the probability of a buffer zone of terrestrial vegetation around a wetland is not independent of wetland class.
Specifically Urban Open Space wetlands were more likely to have a terrestrial buffer zone ($x^2 = 7.44, P = 0.02$). A 2x2 Chi-squared test on just the Low and High Urbanization wetland categories failed to reject the null hypothesis of independence between wetland category and the presence or absence of a terrestrial vegetation buffer zone ($x^2 = 0.94, P = 0.33$).

**Anuran Richness and Diversity**

During the 5 month sampling period at 51 wetland sites, 12 species of anuran were detected. Of the 12 species detected, 3 are listed under the South Carolina Department of Natural Resources Wildlife Action Plan as species of priority [Pickerel Frogs (*Lithobates palustris*), Northern Cricket Frogs (*Acris crepitans*), and Upland Chorus Frogs (*Pseudacris feriarum*)] (South Carolina Department of Natural Resources, 2015). Green Frogs (*Lithobates clamitans*) were the most common anuran, occurring at 84% of sites. An additional 5 species were also found at > 50% of sites [Fowler’s Toads (*Anaxyrus fowleri*) 57%, Gray Treefrogs (*Hyla versicolor*) 65%, American Bullfrogs (*Lithobates catesbeianus*) 71%, Southern Leopard Frogs (*Lithobates sphenoocephalus*) 78%, and Spring Peepers (*Pseudacris crucifer*) 59%]. Within wetland categories, 3 species were found to be present at over 50% of Urban Open Space wetlands: American Bullfrogs (63%), Southern Leopard Frogs (74%), and Green Frogs (79%; Table 1.1).

Species richness ranged from 0 – 10 species/site and was greatest in Low and High Urbanization sites (Figure 1.4), the estimates of richness within categories varied little (< 1 species) from observed values. All 12 species were detected in Low Urbanization wetlands and in High Urbanization wetlands, and 10 detected in Urban
Open Space wetlands (Table 1.1). As the percentage of impervious surface surrounding wetlands increased, species richness decreased at both the 300 m ($R^2 = 0.06$, $P = 0.09$; Fig. 1.5A) and 750 m ($R^2 = 0.12$, $P = 0.01$; Fig. 1.5B) scales, but the response to impervious surface was weak. Similarly, as road density surrounding a wetland increased, species density decreased at both the 300 m ($R^2 = 0.15$, $p = 0.005$; Fig. 1.6A) and 750 m ($R^2 = 0.23$, $P < 0.001$; Fig. 1.6B) scales. Analysis of $\beta$-diversity among wetlands within each class showed that Urban Open Space wetlands tend to be more variable from one another than either Low or High Urbanization wetlands (Figure 1.7). Indicator species analysis revealed Spring Peepers ($P < 0.001$), American Toads ($P = 0.002$), and Northern Cricket Frogs ($P = 0.01$) were significantly associated with Low Urbanization wetlands. Indicator value is derived from two components that are each calculated between 0 – 1: specificity, which is highest when a species is only present at sites within a category and fidelity, which is highest when a species is present at all sites within a category. Fidelity values for Spring Peepers and Northern Cricket Frogs were relatively high (0.87 and 0.68, respectively) compared to that of American Toads (0.56). Specificity values were at least 0.60 for all three species, but were particularly high for American Toads (0.79). No species were significant indicators for High Urbanization or Urban Open Space wetlands. The ordination of the entire species assemblage through RDA revealed that very little variation in structure (0.10) can be explained based on the three wetland land use categories.
DISCUSSION

Urbanization and road density had a weak impact on anuran communities; however, the direction of effect was consistent with what others have reported (Rubbo & Kiesecker 2005; Pillsbury & Miller 2008; Hamer & Parris 2011). Urban Open Space wetlands (wetlands within the urban matrix but immediately surrounded by greenspace) had lower species richness than wetlands in either High or Low Urbanization areas. The lower amount of impervious surface surrounding an Urban Open Space wetland at the core habitat may enable the wetland and surrounding area to function similarly to that of a Low Urbanization wetland, but the presence of a higher road density at this level may limit dispersal and effectively favor only a small subset of the available anuran community.

Development within Urban Open Space wetlands in our study area was indistinguishable from High Urbanization wetlands at larger spatial scales; however, Urban Open Space wetlands were surrounded by levels of development intermediate between low and High Urbanization sites at scales closer to the core habitat used by anurans (300 m). Road densities surrounding Urban Open Space wetlands are higher at both core habitat and average maximum migration scales than at either scale for Low and High Urbanization wetlands. This likely results from the smaller networks of roads that are abundant in areas supporting urban open spaces. Whereas a High Urbanization wetland may have more impervious surface surrounding it, little of that impervious surface is made up of roads, and if it is, it is most likely a larger highway running nearby the wetland. Urban open spaces on the other hand are generally located in areas
containing larger networks of smaller roads, and usually have grids of these small roads running through the open space themselves (walking and driving paths, cart paths, entrances and exits to the open space, etc.). While Urban Open Space wetlands may harbor less impervious surface in the surrounding upland habitat at the core level, this larger network of small roads can further fragment the landscape and act as a large barrier to dispersal of small bodied animals (Findlay et al. 2001; Eigenbrod et al. 2008).

