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Avian communities in an urbanizing region: Abundance patterns and effects of local habitat features

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AVIAN COMMUNITIES IN AN URBANIZING REGION: ABUNDANCE PATTERNS AND EFFECTS OF LOCAL HABITAT FEATURES

A Thesis

by

JOHN S. BRUSH

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The University of Texas Rio Grande Valley
In partial fulfillment of the requirements for the degree of

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AVIAN COMMUNITIES IN AN URBANIZING REGION:ABUNDANCE PATTERNS AND EFFECTS OF LOCAL HABITAT FEATURES

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December 2016

ABSTRACT

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The Lower Rio Grande Valley of Texas has experience rapid population growth and increase of urban area since the mid-20th century. The goal of this thesis were to: 1) explore patterns of avian communities in suburban and remnant urban woodlands, with particular interest in native forest species, and 2) to assess how local habitat variables, such as number of native trees, affect avian communities in suburban habitats, and 3) provide recommendations to municipalities and homeowners to enhance urban habitats for birds. Results indicated that remnant urban woodlands provided habitat for common native forest bird species that otherwise avoided suburban habitats, and therefore should be conserved. Increased vegetation cover had a positive effect on bird diversity and native forest bird species richness while increased impervious cover and grass cover decreased both avian community metrics. Local habitat features were correlated with bird species abundances and bird diversity metrics.

DEDICATION

I would not have been able to complete this thesis without the support of family and friends. My parents, Timothy and Catherine Brush, along with my sisters Elizabeth and Laura, have provided constant love and support. I also dedicate this work to my friends, Stephen and Amber, for their continued friendship. And lastly, I dedicate this thesis to the winged wonders.

ACKNOWLEDGMENTS

I will always be grateful to Dr. Alexis Racelis, chair of my thesis committee, for his mentoring and advice. From recruitment on he has provided the guidance and clarity of thought essential to this process. My thanks also go out to my thesis committee members: Dr. Timothy Brush and Dr. Frank Dirrgl. Their input on methodology and the manuscript has been vital, and their frequent communication has been kind, poignant, and encouraging. I also thank Dr. Kenneth R. Summy for access to his lab and assistance with GIS software.

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CHAPTER I

INTRODUCTION

Urbanization and Biodiversity

The continuous and rapid growth of human populations in recent centuries has led to the Anthropocene, the modern era in which human activities have dominated the biosphere (Crutzen, 2006; Steffen et al., 2007). Proposed to have begun during the Industrial Era (circa 1800), the Anthropocene is defined by the idea that "…the Earth has now left its natural geological epoch, the present interglacial state called the Holocene" (Steffen et al., 2007) and is now greatly impacted by humans.

During the Anthropocene, human populations have experienced exponential growth. The global population has more than doubled since the mid-20th century (Steffen et al., 2007), and since the early 2000's has increased by about 1 billion (United Nations Population Division, 2015). This human dominance is accompanied and substantiated by the alteration of the Earth's environment (Vitousek et al., 1997). For instance, more land has been converted to cropland over the past 30 years than was converted between 1700 and 1850 (Millennium Ecosystem Assessment, 2005), and cropland and pasture now covers approximately 40% of the world's land surface (Foley et al., 2005).

In addition to agricultural land conversions, urban area has also increased rapidly during the Anthropocene. Between 1970 and 2000, global urban area quadrupled to 58,000 km^2, and is projected to increase to 1,527,000 km^2 (as noted by the authors, an area roughly the size of Mongolia) by 2030 (Seto et al., 2011; United Nations, 2014). In developing countries this urban expansion often takes place on agricultural lands (Seto et al., 2011), reflecting a global human population shift from rural to urban. As of 2014, 54% of people lived in urban areas, a number which is projected to increase to 66% by 2050 (United Nations, 2014).

Urbanization is "... a multidimensional process that manifests itself through rapidly changing human population and changing land cover" (Seto et al., 2013). What qualifies as an urban area changes varies between countries and organizations, but Seto et al. (2013) note that in North America and Europe it is generally characterized by having greater than 50% of the surface built, with adjacent areas having between 30-50% built surface, and a human population density of greater than 10 people per hectare. Marzluff et al. (2001) also qualify urban as having greater than 50% surface built, but go further to standardize terms of suburban (30-50% built surface, > 10 people/hectare), rural/exurban (5-20% surface built, 1-10 people/hectare), and wildland (0-2% built, < 1 person/hectare).

Globally, and especially in urbanized, high density population centers, land use change has important implications for human and environmental health, along with providing opportunities to study impacts on ecological processes and the ecosystem services associated with them.

The land transformations and population shifts resulting from urbanization produce a complex mosaic of habitats, and have notable and multi-scalar effects on the environment. Human actions have considerable negative impacts on biodiversity at a global scale (Dirzo et al., 2014). Current species extinction rates across a variety of taxa are higher than historical background rates (Millennium Ecosystem Assessment, 2005). Of the species evaluated by the

IUCN Redlist, 22% of vertebrates are threatened with extinction. This includes 31% of reptiles, 30% of amphibians, 21% of mammals, and 12% of birds (Vié et al., 2009). If threatened and endangered species continue to be lost the earth could be facing its 6th mass extinction (Barnosky et al., 2011), and most indicators of biodiversity continue to decline in the face of increased anthropogenic pressures (Butchart et al., 2010).

The predominant pressure for the majority of these taxa is habitat loss and degradation (Vié et al., 2009). Urbanization often generates increased extinction rates and loss of native species (McKinney, 2002), which is exacerbated by the trend for faster urbanization rates in biodiverse, low-elevation, and coastal regions (Seto et al., 2013). Urbanization affects local biodiversity through habitat loss and biotic community changes (e.g. increase of non-native species, increase of predators) associated with the subsequent structure of “replacement habitats” (Blair, 1996; Chace et al., 2006; DeCandido et al., 2004; DeGraaf et al., 1986; Dirzo et al., 2014; Donnelly et al., 2006; Kowarik, 2008; Marzluff et al., 2008; McKinney, 2002). Globally, and especially in these urbanized, high density population centers, land use change has important implications for human and environmental health, along with providing opportunities to study impacts on ecological processes and the ecosystem services associated with them.

Birds, a well-known taxon, make good indicator species to study the effects of urbanization (Blair, 1999). The study of birds in urban environments has been going on for over 75 years, with the number of studies increasing drastically over the past few decades (Marzluff et al., 2001). Many of these studies, along with those on other taxa, are done by examining distributions along an urban-rural gradient (Marzluff et al., 2001; McDonnell et al., 2008). In a literature review, McDonnell and Hahs (2008) found that 63% percent of urban ecology papers used urban-rural gradients, of which nearly half were studies of birds.

Gradient analysis is based on the idea that “all species occur in a characteristic, limited range of habitats; and within their range, they tend to be most abundant around their particular environmental optimum” (Ter Braak et al., 1988). Gradient analysis provides a way to assess and relate biotic community composition to environmental variables (and vice versa) and to address basic ecological questions on multiple scales (McDonnell et al., 1990). Analyses of urban gradients – or the transition between rural to urban environments – examine the complex mosaic of human land-use and vegetated cover that urbanization creates (see for example (Blair, 1996; McDonnell et al., 2008)). Many of these studies, however, focus mainly on describing the pattern of avian communities along the gradient, and do not provide detailed information about the mechanisms driving community changes, such as fecundity, dispersal, and other population dynamics (Marzluff et al., 2001). However, understanding patterns of avian abundance along these gradients is necessary to establish baseline knowledge and generate further inquiry.

Bird species tend to respond to urbanization in one of three ways. As labeled by Blair (1996), bird species fall into the categories of urban exploiters, urban adapters, and urban avoiders.

Urban exploiters (sometimes called synanthropic species) are able to thrive with the changes brought by urbanization, and indeed may not be found (or found in lower abundances) in natural, exurban habitats. In the book “Subirdia”, John Marzluff lists the ‘fab five’ of urban exploiter species; House Sparrow (*Passer domesticus*), European Starling (*Sturnus vulgaris*), Rock Pigeon (*Columba livia*), Mallard (*Anas platyrhynchos*), and Canada Goose (*Branta canadensis*). While some of the most synanthropic species in North America are exotics, native species are capable of exploitation as well. For example, see the significant expansion the Great-tailed Grackle (*Quiscalus mexicanus*) in the south and western United States (Wehtje, 2003). The abundance of urban exploiters often leads to a peak of bird density (though not in species richness) in highly developed urban areas (Beissinger et al., 1982; Blair, 1996; Chace et al., 2006; Emlen, 1974; Evans et al., 2009; Marzluff, 2008).

Urban adapters are species capable of occupying intermediate levels of urbanization. Oftentimes they are early successional species, or generalists with broad habitat tolerances (Emlen, 1974; Marzluff et al., 2008; Marzluff, 2008). This is a wide range of species, but a few common North American examples would be Mourning Dove (*Zenaida macroura*), Northern Mockingbird (*Mimus polyglottos*), and Northern Cardinal (*Cardinalis cardinalis*). These species take advantage of increased resources that suburban habitats provide, such as bird feeders and fruiting plants - including exotic plant species (Belaire et al., 2014; Daniels et al., 2006; Gray et al., 2015).

Urban avoider species are those intolerant of urban development. These are often habitat specialists requiring natural habitats (whether deserts, prairies, or dense forests) – the opposite of lawn-dominated landscapes common in suburban neighborhoods (sometimes called the ‘urban savannah’) (Dorney et al., 1984; Gobster, 1994; Henderson et al., 1998). Aurora et al. (2009) found that bird species dependent on brushy ground and mid-story cover were absent in traditional lawn dominated landscapes in San Antonio, TX. In Tucson, Arizona, where most landscapes are devoid of native flora, native desert species made up only 3% of the bird community (Emlen, 1974). This guild of urban avoider species is therefore a focal point for conservation of bird diversity in urban areas.

Urban exploiters and suburban adapters often belong to omnivorous, granivorous, and ground-foraging guilds, whereas urban avoiders are often insectivorous, foliage and high canopy feeders (Aurora et al., 2009; Beissinger et al., 1982; Burghart et al., 2008; Chace et al., 2006). In several studies in the tropics, frugivores and nectarivores were observed in higher proportions of urban exploiters and adapters (Chace et al., 2006); one study found large numbers of insectivores along with granivores (Escobar-Ibáñez et al., 2015), and another saw increases of omnivorous species associated with human-altered habitats in small patches of cloud forest in Veracruz, Mexico (Rueda-Hernandez et al., 2015) However, data on urban bird communities is lacking in tropical and subtropical regions, as most research on the subject has been done in temperate climates in the United States and Europe (Marzluff et al., 2001).

A number of urban-rural gradient analyses found that bird species richness often peaks in intermediate zones between predominantly built urban centers and rural, natural habitats (Aurora et al., 2009; Blair, 1996; Blair, 1999; Chace et al., 2006; Concepción et al., 2015; Marzluff, 2008). This is in line with the Intermediate Disturbance Hypothesis (Marzluff, 2008; Roxburgh et al., 2004), which suggests that local species diversity is maximized when ecological disturbance (in this case in the form of urbanization) is neither too rare nor too frequent, or where the intensity of disturbance is moderate (Connell, 1978). However, it should be noted that much of the diversity associated with this intermediate level of urbanization (suburban habitats) is from the addition of widespread urban exploiter and urban adapter species. So while local diversity may increase in certain areas, the colonization of urban tolerant species along with the loss of urban avoiders leads to a regional homogenization of avian communities (Blair, 2001; Concepción et al., 2015; Evans et al., 2009; Marzluff, 2008; McKinney, 2006). For example, in a comparison between urban bird communities in Ohio and California, Blair (2001) found that the most developed sites had a species compositional similarity almost three times of that found in the most natural sites. This homogenization is not only concerning due to the loss of predevelopment (potentially localized or endemic) bird species, but also because it affects human perceptions of biodiversity as well (Belaire et al., 2015; McKinney, 2002; Miller, 2005).

To combat the homogenization of avian communities in urban areas, Marzluff (2008) recommends developing separate plans for management of urban exploiters, urban adapters, and urban avoiders, resulting in a heterogeneous mix of low, mid, and highly developed areas. Local scale changes, such as more native vegetation, greater tree coverage, and more diverse vegetative structure can increase the diversity and presence of native bird species, including those urban avoiders associated with predevelopment habitats (Aurora et al., 2009; Belaire et al., 2014; Chace et al., 2006; Chamberlain et al., 2007; Donnelly et al., 2006; Evans et al., 2009; Lerman, 2011; Melles, 2003; Mills et al., 1989). Information on the effects of these habitat features is necessary to inform effective management policies and conservation efforts in the world's growing urban component.

Human Relations to Urban Bird Diversity

Miller (2005) lays out the issue of "shifting baseline syndrome" or "environmental generation amnesia". Generations growing up in an already impoverished state of biodiversity, such as those found in highly developed environments, have their view of biodiversity at a lower baseline. This cycle of shifting baselines leads to a "ratcheting down of expectations regarding the quality and ecological function of natural areas closest to people's home and workplaces" (Miller, 2005). Given the prevalence of people living in neighborhoods of low biodiversity, the continual downward shift of these baselines is almost assured (Turner et al., 2004). Compounding this are the findings of Belaire et al. (2015), who found that people tended to underestimate the bird diversity in neighborhoods in Chicago, Illinois. This downward shift of baselines may have more pronounced effects on people of lower socioeconomic status, as neighborhoods of higher income often have greater biodiversity – a phenomenon known as the ‘luxury effect’ (Kinzig et al., 2005; Lerman, 2011; Martin et al., 2004; Melles, 2005).

