AVIAN POPULATIONS IN HUMAN-DOMINATED LANDSCAPES: AN ANALYSIS OF SPATIO-TEMPORAL DYNAMICS AT THE URBAN-RURAL INTERFACE

by

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To my wife and family, and generations to come.

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ABSTRACT

To measure the ecological effects of urbanization this research focuses on bird-habitat relationships at the urban-rural interface by: investigating static and change relationships between local landscapes and local birds (Chapter 2); proposing and evaluating the use of bioacoustic recording equipment for avian point-sampling in an urbanized environment (Chapter 3); and, testing the relationships between forest birds and the landscape characteristics of forest and developed land covers (measured via development density) that are commonly intermixed in the study region (Chapter 4).

I assessed the relationships between compositions and changes of landscapes and avian abundances in Southeast Michigan using three bird guilds to group species of interest including woodland, grassland, and urban birds. The predominant landscape changes were agricultural abandonment, urbanization, and afforestation. I found that grassland and urban birds experienced the most consistent declines and that both average species richness and total abundance of birds also consistently declined. These results highlight that some bird guilds (e.g., grassland birds) suffered significant declines associated with habitat loss, while other guilds (e.g., woodland birds) did not respond to marked habitat increases. Then, I tested the effectiveness of omni-directional bioacoustic recording equipment versus traditional *in situ* point counts, along an urban-rural gradient. I found that recording-based interpretations were subject to the same ambient noises, and similar resulting levels of distraction associated with those noises, as were field-based observations; and, although not in perfect agreement with field-based observations, recordings can serve as an effective point-count mechanism in urbanizing environments. Finally, to explore how habitats within developed landscapes can be beneficial to birds, I tested community and species-level effects of patch- and matrix-characteristics on bird richness and occurrence. I found that focal-patch area is the primary contributor to a site's overall species richness, but that the addition of matrix tree-cover area influences the ability of the patches to support many forest-obligate species, especially Neotropical migrants. This relationship suggests that the amount of matrix tree cover surrounding woodlots, parks, and other preserved set-asides may play a critical role in supporting area-sensitive species in urbanizing environments.

CHAPTER 1

INTRODUCTION

Background

The pages in this dissertation focus on the ecological effects of one of the most rapid and expansive disturbances the earth has seen in modern times - urbanization by humans. For the majority of human history, anthropogenic influences on biophysical processes and ecological systems have been relatively limited as compared to the effects of natural (i.e. non-human) phenomenon (Alberti et al. 2003). In the last century, however, humans have changed Earth's ecosystems at extraordinary (Alberti et al. 2003) and alarming rates by converting land (Brown et al. 2005), controlling and consuming resources (Vitousek et al. 1986), disrupting hydrological systems (Arnold and Gibbons 1996; Girling and Kellet 2002), and altering habitats and species compositions (McKinney 2002). It is estimated that between one-third and one-half of Earth's landscapes have been transformed by human actions (Vitousek et al. 1997). Transformations, based on contemporary land-use practices, have occurred to such an extent that long-term, local to global ecosystem services may be undermined (Foley et al. 2005). Example modifications include the rapid destruction of virgin rain forests, the elimination of wetlands [over 50% of all wetlands in the contiguous United States (Dahl 1990)], the conversion of rich prairie and grasslands to agriculture, and, of primary importance to this study, the alteration of a diverse array of biological habitats through development and urban sprawl (Gill 1995).

Dramatic changes in land use and land cover can have significant ecological effects, both positive and negative. For example, positive effects include the re-establishment of local forest lands providing increased habitat, potential recreation opportunities, and a myriad of other ecosystem services (Guo et al. 2001). Negative effects include increased impervious surfaces and associated loss of habitat and native biodiversity, and the introduction of exotic species (Collinge 1996). There are generally three competing hypotheses about how urbanization affects biodiversity. First, as human population density increases, species richness (i.e., biodiversity) increases (Luck et al. 2004); i.e., the productivity hypothesis). Second and in direct contradiction to the first, Cam et al. (2000) demonstrated that a monotypic negative relationship exists between relative species richness and the level of urbanization (i.e., the ecosystem stress hypothesis - following the effects of human influences on species diversity in Rapport et al. 1985). The ecosystem stress hypothesis is also supported by Lepczyk et al. (2008) who suggest that both species richness and abundance are closely linked to the degree of human dominance in the Midwest United States (U.S.). Third, at some intermediate level of disturbance there is a peak in biodiversity [i.e., the intermediate disturbance hypothesis; (Connell 1978; McKinney 2002)]. This elevated biodiversity is related to increased heterogeneity as the floristic community expands by introducing non-native and edge species through processes like urbanization (Blair 1996).

One taxonomic group that is often used to investigate the ecological effects of human-induced landscape change is birds. The rationale for using birds is threefold. First, birds respond to a number of structural and functional elements of the environment based on a diverse array of microhabitat requirements (Eiswerth and Haney 2001). Second, unlike other terrestrial animals that reside within human-dominated landscapes, birds can relocate to more suitable habitats with relative ease, making bird abundance a sensitive measure for understanding the effects of anthropogenic landscape disturbances on ecological integrity (O'Connell et al. 1998). Third, birds are readily identifiable by sight and sound, making them ideal subjects for study. Birds have another quality that makes them fitting as subjects for study in relation to human-induced landscape change - people have a great affinity for them. This is supported by the number of households with bird feeders in their yards (64% in a study in Southeast Michigan; Lepczyk et al. (2004)); the number of birding organizations (682 across 49 of the 50 U.S. States; http://www.birdwatchersdigest.com, 2005); and the emergence of a new sector of the travel industry - avitourism (U.S. Department of the Interior et al. 2001).

Public attention to birds has no bearing on science directly, but offers a unique opportunity for the integration of science and policy formation. Following the opening words of a famous quote by the Senegalese environmentalist, Baba Dioum (1968), “In the end we will conserve only what we love…,” I posit that the pronounced societal interest in birds may offer an opportunity to convey knowledge about how land-use policies, resulting land-use choices, and human-induced landscape disturbances may influence bird populations, and in turn, the larger natural community. While the intent of this dissertation is not to develop policy, I subscribe to the notion proposed by Cash et al.

[2003; cited in Palmer et al. (2005) pg. 7], "It is no longer enough just to do the science; knowledge must be conveyed in a way that allows policy makers and the public to translate science into action." I suggest that science that has the "…ultimate goal of providing empirically based guidelines to policy makers…" (Miller et al. 2003 pg. 1057) has direct societal value. Societal value is especially high for the case of urbanization (conversion of natural areas to human settlements), because urbanization can have significant long-term effects on wildlife (and humans) resulting from its permanence within the landscape (Marzluff and Ewing 2001). Since urbanization and associated habitat loss and fragmentation are likely to continue in some form in perpetuity (Marzluff and Ewing 2001), land planners and policy makers must understand how their policies ultimately affect the persistence of natural communities. Understanding the outcomes of land-use policies is especially important for communities like those found in my study area, where increased land-use planning authority has been given to the local units of government by the State of Michigan following Home Rule, self-governing powers (Citizen Research Council of Michigan 1994-October). Michigan is one of 11 states that are categorized as Home Rule states. The other thirty-nine states are categorized as Dillon's Rule states. In Dillon's Rule states, state governments generally have more power to coordinate regional land uses (Richardson et al. 2003). In the Home Rule type of political landscape, adjacent local governments commonly do not coordinate land-use planning efforts and therefore increase the likelihood of decentralized urbanization and increased fragmentation within the landscape. By focusing the discussion about the ecological effects of local land-use planning and urbanization on birds - local residents may better understand the ecological effects of land development patterns and the need for cooperation between surrounding municipalities.

The chapters within this dissertation add to bird-habitat studies at the urban-rural interface by: investigating static and change relationships in local landscapes and local birds (Chapter 2); proposing and evaluating the use of bioacoustic recording equipment for avian point-sampling in an urbanized environment (Chapter 3); and, testing the relationships between forest birds and the contradictory land covers of forest and impervious surface that are commonly intermixed in the study region (Chapter 4).