Anuran communities appeared to be most influenced by landscape-scale factors rather than habitat specific factors at the individual wetland scale, particularly for Urban Open Space wetlands. The density and placement of roads in the surrounding landscape, and the fragmentation of upland habitats that coincide with this density affect anuran dispersal and species tend to have a negative relationship with urbanization (Rubbo & Kiesecker 2005; Pillsbury and Miller 2008). Species richness declines have been described for amphibians in other settings with fragmented landscapes, similar to the decline in species richness found at both landscape scales assessed in this study (Knutson et al. 1999; Lehtinen et al. 1999; Villaseñor et al. 2017). Generally, fewer species were found in Urban Open Space wetlands than were found in both Low and High Urbanization wetlands. Measures of $\beta$-diversity among wetland types also showed that wetlands located in urban open spaces tend to be more variable in species composition from one another than those in low or High Urbanization areas. Reviews of the literature show that the isolation of urban open space from other natural areas can lead to a shift in species assemblage as generalist species (urban adapters) continue to colonize whereas sensitive and specialist species disappear (Nielsen et al. 2014). Studies have also shown
that an increase in the amount of human recreation in an area can have negative effects, directly and indirectly on reptile and amphibian species (Larson et al. 2016; Larson et al. 2018). Variations in management strategies, and wetland/habitat quality among urban open spaces may function alongside isolation and size of the open space to affect the richness and diversity of vertebrates found therein.

Though the immediate surrounding habitat may be suitable for anurans, the isolation of this habitat by roads and the fragmenting of land between wetland sites can have an effect on the number of species present at Urban Open Space wetlands (Parris 2006). Those species that inhabit higher proportions of Urban Open Space wetlands tend to be those with overall larger body size. These species are also those that tend to exhibit postmetamorphic life stages that are primarily associated with aquatic habitats (He cnar & M’Closkey 1998; Rubbo & Kiesecker 2005) such as Green Frogs, American Bullfrogs, and Southern Leopard Frogs, though this is not always the rule. These species may not be as vulnerable to increases in road density surrounding wetlands as they tend to remain within or relatively near their aquatic habitats and are less affected by road mortality than some smaller bodied species (Spring Peepers, Green Treefrogs, Gray Treefrogs, etc.).

The extensive use of upland habitats by species such as Gray Treefrogs, Spring Peepers, and American Toads may be hindered by the higher presence of roads surrounding their breeding wetlands and may eventually drive them out of a wetland altogether (Rubbo & Kiesecker 2005; Parris 2006). A reduction in successful migration rates of juveniles and adults between ponds or to upland habitats can, over time, lead to a decline in the numbers of anuran species present within these environments.
CONCLUSION

This study shows the importance of understanding how factors in the landscape surrounding a wetland at different spatial scales drives anuran richness and community composition. I did not detect significant differences in community structure among wetland types, though I did see a reduction in species richness within Urban Open Space wetlands and with increasing levels of urbanization. The negative relationship between species richness and impervious surface (Fig. 1.5) reflects the findings of others. Such declines likely are a result of landscape alterations, decreased wetland/upland connectivity, and decreased wetland availability (Lehtinen et al. 1999, Rubbo & Kiesecker 2005, Parris 2006, Gagne & Fahrig 2007). There may be other important factors at the local or landscape scale that drive shifts in species richness and community structure that I did not take into consideration. The percentage of impervious surface surrounding the wetlands within the study provided the basis for their assignments into each category: Low Urbanization, High Urbanization, or Urban Open Space. The fact that I noticed a negative relationship between species richness with increasing urbanization, but no discernable trend among wetland categories, may indicate that responses to development are continuous rather than threshold-like.

If anuran diversity is to be maximized and maintained, better understanding of the key factors that drive their populations within urban environments is imperative. Management strategies should take into account the level of urbanization and the density of roads surrounding a wetland. Where possible, minimizing the density of roads surrounding urban open spaces may serve to benefit anurans as they move within and
between open spaces themselves and the surrounding landscape. Additionally, understanding how the utilization of differing management strategies within urban open spaces affects anurans may further enhance the conservation potential for these spaces.
Table 1.1. Proportion of sites occupied for anuran species among wetland types observed during surveys February-July 2018 in the South Carolina Piedmont ecoregion. Wetlands were divided into one of three categories [Low Urbanization (L), High Urbanization (H), and Urban Open Space (UOS)] based on percentage of impervious surface at the 750 m scale (*sensu* Calhoun et al. 2005).

<table>
<thead>
<tr>
<th>Taxon</th>
<th>Common Name</th>
<th>Wetland Type</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Acris crepitans</em></td>
<td>Northern Cricket Frog</td>
<td>0.69 0.44 0.37</td>
</tr>
<tr>
<td><em>Anaxyrus americana</em></td>
<td>American Toad</td>
<td>0.56 0.25 0.16</td>
</tr>
<tr>
<td><em>Anaxyrus fowleri</em></td>
<td>Fowler’s Toad</td>
<td>0.75 0.50 0.47</td>
</tr>
<tr>
<td><em>Gastrophryne carolinensis</em></td>
<td>Eastern Narrow-mouthed Toad</td>
<td>0.19 0.38 0.21</td>
</tr>
<tr>
<td><em>Hyla cinerea</em></td>
<td>Green Treefrog</td>
<td>0.50 0.56 0.32</td>
</tr>
<tr>
<td><em>Hyla versicolor</em></td>
<td>Gray Treefrog</td>
<td>0.81 0.81 0.37</td>
</tr>
<tr>
<td><em>Lithobates catesbeianus</em></td>
<td>American Bullfrog</td>
<td>0.75 0.75 0.63</td>
</tr>
<tr>
<td><em>Lithobates clamitans</em></td>
<td>Green Frog</td>
<td>0.88 0.88 0.79</td>
</tr>
<tr>
<td><em>Lithobates palustris</em></td>
<td>Pickerel Frog</td>
<td>0.19 0.06 0.00</td>
</tr>
<tr>
<td><em>Lithobates sphenoecephalus</em></td>
<td>Southern Leopard Frog</td>
<td>0.81 0.81 0.74</td>
</tr>
<tr>
<td><em>Pseudacris crucifer</em></td>
<td>Spring Peeper</td>
<td>0.88 0.63 0.32</td>
</tr>
<tr>
<td><em>Pseudacris feriarum</em></td>
<td>Upland Chorus Frog</td>
<td>0.13 0.13 0.00</td>
</tr>
</tbody>
</table>
FIGURE LEGENDS

Figure 1.1. Sampled wetland sites (red circles) are represented on the main map. The study area was within the South Carolina Piedmont Ecoregion and is shown on the inset.