Yards and neighborhoods often provide the greatest opportunity for urban populations to experience nature, and consequently are of importance in shaping the views of urban residents and combating the “extinction of experience”, which leads growing disconnect and apathy towards ecosystem health and conservation (DeStefano et al., 2003; Lerman, 2011; Pyle, 2003). This disconnect from nature has impacts on human health, and the ‘luxury effect’ may contribute to health inequalities between socioeconomic groups (Fuller et al., 2007; Miller, 2005; Mitchell et al., 2008; Turner et al., 2004).

Birds play an important role in connecting people to nature (Belaire et al., 2015). Bird watching is a popular hobby in the United States, with 47 million people participating in 2011 (Carver, 2013). By far, most of these birdwatchers (birders) did their birding around the home (41 million around the home, 18 million away), and spent 9 times as many days doing so (Carver, 2013). This interest in birds could be a tool to promote bird conservation from individual landowners up to governments, particularly when the sizeable financial impacts of birdwatchers on ecotourism are considered (Sekercioglu, 2002). Carver (2013), for instance, found that bird watching generated $107 billion in total industry output across the United States, including creating 666,000 jobs.

Indeed, successful efforts to attract wildlife to one’s yard (or by extension, city) may further encourage landowners to put more effort into wildlife-friendly practices, resulting in a positive feedback (Goddard et al., 2013). While features at the landscape scale, such as size, connectivity (or isolation), and composition of habitat patches, have significant effects on avian communities (Bolger et al., 1997; Galli et al., 1976; Melles, 2003; Opdam et al., 1984), several studies have shown that local scale habitat structure and composition (such as that of yards) are more important predictors (Chamberlain et al., 2007; Clergeau, 2001; Lerman, 2011). This presents opportunities for bottom-up conservation efforts in urban habitats, where numerous changes at a local scale can add up to larger scale impacts (Colding, 2007; Cooper et al., 2007).

The Lower Rio Grande Valley

Global trends in land conversion and population shift are apparent in the Lower Rio Grande Valley (LRGV) of southernmost Texas, which borders Tamaulipas, in northeastern Mexico. Comprised of Starr, Hidalgo, Willacy, and Cameron counties, the LRGV has undergone substantial land use changes since European-based settlements arrived in the in the 17th century (Brush, 2005; Leslie Jr, 2016). The human population has grown rapidly during the second half of the 1900's, rising from about 400,000 in the 1960s to about 1,350,000 as of 2015 (Leslie Jr, 2016; US Census Bureau, 2015). Population growth is expected to continue, potentially reaching 3 million by 2050 (Leslie Jr, 2016; Stubbs et al., 2003).

The associated land conversion in the LRGV was first from native riparian and thorn forests and savannah to agricultural tracts, a change augmented by the development of modern irrigation systems in the early 1900s and the creation of Falcon Dam in the 1950's (Brush, 2005; Jahrsdoerfer et al., 1988). In recent years urban areas have become a dominant aspect of land conversion, sometimes at the cost of agricultural land. By the late 1980s, an estimated 95% of native habitats had been cleared for agricultural and urban development (Jahrsdoerfer et al., 1988). Between 1993 and 2003, urbanization increased by 46% in Hidalgo, Cameron, and Willacy counties while irrigated land decreased by 7.6% (Huang et al., 2006).

This habitat loss threatens a highly biodiverse community of flora and fauna, including a subset of tropical species that reach their northernmost range limits in South Texas (Leslie Jr, 2016). Some of these species, like Plain Chachalaca (*Ortalis vetula*), White-tipped Dove (*Leptotila verreauxi*), and Couch's Kingbird (*Tyrannus couchii*), are fairly common in forested parts of the region, while others, like Altamira Oriole (*Icterus gularis*) and the rarer Hook-billed Kite (*Chondrohierax uncinatus*), remain more localized and uncommon. These "South Texas Specialty" birds (although some are also found in other limited parts of the United States), along with the speciose set of temperate, desert, and coastal birds found in the region, make the Lower Rio Grande Valley a well-known and highly visited birding destination. A study concluded that nature tourism in the Valley, of which birdwatching is a large component, brings in an estimated impact of $462,998,700 annually, a figure the authors noted as being conservative (Woosnam et al., 2012).

Restoration/re-vegetation efforts of previously agricultural lands by Texas Parks and Wildlife Department and the US Fish and Wildlife Service have been underway for over 30 years, with the focus on enhancing the connection of habitat fragments along the Rio Grande, particularly of riparian and thorn forest tracts. A study done in 2013-2014 found that a variety of forest bird indicator species occupy these restored woodlands with comparable frequency to remnant tracts of mature woodland (Brush and Feria unpublished data). This study was also the first to investigate (though not as its main purpose) the bird communities in suburban habitats and small patches of natural woodlands in cities, and found that some of these tropical affiliated, South Texas focal species readily used some of these areas. The presence of some of these predevelopment forest bird species, along with some primarily urban dwellers (such as the IUCN Redlist endangered Red-crowned Parrot (*Amazona viridigenalis*)), makes further study of these urban bird communities of interest for future conservation efforts. This is particularly true in the face of what is expected to be continued urban development in the region.

The goals of this thesis were to: 1) explore abundance patterns within avian communities in suburban and remnant urban woodlands, with a particular interest in native forest bird species, 2) asses how local habitat variables, such as number of native trees, affect avian communities in suburban habitats, and 3) put these results into the context of bird conservation in the Lower Rio Grande Valley. The ultimate goals of this study are to inspire and inform future lines of inquiry into urban bird research in this subtropical region, as well as to provide local municipalities and homeowners with practical, science-based recommendations to enhance conservation and ecosystem services associated with bird diversity.

CHAPTER II

THE BREEDING BIRD COMMUNITY OF A REMNANT URBAN WOODLAND IN MCALLEN, TX.

Abstract

Bird surveys at the McAllen Nature Center were done in 2015-2016 to assess the breeding bird community found in a remnant tract of thorn-forest in urban McAllen. Both the bird community and vegetative characteristics were compared with that of a large, natural reserve along the Rio Grande. 37 bird species were recorded as having a territory in the McAllen Nature Center, including 15 species not found at the large natural tract. 13 of these predominantly used the more open, human-maintained habitats the northern and western sides of the park, and not the natural central thicket. The presence of many common thorn-forest birds, including a subset of the regions “South Texas specialty” species, is encouraging. Small remnant woodlands such as the McAllen Nature Center may prove vital to keep these species present in the urban avian community of the Lower Rio Grande Valley.

Introduction

The amount of urban area is increasing worldwide, and in conjunction with this is a shift of human populations from rural to urban. Between 1970 and 2000, global urban area quadrupled to 58,000 km^2, and is projected to increase to 1,527,000 km^2 (as noted by the authors, an area roughly the size of Mongolia) by 2030 (Seto et al., 2011). In 2014 54% of people lived in urban areas, a number which is projected to increase to 66% by 2050 (United Nations, 2014). These land transformations and population shifts have notable and multi-scalar effects on the environment, from individual plant and animal species to the entire biosphere. The overall global trend is one of decreasing biodiversity in the face of increasing anthropogenic pressures (Butchart et al., 2010; Dirzo et al., 2014).

Global trends of increased urbanization can be seen in the Lower Rio Grande Valley of southernmost Texas. Comprised of Starr, Hidalgo, Willacy, and Cameron counties, the LRGV has undergone substantial land use changes since European-based settlements arrived in the in the 17th century (Brush, 2005; Leslie Jr, 2016). While early land conversion was primarily for agriculture, in recent years urban growth has become a dominant force of change. Between 1993 and 2003, urban area increased by 46% in Hidalgo, Cameron, and Willacy counties while irrigated land decreased by 7.6% (Huang et al., 2006). The human population has undergone rapid increase during the second half of the 1900's, rising from about 400,000 in the 1960s to about 1,350,000 as of 2015 (Leslie Jr, 2016; US Census Bureau, 2015). Population growth is expected to continue, potentially reaching 3 million by 2050 (Leslie Jr, 2016; Stubbs et al., 2003).

The region is known for its wealth of biodiversity, including a subset of mostly tropical species found little to nowhere else in the United States (Leslie Jr, 2016). Avian examples like the Green Jay, Olive Sparrow, and White-tipped Dove are what many call “South Texas specialty birds”, which are of great interest to visiting bird watchers. Birds are indeed a well-known taxon that is the source of much ecotourism, and are one of the main natural attractions of the Lower Rio Grande Valley (Mathis, 2004; Woosnam et al., 2012). Over 500 species of bird have been documented in the four counties (Starr, Hidalgo, Willacy, and Cameron), which when compared with entire states would make it one of the top 10 most speciose in the United States (eBird, 2016). Yet little is known about the avian community composition in the growing urban component of a region with only 5% of its native habitat remaining (Jahrsdoerfer et al., 1988).

McAllen is the largest city in Hidalgo County (the largest county in the LRGV) with an estimated population of 140,000 (US Census Bureau, 2016). Yet the city owns only two small publically accessible parks with remnant native vegetation; Quinta Mazatlan World Birding Center and the McAllen Nature Center. The latter, containing a larger portion of native habitat, had been closed to the public for a period between 2007 and 2014, only recently reopening to the public, and little is known about its bird community. As part of an Interlocal Cooperative Agreement with the City of McAllen, the author (JSB) did surveys of the bird community and vegetation to inform the conservation and management of the McAllen Nature Center (hereafter, MNC).

Study Area

The McAllen Nature Center is an about 20 acre (8 hectare) park located in an area of mixed development in McAllen, with large shopping plazas, a convention center, a sports and park complex, and neighborhoods nearby. The nature center has three distinct sections; an open park-like area with ringed by trees and denser scrub around the entrance (~ 2 ha), a “savannah” section with scattered trees on the west side (~ 1.5 ha), and a dense thicket of native thorn-forest in the center (~ 5 ha) (figure 2.1).

Figure 2.1: Aerial image of the McAllen Nature Center (January 2016).

Although the long-term history of the site is not fully documented, about half of this thicket had not been cleared since before the 1930s, and all of the thicket section has been left intact since the 1960s (The Valley Land Fund, 2012).

Methods

Bird Surveys

Surveys were done via the territory mapping method (Bibby et al., 1992). As Bibby et al. note, this method can provide estimates for absolute numbers of birds in an area. This method was also used in two earlier surveys of a 8 ha section of Santa Ana National Wildlife Refuge (SANWR, total size of refuge is 800 ha) in the 1970s and 1990s (Brush et al., 1998; Gehlbach, 1987), which allows for comparison between the bird community found in the McAllen Nature Center and that representative of a large natural tract only 17.5 km away. 10-12 bird surveys were done each year between late March and late July. Surveys took 1.5-2 hours to complete, and were conducted in the morning between 0800 and 1030 hrs. Observations of birds and their behavior (such as singing, gathering nesting material, and territorial interactions) were mapped, with particular effort made to document simultaneous observations of birds. This allows for greater accuracy in distinguishing separate territories. All common names follow the Fifty-seventh Supplement to the American Ornithologists' Union Check-list of North American Birds (Chesser et al., 2016).

At the end of each survey season, all data were entered into a web-based mapping software (Google Earth^{TM}, Menlo Park CA) and territories were drawn and counted. Clusters of bird observations by species were conservatively marked as territories, requiring at least three observations (preferably more) to qualify as a territory. Time of year for the observations was also taken into account, with more consideration for territories placed on middle of breeding season records than those on either the early or late extremes. However, a single record of a nest with eggs or young is sufficient to mark down a territory. For species that do not have traditional territories, or those that only defend a small area immediate to the nest, different methods were used. For species such as White-winged Dove (*Zenaida asiatica*), half of the maximum count of individuals was used to determine number of territories (which assumes that half of the individuals were males). For those species like European Starling (*Sturnus vulgaris*), which only defend an area immediately around their nests, the number of nests found was used. The Birds of North America Online species accounts (Rodewald, 2015) were used as references for the spacing and territoriality of birds found during surveys, along with looking at known habitat preferences. The number of territories was averaged over the two-year survey period.

Bird species lists and number of territories were compared with those of Brush and Cantu (1998), although observer differences may account for differences in number of territories estimated. The roughly 20 year difference between study periods is also worth noting. Thus, more emphasis was placed on comparisons between avian communities as a whole, along with the trends of abundances within the habitats of the MNC.

Vegetation Surveys

Survey methods were modified from those of Brush and Cantu (1998), ultimately based on the method of quantitative habitat description (James et al., 1970). Ten 0.05 ha circular plots (diameter = 25 m) were placed in the MNC. Six were randomly placed in the center thicket, two randomly placed in the park-like entrance, and two randomly placed in the savannah-like plot on the west side. Tree density, frequency, and percent canopy cover were determined along with shrub density and percent ground cover. At each of the ten plots all trees with > 8 cm diameter at breast height (DBH) were measured for DBH and height. Tree height was measured using an extendable pole marked in 0.5 m intervals, from 0-0.5 up to 14-14.5 m above the ground (estimated at heights > 5.5 m). Percent canopy cover was measured using the Canopy App (Version 1.0.2.) from the University of New Hampshire. Percent canopy cover was the average of 5 readings, one taken from the center of each plot and then four at points 12 meters out in each cardinal direction. Percent ground cover was measured using the Canopeo app (Version 2.0) from Oklahoma State University. Percent herbaceous ground cover was the average of 13 readings taken every 3 meters in cardinal directions from the center of the plot (one reading at the center of the plot as well). Shrub density (plants with < 8 cm DBH) and composition were measured by counting the number of stems intercepted by arms out-stretched while walking. Two transects through each plot were done (east-west and north-south). The number of “hits” per plot reflects the density and relative abundance of woody shrub species.