Research Objective and Hypothesis

The objectives of this research were to:

(1) Understand how avian communities at the urban-rural interface are related to land cover, and changes in land cover, at the local scale;

(2) Evaluate effectiveness of acoustic surveys as a method for avian point sampling in a human-dominated landscapes, and to discern if thresholds in development density and/or urban noise influence the effectiveness of the recording system; and,

(3) Determine the effects of urbanization and tree cover in the surrounding matrix on the presence of forest-bird species in set-aside forest patches.

Specific hypotheses that were tested include:

Hypothesis 1: Guild-based (e.g., forest and urban) bird communities will exhibit land-cover-specific habitat associations and, therefore, will be affected by habitat amounts and changes to those amounts.

This hypothesis is exploratory in nature and was intended to investigate the effects of the major trends in land-cover change that were observed using aerial photography on three avian guilds - woodland, grassland, and urban - in the Southeast Michigan study region.

Hypothesis 2: Changes in bird communities will be related to changes in land cover in ways that mirror relationships determined by static-value comparisons.

The prediction that a space-for-time substitution analyses will yield similar results (i.e., bird-habitat relationships) as compared to long-term studies at single sites through time was tested.

Hypothesis 3: The bioacoustic recording method is not an effective alternative to traditional point counts, because of the decreased quality of recordings in urbanized environments.

This hypothesis is based on an initial field season using recordings in the study regions. It is not clear whether this approach is effective in more human-dominated landscapes because of noise associated with human activity.

Hypothesis 4: Locational perception is hindered and the opportunity for pin-pointing and following individual birds is reduced when interpreting bioacoustic recordings.

It is readily obvious that when listening to recordings in mono versus stereo mode three-dimensional perception is lost. Predictions related to this hypothesis test the influence of locational perception on bioacoustic recording effectiveness.

Hypothesis 5: A forest patch's location (and inherently the characteristics of the patch and the surrounding matrix) influences what avian species occur within the patch.

I tested for a relationship between species richness and the landscapes surrounding a forest habitat patch. Predictions related to this hypothesis test the influences of intermixed tree-cover and urbanization on a focal patch's ability to support forest-obligate avian species.

Structure of Dissertation

The body of this dissertation contains three interrelated, yet independent, primary chapters. The primary chapters are book ended by an introduction (this chapter, introducing the theme of the dissertation) and a conclusion (summarizing each chapter's results). These primary research chapters (2-4) have been prepared as manuscripts for publication.

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CHAPTER 2

RELATIONSHIPS BETWEEN LANDSCAPES AND AVIAN ABUNDANCES IN SOUTHEAST MICHIGAN, U.S.A.

Introduction

For the majority of human history, anthropogenic influences on biophysical processes and ecological systems had been relatively limited as compared to the effects of natural (i.e., non-human) phenomena (Alberti et al. 2003). In the last century, however, humans have changed Earth's ecosystems at extraordinary (Alberti et al. 2003) and alarming rates by converting land (Brown et al. 2005), controlling and consuming resources (Vitousek et al. 1986), disrupting hydrological systems (Arnold and Gibbons 1996; Girling and Kellet 2002), and altering habitats and species compositions (McKinney 2002). As a result, between one-third and one-half of Earth's landscapes have been transformed by human actions (Vitousek et al. 1997). Moreover, the transformations due to contemporary land-use practices have likely impaired ecosystem services from local to global scales (Foley et al. 2005). Understanding the ecological effects of human-induced (i.e., anthropogenic) land-use and land-cover changes is important because urbanization and its associated habitat changes are likely to continue in some form in perpetuity (Marzluff and Ewing 2001).

One taxonomic group often used to investigate the ecological effects of human-induced landscape change is birds. The rationale for the common use of birds is threefold. First, birds respond to a number of structural and functional elements of the environment based on a diverse array of microhabitat requirements (Eiswerth and Haney 2001). Second, unlike other terrestrial animals that reside within human-dominated landscapes, birds can relocate to more suitable habitats with relative ease, making bird abundance a sensitive measure for understanding the effects of anthropogenic landscape disturbances on ecological integrity (O'Connell et al. 1998). Third, birds are readily identifiable by sight and sound, making them ideal subjects for study.

Over the last two decades, landscape-scale research on the human influences on bird communities has increased markedly (Marzluff et al. 2001). Bird-urbanization research has ranged from the effects of fragmentation (Jokimaki and Huhta 1996; Boulinier et al. 1998; Coppedge et al. 2001; Donovan and Flather 2002), to the differential effects along urban to rural gradients (Blair 1996; Germaine et al. 1998; Crooks et al. 2004), to the effects of residential development (Odell and Knight 2001; Hansen et al. 2002; Pidgeon et al. 2007). While numerous studies focused on habitat composition, structure, and context as determinants of avian population persistence, most of these studies are short-term (e.g., 1-5 years) and focus on bird-habitat static relationships (i.e., the bird-habitat relationships regardless of actual changes through time). Thus, in lieu of space-for-time substitution techniques [SFTS; Pickett (1989)], little is known about how human landscape change has influenced birds. Therefore, my goal was to understand how both amounts and changes in the amounts land cover affect avian communities at the urban-rural interface, at the local scale.

In addressing my primary goal I investigated the effects of the major trends in land-cover change on three guilds: woodland, grassland, and urban birds. Woodland and grassland birds were selected because they are commonly thought to be declining in North America (Sauer et al. 2004), while urban birds, which are commonly associated with development, could be replacing woodland and grassland birds as habitats are converted through urbanization. Considering the three selected guilds, I hypothesized that (1a) increased amounts of impervious surfaces or hardscapes (acting as a surrogate measure of human presence) would negatively affect total abundance (i.e., the total number of birds counted) and total richness (i.e., the total number of unique species), and (b) will negatively influence counts for the woodland and grassland guilds as habitat is displaced, but (c) positively influence counts for urban birds as human dominance increases; and, (2a) increased tree cover due to secondary succession of abandoned agricultural land and low-density residential development with its associated landscape plantings, will increase the total abundance and total richness of birds, and will (b) positively influence the woodland and urban guilds (the latter because much of the afforestation is occurring within urbanizing areas), and (c) negatively influence grassland birds. I evaluated both static-value and change-value comparisons and, therefore, further hypothesize that (3) change-value relationships will mirror those determined by static-value comparisons.

Methods

Study Area

The study area was a portion of Southeastern Michigan, USA (Figure 2.1), the most densely populated portion of the state and having >90% of land in private ownership (Lepczyk et al. 2004). I used North American Breeding Bird Survey (BBS) data for the three townships, where both land-cover data were available (Brown et al. 2008), and routes with "acceptable" (defined in the methods) data were continuously sampled since the 1970s. The selected BBS route numbers (with sample years) and their associated townships were 49066 (1969-71, 1978-80, 1988-90) - Pittsfield Township; 49167 (1988-90, 1998-2000) - Scio Township; and, 49072 (1978-80, 1990-92) and 49073 (1968-70, 1977-79, 1990-92, 1998-2000) - both in Ray Township.

Landscape Data and Metrics

Using aerial photos (1-3 m resolution) sampled within the three townships, patch-level interpretations of land covers (i.e., biophysical features such as tree cover) were completed using a framework of 10 classes selected to represent the distribution of land covers in the study region. These classes included: agriculture lands under active row cropping; deciduous, coniferous, and mixed tree covers; impervious surfaces (both structures and transportation features); maintained lawns; open natural areas; wetlands; and, open water. All land covers were interpreted and digitized at a working scale of 1:3,000, using a linear, minimum mapping unit of 10 m. In an effort to better match the intended guild-level resolution of the avian data, I aggregated these initial 10 classes into six categories: tree cover; open natural; agriculture (row crop); maintained; wet; and those areas with little or no habitat value, i.e., impervious surfaces (Table 2.1). Final vector datasets were converted to 1 m raster grids for spatial-pattern analyses and segmented to create circular landscapes with a radius of 400 m around each BBS stop location. This landscape sampling distance (i.e., 400 m) is equivalent to bird sampling distances for BBS surveys (Sauer et al. 2004).

