USE OF EDGE AND INTERIOR HABITAT OF URBAN FOREST REMNANTS BY
AVIFAUNA AND HERPETOFAUNA

By

DANIEL EUGENE DAWSON

A THESIS PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
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To my family, especially my mother and father, who have gone far beyond their obligation as parents to help me succeed academically and develop professionally

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF OBJECTS

Abstract of Thesis Presented to the Graduate School
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Requirements for the Degree of Master of Science

USE OF EDGE AND INTERIOR HABITAT OF URBAN FOREST REMNANTS BY
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By

Daniel Eugene Dawson

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Urban forest remnants are utilized by various wildlife species, but little research has been

conducted on whether certain species avoid or prefer edges of urban remnants. The objective of

this study was to determine whether avifauna and herpetofauna differentially use edge and

interior habitat of urban forest remnants.

With avifauna, I used point counts to survey 6 urban forest remnants (2.6–16.6 ha) in

Gainesville, Florida from November 2004 through October 2005. I compared the average daily

relative abundances of individual species and residency groups within the winter, spring,

summer, and fall seasons at edge locations (40 m from edge) and interior locations (beyond 40 m

from edge). I measured a suite of vegetative structure characteristics at edges and interiors during

both the dormant and growing seasons. Out of 77 species sighted, only a few individual bird

species and residency groups were found to use edges and interiors differently. During the

summer, the two groups of all and uncommon year-round residents had higher relative

abundances at edges. In addition, during the fall, groups containing all migrants, common

migrants, summer migrants, and uncommon migrants had higher abundances at interiors.

Analyses of vegetative structure revealed very few differences between edges and interiors

during either the growing season or the dormant season.

With herpetofauna, I used pitfall/funnel trap-PVC pipe sampling arrays to survey 5 urban forest remnants (3.0–16.6 ha) in Gainesville, Florida during the summers of 2005 and 2006. I compared the average daily relative abundances of individual species and taxa groups (Order and Suborder-level; Family-level), as well as species richness at edge locations (40 m from edge) and interior locations (beyond 40 m from edge). Results showed that neither the relative abundances of individual species and taxa groups, nor species richness was significantly different between edges and interiors. However, the relative abundances of the species *Hyla squirella*, the Ranid and Anuran groups, and the Sub-order Serpentes group (Snakes), as well as species richness were significantly greater in some remnants than others.

Overall, results show that there is little segregation in the use of edge and interior habitat of these urban forest remnants by either birds or herpetofauna, which may be partially driven by similarities in vegetative structure between edges and interiors. For avifauna, however, the greater use of interiors by fall migrants suggests that the interiors of these small urban forest fragments should be managed to reduce future levels of human disturbances. For herpetofauna, larger, connected remnants with wetlands may have led to increased species richness and greater relative abundances of *Hyla squirella*, Ranids, Anurans, and snakes.

USE OF EDGE AND INTERIOR HABITAT OF URBAN FOREST REMNANTS BY
AVIFAUNA

Introduction

The impact of urbanization on wildlife and the natural environment is of growing interest

as the level of urbanization continues to increase in the world (Marzluff 2001, Alig, Jeffrey, and

Lichtenstein 2004). Birds are among the best studied urban wildlife because they are the most

visible and easy to study, they are charismatic, and they are sensitive to factors at different

temporal and spatial scales in urban environments (Mensing, Galatowitsch, and Tester 1998,

Hostetler 1999, Hostetler and Knowles-Yanez 2003, Hostetler, Scot, and Paul 2005, Atchinson

and Rodewald 2006). Urban birds are often transient, seasonally-varying organisms that are in

close proximity to human disturbance. Urban landscapes are frequently made up of fragmented

forest remnants with large edge-to-interior ratios, and these remnants are often isolated from

other remnants or larger tracks of habitat. Despite these challenges, a number of bird species do

utilize habitat fragments in a variety of urban/suburban environments throughout the year.

Edge Effects and Urban Effects on Habitat Use

Edges of habitats have been long recognized as often having higher densities, and higher

diversities of species than interior forests (Lay 1938, Gates and Gysel 1978, Noss 1991). For

example, Noss (1991) found that in a large tract of mature upland deciduous forest near

Gainesville, Florida, bird densities were significantly higher in both edges next to roads and

edge-gaps within interior forests than within interior forests themselves during most seasons.

Edge habitats often have more sunlight exposure and more emergent vegetation than interiors,

providing good opportunities for foraging on fruit and invertebrates (Noss 1991, McCollin 1998,

Rodewald and Brittingham 2004). Therefore, habitat edges are important habitat components,

and are associated with a number of species. However, some bird species have been shown to be

edge-indifferent, and others have been shown to require or prefer forest interior habitat and actively avoid edge habitat (Whitcomb et al. 1981, Noss 1991). Interior forest species tend to nest at forest interiors (Whitcomb et al. 1981), or prefer to forage on more moisture dependent insects and shade dependent plants found in the interior (Villard 1998). In addition, species associated with forest interiors may be negatively affected by disturbances such as traffic noise (Reijnen et al. 1997, Fernández-Juricic 2001), and pedestrian presence (Fernández-Juricic and Tellaria 2000). In urban forest remnants, there is often a disproportionately large amount of forest-edge habitat compared to forest-interior habitat, leading to more prevalent use by edge or edge-indifferent species than by interior-associated birds (Whitcomb et al. 1981, McIntyre 1995, Chase 2006).

Seasonal Influence on Use of Urban Forest Remnants

In temperate North America, seasonal differences in habitat use by birds depend upon breeding, wintering, or migration stop-over needs. During the breeding season, species have a broad range of habitat requirements and limitations, depending upon the breeding and nesting strategies they employ. For instance, during the breeding season, interior-forest specialists may not use the edges of habitats because they have inappropriate cover and nesting substrates, are prone to nest predation or parasitism, or are too close to human disturbance (Tilghman 1987a, McIntyre 1995, Villard 1998, Mörtberg 2001). Likewise, edge-specialists species may not use patch interiors if nesting substrates and food resources are more readily available on edges (McIntyre 1995, Fernández-Juricic 2001). In contrast with the breeding season, habitat use of forest patches in temperate North America during the winter and fall/spring migration periods generally revolves around access to food resources. During fall/spring migration, food resources at stop-over locations are especially important to birds because of the need to refuel for travel (Moore, Gauthreaux, Kerlinger, and Simons 1995). In addition, there may be an underlying and

innate strategy for arriving and leaving stop-over foraging habitat in order to maximize the chances of gaining access to higher energy foods at other stop-over locations, and arriving at end destinations faster to gain access to better quality breeding or wintering habitat (Moore et al. 1995). Likewise, the winter habitat use of many species also revolves around food intake. Because of this, habitat requirements for both stop-over migrants and winter migrants can be more flexible than during the breeding season, and birds utilize urban/suburban landscapes (Yaukey 1996, Jokimäki and Suhonen 1998, Hostetler and Holling 2000, Rodewald and Brittingham 2004, Atchinson and Rodewald 2006). Specifically, it has been hypothesized that birds may use edge-dominated, urban/suburban habitat during the winter and stop-over periods because of access to fruit-bearing ornamental plants, access to human-supplied feeders, and warmer average temperatures than non-urban habitat (Atchinson and Rodewald 2006, Shochat, Warren, Faeth, McIntyre, and Hope 2006).

Use of Edge Versus Interior Habitat

Despite the considerable amount of research on the use of urban remnants by birds, little research has compared edge versus interior habitat use in any season. In one such study by Fernández-Juricic (2001), avian habitat use in Madrid, Spain was compared between the edge and interiors of urban parks during the breeding season. He determined that species generally more habituated to human contact with generalist habitat requirements used edges significantly more than interiors, whereas species with specific forest habitat requirements used interior habitats significantly more than edges. This result suggests that certain birds differentially use edges and interiors, but similar studies have not been replicated in other urban environments, especially across seasons.

Objective

The objective of my study was to determine if birds differentially use edges and interiors of urban forest remnants during winter, spring, summer, and fall.

Methods

Study Site

The University of Florida Gainesville campus is located in north-central peninsular Florida. This study took place in 6 urban forest remnants on the University of Florida campus: Harmonic Woods (3.7 ha), Fraternity Wetlands (2.6 ha), Graham Woods (3.0 ha), Bartram-Carr Woods (3.5 ha), Lake Alice Conservation Area (11.3 ha), and Biven's Arm Forest (16.6 ha) (Figure 1-1). Three of the four smallest remnants (Harmonic Woods, Fraternity Wetlands and Bartram-Carr Woods) included largely upland mixed pine-hardwood forest, with all containing or being immediately adjacent to small streams or low-lying areas. Graham woods consisted of a mixture of low-lying bottomland hardwood and upland mixed pine-hardwood forest, and contained a small network of streams. One of the two largest remnants, Lake Alice Conservation Area, consisted largely of upland mixed pine-hardwood forest, had some regenerating clear-cut habitat, and was adjacent to a large marsh, and therefore contained some flood-plain forest as well. The other large remnant, Biven's Forest, consisted of mostly bottomland-hardwood swamp in its interior, but was ringed by mixed pine-hardwood forest on three of its four edges, with its fourth edge being adjacent to a lake. Biven's Forest, Graham Woods, and portions of Lake Alice Conservation Area and Bartram-Carr Woods were subject to occasional flooding.

Sampling Methods

Avian sampling

To compare the use of edge versus interior locations by birds, I considered the first 40 m from the remnant boundary toward the interior as "edge", and all space beyond 40 m from the

boundary of the remnant I considered as "interior" (Fig 1-2). Edge habitat was within 40 m of remnant boundaries because most remnants were small (< 4 ha), and this width was similar to the 50 m width used by Fernández-Juricic (2001).

To survey birds, I used point counts randomly located at the edge and interior of each forest remnant. To assure some degree of equal sampling effort per forest remnant, I allocated a one point per 2 ha ratio, with a maximum of 10 points given to a remnant. I chose a 20 m radius point count sampling area for each point count location so that the diameter of the edge point sampling radius was completely contained within edge habitat. To ensure that interior counts were entirely enclosed within interior habitat, I selected all interior points between 60 m to approximately 100 m from the edge. To reduce the possibility of double counting, I designed points to be 140 m from each other, which, when the 20 m sampling radius is factored in, gives at least 100 m between sampling radii within remnants. However, because of remnant-size limitations and additional points added during the spring and fall, a couple of points were located less than 140 m apart; I sampled these on different days to eliminate the possibility of double counting.

The point count sampling technique used was similar to the technique used by Smith et al. (1993). For each of the four seasons, all birds that were heard and/or seen within a fixed, 20 m radius over a 10 min count, excluding fly-overs, were recorded. I conducted all counts in the first 3 ½ hours after sunrise. To reduce sampling bias due to time of day, I systematically rotated the time each point location was surveyed during each sampling morning. Because I wanted to capture as much diversity as possible, I varied sampling intensity and frequency per season. In particular, I increased sampling efforts during the spring and fall to account for anticipated increased migrant diversity during those seasons. During the winter (11/04–4/05), I conducted

counts at a total of 12 points twice a week, every other week, with the exception of 4 points in the Lake Alice Conservation Area, which I only sampled once a week. During the spring (4/05–5/05), I surveyed 32 points once a week. During the summer (5/05–8/05), I surveyed 20 points once a week, every other week. In the fall (9/05–11/05), I again surveyed 32 points once a week.

Vegetation sampling

To determine whether structural differences occurred, I conducted vegetation sampling at both edge and interior locations during both the growing (spring, summer, and fall) and dormant seasons (winter). Because woody stem density, tree density, and standing snag measures were considered perennial habitat features, these were only sampled during the dormant season. I carried out vegetation sampling only at point count locations used during the winter season. I did this because the winter had the least sampling intensity in terms of the number of point count locations sampled, and therefore was the most logistically practical set of points to collect data from. I sampled woody shrub stem density (≥ 1 m in height, < 8 cm dbh) on two, perpendicular, randomly assigned, 20 m transects running from the center of the point count sampling radii to the margin (James and Shugart 1970). Following modified procedures from Tilghman (1987a) and James and Shugart (1970), I randomly established four, 1 m^2 subplots within each point count sampling radii, and estimated several measures at each subplot. I counted woody shrub stems (< 8 cm dbh) to document shrubs less than 1m in height. I visually estimated ground cover for cover classes representing percentages of cover (including, 0%, >0–10%, 10–25%, 26–50%, 51–75%, > 75%) of bare ground, grass, dead debris, forbs, shrubs (woody or herbaceous), trees (woody stems >8 cm dbh), and vines. The proportion of occurrence (i.e., how many 1 m^2 subplots a cover class occurred in) of each cover class per cover variable was averaged over the four 1 m^2 subplots per 20 m point count sampling radii. I visually noted vertical vegetative structure for each type of vegetation that was at < 1 m in height, ≥ 1 m and < 5 m in height, and $\geq$

5m in height. I measured over-story canopy cover using a spherical densiometer. If there was a significant mid-story (< 5 m) that prevented reasonable sighting of the over-story canopy, then I used the location within 5 m of the point that presented the most un-obstructed view of the canopy was used. I observed canopy cover in all cardinal directions, and averaged it per 1 m^2 subplot. I measured visual obstruction between 0–2 m in height by recording the number of decimeters in each ½ m section of a marked sighting pole that were > 25% obstructed by vegetation. I placed the pole at the center of each 1 m^2 subplot and observed at a distance of 4 m, at a height of 1 m, and I observed it in each cardinal direction. I averaged these data per ½ m section, per 1 m^2 subplot (Robel, Briggs, Dayton, and Hurlbert 1970). I averaged all data collected at 1 m^2 subplots over all four subplots per 20 m radius plot. I measured the number of trees (> 8 cm dbh) and standing snags in a 10 m radius subplot, stemming from the center of each 20 m radius plot. I scaled all measures of shrub and tree density to densities per ha.