Results

Bird Surveys

37 bird species were recorded as having at least one partial territory in the McAllen Nature Center (table 2.1). Three of the most abundant bird species were doves, with White-winged Dove, White-tipped Dove, and Mourning Dove (averaging 12, 8, & 7.5 territories respectively), equaling 23% of all territories. Fifteen species of the total observed (40.5%) most commonly established territories in the open-park section of the MNC (table 2.1), 11 species were more found more commonly in the thicket, and only 1 in the savannah. The savannah section held by far the fewest number of total territories (9.75) when compared with the open-park (46.75), and central thicket (41.25) sections. Of the 10 remaining species, 6 averaged equal numbers of territories in at least two of the sections. Four were undetermined due to mapping methodology (for example, White-winged Dove was undetermined due to its number of territories derived from total abundance across the entirety of the MNC).

Comparisons in the Breeding Bird Communities

Only three bird species reported from the SANWR study area by Brush and Cantu (1998) were not found at the MNC – Carolina Wren, Black-bellied Whistling-Duck, and Red-shouldered Hawk (table 2.2). These three species were found in low numbers at the SANWR study area. An additional 15 species (equaling 25% of total territories found in the MNC) were found to have territories at the McAllen Nature Center that were not found during the 1994-1996 study at SANWR. Eleven of these additional species had the greatest number of territories in the open park section of the MNC (House Sparrow, Purple Martin, European Starling, Curve-billed Thrasher, Lesser Goldfinch, Clay-colored Thrush, Great-tailed Grackle, Western Kingbird, Green Parakeet, Tropical Kingbird, and House Finch). Three had their greatest number of territories in the thicket (Black-chinned Hummingbird, Verdin, and Cactus Wren). Only Brown-headed Cowbird had equal territories in two of the habitat types; park and thicket. All told, there was an overlap of 22 species between SANWR and the MNC, with three species only found at the former and 15 additional species found at the latter.

Table 2.1: The average number of bird territories at the MNC. Species ranked from largest to smallest. Asterisks indicate separation of territories into sections not possible.

Common Name	**Avg Total**	**Avg Park**	**Avg Savannah**	**Avg Thicket**
White-winged Dove	12	*	*	*
Northern Mockingbird	9.5	4	2	3.5
White-tipped Dove	8	1.5	0	6.5
Mourning Dove	7.5	*	*	*
House Sparrow	7	7	0	0
Olive Sparrow	7	2	0	5
Purple Martin	7	7	0	0
Northern Cardinal	7	1.5	0	5.5
Golden-fronted Woodpecker	5	3.5	0.5	1
Brown-crested Flycatcher	4.5	2	0.75	1.75
Plain Chachalaca	3.5	1.75	0	1.75
Great Kiskadee	3.5	1	1	1.5
Couch's Kingbird	3	1	1.5	0.5
Yellow-billed Cuckoo	3	0.5	1	1.5
European Starling	2.75	2.5	0.25	0
Long-billed Thrasher	2.5	0	0	2.5
Curve-billed Thrasher	2.25	1.5	0.25	0.5
Brown-headed Cowbird	2	1	0	1
Bronzed Cowbird	2	1.25	0	0.75

Ladder-backed Woodpecker	2	0.5	0.5	1
Lesser Goldfinch	2	1	1	0
White-eyed Vireo	2	0.5	0	1.5
Groove-billed Ani	1.5	1	0.5	0
Verdin	1.5	0	0	1.5
Black-chinned Hummingbird	1.25	0.5	0	0.75
Clay-colored Thrush	1.25	1.25	0	0
Common Ground-Dove	1.25	0	0	1.25
Buff-bellied Hummingbird	1	*	*	*
Black-crested Titmouse	1	0.5	0	0.5
Green Jay	1	0	0	1
Great-tailed Grackle	1	0.5	0.5	0
Western Kingbird	1	1	0	0
Greater Roadrunner	0.5	*	*	*
Green Parakeet	0.5	0.5	0	0
Tropical Kingbird	0.5	0.5	0	0
Cactus Wren	0.25	0	0	0.25
House Finch	0.25	0	0	0.25

Table 2.2: Average number of territories for SANWR and the MNC. Species ranked from most breeding territories to least at SANWR (Brush et al., 1998).

Species	**SANWR (1994-1996)**	**MNC (2015-2016)**
White-winged Dove	35	12
Olive Sparrow	17.7	7
Mourning Dove	13.2	7.5
White-tipped Dove	12.3	8
Plain Chachalaca	10.8	3.75
Golden-fronted Woodpecker	8.3	5
Long-billed Thrasher	7.8	2.5
White-eyed Vireo	5.5	2
Couch's Kingbird	5.3	3
Ladder-backed Woodpecker	5	2
Brown-crested Flycatcher	5	4.5
Black-crested Titmouse	4.7	1
Green Jay	4	1
Great Kiskadee	3.5	2.5
Yellow-billed Cuckoo	3	3
Bronzed Cowbird	3	2

Northern Cardinal	2.8	6.5
Groove-billed Ani	2.3	1.5
Northern Mockingbird	1	9.5
Common Ground-Dove	0.3	1
Greater Roadrunner	0.3	1
Buff-bellied Hummingbird	0.3	1
Additional SANWR Species		
Black-bellied Whistling-Duck	0.3	
Red-shouldered Hawk	0.2	
Carolina Wren	1	
Additional MNC Species		
House Sparrow		7
Purple Martin		7
European Starling		2.75
Curve-billed Thrasher		2.25
Brown-headed Cowbird		2
Lesser Goldfinch		2
Verdin		1.5
Black-chinned Hummingbird		1.25
Clay-colored Thrush		1.25
Great-tailed Grackle		1
Western Kingbird		1
Green Parakeet		0.5
Tropical Kingbird		0.5
Cactus Wren		0.25
House Finch		0.25

Vegetation Surveys

Vegetation in the MNC varied markedly between the three sections. Vegetation was densest in the central thicket, with greater values of percent canopy cover, tree density, and shrub density (table 2.3). Tree species richness was also higher in the central thicket as compared with the open sections of the MNC. Mean tree height and percent herbaceous ground cover were lowest in the central thicket, which was predominantly covered by leaf litter.

Table 2.3: Vegetation measurements taken at the MNC.

		Section	
Vegetation Variable	**Park**	**Savannah**	**Thicket**
Percent Canopy Cover	38.1	27.9	49
Mean Canopy Height (m)	10.6	6.1	5.5
Tree Density (n/ha)	40	60	147
Mean Tree DBH (cm)	64.3	23.1	25.4
Tree Species Richness	3	2	5
Shrub Hit Density (n/ha)	0	10	6180
Percent Ground Cover	27%	11.6%	5.7%

The most common tree was mesquite (*Prosopis glandulosa*), which made up 65% of all trees (table 2.4). 94% of all trees were natives, with the only non-native trees found in the northern park-like section. Granjeno (*Celtis pallida*) was the most abundant shrub species with 56% of all observations belonging to this species, followed up by snake eyes (*Phaulothamnus spinescens*), lotebush (*Ziziphus obtusifolia*) and coyotillo (*Karwinskia humboldtiana*) (table 2.5). Nearly all of the shrubs "hits" were in the thicket section.

Table 2.4: Frequency of tree species at the MNC. Asterisk indicates non-native species.

Tree Species	**Count**
mesquite (*Prosopis glandulosa*)	33
Texas ebony (*Ebenopsis ebano*)	4
coma (*Sideroxylon celastrinum)*	3
granjeno (*Celtis pallida*)	3
live oak (*Quercus virginiana)**	2
tepeguaje (*Leucaena pulverulenta*)	2
anacua (*Ehretia anacua*)	1
brasil (*Condalia hookeri*)	1
Montezuma bald cypress (*Taxodium mucronatum*)	1
Washington fan palm (*Washingtonia robusta*)*	1

Table 2.5: Relative abundance of shrub species in the MNC.

Shrub Species	**Observations**
granjeno (*Celtis pallida*)	1038
snake eyes (*Phaulothamnus spinescens*)	531
lotebush (*Ziziphus obtusifolia*)	165
coyotillo (*Karwinskia humboldtiana*)	44
brasil (*Condalia hookeri*)	34
guayacan (*Guaiacum angustifolium)*	11
Texas persimmon (*Diospyros texana*)	10
Barbados cherry (*Malpighia glabra*)	9
lime prickly-ash (*Zanthoxylum fagara*)	8
Texas lantana (*Lantana urticoides*)	4
Turk's cap (*Malvaviscus arboreus*)	1

Comparisons in Vegetation

Because the park and savannah sections of the MNC are mowed and maintained, comparisons with SANWR were only done with the central thicket. Percent canopy cover was similar between locations with 47% at SANWR and 49% at the MNC (table 2.6). More and smaller trees were found at SANWR, as evidenced by greater tree density at SANWR but greater mean DBH at the MNC (figure 2.2). Tree species richness was considerably greater at SANWR.

Table 2.6: Comparison of vegetation variables between SANWR and the MNC.

Vegetation Variable	**SANWR**	**MNC**
Canopy Cover	47%	49%
Mean Canopy Height (m)	6.1	5.5
Tree Density (n/ha)	396	147
Mean DBH (cm)	17.9	25.4
Tree Species Richness	13	5
Shrub Density n/ha	14180	6180
Percent Ground cover	No data	5.7

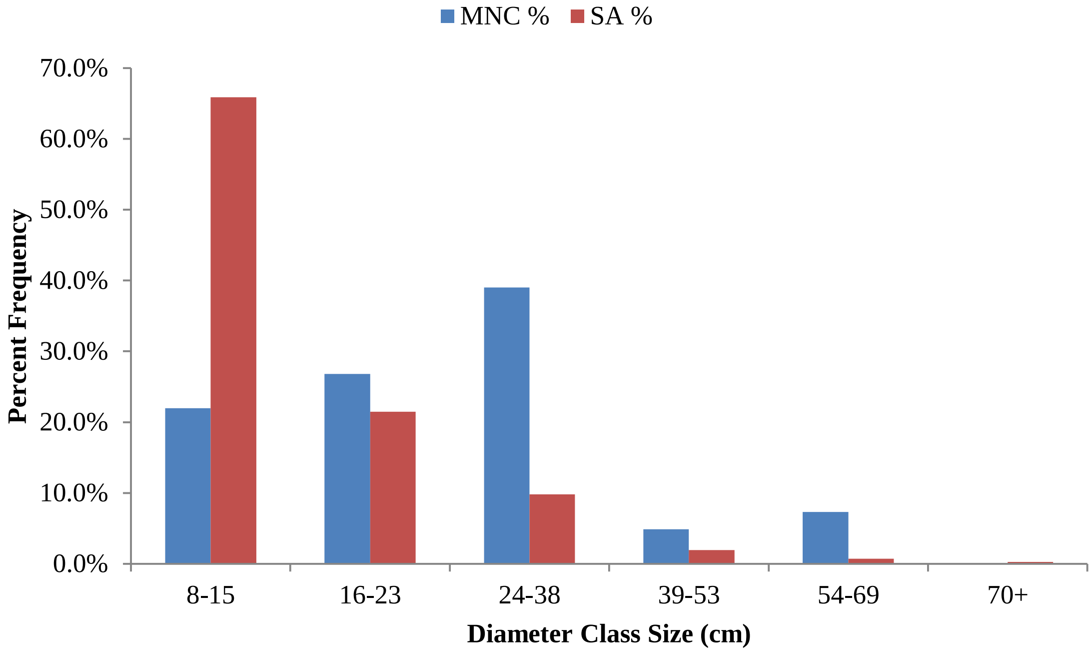

Figure 2.2: Frequency of trees by diameter class size. SANWR n = 419, MNC n = 41.

Discussion

The bird community at the MNC is a mix of species commonly found in thorn-forest habitats, lower thorn-scrub, and urban settings. The central thicket, dominated by thorny plant species such mesquite and granjeno, supports many of the thorn-forest birds commonly found at large, exurban reserves like SANWR. South Texas species dependent on dense habitats such as Olive Sparrow, Long-billed Thrasher, and White-tipped Dove predominantly utilized the central thicket, although the ring of dense vegetation lining the open-park section also housed smaller numbers of these species.

There were 3 species with established territories only at SANWR (not observed at the MNC), but there were 15 species found at the MNC but not SANWR (hereafter “MNC-only” species). Six of these MNC-only species are well-known to exploit urban habitats. House

Sparrows were dependent on nest sites provided by the Purple Martin houses in the open park-like habitat, and European Starlings similarly exploited nest sites excavated by Golden-fronted Woodpeckers and the forage opportunities provided by open grassy areas as well as outside the park. While present in low numbers on the edges of the MNC, Great-tailed Grackle seemed to predominantly use the open areas of the park as foraging grounds, presumabley nesting just off-site. Green Parakeet, a Mexican species presumed to have naturally spread but is limited to urban areas of south Texas (Brush, 2005), had one probable nest in 2015, but was not seen again in 2016. One other common urban exploiter throughout most of the United States, the House Finch, has begun appearing with more frequency in the Lower Rio Grande Valley, but no breeding records have been obtained. While a pair was seen on two occasions (including one in which they appeared to be examining potential nest sites in a Washintonian palm), no nest was found. However, the species met the requirements for a cluster, and it is likely they bred elsewhere on site or just off it.

