

JENNINGS, ANDREW JAMES BLEVINS, M.S. The Effects of Preformed Scour Holes on Anuran Biodiversity in the North Central Piedmont Region. (2013)
Directed by Dr. Stanley H. Faeth, 65 pp.

Urbanization negatively affects many biotic communities throughout the world. In the southeast United States, cities are expanding into previously rural and forested areas, reducing quality and quantity of habitat which may lead to declines in species diversity. To mitigate the impact that urbanization has on natural systems, the North Carolina Department of Transportation (NCDOT) installed a new stormwater control, the Preformed Scour Hole (PSH). However the impact of PSHs on aquatic and amphibious populations and communities, including anurans (frogs and toads), is unknown. Using the manual calling survey protocol, I surveyed the anuran populations in PSHs in Guilford, Alamance, Caswell, and Randolph counties. I correlated species richness and individual species presence with local factors associated with patch quality and regional factors associated with connectedness.

I found that, as predicted, degree of urbanization was negatively associated and PSH surface area and the presence of riparian vegetation was positively associated, with the total number of species present in PSHs. Additionally, I found that the presence of different genera were either positively (in the case of *Lithobates*) or negatively (in the case of *Pseudacris*) correlated with increases in urbanization.

The results of this study suggest that PSHs may help to mitigate anuran diversity loss due to urbanization. Furthermore, my results suggest that genera-specific models should be created to explain patterns of anuran diversity in urban areas. However, this study was unable to determine PSHs effect on anuran individual fitness or population size.

THE EFFECTS OF PREFORMED SCOUR HOLES ON ANURAN
BIODIVERSITY IN THE NORTH CENTRAL
PIEDMONT REGION

by

Andrew James Blevins Jennings

A Thesis Submitted to
the Faculty of The Graduate School at
The University of North Carolina at Greensboro
in Partial Fulfillment
of the Requirements for the Degree
Master of Science

Greensboro
2013

Approved by

Committee Chair

APPROVAL PAGE

This thesis written by Andrew James Blevins Jennings has been approved by the following committee of the Faculty of The Graduate School at The University of North Carolina at Greensboro

Committee Chair _____

Committee Members _____

Date of Acceptance by Committee

Date of Final Oral Examination

ACKNOWLEDGMENTS

I thank my advisor, Dr. Stan Faeth, and committee members Dr. Matina Kalcounis-Ruppell and Dr. Gideon Wasserberg, for their assistance with my research and with my thesis. I thank Dr. Roland Deustch for his assistance with my statistical analysis. I thank Dr. Michael Dorcas, Mr. Jeff Beene, and Dr. Steven Price for their assistance in establishing the species mix for this study and for answering questions in regards to anuran dispersal. I thank the North Carolina Department of Transportation for providing study sites. I thank Danielle Hayes, Ron Baron, Nicole Owens, Laura Fondario, Austin Craven, Marcelo Schwartz, Adam Speen, and Reece Jennings for their assistance during the field season of 2012. I thank the UNCG Biology Department for funding my research.

TABLE OF CONTENTS

	Page
LIST OF TABLES	v
LIST OF FIGURES	vi
CHAPTER	
I. INTRODUCTION	1
II. MATERIALS AND METHODS.....	23
III. RESULTS	30
IV. DISCUSSION	34
REFERENCES	57

LIST OF TABLES

	Page
Table 1. The anuran species pool for each county.....	49
Table 2. Values of local scale factors that were measured in the study.....	50
Table 3. Values of regional scale factors measured in the study.....	51
Table 4. Species presence and activity level for each site for each sampling period in 2012.....	52
Table 5. Nested matrix of species presence.....	54
Table 6. The biodiversity model for the presence of all species.....	55
Table 7. The logit model for <i>P. crucifer</i> presence.....	55
Table 8. The logit model for <i>P. feriarum</i> presence.....	55
Table 9. The logit model for <i>L. catesbianna</i> presence.....	56
Table 10. The logit model for <i>L. clamitans</i> presence.....	56

LIST OF FIGURES

	Page
Figure 1. The four counties and locations of PSH within.....	42
Figure 2. Total and average activity level for all species for each sampling period.....	43
Figure 3. Activity level for each species during each sampling period.....	44
Figure 4. Association of PSH surface area and anuran biodiversity.....	45
Figure 5. Species-Area Relationship curve for anuran diversity	46
Figure 6. Association of the presence of riparian vegetation and anuran biodiversity	46
Figure 7. Association of urbanization and anuran biodiversity	47
Figure 8. Association of urbanization and the presence of individual species.....	48

CHAPTER I

INTRODUCTION

Throughout the world, human populations are becoming more urban and less rural. Currently more than 50% of the world's population lives in urban areas and that number is expected to rise to over 60% by 2035 (UN 2012). In the United States, over 80% of the population lives in cities and that percentage is expected to increase to over 85% by 2025 (UN 2012). Urbanization is associated with habitat fragmentation and loss (McKinney 2006), hydrographic changes (Walsh et al. 2005), changes in nutrient availability (Lewis et al. 2006), and introduction of non-native species (McKinney 2008). One consequence of these changes due to urbanization is the loss of biodiversity of native flora and fauna (e.g., Faeth et al. 2011). As the degree of urbanization increases, native biodiversity usually decreases (McKinney 2008; Faeth et al. 2011; Hamer and McDonnell 2009; Hamer and McDonnell 2008).

Trends in Urban Biodiversity

Within urban environments there is a dramatically lower level of terrestrial faunal biodiversity, measured as species richness, compared to surrounding areas (eg. Shochat et al. 2010), and this decline has been observed in birds (see Chace and Walsh 2006; McKinney 2008), arthropods (see Raupp et al 2010; Faeth et al. 2011), mammals (see Wenguan et al. 2008), and reptiles and amphibians (see Hamer and McDonnell 2008; Mitchell et al. 2008). Explanations for these observed declines in biodiversity have

included local level factors, such as changes to abiotic factors including temperature gradients, hydrography, and nutrient availability (see Shochat et al 2006), local habitat loss and fragmentation (McKinney 2008), and regional factors such as increased isolation, and decreased connectivity due to fragmentation (e.g., Faeth and Kane 1978, Leibold et al. 2004). The changes in local and regional factors (outlined below) have been shown to negatively impact urban biodiversity.

Local Scale Environmental Change – Abiotic Factors

Abiotic factors, such as hydrology, climate and nutrient availability, are often highly altered in urban areas. In urban environments, the hydrographic period and hydrography are altered mainly by the extensive use of impervious surfaces and channelized runoff (Walsh et al. 2005). Urban environments tend to have higher instances of flash floods and lower infiltration rates of rain water (Walsh et al. 2005). This leads to “flashier” hydroperiods and increased bank erosion in urban streams (Walsh et al. 2005). This change in hydrography, often referred to as the Urban Stream Syndrome, leads to increases in run-off and has negative effects on water quality in urban streams due to elevated concentrations of nutrients and pollutants and increased suspension of solids (Walsh et al. 2005).

In addition to changed hydrography, urban environments also have different temperature gradients compared to non-urban environments. One effect, known as the Urban Heat Island, has been observed in cities in various climates (Brazel et al. 2000; Partecke and Gwinner 2007). Urban areas, especially the core of the city, are generally warmer than their neighboring non-urban habitats (Brazel et al. 2000). This temperature

gradient is due to the three dimensional built-up structure of urban environments, high density of paved roadways, and the building materials used in cities (Parlow 2011). Urban areas tend to absorb infrared radiation during the day and reflect back that radiation during the evening, maintaining warmer temperatures, especially at night (Parlow 2011). The intensity of the Urban Heat Island effect depends on many factors such as where the city is located, the type of building materials, and the amount of greenspace (Kuttler 2008). Urban areas show higher average low temperatures than neighboring non-urban areas, and in most climatic zones, urban areas show higher average high temperatures than neighboring non-urban areas (Brazel et al. 2000). The urban heat island moderates winter temperatures and increases summer high temperatures for many cities worldwide, especially in temperate zones where many large cities are located (Shochat et al 2006).

Another abiotic factor, nutrient availability, differs in urban environments compared to non-urban environments. Urban environments usually have more available carbon, nitrogen, and phosphorous than non-urban areas (Lewis et al. 2006). Sources of these nutrients include lawn fertilization, emissions from cars and factories, waste water, and refuse (Robbins and Sharp 2003; Kaye et al. 2006). This increase in nutrients is typical in most urban environments and may have a homogenizing effect on urban biodiversity, favoring non-native species over native species and r-selected, highly mobile, generalists over k-selected, stationary, specialists (McKinney 2006).

Local Scale Processes - Habitat Fragmentation

Land use conversion and road building activities in urban environments lead to habitat fragmentation that characterizes urban, suburban and exurban areas. Both of these activities have effects on patch connectivity (see Regional Processes below) as well as effects on patch size and quality. Patch size is directly correlated with biodiversity and has been examined using species-area relationships (SPAR) elsewhere (see Rosenzweig 1995). As urbanization increases, habitat patches shrink or disappear. SPAR predicts that smaller habitat patches harbor less biodiversity than larger patches as the number of niches decrease and “core” species become locally extinct (Rosenzweig 2003; Marzluff 2005). Habitat loss within cities may be exacerbated because cities are often built in areas that have high initial levels of biodiversity (Kuhn et al 2004). Habitat fragmentation can also reduce dispersal (see Regional Processes below) In the case of organisms with naturally low dispersal capabilities, such as amphibians, habitat fragmentation can remove entire metapopulations (Pope et al. 2000). Amphibious organisms require both aquatic and terrestrial habitats. Thus removing or reduction of either could have the same effect on presence of species as removing both (Ficetola and De Bernardi 2004; Scheffers and Paszkowski 2012).

Local Processes – Changes in Disturbance Regimen

Each ecosystem has a natural disturbance regimen which can include regional scale catastrophic disturbances, such as hurricanes or large wild fires, and local scale disturbances, such as a fallen tree. To explain how disturbance and biodiversity are related, Connell (1978) developed the Intermediate Disturbance Hypothesis (IDH). The

IDH states that biodiversity will be highest at intermediate levels of disturbance in both frequency (how often) and severity (magnitude of the disturbance) (Connell 1978). At low levels of disturbance, species interactions, such as competition, become more intense and cause local extinction, thus reducing diversity. At high levels of disturbance, populations cannot become established or do not persist due to severe and frequent disruptions (Connell 1978).

Although originally developed for explaining diversity in natural systems, IDH has also been used to explain biodiversity in human dominated landscapes, such as cities (McKinney 2008). Urban environments may lack some type of natural disturbances, such as wildfires, and have instead human-caused disturbances. Urban disturbance is mainly due to land use changes and can include construction of buildings, creation of athletic parks, road paving, stream diversion, and land grading (McKinney 2008). Additionally, urban disturbance is often permanent and non-reversible, leading to less succession within the urban environment.

Beyond the types of disturbance, urban disturbance regimens are often different than their natural counterparts as they are human mediated and controlled. Individuals, institutions, and governments also control patch destruction and creation by deciding when and where new construction occurs (Imhoff et al. 2004). Additionally, when a disturbed habitat is covered in impervious surface (e.g., asphalt, roads, buildings) the habitat is largely removed for most organisms and biological succession is indefinitely suspended.

Urban bird and arthropod communities have been shown to follow IDH predictions along the urban-rural gradient with species diversity peaking at intermediate levels of urbanization (i.e., suburban areas) (Blair and Launer 1997; McKinney 2008; Shochat et al. 2010). These communities show low levels of diversity in urban cores, where communities are dominated by synanthropic species (species that are associated with humans and are often exotic), and low levels of diversity in wild areas, where communities are dominated by native species (Blair 1996; Blair and Launer 1997). However, in areas of moderate urbanization, both the native species and synanthropic species are able to persist due to the increased niches and additional resources in moderately urbanized habitats (Shochat et al. 2010).

Local Processes – Changes in Net Primary Productivity

Urban areas have higher overall nutrient availability than wild areas due to inputs from fertilization and emissions related to fossil fuel combustion. Productivity and biodiversity are related in a similar manner as disturbance and biodiversity, with low levels of biodiversity occurring at low and high levels of productivity, and high levels of biodiversity occurring at intermediate levels of productivity, albeit due to different mechanisms (Shochat et al. 2006). Biodiversity changes due to IDH are driven by niche availability (Connell 1978), while the changes in biodiversity due to productivity levels are driven by competitive exclusion and resource exploitation (Faeth et al 2011). Gregg et al. (2003) examined net primary production in urban areas in New York City versus nearby non-urban areas, and found that urban areas have an overall higher level of net primary production compared to non-urban areas. An urban patch that is composed of

solely impervious surface, (e.g., a parking lot, street, or building) has near zero net primary production, while a fertilized and watered lawn may have very high net primary productivity (Shochat et al. 2006). These extremes of net primary productivity create heterogeneity and patchiness in productivity that may result in uneven distribution of individuals and biodiversity that characterizes urban environments (Hope et al. 2003).

Regional Scale Processes – Island Biogeography Theory

Whereas local scale processes explain biodiversity changes by using patch quality, abiotic factors and species interactions, regional (landscape) scale processes explain biodiversity changes by using dispersal and patch connectivity. One of the most recognizable theories used to explain biodiversity changes at a regional scale is island biogeography theory (IBT) proposed by MacArthur and Wilson (1967). This theory assumes some impediment to dispersal (e.g., ocean, impervious cover) between islands (patches) and states that biodiversity of an island is a function of the colonization rate, determined by connectedness (distance to source of species pool), of the island and the extinction rate, determined by the size of the island (MacArthur and Wilson 1967).