Analyses

Individual species

I analyzed bird count data for each season. I generated average daily relative abundances of birds for the edge and interior of each forest remnant by summing the total count data per species for the edge and interior point locations of a given forest remnant, and then dividing by the total number of survey days carried out at the edge and interior locations of that remnant. For example, if a remnant had two edge point count locations that were each sampled 5 days apiece, then I would sum the count data for a species for those two points, and divide by 10 (the total number of sample days for that remnant's edge) to produce the average daily relative abundance per that remnant edge. I removed one point at the edge of the Lake Alice Conservation Area from the analysis because it was inadvertently placed too far away from the edge. I entered data into a one-way ANOVA model blocked for forest remnant in which relative abundance was the

response variable and location (Edge or Interior) was treated as the independent variable. Prior to analysis, the data were checked for the assumptions of normality and equal-variance with Ryan-Joiner and Levine tests. Data were square-root if they did not meet the assumptions. Some species did not meet normality and equal-variance assumptions regardless of transformation, and I tested them with the non-parametric equivalent of the randomized block ANOVA, the Friedman test. Because of the effects on the relative abundance distribution of many zeros, and because I wanted to limit individual analysis to fairly well-represented species, I only individually analyzed species that occurred in at least 6 out of the possible 12 edge and interior areas across the 6 remnants. An alpha of 0.1 was used for all statistical tests.

Residency groups

I grouped species according to residency status per season, and average daily relative abundances of each residency group at the edge and interior of each forest remnant were calculated per season. Residency groups included year-round residents, winter migrants (those that only wintered in the Gainesville area), summer migrants (those that bred in the Gainesville area but migrated south), stop-over migrants (those that only use the Gainesville area as stop-over habitat during spring and fall migration), and all migrants. Residency status was assigned based on species information and range maps as reported in Poole (2005). To reduce the influence of under- and over-represented species, I further sub-grouped residency groups into three categories per season, including: only species that weren't abundant enough to be tested individually (uncommon group), only species that were abundant enough to be tested individually (common group), and all species combined. I analyzed all residency groups as described above for individual species. Due to the low number of occurrences of stop-over migrants during the spring and the fall seasons, I calculated a combined relative average daily abundance for stop-over migrants during those seasons. I analyzed them with the non-parametric

Friedman test, as they failed tests for non-normality and transformations were not effective (alpha = 0.1).

Vegetation sampling analysis

I analyzed measures of shrub and tree densities, canopy cover, and visual obstruction in each ½ m height section with the same ANOVA model as described for the bird analysis. Normality and equal variance assumptions were checked in a similar way. To analyze ground cover, I separately compared each cover class of each ground cover variable between edges and interiors (e.g., for grass, I compared the 25–50% cover class between remnant edges and interiors). To do this, I took the average proportion of occurrence of each cover class per cover variable that was calculated previously for each point count location, and calculated the average per remnant edge and interior. I then entered the data into the same ANOVA model previously described. Normality and equal variance assumptions were checked as described above, and non-normal distributions were tested with the non-parametric Friedman test. Due to an inconsistency in data collection during the growing season, I was unable to analyze the > 0–10%, and >10–25% cover classes for ground cover variables for that season. Vertical structure was analyzed both by individual vegetation and structure components, and by groups containing all vegetation (vegetation - dead debris), and all structure (vegetation + dead debris) in case the absence or presence of dead debris in overall vegetative structure was of significance. In a manner similar to Tilghman (1987a) and Karr (1968), I analyzed the vertical structure offered by individual structure components by considering the total of three layers to be an index of presence/absence between 0–300 for each structure component per point. I analyzed total vegetation structure per vertical section (< 1 m in height, ≥ 1m and < 5 m in height, and ≥ 5 m in height), and I calculated it for each sample point as an index between 0–500, with the presence/absence of each vegetation component representing 1/5 of the total index value, not including dead debris. I

20

analyzed total structure (vegetation + dead debris) in a similar way, but each vertical section was calculated out of an index between 0–600 because of the addition of dead debris to the analysis. I analyzed resultant index values for each category with the same ANOVA model as previously described in the bird analyses, and normality and equal variance assumptions were checked similarly. An alpha of 0.1 was used for all statistical tests.

Results

Birds

I observed a total of 77 species across all four seasons. A list of species detected in all four seasons, along with their residency status, their residency sub-group status, and their abbreviations can be found in Appendix A. During the winter, I observed a total of 45 species. Of 21 species common enough to be analyzed individually, 4 species had significantly higher relative abundances at edges than interiors, including the Carolina Chickadee, Cedar Waxwing, Blue Jay, and Northern Mockingbird (Table 1-1). With residency status categories, no group was shown to have significantly higher relative abundances at edges or interiors (Table 1-1).

During the spring, I observed a total of 42 species. Of 14 species common enough to be individually analyzed, Carolina Wren and Ruby-crowned Kinglet had significantly higher relative abundances at interiors than edges (Table 1-2). With residency status categories, the common summer migrants group was shown to have significantly higher relative abundances at edges than interiors (Table 1-2).

During the summer season, I observed a total of 31 species. Of 9 species common enough to be individually analyzed, no individual species had significantly different relative abundances between edges and interiors (Table 1-3). With residency status categories, both the year-round resident uncommon group and the year-round resident all species group were found to have significantly higher relative abundances at edges than interiors (Table 1-3). The year-round

21

resident uncommon group was made up of 17 species; Northern Mockingbird was the most widely represented member of the group and represented, on average, 24.4% and 23.4% of the cumulative edge and interior relative abundance of the group. To test for the effect of this individual species on the group, I re-analyzed the year-round resident uncommon group with Northern Mockingbird excluded. After this, the pattern of significantly higher relative abundances at edges was no longer present ($P = 0.126$), though the overall pattern still persisted for the group.

During the fall season, I observed a total of 42 species. Of 14 species common enough to be individually analyzed, 3 species were found to have significantly higher relative abundances at edges than interiors, including the Northern Mockingbird, the Red-bellied Woodpecker, and the Downy Woodpecker (Table 1-4). In addition, Eastern Tufted Titmouse had significantly higher relative abundances at interiors than edges. When I grouped species together according to various residency status categories, the groups that included all migrants, the migrant uncommon group, the migrant common group, and the summer migrant group were shown to have significantly higher relative abundances at interiors than edges (Table 1-4).

The uncommon migrant group was made up of 14 species: the combined abundances of Ruby-crowned Kinglet and Baltimore Oriole made up, on average, 25.0% and 50.3% of the cumulative edge and interior relative abundances of the group, respectively. To test for the effect of these individual species on the group, I reanalyzed the uncommon migrant group with Ruby-crowned Kinglet and Baltimore Oriole excluded. After this, the significant pattern of higher relative abundances at interiors was no longer present ($P = 0.46$). The migrant common group was made up of four species: the Gray Catbird made up, on average, 57.6% and 53.2% of the cumulative edge and interior abundances of the group, respectively. After reanalyzing the

migrant common group without the Gray Catbird, the pattern of significantly higher relative abundances at interiors was no longer present ($P = 0.215$); though the general pattern still persisted for the group. The summer migrant group is made up of 5 species: the Red-eyed Vireo contributed, on average, 60% and 66% of the cumulative relative abundance at edges and interiors of the group, respectively. With the Red-eyed Vireo removed a non-significant result was found for the summer migrant uncommon group ($P = 0.699$).

Lastly, the results of the non-parametric Friedman test for the combined spring/fall stop-over migrants showed no significant difference in the relative abundances of migrants between edges and interiors ($P = 0.414$). However, only 2 migrants (American Redstart and Prairie Warbler) of the 7 species recorded (Appendix A) occurred at edges, while all 7 recorded migrants occurred at interiors.

Vegetation

Analysis of average shrub stem density < 1 m and ≥ 1 m, canopy cover, visual obstruction, and density of trees and snags showed no significant differences in vegetation characteristics between edge and interior areas in either season. When I analyzed vertical structures during the winter, dead-debris was found to be significantly more present in the vertical strata in interiors than in edges during the winter. When I analyzed ground cover during the dormant season, there were a significantly greater occurrence of bare ground making up 25–50% of the ground cover at interiors than edges, and a significantly greater occurrence of grass making up < 10% of the ground cover on edges then interiors (Table 1-5). During the growing season, there was a significantly higher presence of vegetation < 1 m in height, and presence of shrubs in the vertical strata at interiors than edges. During the growing season, there was significantly greater occurrence of vines making up between 25–50% of the ground cover at interiors than edges (Table 1-6).

Discussion

Only a couple of bird species and a few bird groups used habitat edges significantly differently than interiors in any season. A possible reason that few species preferred edge or interior locations was that edges were largely not vegetatively different from interiors, and that remnants may have been viewed as "edge" habitat. This assertion is supported in a review study by McCollin (1998) that found small forest remnants to be predominantly edge habitat. In addition, Fernández-Juricic (2001) found a similar lack of vegetation structure differences between interior and edge habitats of urban remnants, though several bird species still differentiated in use between edges and interiors in that study. Likewise in my study, some species and groups of birds did differentiate between edge and interior habitats. I discuss possible reasons for this below.

Individual Species

The Blue Jay during the winter and the Northern Mockingbird during the winter and the fall exhibited significantly higher relative abundances at edge habitats. These are commonly observed species in urban habitats in Northern Florida and are often associated with habitat edges (Derrickson and Breitwisch 1992, Tarvin and Wolfenden 1999, Poole 2005). The Northern Mockingbird is highly associated with open habitat, apparently preferring very low grass or bare substrate to lunge at insects just above the ground (Breitwisch, Diaz and Lee 1987, Derrickson and Breitwisch 1992). Roth (1979) even suggested that too much cover inhibits foraging success for this species. Though edges in my study were not overly open as a rule, adjacent matrix was often open urban surfaces, such as maintained grass. In addition, analysis of vegetation showed that interior habitats during the non-dormant seasons had significantly higher representation of shrubs in the vertical strata, and higher vegetation representation < 1 m in height than edges, which may have negatively influenced the Northern Mockingbird. Further, Blue Jay is a

24

generalized forager, and it has often been observed foraging in urban lawns for insects (Tarvin and Wolfenden 1999). Both species take fruit as well (Poole 2005), and forest edges also typically have more fruiting plants than forest interiors (Noss 1991, McCollin 1998, Rodewald and Brittingham 2004).

The Cedar Waxwing was also shown to occur more often on edges during the winter. However, the flocking behavior of this species may be the cause of this result. Cedar Waxwing often occurs in large flocks in Florida during non-breeding seasons (Kale and Maeher 1990). In this study, flocks were not seen that often but when they were, a large number of individuals were recorded. They may have occurred in the interior and were not recorded because of limited observation time. Throughout the year, and especially during the winter, Cedar Waxwing is highly associated with the presence of fruit-bearing plants (Witmer 1996a , Witmer, Mountjoy, and Elliot 1997), and is commonly seen in urban matrices during the winter, feeding on the fruits and flowers of cultivated and/or ornamental shrubs (McPherson 1987, Witmer 1996b, Witmer et al. 1997). Though I did not survey remnant edges, interiors, or the urban matrix surrounding the forest remnants for fruit or fruiting species abundance, the presence of fruiting cultivated trees and bushes in the surrounding matrix may have influenced Cedar Waxwing to use edges more than interiors.

The Carolina Chickadee had higher relative abundances at edges than interiors during the winter and same for the Red-bellied Woodpecker and Downy Woodpecker during the fall. Unlike Northern Mockingbird and Blue Jay, which are generally considered edge species, Carolina Chickadee, Red-bellied Woodpecker, and Downy Woodpecker are generally considered to be edge-indifferent (Poole 2005), utilizing a variety of habitat types ranging from mature forests to urban parks throughout the year. For Carolina Chickadee, though, there is some

evidence that fruit becomes more important as a food source during the winter (Brewer 1963, Mostrom, Curry, and Lohr 2002). More fruiting bushes and trees, as well as possibly more invertebrates, may exist on edges during the winter (Noss 1991). Red-bellied woodpeckers utilize a very broad array of habitats and are generally arthropod-eaters (Shackelford, Brown, and Conner 2000). However, this species has been observed foraging on fruit trees in suburban habitats in south Florida at the same rate as on tree trucks for insects (Breitwisch 1977, Shackelford et al. 2000). If more fruits occurred on edges of these remnants, then the Red-bellied woodpeckers may have concentrated on edges during the fall. Therefore, determining fruit abundance in future studies may be important in determining habitat use patterns for these species.

Carolina Wren and Ruby-crowned Kinglet used interiors significantly more than edges during the spring, and Eastern Tufted Titmouse used interiors significantly more than edges during the fall. Greater occurrence of shrubs < 1 m in height and of shrubs in the vertical strata might have contributed to higher interior use by Carolina Wren, as it prefers dense shrub cover (Haggerty and Morton 1995). However, this pattern is curious because these species all typically use a wide variety of habitat types during these seasons (Poole et al. 2005), and Carolina Wren and Eastern Tufted Titmouse are both year-round resident species that did not exhibit a similar pattern in other seasons. This pattern may be explained by factors not considered in this study, and additional research is required to clarify it.

Residency Status

During the summer, year-round residents may use edges more than interiors, as evidenced by the groups of all year-round residents and uncommon year-round residents having significantly higher relative abundances at edges. However, this pattern was partially a result of the high counts of the Northern Mockingbird. Of the 9 species making up the summer common

group, only the Northern Parula is not considered an edge-preferring or edge-indifferent species (Poole et al. 2005). This is consistent with Dunford and Freemark (2004), who found that many resident breeding birds in suburban habitat in Canada were generally edge, or edge-indifferent species. In addition, though there was no significant difference for common year-round residents, the relative abundance at edges for this group was higher than the relative abundance on interiors. Some vegetative differences occurred between edge and interior areas; edges had significantly lower vegetation < 1 m in height and lower amounts of shrubs in the vertical strata than interior locations. Edges were closer to the more open, sunnier matrix than interior locations. So, while increased amounts of understory cover at interiors may have provided predator protection, the condition and matrix-adjacent position of edges may have created more opportunities for foraging on fruiting plants and invertebrates. In addition, these species are probably tolerant of human disturbance, as increased human presence and increased traffic noise on edges has been shown to influence habitat use by birds (Reijnen, Foppen, and Veenbaas 1997, Fernández -Juricic and Tellaria 2000, Brontos and Horrondo 2001, Fernández-Juricic 2001).