Figure 1.2. Box and whisker plots of percent of impervious surface within (a) a 750 m and (b) 300 m buffer of land surrounding wetlands in the South Carolina Piedmont ecoregion. Wetlands were divided into one of three categories (Low Urbanization, High Urbanization, and Urban Open Space) based on percentage of impervious surface at the 750 m scale (*sensu* Calhoun et al. 2005). Dark bars represent median value, boxes represent the lower (25%) and upper (75%) quartiles, and whiskers represent the lowest and highest observed values up to 1.5 times the inter-quartile range. Values > 1.5 times the inter-quartile range are represented by open circles. Letters within boxes indicate pairwise significant differences determined through Tukey HSD test.

Figure 1.3. Box and whisker plots of percent of road density in meters within (a) a 750 m and (b) 300 m buffer of land surrounding wetlands in the South Carolina Piedmont ecoregion. Dark bars represent median value, boxes represent the lower (25%) and upper (75%) quartiles, and whiskers represent the lowest and highest observed values up to 1.5 times the inter-quartile range. Values > 1.5 times the inter-quartile range are represented
by open circles. Letters within and outside boxes indicate pairwise significant differences
determined through Tukey HSD test.

Figure 1.4. Chao estimated anuran species richness across three wetland types in the
South Carolina Piedmont ecoregion.

Figure 1.5. Observed anuran species richness by percentage of impervious surface within
(a) a 750 m ($r^2 = 0.1198$, $p = 0.01$) and (b) 300 m ($r^2 = 0.0565$, $p = 0.09$) buffer of land
surrounding wetlands in the South Carolina Piedmont ecoregion.

Figure 1.6. Observed anuran species richness by road density in meters within (a) a 750
m ($R^2 = 0.2327$, $p < 0.001$) and (b) 300 m ($R^2 = 0.1461$, $p = 0.005$) buffer of land
surrounding wetlands in the South Carolina Piedmont ecoregion.

Figure 1.7. Box and whisker plot showing measurements of $\beta$-diversity (as measured by
Bray-Curtis distance) among wetlands types in the South Carolina Piedmont ecoregion.
Dark bars represent median value, boxes represent the lower (25%) and upper (75%)
quartiles, and whiskers represent the lowest and highest estimate of $\beta$-diversity.
Figure 1.1
Figure 1.2

A) 

B) 

% Impermeable Surface in 750 m

High   Low   Urban Open Space

Wetland Type

% Impermeable Surface in 300 m

High   Low   Urban Open Space

Wetland Type
Figure 1.3

A)

B)
Figure 1.4
Figure 1.5

A)  

B)
Figure 1.6

A) 

B)
Figure 1.7

Box plots showing β-diversity (Bray-Curtis Index) across different wetland types:

- High
- Low
- Urban Open Space

Y-axis: β-diversity (Bray-Curtis Index)
X-axis: Wetland Type
REFERENCES


CHAPTER TWO

Do open spaces within an urban matrix increase anuran abundance probability?

ABSTRACT

While many threats face amphibian populations, urbanization is among the leading causes of amphibian declines worldwide. Urban open spaces, such as golf courses and parks, can potentially provide suitable habitats to amphibians within urbanized matrices. During the spring and summer of 2018, I conducted active call surveys to determine the relative abundance of anurans at 51 wetland sites within the Piedmont ecoregion of South Carolina. Approximately one-third of these wetlands are located within urban open spaces, while the remainder are situated along a gradient of development. I evaluated species-specific relative abundances as a function of wetland type (Urban Open Space, High Urbanization, or Low Urbanization), several within-wetland variables and two landscape-scale covariates. No species were influenced by wetland type, however impervious surface had a negative influence on the abundance of one species whereas road density negatively affected three species of anurans. Roads negatively impact anurans through direct mortality, dispersal limitations, and habitat fragmentation. Conservation efforts conducted within open spaces should attempt to focus on issues not only at an individual wetland scale, but also at scales outside of the open spaces themselves. Understanding how urbanization at various spatial scales effects anuran species can inform efforts to bolster amphibian conservation efforts in urban matrices.
INTRODUCTION

Amphibian populations are declining world-wide, with North America being no exception (Stuart et al. 2004; Adams et al. 2013; Miller et al. 2018). While there are multiple drivers of these declines, urbanization is a key process threatening amphibians (Hamer & McDonnel 2008; Marsh et al. 2017). Urbanization is widespread (Moore et al. 2003; Wang et al. 2012; World Bank 2018) and leads to rapid replacement of natural habitat with urban infrastructure such as buildings, houses, roads, paved areas, and other impervious surfaces (McDonnell & Pickett 1990; Hamer & McDonnell 2010). Urbanization can negatively affect amphibian populations via outright habitat destruction (Rubbo & Kiesecker 2005), a reduction of habitat quality (Pope et al. 2000; McKinney 2002), an increase in vehicle-related mortality and isolating otherwise suitable habitats (Lehtinen et al. 1999; Elzanowski et al. 2009; Smallbone et al. 2011).