As a result of fragmentation, cities are a patchwork of differing land-use types and habitats, ranging from urban cores, where impervious cover is high and net primary productivity and species richness is low (Gregg et al 2003), to remnant patches of natural habitats, such as urban parks, where productivity and biodiversity are relatively high. Impervious cover (e.g., roads, buildings, parking lots) that separate habitat patches often acts as an impediment to dispersal and may have similar effects on dispersal and connectivity as do oceans. Marzluff (2005) proposed that island biogeography theory

could be used to explain the changes in biodiversity along the urban gradients using colonization and extinction rates. As urbanization increased, native bird species went locally extinct, lowering the biodiversity of native bird species in urbanized areas (Marzluff 2005). Other studies have examined the effects of patch size in urban environments on biodiversity and have found that patch size is positively correlated with biodiversity and that urban patch biodiversity can be modeled similarly to oceanic islands (Faeth and Kane 1978; Donnelly and Marzluff 2006;).

Regional Scale Processes – Metacommunity dynamics

An expansion of the island biogeography theory is the concept of metacommunities, or a population of communities. A metacommunity is commonly defined as “as a set of local communities that are linked by dispersal of multiple potentially interacting species” (Leibold et al 2004). Metacommunity theory describes processes that occur at the metacommunity scale (regional or landscape scale), the same scale as island biogeography (Leibold et al. 2004). Whereas both island biogeography and metacommunity theory are designed to explain biodiversity trends at the regional scale and both examine components of dispersal and connectedness, island biogeography theory assumes a fixed mainland pool of species from whence all colonization originates. In contrast, metacommunity theory assumes migration among the cluster of isolated communities, allowing for local extinctions but regional persistence of a species (Leibold et al. 2004). Thus, dispersal and the factors that affect dispersal become the main focus of metacommunity models.

In the case of a community that has no dispersal or colonization, basic models of species interactions, including Lotka-Volterra models (Murdoch and Oaten 1975; Abrams and Walters 1996) are appropriate to explain biodiversity. However, most communities have some level of dispersal and colonization between patches and communities (Leibold et al. 2004). For communities with low dispersal rates, regional level dispersal involving colonization dictates diversity as in IBT (Leibold et al. 2004). Colonization effects regulate the species mix of individual patches and can affect the assembly history of local communities (MacArthur and Wilson 1967; Drake 1991; Leibold et al. 2004). If dispersal rates are high, other effects, such as mass and rescue effects can alter species interactions and abundances and determine diversity (Leibold et al. 2004). Mass effects occur when species colonize patches and are unable to maintain a population within the patch, thus relying on constant influx from an adjacent patch to persist (Shmida and Wilson 1985). One example of mass effects is sink-source dynamics. Rescue effects are similar in that an influx of colonists from other patches prevents species from becoming extinct within a given patch (Brown and Kodric-Brown 1977). Both of these effects are dependent on the connectedness of patches which is often measured as distance from patch to patch. Metacommunity theory, like island biogeography, takes into account dispersal and colonization but also acknowledges the importance of patch quality and local scale effects that can lead to successful colonization.

Metacommunity models can be used to explain trends in biodiversity in urban areas. Parris (2006) used metacommunity models (as described by Leibold et al. 2004) to explain biodiversity changes in anuran (frog and toad) communities in Melbourne,

Australia. She found that pond size, isolation (measured as surrounding road cover), and presence of a vertical pond wall had the greatest effect on biodiversity (Parris 2006). Her results were consistent with the species-sorting perspective (a metacommunity perspective which emphasizes spatial niche separation over spatial dynamics and emphasizes both patch quality and dispersal), and the mass effect perspective (a metacommunity perspective which focuses on the effects of immigration and emigration, emphasizing spatial dynamics), two of the four metacommunity perspectives outlined by Leibold et al. (2004) (Parris 2006). Additionally, her results show that patch quality (as measured by physical and chemical characteristics) and isolation both play a role affecting biodiversity in urban environments. Parris (2006) argued that metacommunity theory should be considered for future studies of urban anuran communities.

Urban Biodiversity Summary

Urban environments are highly fragmented and patch quality and size are notoriously heterogeneous. The patches themselves are highly controlled by human inputs, either direct (application of fertilizing, choice of plants), or indirect (fertilization via airborne emissions, increasing stream flashiness) and are connected through remnant riparian zones, parks, median strips, roadside green spaces, and direct human selection/movement (Swan et al. 2011). Generally, native flora and fauna lose biodiversity as urban areas expand, with the total number of species declining as urbanization increases (Shochat et al. 2010; Shochat et al. 2006; Marzluff 2005; Faeth et al. 2011; McKinney 2008; McKinney 2006). In the case of anurans, Parris et al. (2006) argued that a model including local (patch quality) and regional (connectivity) factors

best explains the patterns of anuran biodiversity in urban areas. Previous work in Italy has also shown that both local scale and regional scale processes (including urbanization) should be modeled together to explain anuran biodiversity trends (Ficetola and De Bernardi 2004). More recent studies also show that a model that includes both local and regional scale processes is most effective at explaining anuran biodiversity changes in urban areas (Birx-Raybuck et al. 2009; Hamer and McDonnell 2008).

Anuran Biodiversity Declines

Worldwide anuran biodiversity has been declining for several decades (Burrowes et al. 2004; Alford and Richards 1999; Pounds and Crump 1994). There are a number of causes for these declines, including habitat destruction and degradation, introduced competitors and predators, climate change, and infectious disease (Alford and Richards 1999; Hamer and McDonnell 2008). Habitat loss has been shown to directly cause anuran biodiversity decline through destruction of temporal pools (breeding habitat) and upland forest (adult refugia habitat) (Hecner and M'Closkey 1997). Habitat degradation, typically the draining or polluting (such as increases in toxicants and acidity) of breeding sites, lowers the opportunities for recruitment (drainage) and the probability of successful recruitment (pollution) (Alford and Richards 1999; Brand et al. 2010).

The introduced and invasive, *Lithobates catesbianna* (North American bullfrog) has been implicated in the decline of anuran species outside of its natural range in several locations worldwide (Alford and Richards 1999). Additionally, introduced predatory fish species have been shown to reduce anuran reproduction through the consumption of tadpoles before metamorphosis has occurred (Fisher and Shaffer 1996).

Climate change has a negative impact on anuran biodiversity (Pounds and Crump 1994; Alford and Richards 1999; Lips et al. 2008) and has been associated with the decline and possible extinction of several species in the Monteverde region of Costa Rica (Pounds and Crump 2004). The observed decline has been linked to increases in temperature and decreases in available moisture, two important abiotic factors that impact anuran breeding success and adult survival (Lips et al. 2008).

The major infectious disease implicated in anuran decline is chytridiomycosis (chytrid), which is caused by the fungal pathogen *Batrachochytrium dendrobatidis* (Lips et al. 2008). Chytrid is a highly invasive disease that causes adult mortality in anurans and has been found in tropical areas of South America, Australia, and within temperate regions of the United States (Lips et al. 2008).

Anurans have been widely used as an indicator group to indicate environmental quality for other vertebrates (Lawler et al. 2003). The use of anurans as indicators is due to their complex life cycle, requiring both terrestrial and aquatic habitat, and their sensitivity to pollutants (Vitt et al. 1990). This can lead to the “canary in the coal mine” effect where anuran presence and biodiversity can indicate the relative health of an environment for other species. and applies to urban as well as non-urban environments (Lawler et al. 2003; Scheffers and Paszkowski 2011). Urbanization causes extensive and intensive environmental changes (e.g., Faeth et al. 2011; Grimm et al. 2008) at both local and regional scales, which often leads to declines in anuran biodiversity in urban settings.

Anuran Declines in Urban Areas

The biodiversity of anurans generally decreases with increases in urbanization (Hamer and McDonnell 2008; Knutson et al 2000; Dodd and Smith 2003; Scheffers and Paszkowski 2012). Hamer and McDonnell (2009) found that urbanization was correlated with a 7% decrease in total anuran biodiversity in Melbourne, Australia. Scheffers and Paszkowski (2012) reviewed 24 North American anuran studies, which examined 144 total anuran responses to urbanization, and found that many anurans had negative responses to increasing urbanization (31%), few had positive responses to increasing urbanization (4%), and others had either a neutral response (17%) or an unknown response (48%) to increases in urbanization. However, the causes of these declines were not been identified in the study.

Local Effects on Anuran Declines

The most commonly proposed cause for these declines are local processes related to habitat loss and degradation (Hamer and McDonnell 2008; Ostergaard et al 2008). Anurans are impacted by alterations to hydrography, pollutants, temperature, and habitat fragmentation (Brand et al 2010; Lips et al. 2008). Whereas most organisms suffer some loss in biodiversity due to habitat loss, anurans are uniquely impacted compared to other terrestrial animals. The dual lifestyle of most anurans means that anurans require two different habitats, terrestrial and aquatic, and the quantity and quality of both impact anuran biodiversity (Hamer and McDonnell 2008). First, adults require suitable terrestrial habitat during the non-breeding season for survival and then dispersal to aquatic habitats during the breeding season (Semlitsch and Bodie 2003). Second, most

anurans require aquatic habitats for breeding and larval survival. Aquatic habitats are often located near upland forest habitats (the preferred terrestrial habitat for many species) and can include streams, lakes, and seasonal pools (Hamer and McDonnell 2008). Anurans use aquatic habitats for reproduction, which includes calling, amplexus (copulation), and oviposition. Upon hatching, tadpoles are aquatically bound due to having gills and being legless. Tadpoles will remain in the aquatic habitat for a few weeks to several years, depending on the species (Dorcas and Gibbons 2008). The aquatic habitat's quality is very important for anurans, as tadpoles have long been considered organisms that are sensitive to pollution (Phillips 1990). The declining availability and quality of both terrestrial and aquatic habitats as well as disease (including chytrid), increases in temperature (due to the Urban Heat Island effect), and noise pollution that affects mating calls (e.g. Kaiser and Hammers 2009), have all contributed to the decline in anuran biodiversity in urban areas (Hamer and McDonnell 2008).

At the regional level, the connectivity of, and dispersal among, terrestrial and aquatic patches is critical in determining anuran biodiversity. This connectivity is often disrupted in urban environments due to construction of buildings and roads, and alterations of streams. Even non-urban roads can have negative effects on dispersal as roads are often implicated in direct mortality of adult anurans (van der Ree et al 2011). As the connectivity between patches decreases, the persistence of a species within an area decreases as there is less likelihood for rescue effects to occur following a local extinction event (Parris 2006; Leibold et al 2004). This isolation of not only adult upland

forest but also aquatic habitat used for breeding, can lead to declines in anuran biodiversity within the highly fragmented urban environment.

Correlative studies suggest that anuran biodiversity changes are due to both local and regional processes (see Ficetola and De Bernardi 2004; Parris 2006; Birx-Raybuck et al 2009; Barrett and Guyer 2008; Gagné and Fahrig 2007; Scheffers and Paszkowski 2012). However, testing the various hypotheses (patch quality, island biogeography, metacommunity theory) is difficult, especially in urban environments. Many factors may affect patch quality, including the presence of aquatic and terrestrial habitat, the levels of pollution, wetland depth, wetland surface area, the slope of the bank of the wetland, sun exposure, surrounding vegetation, aquatic vegetation, age of a wetland, and the presence of fish (Birx-Raybuck et al 2009; Ficetola and De Bernardi 2004). Some of these factors likely interact with each other. For example, sun exposure and depth affect vegetation and the presence of vegetation may affect the presence of fish.

On a regional scale, connectivity of patches that influences dispersal is also important to anuran biodiversity (Parris 2006; Ficetola and De Bernardi 2004). However, measuring connectedness may be complicated, especially in an urban environment. Some studies measure connectedness as a function of distance from one location to another, with little attention given to the physical make-up of the landscape (Ficetola and De Bernardi 2004). Alternatively, Brand and Snodgrass (2010) focused on the land use (roads, buildings, agriculture) without examining distance from patch to patch. Parris (2006) used land-use as an indirect measure of connectivity (more impervious cover is assumed to reflect less connectivity). However, the latter method does not consider traffic

volume which likely affects connectivity and thus dispersal (Parris 2006). Currently, there is not a standard practice for measuring connectivity for anurans in an urban environment.

Anthropogenic Ponds

Anthropogenic ponds associated with roadways and urbanization may positively affect anuran diversity by providing required habitat for survival and breeding. These ponds are primarily built for retention, detention, and erosion control with little consideration of possible ecological benefits or detriments (Brand and Snodgrass 2010). However, Battin (2004) hypothesized that anthropogenic ponds could negatively affect biodiversity by serving as ecological traps for breeding amphibians. Ecological traps are poor-quality habitats that are chosen by an organism over better quality habitats, leading to decreased fitness and local extinction (Gilroy and Sutherland 2007).

The ecological trap hypothesis, proposed by Battin (2004), was tested by Brand and Snodgrass (2010). They found a general decline in anuran biodiversity along an urban gradient as urbanization increased. But Brand and Snodgrass (2010) also found that anthropogenic ponds had a higher level of anuran biodiversity when compared to naturally formed ponds with the same level of urbanization, thus challenging Battin's (2004) hypothesis that such ponds were ecological traps. These results suggest that man-made stormwater controls (i.e., detention and retention ponds) may mitigate anuran biodiversity loss caused by urbanization.