During the fall, migrants may use interiors more than edges, as evidenced by the higher relative abundances at interiors for the all migrant group, the uncommon migrant group, the common migrant group, and the summer migrant group. However, the common migrant group result is partially driven by the relative abundance contributed by Gray Catbird. In addition, the uncommon migrant group is partially driven by the relative abundances contributed by Ruby-crowned Kinglet and Baltimore Oriole. Lastly, the summer migrant group result is partially driven by the relative abundances contributed by the Red-eyed Vireo.

Despite the ambiguities mentioned above, a reason to consider the biological relevancy of the findings is that 14 of the 18 total migrant species during the fall have relative abundances that

are at least slightly higher at interiors than edges. This suggests that an underlying ecological cause might be present that drives migrant species to use interiors more, or avoid edges of these urban forest remnants. There are several studies that show that wintering birds and fall migrants use a variety of habitat types, including urban/suburban sites (Tilghman 1987b, Winker, Warner, and Weisbrod 1992, Yaukey 1996, Rodewald and Brittingham 2002, Rodewald and Brittingham 2004, Atchison and Rodewald 2006). In my study, migrants may select foraging and resting areas away from edges of urban forest remnants. The patterns of higher vegetation in the under-story (< 1 m) and a higher representation of shrubs in the vertical strata at interiors than edges may be contributing factors, as these structural features provide better cover from predators. In addition, remnant interiors may provide protection from human disturbances near edges.

Combined Fall and Spring Migrants

No difference in edge and interior habitat use by stop-over migrants was demonstrated, but the analyses were limited because of few observations of these birds. Similar to a previous avian study in Gainesville (Hostetler et al. 2005), I observed few occurrences of stop-over migrants in the urban forest remnants. It was noted, though, that only 2 of 7 species occurred at edges and that all 7 migrants occurred at interiors. Two of those species, Acadian Flycatcher and Blackpoll Warbler, are generally considered interior forest birds (Poole 2005) and the others are edge-indifferent or edge preferring. Given the limited nature of the data, an underlying trend of edge avoidance by stop-over migrants is problematic, but it warrants further study. I found that the vegetation structure at edges and interiors during the spring and fall were only different in a few ways (e.g., significantly greater presence of vegetation < 1 m in height, and significantly greater vertical presence of shrubs in general on interiors). These differences could have lent to better protection from predators on interiors than edges, which Moore et al. (1995) mentions as being an important feature of stop-over habitats. Stop-over habitat is essential for migrating birds to

replenish fat supplies, to rest before the next leg of migration, and to provide protection against predators (Moore et al. 1995). Urban green space can serve as important stop-over habitat because these spaces may serve as more productive habitat, or as the only stop-over habitats for birds along some migratory routes (Moore et al. 1995).

Summary and Conclusions

Most results indicated a lack of differentiation by birds between edges and interiors, possibly due to vegetative similarities between edges and interiors. However, results indicated that year-round residents as a group may use edges more than interiors during the summer and that some migrants might use interiors more than edges during fall migration. However, because of the dominance of one or a few species in each group, it is not clear whether these patterns of the greater use of edge and interior habitats during the summer and fall are biologically relevant. Despite this, the data indicates that certain birds may differentially use interior areas (greater than 40 m from an edge) and/or edges (less than 40 m from an edge) of small urban forest remnants, and that this pattern of habitat-use may be modified by season. These results suggest that the interiors of these urban forest remnants may be managed for migrant species during the fall, possibly by reducing human disturbance in the interior. Lastly, the study results point out that urban forest remnants are used by a number of different species throughout the year, and that their conservation contributes toward the diversity of the surrounding urban environment.
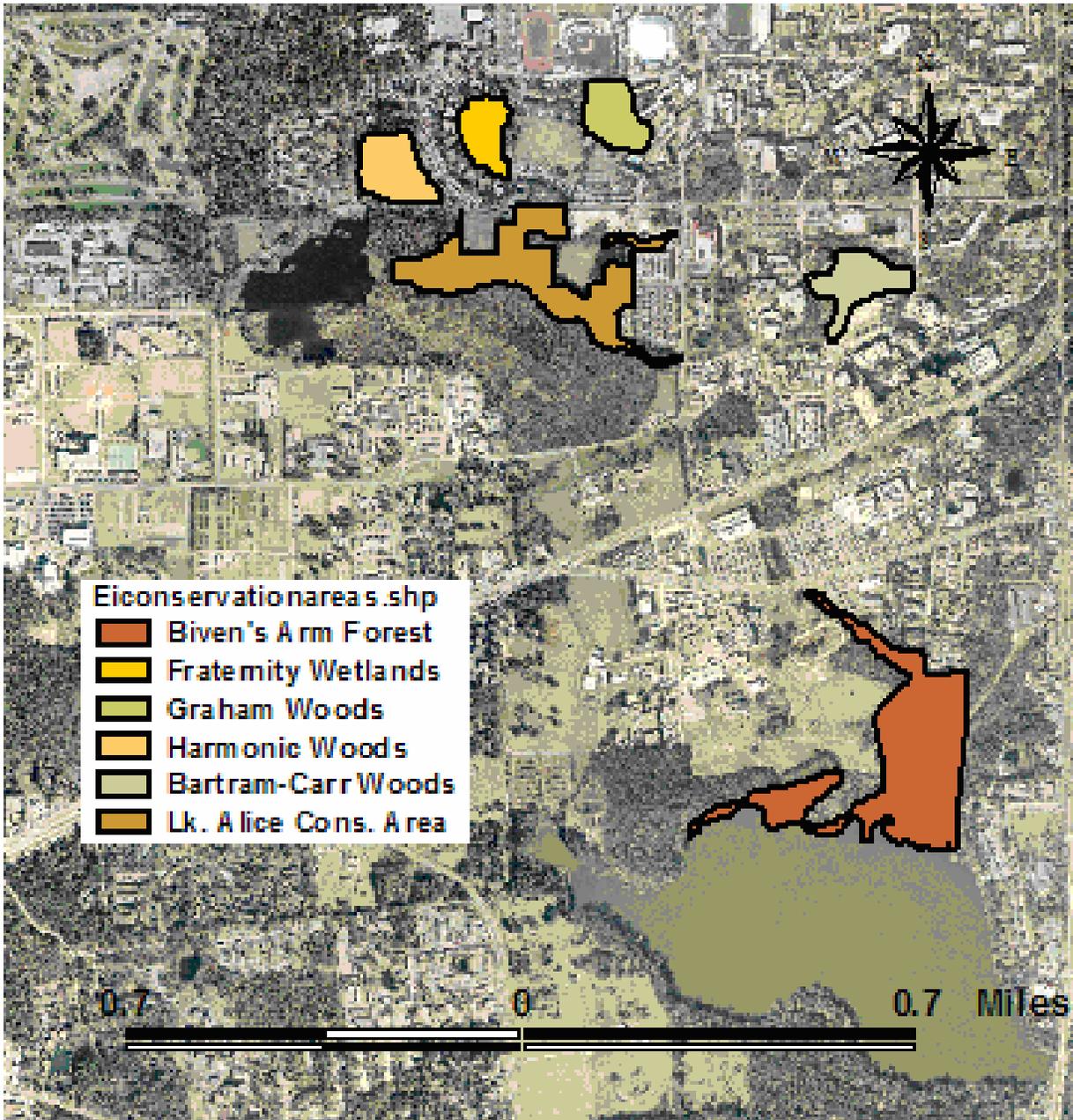
Figure 1-1. Urban forest remnants on the University Florida Campus in Gainesville, Florida.
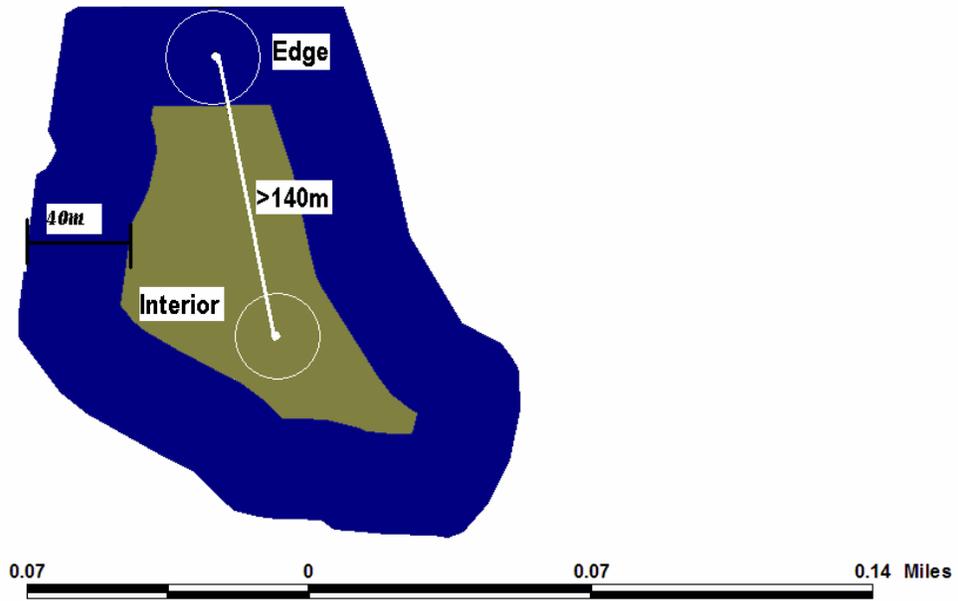
Figure 1-2. Illustration of edge and interior point count locations for bird surveys within forest remnants, in Gainesville, Florida. Edge was defined as the habitat < 40m from the remnant boundary. Interior was defined as all habitat > 40m from the remnant boundary.

Table 1-1. Individual species and residency groups analyzed for winter bird count surveys of edge and interior locations within forest remnants in Gainesville Florida. Shown are overall average daily relative abundances with accompanying standard error (SE) values at edges and interiors, and test statistics (T.S.) results with accompanying *P*-values. Unless noted, statistical test is one way ANOVA. For all tests, df = 1, and n = 6 for edge and interior areas. For species abbreviations, see appendix A. Residency codes: WR=winter resident, SM=summer migrant, AM=all migrants, and YR=year-round residents.

Subgroup	Number of species per subgroup	Residency Group/Species	Edge		SE	Interior		SE	T.S.	*P*
All Species	15	WR	2.86	±	0.41	2.63	±	0.67	0.09	0.781
	18	AM	2.89	±	0.41	3.11	±	0.45	0.17	0.697
	28	YR**	3.47	±	0.68	2.48	±	0.41	0.67	0.414
Common	10	WR	2.78	±	0.38	2.95	±	0.46	0.12	0.247
	11	YR*	1.72	±	0.11	1.49	±	0.14	1.72	0.743
Uncommon	5	WR	0.090	±	0.054	0.095	±	0.016	3.25	0.105
	2	SM	0.013	±	0.009	0.096	±	0.054	0.19	0.670
	8	AM**	0.111	±	0.054	0.112	±	0.040	2.67	0.102
	17	YR**	0.784	±	0.394	0.399	±	0.075	0.67	0.414
		AMCR	0.04	±	0.02	0.04	±	0.02	0.00	0.958
		AMGO**	0.24	±	0.15	0.55	±	0.36	0.00	1.000
		AMRO**	0.43	±	0.20	0.57	±	0.43	0.67	0.414
		BAWW	0.04	±	0.02	0.09	±	0.04	0.62	0.467
		BGGC**	0.19	±	0.08	0.17	±	0.03	2.67	0.102
		BLJA	0.26	±	0.06	0.12	±	0.04	4.64	0.084
		CACH**	0.16	±	0.05	0.01	±	0.01	6.00	0.014
		CARW	0.44	±	0.10	0.54	±	0.18	0.15	0.718
		CEWA**	0.21	±	0.08	0.00	±	0.00	6.00	0.014
		DOWO	0.06	±	0.02	0.05	±	0.01	0.13	0.734
		EAPH	0.06	±	0.03	0.05	±	0.02	0.10	0.769
		ETTI	0.26	±	0.06	0.20	±	0.07	0.53	0.499
		GRCA**	0.11	±	0.03	0.12	±	0.06	0.00	1.000
		MODO*	0.19	±	0.09	0.04	±	0.02	2.25	0.194
		NOCA	0.81	±	0.11	0.85	±	0.15	0.04	0.845
		NOMO	0.25	±	0.07	0.06	±	0.03	4.83	0.079
		PAWA	0.07	±	0.04	0.06	±	0.03	0.53	0.508
		RBWO	0.32	±	0.11	0.23	±	0.06	0.55	0.490
		RCKI**	0.76	±	0.10	0.59	±	0.08	0.67	0.414
		YBSA	0.03	±	0.02	0.05	±	0.02	0.45	0.534
		YRWA*	0.82	±	0.21	0.88	±	0.17	0.38	0.567

*square-root transformed
**tested with non-parametric Friedman test.

Table 1-2. Individual species and residency groups analyzed for spring bird count surveys of edge and interior locations within forest remnants in Gainesville Florida. Shown are overall average daily relative abundances with accompanying standard error (SE) values at edges and interiors, and test statistics (T.S.) results with accompanying *P*-values. Unless noted, statistical test is one way ANOVA. For all tests, df = 1, and n = 6 for edge and interior areas. For species abbreviations, see appendix A. Residency codes: WR=winter resident, SM=summer migrant, AM=all migrants, and YR=year-round residents.