I calculated two types of metrics using Fragstats 3.3 (McGarigal et al. 2002), those that describe the landscape-scale patterns (i.e., an entire study site) and those that describe class-level patterns (i.e., single land-cover classes, e.g., forests). Metrics were chosen to capture horizontal landscape factors generally accepted as important to birds, including habitat area (Turner 1989; Burke and Nol 2000); the number of patches (i.e., an area of relatively homogenous habitat), commonly used as a surrogate for fragmentation (Boulinier et al. 1998; Donovan and Flather 2002); amount of edge (Burke and Nol 2000; Fahrig 2001); and, spatial isolation (Bellamy et al. 2003; Ahlering and Faaborg 2006). Specific landscape-level metrics included the number of patches (i.e., patch number), totaled across all land-cover types; the average patch size and interspersion, including all types of patches; a measure of contrast at the various patch boundaries (i.e., total edge contrast index); and, Shannon's diversity index. Calculated class-level metrics included class area; patch number of each class; total edge; and, mean proximity to the nearest neighbor of the same class. Since class-level factors were calculated for each of the five selected land-cover classes, there were 20 class-level variables along with 5 landscape-level variables, for a total of 25 independent variables.

I performed a factor analysis to reduce the number of variables for two reasons. First, when reviewing the correlation coefficients, many to most pairs of metrics were highly correlated (Riitters et al. 1995). Second, following a recommended 5 to 10 samples per variable (Kleinbaum et al. 1988, cited in Norman and Steiner 2000), a minimum of 125 observations would be needed to use all 25 variables; far greater than the sample size of this study (see below). The factor analysis was completed using a Varimax rotation and the factor scores extracted in SPSS 15.0 (2006). The first six factors selected based on having eigenvalues greater than 1 (Preisendorfer et al. 1981) described 80.8% of the variation in the model. These six factors could easily be interpreted using the areas of the five original land covers; the sixth component focused on the patch number, edge, and richness at the landscape level. Thus the original landscape variables were used for ease of interpretation. The final landscape variables, then, were the areas for each of the land covers including impervious surface, maintained, open natural, tree cover, and wet, and patch number, as a measure of landscape fragmentation and diversity. Both correlation coefficients and variance inflation factors (VIF) were examined to assess the possibility of collinearity among the landscape variables.

Bird Data

Bird-count data were obtained from the North American Breeding Bird Survey (BBS; http://www.pwrc.usgs.gov/bbs/). To maintain collection and reporting consistency within BBS data, a review process that accounts for factors that may influence the uniformity of data collection (e.g., start time, total time for survey of the entire route, weather, number of cars passing while surveying) is performed. Upon review, all routes are designated as acceptable (meets all BBS criteria) or unacceptable (fails for one or more reasons) in regards to the methodological criteria of the BBS program (Sauer et al. 2004). BBS stop-level data were compiled from 38 stops (i.e. individual sample locations) in the study area that were designated “Acceptable” by the BBS, had multiple visits corresponding to the available aerial photo data, and were not the result of a first-time observer (described below). These 38 stops were selected from four different BBS routes over many time steps from 1968 to 2000. This combination of sites and time steps provided a total sample size of 107 observation occurrences (rte 66, 7-sites in 3 decades; rte 72, 10-sites in 2 decades; rte 73, 11-sites in 4 decades; rte 167, 11-sites in 2 decades). To compile the BBS data set, annual observations for each of the stops were manually transferred from BBS field log-sheets (paper logs acquired from USGS Patuxent Wildlife Research Center), to an electronic database.

Biases within the data not typically accounted for by BBS methodology itself were addressed prior to analysis. Specifically, for each observation occurrence I aggregated three years of species counts at each survey stop by summing counts across years. The three-year-aggregated observations should account for many of the yearly fluctuations that can result from observer bias. With the exception of using sums instead of averages, this aggregation process is similar to Coppedge et al. (2001), who also analyzed BBS data at the stop-, rather than route-, level. Kendall et al. (1996) found that removing each BBS observer’s first-year observations decreased population trends (by an average of 1.8% per year) for 90% of the species analyzed. Additionally, 10.4% of the species analyzed had trends that switched from either positive to negative, or negative to positive (Kendall et al. 1996). Based on these findings, any first-year observations were removed from the analysis; and furthermore, I omitted any three-year aggregated observations that included an annual observation-occurrence from a first-year observer. Additional potential biases within the data were addressed by clustering species into ecologically based life-history guilds, which likely reduces data-collection biases because “differences in individual species trends may 'average out' when species are grouped” (Thomas and Martin 1996 pg. 488). It should be noted that clustering species into generalized habitat guilds (as compared to guilds that specify specific, microhabitat needs) necessarily reduces the resolution of potential bird-habitat comparisons, and therefore limits meaningful relationships that can be discerned from any bird-habitat study. As such, my choice to use guilds for this analysis was not ecologically based, but was driven by the need for larger sample sizes because of the low numbers of birds counted at many sites, even when using three-year aggregates. Guild definitions, that is, which species was assigned to each guild was based on nesting and foraging habitats and was similar to Peterjohn and Sauer (1993), Coppedge et al. (2001), and Brewer et al. (1991), except for the exclusion of migration status.

Guilds (Appendix 2.1) were delimited by including only fully identified species censused on the BBS routes (i.e., incompletely or unidentified species were excluded; Lepczyk et al. 2008). Additionally, to limit the analysis to those individuals that are most likely affected by local (versus regional) land-use and land-cover changes around the BBS stops, the guilds include only species having an average body mass < 150 g. This selection was based on the allometric relationship between body mass and foraging territories (i.e., home ranges) which indicated a 150 g bird will forage in less than 20% of the area surveyed at a BBS stop (Schoener 1968). Because small songbirds are more likely to be counted relatively near the center of the survey site, the mass limitation allows the survey samples to initially be treated as spatially independent for statistical analyses. That is, by limiting body mass to 150 g, the likelihood of an observer double-counting a bird is reduced because the survey stops are effectively spatially disjointed regions, thereby ensuring that any calculated correlations between landscape and avian population are based on local landscapes that would directly affect local species persistence.

Change Values

To describe landscape and bird changes (n = 69), differences in the amount of land covers, number of patches, community metrics, and guild-based bird counts were calculated for the time steps between adjacent decades for all 107 sample locations. To characterize landscape changes at each individual survey site, I determined the number and percentage of sites with positive changes, negative changes, or no changes in each of the landscape variables. Because I had a complete enumeration of land covers within each site for each available time period (i.e., no sampling), I had no need to test for statistical differences between landscape metric values at the individual site level.

To establish a baseline understanding of bird-community changes in the region, I compared the percent of sample locations with at least one individual of each guild present at each time step. Then, to illustrate site-level changes in bird counts between individual time steps, I determined the number and percentage of sites with positive changes, negative changes, or no changes in bird counts. As with the landscapes, I analyzed the entire population.