Preformed Scour Holes

The most common type of stormwater control used in studies of urban anuran diversity is the retention pond (Birx-Raybuck et al. 2009; Brand and Snodgrass 2010; Ostergaard et al. 2008). While conspicuous and fairly common, retention ponds are but one of many types of stormwater control use by cities, counties and states. . The North Carolina Department of Transportation (NCDOT) has begun to use a new stormwater control, the preformed scour hole (PSH), to minimize erosion caused by roadside scour. The PSH is a human dug depression lined with riprap, designed to dissipate energy from roadway point discharge and provide a stable impact point for peak flow (NCDOT 2008). PSHs are pre-shaped basins that are located downhill from a stormwater outflow with permanent soil reinforcement matting located on the downhill side of the PSH to prevent erosion downhill from the PSH (NCDOT 2008). The primary purpose of the PSH is to limit erosion, however the secondary purpose is to promote runoff infiltration (NCDOT 2008). This secondary purpose, promoting runoff infiltration, allows stormwater to gather and form temporary pools which may have biological implications for amphibious organisms.

PSHs are an ideal stormwater control to study for their effects on biodiversity because they are associated with new road construction and are found along an urban to rural gradient. Additionally, the effects of PSHs on biodiversity, including anuran biodiversity, have not been previously examined.

Regional Anuran Species

North Carolina is home to 29 species of anuran from seven genera and five families (Dorcas and Gibbons 2008). The Piedmont area surrounding Greensboro, NC has 12 known species from six genera and four families (Dorcas and Gibbons 2008). The species pool for Greensboro, NC region consists of *Anaxyrus americanus* (American Toad), *Anaxyrus fowleri* (Fowler's Toad), *Acris crepitans* (Northern Cricket Frog), *Hyla chrysoscelis* (Cope's Gray Tree Frog), *Hyla versicolor* (Common Gray Tree Frog), *Gastrophryne carolinensis* (Eastern Narrowmouth Toad), *Pseudacris feriarum* (Upland Chorus Frog), *Pseudacris crucifer* (Spring Peeper), *Lithobates sphenoccephala* (Southern Leopard Frog), *Lithobates palustris* (Pickeral Frog), *Lithobates clamitans* (Green Frog), and *Lithobates catesbeiana* (Bullfrog). For more information about how the species pool for Greensboro, NC was calculated see Chapter II. All 12 species are carnivorous as adults, generally insectivorous, but will eat smaller anurans. All are herbivorous as tadpoles (Dorcas and Gibbons 2008). Each of these species has specific habitat preferences and specific breeding period, outlined below.

Both species of true toad (*A. americanus* and *A. fowleri*) share life history characteristics and have been known to hybridize (Green and Parent 2003). Both toad species adults are large, over 5 cm in snout to vent length (SVL), have highly variable colorations and dry skin with warts along the back as well as 2 parotid glands, which contain toxin, located behind each eye (Dorcas and Gibbons 2008). *A. americanus* breeds in late-winter to mid-spring (late February to early April) while *A. fowleri* breeds from mid-spring to early-summer (early April to early July) (Dorcas and Gibbons 2008).

Both species prefer to breed in ephemeral pools and generally take 2 months for metamorphosis to occur (Dorcas and Gibbons 2008). Additionally, the toads prefer upland forest for non-breeding habitat and are generally only found near water during the breeding season (Dorcas and Gibbons 2008).

A. crepitans is the only member of the *Acris* (cricket frog) genus found in and around Greensboro, NC. *A. crepitans* is small, usually under 2.5 cm SVL, highly variable coloration and pattern, with a distinctive dark triangle found between the eyes (Dorcas and Gibbons 2008). *A. crepitans* is an opportunistic breeder and will breed from early spring to late summer (late March to late August) in permanent bogs or ephemeral pools and take between 1 and 3 months to complete metamorphosis (Dorcas and Gibbons 2008). Adult *A. crepitans* prefer moist habitats, including permanent bogs but can also be found in low-lying forests that are near a permanent water source, with plenty of ground cover including low-lying plants and downed trees (Dorcas and Gibbons 2008).

The two members of the *Hyla* (tree frog) genus (*H. chrysoscelis* and *H. versicolor*) have overlapping ranges but are unable to hybridize as *H. chrysoscelis* is diploid and *H. versicolor* is tetraploid. Nonetheless, they share identical life-history traits and only differ based on range, call, and chromosome number (*H. versicolor* is tetraploid and *H. chrysoscelis* is diploid) (amphibiaweb.org). *H. versicolor* is a northern species and is only found in NC in disjunct populations in Caswell and Warren counties, while *H. chrysoscelis* is a southern species and is found throughout the state of NC (Dorcas and Gibbons 2008). Both tree frogs are large, around 5 cm SVL, highly variable with coloration and pattern, have large toe pads, and a distinct yellow-orange patch on the

inside of both thighs (Dorcas and Gibbons 2008). Both species breed from early-spring to mid-summer (late March to mid-July), prefer to breed in ephemeral pools, and generally take 2 months to complete metamorphosis (Dorcas and Gibbons 2008). Adult tree frogs prefer upland forest habitat and generally only come down from the treetops for breeding purposes, preferring to spend time in a more arboreal habitat. However, *H. chrysoscelis* is often found within urban areas and will use human-made structures as well as trees for non-breeding habitat (Dorcas and Gibbons 2008).

G. carolinensis is the only narrowmouth toad in the southeast United States, as well as the only member of the family Microhylidae in North Carolina (Dorcas and Gibbons 2008). *G. carolinensis* is small, 2.5 cm SVL, comes in a variety of colors, including red, black, brown, and gray, with various patterns, and has a distinct rounded body with a pointed nose and a fold of skin behind the eyes (Dorcas and Gibbons 2008). *G. carolinensis* breed from late-spring to early-fall (early May to mid-September), prefer to breed in ephemeral pools, and generally take 3 to 10 weeks to complete metamorphosis (Dorcas and Gibbons 2008). Adults generally spend time in underground burrows or beneath leaf litter found along forest floors (Dorcas and Gibbons 2008).

P. feriarum and *P. crucifer* are both members of the genus *Pseudacris* (chorus frogs) and have similar life history. Both are small, 2.5 cm SVL, and *P. feriarum* is highly variable in coloration and pattern, but has a distinctive light stripe above the upper lip as well as dark coloration around the eye (Dorcas and Gibbons 2008). *P. crucifer* is generally tan or brown in coloration with a distinctive “X” shape along the back that stretches from each eye to the opposite hip (Dorcas and Gibbons 2008). *P. feriarum* and

P. crucifer breed from late-winter to early-spring (mid-February to early April), prefer to breed in ephemeral pools away from flood plain forests, and take between 10 weeks to 2 months to complete metamorphosis (Dorcas and Gibbons 2008). Adult *P. feriarum* spend most of their time amongst forest undergrowth, often hidden under leaf litter, while *P. crucifer* also spends time amongst leaf litter, *P. crucifer* is also known to overwinter in arboreal settings like the tree frogs (Dorcas and Gibbons 2008).

All 4 members of the *Lithobates* (true frogs) genus in Greensboro have similar habitat requirements and preferences, but have unique identifying characteristics. All four are large, and range in SVL length from 7.5 cm (*L. palustris* (pickerel frog), *L. sphenoccephala*, and *L. clamitans*) to 15 cm (*L. catesbianna*) (Dorcas and Gibbons 2008). *L. paulstris* (pickerel frog), *L. sphenoccephala* (leopard frog), and *L. clamitans* (green frog) all have a distinct dorsolateral fold running from eye to same hip, while *L. catesbianna* (bullfrog) lacks this characteristic (Dorcas and Gibbons 2008). The pickerel frog has two rows of large square spots that run the length of its back and a rounded snout, while the leopard frog has irregular spots along its back and a sharply pointed snout (Dorcas and Gibbons 2008). Both the green frog and bullfrog are highly variable in coloration but generally are some combination of green and brown. Green frogs generally have a mottled underside, and male green frogs have a spot in the tympanum (Dorcas and Gibbons 2008).

The leopard frog is an opportunistic breeder, but will generally breed from late fall to late spring (mid-November to mid-May) and takes 3 months to complete metamorphosis. The pickerel frog breeds from early spring to late spring (mid-March to

late May) and takes 2-3 months to complete metamorphosis (Dorcas and Gibbons 2008). The green frog breeds from early spring to late summer (mid-March to late August) and takes 3-6 months to complete metamorphosis (Dorcas and Gibbons 2008). The bullfrog breeds from early spring to late summer (mid-March to late August) and takes between three months and two years to complete metamorphosis (Dorcas and Gibbons 2008). All four *Lithobates* species prefer to breed in permanent bodies of water, and adults of all four species prefer to overwinter in and around permanent aquatic habitats.

Research Question and Hypothesis

This study examines anuran biodiversity found in Preformed Scour Holes (PSH) a specific North Carolina Department of Transportation (NCDOT) stormwater control. In addition the study ascertains whether local and regional-level factors associated with PSH's are correlated, with changes in anuran biodiversity. The aims of this study are 1) to assess anuran biodiversity in PSHs that vary in degree of urbanization and 2) to determine which local and regional level factors associated with PSHs correlate with anuran biodiversity in the north central piedmont of North Carolina. Based on previous studies (including Parris 2006 and Ficetola and De Bernardi 2004) and IBG and metacommunity theory, I hypothesize that PSH surface area and urbanization will affect anuran biodiversity. I predict surface area and urbanization will be positively and negatively correlated with anuran diversity, respectively. At the species level, I hypothesize that local and regional factors will vary on their effects on different species depending upon the species life histories.

CHAPTER II

MATERIALS AND METHODS

Study Sites

Partnering with the North Carolina Department of Transportation (NCDOT), I was given access to Preformed Scour Hole (PSH) erosion control sites throughout central North Carolina. A Preformed Scour Hole is a “structural stormwater control designed to dissipate energy and promote diffuse flow” (NCDOT 2008). Each PSH is pre-shaped, stabilized with filter fabric, and lined with rip-rap (NCDOT 2008). Rip-rap consists of medium sized stones around 20 cm in diameter. The PSH mimics natural scour holes that prevent road run-off erosion from point discharges. The intended water quality benefits of a PSH are to “reduce the amount of end-of-pipe erosion by eliminating unabated scour” and “promote runoff infiltration and reduce downgrade erosion” (NCDOT 2008). The NCDOT has not conducted any studies of the potential benefits to biodiversity of arthropods, amphibians, or any other animals or semi-aquatic/aquatic plants.

Greensboro was considered the urban center for this research. All of the PSH in Guilford, Alamance, Randolph, and Caswell counties were prescreened to determine whether they might be used for amphibian breeding. The criterion for prescreening was the ability for a site to hold water for at least two months during the predominant anuran breeding season (February to June). After being initially screened in February 2012, each site was re-examined in early May 2012 to test whether the original criterion was

sufficient. Of the 54 PSH found in the study area, 21 were found to hold water for longer than two months (Figure 1).

Study Organisms

In the study area, there are 12 anuran species, each with a unique and distinct call. Anurans typically call only during the breeding season. Thus, it can be assumed that calling activity is an indication of reproductive activity and not simply anuran migration. The 12 species of anuran found with the study area include: *Anaxyrus americanus*, *Anaxyrus fowleri*, *Acris crepitans*, *Hyla chrysoscelis*, *Hyla versicolor*, *Gastrophryne carolinensis*, *Pseudacris feriarum*, *Pseudacris crucifer*, *Lithobates sphenoccephala*, *Lithobates palustris*, *Lithobates clamitans*, and *Lithobates catesbeiana*.

To determine the likely pool of anuran species in the area surrounding each PSH, I used calling data collected by Davidson College (Davidson HerpLab 2012; Price and Dorcas 2011), collection data from the North Carolina Museum of Natural Sciences (Beene 2012), and the herpsfnc.org (Dorcas 1999) website that is managed by Davidson College. A species was considered as part of the potential species pool for the site if it was found within a county in 2 of the 3 data sets. See Table 1 for each county's species mix. The field season coincided with the breeding season for all species.

Species Presence

Each species of anuran found within the study area has a unique and distinct call. There are no auditory cryptic species. All species were identified in situ and recorded and re-listened to for confirmation using the database created and managed by Davidson Herpetology (Dorcas 1999). For recording purposes, the Sony ICD-PX312 (digital flash

recorder) was used in conjunction with the Audio-Technica ATR-3350 Lavalier Omnidirectional Condenser Microphone. All recordings have been stored on my personal computer and are available upon request. Beyond auditory identification, visual inspections confirmed species presence.

Each site was between 3.5m² and 51.7 m² and could be easily circled to ensure any calling that was detected was from the site and not from the surrounding area. Only individuals of species that were calling from the site were considered present. If an individual (or individuals) of a species was detected then the species was considered present. This type of auditory survey based on presence/absence identification is known as Manual Calling Surveys (MCS) and has been identified as a very efficient method to gather data from many sites within a short time frame (Wright and Wright 1949). The MCS method has been outlined in Marsh and Trenham (2008) and used in other anuran studies of diversity including Todd et al (2003), Ficetola and De Bernardi (2004) Gooch et al. (2006), Kirlin et al (2006), and Brand and Snodgrass (2010).