Subgroup	Number of species per subgroup	Residency group/Species	Edge	±	SE	Interior	±	SE	T.S	*P*
All Species	4	SM	0.63	±	0.20	0.49	±	0.09	0.50	0.513
	13	WR**	1.04	±	0.20	1.20	±	0.29	0.67	0.414
	21	AM	1.70	±	0.18	1.73	±	0.22	0.09	0.778
	21	YR	3.16	±	0.54	3.64	±	0.57	0.40	0.557
Common	3	SM*	0.62	±	0.14	0.47	±	0.31	5.04	0.075
	3	WR*	0.75	±	0.08	1.01	±	0.21	0.95	0.375
	6	AM**	1.37	±	0.13	1.48	±	0.42	0.20	0.655
	7	YR	2.51	±	0.43	3.16	±	0.43	1.27	0.312
Uncommon	10	WR*	0.29	±	0.15	0.19	±	0.07	0.14	0.721
	15	AM*	0.33	±	0.14	0.28	±	0.08	0.04	0.855
	14	YR	0.65	±	0.20	0.47	±	0.15	0.63	0.464
		BHCO*	0.19	±	0.10	0.25	±	0.11	2.10	0.207
		BLJA*	0.24	±	0.11	0.10	±	0.04	1.34	0.299
		CARW*	0.42	±	0.06	0.83	±	0.25	5.77	0.061
		DOWO**	0.08	±	0.03	0.13	±	0.09	0.00	1.000
		ETTI**	0.21	±	0.09	0.09	±	0.09	2.67	0.102
		GCFL	0.35	±	0.05	0.22	±	0.08	1.97	0.219
		GRCA	0.25	±	0.09	0.26	±	0.10	0.00	0.994
		MODO**	0.08	±	0.02	0.06	±	0.04	0.33	0.564
		NOCA**	1.11	±	0.16	1.34	±	0.27	0.20	0.655
		NOPA**	0.16	±	0.08	0.15	±	0.08	0.17	0.564
		RBWO**	0.42	±	0.14	0.47	±	0.08	0.67	0.414
		RCKI*	0.31	±	0.08	0.63	±	0.15	5.03	0.075
		REVI*	0.12	±	0.07	0.09	±	0.04	0.00	0.949
		YRWA	0.19	±	0.07	0.11	±	0.06	0.50	0.511

*square-root transformed
**tested with non-parametric Friedman test.

Table 1-3. Individual species and residency groups analyzed for summer bird count surveys of edge and interior locations within forest remnants in Gainesville, Florida. Shown are overall average daily relative abundances with accompanying standard error (SE) values at edges and interiors, and test statistics (T.S.) results with accompanying *P*-values. Unless noted, statistical test is one way ANOVA. For all tests, df = 1, and n = 6 for edge and interior areas. For species abbreviations, see Appendix A. Residency codes: WR=winter resident, SM=summer migrant, AM=all migrants, and YR=year-round residents.

Subgroup	Number of species per subgroup	Residency group/Species	Edge		SE	Interior		SE	T.S	*P*
All Species	3	SM	0.45	±	0.11	0.34	±	0.11	0.44	0.535
	7	AM*	0.48	±	0.14	0.37	±	0.10	0.34	0.584
	24	YR	4.77	±	0.57	3.60	±	0.60	5.46	0.067
Common	2	SM	0.40	±	0.11	0.31	±	0.12	0.24	0.645
	7	YR	3.59	±	0.28	3.24	±	0.54	0.35	0.580
Uncommon	5	AM	0.08	±	0.04	0.05	±	0.03	0.32	0.598
	17	YR	1.18	±	0.52	0.36	±	0.09	8.11	0.036
		BHCO*	0.08	±	0.05	0.15	±	0.09	0.33	0.593
		BLJA*	0.55	±	0.22	0.14	±	0.07	4.00	0.102
		CARW**	1.30	±	0.33	1.25	±	0.21	0.67	0.414
		DOWO**	0.21	±	0.04	0.06	±	0.05	2.67	0.102
		ETTI	0.18	±	0.06	0.21	±	0.10	0.07	0.809
		GCFL	0.35	±	0.10	0.25	±	0.09	0.42	0.543
		NOCA	0.78	±	0.20	1.11	±	0.24	0.71	0.437
		NOPA*	0.05	±	0.02	0.07	±	0.05	0.02	0.891
		RBWO	0.49	±	0.12	0.32	±	0.07	2.67	0.163

*square-root transformed
**tested with non-parametric Friedman test.

Table 1-4. Individual species and residency groups analyzed for fall bird count surveys of edge and interior locations within forest remnants in Gainesville, Florida. Shown are overall average daily relative abundances with accompanying standard error (SE) values at edges and interiors, and test statistics (T.S.) results with accompanying *P*-values. Unless noted, statistical test is one way ANOVA. For all tests, df = 1, and n = 6 for edge and interior areas. For species abbreviation, see appendix A. Residency codes: WR=winter resident, SM=summer migrant, AM=all migrants, and YR=year-round residents.

Subgroup	Number of species per subgroup	Residency group/Species	Edge		SE	Interior		SE	T.S	*P*
All Species	5	SM**	0.08	±	0.02	0.21	±	0.08	6.31	0.054
	8	WR	0.31	±	0.10	0.58	±	0.15	3.60	0.116
	18	AM	0.47	±	0.15	0.86	±	0.19	8.04	0.036
	26	YR	5.51	±	0.61	4.39	±	0.63	2.67	0.102
Common	2	WR	0.23	±	0.09	0.43	±	0.13	2.24	0.195
	3	AM	0.33	±	0.12	0.62	±	0.15	4.78	0.080
	9	YR*	4.80	±	0.46	3.92	±	0.55	2.10	0.207
Uncommon	4	SM	0.05	±	0.03	0.06	±	0.02	0.17	0.699
	6	WR*	0.08	±	0.04	0.14	±	0.05	3.99	0.102
	14	AM*	0.14	±	0.04	0.25	±	0.08	14.01	0.013
	17	YR	0.70	±	0.21	0.47	±	0.10	1.28	0.309
		AMRE**	0.07	±	0.03	0.03	±	0.01	0.20	0.655
		BAWW**	0.03	±	0.02	0.06	±	0.03	1.00	0.317
		BLJA*	0.55	±	0.13	0.37	±	0.07	2.48	0.176
		CARW	1.14	±	0.19	1.18	±	0.28	0.04	0.843
		DOWO	0.22	±	0.05	0.07	±	0.03	4.06	0.100
		ETTI	0.11	±	0.03	0.28	±	0.07	5.68	0.063
		GRCA	0.20	±	0.08	0.37	±	0.12	1.60	0.261
		MODO	0.05	±	0.03	0.06	±	0.03	0.03	0.867
		NOCA	1.63	±	0.26	1.42	±	0.20	0.38	0.566
		NOMO*	0.37	±	0.12	0.09	±	0.05	21.97	0.005
		RBWO	0.53	±	0.08	0.29	±	0.06	10.43	0.023
		REVI**	0.03	±	0.02	0.15	±	0.06	1.80	0.180
		WEVI**	0.19	±	0.06	0.16	±	0.05	1.80	0.180

 *square-root transformed
**tested with non-parametric Friedman test

Table 1-5. Vegetation analysis results for edge and interior locations of urban forest remnants during the dormant season in Gainesville, Florida. Shown are overall averages and accompanied SE values for both edge and interior locations for each listed variable measured, and (T.S.) test statistics and accompanying P-values. Unless noted, statistical test is one way ANOVA. For all tests, df = 1, and n = 6 for edge and interior areas.

Variable	Sub-variable	Edge		SE	Interior		SE	T.S.	P
Woody shrub	0-2.5 cm dbh*	108333.33	±	72029.12	125208.33	±	48457.90	0.16	0.263
density per ha	2.5-8 cm dbh*	2500.00	±	1118.03	3750.00	±	1796.99	0.26	0.630
	< 8 cm dbh, > 1 m in height*	6020.83	±	1841.41	5885.42	±	1143.42	0.11	0.754
Trees per hectare		381.25	±	96.16	312.50	±	107.24	0.02	0.883
Snags per hectare		22.92	±	12.25	35.42	±	14.58	0.29	0.611
Visual Obstruction	0-0.5 m**	3.22	±	0.47	2.80	±	0.59	2.67	0.102
	0.5–1 m	2.55	±	0.53	2.04	±	0.50	0.03	0.875
	1–1.5 m	1.84	±	0.42	1.48	±	0.42	0.00	1.000
	1.5–2.0 m	1.53	±	0.45	1.57	±	0.46	0.79	0.416
Overstory Density		73.53	±	6.01	76.28	±	6.93	0.14	0.727
Index of vertical	Dead Debris	120.83	±	11.49	175.00	±	17.38	10.97	0.021
vegetation structure	Forbes	89.58	±	6.78	85.42	±	6.78	0.29	0.611
(0-300)	Grass	56.25	±	15.73	33.33	±	17.87	0.89	0.388
	Shrubs	125.00	±	29.76	160.42	±	13.85	1.09	0.344
	Trees	95.83	±	19.81	87.50	±	24.79	0.16	0.709
	Vines*	83.33	±	22.05	129.17	±	32.54	0.86	0.396
Index of vertical	All vegetation (< 1 m)	272.92	±	27.08	243.75	±	32.56	1.52	0.272
vegetation structure	All vegetation (>1 m, ≤ 5 m)	106.25	±	34.12	166.67	±	17.87	2.11	0.206
(0-500)	All vegetation (> 5 m)	70.83	±	18.73	85.42	±	14.58	0.38	0.567
Index of vertical	All structure variables (< 1 m)	364.58	±	28.94	343.75	±	32.56	0.78	0.419
vegetation structure	All structure variables (> 1 m, ≤ 5 m)	131.25	±	43.27	225.00	±	22.36	3.46	0.122
(0-600)	All structure variables (> 5 m)	75.00	±	20.67	102.08	±	13.85	1.40	0.290
Distribution of ground	Bare Ground (0%)*	79.17	±	9.50	58.33	±	12.36	3.29	0.129
cover	Bare Ground (<10%)	16.67	±	7.68	33.33	±	13.57	2.39	0.183
	Bare Ground (10–25%)**	0.00	±	0.00	14.58	±	8.18	3	0.083
	Bare Ground (25–50%)**	0.00	±	0.00	2.08	±	2.08	1	0.317
	Bare Ground (50–75%)**	4.17	±	4.17	0.00	±	0.00	1	0.317
	Bare Ground (75-100%)**	0.00	±	0.00	8.33	±	8.33	1	0.317
	Dead Debris (0%)**	8.33	±	5.27	4.17	±	4.17	1	0.317
	Dead Debris (< 10%)**	8.33	±	5.27	6.25	±	4.27	0	1.000
	Dead Debris (10-25%)	14.58	±	8.18	18.75	±	6.25	0.29	0.611
	Dead Debris (25-50%)	20.83	±	6.97	27.08	±	8.18	0.32	0.597
	Dead Debris (50–75%)	27.08	±	11.37	16.67	±	8.33	0.39	0.558
	Dead Debris (75–100%)	20.83	±	7.68	35.42	±	9.36	1.24	0.315
	Forbes (0%)**	10.42	±	6.78	18.75	±	8.98	0.33	0.564
	Forbes (< 10%)	50.00	±	7.22	62.50	±	12.08	1.36	0.296
	Forbes (10–25%)	18.75	±	7.74	10.42	±	6.78	0.62	0.465
	Forbes (25–50%)**	10.42	±	5.02	6.25	±	4.27	0.33	0.564
	Forbes (50–75%)**	2.08	±	2.08	6.25	±	4.27	1	0.317
	Forbes (75–100%)**	8.33	±	8.33	4.17	±	4.17	0	1.000
	Grass (0%)	43.75	±	15.73	75.00	±	12.91	2.95	0.146
	Grass (< 10%)**	18.75	±	8.98	4.17	±	4.17	3	0.083

Table 1-5. Continued

Variable	Sub-variable	Edge		SE	Interior		SE	T.S.	P
	Grass (10–25%)**	10.42	±	8.18	8.33	±	5.27	0.33	0.564
	Grass (25–50%)	12.50	±	4.56	10.42	±	5.02	0.09	0.771
	Grass (50–75%)**	6.25	±	4.27	2.08	±	2.08	1	0.317
	Grass (75–100%)**	8.33	±	8.33	0.00	±	0.00	1	0.317
	Shrubs (0%)	31.25	±	16.38	18.75	±	6.25	0.79	0.415
	Shrubs (< 10%)	22.92	±	11.37	25.00	±	6.45	0.03	0.880
	Shrubs (10–25%)	16.67	±	8.33	25.00	±	6.45	0.36	0.576
	Shrubs (25–50%)	22.92	±	11.37	10.42	±	5.02	1.25	0.314
	Shrubs (5–75%)**	6.25	±	4.27	8.33	±	5.27	0	1.000
	Shrubs (75–100%)**	0.00	±	0.00	4.17	±	4.17	1	0.317
	Trees (0%)**	87.50	±	5.59	77.08	±	15.95	0.33	0.564
	Trees (< 10%)**	0.00	±	0.00	4.17	±	4.17	1	0.317
	Trees (10–25%)**	12.50	±	5.59	4.17	±	4.17	2	0.157
	Trees (25–50%)**	0.00	±	0.00	0.00	±	0.00	0	1.000
	Trees (50–75%)**	0.00	±	0.00	2.08	±	2.08	1	0.317
	Trees (75–100%)**	0.00	±	0.00	0.00	±	0.00	0	1.000
	Vines (0%)	54.17	±	16.67	29.17	±	16.35	0.92	0.381
	Vines (> 10%)	22.92	±	8.18	45.83	±	18.73	0.48	0.520
	Vines (10–25%)	22.92	±	9.36	8.33	±	5.27	1.24	0.315
	Vines (25–50%)**	0.00	±	0.00	4.17	±	4.17	1	0.317
	Vines (50–75%)**	0.00	±	0.00	0.00	±	0.00	0	1.000
	Vines (75–100%)**	0.00	±	0.00	0.00	±	0.00	0	1.000

*square-root transformed
**tested with non-parametric Friedman test

Table 1-6. Vegetation analysis results for edge and interior locations of urban forest remnants during the growing seasons in Gainesville, Florida. Includes the overall average and accompanied SE values for both edge and interior locations for each listed variable measured, and test statistics (T.S.) and associated *P*-values. Unless noted, statistical test is one way ANOVA. For all tests, df = 1, and n = 6 for edge and interior areas.