Seven other MNC-only species, including Western Kingbird, Curve-billed Thrasher, and Lesser Goldfinch, are urban adaptable species that do well in mixed habitats with treed and open spaces. One species, the Clay-colored Thrush, is a relatively recent arrival to the Lower Rio Grande Valley, with the first breeding record in the 1980s (Brush, 2005). It is now well-documented at a variety of urban woodlands across the region as well as at SANWR (eBird, 2016). Cactus Wren and Verdin were the sole MNC-only species that were documented only using the central thicket. These two species breed at SANWR in smaller numbers, in areas dominated by lower thorn-scrub vegetation (Brush, 2005)

Surprisingly, given their use of human-altered environments, no Black-bellied Whistling-Ducks attempted to nest on-site. The probable cause is the lack of large cavities suitable for the species. Carolina Wren, which is known to breed in other urban woodlands in the LRGV (JSB personal observation), was also absent. This species may require lusher riparian and associated woodlands (Brush, 2005). The other absentee, Red-shouldered Hawk, is a locally rare breeder, having declined since the 1950s (Brush, 2005), and is currently known to nest in only one or two locations in the LRGV (Brush, 2008). All three absent species are known to be urban adaptable species in other parts of their range.

Conclusions & Management Implications

The presence of many common thorn-forest birds in the McAllen Nature Center is encouraging, showing that even small islands of natural habitat in the increasingly urban landscape can provide suitable habitat for some of our "South Texas specialty" birds. Conserving remnant urban woodlands may prove the best conservation strategy for keeping birds dependent on dense-vegetation, such as Long-billed Thrasher and Olive Sparrow, in the urban avian community. Urban woodlands, like the MNC, may also act as stepping stones, islands, or corridors (or all three) depending on their proximity to other urban woodlands and individual species requirements (Davis et al., 1978), particularly relevant when considering metapopulations dynamics. However, it is important to note that the thorn forest species present in the MNC are representative of abundant to common birds in native forests in the region; uncommon regional specialties such as Altamira Oriole, Northern Beardless-Tyrannulet (*Camptostoma imberbe*), and Gray Hawk (*Buteo plagiatus*), were not detected in urban woodlands during this study. Future investigations on common forest bird species dispersal, immigration and emigration in these urban woodlands "islands" are warranted.

While European Starlings and House Sparrows are non-native, invasive species, their presence in the park is not expected to have broad negative effects on native cavity nesters in the

MNC. Although European Starlings are known to compete with native woodpecker and secondary cavity-nester species (Kerpez et al., 1990; Weitzel, 1988), a meta-analysis concluded that only sapsuckers (*Sphyrapicus* species) exhibited declines that could potentially be related to starling competition (Koenig, 2003). In Seattle, lower densities of cavity-nesting species relative to nearby wildlands was attributed to lack of live trees and snags, while competition from European Starlings accounted for only 5 nest failures out of 117 nests found (77% were reproductively successful) (Blewett et al., 2005). House Sparrows, however, present a problem for Purple Martins (Jackson et al., 1974), and removal of their nests from Purple Martin houses is recommended.

Should revegetation projects be undertaken to expand the thorn-forest of the MNC, plantings of mesquite, granjeno, snake eyes, lote bush, and other common woody species already found in the dense remnant thicket are recommended, and could expand habitat available to "South Texas specialty" birds. The savannah section of the MNC harbored the fewest number of total territories, so expansion of native thorn forest to already treed portions of this section is encouraged. Invasive grasses, guineagrass (*Urochloa maxima*) and bufflegrass (*Pennisetum ciliare*) were a dominant presence in the savannah section and the edges the MNC, but were largely absent from the densest thickets of native vegetation. Removal of invasive plants and avoiding habitat disturbance in and around the central thicket are recommended to preserve the integrity of the native habitat.

The presence of urban adapters and urban exploiters, subsequently raising species richness, appears in line with previous findings in that areas of intermediate disturbance can support increased bird diversity (Marzluff, 2008). In addition, the park-like and savannah habitats at the MNC provided suitable habitat for many of these species. Other urban woodlands with somewhat different habitat conditions should be studied to get an idea of the generality of this study across the LRGV and similar areas. The scale of this study does not allow for a true comparison of bird diversity across a gradient of highly urban to natural exurban habitats. Future research should focus on population dynamics, comparing ecological processes affecting birds in these small urban woodlands to those of natural exurban tracts.

CHAPTER III

SUBURBAN BIRD DIVERSITY OF A SUBTROPICAL REGION: ABUNDANCE PATTERNS AND EFFECTS OF LOCAL HABITAT FEATURES

Abstract

Breeding bird surveys were conducted in residential neighborhoods and urban woodlands in McAllen and Edinburg, Hidalgo County, Texas, in 2015 and 2016. Overall bird diversity increased linearly with vegetative cover and decreased with increasing impervious surface and grass cover. While the majority of species found in residential habitats can be considered urban adapters, retention of native forest birds was significantly related to percent vegetative cover. Forest birds that were uncommon in residential habitats were detected more frequently at sites with greater than 45% vegetative cover, while others were only found in small urban woodlands. Increased tree diversity, particularly of native trees, was shown to have significant positive effects on the avian diversity and native forest bird richness. The abundance of ash trees (*Fraxinus* species) was found to have a significant positive effect on cavity-nesting bird abundance. Simple management decisions like increasing vegetative cover and the number of native trees planted can help support native bird diversity in the Lower Rio Grande Valley's growing urban component.

Introduction

Urbanization, in which human populations transform landscapes and shift to city and town environments, has increased drastically over the past 50 years. More than half of the global population now live in urban areas (United Nations, 2014), densely occupying an area over 58,000 km^2 (Seto et al., 2011). These figures are projected to increase to over 66% of people and 1,527,000 km^2 by the mid-21st century (Seto et al., 2011; United Nations, 2014). Uttara et al. (2012) detail many of the environmental impacts resulting from the concentration of population in these urban areas, including implications of changes to the atmosphere, hydrosphere, and biosphere. For example, through this drastic habitat modification, urbanization is often associated with increased extinction rates and loss of native species (McKinney, 2002), especially since global populations tend to concentrate in low elevation, and coastal regions (Seto et al., 2013).

Birds, a well-known taxon, make good indicator species to study the effects of urbanization (Blair, 1999). Patterns of avian communities are often studied along urban-rural gradients, looking at how communities change from highly developed urban cores to rural native habitats (Beissinger et al., 1982; Blair, 1996; Bolger et al., 1997; Donnelly et al., 2006; Marzluff et al., 2001; McDonnell et al., 2008; Melles, 2003). Bird species tend to respond to urbanization in one of three ways. Blair (1996) grouped birds into the categories of urban exploiters, urban adapters, and urban avoiders. Urban exploiters (sometimes called synanthropic species) are able to thrive with the changes brought by urbanization, and indeed may not be found (or found in lower abundances) in natural, exurban habitats. The abundance of urban exploiters often leads to a peak of bird density (though not in species richness) in highly developed urban areas (Beissinger et al., 1982; Blair, 1996; Chace et al., 2006; Emlen, 1974; Marzluff, 2008).They often are belong to granivorous and omnivorous foraging guilds ((Beissinger et al., 1982; Chace et al., 2006; McKinney, 2002).

Urban adapters are species capable of occupying intermediate levels of urbanization. Oftentimes they are early successional species, or generalists with broad habitat tolerances (Emlen, 1974; Marzluff, 2008). Urban avoider species are those intolerant of urban development, requiring natural habitats (whether they are deserts, prairies, or forests), and often belong to insectivorous foraging guilds (Chace et al., 2006; McKinney, 2002). Consequently, they should be a focus for conservation of bird diversity in urbanizing areas (Blair, 2001).

While human actions have considerable negative impacts of biodiversity at a global scale (Dirzo et al., 2014), some research has shown that bird species richness peaks in intermediate zones of urbanization, such as suburbs/residential areas (Blair, 1996; Chace et al., 2006; Concepción et al., 2015; Marzluff, 2008; McKinney, 2002). This phenomenon, often attributed to the intermediate disturbance hypothesis, occurs through the colonization of urban exploiter and urban adaptor species, and the retention of some urban avoider species (Marzluff, 2008). However, much of this research has been done in temperate climates, while studies in tropical and subtropical regions is scant (Marzluff et al., 2001).

Features at both landscape (such as size, connectivity, and composition of habitat) and local scales (such as number of trees, native vs non-native plants) have been shown to impact avian community responses (Bolger et al., 1997; Chamberlain et al., 2007; Galli et al., 1976; Melles, 2003; Opdam et al., 1984), though some studies have shown that local features may be more important predictors (Clergeau, 2001; Clergeau et al., 1998; Lerman, 2011; Miller et al., 2003). These local scale changes, such as increased native vegetation, greater tree coverage, and more diverse vegetative structure, can increase the diversity and presence of native bird species, including those associated with predevelopment habitats (Aurora et al., 2009; Belaire et al., 2014; Chace et al., 2006; Chamberlain et al., 2007; Donnelly et al., 2006; Evans et al., 2009; Lerman, 2011; Melles, 2003; Mills et al., 1989).

The Lower Rio Grande Valley of Texas, USA, is a subtropical region known for its wealth of biodiversity, including a subset of mostly tropical species found nowhere else or in very limited areas elsewhere in the United States (Leslie Jr, 2016). Yet an estimated 95% of native habitat has been cleared for agriculture, and increasingly, urban development (Jahrsdoerfer et al., 1988). In an effort to inform conservation efforts in this region, bird surveys were conducted in residential areas in McAllen and Edinburg, TX, two of the largest cities in this region, to determine patterns of response of the local avian community to landscape features associated with urbanization. In this chapter, this research is reported in two parts: (Part A) To determine patterns of avian community response across a gradient of urbanization, avian abundance was compared to local-scale land cover in neighborhoods and small urban woodlands. This data enabled further exploration of the effects of vegetation structure and composition in yards on various bird guilds. As a result, with emphasis on providing local municipalities and homeowners with useful information for management, a Canonical Correspondence Analysis (CCA) was performed with selected vegetation variables as an exploratory method, which formulated the analyses in Part B.

Part A: Patterns of Suburban Bird Diversity

Methods

Study Area

Data collection of bird abundance and diversity was conducted in residential neighborhoods and tracts of urban woodlands within the city limits of McAllen, TX, and Edinburg, TX. Sample points were placed in non-randomly selected neighborhoods (but points themselves were random) of predominantly single-family residences of varying tree cover, with at least 200 meters between each point. Five of the points were placed in tracts of urban woodlands ranging from 1-5 hectares in size, and 50 were placed in residential neighborhoods for a total of 55 points (figure 3.1A). For each point, percent cover of vegetation (trees, large woody shrubs), grass, and impervious structure in a 100 m radius was estimated by an unsupervised image classification of 2014 satellite imagery downloaded from the National Agriculture Imagery Program (1 m^2 pixels) (ArcMap V. 10)

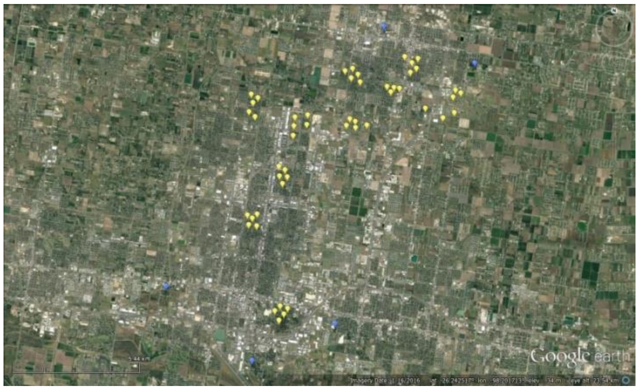

Figure 3.1A: Survey points in McAllen and Edinburg. Yellow = residential points, blue = urban woodland points.

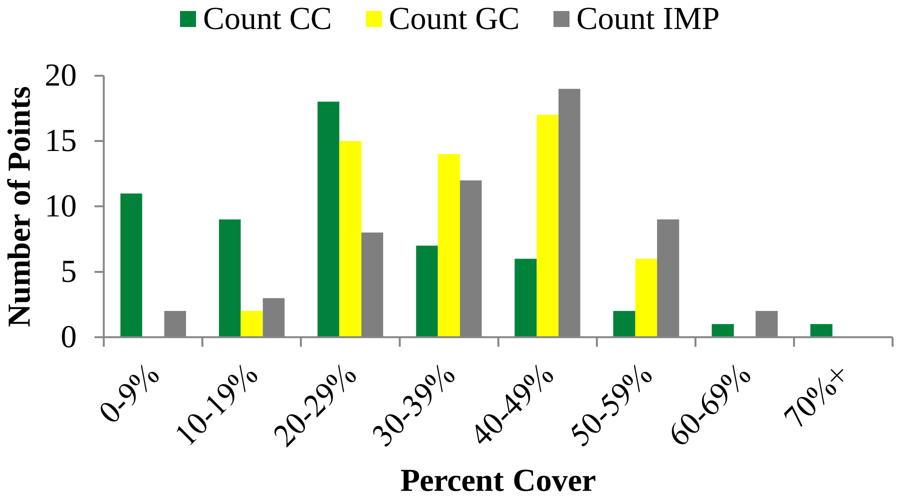

Figure 3.2A: Distribution of points by percent cover. CC = canopy cover, GC = grass cover, IMP = impervious cover.

Points ranged from 2-88% vegetation cover, 12-58% grass cover, and 16-61% impervious cover (figure 3.2A).