For this study, a modified MCS protocol was used. Researchers approached each site and placed themselves between 6 and 10 meters from the edge of each PSH. For recording purposes researchers used the same location at each site visit to help standardized recordings. Once researchers were in place, a 3 minute silent blackout period began in order to allow anurans a chance to acclimate themselves to the presence of the researchers. After 3 minutes, recording began for 3 minutes. After recording, researchers performed a visual inspection of each PSH with head lamps and hand-held flashlights. After collection, the recordings were stored on the researcher's computer. To

confirm species' presence and gain a measure of abundance, the researcher re-listened to each recording, identified the species, and assigned a relative abundance classification using the MCS abundance 1,2,3 classification system in which 1 refers to a single individual calling, 2 is multiple discernible individuals, and 3 denotes multiple indistinguishable individuals often referred to as a chorus (Dorcas et al 2009). Additionally, if a species was observed visually but not recorded calling, that species was given an MCS number of 1. For the purposes of analysis, each species was given a 1, 2, or 3 at each site based on its highest MCS abundance number.

Data collection was conducted from late February 2012 to late June 2012. Each site was visited once every two weeks from March 1, 2012 to April 24, 2012, and each site was visited once a week from April 30, 2012 to June 28, 2012, for a total of 12 site visits for each site throughout the breeding period of 2012. Due to the frequency of site visits it is unlikely that a species went undetected at a given site, as such detectability is not found within any of the models this study generated.

Explanatory Factors

I examined local and regional (or landscape) factors that may influence anuran biodiversity. Ficetola and De Bernardi (2004) performed a similar study that examined environmental (local level) factors and isolation or dispersal (regional level) factors. Using this previous study as a guide, I measured similar local level and regional level factors that may correlate with anuran biodiversity.

The local level factors that were measured included: presence or absence of 1) human-made additional wetland at the PSH, 2) submerged non-aquatic vegetation

(including detritus), 3) floating non-algal vegetation, 4) riparian vegetation (common riparian species such as cattails), 5) algal bloom (if a bloom occurred during the sampling period then algal blooms were considered present), 6) soil in PSH (categorized as no soil if the bottom of PSH was rip-rap or stone), and 6) fish (considered present if a fish was found in a PSH at any time during sampling period).

Other local level factors that were included were: 1) surrounding vegetation and 2) shade percentage. Surrounding vegetation was placed into one of three categories, grass (no woody vegetation surrounding PSH), scrub (young growth woody vegetation and grass are present surrounding PSH), and forest (old growth woody vegetation and no grass surround PSH). Shade percentage was placed into one of 4 categories (0-25%, 25-50%, 50-75%, and 75-100%) based on shade coverage within 1 hour before and after the solar noon on a sunny day in May.

Additional continuous local level factors measured included: 1) PSH surface area, 2) depth at center of PSH, and 3) angle of the slope of the bank of PSH. PSH area was measured as the surface area of standing water less than a week after a rain storm in March. This area also includes the surface area of additional human-made wetlands to which a PSH drains into. Depth at center was measured at the same time as surface PSH area. Using depth and radius measurements, it was possible to determine the angle of the slope of the bank of a PSH.

Regional or landscape level factors include: 1) an estimate of the degree of urbanization, 2) distance from a riparian zone, 3) distance from road, 4) distance to nearest upland forest patch, and 5) distance from nearest PSH (to control for spatial

autocorrelation). Upland forest patches were defined as any patch of canopy producing trees that covered a minimum of 450 m². Riparian zones were defined as areas surrounding permanent flowing or standing water. This includes, but is not limited to, streams, rivers, and lakes. Distance from each PSH to road, forest patch, riparian zone, and next PSH was determined using the ruler function on Google Earth and the most recent satellite image.

Urbanization was estimated at two scales using the following technique. Using the most recent data on impervious land cover from the National Land Cover Database (USGS 2006), I measured the percent of impervious cover surrounding each site at radii of 100 and 1000 meters. Percent of impervious cover was measured as the number of 30m x 30m pixels within each circle that were covered in impervious cover divided by the total number of pixels in each circle. Pixel size used was the smallest pixel size available using the National Land Cover Database (USGS 2006). Impervious cover was chosen as the estimate of urbanization to ensure consistency for each PSH. Impervious cover increases with urbanization (Pauleit and Breuste 2011), and has been used as an indicator of urbanization (Pauleit and Breuste 2011) and allows for a quantitative comparison of sites. Urbanization was measured using ArcMap.

Statistical Analysis: Biodiversity

To determine which of the local and regional factors were associated with species diversity, a step-wise linear regression model was used (R version 2.15.1). PSH surface area, distance from riparian zone, distance from upland forest patch, distance from road, and distance from nearest PSH were log-transformed to ensure normalcy. Using AIC

criteria, a model was created using forward/backward stepwise linear regression. The original fitted model was created without including either measure of urbanization and only examined the local and regional factors. Both scales of urbanization were added back into the fitted model independently and together and the resulting models were compared.

Statistical Analysis: Species Specific Analysis

Of the ten species observed, only 4 were viable candidates to perform species specific analyses. Eight species were not included for the following reasons. *H. chrysoseelis* was found at all of the sites while *A. fowleri* and *L. sphenoccephala* were not found at any sites, *L. palustris*, *H. versicolor*, and *Acris crepitans* were found at only one site each, and *G. carolinensis* and *A. americanus* were only found at 2 sites. The 4 remaining species, *P. feriarum*, *P. crucifer*, *L. clamitans*, and *L. catesbianna*, were examined using a binomial logit model and forward stepwise regression to determine a model of best fit. The criterion used to build the species-specific model was a p-value less than 0.2. Once the initial model was built, forward/backward stepwise regression using AIC criterion was performed on the initial model to find the model of best fit.

Additionally, I examined whether the presence or absence of one species was affected by the presence or absence of another species. For this analysis, a two-way contingency table was created using p-values from Fisher's exact test. All statistical analysis was completed using the R statistical package version 2.15.1.

CHAPTER III

RESULTS

Explanatory Factors

Local and regional scale factors associated with each site are presented, respectively, in Table 2 and Table 3. These factors were used in the model building process outlined in the previous section.

Species Presence

The biodiversity of the 21 PSHs ranged from 1 to 6 species. The average number of species at a given site was 3 ± 1.10 SD species. Median number of species was also three. One species was detected at all sites (*Hyla chysoscelis*) while two species were not detected at any site (*Anaxryus fowleri* and *Lithobates sphenoccephala*) (Table 4). The number of sites where a species was found and the maximum MCS abundance number are found in Table 4. Additionally, a nested matrix of species presence can be found in Table 5.

Biodiversity Model

The best fit stepwise linear regression model to explain species richness included urbanization at a 100 meter radius ($r \pm SE, p\text{-value}$) ($-0.050 \pm 0.013, 0.001$), log (surface area) of each scour hole ($1.487 \pm 0.469, 0.005$), and the presence of riparian vegetation ($4.069 \pm 1.191, 0.003$). This model began with all available explanatory factors minus both urbanization levels and no interactions to determine which non-urban

factors affect biodiversity. Following the step-wise regression, models containing both urbanization levels individually and collectively were compared. Local scale urbanization (100 meter radius) alone provided the model of best fit (Table 5).

Pseudacris crucifer

The logistic model of best fit for the presence of *P. crucifer* contained (r +/- SE) urbanization at a 100 meter radius (-0.319 +/- 0.2129) and log (surface area) of each scour hole (14.509 +/- 9.397). Local scale urbanization (100 meter radius) is negatively correlated with *P. crucifer* presence but not significantly so (p-value = 0.134). Surface area of PSH is positively correlated with *P. crucifer* presence but not significantly so (p-value = 0.123) (Table 7).

Pseudacris feriarum

The logistic model of best fit for the presence of *P. feriarum* contained (r +/- SE) urbanization at a 1000 meter radius (-0.112 +/- 0.069), log (surface area) of each scour hole (2.166 +/- 1.215), and the presence of *L. clamitans* (-7.189 +/- 4.652). Regional scale urbanization (1000 meter radius) is negatively correlated with *P. feriarum* presence and is marginally significant (p-value = 0.1028). Surface area of PSH is positively correlated with *P. feriarum* presence and is marginally significant (p-value = 0.0745). The presence of *L. clamitans* is negatively correlated with *P. feriarum* presence but not significantly so (p-value = 0.1223) (Table 8).

Lithobates catesbianna

The logistic model of best fit for the presence of *L. catesbianna* contained (r +/- SE) the angle of the slope of the bank (0.578 +/- 0.319), the log (distance from a natural

riparian zone) (1.446 +/- 0.762), the presence of submerged non-aquatic vegetation (-3.201 +/- 1.971), urbanization at a 1000 meter radius (0.0874 +/- 0.0519), and the presence of an additional man-made wetland (6.495 +/- 3.088). The angle of the slope of the bank is positively correlated with *L. catesbianna* presence and is marginally significant (p-value = 0.0699). The distance from a natural riparian zone is positively correlated with *L. catesbianna* presence and is marginally significant (p-value = 0.0579). The presence of submerged non-aquatic vegetation is negatively correlated with the presence of *L. catesbianna* but is not significant (p-value = 0.1044). Regional scale urbanization (1000 meter radius) is positively correlated with *L. catesbianna* presence and is marginally significant (p-value = 0.0925). The presence of additional man-made wetland is positively correlated with the presence of *L. catesbianna* and is significant (p-value = 0.0354) (Table 9).

Lithobates clamitans

The logistic model of best fit for the presence of *L. clamitans* contained (r +/- SE) urbanization at a 100 meter radius (0.0827 +/- 0.0573), the log (distance from road) (-25.731 +/- 13.505), and the presence of soil in a PSH (23.745 +/- 12.327). Local scale urbanization (100 meter radius) is positively correlated with the presence of *L. clamitans* but is not significant (p-value = 0.1484). The distance from the road to the PSH is negatively correlated with the presence of *L. clamitans* and is marginally significant (p-value = 0.0568). The presence of soil in a PSH is positively correlated with the presence of *L. clamitans* and is marginally significant (p-value = 0.0541) (Table 10).

Fisher's Exact Test

The presence or absence of a species did not predict the presence or absence of another species. However, *Anaxyrus americanus* and *Gastrophryne carolinensis* are positively correlated with each other, although not significantly so (p-value = 0.1857) and the presence of *Lithobates clamitans* and *Pseudacris feriarum* are inversely correlated with each other, although only marginally significantly so (p-value = 0.0635).

CHAPTER IV

DISCUSSION

My results show that PSH associated with road building may provide habitat for anuran species and PSH can maintain regional biodiversity (Figure 2). These findings are consistent with other studies showing that stormwater controls can provide habitat for anurans in urban areas (Parris 2006; Bix-Raybuck et al. 2009). Parris (2006) found 10 species of anuran using stormwater controls for breeding in Melbourne, Australia, and Bix-Raybuck et al. (2009) found 5 species using stormwater controls for breeding in the western Piedmont of North Carolina, USA. Most PSH harbored more than one species during the breeding season (average number of species at each site was 3), indicating that PSH have features that are attractive to multiple species of anurans.

The suitability of PSHs as breeding habitat appeared to vary among anuran species. For example, *H. chrysozelis* was found at all sites (Figure 3). *H. chrysozelis* is a fairly common anuran species which is tolerant of many landscapes and environments (Brand and Snodgrass 2010; Brand et al. 2009). Seven of the other 11 species were found at multiple sites and only three species were observed at a single site. Of the probable regional pool of anuran species (Dorcas and Gibbons 2008), only *A. fowleri* and *L. sphenoccephala* were not observed. However the closely related species, *A. americanus* and *L. palustris*, respectively, were observed in some PSHs. It is possible that hybrids of *A. fowleri* and *A. americanus* were found and identified as *A. americanus*, because the

study area was located within the hybridization zone for these two species (Green and Parent 2003). Ecologically, there is little difference between *A. fowleri* and *A. americanus* as they share common life history traits and adult body size and fill the same ecological niche (Green and Parent 2003).

Factors Effecting Anuran Biodiversity – PSH Surface Area and Riparian Vegetation

My prediction concerning the relationship between PSH surface area and anuran biodiversity was confirmed. Overall anuran species diversity was positively related to PSH's area (Figure 4 and Figure 5). Additionally, I found that one other local scale factor, presence of riparian vegetation, was also positively correlated with anuran biodiversity (Figure 6). While this factor was not predicted to be correlated with anuran biodiversity, it is not unexpected as riparian vegetation has biological relevance for anurans. These results are consistent with other studies that show larger aquatic habitats with vegetation support higher diversity of anurans than smaller habitats with less vegetation (e.g., Hanski 1994; Ficetola and De Bernardi 2004; Parris 2006). Larger sites may support higher population sizes and thus reduce local extinction as predicted by Island Biogeography Theory (MacArthur and Wilson 1967). Larger and more vegetated sites also provide more structural and habitat complexity for breeding and feeding that support a wider diversity of species. Many adult anurans use riparian vegetation as oviposition sites, members of the *Hyla* and *Pseudacris* genera use vertical vegetation as calling sites to prevent detection from predators and to allow their call to carry over a larger area, and all the anurans in the study area are herbivorous until metamorphosis. Therefore, it is not surprising to find that riparian vegetation is positively correlated with

anuran biodiversity. However, larger sites may also carry more risk than a smaller site because they support more aquatic predators such as fish (Ficetola and De Bernardi 2004). PSHs, however, are separated from larger water bodies and are ephemeral pools, and therefore usually do not harbor fish.