Variable	Sub-variable	Edge		SE	Interior		SE	T.S.	P
Overstory Density		72.42	±	10.87	85.01	±	5.96	1.04	0.354
Visual Obstruction	0–0.5 m*	4.89	±	1.53	3.76	±	0.34	0.18	0.690
	0.5–1 m*	4.03	±	1.47	3.04	±	0.41	0.01	0.933
	1–1.5 m	3.85	±	1.46	2.54	±	0.36	0.97	0.371
	1.5–2.0 m**	3.51	±	1.31	2.58	±	0.24	0.67	0.414
Index of vertical	Dead Debris	119.44	±	20.91	104.17	±	10.54	0.42	0.545
vegetation structure	Forbes	51.39	±	13.89	66.67	±	10.54	1.45	0.282
(0-300)	Grass	39.58	±	15.95	56.25	±	15.05	1.04	0.355
	Shrubs	136.11	±	24.69	179.17	±	11.93	5.04	0.075
	Trees	111.11	±	18.31	125.00	±	14.43	0.59	0.478
	Vines	140.28	±	30.69	141.67	±	32.06	0.00	0.977
Index of vertical	All vegetation (< 1 m)	218.75	±	27.34	291.67	±	30.73	4.54	0.086
vegetation structure	All vegetation (> 1, ≤ 5 m)	172.92	±	29.30	170.83	±	11.93	0.01	0.934
(0-500)	All vegetation (> 5 m)	86.81	±	18.83	106.25	±	11.06	0.84	0.402
Index of vertical	All structure variables (<1 m)	302.08	±	37.97	379.17	±	35.01	2.74	0.159
vegetation structure	All structure variables (> 1 m, ≤ 5 m)	206.94	±	36.79	187.50	±	11.18	0.43	0.540
(0-600)	All structure variables (> 5 m)	88.89	±	18.96	106.25	±	11.06	0.63	0.465
Distribution of	Bare Ground (0%)	54.17	±	13.94	42.36	±	12.19	0.95	0.374
ground cover	Bare Ground (25–50%)**	8.33	±	8.33	10.42	±	8.18	1.00	0.317
	Bare Ground (50–75%)**	0.00	±	0.00	0.00	±	0.00	0.00	1.000
	Bare Ground (75–100%)**	0.00	±	0.00	0.00	±	0.00	0.00	1.000
	Dead Debris (0%)**	12.50	±	12.50	4.17	±	4.17	0.00	1.000
	Dead Debris (25–50%)	18.75	±	8.98	31.94	±	11.47	1.31	0.303
	Dead Debris (50–75%)	16.67	±	6.18	20.83	±	7.68	0.25	0.638
	Dead Debris (75–100%)	25.00	±	15.81	15.97	±	5.73	0.21	0.663
	Forbes (0%)	45.83	±	13.57	22.22	±	12.11	3.05	0.141
	Forbes (25–50%)**	2.08	±	2.08	8.33	±	5.27	2.00	0.157
	Forbes (50–75%)**	10.42	±	6.78	2.08	±	2.08	2.00	0.157
	Forbes (75–100%)**	0.00	±	0.00	2.08	±	2.08	1.00	0.317
	Grass (0%)	62.50	±	17.97	39.58	±	14.50	3.25	0.131
	Grass (25–50%)**	4.17	±	4.17	4.17	±	4.17	0.00	1.000
	Grass (50–75%)**	4.17	±	4.17	0.00	±	0.00	1.00	0.317
	Grass (75-100%)**	18.75	±	16.38	5.56	±	5.56	2.00	0.157
	Shrubs (0%)	35.42	±	17.20	11.81	±	5.93	2.02	0.215
	Shrubs (25–50%)	8.33	±	5.27	24.31	±	8.36	2.20	0.199
	Shrubs (50–75%)	12.50	±	6.45	12.50	±	6.45	0.00	1.000
	Shrubs (75–100%)**	2.08	±	2.08	4.17	±	2.64	1.00	0.317
	Trees (0%)	91.67	±	8.33	74.31	±	11.20	2.65	0.165
	Trees (25–50%)**	8.33	±	8.33	8.33	±	8.33	0.00	1.000
	Trees (50–75%)**	0.00	±	0.00	0.00	±	0.00	0.00	1.000
	Trees (75–100%)**	0.00	±	0.00	0.00	±	0.00	0.00	1.000
	Vines (0%)	27.08	±	15.28	25.00	±	15.81	0.01	0.939
	Vines (25–50%)**	4.17	±	4.17	25.69	±	9.65	4.00	0.046

38

Table 1-6. Continued

Variable	Sub-variable	Edge		SE	Interior		SE	T.S.	*P*
	Vines (50–75%)**	2.08	±	2.08	0.00	±	0.00	1.00	0.317
	Vines (75–100%)**	8.33	±	6.18	5.56	±	5.56	0.33	0.564

*square-root transformed
**tested with non-parametric Friedman test

CHAPTER 2
USE OF EDGE AND INTERIOR HABITATS OF URBAN FOREST REMNANTS BY
HERPETOFAUNA

Introduction

Reptiles and amphibians face numerous challenges in coexisting with an urbanizing world
(Rubbo and Kiesecker 2004, McKinney 2006). Research has shown that urbanization can
negatively affect herpetofauna because of the increased amount of impervious surfaces (Richter
and Azous 1995), habitat isolation caused by dispersal barriers such as roads (Ficetola and De
Bernardi 2004, Parris 2006), the degradation of wetlands through the destruction of habitat, and
the alteration of hydroperiod and flow regimes (Delis, Mushinsky, and McCoy 1996, Riley et al.
2005). For amphibians, the effect of urbanization has been paid particular attention because of
their need for access to water to breed in. Both reptiles and amphibians are at risk from habitat
fragmentation and other anthropogenic threats on a global scale (Gibbons et al. 2000), with the
IUCN estimating that 1/3 of herpetofaunal species worldwide are threatened with extinction
(Baillie, Hilton-Taylor, and Stuart 2004, and Cushman 2006).

Urban and Edge Effects on Herpetofauna

Urbanization often causes habitat fragmentation, and reptiles and amphibians can persist
within forest remnants (Demayandier and Hunter 1998, Schlaepfer and Gavin 2001), including
habitat remnants within urban matrices (Enge, Robson, and Krysko 2004, Ficetola and De
Bernardi 2004, Rubbo and Kiesecker 2005, Parris 2006). Habitat fragmentation creates a higher
amount of edge habitat in relation to the amount of interior habitat. From a habitat use
standpoint, this is important because habitat edges are often used differently than habitat
interiors. Indeed, edges have long been recognized for often having higher diversities and higher
abundances of species than habitat interiors, particularly of game species and birds (Lay 1938,
Yahner 1988). This pattern is partially due to factors such as increased sunlight exposure,

increased emergent vegetation, and increased abundances of invertebrates along edges. However, for herpetofauna, particularly amphibians, interior habitats generally offer cooler, moister conditions, and therefore may be more conducive to survival, particular during dry periods (Lehtinen, Ramanamanjato, and Raveloarison 2005).

Research comparing edge and interior use of forest remnants has shown that herpetofauna can respond to edge differentially, and may partition their species assemblages into edge-associated, interior-associated, and edge-indifferent species (Schlaepfer and Garvin 2001, Urbina-Cardona, Olivares-Perez, and Reynoso 2006, Lehtinen et al. 2005). These findings have varied depending upon the ecological system that was studied, as well as the season it was studied in. For example, Lehtinen et al. (2005) and Schlaepfer and Garvin (2001) found herpetofauna to differentially use edges and interiors of remnants within desert and pasture matrices, respectively, but that these results were highly dependent upon season. Contrastingly, Urbina-Cardona et al. (2006) found differential habitat use by herpetofauna at edges and interiors in remnants within another pasture matrix, but found that these effects were largely not influenced by season. Currently, very little is known about whether individual species, or taxa-groups avoid edges and preferentially utilize interior areas of urban forest remnants.

Objective

The objective of my study was to determine whether species and taxa-groups of amphibians and reptiles differentially use edge and interior habitat within urban forest remnants during the summer.

Methods

Study Site

This study took place in 5 forest remnants on the University of Florida campus, located in Gainesville, Florida. They included Harmonic Woods (3.7 ha), Graham Woods (3.0 ha),

Bartram-Carr Woods (3.5 ha), Lake Alice Conservation Area (11.3 ha), and Biven's Arm Forest (16.6 ha) (Figure 2-1). The University of Florida Gainesville campus is located in north-central peninsular Florida, which experiences hot, humid, and generally rainy summers. Two of the three smallest remnants (Harmonic Woods and Bartram-Carr Woods) included largely upland mixed pine-hardwood forest habitat, with all containing or being immediately adjacent to small streams or low-lying areas. The third small patch (Graham Woods) consisted of a mixture of low-lying bottomland hardwood and upland mixed pine-hardwood habitat, and contained a small network of streams. One of the two largest remnants, Lake Alice Conservation Area, consisted of upland mixed pine-hardwood forest, some regenerating clear-cut habitat, and was adjacent to a large marsh (25 ha), and therefore contained some flood-plain forest as well. The other large remnant, Biven's Forest, consisted of mostly bottomland-hardwood swamp in its interior, but was ringed by mixed pine-hardwood forest on three of its four edges, with the fourth edge being adjacent to a lake. All remnants except Harmonic Woods were subject to occasional flooding.

Herpetofaunal Sampling

I sampled Herpetofauna during the summers of 2005 and 2006 from May until August, using drift fence arrays with pitfall traps and funnel traps, along with Poly Vinyl Chloride (PVC) pipe refugia to sample for tree frogs. I made drift fences out of approximately 30 cm wide silt fencing (Enge 1997). Following a modification of a design by Moseley, Castleberry, and Schweitzer (2003), I formed arrays in the shape of a Y, with the three, 7.6 m long wings conjoined around a single pitfall trap, and placed at 120° angles to each other. I placed funnel traps at the distal ends of each wing, making sure they were flush to the ground (Johnson, S., Personal communication). I made pitfall traps of 19.1 L plastic buckets. For funnel traps, I used a modification of the format described by Enge (1997), using aluminum window screening approximately 76 cm in length to build cylindrical traps of the same length with a funnel in one

end, and with the other end closed. To prevent desiccation of captured specimens, I placed a dampened sponge inside each trap. Originally, I drilled holes into the bottom of buckets for drainage. However, in remnants with high levels of ground water, water would flood the bucket from the bottom up. Therefore, in these remnants and places that tended to flood, I installed buckets without holes in the bottom, and I used iron rebar stakes to hold buckets in the ground against hydro-static water pressure (Enge, K., personal communication). I scooped out rainwater collected in pitfall traps each sampling day as necessary. PVC pipe refugia were used to attract various species of tree frogs. I used pipes of both 2.5 cm and 5 cm diameter-widths, with lengths of about 76 cm. I drove pipes into the ground at a depth suitable for the pipe to stand up on its own (Zacharow, Barichivich, and Dodd 2003). I placed one pipe of each diameter width between each wing of the Y-shaped fence array (Moseley et al. 2003), resulting in 6 total PVC pipes per sampling array.

This design was well suited to sampling in multiple small-sized areas, because it was compact as well as cost effective. Because this sampling method does not rely on human observations, detection probabilities for species should have been similar within remnants, assuming that species moved in the same manner throughout remnants. In addition, as sampling methods should reflect the detection probability of the study subjects in order to be effective (MacKensie and Royle 2005), this method allowed me to sample for a relatively wide amount of diversity, given resource constraints.

To compare edge and interior locations (Figure 2-2), I considered the first 40 m from the remnant boundary toward the interior as "edge", and I considered all space beyond 40 m from the boundary of the remnant to be "interior". I decided to place arrays at edge locations between 20–40 m from boundary edges due to the close proximity of remnants to the urban environment,

and the potential for human interference. Except for this specification, I placed sampling arrays randomly within edge and interior areas of remnants. To assure some degree of equal sampling effort per forest remnant, I allocated a one array per 2 ha ratio, up to a maximum of 4 arrays per remnant. I specified all arrays within remnants to be at least 100 m apart from each other (Campbell and Christman 1982), though in two remnants (Lake Alice Conservation Area and Biven's Forest), logistical difficulties (unsuitable substrates) only allowed a maximum distance of 80 m between sampling arrays. Using these parameters, I placed a total of 7 interior and 7 edge arrays in 5 forest remnants around the University of Florida campus.

When I sampled herps, I opened traps for periods of four days apiece, and checked them every day. I opened and checked traps in a systematic sequence so that they were checked at the approximate corresponding time they were set on the previous day. This assured that all traps would be open to sample for the same amount of time each day (approx. 24 hr), allowing for equal sampling effort per trap. On the fourth day, I closed traps until the next sampling period. Each day, I identified captured specimens to species, and then promptly released them. I operated sampling arrays from May through August, every other week. Occasionally, heavy rains forced the closure of some traps due to flooding. In this situation, closed traps were re-opened during the same week for the amount of sampling time lost to inclement weather. The presence of ants at sampling locations also necessitated the closure of funnel traps indefinitely when trapped individuals were negatively affected.

Analysis

I conducted data analyses comparing average daily relative abundance of individual species, at the Order/Suborder taxa-level (including Snake, Frog, Lizard), and the Family taxa-level (Ranid, Hylids, Skinks, Anoles) between edges and interiors. I also conducted an analysis of overall species richness between edges and interiors. I calculated average daily relative

abundance per species for each edge or interior location within a forest remnant (e.g., Harmonic Woods) by dividing the summed count data per remnant edge and interior by total trap effort (i.e., number of trap nights). Total trap effort was modified by sampling methodologies utilized (e.g., 3 funnel traps and 1 pitfall=4/4, or 100% operational) per trap night. For example, if a total of 10 frogs were caught over 4 nights in which the pitfall trap and only 2 of the 3 funnel traps were open, then I would calculated this average as: $10/(4 * [3/4]) = 3.33$.