Bird-landscape Static Relationships

The relationships between the number of birds counted at a site in each guild (i.e., guild abundance) over three-years and the landscape metric values were calculated across all sample sites (n = 107), regardless of location or time-step. Because it is likely that the avian communities underwent complete exchange, possibly many times over, during the 10-year time steps, data measured at each time step are assumed to be independent observations, and were therefore pooled for analysis. In other words, while the independent variables (i.e., land-cover proportions measured at the same location at different times) are arguably repeated measures, the dependent variables (i.e. birds) were considered independent for statistical analysis. The relationships between bird count data and landscape metrics were analyzed in SPSS 15.0 (SPSS Inc. 2006) as a generalized linear model, based on a negative binomial distribution. The negative binomial model is commonly chosen for analysis of count data because it contains a model factor that can adjust the variance independent of the mean, and allows for the response variables to deviate from a normal distribution (pers. comm., L. Zhang, University of Michigan Center for Statistical Consulting and Research). The dependent variables for these analyses were bird abundance by guild. For those dependent variables (i.e., richness and total abundance) that were normally distributed, stepwise linear regression was performed to determine which, if any, predictor variables were significantly explanatory. In either case (i.e., negative binomial or stepwise linear), only final models with all significant (described below) predictor variables were accepted. If a model outcome was significant, but a predictor variable was not, the model was re-run with the non-significant predictor excluded.

Because of the clustered nature of BBS routes, there is the potential for spatial autocorrelation. That is, two BBS stops that are closer to one another are more likely to have similar landscape features, and therefore have more similar bird communities, than two sites farther apart. If positive spatial autocorrelation is present, test statistics are too often declared significant (i.e., p-values are artificially lowered); conversely, if negative spatial autocorrelation is present, test statistics are too often declared non-significant (i.e., p-values are inappropriately inflated; Legendre 1993). Spatial autocorrelation was investigated by calculating Moran's I index on residual values from all bird-habitat regression-based static and change-value (described below) comparisons. For several guilds (i.e., Urban and Woodland), the selected set of landscape variables captured the spatial variation within the samples and removed spatial autocorrelation from the model residuals (Smith 1994; Pidgeon et al. 2007). For the balance of the comparisons (i.e., Grass, Richness, and Total Counted), positive spatial autocorrelation was present within the residuals. While the measured spatial autocorrelation was significant ($p < 0.01$) for these models, the amount of spatial autocorrelation was generally small (Moran's I range: 0.01 to 0.18) in comparison to the maximum range for the Moran's I index of -1 to 1. Therefore, as opposed to complicating the regression models, for those guilds with positive spatial autocorrelation, I chose to address the potential effects of spatial autocorrelation by reducing the critical values from $\alpha = 0.10$ to $\alpha = 0.05$. This critical-value reduction effectively reduces the degrees of freedom associated with the analysis and should reduce the possibility of test statistics being declared significant, when they are not (Ricketts et al. 1999). Predictor variables for models with no spatial autocorrelation were evaluated based on $\alpha = 0.10$.

It should be noted that because the land-cover proportions summed to one, I dropped the percentage of agriculture as an independent variable, and land-cover variables can all be interpreted in relation to agriculture. Agriculture was chosen because it was the most consistently decreasing land cover in the region (making Ag appropriate for comparative analysis) and because it demonstrates a highly significant ($p < 0.01$ in each case), negative correlation with four of the five other land covers including: tree cover (-0.49), impervious surface (-0.59), other natural (-0.72), and wet areas (-0.39).

Bird-landscape Change Relationships

To determine if changes in guild-level bird abundances (i.e., t2 abundance – t1 abundance) were related to changes in the site-level landscape variables (i.e., t2 variable value – t1 variable value), I performed stepwise linear regression on the change values ($n = 69$). My intent in comparing change-values was to capture the dynamics across individual sites, i.e., to capture specific land-cover changes that elicit a response in bird counts. While still correlative in nature, comparing change-values moves the analysis closer to inference about causation, because it measures the dynamics of how the landscape changed and how birds responded. Similar to the analysis with static values, all data from all locations and times were included in the analysis, and only final models with all significant predictor variables were accepted. Also similar to static-value comparisons, all regression residuals were tested for spatial autocorrelation. All change-value residuals contained some level of negative spatial autocorrelation. The measured spatial autocorrelation was significant ($p < 0.01$ to $p < 0.10$) for these models, but like above, the amount of dispersed spatial autocorrelation was generally small (range -0.11 to

-0.16) in comparison to the maximum range for the Moran's I index of -1 to 1. Therefore, to address the spatial autocorrelation issue, I contemplated increasing the critical values from α = 0.10 to > 0.10 to reduce the possibility of test statistics being declared non-significant, when they are, in fact, significant. In the end, the dispersed pattern within the data was minimal, and that the original critical value of α = 0.10 was liberal to begin with; therefore I made no additional critical-value correction for the potential effects of spatial autocorrelation in change values.

For all comparisons (i.e., static and change analysis), regression models were completed for each guild independently (i.e., the predictor variables were tested for forest species response alone, and then for grassland species alone, and so on). Since the regression models were completed independently of one another, and any individual regression model result would stand on its own in the absence of other models, p-value corrections (e.g., Bonferroni) were determined to be inappropriate, and unnecessarily restrictive.

Results

Landscape & Bird Changes

The two land covers most commonly decreasing between decades were agriculture and open natural (Table 2.2). Agriculture decreased for 75.4% of the site-time observations while open natural decreased for 65.2% of the cases. Tree cover, impervious surface, and maintained lawns increased for 84.1%, 87.0%, and 71.0% of the site-time combinations, respectively. The number of patches increased at 88.4% of the site-time observations, with only 10.1% measuring a decreased patch count.

Agriculture and open natural were the dominant land covers at all sites in the 1970's (60% and 40% of the sites, respectively; data not shown). In 1980, 1990, and 2000, the number of sites whose majority land cover was agriculture or open natural decreased (from 60% to 40% and from 40% to 35% of the sites, respectively), while the number of sites with a majority of tree cover increased (from 0% to 20% of the sites). Although the percentage of agriculture tended to decline, analyzing these results on a site-by-site basis demonstrated that sites that were predominately agricultural remained primarily agriculture a majority of the time (77.7%), but also transitioned to predominately tree cover (2.8%), open natural (16.7%), or maintained lawns (2.8%). Similarly, while the amount of open natural decreased for a large percentage of the site-time observations (65.2%), sites that began with a majority of open natural remained that way for 78.6% of the cases. A transition from predominantly open natural to predominately agricultural or impervious surface occurred equally 10.7% of the time, for each. Sites that began as either predominately tree cover or impervious surface remained that way (i.e., did not transition to largely a second land cover). No sites began with a majority of maintained lawns or wet areas.

Reviewing site-level changes in bird counts between individual time steps, the only two guilds that experienced consistent directional changes (at > 60% of sites and times) were grassland and urban birds, which were both negative changes (Table 2.3). Both species richness and abundance declined in more than half of the site-time comparisons, 56.6% and 69.6% of the cases, respectively.

Bird-landscape Static Relationships

Tree cover and impervious-surface area were the most influential landscape factors related to bird counts (Table 2.4). Total tree-cover area positively influenced the abundance of birds in the woodland guild ($p < 0.000$) and both total richness ($p = 0.025$) and total birds counted ($p = 0.001$). Conversely, total tree-cover area negatively influenced birds in the grassland ($p = 0.007$) and urban ($p = 0.009$) guilds. The amount of impervious surface negatively affected the number of birds counted within the woodland ($p = 0.084$) and grassland ($p < 0.001$) guilds, as well as negatively influenced both total species richness ($p < 0.001$) and total birds counted ($p = 0.003$). Lastly, maintained area negatively affected the abundance of grassland birds ($p = 0.015$). All other potential relationships were excluded from the regression models because of a lack of statistical significance.

Bird-landscape Change Relationships

Change in tree cover area at sites demonstrated a significant negative relationship with changes in grassland bird abundances ($p = 0.050$; Table 2.5). Change in amount of maintained lawns positively influenced changes in urban bird abundance ($p = 0.043$), but negatively influenced changes in the abundance of both woodland birds ($p < 0.000$) and total species richness ($p < 0.001$). Open natural-area change had a significant positive relationships with changes in both wetland and open water birds ($p = 0.031$) and the change in wet area at a site was positively related to changes in total species richness ($p = 0.014$). All other potential relationships were excluded from the regression models because of a lack of statistical significance.