Studies on other organisms have shown the importance of patch size on population persistence, specifically in regards to the Allee effect, or the positive effect conspecifics have on population growth (Stephens and Sutherland 1999). While anurans do not cooperatively breed, anurans do aggregate at breeding sites for calling and mate selection purposes, and larger breeding patch size would provide more individuals available for successful mating. In larger patches, there may be enough individual male and females to avoid the Allee effect (a reduction in population growth rate at low densities due to insufficient availability of mates) that may occur in small patches with few individuals (Stephens and Sutherland 1999). The Allee effect provides an explanation for why PSH size was found to be an important factor for anuran biodiversity. PSH surface area only measured the size of breeding habitat and did not measure the size of the overwintering habitat (upland forest or aquatic habitat) thus the Allee effect may be the mechanism through which PSHs may mitigate anuran population declines by providing breeding habitat throughout the urbanizing landscape.

Factors Effecting Anuran Biodiversity – Urbanization

Whereas anuran diversity was positively correlated with size and vegetation, diversity was negatively correlated with degree of urbanization at 100-meter radius as predicted (Figure 7). Urbanization at 100-meter radius was the only regional factor to

remain in the best fit regression model. However, urbanization at different scales (100 meter and 1000 meter radii) improved the model and both are highly correlated with one another. Urbanization was measured as percent of impervious surfaces and is thus likely an indirect measure of other regional factors such as reduced connectivity, via upland forest loss, and increased impediment to dispersal. Urbanization may also be associated with local factors such as increased mortality from air and water pollutants, altered climate (i.e., heat island effects), and reduced reproduction due to elevated noise or light pollution that interferes with mating (Kaiser and Hammers 2009). These results are consistent with previous research showing that urbanization generally has a negative effect on anuran biodiversity (Brand and Snodgrass 2010; Birx-Raybuck et al. 2009; Ficetola and De Bernardi 2004; Parris 2006; Scheffers and PDOG 2012). For example, Parris (2006) found that as road cover (or degree of urbanization) increased surrounding the stormwater control, anuran biodiversity decreased with a correlation coefficient of -7.38 (SD = 2.61) in Melbourne, Australia.

My results are similar to those found by Parris (2006) in that a model that includes two local factors and one regional factor best explains the trends in biodiversity. In fact, the model proposed by Parris (2006) included two of the same three factors found in my model of anuran biodiversity, surface area of stormwater control and degree of urbanization as measured by amount of impervious surface cover. However, the regional factor, urbanization, as measured here, likely also affects local factors. Because my study was correlational, the specific local and regional factors that affect anuran diversity associated with PSHs cannot be disentangled without further studies and controlled

experiments. Nonetheless, my results are consistent with predictions from metacommunity theory, specifically the species sorting perspective as outlined in Leibold et al. (2004). There was strong evidence that patch specific factors affected anuran biodiversity as well as strong evidence that connectivity, as measured by urbanization, affects anuran biodiversity. Thus, metacommunity theory may be a good framework to examine anuran biodiversity in human dominated environments because it incorporates local and regional processes (Ficetola and De Bernardi 2004; Parris 2006; Birx-Raybuck et al 2009).

Species Specific Effects

Urbanization, as estimated by amount of impervious surfaces, was a significant factor in all 4 species-specific models but at different scales. However, because the levels of urbanization (100-meter or 1000-meter radius) were highly correlated, the differential effect of scale on individual species cannot be distinguished. For *Pseudacris* species (*P. feriarum* and *P. crucifer*), urbanization was negatively associated with species presence, whereas the presence of *Lithobates* species (*L. catesbianna* and *L. clamitans*) were positively correlated with urbanization (Figure 8). Although these correlational effects are not statistically significant (Table 5-8), probably due to small sample size, the direction of the effects suggests that urbanization has genera-specific effects.

These genera-specific effects may be linked to life history and preferred overwintering habitats of the two genera. *L. catesbianna* is a synanthropic species (a species associated with humans and human activities) and has been positively correlated with anthropogenic change, as well as urbanization, in previous work (Delis et al. 1996).

Delis et al. (1996) found that *L. catesbianna* had increased abundance in urbanized areas, compared to non-urban areas, and hypothesized that this was due to a preference for permanent bodies of water which were more prevalent in urban areas in Tampa Palms, Florida. *L. clamitans* is similar in size, ecological role, and habitat requirements to *L. catesbianna*. Both species prefer to overwinter near permanent water and do not use upland forest as overwintering sites. Therefore, upland forest loss in urban environments should have diminished negative effect on these species compared to other anuran species (Delis et al. 1996). These results are consistent with research conducted on *Rana temporaria* and *Bufo bufo* in England. In the upland forest dwelling, *B. bufo* declined in urban areas while *R. temporaria*, which prefers permanent water bodies, did not (Carrier and Beebee 2003). Alternatively, *P. feriarum* and *P. crucifer* tend to overwinter in upland forest and use standing pools of water only for breeding. Because urbanization, as measured here, is associated with upland forest loss, it is not surprising that species that use upland forest habitat decrease as urbanization increases. These results suggest that species life histories may largely dictate their persistence in urban environments (Delis et al. 1996; Brand and Snodgrass 2010; Carrier and Beebee 2003).

Stormwater Controls and Ecological Reconciliation

My results combined with other studies (see Brand et al. 2010; Brand and Snodgrass 2010; Birx-Raybuck et al 2009; Parris 2006) suggest that stormwater controls can be effectively used as breeding sites for anurans in urban areas. Many anuran species are declining due to habitat loss, the creation of anthropogenic ponds and stormwater controls may mediate, and possibly halt, some of the loss in biodiversity due to

urbanization. The concept of ecological reconciliation is not new and was defined by Rosenzweig (2003) as “redesigning anthropogenic habitats so that their use is compatible with use by a broad array of other species”. Examples given by Rosenzweig (2003) include backyard wildlife habitats (where homeowners are encouraged to provide habitat for local wildlife species), long-leaf pine forests in Florida (managed for hunting, fishing, logging, and species preservation), and eastern blue bird nesting boxes (in which humans build nest boxes that exclude invasive species that negatively impact blue bird survival). However, the principles of ecological reconciliation are receiving more attention in urban planning from groups such as the NCDOT, as a way to minimize the impact of urbanization on native flora and fauna (Rosenzweig 2003). The use of stormwater controls could help to maintain moderate to high levels of native anuran biodiversity in urban areas and could maintain high levels of anuran biodiversity in non-urban areas (Brand and Snodgrass 2010).

My research also indicates that stormwater controls should be designed to be as large as possible and contain riparian vegetation to promote anuran use of stormwater controls for breeding. My results did not ascertain if there is a threshold size for stormwater controls, a size where biodiversity either increases or declines, as has been found for wetland areas in general (Ficetola and De Bernardi 2004). My results also suggest that species and genus-specific responses to urbanization need to be considered in designing and implementing measures to maintain anuran biodiversity in urban environments.

There are limitations and caveats for the results of this study. My study was observational and correlational thus causes that underlie patterns on anuran biodiversity cannot be ascertained without additional studies. This study encompassed only one field season and was limited to 21 sites. Thus caution is required in extrapolating to different urban environments, larger spatial scales and longer time frames. Also this study did not address fitness of anurans. Although it appears that PSH's can mediate anuran biodiversity loss, my study cannot exclude the possibility that PSH's act as ecological traps (e.g. Battin 2004). Future studies should thus examine fitness effects of stormwater controls such as PSH's. Additionally, future studies should test whether local or regional factors contribute more to anuran biodiversity loss. Lastly, it is clear that urbanization affects anuran species and genera in different ways, therefore, future studies of urbanization effects should focus on not only the community as a whole but also on at least genera and preferably individual species.

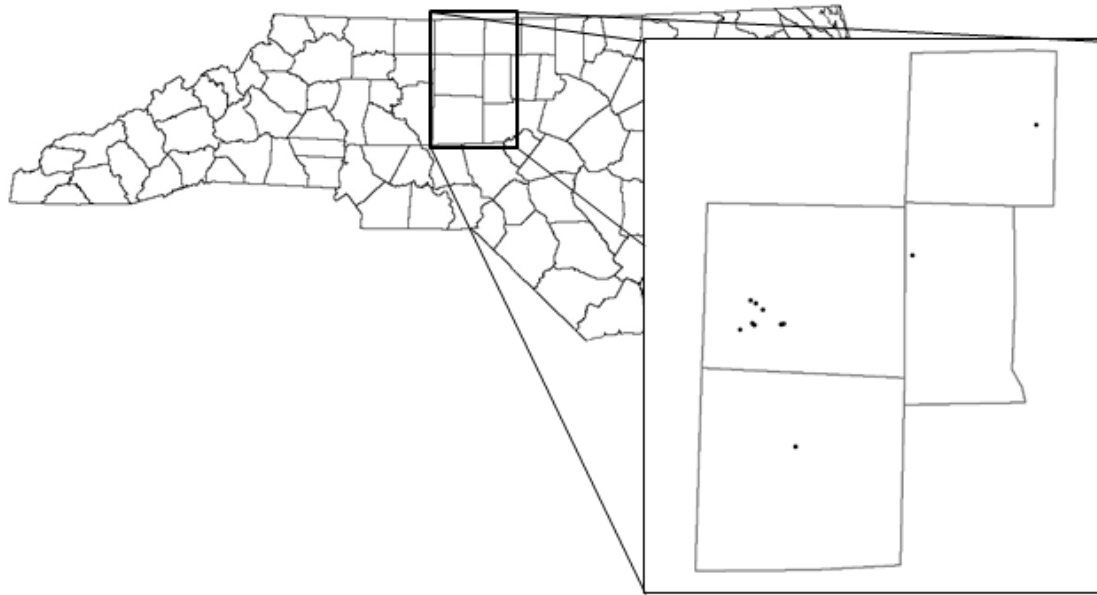


Figure 1. The four counties and locations of PSH within. Top: Caswell County (n = 1), middle: Guilford County (n = 18), Alamance County (n = 1), and bottom: Randolph County (n = 1).

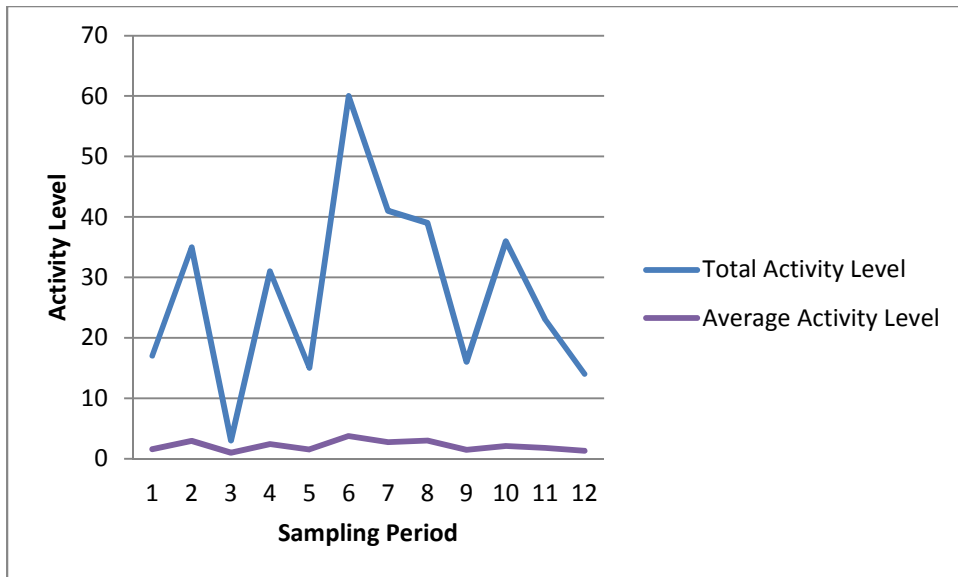


Figure 2. Total and average activity level for all species for each sampling period. Total activity level was calculated by summing the 1,2,3 MCS number for each species present during each sampling period. Average activity level was calculated as total activity level divided by the number sites where anuran presence was detected during each sampling period.