For most species, three funnel traps and one pitfall trap per array constituted the applicable sampling methodologies at each array. For tree frogs, the sampling involved only the 6 PVC pipes per array (e.g., 6 pipes = 6/6 or 100% operational). For *Anolis sagrei*, which were caught using all sampling methodologies, the applicable sampling devices included the pipes, the funnel traps, and the pitfall traps (e.g., 3 funnel traps, 1 pitfall, and 6 pipes = 10/10 or 100% operational). I used this analytical approach because sampling effort per array was occasionally reduced when traps or pipes were lost temporarily due to extreme weather or unknown disturbances (i.e. raccoon interference), or intentionally removed due to ant predation.

In Harmonic Woods, Graham Woods, and Bartram-Carr Woods, there were only 2 sampling arrays (1 at edge, 1 at interior). Lake Alice Conservation Area and Biven's Forest were larger and had two sampling arrays per edge and interior. However, I inadvertently placed 1 edge location in each of the larger remnants (Biven's Forest and Lake Alice Conservation Area) too far from boundaries of these remnants (i.e., > 20–40 m from patch boundaries). I excluded these arrays from analysis to prevent undue bias on any actual edge effect. In addition, I only sampled Bartram-Carr Woods through the first week of July in 2006 because of building construction that began in that remnant.

I entered calculated data into a one-way ANOVA model blocked for forest remnant in which average daily relative abundance was the dependent variable, and edge or interior location was the independent variable. Because I was not interested in the effect of sampling year, I averaged the relative abundances for each analyzed species and group between both years. I tested the data for normality with the Ryan-Joiner test, and for equal-variance with the Levine test. I square-root transformed non-normal and heteroskedastic distributions for individual species and groups. I used the non-parametric Friedman test to analyze species and groups unable to meet parametric test assumptions after transformation. Because there were 5 sampled forest remnants with both edge and interior locations, this resulted in a total of 10 possible forest remnant locations. In order to deal with non-normality issues due to having too many zeros in the data, I only statistically analyzed individual groups in each level of analysis if they were present in at least half (5) of the 10 possible forest remnant locations. An alpha of 0.1 was used for all statistical tests.

I calculated species richness (edge and interior) per forest remnant and entered it into a one-way ANOVA model blocked by forest remnant in which number of species was the dependant variable, and edge or interior location was the independent variable. Similar to the count data, I averaged species richness data between both years. I tested normality and variance assumptions as previously described (alpha=0.1).

Lastly, in order to gauge similarities in species assemblages at edges and interiors, I computed Horn similarity index values between edges and interiors within each remnant. To do this, I used R Statistical Program, using the Vegan Community Analysis package.

Results

Over the summers of 2005 and 2006, I checked 12 arrays a total of 552.5 trap nights for tree frogs, 548.6 trapping nights for *Anolis sagrei*, and 542.75 trapping nights for all remaining species. I caught a total of 23 species in sampling arrays, shown in Appendix B.

Individual Species

Only 6 species were present in enough forest remnant locations in both years to be individually analyzed. After analyzing *Anolis sagrei, Eleutherodactylus planirostris sp., Hyla cinera, Hyla squirella, Rana clamitans,* and *Scincella lateralis*, none were found to have significantly higher relative abundances at either edges or interiors (Table 2-1). *Hyla squirella* was found to be significantly affected by remnant (Table 2-2), with 89.76% of the cumulative average daily relative abundance of this species found in Biven's Forest.

Taxa-Groups

General taxa-subgroup

When I grouped the species into general-taxa subgroups, including the order Anura (frogs), and the suborders Serpentes (snakes) and Lacertilia (lizards) of the order Squamata, there were no groups that had significantly higher relative abundances at edges or interiors (Table 2-1). Frogs were significantly affected by forest remnant, with 80.52% of this group's cumulative average relative abundance found in Biven's Forest and Lake Alice Conservation Area (Table 2-2). Snakes showed a similar remnant effect, with 60.98% of the group's cumulative average relative abundance represented in Biven's Forest and Lake Alice Conservation area, and 32.69% of the group's cumulative average relative abundance represented in Harmonic Woods (Table 2-2).

Specific taxa-subgroup

When I grouped species into more specific-taxa subgroups, including Ranidae (true frogs), Scincidae (skinks), Hylidae (tree-frogs), and Polychrotidae (Anolis lizards), no groups had significantly higher daily relative abundances at edges or interiors (Table 2-1). Ranids were significantly affected by forest remnant, with 69.42% of this group's cumulative relative abundance found in Biven's Forest and Lake Alice Conservation Area (Table 2-2).

Species Richness

The number of species between edge and interior locations was not significantly different (Table 2-1). Species richness was significantly affected by forest remnant, with the most species occurring in Biven's Forest (19 species) and Lake Alice Conservation Area (16 species) (Table 2-2).

Species Composition

The Horn similarity index is based on a scale of 0–1, with 0 representing a completely different species composition, and 1 representing completely identical compositions. When I calculated the similarities between the edges and interiors of individual remnants, it was found that similarity values ranged from 0.520–0.890, with a mean value of 0.775 (Table 2-3).

<div align="center">

Discussion

</div>

Edge vs. Interior Habitat Use

For herpetofauna, I found no difference in the use of edge or interior habitat for any individual species, family-level taxa group, or order/suborder-level taxa group. I also found no difference in species richness between edges and interiors. Further, species similarity indices between edges and interiors ranged from moderately similar to highly similar. Taken together, herpetofauna analyzed in my study do not appear to differentially use edges or interiors of these small urban remnants. Though edge effects for herpetofauna in urban matrices have not been

well studied, there has been previous evidence of differential use of edge and interior habitat of forest remnants in rural settings (Schlaepfer and Gavin 2001, Lehtinen et al. 2003, Urbina-Cardona et al. 2006). A common finding is that canopy cover tends to increase with distance from edge (Urbina-Cardona et al. 2006, Schlaepfer and Gavin 1999). This generally contributes toward interior forest remnant conditions of lower temperatures and increased levels of relative humidity than edges (Urbina-Cardona 2006, Lehtien et al. 2003, Schlaepfer and Gavin 1999). This leads to higher use of forest interior by moisture-sensitive amphibians, and an even some species of reptiles for higher breeding success (Schlaepfer and Gavin 1999). In addition, significant differences in under-story density between edges and interiors may favor species that prefer sparser vegetation densities typically found at interiors, or greater vegetation densities typically found at edges (Schlaepfer and Gavin 2001, Urbina-Cardona et al. 2006).

In my study, one reason for a lack of segregation could be the small amount of structural habitat differences between edge and interior habitats, particularly over-story density (Chapter 1). Further, herpetofauna in my study were only sampled during the summer rainy season, and species during this season, particularly amphibians, may have been inclined to use the entire forest remnant if they were dispersing in search of wetlands for breeding activities. This is consistent with seasonal differences in habitat use by herpetofauna found by Lehtinen et al. (2003) in tropical fragments, and suggests that seasonality may need to be accounted for in future research in my study area. Lastly, only 5 species were sufficiently common to be analyzed individually. The sampling methodology may not have been effective in capturing other species or other herps may not be abundant in these urban remnants.

Habitat Use among Forest Remnants

Although no edge effects were detected for herpetofauna in this system of remnants, species richness was greater and more *Hyla squirella*, Ranids, Anurans (Frogs), and Snakes,

49

were found in certain forest remnants. Several habitat features within particular remnants may have contributed to a forest remnant effect. First, the presence or amount of wetlands in or adjacent to the remnant may have been an important contributor. Biven's Forest and Lake Alice Conservation Area contained or were directly adjacent to the largest amount of wetlands out of the 5 remnants. Biven's Forest was adjacent to a lake and was largely made up of bottomland hardwood habitat, while Lake Alice Conservation Area was adjacent to a marsh and a lake, and was made up largely of mesic, mixed pine-hardwood forest. These two remnants contained the highest relative abundances of Ranids and Anurans, with Biven's Forest containing the highest counts of *Hyla squirella* and the most species. This is consistent with previous findings that these groups of herpetofauna are often most abundant near wetlands (Houlahan and Findlay 2003). My study was also conducted during the rainy season, and frogs would be most attracted to wet areas for breeding.

Second, remnant size might also be a factor, as Biven's Forest (16.59 ha), and Lake Alice Conservation Area (11.27 ha) were the largest remnants out of the 5. The relative abundance of the Snake group was the highest in Biven's Forest and Lake Alice Conservation Area. Therefore, for the Anuran group, the Ranid group, and the Snake group, larger remnants may attract more individuals and species. This is consistent with previous research that found higher abundances of herpetofauna in larger forest remnants (Gibbs 1998, Cushman 2006, Parris 2006).

Finally, these two areas were also the least isolated of all the 5 remnants, with Lake Alice Conservation Area being directly adjacent to a large marsh and Biven's Forest being located next to agricultural fields, forested land, and a lake. Roads or buildings circumscribed the other three remnants. Isolation, particularly by urban, impervious surfaces such as roads have been implicated as being negatively associated with species richness and abundance in remnants

(Houlahan and Findlay 2003, Ficetola and De Bernardi 2004, Parris 2006) because roads are a barrier to dispersal (Gibbs 1998, deMaynadier and Hunter 2000, Cushman 2006).

Conclusion

No differences in relative abundance and species richness for any species or group of herpetofauna, as well as similar species compositions between edges and interiors of remnants suggests that herpetofauna may not differentiate between edge and interior areas of these forest remnants. This is possibly due to the fact that herps may widely disperse during the summer, searching for breeding sites. Also, vegetative characteristics were similar between edge and interior habitats in this system. However, a couple of remnants had more herpetofauna, suggesting that remnant size, wetland presence, and isolation by urban surfaces of certain may influence the distribution of herpetofauna.
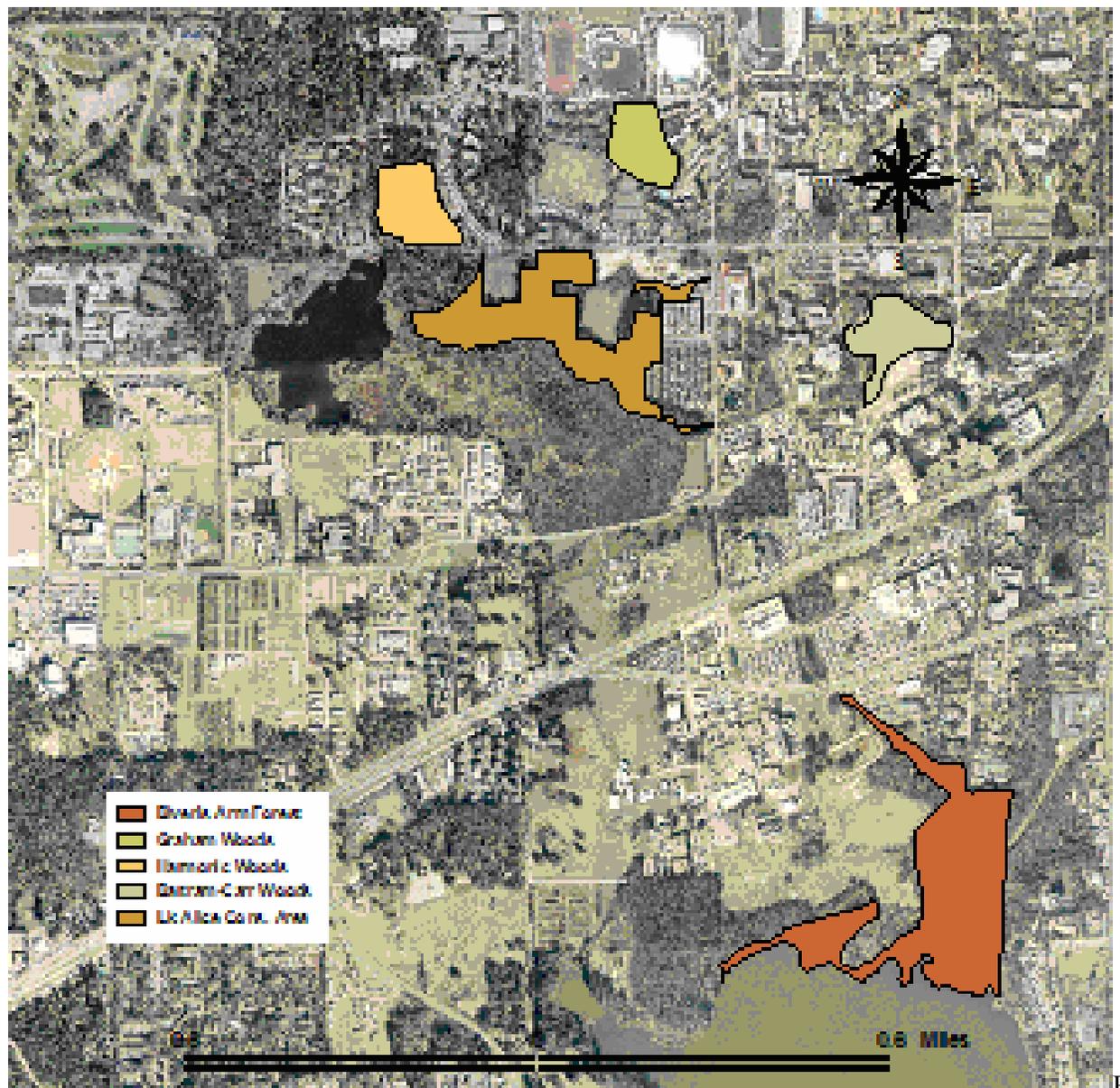
Figure 2-1. Forest remnants on the University of Florida campus in Gainesville, Florida.
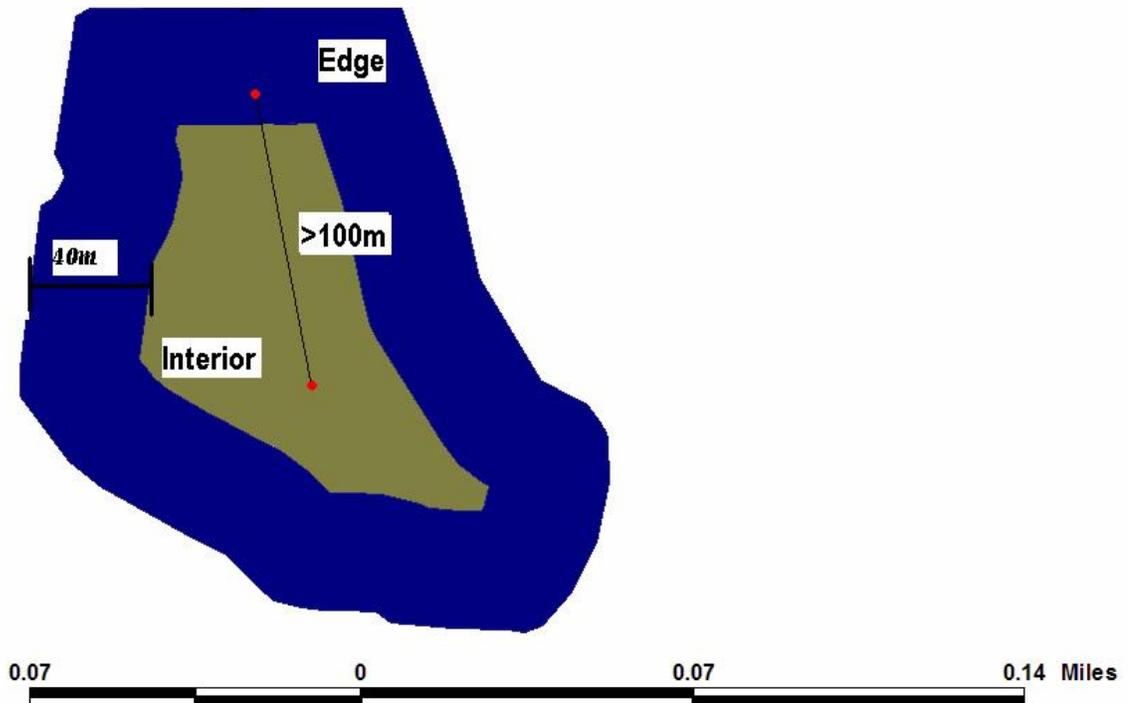
Figure 2-2. Illustration of edge and interior location of herpetofauna sampling arrays within forest remnants in Gainesville, Florida. An edge array was within 20–40 m from the boundary of a remnant and an interior array was situated greater than 40 m from a remnant boundary. Arrays were positioned to be at least 100 m apart to maintain independence from each other.