Discussion

Overall, guild-specific bird abundances and changes in abundance were both correlated (although non-synchronously) to abundances and changes in local land covers in the study region. The observed relationships between birds and landscapes are important because of the rate and extent of land-cover change, especially habitat loss, associated with urbanization. For example, agricultural and open natural lands, which have been the predominant land cover in the region for the last half century, and are home to Southern Michigan's grassland birds, decreased at the majority of the sample locations. The rest of the land covers, some of which may serve has viable habitats, typically increased (tree cover, impervious surface, and maintained lawns) or remained relatively unchanged (wet areas) for the majority of the comparisons. In essence the study sites were experiencing agricultural abandonment, leading to urbanization, and/or afforestation. Additionally, because of the replacement of agriculture with patches of trees, open fields, and other natural features, including maintained lawns and residential developments, the landscapes were becoming more diverse, contrasting, and edge-filled, as indicated by an increased number of land-cover patches observed at the various sites. These general land-cover trends are consistent with results of analyses across whole townships in Southeastern Michigan (Rutledge and Lepczyk 2002; Brown et al. 2008), suggesting that changes found within the BBS stop extents are consistent with changes measured across the entire township scale (36 miles^2, nominally). A high degree of correlation between the two scales allows for changes measured at BBS stops to serve as an indicator for avian ecological changes across the larger region, thus providing important information for conservation efforts within the region.

I posited working hypotheses related to both static-value relationships (i.e., comparisons between bird counts and static landscape configurations), and then in relation to change values (i.e., comparisons between simultaneous changes in bird counts and measured landscape changes at single sites though time). Static-value comparisons (Table 2.4) suggested that there was a significant negative relationship between the amount of impervious surface at a site, and the total abundance of birds counted and total species richness. Using the amount of impervious surface as an indicator for urbanization level, this result supports the ecosystem stress hypothesis (Rapport et al. 1985; McKinney 2002), which states that human influences negatively affects species richness. The results also demonstrated statistical support for the predictions that increased impervious surface would negatively affect both grassland and woodland birds. Overall, my hypothesis addressing the relationship between the amount of impervious surface and the number of urban birds was not supported. This finding is contrary to the intuition that sites with greater imperviousness (i.e., more developed landscapes) would have more urban birds. While the data do not support a relationship between urban birds and impervious surfaces, the result could simply be a product of species included in the guild definitions (e.g., Blue Jay [*Cyanocitta cristata*] and American Robin [*Turdus migratorius*] could be considered both urban and forest species; see Appendix 2.1). All told, the amount of impervious surface at a site demonstrated significant negative relationships with half of the dependent variables (i.e., bird guilds and community metrics).

I found support for relationships between the amount of tree cover at a site and both total abundance and richness (Table 2.4). Additionally, there were statistically significant relationships between tree cover and the three guilds of interest: grass, urban, and woodland. My predictions that woodland species would be positively influenced and grassland birds would be negatively influenced were both supported. However, the prediction that urban birds would be positively affected by tree cover, because afforestation is partially tied to residential development, was not supported. In fact, the amount of tree cover at a site and the abundance of urban birds yielded a significant negative relationship. Woodland birds were positively associated with the amount of tree cover and negatively related to the amount of impervious surface. Since tree cover and impervious surfaces are commonly mixed at the urban-rural interface, a conservation question of interest is: how do tree cover and impervious surfaces interact to support or hinder woodland bird populations? Further research is needed within the region to address such a question and is the focus of Chapter 4.

In testing the relationships with changes in impervious surfaces, none of my predictions were supported. That is, none of the dependent variable changes demonstrated even weak relationships with changes in imperviousness (Table 2.5). Change in tree cover at sites demonstrated significant relationships with one of the eight dependent variables, which supported only one of the predictions (i.e., that a change in the amount of tree cover has a significant negative relationship with change in grassland bird abundances). Based in my findings, I hypothesize that as agricultural lands decline through conversion to other uses, the habitat available for grassland birds also declines.

While my primary objectives and hypotheses focused on tree cover and impervious surface, I found that maintained lawns demonstrated more significant change-relationships than any other factor. Change in the amount of maintained lawn was significantly related to changes in urban birds, woodland birds, and species richness. An increase in the amount of maintained area was related to increases in the abundance of urban birds, and decreases in both woodland bird abundance and overall species richness. The above relationships are logical because amount of maintained lawn is likely an indicator of factors that can directly impact birds species, including alteration or maintenance of vegetation (both native and exotic), introduction of subsidized predators (e.g., domestic house cats), chemical applications, and nesting and food supplementation (Lepczyk et al. 2004).

My investigation relied on BBS data collected at each individual stop, as opposed to data aggregated across all stops in a route (Flather and Sauer 1996; Donovan and Flather 2002). Few have used BBS stop-level data (i.e., actual 3-minute species counts along a BBS route) for research and analysis, although, Coppedge et al. (2001; 2004) and Niemuth et al. (2007) have applied, non-aggregated, stop-level BBS data to bird-habitat research. I suggest that using stop-level data in conjunction with high-resolution landscape data is appropriate, and produces results at a scale that is beneficial to local land-use and land-cover conservation planning. Also, stop-level analyses provide a more detailed picture of the relationships between bird counts and landscape changes than studies conducted at a more regional level, or those completed across entire BBS routes, where bird counts are averaged over a larger, more aggregated geographic extent.

I compare how the amount of land cover is associated with the abundance of birds, but also how changes in the amount of a land cover influences changes in bird abundance. While the comparisons are outwardly similar, the former describes relationships as a static measure, (i.e., the number of birds at a site is associated with an amount of land cover), whereas the latter tries to capture the dynamics of change. If the amount of habitat area changes by a given amount, birds respond by increasing or decreasing accordingly. While still correlative in nature, and limited by a lack of information about which change came first (i.e., land cover versus birds), comparing change values comes closer to exploring causal relationships because it measures the dynamic of what changed and what responded. Because of the spatial and temporal stratification of the study sites, I was able to compare influences of land covers on bird guilds based on static landscape characteristics, but also based on changes at the same site through time. Each adds to the description of how land cover may be influencing the local bird communities in the region, although, there are discrepancies between the static measures and change values, consequently my third hypothesis was not supported. For example, the woodland bird guild exhibited a significant relationship ($p < 0.001$) with the amount of tree cover at a site. Based on this relationship, I expected that sites which added more trees through time would have increased abundances of woodland birds. Even with such a strong relationship with the amount of tree cover at a site regardless of time, changes in woodland birds showed no relationship with changes in the amount of tree cover through time. This difference suggests that landscapes that have more tree cover (combined with relatively lower amounts of impervious and maintained areas) will have more woodland birds, but simply increasing the amount of tree cover in the landscape will not necessarily elicit an increase in the numbers of woodland birds. The lack of a relationship with the change values may be simply a result of inaccurate counts of woodland birds at the study sites, or incorrect species compositions for each guild, or possibly inadequate increases in the amount of tree cover in any one site over the ten-year time step (e.g., the additional amount of tree cover has not added enough habitat to support further territorial species). Alternatively, the amount of tree cover could have increased sufficiently to produce increases in woodland bird counts, but there may be thresholds and non-linearities related to changes in the amounts of land covers, the amount of tree-cover core area, landscape-matrix differences at scales other than the individual site, conspecific attraction, and patch connectivity.