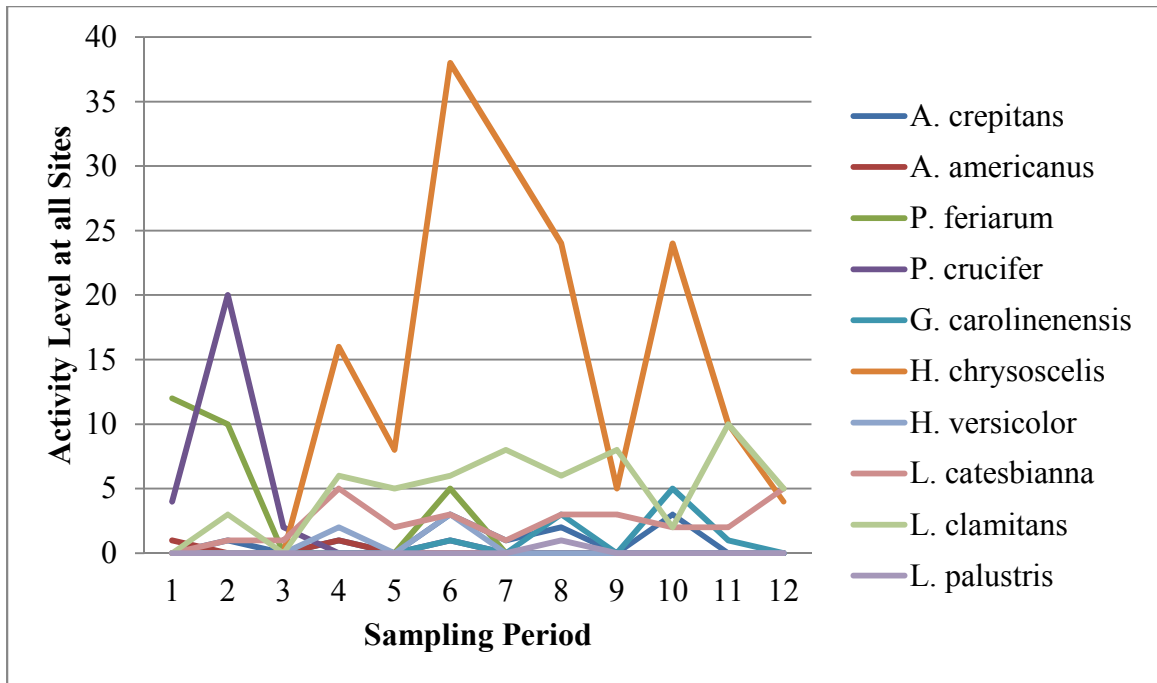


Figure 3. Activity level for each species during each sampling period. Activity level was calculated by summing the 1,2,3 MCS number for all sites a species was present at during each sampling period.

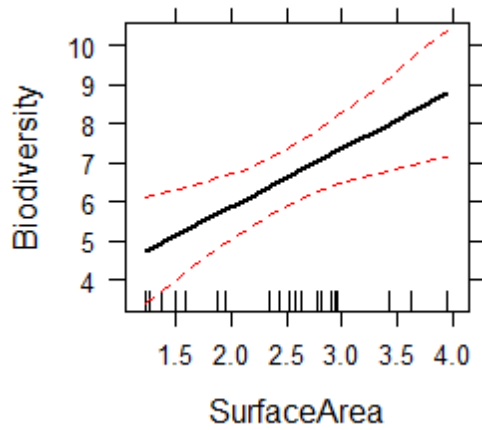


Figure 4. Association of PSH surface area and anuran biodiversity. Surface area is log-transformed and anuran biodiversity is measured as the sum of the number of species multiplied by each species maximum 1,2,3 MCS number. The black line is the line of best fit and the red dashed lines are 95% confidence intervals.

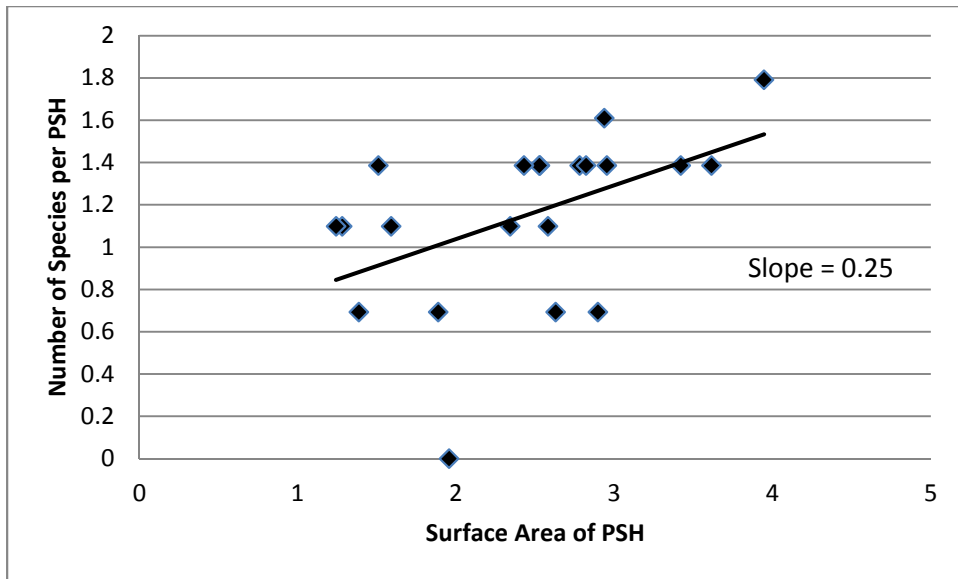


Figure 5. Species-Area Relationship curve for anuran diversity. Log-transformed number of species at each site versus log-transformed PSH surface area.

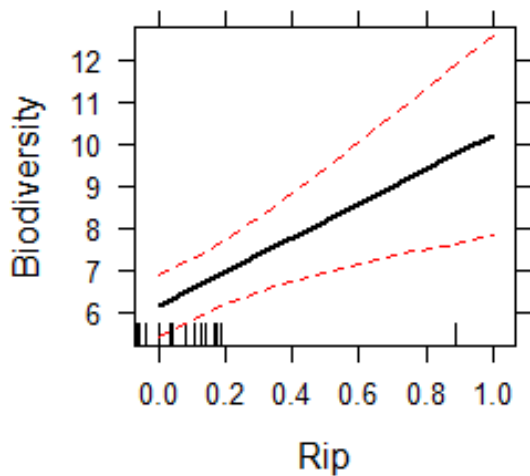


Figure 6. Association of the presence of riparian vegetation and anuran biodiversity. Riparian vegetation is measured as present (1) or absent (0) and anuran biodiversity is measured as the sum of the number of species multiplied by each species maximum 1,2,3 MCS number. The black line is the line of best fit and the red dashed lines are 95% confidence intervals.

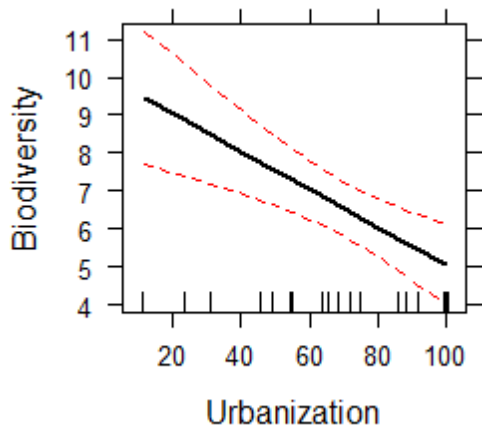


Figure 7. Association of urbanization and anuran biodiversity. Urbanization is measured as the percent impervious cover surrounding each PSH at a 100-meter radius and anuran biodiversity is measured as the sum of the number of species multiplied by each species maximum 1,2,3 MCS number. The black line is the line of best fit and the red dashed lines are 95% confidence intervals.

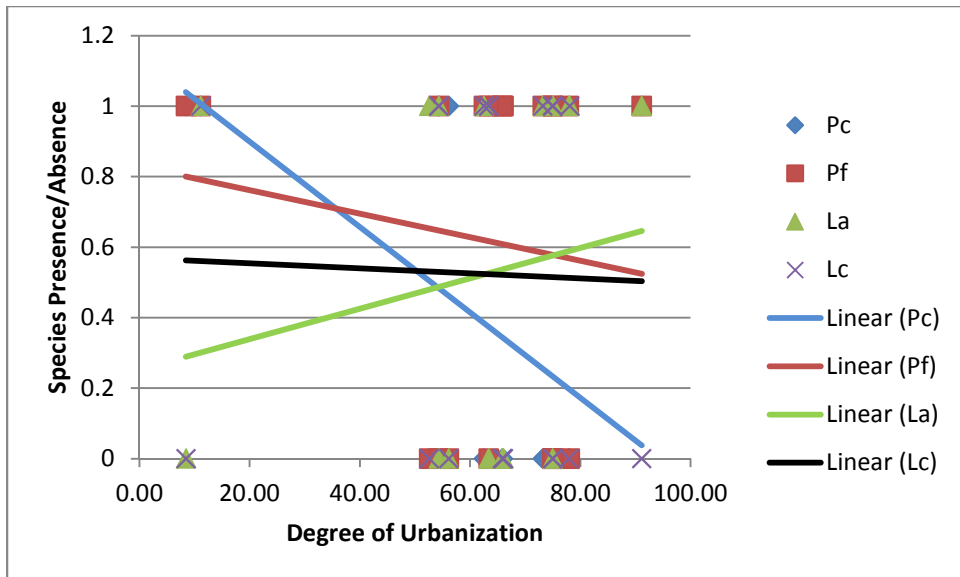


Figure 8. Association of urbanization and the presence of individual species. The presence of Pc (*P. crucifer*, blue diamond/blue line), Pf (*P. feriarum*, red square/red line), La (*L. catesbianna*, green triangle/green line), and Lc (*L. clamitans*, blue X/black line) as correlated with urbanization. Trendlines were added to assist with visual representation.

Table 1. The anuran species pool for each county. X = species is within that county's species pool.

Species	Guilford	Alamance	Randolph	Caswell
<i>Anaxyrus americanus</i>	X	X	X	X
<i>Anaxyrus fowleri</i>	X	X	X	X
<i>Acris crepitans</i>	X	X	X	X
<i>Hyla chrysoscelis</i>	X	X	X	X
<i>Hyla versicolor</i>				X
<i>Pseudacris feriarum</i>	X	X	X	X
<i>Pseudacris crucifer</i>	X	X	X	X
<i>Gastrophyrne carolinensis</i>	X	X	X	
<i>Lithobates sphenoccephala</i>	X	X	X	
<i>Lithobates palustris</i>	X	X	X	X
<i>Lithobates clamitans</i>	X	X	X	X
<i>Lithobates catesbianna</i>	X	X	X	

Table 2. Values of local scale factors that were measured in the study. PSH are identified using the number assigned by the NCDOT. Area = PSH surface area, Depth = depth at center of each PSH, Angle = angle in degrees of the incline of the bank of each PSH, Wetland = presence (1) or absence (0) of an additional man-made drainage area at each PSH, Soil = presence (1) or absence (0) of soil in each PSH, SubVeg = presence (1) or absence (0) of submerged terrestrial vegetation in each PSH, Float = presence (1) or absence (0) of floating non-algal vegetation in each PSH, Rip = presence (1) or absence (0) of riparian (aquatic) vegetation in each PSH, Surround = the type of terrestrial vegetation surrounding each PSH (1 = grass, 2 = scrub, non-woody vegetation, 3 = woody forest), and Shade = shade covered at each PSH during solar noon in May (0 = full sun/no shade, 1 = < 25% shade, 2 = 25-50% shade, 3 = 50-75% shade, 4 = > 75% shade). Area was log-transformed for statistical analysis.

PSH	Area	Depth	Angle	Wetland	Soil	SubVeg	Float	Rip	Surround	Shade
2284	3.60	0.27	14.93	0	1	1	1	0	1	4
2278	30.55	0.04	1.32	1	0	0	0	0	1	1
2276	12.57	0.58	16.28	0	1	1	1	0	2	1
2283	4.52	0.27	12.53	0	0	0	0	0	3	3
2059	4.00	0.33	18.27	0	0	0	0	0	3	4
2487	37.11	0.17	6.97	1	1	0	1	0	1	2
2492	10.40	0.38	10.79	0	1	0	1	0	1	0
2493	4.91	0.17	7.52	0	0	0	1	0	1	0
2494	6.61	0.19	7.48	0	0	0	0	0	1	1
2495	16.15	0.23	7.66	1	1	0	1	0	1	0
2496	7.07	0.13	4.84	0	0	1	0	0	1	0
2500	16.80	0.13	5.64	1	1	1	0	0	1	2
2501	12.50	0.37	16.49	0	1	1	0	0	1	1
2286	18.86	0.56	12.85	0	1	1	1	1	1	3
2287	13.85	0.13	3.46	0	1	0	0	0	3	3
2288	18.10	0.24	5.74	0	0	0	0	0	3	3
2289	13.20	0.51	14.08	0	0	0	0	0	2	0
2290	19.15	0.27	7.24	1	0	0	1	0	2	1
2505	3.46	0.25	13.27	0	0	0	0	0	1	0
2532	51.70	0.25	11.66	1	0	0	0	0	1	0
1344	11.34	0.20	8.75	1	0	1	1	1	2	2

Table 3. Values of regional scale factors measured in the study. PSH are identified using the number assigned by the NCDOT. UTM North, East, and Zone are location data provided by the NCDOT. County = county each PSH is located in, Road = distance (in meters) from center of each PSH to the nearest edge of the road, Patch = distance (in meters) from center of each PSH to the nearest patch of forest, H2O = distance (in meters) from center of each PSH to the nearest riparian zone, DistPSH = distance (in meters) from center of each PSH to the center of the nearest PSH, Urb100 = percent of impervious cover within a 100-meter radius of each PSH, and Urb1000 = percent of impervious cover within a 1000-meter radius of each PSH. Road, Patch, H2O, and DistPSH were all log-transformed for statistical analysis.