Table 2-1. Average daily relative abundance of herpetofauna species and groups, as well as species richness between edges and interiors of 5 urban forest remnants in Gainesville, FL. Shown are the means and standard error (SE) values for the average daily abundances and species richness of both edges and interiors, the test statistics (T.S) and associated *P*-values for all individually analyzed species and groups. Also shown are the number of species per taxa group. Unless noted, statistical test is one way ANOVA. For all tests, df = 1, and n = 5 for edge and interior areas.

Taxa Group	Number of Species per group	Taxa Group/ Species/ Species Richness	Edge		SE	Interior		SE	T.S.	*P*
Order-level	10	Anura	0.59	±	0.27	0.62	±	0.33	0.19	0.69
	7	Squamata, suborder Serpentes	0.03	±	0.02	0.04	±	0.01	0.17	0.70
	5	Squamata, suborder Lacertilia	0.43	±	0.13	0.23	±	0.09	1.43	0.30
Family-level	2	Hylidae**	0.27	±	0.19	0.36	±	0.32	0.00	1.00
	2	Polychrotidae*	0.15	±	0.04	0.12	±	0.06	0.19	0.69
	3	Ranidae	0.21	±	0.08	0.17	±	0.07	3.39	0.14
	3	Scincidae	0.27	±	0.11	0.11	±	0.05	2.63	0.18
		Anolis sagrei*	0.15	±	0.03	0.12	±	0.06	0.42	0.55
		Eleutherodactylus planirostris*	0.07	±	0.03	0.05	±	0.03	0.94	0.39
		Hyla cinera**	0.02	±	0.02	0.03	±	0.01	1.00	0.32
		Hyla squirella*	0.25	±	0.19	0.33	±	0.31	0.33	0.56
		Rana clamitans	0.13	±	0.04	0.13	±	0.04	0.01	0.95
		Scincella lateralis	0.24	±	0.12	0.09	±	0.05	1.84	0.25
		Species Richness	7.40	±	1.75	8.20	±	2.03	1.43	0.30

*square-root transformed
**tested with non-parametric Friedman test

Table 2-2. Herpetofauna species and groups shown to be significantly affected by remnant in urban forest remnants in Gainesville, Florida. Shown are average daily abundances of each species or group in each remnant, the remnant mean accompanied by the standard errors (SE), test-statistics (T.S.) and associated P-values. The number of species per taxa group are also shown. Unless noted, statistical test is one way ANOVA. For all tests, df = 4, and n = 5 for remnants sampled. Remnant abbreviations: HW=Harmonic woods, GW=Graham Woods, BF=Biven's Forest, HCP=Health Center Park, and LACA=Lake Alice Conservation Area.

Taxa Group	Number of Species per group	Taxa group/ Species/ Species richness	HW	GW	HCP	BF	LACA	Mean	SE	T.S.	P
Order-level	10	Anura	0.34	0.10	0.15	0.75	1.70	0.61	0.30	71.52	<0.00
	7	Squamata, suborder Serpentes	0.05	0.00	0.01	0.07	0.03	0.03	0.01	5.00	0.07
Family-level	3	Ranidae	0.18	0.10	0.01	0.43	0.22	0.19	0.07	63.19	<0.00
		Hyla squirella*	0.00	0.00	0.03	1.29	0.12	0.33	0.29	8.00	0.09
		Species Richness	7.00	2.50	6.00	12.50	11.50	7.90	1.84	19.34	0.01

*tested with non-parametric Friedman test

Table 2-3. Horn compositional similarity values for species assemblages between edges and interiors within urban forest remnants in Gainesville, Florida. Values closer to 1 indicate similar species composition.

Remnant	Horn Similarity Index Value
Harmonic Woods	0.855
Graham Woods	0.741
Bartram-Carr Woods	0.863
Biven's Forest	0.897
Lake Alice Conservation Area	0.520
Mean	0.775

SPECIES ABBREVIATIONS, RESIDENCY STATUS, AND INCLUSION IN COMMON OR UNCOMMON GROUPS FOR ALL BIRD SPECIES OBSERVED PER SEASON.

Table A-1. Species abbreviations, residency status, and inclusion in common or uncommon groups for all bird species observed per season. Residency codes: WR=winter resident, SM=summer, migrant, SO = stopover migrants and YR=year-round residents. "C" indicates it was included in the "common" subgroup during a given season. "U" indicates it was included in the "uncommon" subgroup during a given season.

Species	Abbreviation	Status	Winter	Spring	Summer	Fall
Acadian Flycatcher	ACFL	SO				U
American Crow	AMCR	YR	C	U	U	U
American Goldfinch	AMGO	WR	C	U		
American Redstart	AMRE	SO		U	U	C
American Robin	AMRO	WR	C	U		
Baltimore Oriole	BAOR	WR	U	U		U
Black and White Warbler	BAWW	WR	C	U		C
Barred Owl	BDOW	YR				U
Belted Kingfisher	BEKI	YR			U	
Blue-Gray gnatcatcher	BGGC	YR	C			U
Brown-headed cowbird	BHCO	YR	U	C	C	U
Blue-headed Vireo	BHVI	WR	U	C		
Blue Jay	BLJA	YR	C	U	C	C
Blackpoll Warbler	BPWA	SO		U		
Brown Thrasher	BRTH	YR	U	U	U	U
Boat-tailed Grackle	BTGR	YR	U			
Carolina Chickadee	CACH	YR	C	U	U	U
Carolina Wren	CARW	YR	C	C	C	C
Cedar Waxwing	CEWA	WR	C	U		
Chimney Swift	CHSW	SM		U		
Common Grackle	COGR	YR		U	U	U
Common Yellowthroat	COYE	YR		U		U
Downy Woodpecker	DOWO	YR	C	C	C	C
Eastern Phoebe	EAPH	WR	C			U
Eurasian-collared dove	ECDO	YR				U
Eastern Tufted Titmouse	ETTI	YR	C	C	C	C
Fish Crow	FICR	YR	U	U	U	U
Great Blue Heron	GBHE	YR	U			
Great Crested Flycatcher	GCFL	SM		C	C	
Gray Catbird	GRCA	WR	C	C	U	C
Hermit Thrush	HETH	WR	U		U	
House Finch	HOFI	YR	U	U	U	U

Table A-1. Continued

Species	Abbreviation	Status	Winter	Spring	Summer	Fall
House Wren	HOWR	WR	U	U		U
Indigo Bunting	INBU	SO		U		
Loggerhead Shrike	LOSH	YR			U	
Magnolia Warbler	MAWA	SO				U
Mourning Dove	MODO	YR	C	C	U	C
Northern Cardinal	NOCA	YR	C	C	C	C
Northern Flicker	NOFL	YR	U			U
Northern Mockingbird	NOMO	YR	C	U	U	C
Northern Parula	NOPA	SM	U	C	C	U
Orange Crowned Warbler	OCWA	SO		U		
Oprey	OSPR	YR	U	U	U	
Ovenbird	OVEN	WR	U			U
Palm Warbler	PAWA	WR	C		U	U
Pine Warbler	PIWA	YR	U		U	
Pileated Woodpecker	PIWO	YR	U	U	U	U
Prairie Warbler	PRWA	SO		U		U
Red-bellied Woodpecker	RBWO	YR	C	C	C	C
Ruby-crowned Kinglet	RCKI	WR	C	C		C
Red-eyed Vireo	REVI	SM	U	C		C
Red-headed Woodpecker	RHWO	YR				U
Rock Dove	RODO	YR	U			
Red-Shouldered Hawk	RSHA	YR	U		U	U
Red-Tailed Hawk	RTHA	YR			U	U
Red-winged Blackbird	RWBB	YR	U			
Summer Tanager	SUTA	SM			U	U
Swainson's Thrush	SWTH	SO				U
Tree Swallow	TRES	WR		U		
White-eyed Vireo	WEVI	YR	U	U	U	C
Wild Turkey	WITU	YR	U			
Wood Thrush	WOTH	SM				U
Yellow-breasted Chat	YBCH	SO	U			
Yellow-billed Cuckoo	YBCU	SO				U
Yellow-bellied Sapsucker	YBSA	WR	C	U	U	
Yellow-rumped Warbler	YRWA	WR	C	C		
Yellow-throated Warbler	YTWA	YR	U	U		U

APPENDIX B
ALL SPECIES OF HERPETOFAUNA DETECTED BY HERPETOFAUNAL SAMPLING ARRAYS DURING THE SUMMERS OF 2005 AND 2006.

Table B-1. All species of herpetofauna detected by herpetofaunal sampling arrays during the summers of 2005 and 2006.

Species	Order-level taxa group	Family-level taxa group
Anolis carolinensis	Lizard	Polychotidae
Anolis sagrei	Lizard	Polychotidae
Apalone ferox	Turtle*	N/A
Bufo terrestris	Anura	Bufonidae*
Bufo quercicus	Anura	Bufonidae*
Coluber constrictor	Snake	N/A
Diadolphus punctatus	Snake	N/A
Eleutherodactylus planirostris sp.	Anura	N/A
Eumeces fasciatus	Lizard	Scincidae
Eumeces laticeps	Lizard	Scincidae
Farancia abacura	Snake	N/A
Gastrophryne carolinenis	Anura	N/A
Hyla cinera	Anura	Hylidae
Hyla squirella	Anura	Hylidae
Rana catesbiana	Anura	Randiae
Rana clamitans	Anura	Randiae
Rana sphenocephalus	Anura	Randiae
Rhadinaea flavilata	Snake	N/A
Scaphiopus holbrookii	Anura	N/A
Scincella lateralis	Lizard	Scincidae
Storeria dekayi victa	Snake	N/A
Thamnophis sauritus	Snake	N/A
Thamnophis sirtalis	Snake	N/A
Trachemys Scripta	Turtle*	N/A

*Insufficient data for analysis

APPENDIX C
UNIVERSITY OF FLORIDA WILDLIFE SURVEY AND MONITORING PROGRAM: ONE
YEAR RESULTS AND DATA SUMMARY

Object C-1. PDF of University of Florida wildlife survey and monitoring program: one year
results and data summary

LIST OF REFERENCES

Alig. R.J., D.K. Jeffrey, and M. Lichtenstein. 2004. Urbanization on the US landscape: Looking ahead in the 21st century. Landscape and Urban Planning 69: 219–234.

Atchinson, K.A. and A.D. Rodewald. 2006. The value of urban forests to wintering birds. Natural Areas Journal 26: 280–288.

Baillie, J.E.M, C. Hilton-Taylor, S. N. Stuart. 2004. IUCN Red List of Threatened Species. Global Species Assessment. IUCN Gland, Switzerland and Cambridge, UK.

Breitwisch, R. J. 1977. The ecology and behavior of the Red-bellied Woodpecker, *Centurus carolinus* (Linnaeus; Aves: Picidae), in south Florida. Thesis, University of Miami, Coral Gables, FL.

Breitwisch, M. Diaz, and R. Lee 1987. Foraging efficiencies and techniques of juvenile and adult Northern Mockingbirds (*Mimus polyglottos).* Behaviour 101: 225–235.