Bird Surveys

Three fixed-radius (50 m) point count surveys of breeding birds were conducted at each location in April-July of 2015 and 2016 (for a total of 6 counts per site during the observation period). Fixed point counts are useful in that inferences can be made about habitat preferences of bird species and/or communities (Bibby et al., 1992), and offer a simple yet effective approach to control for distance-based detectability bias (Hutto, 2016). All birds detected via sight and sound were recorded for 10 minutes at each point. Birds were only recorded if they were detected in the habitat within 50 meters, thus, birds flying through the circle were not counted. This method excluded swallows and swifts (Purple Martin was also not included due to their reliance on the presence of martin houses). Migratory and non-breeding species, such as various warblers, and the breeding species Black-bellied Whistling-Duck and Yellow-crowned Night-Heron, were also excluded.

Data Analysis

Bird species relative abundance was expressed as an average of the six surveys per point, and total abundance at each point was the sum of these averages for all species. A category of native forest bird species (n = 19) was designated; this category was determined by taking species that had greater than 25% frequency (Brush and Feria unpublished data) in native thorn-forest habitats at Santa Ana National Wildlife Refuge, an 800 ha tract along the Rio Grande. Bird species were grouped into (a) foraging guilds and (b) foraging height guilds based off previous studies reviewed in the Birds of North America Online (Rodewald, 2015). For species not included in this review, additional literature review and personal observations were considered. Due to bird surveys being conducted during the breeding season, emphasis was placed on their foraging habits in the breeding cycle.

Avian diversity was measured for each point using total bird species richness (S), total abundance (A), and Shannon diversity (H) The Shannon diversity of bird communities at each point was also calculated. Relationships between bird diversity and urbanization features (percent vegetation cover, percent grass cover, and percent impervious cover within 100 m around each point) were explored using linear regressions. Forest species retention was explored by developing a presence-absence matrix and estimating the presence thresholds for native forest species. Thresholds were determined based off of logistic regressions (the point where probability of presence of a species exceeded that of absence) performed for bird species that were neither ubiquitous nor scarce. Here, a ubiquitous species is defined as being present at 50 or more points (> 90% frequency), and scarce is defined as being present at 5 or fewer points (< 10% frequency). All analyses were performed with JMP 13 Pro statistical software (Cary, NC).

Results

Thirty-nine bird species were observed across all sites at least once between 2015 and 2016 (table 3.1A). All names follow the Fifty-seventh Supplement to the American Ornithologist's Union Check-list of North American Birds (Chesser et al., 2016). Three species were present at all points (Great-tailed Grackle, House Sparrow, and Northern Mockingbird), while four were only seen once (Blue Jay, Groove-billed Ani, Lark Sparrow, and Verdin). Of the nineteen forest species, 18 (94.7%) were detected in these urban sites, with eight of them occurring on more than half (≥ 28) of the study points – the only bird undetected from the from the native forest species category was Carolina Wren. Common urban exploiter and adapter species (House Sparrow, White-winged Dove, and Great-tailed Grackle) were most abundant.

Table 3.1A: Bird species detected on 2015-2016 surveys. Sorted by most sites detected to least. * = forest bird.

Code	Common Name	Scientific Name	No. Sites Detected	Abundance	StdDev
GTGR	Great-tailed Grackle	*Quiscalus mexicanus*	55	3.65	2.41
HOSP	House Sparrow	*Passer domesticus*	55	5.11	3.43
NOMO	Northern Mockingbird*	*Mimus polyglottos*	55	1.44	0.93
WWDO	White-winged Dove*	*Zenaida asiatica*	53	5.04	4.31
GKIS	Great Kiskadee*	*Pitangus sulphuratus*	50	0.75	0.87
GFWO	Golden-fronted Woodpecker*	*Melanerpes aurifrons*	49	1.12	1.01
BBEH	Buff-bellied Hummingbird	*Amazilia yucatanensis*	48	0.46	0.62
MODO	Mourning Dove*	*Zenaida macroura*	43	0.46	0.81
EUST	European Starling	*Sturnus vulgaris*	38	0.82	1.48
INDO	Inca Dove	*Columbina inca*	38	0.46	0.76
COKI	Couch's Kingbird*	*Tyrannus couchii*	37	0.33	0.63
LEGO	Lesser Goldfinch	*Spinus psaltria*	32	0.34	0.74

BCTI	Black-crested Titmouse*	*Baeolophus atricristatus*	30	0.22	0.59
BCFL	Brown-crested Flycatcher*	*Myiarchus tyrannulus*	30	0.28	0.60
CCTH	Clay-colored Thrush	*Turdus grayi*	28	0.27	0.55
EUCD	Eurasian Collared-Dove	*Streptopelia decaocto*	20	0.28	0.70
LBWO	Ladder-backed Woodpecker*	*Picoides scalaris*	18	0.08	0.27
CBTH	Curve-billed Thrasher	*Toxostoma curvirostre*	17	0.11	0.36
WEKI	Western Kingbird	*Tyrannus verticalis*	13	0.07	0.28
PLCH	Plain Chachalaca*	*Ortalis vetula*	10	0.14	0.61
NOCA	Northern Cardinal*	*Cardinalis cardinalis*	9	0.11	0.46
GREJ	Green Jay*	*Cyanocorax yncas*	7	0.03	0.20
LOSH	Loggerhead Shrike	*Lanius ludovicianus*	6	0.02	0.13
TRKI	Tropical Kingbird	*Tyrannus melancholicus*	6	0.03	0.22
BHCO	Brown-headed Cowbird	*Molothrus ater*	5	0.05	0.26
WEVI	White-eyed Vireo*	*Vireo griseus*	5	0.05	0.30
YBCU	Yellow-billed Cuckoo*	*Coccyzus americanus*	5	0.02	0.13
BROC	Bronzed Cowbird*	*Molothrus aeneus*	4	0.02	0.13
LBTH	Long-billed Thrasher*	*Toxostoma longirostre*	4	0.05	0.29
WTDO	White-tipped Dove*	*Leptotila verreauxi*	4	0.05	0.29
BCHU	Black-chinned Hummingbird	*Archilochus alexandri*	2	0.01	0.08
GREP	Green Parakeet	*Psittacara holochlorus*	2	0.05	0.63
HOOR	Hooded Oriole	*Icterus cucullatus*	2	0.01	0.08
OLSP	Olive Sparrow*	*Arremonops rufivirgatus*	2	0.05	0.27
ROPI	Rock Pigeon	*Columba livia*	2	0.02	0.17
BLJA	Blue Jay	*Cyanocitta cristata*	1	0.00	0.06
GBAN	Groove-billed Ani	*Crotophaga sulcirostris*	1	0.00	0.06
LASP	Lark Sparrow	*Chondestes grammacus*	1	0.00	0.06
VERD	Verdin	*Auriparus flaviceps*	1	0.00	0.06

Eight of the 10 most abundant species were granivorous and omnivorous (table 3.2A). However, 16 (43.3%) of primarily insectivorous species (which include species that eat large numbers of spiders and other arthropods) were present at lower abundances. Of common birds occupying nectarivorous and herbivorous foraging guilds, only Buff-bellied Hummingbird and Plain Chachalaca (respectively) were considered common.

Ground foragers were the most abundant, but several mid-story to canopy feeding species were also present at a number of points, such as Black-crested Titmouse and Brown-crested

Flycatcher. Across all sites, granivores were 5.1 times (sum = 11.99) more abundant than insectivores (with a summed abundance of 11.99 versus 2.35, Table 3.2). Omnivores were 3 times more abundant than insectivores.

Table 3.2A: Foraging and foraging height guilds for 39 species detected on 2015-2016 point counts. * = forest bird species.

Common Name	**Food Habit**	**Foraging Height**	**No. Sites Detected**	**Abundance**
Great-tailed Grackle	Omnivore	Ground	55	3.65
House Sparrow	Granivore	Ground	55	5.11
Northern Mockingbird*	Omnivore	Ground	55	1.44
White-winged Dove*	Granivore	Ground-Low	53	5.04
Great Kiskadee*	Omnivore	Low-Mid	50	0.75
Golden-fronted Woodpecker*	Omnivore	Ground-High	49	1.12
Buff-bellied Hummingbird	Nectarivore	Low-Mid	48	0.46
Mourning Dove*	Granivore	Ground	43	0.46
European Starling	Insectivore	Ground	38	0.82
Inca Dove	Granivore	Ground	38	0.46
Couch's Kingbird*	Insectivore	Mid-High, Aerial	37	0.33
Lesser Goldfinch	Granivore	Ground-Low	32	0.34
Black-crested Titmouse*	Insectivore	Mid-High	30	0.22
Brown-crested Flycatcher*	Insectivore	Mid-High	30	0.28
Clay-colored Thrush	Insectivore	Ground-High	28	0.27
Eurasian Collared-Dove	Granivore	Ground	20	0.28
Ladder-backed Woodpecker*	Insectivore	Low-High	18	0.08
Curve-billed Thrasher	Insectivore	Ground	17	0.11
Western Kingbird	Insectivore	Low-Mid, Aerial	13	0.07
Plain Chachalaca*	Frugivore, Herbivore	Low-High	10	0.14
Northern Cardinal*	Granivore, Insectivore	Ground-Mid	9	0.11
Green Jay*	Omnivore	Ground-High	7	0.03
Loggerhead Shrike	Insectivore, Carnivore	Ground-Low	6	0.02
Tropical Kingbird	Insectivore	High, Aerial	6	0.03
Brown-headed Cowbird	Granivore	Ground	5	0.05
White-eyed Vireo*	Insectivore	Mid	5	0.05
Yellow-billed Cuckoo*	Insectivore	Mid-High	5	0.02
Bronzed Cowbird*	Granivore	Ground	4	0.02
Long-billed Thrasher*	Insectivore	Ground-Low	4	0.05
White-tipped Dove*	Granivore, Frugivore	Ground	4	0.05

Black-chinned Hummingbird	Nectarivore	Low-High	2	0.01
Green Parakeet	Frugivore, Granivore	Mid-High	2	0.05
Hooded Oriole	Insectivore	Low-High	2	0.01
Olive Sparrow*	Granivore, Insectivore	Ground	2	0.05
Rock Pigeon	Granivore	Ground	2	0.02
Blue Jay	Omnivore	Ground-High	1	0
Groove-billed Ani	Insectivore	Ground-Mid	1	0
Lark Sparrow	Granivore	Ground	1	0
Verdin	Insectivore	Low-Mid	1	0

Species diversity generally decreased with increases in impervious surfaces. Shannon diversity (H) ($F(1,53) = 35.7$, $p < 0.0001$, $r^2 = 0.40$) and total species richness (S) ($F(1,53) = 30.0$, $p < 0.0001$, $r^2 = 0.36$) exhibited significant negative relationships with increasing cover by impervious surfaces (figure 3.3A), while total abundance (A) had a negative but nonsignificant relationship.

Relatedly, both Shannon diversity ($F(1,53) = 64.9$, $p < 0.0001$, $r^2 = 0.55$) and total species richness ($F(1,53) = 47.9$, $p < 0.0001$, $r^2 = 0.47$) had positive, significant relationships with increasing canopy cover (figure 3.4A). Total abundance had a positive but nonsignificant relationship, although a curvilinear relationship may have been present where a peak of total abundance occurred at points with between 22% and 33% canopy cover (figure 3.5A).

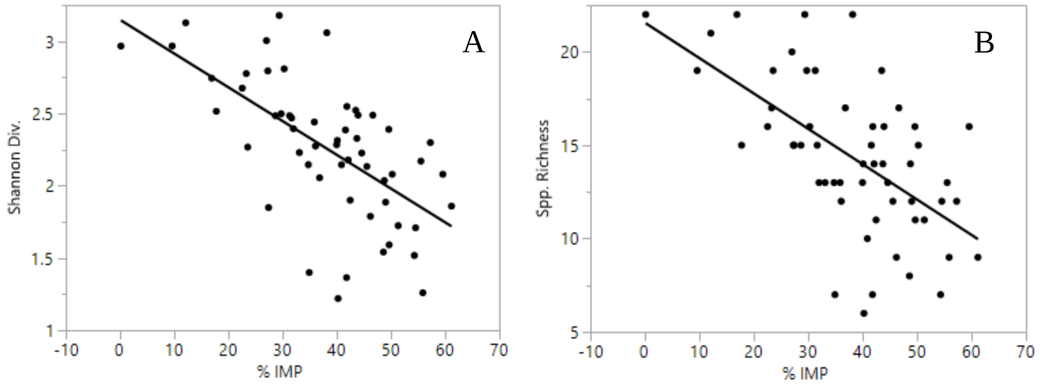

Figure 3.3A: (A) Shannon diversity and (B) species richness regressed against percent impervious cover.

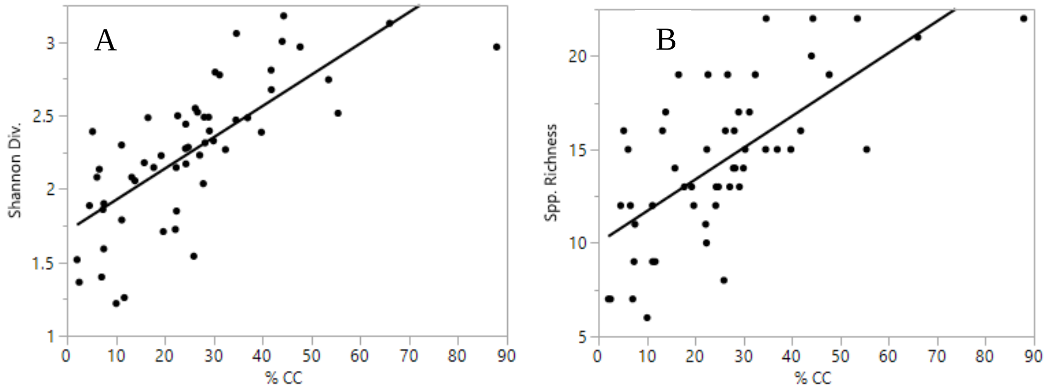

Figure 3.4A: (A) Shannon diversity and (B) species richness regressed against percent canopy cover.