PSH	UTM North	UTM East	UTM Zone	County	Road	Patch	H2O	DistPSH	Urb100	Urb1000
2284	3988064	600221	17	Guilford	15.70	2.22	29.93	136.13	48.70	78.09
2278	3989615	598585	17	Guilford	16.70	17.49	28.05	1221.31	91.67	91.16
2276	3990256	597436	17	Guilford	20.60	30.89	417.42	1221.31	74.48	73.20
2283	3988140	600107	17	Guilford	8.90	1.00	90.65	136.13	11.46	77.92
2059	3983680	595081	17	Guilford	17.00	1.00	29.74	3007.53	88.81	52.57
2487	3984830	598022	17	Guilford	18.40	3.61	63.71	171.28	100	62.51
2492	3984741	598169	17	Guilford	13.00	11.41	103.43	22.06	100	75.00
2493	3984729	598190	17	Guilford	12.30	16.33	120.48	22.06	100	75.00
2494	3984715	598209	17	Guilford	11.80	6.91	149.76	24.60	100	75.00
2495	3984692	598240	17	Guilford	11.40	31.98	189.23	36.86	100	75.00
2496	3984665	598274	17	Guilford	10.70	75.79	229.61	41.32	100	75.00
2500	3985052	597787	17	Guilford	23.10	5.12	304.95	41.31	54.43	54.32
2501	3985080	597753	17	Guilford	21.40	4.74	266.44	41.31	45.84	54.32
2286	3984832	604274	17	Guilford	30.70	1.00	64.73	77.17	71.62	66.06
2287	3984842	604352	17	Guilford	26.10	1.00	7.45	30.11	68.75	65.89
2288	3984858	604379	17	Guilford	23.80	1.00	25.75	30.11	65.89	65.89
2289	3984915	604757	17	Guilford	9.10	16.01	4.47	78.08	63.03	63.40
2290	3984919	604838	17	Guilford	9.00	10.28	7.76	78.08	54.43	63.40
2505	3957333	607539	17	Randolph	11.50	5.44	15.60	27389.29	85.94	56.15
2532	4000325	633813	17	Alamance	7.00	9.42	15.22	32744.82	22.92	11.14
1344	4029610	661645	17	Caswell	15.30	13.33	60.62	40302.57	31.51	8.45

Table 4. Species presence and activity level for each site for each sampling period in 2012. PSH's are identified using the number assigned by the NCDOT. Activity level is shown using the 1,2,3 MCS abundance level (1 = single individual calling/visual of adult/visual of distinct egg-mass, 2 = multiple distinguishable individuals calling, 3 = multiple indistinguishable individuals calling) Hc = *Hyla chrysoscelis*, Pf = *Pseudacris feriarum*, La = *Lithobates catesbianna*, Lc = *Lithobates clamitans*, Pc = *Pseudacris crucifer*, Ba = *Anaxrynus americanus*, Gc = *Gastrophyrne carolinensis*, Lp = *Lithobates palustris*, Hv = *Hyla versicolor*, Ac = *Acris crepitans*, Bf = *Anaxrynus fowleri*, and Ls = *Lithobates sphenoccephala*.

Sampling Period	1 March 1-7	2 March 21-30	3 April 11-24	4 April 30 - May 3	5 May 7-10	6 May 14-17	7 May 21-24	8 May 28-31	9 June 4-7	10 June 11-14	11 June 18-21	12 June 25-28
2284							Hc (3), La (1)		Hc (3), Lc (1)		Hc (1), La (1)	
2278	Pf (1), Pc (1)	Pc (2)					Hc (2)		Hc (2)			
2276	Pf (1), Pc (1)					Hc (2), La (1)	Hc (2)		La (1)	Hc (2), Lc (1)	Hc (1), Lc (1)	Lc (1)
2283	Pf (2)			Hc (2), La (1)		Hc (1)	Hc (3)	Hc (2), La (1), Lp (1)	La (1)		Hc (3)	La (1)
2059							Hc (3)	Hc (2), La (1)		Hc (2), La (1)	Hc (1)	
2487	Pf (2)	Lc (1)		La (1), Lc (1)	La (1), Lc (1)	Hc (3), La (1), Lc (1)	Hc (3), Lc (1)	Hc (2), Lc (1)	Lc (1)	Hc (1*)	Lc (1)	La (1), Lc (1)
2492		Lc (1)	La (1)	La (1), Lc (1)	Lc (1)	Hc (2), Lc (1)	Hc (2), Lc (2)	Hc (2), Lc (2)	Lc (1)	Hc (1*)	La (1), Lc (1)	La (1)
2493	Pf (1)				Lc (1)	Pf (2), Hc (2)	Hc (2)	Hc (2)		Hc (1*)		La (1)
2494		Lc (1)		Lc (1)	Hc (1)	Hc (3), Lc (1)		Hc (3)		Hc (1*)	Hc (1)	
2495	Pf (1)			La (1)	Hc (1)	Hc (2), La (1)	Hc (2)	Hc (2)		Hc (1*)	Lc (1)	
2496						Hc (2)						
2500		Pc (3)		Hc (2), Lc (1)	Lc (1)	Hc (3), Lc (1)	Hc (3), Lc (2)	Hc (3), La (1), Lc (1)	La (1), Lc (1)	Hc (1*)	Lc (1)	La (1), Lc (1)
2501	Pf (1)	Pc (3)	Pc (2)	Hc (1), Lc (1)		Hc (3), Lc (2)	Hc (3), Lc (1)	Hc (1), Lc (1)	Lc (1)	Lc (1*)	Lc (3)	Lc (1)
2286		Pf (2), Pc (3)		Aa (1)		Gc (1), Hc (3)				Gc (3), Hc (3)		Hc (1)
2287		Pf (3)			Hc (1)	Pf (2), Hc (3)				Hc (2)		
2288		Pf (1)				Pf (1)		Hc (2)		Hc (2)		Hc (1)

2289				Lc (1)			Lc (1)	Ge (3), Hc (2), Lc (1)	Lc (1)	Ge (2), Lc (1)	Ge (1), Lc (1)	Lc (1)
2290	Pf (1)	Pc (3)		Hc (2)	Hc (2)		Lc (1)		Lc (1)	Hc (3)	Hc (2), Lc (1)	Hc (2)
2505	Pc (1), Aa (1)			Hc (3)	Lc (1)	Hc (3)	Hc (2)			Hc (2)	Dry	Dry
2532	Pf (1), Pc (1)	Pf (1), Pc (3), Ac (1), La (1)		Ac (1), Hc (3), La (1)	Hc (3), La (1)	Ac (3), Hc (3)	Ac (1), Hc (1)	Ac (2)	Lc (1)	Ac (3), Hc (2), La (1)	Hc (1)	
1344	Pf (1)	Pf (3), Pc (3)		Hc (3), Hv (2)		Pc (1), Hc (3), Hv (3)		Hc (1)				

Table 5. Nested matrix of species presence. PSH's are identified using the number assigned by the NCDOT and are arranged from largest to smallest based on surface area. Hc = *Hyla chrysoscelis*, Pf = *Pseudacris feriarum*, La = *Lithobates catesbianna*, Lc = *Lithobates clamitans*, Pc = *Pseudacris crucifer*, Ba = *Anaxyrus americanus*, Gc = *Gastrophyrne carolinensis*, Lp = *Lithobates palustris*, Hv = *Hyla versicolor*, Ac = *Acris crepitans*, Bf = *Anaxyrus fowleri*, and Ls = *Lithobates sphenoccephala*.

PSH	Hc	Pf	La	Lc	Pc	Ba	Gc	Lp	Hv	Ac	Bf	Ls
2532	X	X	X	X	X					X		
2487	X	X	X	X								
2278	X	X	X		X							
2290	X	X		X	X							
2286	X	X			X	X	X					
2288	X	X										
2500	X		X	X	X							
2495	X	X	X	X								
2287	X	X										
2289	X			X			X					
2276	X	X	X	X								
2501	X	X		X	X							
1344	X	X			X				X			
2492	X		X	X								
2496	X											
2494	X			X								
2493	X	X	X									
2283	X	X	X					X				
2059	X		X									
2284	X		X	X								
2505	X					X						

Table 6. The biodiversity model for the presence of all species. logArea =log transformed surface area of PSH, RipVeg = the presence of riparian vegetation, and Urb100 = urbanization at 100 meter radius). Adjusted R² = 0.7003.

	Estimate	Std. Error	t value	Pr(> t)
Intercept	5.996	1.548	3.873	0.001
logArea	1.487	0.459	3.241	0.005
RipVeg	4.069	1.191	3.416	0.003
Urb100	-0.050	0.013	-3.815	0.001

Table 7. The logit model for *P. crucifer* presence. Urb100 = urbanization at 100 meter radius and of logArea = log transformed surface area of PSH.

	Estimate	Std. Error	z value	Pr(> z)
(Intercept)	-20.3715	13.2703	-1.535	0.125
Urb100	-0.319	0.2129	-1.498	0.134
logArea	14.5095	9.3967	1.544	0.123

Table 8. The logit model for *P. feriarum* presence. Urb1000 = urbanization at 1000 meter radius, logArea = log transformed surface area of PSH and Clam = presence of *L. clamitans*.

	Estimate	Std. Error	z value	Pr(> z)
(Intercept)	2.7519	3.9812	0.691	0.4894
Urb1000	-0.1124	0.0689	-1.632	0.1028
logArea	2.1663	1.2147	1.784	0.0745
Clam	-7.1889	4.6522	-1.545	0.1223

Table 9. The logit model for *L. catesbianna* presence. Angle = the angle in degrees of the slope of the bank of each PSH, logH2O = log transformed distance to nearest natural riparian zone), SubVeg = presence of submerged terrestrial/non-aquatic vegetation, Urb1000 = urbanization at 1000 meter radius, and Wetland = presence of additional human-made wetland drainage for PSH.

	Estimate	Std. Error	z value	Pr(> z)
(Intercept)	-18.0583	8.55702	-2.11	0.0348
Angle	0.57762	0.31869	1.813	0.0699
logH2O	1.44578	0.76235	1.896	0.0579
SubVeg	-3.20118	1.97139	-1.624	0.1044
Urb1000	0.08742	0.05196	1.683	0.0925
Wetland	6.49478	3.08758	2.104	0.0354

Table 10. The logit model for *L. clamitans* presence. Urb100 = urbanization at 100 meter radius, logRoad = log-transformed distance to the nearest road to each PSH, and Soil = presence or absence of soil in the PSH).

	Estimate	Std. Error	z value	Pr(> z)
(Intercept)	53.71052	28.14288	1.908	0.0563
Urb100	0.08274	0.05725	1.445	0.1484
logRoad	-25.7305	13.50527	-1.905	0.0568
Soil	23.74503	12.32747	1.926	0.0541

REFERENCES

- Abrams, P.A. & Walters, C.J. 1996. Invulnerable prey and the paradox of enrichment. *Ecology* **77**, 1125–1133
- Alford, R. A. & Richards, S. J. 1999. Global amphibian declines: a problem in applied ecology. *Annual review of Ecology and Systematics* **30**, 133–165
- AmphibiaWeb Information on amphibian biology and conservation. 2013. Berkeley, California. <http://amphibiaweb.org/>. Retrieved, June 26, 2013.
- Barrett, K. & Guyer, C. 2008. Differential responses of amphibians and reptiles in riparian and stream habitats to land use disturbances in western Georgia, USA. *Biological Conservation* **141**, 2290–2300
- Battin, J. 2004. When good animals love bad habitats: Ecological traps and the conservation of animal populations. *Conservation Biology* **18**, 1482–1491
- Beene, J. 2012. [Anuran presence in Guilford, Caswell, Alamance, and Randolph Counties, NC]. Unpublished raw data. Accessed January 18, 2013.
- Birx-Raybuck, D. A., Price, S. J. & Dorcas, M. E. 2009. Pond age and riparian zone proximity influence anuran occupancy of urban retention ponds. *Urban Ecosystems* **13**, 181–190
- Blair, R.B. 1996. Land use and avian species diversity along an urban gradient. *Ecological Applications*. **6**, 506–519
- Blair, R.B. & Launer, A.E.. 1997. Butterfly diversity and human land use: species assemblages along an urban gradient. *Biological Conservation*. **80**, 113–125
- Brand, A. B. & Snodgrass, J. W. 2010. Value of Artificial Habitats for Amphibian Reproduction in Altered Landscapes. *Conservation Biology* **24**, 295–301

- Brand, A.B., Snodgrass, J.W., Gallagher, M.T., Casey, R.E., & Van Meter, R. 2010. Lethal and sublethal effects of embryonic and larval exposure of *Hyla versicolor* to stormwater pond sediments. *Archives of Environmental Contamination and Toxicology* **58**, 325–331
- Brazel, A., Selover, N., Vose, R. & Heisler, G. 2000. The tale of two climates-Baltimore and Phoenix urban LTER sites. *Climate Research* **15**, 123–135
- Brown, J.H. & Kodric-Brown, A. 1977. Turnover rates in insular biogeography: effect of immigration on extinction. *Ecology*. **58**, 445–449
- Burrowes, P. A., Joglar, R. L. & Green, D. E. 2004. Potential causes for amphibian declines in Puerto Rico. *Journal Information* **60**, 141-154
- Carrier, J. & Beebee, T. J. C. 2003. Recent, substantial, and unexplained declines of the common toad *Bufo bufo* in lowland England. *Biological Conservation* **111**, 395-399
- Chace, J. F. & Walsh, J. J. 2006. Urban effects on native avifauna: a review. *Landscape and Urban Planning* **74**, 46–69
- Connell, J. H. 1978. Diversity in tropical rain forests and coral reefs. *Science* **199**, 1302-1310
- Davidson HerpLab. 2012. [Anuran calling data for North and South Carolina]. Unpublished raw data. Accessed January 18, 2013.
- Delis, P.R., Mushinsky, H.R., & McCoy, E.D. 1996. Decline of some west-central Florida anuran populations in response to habitat degradation. *Biodiversity and Conservation*. **5**, 1579–1595
- Dodd, C.K., Smith, L.L., 2003. Habitat destruction and alteration: historical trends and future prospects for amphibians. In: Semlitsch, R.D. (Ed.), *Amphibian Conservation*. Smithsonian Institution, Washington, DC, pp. 94–112
- Donnelly, R. & J.M. Marzluff. 2006. Relative importance of habitat quantity, structure, and spatial pattern to birds in urbanizing environments. *Urban Ecosyst.* **9**, 99–117