As the density of urbanization increases, the need to pinpoint areas that can be utilized for the conservation of species that would otherwise be negatively affected becomes apparent. Urban open spaces (also referred to as green spaces) may act as important areas of biodiversity conservation (Goddard et al. 2009). Urban open spaces area defined within this study as publicly accessible, managed outdoor spaces that are partially to completely covered by a significant amount of vegetation that exist primarily as semi-natural areas within an urbanized matrix (Jim & Chen 2003; Kong et al. 2009; United States Environmental Protection Agency 2017). These open spaces include community gardens, public parks, sports recreation zones (e.g. golf courses), or cemeteries. Urban open spaces can aid in alleviating the negative effects that habitat loss,
fragmentation, and isolation have on native biodiversity (Kong et al. 2009) and can sometimes create new habitat for wetland breeding amphibians (Birx-Raybuck et al. 2010; Brand & Snodgrass 2010; Marsh et al. 2017).

Anurans represent a group of amphibians that may benefit from the proper management of open spaces and the wetlands that they contain. At a higher risk of extinction than other taxa (Wake & Vredenburg 2008; Hamer & McDonnell 2008), their relatively small home ranges and small body size make them excellent candidates for studies focusing on contained effects of urbanization. Anurans are largely wetland-associated amphibian species which require ample space in and around aquatic and terrestrial habitats to carry out essential life-history processes such as reproduction, sheltering, and foraging. Reviews of the literature indicate that while urban research concerning anurans is heavily focused on assemblage-level response, many species-specific responses to urbanization are poorly understood or unknown altogether (Scheffers & Paszkowski 2012). Different species of anurans are likely to be affected by urbanization in different ways as dispersal needs and capabilities, body size, and breeding strategies vary (Gagne & Fahrig 2010; Marsh et al. 2017). Overall, many anuran communities have been shown to respond negatively to urbanization (Hamer & McDonnell 2008; Scheffers & Paszkowski 2012), but assessing species-specific responses to urbanization can facilitate more efficient population management strategies (Cushman 2006).

Reducing declines in anuran diversity depends on managing and protecting existing habitat to mitigate negative impacts toward wildlife (Puglis & Boone 2012).
Some man-made wetlands in urban open spaces are inhabited by anurans as they are similar to wetlands in otherwise natural habitats (Brand & Snodgrass 2010). These spaces have the potential to provide habitat and offer habitat connectivity for anurans in an otherwise unsuitable, urbanized matrix (Hamer & Parris 2011; Puglis & Boone 2012). A majority of studies have examined variables affecting amphibians along a forested–urban gradient, with very few comparing forested or urban sites with open spaces such as golf courses (Scheffers & Paszkowski 2012).

I identified 51 wetlands along a rural-urban gradient in the upstate of South Carolina that vary in the amount of urbanization surrounding the wetland. Literature reviews indicated that wetland-breeding amphibians require anywhere from 300 – 750 m of upland habitat surrounding the breeding site to complete necessary life-history processes (Semlitsch 2000; Semlitsch & Bodie 2003; Parris 2006; Birx-Raybuck et al. 2009). I assessed the amount of urbanization surrounding these wetlands by overlaying buffers of 300 m (core habitat) and 750 m (average maximum dispersal). The amount of impervious surface within the large buffer as well as the presence of open space (as defined above) was used to place wetlands into one of three categories: Urban Open Space, Low Urbanization, or High Urbanization. My goal was to evaluate how varying amounts of urbanization and road density in the surrounding landscape, as well as wetland-specific attributes, affected the abundance of various anuran species. Call survey data were used to obtain relative abundance of anuran species. I developed a priori hypotheses for relationships I expected to find for each species based on habitat preferences described in the literature. I expected that anurans preferring to breed in
temporary wetlands would be associated with wetlands with shorter hydroperiods whereas those preferring wetlands with longer hydroperiods would be associated with larger more permanent wetlands. I hypothesized that species with overall larger body sizes would be less affected by higher levels of urbanization and road densities than those with comparatively smaller body sizes as they may have greater dispersal and migration abilities, comparatively. I also predicted that all species observed would have an increased probability of occupancy at sites surrounded by open space compared to High Urbanization wetlands.

**METHODS AND MATERIALS**

*Study Area and Wetland Categorization*

I identified 73 potential wetland sites using Google Earth Pro and a file of South Carolina wetlands provided by the National Wetlands Inventory (NWI; https://www.fws.gov/wetlands/Data/State-Downloads.html). Using this wetland geodatabase and various county assessor websites, I was able to obtain landowner information for privately owned wetlands. I narrowed the list of wetlands to 51 by contacting landowners to seek permission to access their property and through subsequent wetland visits. Wetlands were chosen for inclusion in the study based on accessibility and with the goal of creating a gradient of development surrounding them (assessed visually using Google Earth Pro). Study wetlands were in Anderson, Oconee, and Pickens counties, South Carolina and range from rural areas to areas of high urbanization. I specifically targeted
wetlands within urban open spaces (golf courses, parks, and gardens) in order to address the objectives of my study.