The lack of paired outcomes between the static and change comparisons may simply be based on too small of a sample size, or too narrow of a geographic sample-location distribution. However, the difference in results that I have observed highlight the differences between long-term ecological studies at single sites and studies subscribing to the space-for-time substitution (SFTS) methodology (Pickett 1989). Long-term analysis can reveal subtle trends in data that short-term analysis may hide or possibly reverse, and also allows for the exploration of uncommon events (e.g., destructive weather) to determine their frequency and importance (Fleming 1999). In applying SFTS techniques, the general idea is to focus on structural and compositional aspects of systems and neglect functional dynamics (Pickett 1989). By removing time, factors that could be essential to understanding species dynamics may be avoided altogether, e.g., the historical persistence of any given species or the effects of landscape change at a single location through time. Of particular importance here is the phenomenon of time-based landscape change, something missing from many landscape ecology studies of wildlife. Landscapes can change along “different trajectories with different rates of habitat loss, with very different consequences for species occupying those landscapes” (Schrott et al. 2005 pg. 503). If the change is significant, a species-habitat relationship may not exist, or may be altered in adjacent years. If the change is slow enough, species may be able to co-evolve or adapt with the landscape changes allowing for local persistence. The ability to document how any single landscape is changing, along with how bird species are changing, is critical to understanding how landscape change is influencing habitat-based population dynamics. That is, ecological legacy matters. Therefore, SFTS techniques alone, are seemingly limited in their ability to provide answers related to the functional dynamics of systems, but are more appropriate for generating future testable hypothesis and questions (Pickett 1989).

Conclusions

The predominant landscape changes the study sites experienced were agricultural abandonment, urbanization, and afforestation; and, both average total species richness and total abundance of birds declined for the majority of the sample locations. The latter result is similar to that of Valiela and Martinetto (2007), who estimated that the abundance of birds recorded during BBS surveys have declined linearly between 1966 and 2005. I found that both grassland and urban birds experienced declines for the majority of the sites. All other guilds demonstrated marginal increases and decreases depending on location.

I evaluated potential relationships between the landscapes and bird data from two perspectives, first in relation to static-value relationships, and then in relation to change values. The outcomes describing relationships between the number of birds and the amount of land cover were not consistent with described relationships between a change in the number of birds and a change in the amount of a land cover. From an avian conservation and land-use planning perspective then, when designing bird friendly landscapes it is important to account for both static and change-relationships. Alone, each describes only a portion of the necessary understanding, but together static and change-relationships describe bird-habitat-area interactions and also provide insight into the changes that must take place at a given site to elicit a bird count response. In a region (e.g., S.E. Michigan) dominated by private ownership and human development, the critical goal should be to understand where the habitat thresholds are so that urbanization can be channeled down a more ecologically sensitive path. Such a goal is highlighted by Rosenzweig (2003) who calls for the institution of a "reconciliation ecology" (i.e., "…the science of inventing, establishing, and maintaining new habitats to conserve species diversity in places where people live, work, and play;" pg. 7).

The urbanization I found during this study resulted in altered habitats, decreased habitats, and increased habitats with varying influences on different bird communities. Some bird guilds (e.g., grassland birds) suffered significant declines associated with habitat loss while other guilds (e.g., woodland birds) have not responded to marked increases in habitat. Therefore, not all seemingly negative changes associated with urbanization are detrimental to avian species, but also, not all seemingly positive changes (e.g., afforestation) are by themselves beneficial. We need to better understand how the specifics of relatively expansive and rapid urbanization (as compared to historical extents and rates) are affecting avian communities. To accomplish this, additional, higher-resolution avian sampling (i.e., more detailed than the BBS) and landscape studies, in the form of long-term research, to elucidate habitat area thresholds and matrix effects in urbanizing environments, following a gradient paradigm, is needed. Only then can we hope to provide useful science for the education of land managers, developers, and private citizens on the means to achieve an avian-based version of Rosenzweig's reconciliation ecology.

Table 2.1. Land-cover classes used for interpreting aerial photographs.

Class	Code	Description
Tree Cover	Tcov	All trees including deciduous and coniferous types delineated by tree canopy extent.
Open Natural	Onat	Primarily including open and fallow fields, but also other indiscernible natural areas.
Agriculture	Ag	Agricultural row crops including agricultural bare soils.
Maintained Lawn	Maint	Manicured areas adjacent to commercial and/or residential structures, including golf courses and other maintained areas.
Wet Areas	Wet	All wet areas including lakes, rivers, streams, and wetlands.
Impervious	Imp	All impervious surfaces including housing and commercial structures, driveways, and transportation networks.

Table 2.2. The number and percent of site time-change observations (out of 69 possible) exhibiting either an increase in land cover area or patch number (positive change), a decrease in land cover area or patch number (negative change), or no change in land-cover area or patch number. See Table 2.1 for class abbreviations.

Direction	Ag	Tcov	Imp	Onat	Maint	Wet	PN
pos. change	12	58	60	24	49	35	61
neg. change	52	11	9	45	20	17	7
no change	5	0	0	0	0	17	1
% pos. change	17.4	84.1	87.0	34.8	71.0	50.7	88.4
% neg. change	75.4	15.9	13.0	65.2	29.0	24.6	10.1
% no change	7.2	0.0	0.0	0.0	0.0	24.6	1.4

Table 2.3. The number of site-time change observations (out of 69 possible) exhibiting either positive change (+), negative change (-), or no change in guild-level abundance, species richness, and total abundance (count). See Table 2.1 for class abbreviations.

Direction	Grass	Open	Scrub	Urban	Wood	Wet-Open	Richness	Count
pos. change	19	23	33	23	18	27	27	18
neg. change	42	32	27	45	21	40	39	48
no change	8	14	9	1	30	2	3	3
% pos. change	27.5	33.3	47.8	33.3	26.1	39.1	39.1	26.1
% neg. change	60.9	46.4	39.1	65.2	30.4	58.0	56.5	69.6
% no change	11.6	20.3	13.0	1.4	43.5	2.9	4.3	4.3

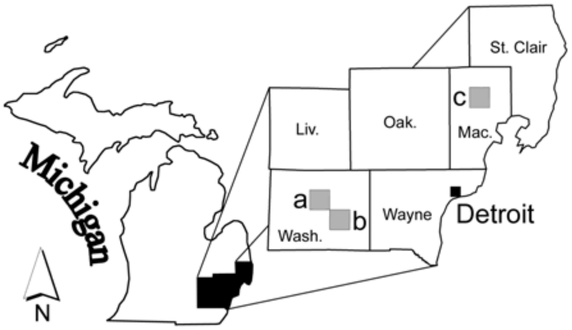
Table 2.4. Independent model results (i.e., models were calculated for each guild separately) for static-value comparisons between the landscape variables and bird counts (n = 107) and community metrics (i.e., richness and total counted). Generalized linear models, based on a negative binomial distribution, were completed for bird counts; and stepwise linear regressions were completed for the community metrics. The direction of the relationship (+ or -) and p-values are reported for overall models and predictor variables that were significant.

Regression		Landscape Variables					
Method	Class/ (Model Fit)	Land_NP	Tcov_Area	Imp_Area	Maint_Area	Onat_Area	Wet_Area
Neg. Binomial	Grass	--	(-) $p = 0.007$	(-) $p < 0.000$	(-) $p = 0.015$	--	--
	($\chi2 = 46.570$; df = 3; $p < 0.000$)		$R = -0.181$	$R = -0.320$	$R = -0.251$		
	Open (all variables excluded)	--	--	--	--	--	--
	Scrub (all variables excluded)	--	--	--	--	--	--
	Urban	--	(-) $p = 0.009$	--	--	--	--
	($\chi2 = 6.215$; df = 1; $p < 0.013$)		$R = -0.332$				
	Wood	--	(+) $p < 0.000$	(-) $p = 0.084$	--	--	--
	($\chi2 = 21.851$; df = 2; $p < 0.000$)		$R = 0.411$	$R = -0.162$			
	Wet (all variables excluded)	--	--	--	--	--	--
Linear	Richness	--	(+) $p = 0.025$	(-) $p < 0.000$	--	--	--
	(adj. $R^2 = 0.158$; $F_{2,104} = 10.920$; $p < 0.000$)						
	Total Counted	(-) $p = 0.043$	(-) $p = 0.001$	(-) $p = 0.003$	--	--	--
	(adj. $R^2 = 0.197$; $F_{1,105} = 9.655$; $p < 0.000$)						

Table 2.5. Stepwise linear regression results for change-value comparisons (n = 69) between changes (Δ) in the individual landscape variables and changes in guild-level bird counts and community metrics. The direction of the relationship (+ or -) and p-values are reported for overall models and predictor variables that were significant.