Brewer, R. 1963. Ecological and reproductive relationships of Black-capped and Carolina chickadees. Auk 80: 9–47.

Campbell, H.W. and S.P. Christman. 1982. Field techniques for herpetofaunal community analysis. Pages 193–200 *in* N. J. Scott, Jr. (ed.), Herpetological Communities. U.S. Fish and Wildlife Service. Wildlife Research Report 13, Washington, D.C., USA.

Chase, J.F. and J.J. Walsh. 2006. Urban effects on native avifauna: a review. Landscape and Urban Planning 74: 46–69.

Cushman, S. A. 2006. Effects of habitat loss and fragmentation on amphibians: A review and prospectus. Biological Conservation 128: 231–240.

Delis, P.R., H.R. Mushinsky, and E.D. McCoy. 1996. Decline of some west-central Florida anuran populations in response to habitat degradation. Biodiversity and Conservation 12: 1579–1595.

DeMaynadier, P.G., M.L. Hunter Jr. 2000. Road effects on amphibian movements in a forested landscape. Natural Areas Journal 20: 56–65.

Derrickson, K. C. and R. Breitwisch. 1992. Northern Mockingbird. Account no. 7 *in* A. Poole, P. Stettenheim, and F. Gill, editors. The Birds of North America Online, The Academy of Natural Sciences, Philadelphia, Pennsylvania and The American Ornithologists' Union, Washington, DC., USA.

Dunford, W. and K. Freemark. 2004. Matrix matters: Effects of surrounding land uses on forest birds near Ottawa, Canada. Landscape Ecology 20: 497–511.

Enge, K.M. 1997. Use of silt fencing and funnel traps for drift fences. Herpetological Review 28: 30–31.

Enge, K. M., M. S. Robson, and K. L. Krysko. 2004. Reptile surveys on pine rockland habitat in six Miami-Dade county parks. Biological Sciences 67: 195–204.

Enge, Kevin M. 2005. Herpetological biologist. Florida Fish and Wildlife Conservation Commission. Florida, USA.

Fernández, J.C. and J.L. Tellaria. 2000. Effects of human disturbance on spatial and temporal Feeding patterns of black bird *Turdus merula* in urban parks in Madrid, Spain. Bird-Study. 2000 47: 13–21.

Fernández-Juricic, E. 2000. Bird community composition patterns in urban parks in Madrid: The role of age, size and isolation. Ecological Research 15: 373–383.

Fernández -Juricic, E. 2001. Avian spatial segregation at edges and interiors of urban parks in Madrid, Spain. Biodiversity and Conservation 10: 1303–1316.

Ficetola, G. F. and F. De Bernardi. 2004. Amphibians in a human-dominated landscape: The community structure is related to habitat features and isolation. Biological Conservation 199: 219–230.

Gates, J.E. and L. W. Gysel. 1978. Avian nest dispersion and fledging success in field-forest ecotones. Ecology 59: 871–883.

Gibbs, J.P. 1998. Amphibian movements in response to forest edges, roads, and streambeds in southern New England. Journal of Wildlife Management 62: 584–589.

Gibbons, J.W., D.E. Scott, T.R. Avis, J. Ryah, K. A. Buhlmann, T.D. Tuberville, B. S. Metts, J.L. Greene, T.Mills, Y. Leiden, S. Poppy, and C. T. Winne. 2000. The global decline of reptiles, deja vu amphibians. Bioscience 50: 253–266.

Haggerty, T. M., and E.S. Morton. 1995. Carolina Wren (*Thryothorus ludovicianus*). Account 188 *in* A. Poole and F. Gill, editors. The Birds of North America Online, The Academy of Natural Sciences, Philadelphia, Pennsylvania and The American Ornithologists' Union, Washington, D.C., USA.

Hostetler, M.E. 1999. Scale, birds, and human decisions: A potential for integrative research in urban ecosystems. Landscape and Urban Planning 45: 15–19.

Hostetler, M.E., and C.S. Holling. 2000. Detecting the scales at which birds respond to structure to structure in urban landscapes. Urban Ecosystems 4: 25–54.

Hostetler, M.E. and K. Knowles-Yanez. 2003. Land use, scale, and bird distributions in the Phoenix metropolitan area. Landscape and Urban Planning 62: 55–68.

Hostetler, M.E., D. Scot, and J. Paul. 2005. Post-construction effects on an urban development on migrating, resident, and wintering birds. Southeastern Naturalist 4: 421–434.

Houlahan J.E. and S. C. Findlay. 2003. The effects of land use on wetland species and community composition. Canadian Journal of Fisheries and Aquatic Sciences 60: 1078–1094.

James, F.C., and H.H. Shugart. 1970. A quantitative method of habitat description. Audubon Field Notes 24: 727–735.

Johnson, Steven Albert. 2005. Assistant professor in the department of Wildlife Ecology and Conservation. University of Florida. Gainesville, Florida, USA.

Jokimäki, J. and J. Suhonen. 1998. Distribution and habitat selection of wintering birds in urban environments. Landscape and Urban Planning 39: 253–263.

Kale II, H. W. and Maehr, D. S. 1990. Florida's Birds. A handbook and reference. Pineapple Press. Sarasota, Florida. USA.

Karr, J.R. 1968. Habitat and avian diversity on strip-mined land in east-central Illinois. Condor 70: 348–357.

Lay, D. 1938. How valuable are woodland clearings to birdlife? Wilson Bulletin 50: 254–245.

Lehtinen, R.M., J.B. Ramanamanjato, and J.G. Raveloarison. 2003. Edge effects and extinction proneness in a herpetofauna from Madagascar. Biodiversity and Conservation 12: 1357–1370.

MacKenzie, D.I., and J.A. Royle. 2005. Designing occupancy studies: General advice and allocating survey effort. Journal of Applied Ecology 42: 1105–1114.

Marzluff, J.M. 2001. Worldwide urbanization and its effects on birds. Pages 19–48 *in* J.M. Marzluff, R. Bowman, and R. Donnelly, editors. Avian ecology and conservation in an urbanizing world. Kluwer Academic Publishers. Boston, Massachusetts, USA, Dordrect, GER, and London, UK.

McCollin, D. 1998. Forest edges and habitat selection by birds; A functional approach. Ecography 21: 247–260.

McIntyre, N.E. 1995. Effects of forest patch size on avian diversity. Landscape Ecology 10: 85–99.

McKinney, M.L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127: 247–260.

McPherson, J.M. 1987. A field study of winter fruit preferences of Cedar Waxwings. Condor 89: 293–306.

Mensing, D.M., S.M. Galatowitsch, and J.R. Tester. 1998. Anthropogenic effects on the biodiversity of riparian wetlands of a northern temperate landscape. Journal of environmental management 53: 349–377.

Moore, F.R, S. A. Gauthreaux, P. Kerlinger, and T.R. Simons. 1995. Habitat requirements during migration: Important link in conservation. 121–144 in T.E. Martin and D.M. Finch, editors. Ecology and management of neotropical migratory birds. Oxford University Press, New York, New York, USA.

Mörtberg, U.M. 2001. Resident bird species in urban forest remnants: Landscape and habitat Perspectives. Landscape Ecology 16: 193–203.

Moseley, K.R., S.B. Castleberry, and S.H. Schweitzer. 2003. Effects of prescribed fire on herpetofauna in bottomland hardwood forests. Southeastern Naturalist 2: 475–486.

Mostrom, A. M., R. L. Curry, and B. Lohr. 2002. Carolina Chickadee (*Poecile carolinensis*). Account 636 in A. Poole and F. Gill, editors. The Birds of North America, The Academy of Natural Sciences, Philadelphia, Pennsylvania, and The American Ornithologists' Union, Washington, D.C., USA.

Noss, R. 1991. Effects of edge an internal patchiness on avian habitat use in an old-growth Florida hammock. Natural Areas Journal 11: 35–47.

Parris, K. 2006. Urban amphibian assemblages as metacommunities. Journal of Animal Ecology 75: 757–764.

Poole, A. (Editor). 2005. The Birds of North America, The Academy of Natural Sciences, Philadelphia, PA, and The American Ornithologists' Union, Washington, D.C., USA.

Reijnen R, R. Foppen, and G. Veenbaas. 1997. Disturbance by traffic of breeding birds: Evaluation of the effect and considerations in planning and managing road corridors. Biodiversity and Conservation 6: 567–581.

Richter, K. O., and A.L. Azous. 1995. Amphibian occurence and wetland characteristics in the pudget sound basin. Wetlands 15: 305–312.

Riley, S.P.D., G. T. Busteed, L. B. Kats, T L. Vandergon, L. F. S. Lee, R. G. Dagit, J. L. Kerby, R. N. Fisher, and R. M. Sauvajot. 2005. Effects of urbanization on the distribution of amphibians and invasive species in southern California streams. Conservation Biology 19: 1894–1907.

Robel, R.J., J.N. Briggs, A.D. Dayton, and L.C. Hurlbert. 1970. Relationships between visual obstruction and weight of grassland vegetation. Journal of Wildlife Management 23: 295–297.

Rodewald, P.G. and M.C. Brittingham. 2002. Habitat use and behavior of mixed species landbird flocks during fall migration. Wilson Bulletin 114: 87–98.

Rodewald, P. G., and M.C. Brittingham. 2004. Stopover habitats of landbirds during fall: Use of edge- dominated and early successional forests. The Auk 121: 1040–1055.

Roth, R.R. 1979. Foraging behavior of mockingbirds: The effect of too much grass. The Auk 96: 421–422.

Rubbo, M. J. and J.M. Kiesecker. 2005. Amphibian breeding distribution in an urbanized landscape. Conservation Biology 19: 504–511.

Schlaepfer M. A. and T.A. Gavin. 2001. Edge effects on lizards and frogs in tropical forest fragments. Conservation Biology 15: 1079–1090.

Shackelford, C. E., R. E. Brown, and R. N. Conner. 2000. Red-bellied Woodpecker (*Melanerpes carolinus*). Account 500 *in* A. Poole and F. Gill, editors. The Birds of North America, The Academy of Natural Sciences, Philadelphia, Pennsylvania, and The American Ornithologists' Union, Washington, D.C., USA.

Shochat, E., P.S. Warren, S.H. Faeth, N.E. McIntrye, and D. Hope. 2006. From patterns to emerging processes in mechanistic urban ecology. Trends in Ecology and Evolution 21: 186–191.

Smith, W.P., D.J. Twedt, D.A. Widenfeld, P.B. Hamel, R.P. Ford, R.J. Cooper. 1993. Point counts of birds in bottomland hardwood forests of the Mississippi alluvial plain. U.S. Forest Service, Forest Service Southern Forests Experimentation Station, Research Paper SO-274, Washington, D.C., USA.

Tarvin, K. A., and G.E. Wolfenden. 1999. Blue Jay (*Cyanocitta cristata*). Account 469 *in* A. Poole and F. Gill, editors. The Birds of North America, The Academy of Natural Sciences, Philadelphia, Pennsylvania, and The American Ornithologists' Union, Washington, D.C., USA.

Tilghman, N.G. 1987a. Characteristics of urban woodlands affection breeding bird diversity and abundance. Landscape and Urban Planning 14: 481–495.

Tilghman, N.G. 1987b. Characteristics of urban woodlands affection winter bird diversity and abundance. Forest Ecology and Management 21: 163–175.

Urbina-Cardona, J. N., M. Olivares-Perez, V. H. Reynoso. 2006. Herpetofauna diversity and microenvironment correlates across a pasture-edge-interior ecotone in tropical rainforest fragments in the Los Tuxtlas Biosphere Reserve of Veracruz, Mexico. Biological Conservation 132: 61–75.

Villard, M. A. 1998. On forest interior species, edge avoidance, area sensitivity and dogmas in avian conservation. The Auk 115: 801–805.

Whitcomb. R.F., C.S. Robbins, J.F. Lynch, B.L. Whitcomb, K. Limkiewicz, and D. Bystrak. 1981. Effects of forest fragmentation on avifauna of the eastern deciduous forest. 125–205 *in* R.L. Burgess and D. M. Sharpe, eds. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, New York. USA.

Winker, K., D.W. Warner, and A. Weisbrod. 1992. Migration of woodland birds at a fragmented woodland stopover site. The Wilson Bulletin 104: 580–592.

Witmer, M.C. 1996a. Annual diet of Cedar Waxwing based on U.S. Biological Survey records (1885–1950) compared to diet of American Robin: contrasts in dietary patterns and natural history. The Auk 113: 414–430.

Witmer, M.C. 1996b. Consequences of an alien shrub on the plumage coloration and ecology of Cedar Waxwings. The Auk 113: 735–743.

Witmer, M. C., D. J. Mountjoy, and L. Elliot. 1997. Cedar Waxwing (*Bombycilla cedrorum*). Account 309 *in* A. Poole and F. Gill, editors. The Birds of North America, The Academy of Natural Sciences, Philadelphia, Pennsylvania, and The American Ornithologists' Union, Washington, D.C., USA.

Yaukey, P.H. 1996. Patterns of avian population density, habitat use, and flocking behavior in urban and rural habitats during winter. Professional Geographer 48: 70–81.

Yahner, R. H. 1988. Changes in wildlife communities near edges. Conservation Biology 2: 333–339.

Zacharow, M., W.J. Barichivich, and K.C. Dodd, Jr. 2003. Using ground-placed PVC pipes to monitor hylid treefrogs: capture biases. Southeastern Naturalist 2: 574–590.

BIOGRAPHICAL SKETCH

Dan Dawson received his High School Diploma from W.R. Boone High School in Orlando, Florida in the spring of 2000. He attended the University of Florida, where he attained a Bachelor of Science degree in the College of Agriculture and Life Science with a major in wildlife ecology and conservation in the summer of 2004. He further attended the University of Florida for graduate school, and studied wildlife diversity and conservation within the urban environment. He was awarded a Master of Science degree from the College of Agriculture and Life Science with a major in wildlife ecology and conservation in the spring of 2007.