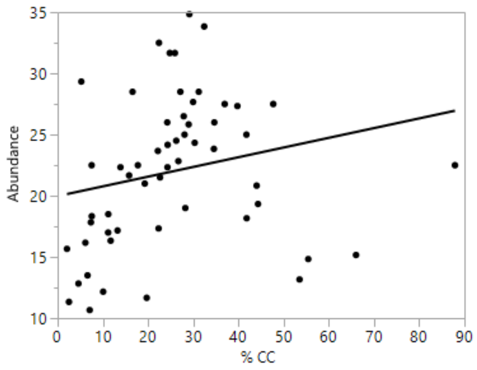

Figure 3.5A: Regression of total bird abundance with percent canopy cover.

Shannon diversity (F(1,53) = 10.1, p = 0.0025, r^2 = 0.16) and species richness (F(1,53) = 7.7, p = 0.0076, r^2 = 0.13) had negative significant relationships with increasing grass cover (figure 3.6A), while abundance exhibited a negative, nonsignificant relationship. Regressions with % vegetation cover exhibited the strongest lines of fit, followed by those with % impervious and % grass cover, respectively.

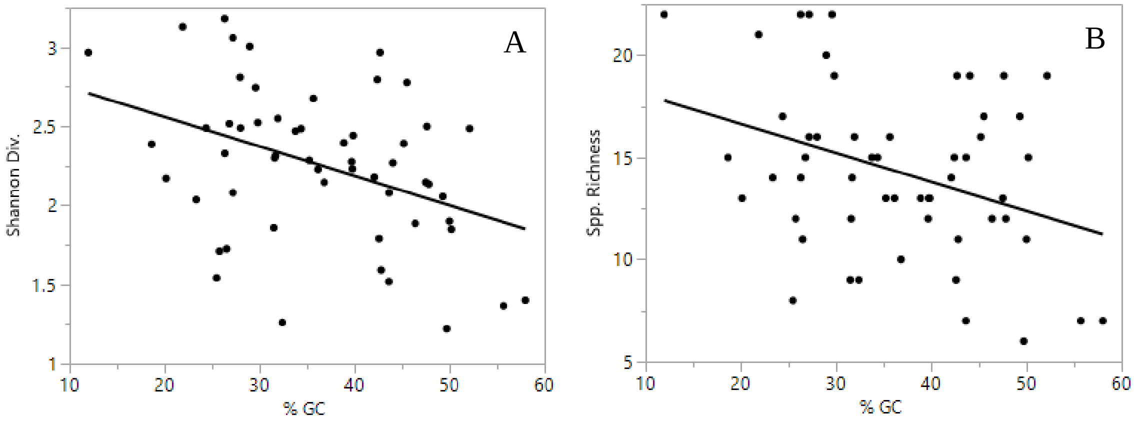

Figure 3.6A: (A) Shannon diversity and (B) species richness regressed against percent grass cover.

Thresholds for 10 native forest bird species were determined by logistic regression, from Mourning Dove (MODO) to Green Jay (GREJ) (figure 3.7A). The curved line is visually estimated to represent the presence thresholds for each species. For example, at values above 15% canopy cover, Couch's Kingbird was expected to be present.

Discussion

In contrast with some previous urban-rural gradient studies, this study focused primarily on the gradient of vegetation cover within residential neighborhoods and its effects on a subtropical bird community at a local scale (3.1 hectares). Some species, such as Great-tailed Grackle and House Sparrow, are known to be urban exploiters in other regions (Blair, 1996; Donnelly et al., 2006; Wehtje, 2003), consistent with results from this study where these species were observed in high relative abundances and associated with more open, urbanized habitats.

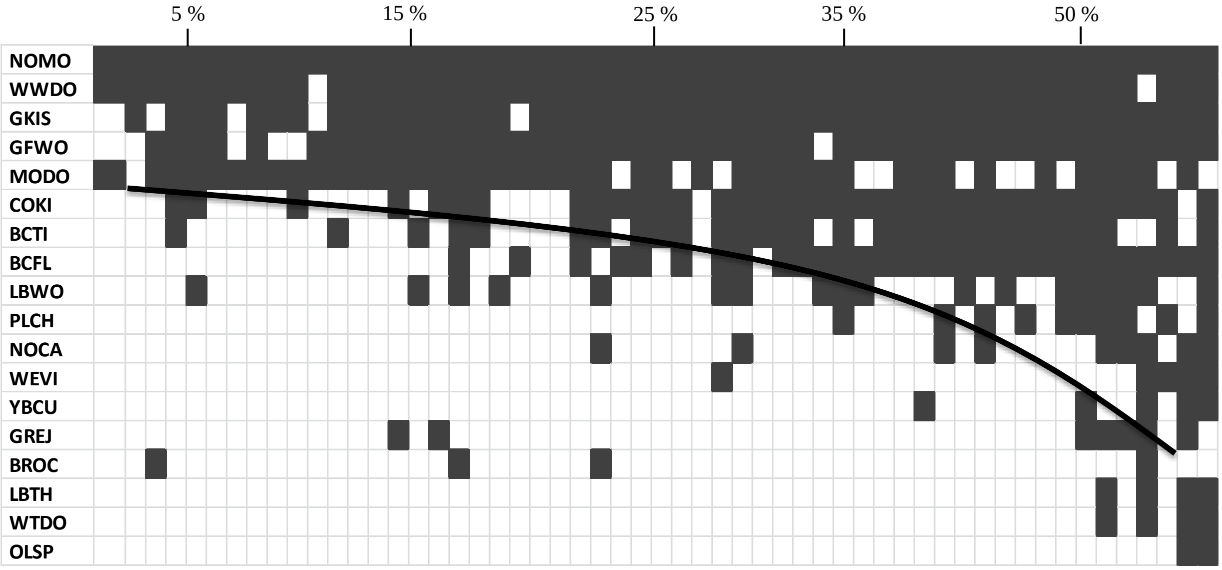

Figure 3.7A: Native forest species retention based off percent canopy cover. Columns = increasing values of percent canopy cover (left to right), darkened squares = species presence.

Given the suburban settings and urban woodland settings of this study, most of the species present can be thought to be urban adapters of varying degrees. Northern Mockingbird, White-winged Dove, Great Kiskadee, and Golden-fronted Woodpecker were nearly ubiquitous across study points. The above are common species in the LRGV, known to be present in a variety of habitats (Brush, 2005). Historically, White-winged Dove used to be limited to the southern regions of south Texas and the southwest but have expanded north, seemingly due to their ability to adapt to human-altered environments (Brush, 2005). The abundance of Golden-fronted Woodpecker may be particularly important for secondary cavity-nesters given that nest boxes were not frequently encountered (JSB personal observation), and should be considered when investigating abundance patterns of urban cavity-nesters.

Other species, such as Plain Chachalaca, Northern Cardinal, and Green Jay, though uncommon, were still recorded in residential areas, typically those areas of greater than 45% vegetative cover. Should the percent cover of vegetation increase, I would expect these species to occur more frequently and in greater abundances. Other adapters included several commonly sought-after species by visiting bird watchers, such as Couch's Kingbird, Tropical Kingbird, and Clay-colored Thrush. The latter two represent recent (within ~ past 30 years) additions to the LRGV's avifauna (Brush, 2005). Clay-colored Thrush in particular has seemed to expand into residential habitats in McAllen and Edinburg, having occurred at more than half of all survey points. Couch's Kingbird was present at 67% of study points (most of these points with greater than 15% canopy cover) and was the most common kingbird species encountered, though Tropical Kingbird populations seem to be increasing in the region (Brush, 2005). Buff-bellied Hummingbirds, another sought-after species by visiting birders, were present at 87% of study points, exemplifying that suburban settings are amiable to the species.

Despite being abundant across the LRGV, thorn-forest species like Olive Sparrow, Long-billed Thrasher, and White-tipped Dove were only present urban woodlands, and can be considered urban avoider species in the context of this study. However, their presence in these urban woodlands is encouraging, and indicates that perseveration of small habitat “islands” in urban areas may allow these South Texas specialty birds to retain a foothold in our cities, as the dense habitat that they require is unlikely to be provided in residential habitats where open lawns and scattered trees dominate. This reiterates results found in Chapter II in that preservation and/or restoration of dense woodlands are expected to be important for this group of species.

The abundance of ground feeding granivorous and omnivorous species is in line with results of studies done in Ohio (Beissinger et al., 1982) and Arizona (Emlen, 1974), and supports general trends of these avian trophic guilds responses to urbanization (Chace et al., 2006; McKinney, 2002). The most abundant insectivore was European Starling, which predominantly feeds on the ground. However, several other insectivores remained fairly common, including Couch’s Kingbird, Black-crested Titmouse, Brown-crested Flycatcher, and Clay-colored Thrush. Overall, however, insectivorous species remained less abundant than granivorous and omnivorous species, which supports conclusions found in related studies. The scarcity of nectarivorous species is expected in context of the region’s avian community, as only two species of hummingbird (Buff-bellied Hummingbird and Black-chinned Hummingbird) regularly breed.

Marzluff (2008) found a unimodal curve related to species richness along an urban-rural gradient, where the peak of species richness was in areas of intermediate development (suburbs) in Seattle. However, percent canopy in those intermediate zones of development ranged between 40-60%, whereas the majority of points in McAllen and Edinburg were below 40% canopy cover (46/55 survey points). This study limitation may account for the linear relationships between species diversity and vegetative cover reported in this study, as I did not investigate the full gradient from predominantly impervious urban centers to natural, exurban habitats. Given the strength of fit between percent vegetative cover and these avian community metrics (as opposed to those of impervious and grass cover), it follows that many neighborhoods in McAllen and Edinburg are on the low end of the urban vegetation gradient. Accordingly, gains in vegetative cover may have greater benefits for bird diversity than simply limiting percent impervious structure, which is strongly correlated with the number of buildings (houses) in residential neighborhoods, as road width remains fairly standard. Indeed, over half (10/18) of native forest bird species tended to be absent when percent vegetative cover dropped below 40% at individual study points.

Some species, for example Loggerhead Shrike and Eurasian Collared-Dove, favored more open, lawn dominant neighborhoods. Loggerhead Shrike is a declining species throughout much of its range in the US (Yosef, 1996), while Eurasian Collared-Dove is a recent non-native invader (Brush, 2005). The shift of neighborhoods over time from newer, more open landscapes to older, lusher ones is something to consider, as the bird community will consequently shift.

Conclusions and Recommendations

Increasing percent vegetative cover in McAllen and Edinburg should be the primary goal to increase bird diversity and forest species richness in residential neighborhoods. As recommended by Donnelly (2006), efforts should focus on native forest species, as they are the most negatively impacted by urbanization. Thresholds for native forest species that were more uncommon (but still present) in residential areas, such as Green Jay and Plain Chachalaca, suggest that aiming for at least 45% canopy cover even at a local scale (< 1 hectare) in neighborhoods will have positive effects for native forest species.

Given the hierarchical nature of linkages between spatial scales (Savard et al., 2000), it is expected that local increases in vegetative cover will scale up to having larger cumulative effects. Chamberlain (2007) found that bird species richness in London greenspaces was increased with greater proportions of adjacent gardens (yards), which effectively expanded the area of green space. While this trend was only present at sites < 1 hectare, it demonstrates cumulative effect of changes at the scale of yard. In Arizona, Lerman (2011) postulated that increased native vegetation in yards might explain the persistence of native birds in Phoenix. It should be noted, however, that creating a homogeneous level of vegetative cover at all landscapes is not recommended. Maintaining habitat heterogeneity is important since thresholds for occurrence of native bird species will vary, with some species preferring less or more densely vegetated habitats. Instead, as noted by Marzluff (2008) and Donnelly (2006), maintaining a heterogeneous mix of habitats will help preserve a greater diversity of native forest and native adapter bird species, thus combating homogenization of bird communities.

For species that were not present in residential neighborhoods, like Long-billed Thrasher and Olive Sparrow, conservation of urban woodlands is important, specifically through their provision of dense vegetation not found in residential neighborhoods. Urban woodlands in many respects act as habitat islands in the urban mosaic sea. As expected by the theory of island biogeography, increasing the size of urban woodland patches accordingly increases the number of bird species present (Donnely, 2003; Galli et al., 1976; Tilghman, 1987). These islands support different foraging and nesting guilds, such as insectivores, carnivores, and ground-nesting species (Tilghman, 1987), which are often less abundant in urban settings (Chace et al., 2006; McKinney, 2002). Increasing the connectivity of these urban woodland islands, whether by decreasing distances between patches or making the urban mosaic more passable for dispersing birds (for example, by increasing vegetative cover in residential neighborhoods), is recommended.

The paucity of urban woodlands in McAllen and Edinburg highlights the need to identify and conserve those that remain, and restore or afforest available greenspaces. Tools such as the Zonation conservation planning software (Gordon et al., 2009; Moilanen et al., 2014), or a combination of eBird (eBird, 2016) and i-Tree (www.itreetools.org) (Lerman et al., 2014) could be used to identify important areas for conserving bird diversity, particularly for uncommon urban adapters and urban avoider species.