A handheld Garmin GPS unit (GPSmap 62s, Garmin, Ltd.) was used to collect coordinates at each wetland and these coordinates were loaded into ArcMap 10.4.1. I then delineated each wetland as a unique polygon in ArcGIS (ESRI, Redlands, CA) using a satellite imagery base map and knowledge of the actual wetland boundaries obtained from ground truthing. Two buffers (300m and 750) were added around each wetland perimeter, representing the average core habitat and maximum average dispersal range from the wetland for the anuran species expected to be found within the study area (Semlitsch & Bodie 2003, Parris 2006, Birx-Raybuck et al. 2009). I placed each wetland into one of three categories, as described in Chapter 1. Wetlands with \( \geq 30\% \) impervious surface within the 750–m buffer were given the label “High Urbanization”, whereas wetlands with < 30% impervious surface were labelled “Low Urbanization”. Wetlands within open spaces were labelled “Urban Open Space”. This resulted in 16 Low Urbanization, 16 High Urbanization, and 19 Urban Open Space wetlands. Percentage of impervious surface surrounding wetlands in urban open spaces was not used in labelling these wetlands; my Urban Open Space category included sites that would have been placed in either of the other two categories. The distinct management strategies associated with wetlands and adjacent terrestrial habitat in these areas create the impetus for assigning them to their own category. Previous analysis (Chapter 1) revealed that Urban Open Space wetlands are intermediate with respect to percent impervious surface relative to Low Urbanization and High Urbanization wetlands within the core buffer (300
m); however, the impervious surface content more closely resembles High Urbanization wetlands at the larger buffer size (Figure 1.1). Road densities surrounding Urban Open Space wetlands are higher than road densities at both Low Urbanization and High Urbanization wetlands at both buffer sizes (Figure 1.2). Low Urbanization wetlands had the highest mean canopy cover of all wetland types and showed the greatest difference between High Urbanization and Urban Open Space wetlands; there was no significant difference in canopy cover between Urban Open Space and High Urbanization wetlands (Chapter 1).

**Wetland Landscape and Site Characteristics**

We collected data on both landscape- and local-level factors hypothesized to influence the abundance (as measured by calling index) of frogs. I obtained data on developed land, impervious surface, and road density from the South Carolina Department of Transportation (SCDOT). Using these data, I was able to calculate levels of urbanization and road density within the radii of both buffer zones. Road density was calculated as the total road length in meters within each buffer, whereas impervious surface and developed land were calculated as the percent of coverage. I obtained a secondary layer of wetlands from the NWI that included all wetlands in South Carolina to determine the distance from the nearest body of water to my study wetlands. This was then split into two separate categories [riverine bodies and freshwater wetlands (freshwater emergent, forested/shrub, and ponds)]. The straight-line distances from each wetland to each of the wetland categories was then calculated. To account for the possibility that study sites may be each other’s closest neighbor I included the study sites
when determining the distance to other freshwater wetlands, as several were not actually included in the NWI layer.

I recorded within-wetland habitat data during daytime dip net surveys that were performed as part of a separate study from March – July 2018 (Chapter 1). Wetland size was determined through ArcGIS via a wetland delineation process by which I created polygons for each wetland individually. Depth (m) was measured at the deepest point using a meter stick, and a depth of >1.2 m was assigned to wetlands deeper than the 1.2 m limit of the depth stick used. I obtained a wetland’s average organic layer depth by measuring the depth (cm) of the submerged organic layer at each dip net stop and then averaging them together for each wetland. I measured canopy cover at each wetland after leaf out (July) by taking photos at each cardinal location using an iPhone 7 (Apple, Cupertino, CA) paired with a fisheye lens attachment (Amir, Shenzhen, Guangdong, China). I only took one photo to obtain canopy cover at wetlands small enough (< 0.01 ha) where multiple photos were unnecessary. I used the Gap Light Analyzer (Cary Institute of Ecosystem Studies, Millbrook, NY) to attain a percentage of canopy cover for each photo. I averaged canopy cover values across all of a wetland’s photos to produce its canopy cover measurement. Aquatic vegetation cover was measured by making visual estimates of the amount of emergent and submerged aquatic vegetation categorizing these estimated (1 = 0-25%, 2 = 26-50%, 3 = 51-75%, 4 = 76-100%). I recorded the presence or absence of a zone of herbaceous terrestrial vegetation surrounding wetlands, and assigned wetland categories according to “edge type” where (1 = edge vegetation present, 0 = edge vegetation absent). A zone of terrestrial vegetation had to be at least 1 m wide.
and present around at least half of the wetland edge be labelled as present. Terrestrial vegetation surrounding wetlands in urban open spaces may provide a more suitable habitat structure for anurans, offering shelter for adults and metamorphs and acting as calling and amplexus sites (Parris 2006; Puglis & Boone 2012). Buffer zones such as these can also help alleviate the effects that applied chemicals can have on a wetland (Puglis & Boone 2012). I visually noted fish presence at wetlands and this was further assessed through dip net surveys. Hydroperiod for a wetland was determined for the study season (February – July 2018) by noting the presence of absence of water during each survey, call and dip net. Wetlands that were dry at any point during the study were assigned a value = 0 (Temporary), whereas those that held water throughout the period of the study were given a value = 1 (Permanent). Water temperature, conductivity, and pH was measured using an Oakton PCTStestr™ (Cole Parmer, Vernon Hills, IL) during each dip net visit. A Kestrel 2500 unit (Nielsen-Kellerman, Boothwyn, PA) was used to gather air temperature, relative humidity, and max wind speed.

I evaluated the Pearson correlation coefficients between all pairs of landscape and local variables and removed one member of the pair when $|r| > 0.70$. Percent developed land was removed as it was highly correlated with impervious surface at both buffer sizes. Both impervious surface and road length at the 750 m scale were removed as site covariates as they were highly correlated with impervious surface and road length at the 300 m scale (0.91 and 0.78, respectively). Wetland depth was removed as it was determined that hydroperiod may be a more important factor than max depth of a wetland.
for anuran species as hydroperiod and max depth are typically highly correlated (Babbitt et al. 2003)