Regression		Landscape Variables					
Method	Class/ (Model Fit)	Land_NP	Tcov_Area	Imp_Area	Maint_Area	Onat_Area	Wet_Area
Linear Reg. on Change Values	Grass	--	(-) $p = 0.050$	--	--	--	--
	(adj. $R^2 = 0.042$; $F_{1,67} = 3.999$; $p = 0.050$)						
	Open (all variables excluded)	--	--	--	--	--	--
	Scrub (all variables excluded)	--	--	--	--	--	--
	Urban	--	--	--	(+) $p = 0.043$	--	--
	(adj. $R^2 = 0.045$; $F_{1,67} = 4.239$; $p = 0.043$)						
	Wood	--	--	--	(-) $p < 0.000$	--	--
	(adj. $R^2 = 0.245$; $F_{1,67} = 24.135$; $p < 0.000$)						
	Wet	--	--	--	--	(+) $p = 0.031$	--
	(adj. $R^2 = 0.053$; $F_{1,67} = 4.838$; $p = 0.031$)						
	Richness	--	--	--	(-) $p = 0.001$	--	(-) $p = 0.014$
	(adj. $R^2 = 0.191$; $F_{2,66} = 9.024$; $p < 0.000$)						
	Total Counted (all variables excluded)	--	--	--	--	--	--

Figure 2.1. Location of study area in Southeastern Michigan, U.S.A. The selected portions of the BBS routes are contained within three townships (a) Scio, (b) Pittsfield, and (c) Ray. Detroit (42.33° N Lat., 83.05° W Long.) anchors the southeast corner of the region.



Appendix 2.1. All the species observed in our study, listed taxonomically, and the guild into which they were placed.

AOU	Common Name	Latin Name	*Guild
1470	Alder Flycatcher	*Empidonax alnorum*	Scrub
4440	Eastern Kingbird	*Tyrannus tyrannus*	Open
4520	Great Crested Flycatcher	*Myiarchus crinitus*	Woodland
4560	Eastern Phoebe	*Sayorris pheobe*	Open
4610	Eastern Wood-Pewee	*Contopus virens*	Woodland
4650	Acadian Flycatcher	*Empidonax virescens*	Woodland
4660	Willow Flycatcher	*Empidonax traillii*	Scrub
4670	Least Flycatcher	*Empidonax minimus*	Woodland
4740	Horned Lark	*Eremophila alpestris*	Grass
4770	Blue Jay	*Cyanocitta cristata*	Urban
4930	European Starling	*Sturnus vulgaris*	Urban
4940	Bobolink	*Dolichonyx oryzivorus*	Grass
4950	Brown-headed Cowbird	*Molothrus ater*	Open
4980	Red-winged Blackbird	*Agelaius phoeniceus*	Wet-Open
5010	Eastern Meadowlark	*Sturnella magna*	Grass
5011	Western Meadowlark	*Sturnella neglecta*	Grass
5070	Baltimore (Northern) Oriole	*Icterus galbula*	Open
5110	Common Grackle	*Quiscalus quiscula*	Urban
5190	House Finch	*Carpodacus mexicanus*	Urban
5290	American Goldfinch	*Carduelis tristis*	Scrub
5400	Vesper Sparrow	*Pooecetes gramineus*	Grass
5420	Savannah Sparrow	*Passerculus sandwichensis*	Grass
5460	Grasshopper Sparrow	*Ammodramus savannarum*	Grass
5600	Chipping Sparrow	*Spizella passerina*	Urban
5630	Field Sparrow	*Spizella pusilla*	Scrub
5810	Song Sparrow	*Melospiza melodia*	Scrub

5870	Eastern Towhee	*Pipilo erythrophthalmus*	Scrub
5930	Northern Cardinal	*Cardinalis cardinalis*	Scrub
5950	Rose-breasted Grosbeak	*Pheucticus ludovicianus*	Woodland
5980	Indigo Bunting	*Passerina cyanea*	Scrub
6040	Dickcissel	*Spiza americana*	Grass
6080	Scarlet Tanager	*Piranga olivacea*	Woodland
6110	Purple Martin	*Progne subis*	Urban
6120	Cliff Swallow	*Petrochelidon pyrrhonota*	Open
6130	Barn Swallow	*Hirundo rustica*	Open
6140	Tree Swallow	*Tachycineta bicolor*	Open
6190	Cedar Waxwing	*Bombycilla cedrorum*	Open
6240	Red-eyed Vireo	*Vireo olivaceus*	Woodland
6270	Warbling Vireo	*Vireo gilvus*	Woodland
6280	Yellow-throated Vireo	*Vireo flavifrons*	Woodland
6520	Yellow Warbler	*Dendroica petechia*	Scrub
6810	Common Yellowthroat	*Geothlypis trichas*	Scrub
6870	American Redstart	*Setophaga ruticilla*	Woodland
6882	House Sparrow	*Passer domesticus*	Urban
7040	Gray Catbird	*Dumetella carolinensis*	Scrub
7050	Brown Thrasher	*Toxostoma rufum*	Scrub
7210	House Wren	*Troglodytes aedon*	Scrub
7250	Marsh Wren	*Cistothorus palustris*	Wet-Open
7270	White-breasted Nuthatch	*Sitta carolinensis*	Woodland
7310	Tufted Titmouse	*Parus bicolor*	Woodland
7350	Black-capped Chickadee	*Poecile atricapilla*	Woodland
7510	Blue-gray Gnatcatcher	*Polioptila caerulea*	Woodland
7550	Wood Thrush	*Hylocichla mustelina*	Woodland
7610	American Robin	*Turdus migratorius*	Urban
7660	Eastern Bluebird	*Sialia sialis*	Open

*From Coppedge et al. 2001, PeterJohn and Sauer 1993, and Brewer et al. 1991.

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CHAPTER 3

COMPARING BIOACOUSTIC AND EXPERT-BIRDER POINT COUNTS OF FOREST BIRDS ALONG AN URBAN TO RURAL GRADIENT

Introduction

Research on bird-habitat relationships within urbanizing areas has increased markedly over the past two decades (Marzluff et al. 2001). These studies have investigated relationships such as the effects of fragmentation [Boulinier et al. (1998), Coppedge et al. (2001), and Donovan and Flather (2002)]; differential population characteristics across urban to rural gradients [Blair (1996), Germaine et al. (1998), and Crooks et al. (2004)]; and, the influence of residential development on reproductive success (Hansen et al. 2002). To study the effects on birds of human-induced changes in land use and land cover, as the aforementioned studies do, investigators must necessarily have a minimum of two data components arrayed across some geographic range: landscape/habitat data and bird counts. The collection and processing of these data can be time consuming, tedious, and often require assistants with specialized skills. Because the availability of skilled individuals is typically less than the research community demands, it is tempting for a researcher to learn the birds of a region, or hire an assistant to learn them, and commence with a study. However, a number of negative effects of first-time observers on research results are well documented (Sauer et al. 1994; Kendall et al. 1996), suggesting that this approach produces usable data, at best, only after the first year of a project. Studies that cover large geographic areas face the additional challenge of deploying field observers over a large area within a biologically relevant time window, which poses additional logistical constraints.

The challenges of collecting field data across a broad geographical region and the shortage of skilled birders prompted Hobson and others to test "…a newly developed omnidirectional microphone system with exceptional recording performance to determine whether this technique could adequately replace a trained observer…" (Hobson et al. 2002, pg. 710) for avian point-count surveying in the southern boreal mixed-woods of central Canada. Hobson et al.'s hope was that unskilled field assistants could collect field samples (i.e., bioacoustic surveys) during the short breeding season and then skilled personnel could review and interpret the recordings during less demanding times of the year. Their methodological comparison was quite successful, finding that bird sampling estimates from the recordings were similar to more traditional methods (Hobson et al. 2002).