- Dorcas, M.E., & Gibbons, J.W. 2008. *Frogs and toads of the southeast*. University of Georgia Press, Athens
- Dorcas, M.E., Price, S.J., Walls, S.C., & Barichivich, W.J. 2009. Auditory monitoring of anuran populations. *Conservation and ecology in amphibians*. Oxford University Press, Oxford, United Kingdom, pp. 281-298
- Dorcas, M.E. 1999. Amphibians and reptiles of North Carolina. *Herpsofnc.org*. Retrieved January 2013, from <http://www.herpsofnc.org>
- Drake, J.A. 1991. Community-assembly mechanics and the structure of an experimental species ensemble. *American Naturalist* **137**, 1–26
- Faeth, S. H., Bang, C. & Saari, S. 2011. Urban biodiversity: patterns and mechanisms. *Annals of the New York Academy of Sciences* **1223**, 69–81
- Faeth, S.H. & T.C. Kane. 1978. Urban biogeography: city parks as islands for Diptera and Coleoptera. *Oecologia*. **32**, 127–133
- Ficetola, G.F. & De Bernardi, F. 2004. Amphibians in a human-dominated landscape: the community structure is related to habitat features and isolation. *Biological Conservation* **119**, 219–230
- Fisher, R.N. & Shaffer, H.B. 1996. The decline of amphibians in California's Great Central Valley. *Conservation Biology* **10**, 1387-1397
- Gagné, S.A., & Fahrig, L. 2007. Effect of landscape context on anuran communities in breeding ponds in the National Capital Region, Canada. *Landscape Ecology*. **22**, 205–215
- Gilroy, J. & Sutherland, W. 2007. Beyond ecological traps: perceptual errors and undervalued resources. *Trends in Ecology & Evolution*. **22**, 351–356
- Gooch, M. M., Heupel, A. H., Price, S. J., and Dorcas, M. E. 2006. The effects of survey protocol on detection probabilities and site occupancy estimates of summer breeding anurans. *Applied Herpetology*. **3**, 129–42

- Green, D. M., & Parent, C. 2003. Variable and asymmetric introgression in a hybrid zone in the toads, *Bufo americanus* and *Bufo fowleri*. *Copeia*. **2003**, 34–43
- Gregg, J.W., Jones, C.G., & Dawson, T.E. 2003. Urbanization effects on tree growth in the vicinity of New York City. *Nature*. **424**, 183-187
- Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X., Bridges, J.M., 2008. Global change and the ecology of cities. *Science*. **319**, 756–760
- Hanski, I.A. 1994. Patch-occupancy dynamics in fragmented landscapes. *Trends in Ecology and Evolution*. **9**, 131–135
- Hamer, A. J. & McDonnell, M. J. 2008. Amphibian ecology and conservation in the urbanising world: A review. *Biological Conservation* **141**, 2432–2449
- Hamer, A. J. & McDonnell, M. J. 2009. The response of herpetofauna to urbanization: Inferring patterns of persistence from wildlife databases. *Austral Ecology* **35**, 568–580
- Hecnar, S.J. & M'Closkey, R.T. 1997. Spatial scale and determination of species status of the green frog. *Conservation Biology* **11**, 670- 682
- Hope, D., Gries, C., Zhu, W., Fagan, W.F., Redman, C.L., Grimm, N.B., Nelson, A.L., Martin, C., & Kinzig, A. 2003. Socioeconomics drive urban plant diversity. *Proceedings of the National Academy of Sciences of the USA* **100**, 8788–8792
- Imhoff, M.L., Bounoua, L., DeFries, R., Lawrence, W.T., Stutzer, D., Tucker, C.J., & Ricketts, T. 2004. The consequences of urban land transformation on net primary productivity in the United States. *Remote Sensing of Environment* **89**, 434-443
- Kaiser, K. & Hammers, J. 2009. The effect of anthropogenic noise on male advertisement call rate in the neotropical treefrog, *Dendropsophus triangulum*. *Behaviour* **146**, 1053–1069
- Kaye, J. P., Groffman, P. M., Grimm, N. B., Baker, L. A., & Pouyat, R. V. 2006. A distinct urban biogeochemistry? *Trends in Ecology and Evolution* **21**, 192-199

- Kirilin, M., Gooch, M. M., Price, S. J., & Dorcas, M. E. 2006. Predictors of winter anuran calling activity in the North Carolina Piedmont. *Journal of the North Carolina Academy of Science* **122**, 10–18
- Knutson, M. G., Sauer, J. R., Olsen, D. A., Mossman, M. J., Hemesath, L. M. & Lannoo, M. J. 1999. Effects of landscape composition and wetland fragmentation on frog and toad abundance and species richness in Iowa and Wisconsin, U.S.A. *Conservation Biology* **13**, 1437–1446
- Kühn, I., Brandl, R. & Klotz, S.. 2004. The flora of German cities is naturally species rich. *Evolutionary Ecology Research*. **6**, 749–764
- Kuttler, W. 2008. The urban climate-basic and applied aspects. In: Marzluff, J. M., Schulenberger, E., Endlicher, W., Alberti, M., Bradley, G., Ryan, C., Simon, U., & ZumBrunnen, C. (eds.), *Urban Ecology: an international perspective on the interaction between humans and nature*. Springer Science+Business Media. New York, NY. pp. 233-248
- Lawler, J. J., White, D., Sifneos, J. C. & Master, L. L. 2003. Rare species and the use of indicator groups for conservation planning. *Conservation Biology* **17**, 875–882
- Leibold, M.A., Holyoak, M., Mouquet, N., Amarasekare, P., Chase, J.M., Hoopes, M.F., Holt, R.D., Shurin, J.B., Law, R., Tilman, D., Loreau, M., & Gonzalez, A. 2004. The metacommunity concept: a framework for multi-scale community ecology. *Ecology Letters* **7**, 601-613
- Lewis, D. B., Kaye, J. P., Gries, C., Kinzig, A. P. & Redman, C. L. 2006. Agrarian legacy in soil nutrient pools of urbanizing arid lands. *Global Change Biology* **12**, 703–709
- Lips, K.R., Diffendorfer, J., Mendelson III, J. R., & Sears, M.W. 2008. Riding the wave: Reconciling the roles of disease and climate change in amphibian declines. *PLoS Biology* **6**(3): e72
- MacArthur, R.H. and Wilson, E.O. 1967. *The Theory of Island Biogeography*. Princeton University Press, Princeton, NJ, USA.

- Marsh, D. M. and Trenham, P. C. 2008. Current trends in plant and animal population monitoring. *Conservation Biology*, **22**, 647–55
- Marzluff, J. M. 2005. Island Biogeography for an Urbanizing World How Extinction and Colonization May Determine Biological Diversity in Human-Dominated Landscapes. *Urban Ecology* 355–371
- McKinney, M. L. 2006. Urbanization as a major cause of biotic homogenization. *Biological Conservation* **127**, 247–260
- McKinney, M. L. 2008. Effects of urbanization on species richness: A review of plants and animals. *Urban Ecosyst* **11**, 161-176
- Mitchell, J. C., Brown, R. E. J., & Bartholomew, B. 2008. *Urban Herpetology*. Society for the Study of Amphibians and Reptiles. Salt Lake City
- Murdoch, W.W. & Oaten, A. 1975. Predation and population stability. *The American Naturalist*, **9**, 1–31.
- North Carolina Department of Transportation 2008. Stormwater best management practices toolbox. NC Department of Transportation Hydraulics Unit, Raleigh, NC
- Ostergaard, E. C., Richter, K. O., & West, S. D. 2008. Amphibian use of stormwater ponds in the Puget lowlands of Washington, USA. In: Mitchell JC, Brown REJ, Bartholomew B (eds) *Urban herpetology*. Society for the Study of Amphibians and Reptiles, Salt Lake City, pp 259–270
- Parlow, E. 2011. Urban climate. In: Niemela, J. (Ed.), *Urban Ecology*. Oxford University Press Inc., New York, NY, pp. 31-44
- Parris, K.M. 2006. Urban amphibian assemblages as metacommunities. *Journal of Animal Ecology* **75**, 757–764
- Partecke, J. & Gwinner, E. 2007. Increased sedentariness in European blackbirds following urbanization: a consequence of local adaptation? *Ecology* **88**, 882–890

Pauleit, S. & Breuste, J. H. 2011. Land-use and surface-cover as urban ecological indicators. In: Niemela, J. (Ed.), *Urban Ecology*. Oxford University Press Inc., New York, NY, pp. 19-30

Phillips, K. 1990. Where have all the frogs and toads gone? *BioScience* **40**, 422–424

Pope, S.E., Fahrig, L., & Merriam, H.G. 2000. Landscape complementation and metapopulation effects on leopard frog populations. *Ecology* **81**, 2498–2508.

Pounds, J.A. & Crump, M.L. 1994. Amphibian declines and climate disturbance: the case of the golden toad and the harlequin frog. *Conservation Biology* **8**, 72-85

Price, S. J. & Dorcas, M. E. 2011. The Carolina Herp Atlas: an online, citizen-science approach to document amphibian and reptile occurrences. *Herpetological Conservation and Biology* **6**, 287–296

Raupp, M.J., Shrewsbury, P.M. & Herms, D.A. 2010. Ecology of herbivorous arthropods in urban landscapes. *Annual Review of Entomology* **55**, 19–38

Robbins, P. & Sharp, J. T. 2003. Producing and consuming chemicals: the moral economy of the American lawn. *Economic Geography* **79**, 425–451

Rosenzweig, M.L. 1995. *Species Diversity in Space and Time*. Cambridge University Press, Cambridge, UK

Rosenzweig, M. L. 2003. Reconciliation ecology and the future of species diversity. *Oryx* **37**, 194-205

Scheffers, B. R. & Paszkowski, C. A. 2011. The effects of urbanization on North American amphibian species: Identifying new directions for urban conservation. *Urban Ecosystems* **15**, 133–147

Semlitsch, R.D. & Bodie, J.R. 2003. Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles. *Conservation Biology* **17**, 1219–1228

Shmida, A. & Wilson, M.V. 1985. Biological determinants of species diversity. *Journal of Biogeography*. **12**, 1–20.

- Shochat, E., Lerman S.B., Anderies, J. M., Warren, P. S., Faeth, S. H., and Nilon, C. H. 2010. Invasion, Competition, and Biodiversity Loss in Urban Ecosystems. *BioScience* **60**, 199–208
- Shochat, E., Warren, P.S., Faeth, S.H., McIntyre, N.E., and Hope, D. 2006. From patterns to emerging processes in mechanistic urban ecology. *Trends in Ecology and Evolution* **21**, 186-191
- Swan, C. M., Pickett, S. T. A., Szlavecz, K., Warren, P., & Willey, K. T. 2011. Biodiversity and community composition in urban ecosystems: coupled human, spatial, and metacommunity processes. In: Niemela, J. (Ed.), *Urban Ecology*. Oxford University Press Inc., New York, NY, pp. 179-186
- Stephens, P. A. and Sutherland, W. J. 1999. Consequences of the Allee effect for behaviour, ecology, and conservation. *Trends in Ecology and Evolution* **14**, 401-405
- Todd, M. J., Cocklin, R. R., and Dorcas, M. E. 2003. Temporal and spatial variation in anuran calling activity in the western Piedmont of North Carolina. *Journal of the North Carolina Academy of Science* **119**, 103–110
- United Nations Population Division, 2012. World urbanization prospects. U.N. Department of Economic and Social Affairs, New York, New York, U.S.A.
- United States Geological Survey. 2006. Multi-Resolution Land Characteristics Consortium (MRLC). Retrieved January 25, 2013, from <http://www.mrlc.gov/index.php>
- van der Ree, R., Jaeger, J. A. G., van der Grift, E. A. & Clevenger, A. P. 2011. Effects of roads and traffic on wildlife populations and landscape function: Road ecology is moving toward larger scales. *Ecology and Society* **16**, 48–48
- Vitt, L.J., Caldwell, J.P., Wilbur, H.M., & Smith, D.C. 1990. Amphibians as harbringers of decay. *Bioscience* **40**, 418
- Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P.M. Groffman, and R. P. Morgan III. 2005. The urban stream syndrome: Current knowledge and the search for a cure. *Journal of the North American Benthological Society* **24** 706-723

Wenguang, Z., Yuanman, H., Jinchu, H., Yu, C., Jing, Z., and Miao, L. 2008. Impacts of land-use change on mammal diversity in the upper reaches of Minjiang River, China: implications for biodiversity conservation planning. *Landscape and Urban Planning* **85**, 195-204

Wright, A. H. and Wright, A. A. 1949. *Handbook of Frogs and Toads of the United States and Canada*. 3rd edn. Comstock Publishing Associates, Ithaca, NY