**Anuran Call Surveys**

Anuran call surveys were conducted once per month February – June 2018 for a total of five call surveys per wetland. These were performed during the evenings beginning approximately 30 minutes after sundown and ending no later than 0100 the following morning. In an effort to maximize detection probability, surveys were only conducted when air temperatures were between 5.6 °C and 30 °C (NAAMP; Wier & Mossman 2005; Steelman & Dorcas 2010). Surveys were not conducted when wind speeds were consistently in excess of 8 – 12 mph or when there was heavy rainfall, as either of those conditions may negatively affect detection of frog calls. Five minutes was spent actively listening at each wetland and I recorded calls as an index of abundance per NAAMP protocol. The call index developed by NAAMP estimates abundance along a 1 – 3 scale with 1 representing a lower abundance and 3 representing the highest abundance (i.e. 1 = individuals can be counted, 2 = there may be overlap of calls but individuals can be distinguished, 3 = full chorus, calls constant, continuous, and overlapping). Studies have evaluated the effectiveness of this call index and its relation to actual abundance using mark – recapture and found positive relationships between call index and abundance in Green Frogs (Nelson & Graves 2004) and Boreal Chorus Frogs (*Pseudacris maculata*; Corn et al. 2000). To account for ambient noise surrounding a wetland I utilized the Massachusetts noise index where (0 = no effect on sampling, 1 = slight effect on sampling, 2 = moderate effect on sampling, 3 = serious effect on sampling, 4 =
profound effect on sampling). I also recorded Julian day as well as start and end times during each survey. Prior to analysis, months in which anurans were not observed calling were removed for each focal species. However, months during which a species was not observed calling were kept in the data set if the species was heard calling the month before and the month after.

**Data Analysis**

Using abundance index data from anuran call surveys I developed N-mixture models to investigate species-specific relationships between call activity and environmental site covariates, while simultaneously accounting for factors that may have influenced detection probability related to frog calls (Royle & Link 2005). We collected data on six detection probability covariates: Julian day, time of day, temperature, humidity, wind and noise. Noise was never incorporated as a detection covariate as there were no sampling occasions where noise levels reached a point that would prevent surveyors from hearing calls. Time of day was recorded as the start time of each five-minute survey. Temperature and humidity were logged as their maximums recorded during each survey. Wind was measured as the maximum wind speed during a survey and then converted into a binary format where “1” was assigned on occasions where wind speed exceeded 10 mph and “0” was assigned otherwise.

I began model evaluation by first examining the support for detection covariates modelled individually. Each of these univariate detection-only models and a null model were compared using Akaike’s Information Criterion (AIC; Burnham and Anderson 2002). Detection covariates with strong support (ΔAIC < 2.0) were evaluated together as
additive models to determine if more complex structures were warranted for the detection process. Once I identified the detection covariate model with the most support (ΔAIC = 0.00), I integrated this covariate in all further models assessing site covariates.

For each species I evaluate the influence of the three wetland categories against four other hypothesized predictors of call index: impervious surface, road density, a global model containing all site covariates except wetland category, and a null model. The relative evidence for each of these hypotheses was compared using AIC (Table 2.1). In the results I focus on relationships between call index and site covariates for models with ΔAIC < 2.0 and significant effect size for the parameter estimate (SE did not overlap with zero). All non-categorical variables were standardized to z-scores (mean = 0, standard deviation = 1). All models were developed using the “p-count” function within the unmarked package (Fiske & Chandler 2011) in Program R Version 3.4.1. (R Core Team 2017). Null and global models were evaluated for each species using each distribution option (Poisson, Negative Binomial, and Zero Inflated-Poisson) and then ranked by AIC to determine which distribution was best suited for each species. The Poisson distribution was used to construct models for American Bullfrogs, Fowler’s Toads, Gray Treefrogs, and Southern Leopard Frogs, whereas the Zero-Inflated Poisson distribution was used to construct models for Northern Cricket Frogs.

RESULTS

During the 5 month sampling period I collected 255 nights of call surveys resulting in 508 detections of 12 anuran species. Of the 12 species detected, 3 are listed
under the South Carolina Department of Natural Resources Wildlife Action Plan as priority species [Pickerel Frogs (*Lithobates palustris*), Northern Cricket Frogs (*Acris crepitans*), and Upland Chorus Frogs (*Pseudacris feriarum*)] (South Carolina Department of Natural Resources, 2015). Detections for these 12 species ranged from 10 – 92 (mean = 42.3, SD = 25.04). Five species observed during the study were chosen for inclusion in N-mixture models because detections were sufficient to allow for parameter estimates: American Bullfrogs (*Lithobates catesbeianus*), Fowler’s Toads (*Anaxyrus fowleri*), Gray Treefrogs (*Hyla versicolor*), Northern Cricket Frogs (*Acris crepitans*), and Southern Leopard Frogs (*Lithobates sphenoecephalus*).

Temperature was an important influence on detection probability for many species (Table 2.2). Detection probability increased for American Bullfrogs, Fowler’s Toads, and Gray Treefrogs but decreased for Southern Leopard Frogs as temperature increased. Along with an increase in temperature, detection of Gray Treefrogs increased with time of day (latest sampling time for the species was 2328). Time of year also had an influence on detection probability, such that detections were higher later in the season for Northern Cricket Frogs, but decreased in later samples for Southern Leopard Frogs.

Impervious surface and roads surrounding a wetland had an effect on calling index (hereafter, relative abundance) of all species (Table 2.2). Increasing impervious surface had a negative impact on the relative abundance of Northern Cricket Frogs and Fowler’s Toads whereas an increase in road density negatively affected the relative abundance of Gray Treefrogs, American Bullfrogs, and Southern Leopard Frogs. American Bullfrogs appeared to be the only species affected by wetland type: Compared
to High Urbanization wetlands, relative abundance increased within Low Urbanization 
wetlands and decreased within Urban Open Space wetlands, though these trends were 
relatively weak. The null model was also present in top ranked models for American 
Bullfrogs, Fowler’s Toads, and Northern Cricket Frogs.