This study serves as a preliminary exploration of abundance patterns in urban avian communities, and can serve as a baseline to direct future explorations. More research examining bird communities across a full urban-rural gradient would be useful in expanding knowledge of avian responses in a subtropical environment, and should be expanded to include cities in Cameron, Starr, and Willacy counties. The growing urban component of the LRGV offers opportunities for conservation efforts, and should continue to be studied as the region continues to develop.

Part B: Effects of Local Vegetation Composition on a Residential Avian Community

Methods

Vegetation Surveys

In addition to bird surveys and land cover seen in Part A, local habitat composition and structure was measured at each residential survey point in 2016. Two concentric circles covering 0.02 and 0.08 hectares (diameters of ~ 16 and ~ 32 meters) were visually estimated around each point (such as in Donnelly & Marzluff, 2006). The roadway itself was not counted as part of the study plots (road width averaged 8.9 m, SD = 1.4 m). In the 0.02 ha plot, I visually estimated the area of coverage by native and non-native shrubs and herbaceous plants (multiplying foliage height by width), and the percent of ground coverage by grass and herbaceous plants. In the 0.08 plot, trees were identified, counted, and heights were estimated to the nearest 0.5 m.

The number of houses within 100 m of each point, and average age of houses was also measured. Houses were counted via satellite imagery, and housing age was quantified by averaging the four nearest residences around each survey point with built dates obtained from the Hidalgo County Appraisal District (www.hidalgoad.org).

Vegetation variables were chosen based off previously explored factors known to affect or be correlated with bird community characteristics, particularly those that would translate easily to management decisions by homeowners and municipalities (table 3.1B).

Table 3.1B: Vegetation variables selected for CCA and further analysis.

Variable Code	**Description**
IMP 100	percent impervious cover within 100m
GC 100	percent grass cover within 100m
CC 100	percent canopy cover within 100m
Avg. Age	average age of 4 nearest houses
#STR	Number of houses within 100 meters
ShrbNat	estimate of area (m^2) covered by native shrubs
ShrbN-n	estimate of area (m^2) covered by non-native shrubs
Tree SR	tree species richness
#NTree	number of native trees within 16 meters
#N-nTre	number of non-native trees within 16 meters
TH 2-6.5m	Count of trees between 2-6.5 meters
TH 7-11.5m	Count of trees between 7-11.5 meters
TH 12-16m	Count of trees between 12-16 meters
THDiv	Tree height shannon diversity
Ash	Count of ash trees within 16 m
LiveOak	Count of live oak trees within 16 m

Data Analysis

Relationships between relative bird species abundance in residential neighborhoods and vegetation variables were explored by Canonical Correspondence Analysis (CCA) with Canoco5 software (Ter Braak et al., 2012). Bird species seen on 4 or less (< 10%) of residential point plots were not included, leaving 23 species included in the CCA (table 3.2B). Vegetation variables were selected for correlations with various bird species and guilds based on the significance ($p < 0.05$) of their simple effects prior to forward-selection of CCA (Monte Carlo randomization test, 499 permutations, Canoco 5) (table 3.3B). See part A for regressions with percent cover within 100 m (CC100, IMP100, and GC100). Correlations were performed with finer-scale variables, following the CCA. All correlations were performed with JMP Pro 13 statistical software.

Table 3.2B: Bird species used in CCA. * = native forest birds.

Code	Common Name	Foraging Guild	Foraging Height
BBEH	Buff-bellied Hummingbird	Nectarivore	Low-mid
BCFL	Brown-crested Flycatcher*	Insectivore	Low-mid
BCTI	Black-crested Titmouse*	Insectivore	Mid-high
CBTH	Curve-billed Thrasher	Insectivore	Ground
CCTH	Clay-colored Thrush	Insectivore	Ground, low
COKI	Couch's Kingbird*	Insectivore	Mid-high
EUCD	Eurasian Collared-Dove	Granivore	Ground
EUST	European Starling	Insectivore	Ground
GFWO	Golden-fronted Woodpecker*	Omnivore	Ground-high
GKIS	Great Kiskadee*	Omnivore	Low-mid
GTGR	Great-tailed Grackle	Omnivore	Ground
HOSP	House Sparrow	Granivore	Ground
INDO	Inca Dove	Granivore	Ground
LBWO	Ladder-backed Woodpecker	Insectivore	Low-high
LEGO	Lesser Goldfinch	Granivore	Ground-low
LOSH	Loggerhead Shrike	Insectivore	Ground-low
MODO	Mourning Dove*	Granivore	Ground
NOCA	Northern Cardinal*	Granivore	Ground-mid

NOMO	Northern Mockingbird*	Omnivore	Ground
PLCH	Plain Chachalaca*	Herbivore	Ground-high
TRKI	Tropical Kingbird	Insectivore	Mid-high
WEKI	Western Kingbird	Insectivore	Low-mid
WWDO	White-winged Dove*	Granivore	Ground-low

Table 3.3B: Vegetation variables selected for correlations determined by forward selection of CCA.

Variable	**Contribution %**	**F**	**P**
CC100	39.1	14.1	0.002
Avrg. Age	31.3	10.6	0.002
TH 7-11.5m	25.8	8.4	0.002
#STR	20.4	6.4	0.002
GC100	17.2	5.3	0.002
IMP100	14.2	4.3	0.002
Live Oak	13	3.9	0.006
TH Div.	11.9	3.6	0.006
Ash	9.4	2.8	0.018
#Ntree	8.9	2.6	0.016

Results

The five most abundant trees were non-native. Live oak (*Quercus virginiana*, 24.9%) and ash (*Fraxinus spp.*, 9.2%) were the most abundant trees, accounting for more than a third (34.1%) of all trees. Royal palm (*Roystonea cubensis*) and *Washingtonia* palms (*Washingtonia* spp.) accounted for 10.7% of all trees, and crape myrtle (*Lagerstroemia indica*, 4.0%) rounded off the top five.

The five most common native trees were wild olive (*Cordia boissieri*, 3.72%), mesquite (*Prosopis glandulosa*, 2.42%), sabal palm (*Sabal mexicana*¸ 2.26%), sugar hackberry (*Celtis laevigata*, 2.10%), Texas ebony (*Ebenopsis ebano*, 1.94%), and anacua (*Ehretia anacua*, 1.62%).

In total, non-natives accounted for 83% of all trees (n = 619), while natives accounted for 17%. Trends of dominance by non-natives continued in area covered by shrubs. Non-native shrubs accounted for 87.2% of all shrub area, while natives accounted for 12.8%.

The results of the CCA show the relationships between bird species and local habitat variables are shown as bird species scores (points) and habitat variables (arrows) (figure 3.1B.). The length of the arrow reflects the relative importance of each variable. Thus, long arrows reflect more important habitat variables than shorter arrows. Axis I, with high correlations with CC100 and AvgAge, reflects a gradient between points in older developments with more vegetation and ones in newer, more open (less vegetation) developments. Axis I explained 25.8% of the variation in the data, while Axis II explained 8.2%. Axis II appeared to reflect differences in vegetation composition, including the number of native trees, species richness, and area of native shrubs.

More than half (14/23) bird species used in the analysis were on the more vegetated side of the gradient. Eight out of ten native forest species were also located on the more vegetated side of the gradient, with only Mourning Dove and Northern Mockingbird populating the less vegetated side.

The effects of Axis I (percent vegetation cover (CC100), percent impervious cover (IMP100), and percent grass cover (GC100)) were analyzed in Part A. Closely correlated relationships, such as percent vegetation cover (CC100) and average age of development (AvgAge), and impervious cover (IMP100) and (#STR), are worthy to note and interpret. Tree species richness (TreeSR), number of native trees (#NTree), and both ash and live oak trees (Ash & LiveOak) appeared to be important along axis II, which is assumed to reflect differences in vegetation composition.

Both bird Shannon diversity (r = 0.43, n = 50, p = 0.0016) and forest species richness (r = 0.40, n = 50, p = 0.0037) had positive, significant correlations with the number of native trees (figure 3.2B), which is turn had a significant positive correlation with tree species richness (see figure 3.1B, r = 0.63, n = 50, p <0 .0001). The two most abundant trees, Live oak (*Quercus virginiana*) and ash (*Fraxinus* species), had non-significant relationships with both bird Shannon diversity and forest species richness.

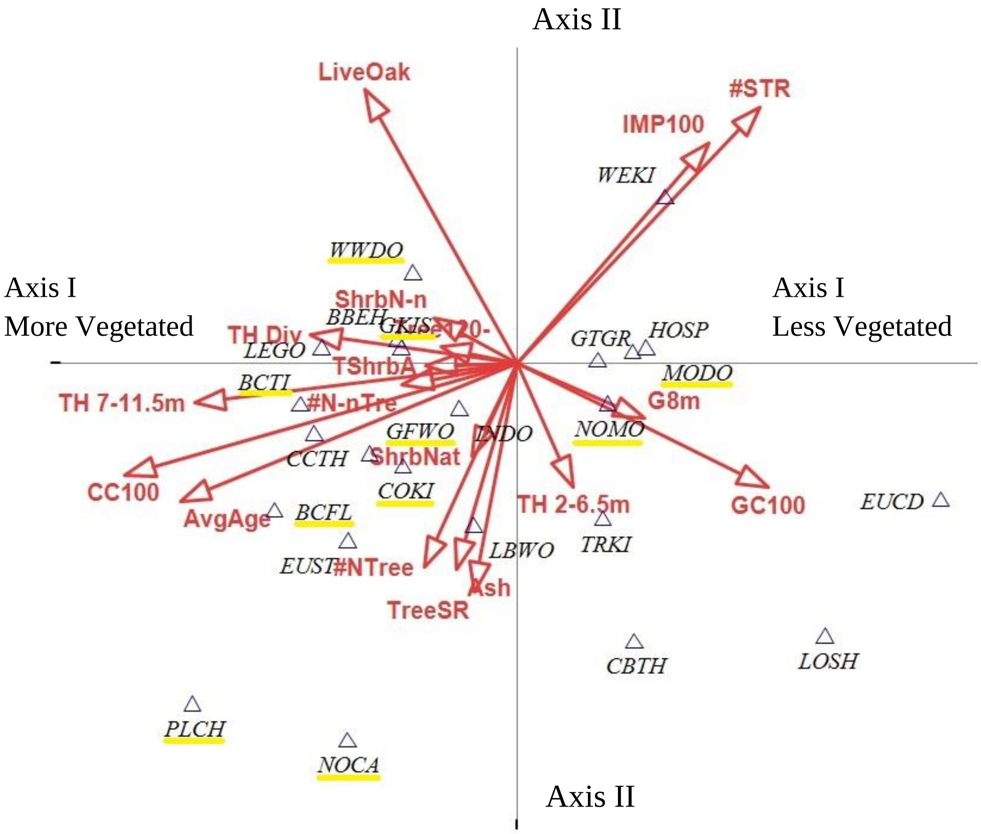

Figure 3.1B: CCA ordination of 25 species as they related to selected local habitat variables. Native forest species are underlined yellow.

However, given the somewhat clustered grouping of cavity nesters on the CCA (EUST, BCFL, BCTI, and GFWO), I further explored cavity nester abundance related to local habitat. In particular, given that ash trees (*Fraxinus* species) are commonly used in the region as landscape plants and readily provide dead knots of wood in which cavity nesters to excavate, I did a regression analysis of cavity nester abundance and number of ash trees. There was a positive, significant relationship between total cavity nester abundance and increasing numbers of ash trees (r = 0.47, n = 50, p = 0.0005) (figure 3.3B).

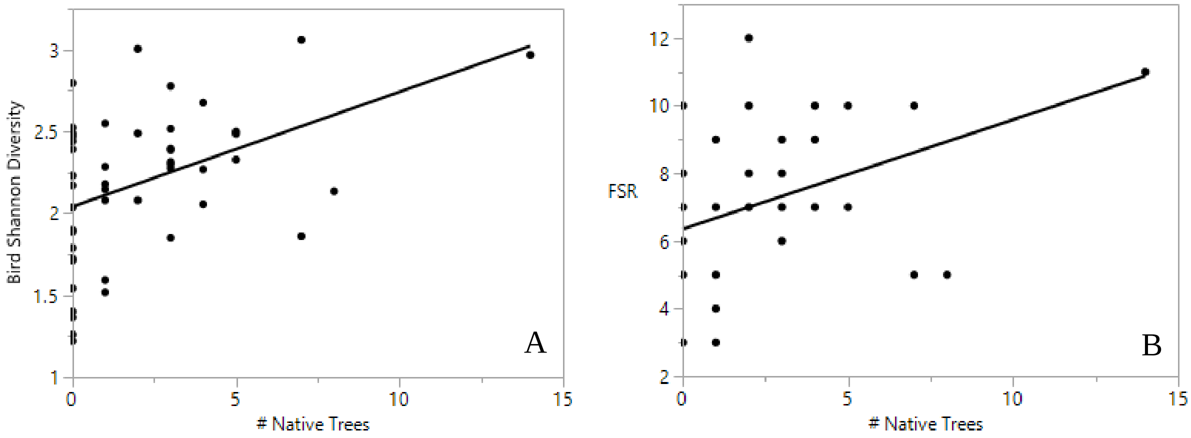

Figure 3.2B: Correlations of bird Shannon diversity (A), and forest species richness (B) with number of native trees.