DISCUSSION

As predicted, impervious surface and road density negatively influenced the 
relative abundance value for most of the wetland-breeding anurans we evaluated; 
however, wetland category was not a highly ranked hypothesis for most species. 
Impervious surface or road density appeared among the top models for four of the five 
species we surveyed, and in all cases the relationship with these variables was negative 
(Figure 2.2). Two relatively small bodied species, Northern Cricket Frogs and Gray 
Treefrogs, had a negative relationship with impervious surface and road density, 
respectively, and road density negatively affected two larger bodied species, American 
Bullfrogs and Southern Leopard Frogs.

Negative effects of road density and impervious surfaces on species richness and 
abundance are well documented (Fahrig et al. 1995; Findlay & Houlan 1997; Knutson 
et al. 1999; Findlay et al. 2001; Marsh et al. 2017). Anurans travelling amongst breeding 
wetlands and upland habitats experience relatively high rates of mortality while crossing 
roads (Ashley & Robinson 1996; Mazerolle 2004; Consentino et al. 2014) which could 
affect a species ability to colonize a wetland due to adult mortality prior to breeding as 
well as adult and juvenile mortality during dispersal. Aside from effects of direct 
mortality, roads produce behavioral and physical obstacles to movement (Bouchard et al.
These obstacles can negatively affect a species ability to make seasonal migrations and disrupt dispersal within metapopulations (Gibbs 1998; Hels & Nachmann 2002; Consentino et al. 2014). Though I hypothesized that road densities would have less of an effect on larger bodied species relative to smaller species, American Bullfrogs and Southern Leopard Frogs were negatively affected by roads. Marsh et al. (2017) showed that at smaller scales (~300 m, the same scale tested in this study) road density negatively affected American Bullfrogs as well as Gray Treefrogs, while Southern Leopard Frogs showed no relationship at any scale. Consentino et al. (2014) found that roads negatively affected Southern Leopard Frogs, though the relationship was not highly significant as standard error estimates for effect size crossed zero. The negative effect of roads on these anuran species may be the result of higher densities of smaller, secondary roads surrounding wetlands acting as dispersal barriers, fragmenting habitat and causing direct mortality. At the 300 m scale, road densities may affect different species of anurans in similar manners, regardless of body size or movement ability (Consentino et al. 2014).

The absence of wetland type (Low, High, or Urban Open Space) from the list of well-supported, species-specific models may have resulted because the impervious surface thresholds used for categorization did not represent biologically relevant changes among the three systems. Wetland type only appeared in the top ranking models for American Bullfrogs, though the trends associated with wetland types were relatively weak. I attempted to define Urban Open Spaces based on the surrounding land use; however, the diversity of uses present within this definition yielded a wide range of wetland environments and surrounding buffers so that not all of the wetlands within my
Urban Open Space label functioned similarly to one another. It has been suggested that urban open spaces, particularly golf courses, should be included in residential designs in a manner that promotes higher levels of biodiversity (Colding et al. 2006). Though the construction of open spaces in a manner that facilitates biodiversity may not always be realized as many urban open spaces are constructed in association with real estate projects (Mulvihill 2001) and are therefore deep within the urbanization matrix. This configuration may cause the wetlands and surrounding terrestrial habitats within such urban open spaces to function similar to High Urbanization wetlands and thus may not provide the same benefits to wildlife species that open spaces with minimal surrounding development can (Price et al. 2013). One study found that higher amounts of residential development within and around the boundaries of golf courses had a negative effect on the abundance of semi-aquatic turtles; whereas, wetlands within golf courses surrounded by lower amounts of residential development supported turtle abundances equal to those in more natural ponds (Price et al. 2013). Similarly, land alterations within an open space can lead to homogenization of the landscape and a reduction in habitat quality (Puglis & Boone 2012), a process not taken into account when assigning wetland categories.

The presence of the null model in the top-ranked models for American Bullfrogs, Fowler’s Toads, and Northern Cricket Frogs suggests that although there are landscape variables affecting relative abundance, they may not be the most important driver of abundances for these species within the study area. There may be other landscape-scale or within site variables that have a greater influence on anuran abundance that were not evaluated within this study.
Observing anuran occupancy and abundance across one field season may be insufficient, as multi-seasonal colonization and extinction patterns can offer insights not described here (Randall et al. 2015). Further, the presence of calling males at a site does not necessarily indicate successful reproduction, only the presence of adult males. Long-term studies would allow for characterization of reproductive success as well as metapopulation dynamics among wetlands. Such efforts are necessary to determine if wetlands in urban open spaces and the open spaces themselves can serve as source populations for amphibians, or if they largely function as sinks instead (Puglis & Boone 2012).

**CONCLUSION**

Development-related variables were consistent predictors of call index, although the well-supported models were variable across species. Multiple factors are known to shape anuran occupancy or abundance at wetlands (Pillsbury & Miller 2008; Hamer & Parris 2011; Birx-Ryback et al. 2010). Other researchers have concluded that landscape variables alone explained < 35% of the variation within their datasets (Bonin et al. 1997; Hecnar 1997; Knutson et al. 1999). However, Beebee (1985) found that landscape variables are better predictors than individual wetland characteristics in determining amphibian diversity. Additional work should be performed to more clearly quantify whether urban open space wetlands provide habitat that is distinct from other wetlands, and if there are subsets of open space that offer more suitable breeding and adjacent upland habitat.
The success of amphibian conservation relies on a continued effort to understand the specific mechanisms that drive community structure and distribution within an urbanized environment. Long term monitoring along multiple spatial scales at a variety of wetland types will only add to our knowledge of these species and the factors that influence them, helping to more effectively conserve and reestablish amphibian populations and the habitat they require.
**TABLES**

Table 2.1. Site and landscape scale call index covariates used in single-species N-mixture models for anurans along a rural-urban gradient in the South Carolina Piedmont ecoregion.

<table>
<thead>
<tr>
<th>Occupancy covariate</th>
<th>Scale</th>
<th>Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>type</td>
<td>Landscape</td>
<td>Categorical</td>
<td>Wetland type based off of % impervious surface at 750 m (Low, High, Open Space)</td>
</tr>
<tr>
<td>%Imp300</td>
<td>Landscape</td>
<td>Continuous</td>
<td>Percentage of impervious surface surrounding a site within 300 m</td>
</tr>
<tr>
<td>Road300</td>
<td>Landscape</td>
<td>Continuous</td>
<td>Road density (m) surrounding a site within 300 m</td>
</tr>
<tr>
<td>river_near</td>
<td>Landscape</td>
<td>Continuous</td>
<td>Distance (m) from site to nearest riverine wetland</td>
</tr>
<tr>
<td>freshwet_near</td>
<td>Landscape</td>
<td>Continuous</td>
<td>Distance (m) from site to nearest freshwater wetland (freshwater emergent, forested/shrub, and ponds)</td>
</tr>
<tr>
<td>aquatic_veg</td>
<td>Site</td>
<td>Categorical</td>
<td>Average aquatic vegetation percent at a site. Combined reeds, aquatic grass, and other aquatic vegetation</td>
</tr>
<tr>
<td>terrestrial_veg</td>
<td>Site</td>
<td>Categorical</td>
<td>Presence/absence of a terrestrial buffer zone of vegetation at least 1 m wide and around at least 1/2 of the wetland edge</td>
</tr>
<tr>
<td>hydroperiod</td>
<td>Site</td>
<td>Categorical</td>
<td>Binary category indicating temporary or permanent wetland</td>
</tr>
<tr>
<td>fish</td>
<td>Site</td>
<td>Categorical</td>
<td>Binary category indicating fish presence or absence</td>
</tr>
<tr>
<td>canopy</td>
<td>Site</td>
<td>Continuous</td>
<td>Average site canopy cover</td>
</tr>
<tr>
<td>org_depth</td>
<td>Site</td>
<td>Continuous</td>
<td>Average depth of organic layer within a wetland</td>
</tr>
<tr>
<td>area</td>
<td>Site</td>
<td>Continuous</td>
<td>Size of site (ha)</td>
</tr>
<tr>
<td>pH</td>
<td>Site</td>
<td>Continuous</td>
<td>Average site pH</td>
</tr>
<tr>
<td>cond</td>
<td>Site</td>
<td>Continuous</td>
<td>Average site conductivity</td>
</tr>
</tbody>
</table>
Table 2.2. Most supported (∆AIC < 2.0) single species N-mixture models that evaluated univariate landscape scale habitat covariates along with a null and global model and the top species-specific detection covariate. Models were applied separately to five species with encounter histories sufficient for parameter estimation. The final column represents the β-estimate and standard error associated with the covariate of relative abundance. In the American Bullfrog section, β-estimate and standard error values are given for Low Urbanization wetlands (Low) and Urban Open Space wetlands (UOS) in relation to High Urbanization wetlands.

<table>
<thead>
<tr>
<th>Model, by species</th>
<th>∆AIC</th>
<th>w</th>
<th>k</th>
<th>β-Estimate and SE</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Northern Cricket Frog</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>λ(Null)p(JD)</td>
<td>0.00</td>
<td>0.4400</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>λ(Impervious)p(JD)</td>
<td>1.05</td>
<td>0.2600</td>
<td>5</td>
<td>-0.237 (± 0.229)</td>
</tr>
<tr>
<td>λ(Road)p(JD)</td>
<td>1.69</td>
<td>0.1900</td>
<td>5</td>
<td>-0.118 (± 0.211)</td>
</tr>
<tr>
<td><strong>Fowler's Toad</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>λ(Impervious)p(Temp)</td>
<td>0.00</td>
<td>0.3860</td>
<td>4</td>
<td>-0.229 (± 0.156)</td>
</tr>
<tr>
<td>λ(Null)p(Temp)</td>
<td>0.30</td>
<td>0.3320</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td><strong>Gray Treefrog</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>λ(Road)p(Temp+TOD)</td>
<td>0.00</td>
<td>0.8995</td>
<td>5</td>
<td>-0.643 (± 0.186)</td>
</tr>
<tr>
<td><strong>American Bullfrog</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>λ(Null)p(Temp)</td>
<td>0.00</td>
<td>0.3280</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>λ(Road)p(Temp)</td>
<td>0.36</td>
<td>0.2744</td>
<td>4</td>
<td>-0.2159 (± 0.176)</td>
</tr>
<tr>
<td>λ(Type)p(Temp)</td>
<td>0.82</td>
<td>0.2172</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>λ(Impervious)p(Temp)</td>
<td>1.21</td>
<td>0.1789</td>
<td>4</td>
<td>-0.133 (± 0.153)</td>
</tr>
<tr>
<td><strong>Southern Leopard Frog</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>λ(Road)p(Temp+JD)</td>
<td>0.00</td>
<td>0.6420</td>
<td>5</td>
<td>-0.522 (± 0.255)</td>
</tr>
</tbody>
</table>
FIGURE LEGENDS

Figure 2.1 Expected call index of A) American bullfrog, B) Gray Treefrogs, and C) Southern Leopard Frogs by road density within a 300 m wetland buffer. D) Northern Cricket Frog call index was best predicted by % impervious surface surrounding a wetland within a 300 m buffer zone. All surveyed wetlands were in the South Carolina Piedmont ecoregion.
REFERENCES