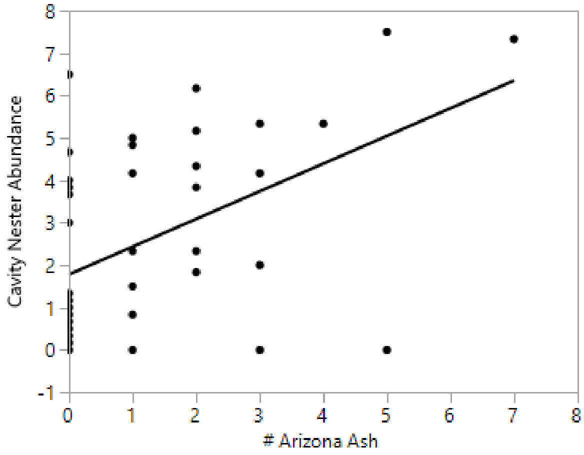

Figure 3.3B: Correlation of cavity nester abundance and number of ash trees.

Both forest species richness and bird Shannon diversity showed positive, significant relationships with the number of trees 7-11.5 meters in height and tree height Shannon diversity figure 3.4B.

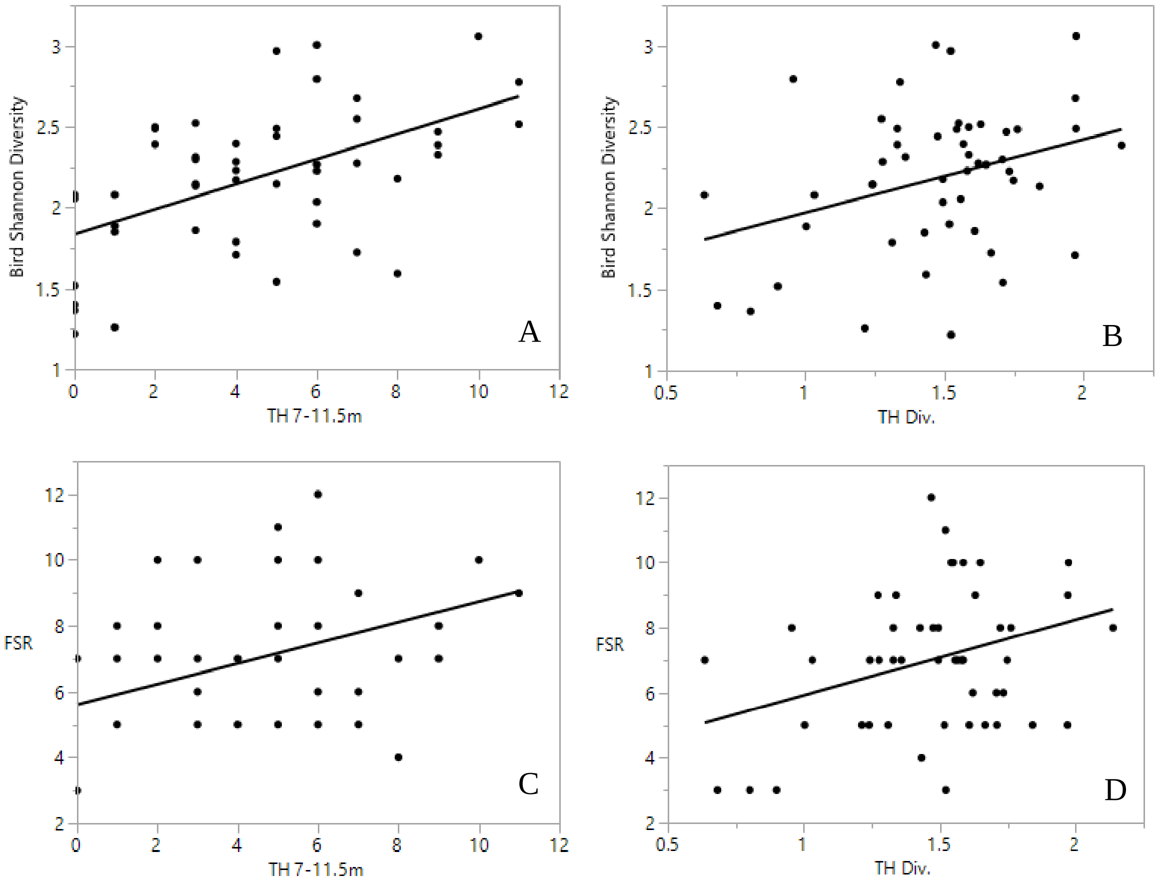

Figure 3.4B: Bird Shannon diversity correlation with trees 7-11.5 m in height (A) and tree height Shannon diversity (B). Forest species richness correlated with trees 7-11.5 m in height (C), and tree height Shannon diversity (D).

Discussion

Home gardens and yards have been shown to play a significant role in determining patterns of abundance in native bird species (Belaire et al., 2014; Burghart et al., 2008; Daniels et al., 2006; Lerman, 2011), as they drive local vegetation structure and composition. This study corroborates those findings in the context of bird community assembly in urban environments of subtropical south Texas. Compositional changes such as increased tree species richness and abundance, particularly of native trees, had significant correlations with bird diversity and native forest birds. This is noteworthy especially for native trees, given their relative scarcity in McAllen and Edinburg neighborhoods. Results also indicate that planting trees between 7 and 11.5 meters in height is recommended for attracting native forest species, although this may simply be an artifact of older, taller trees typically having greater canopy cover. Indeed, trees between 7-11.5 m in height were correlated with local canopy cover.

Shrub abundance and composition (native vs non-native), however, did not show significant correlations with bird diversity and forest species richness. This is contrary to results found in a few other studies that show shrubs have notable effects on urban bird species and communities (Azerrad et al., 2001; Belaire et al., 2014; Burghart et al., 2008; Daniels et al., 2006; Goldstein et al., 1986; Lerman, 2011; Melles, 2003). A potential explanation for this is due to landscaping style in the study areas. Many shrubs are planted along the edge of houses, which on average (a post-study exploration) were 11.1 meters (SD 2.7) from the road. However, while there was a correlation between increasing distance from road and less shrub area, it was not significant ($p = 0.12$). Another factor to consider is that front and back yards may be landscaped differently. In Arizona, front yards frequently took the form of more “socially correct” desert landscaping, while backyards were more lush (Larsen et al., 2006). In temperate Chicago, Belaire et al. (2015) found that backyards provided more resources for wildlife than front yards, where the landscaping was most affected by socioeconomic characteristics of residents. Further investigation of the role of shrubs in residential habitats in the LRGV should be conducted.

These results have important implications for urban planners and homeowners in residential habitats, and will help inform decisions to enhance biodiversity in the LRGV's growing urban component. For example, efforts to plant native trees in urban yards may have tremendous implications for bird conservation in the LRGV, especially for native forest birds that are commonly sought after by bird watchers.

The correlation between average age of neighborhood and canopy cover can be interpreted as the result of the most common development pattern in the region, where existing vegetation is almost entirely removed before housing is constructed. When possible, saving pre-development trees could allow for newer developments to more rapidly support a greater diversity of native birds, and/or potentially mitigate the effects of development.

Ash trees proved to be an important habitat component for the cavity nesting birds. Cavity–nesting species have been found to decrease in some suburban areas (Blewett et al., 2005), so the presence of cavities provided by ash trees may be important in providing nest sites. Particularly, because ash trees can provide cavities while the tree is still alive, they can circumvent the commonplace removal of dead trees in urban areas. It should be noted, however, that the commonly planted Arizona ash is not native, and that the native ash tree (*Fraxinus berlandieriana*) may provide the same benefits, along with likely hosting increased insect diversity.

CHAPTER IV

CONCLUSION AND REVIEW OF RECOMMENDATIONS

Conclusions & Recommendations

During 2015-2016, bird and vegetation surveys were conducted in residential neighborhoods and urban woodlands in McAllen, TX and Edinburg, TX. Results from this study support a number of studies that suggest local scale habitat changes have significant effects on the pattern and abundance of urban bird communities (Bolger, 2001; Burghart et al., 2008; Chamberlain et al., 2007; Clergeau, 2001; Lerman, 2011; Melles, 2005). When these small scale changes are systematically implemented, whether incentives are given top-down from municipalities or bottom-up via grassroots movement, they may scale up to have large scale impacts (Colding, 2007; Cooper et al., 2007).

Describing the pattern of avian community response in the Lower Rio Grande Valley helps fill in the gaps of knowledge about avian responses to urbanization in subtropical regions, and can be useful to local conservation agencies. What follows is a review of three recommendations to support bird diversity and native forest birds for municipalities and homeowners.

1) Increase vegetative cover in suburban areas in the Lower Rio Grande Valley to above at least 45%. Residential neighborhoods in the McAllen and Edinburg area remain low in regards

to percent vegetative cover. The linear nature of relationships between various bird diversity metrics and vegetative is telling in context of the species richness curve found by Marzluff

(2008); neighborhoods I surveyed may not have even reached the intermediate peak of species richness typically associated with suburban habitats.

However, maintaining habitat heterogeneity is important, and local municipalities should work to develop different strategies for the major land-use and land cover types throughout a city, from the dense city core (highly developed) to suburbs (intermediate development), to city edges (low development). As per McKinney (2002), we are in the phase of conservation strategy that requires both restoration of managed lands and the acquisition and protection of natural lands.

Figure 3.8A (re-printed below) should be consulted when planning for increasing the presence of common native forest species.

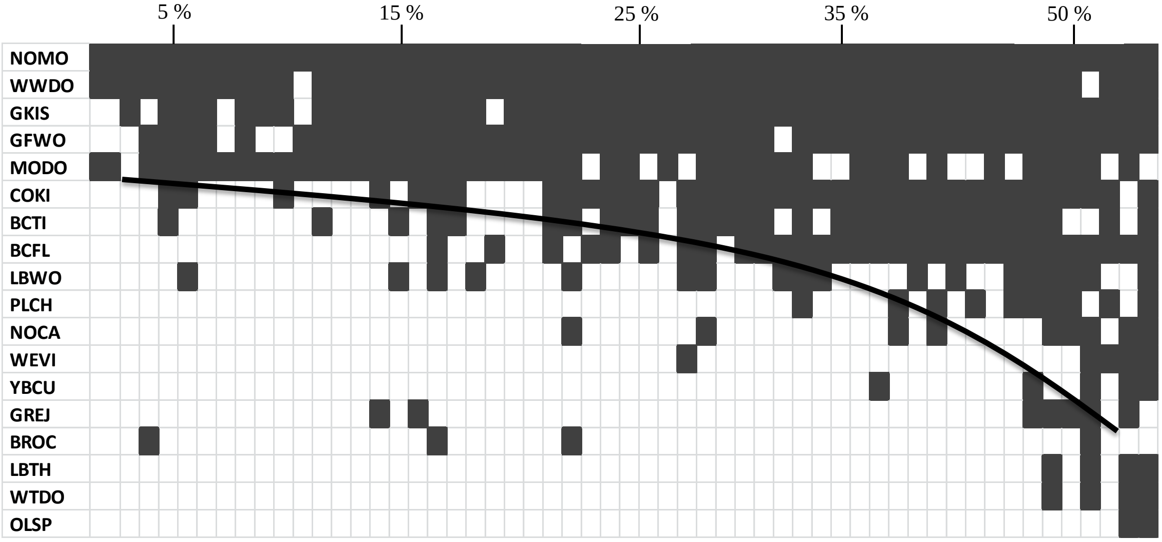

Figure 3.8A: Native forest species retention based off % vegetative cover. Columns represent increasing values of vegetative cover (left to right), and darkened squares indicate species presence.

2) Increase the variety and abundance of trees, particularly that of native trees, in residential habitats. Despite the sparsity of native trees in residential neighborhoods in McAllen and Edinburg, their presence still proved important for native forest birds and overall bird

diversity. The points with the most diverse and greatest numbers of native forest birds had between 3-5 native trees per 0.2 acres, which thus serves as a good baseline for individual homeowners. The increase of specific trees, like ash (*Fraxinus* species), can have significant effects on certain guilds of birds, such as cavity-nesters, and should be considered when developing conservation plans.

3) Conserve existing urban woodland islands in the urban mosaic. The presence of many native forest species in remnant tracts of woodland in urban areas, such as the McAllen Nature Center and Quinta Mazatlan World Birding Center, is encouraging, and may be important to maintaining native bird populations in LRGV cities. For some species, such as Olive Sparrow and Long-billed Thrasher, their presence in urban areas was dependent on these islands of dense vegetation. Increasing the size of urban woodlands through revegetation/afforestation will increase their value for native birds, and will allow them to host an even greater variety from the regional pool of species. Establishing connectivity of these urban woodland islands, whether by decreasing distances between patches or making the urban mosaic more passable for dispersing birds (for example, by increasing vegetative cover in residential neighborhoods), is also recommended.

The rich bird and biodiversity of the LRGV is important both biologically and for local ecotourism. By working to conserve birds in the region’s increasing urban component, we can help ensure that future generations will be able to enjoy some of the avian splendor our native habitats support, along with enhancing the ecosystem services supported by biodiversity. Small changes in our yards and cities can positively impact the bright Green Jays, boisterous Plain Chachalacas, and many other species that make up our local avian community.

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BIOGRAPHICAL SKETCH

John S. Brush graduated with B.S. in Biology from the University of Texas – Pan American in the Fall of 2013. His main area of interest is avian ecology, and he assisted with research done in Mexico, Turkey, and the Lower Rio Grande Valley. John’s passion for birding and ornithology has seen him lead bird tours for the Rio Grande Valley Birding Festival, conducted research with UTPA and UTRGV, and work as a naturalist at Quinta Mazatlan World Birding Center. He completed earned his Master of Science degree in Biology from the University of Texas Rio Grande Valley in the Fall of 2016. John can be contacted at jsbrush10@gmail.com.