While the utility of bioacoustic recording equipment as a replacement for expert birding (and traditional point counts) in remote forests has been confirmed (Hobson et al. 2002), it is unclear whether this approach is effective in human-dominated landscapes. In particular, human-dominated landscapes have the complicating factor of additional sounds associated with human activity which could potentially contaminate the signal. Given the lack of knowledge about the utility bioacoustic surveys in human-dominated landscapes, the goal of this research was to evaluate if acoustic surveys could provide a valid method of sampling outside of remote pristine locations. Based upon this goal, the objectives of the project were to (1) test whether a recording system could serve as a surrogate for an expert birder in a human-dominated environment, and (2) if thresholds in development density and/or urban noise influence the effectiveness of the recording system.

Based on a qualitative assessment of similar recordings from the previous year, I hypothesized *a priori* (1) that the recording method would not be an effective alternative to traditional point counts because of the decreased quality of recordings in urbanized environments. Based upon this initial hypothesis I predicted that: (1) recordings would provide higher estimates of species richness (i.e. total species observed) than field observers because of the extra time available to review unknown species, the lack of adverse environmental conditions (high heat, humidity), and natural distractions (e.g., mosquitoes); (2) as development density increases from rural to urban, an increased disagreement will be observed between the field-observation and recording samples, because sounds associated with human activity increasingly contaminate the recorded signal; and, (3) because of the sensitivity of the microphones, a greater proportion of low frequency sounds at a site will result in increased disagreement between the field-observation and recording samples, with the recordings providing an under-representation.

The second hypothesis was that locational perception is hindered and the opportunity for pin-pointing and following individual birds is reduced when interpreting the recording samples. Based upon this second hypothesis I predicted that the recording reviewers would not be able to fully recreate relative distance and relative direction of observed birds, as compared to the field observations. Furthermore, I predicted that recording-based distances to be shorter (as sounds seem closer in the headset) and directions to be limited (sounds coming from front and back are possibly indistinguishable). Lastly, because I expected 3-dimensional perception to be hindered, I predicted that the recordings are likely to provide reduced counts (i.e., abundances) as compared to the field observations.

Methods

Study Area

The study area is located in Washtenaw County, Michigan, U.S.A., between the area adjacent to City of Ann Arbor, and the state game and recreation lands to the northwest in rural Washtenaw County (Figure 2.1). The County had 323,000 residents in 2000, with two primary cities (Ann Arbor, population 114,000, and Ypsilanti, population 22,360) containing 42% of the total population (http://www.census.gov). Six additional rural communities are distributed throughout the county which covers an area of 186,738 hectares (721 sq. miles; http://www.ewashtenaw.org). In the year 2000, land use within the County was 65.5% rural, 20.5% exurban, 9.5% suburban, and 4.5% urban. To represent the distribution of development densities in the region, I defined rural development densities as an average of ≤ 0.25 units (i.e. built structures)/ha of land (i.e. ≤ 0.10 unit/acre); exurban densities as 0.82 – 0.24 units/ha of land; suburban densities as 3.29 – 0.81 units/ha of land; and urban densities as > 3.29 units/ ha of land (i.e., > 1.33 units/acre). Four land covers predominated in 2000 (SEMCOG; Southeastern Michigan Council of Governments): active agriculture (41.3% of the land); woodlands and wetlands (20.2%); grasslands and shrub (12.8%); and, those land covers associated with single family residential (14.6%).

Birder Selection

To facilitate a comparison of field observations versus bioacoustic recordings, local birders were solicited via local birding mail groups, resulting in one local birder being selected. With limited response to repeated regional, and then national postings (via the Ornithological Society of North America bird jobs list), the second birder was from outside the study region. Both "expert" birders (hereafter Observers) had many years of documented birding experience, were of similar age, were familiar with the birds of the region, and were available. The observers were asked to perform standard avian point counts (described below), and then to interpret, as part of a blind study, the recordings that were collected simultaneously with their point counts. Each observer surveyed birds for approximately 2.5 weeks, with their order of observation (i.e., early or late sampling session) based solely on their availability.

Sampling Design

Site selection for this comparative study focused on set-aside forest patches (i.e., parks, preserves, and other natural areas) along an urban to rural gradient of housing densities within Washtenaw County. I selected sample locations within publicly accessible conservation and recreation lands (i.e., set-asides) that were surrounded by diverse land covers and distributed within four strata of Census block-groups, defined by housing densities (urban, suburban, exurban, and rural) from the 2000 Census. Forty-four locations were compared across the gradient with 8 urban sites, 11 suburban sites, 9 exurban sites, and 16 rural sites (see Urbanization Level below).

In an effort to better represent within-forest characteristics (as opposed to road-side sampling) all sample locations were placed 100 m in from the edge of each selected forest patch. Two samples were completed for each of the 44 sites, totaling 88, 10-minute point counts between May 21 and June 26, 2007. The first surveys were conducted between May 21st and May 29th (completed by Observer A) with the second surveys conducted between June 19th and June 27th (completed by Observer B). Each sample began between 5:40 a.m. and 9:56 a.m. E.S.T. (sunrise ranged between 5:58 a.m. and 6:08 a.m.), with an average of six samples at different plots, completed per day. In an attempt to record different species singing at different times of the morning, the recordings were divided into two time groups, 5:30-7:45 (Time A), and 7:45-10:00 (Time B). The ideal configuration was for each site to be recorded once in each time group. For logistical reasons this was not possible. Thirty-three sites were successfully recorded once each in times A and B, in 10 sites twice during time A, and in one site twice in time B. Sampling required nearly 400 miles of travel to survey each of the sites once, within the daily time constraints (i.e., prior to 10 am), and was purely logistical during the first round. During the second round, sampling order was a combination of travel logistics and whether or not the site had been surveyed in the early time period during the first round.

Urbanization Level

Housing density data from the U.S. Census served as a basis for the sampling design. For analytical purposes I determined actual built-structure density for each sample location by interpreting 2005 high-resolution color aerial photography (courtesy of Washtenaw County) within ArcGIS 9.2 (ESRI, 2007). Built structures (e.g., houses and commercial buildings) were identified on the photographs using a working scale of 1:3000. All built structures within 1 km of each sample location were recorded. To determine the built-structure density for each sample location, the total number of structures was divided by the total area of interest (1 km radius from site). In addition to built structures, total road length within each sample location was calculated by interpreting 2005 aerial photos, guided by the road centerlines from the State of Michigan, (similar to Hawbaker and Radeloff 2004), and tallying the total road lengths within the 1 km-radius area.

Urban Noise

A premise underlying several of my predictions is that more urbanized locations will have higher levels of ambient noise (Slabbekoorn and den Boer-Visser 2006). I assumed that overall ambient noise, and also low frequency noise (commonly associated with traffic), would increase with higher housing and total road length. To investigate noise levels, a noise assessment was performed during the second round (with Observer B) of recordings by having the observer rate the level of ambient noise at each site while in the field and then again from the recordings, on an ordinal scale from 1 (little to no ambient noise) to 5 (high, very distracting ambient noise). Minimum and maximum decibel (dB) levels at each sample location were recorded using an Extech 40-130 db sound-level meter. To assess the accuracy of the estimated ambient noise levels (i.e. the described ordinal scale) for capturing noise levels as recorded by the sound-level meter, I regressed the field-based ambient-level categories against median db levels ($R^2 = 0.40$; p

< 0.000; $F = 28.23$; df = 1, 42). Then, I compared the ambient levels from field observations (determined to match the sound-level meter output) versus ambient ratings from recording interpretations to determine their agreement ($R^2 = 0.71$; $p < 0.000$; $F = 41.67$; df = 1, 42). In performing this test I found that the level of ambient noise as determined in the field (from both the ordinal scale and the sound-level meter) is similar to that found on the recordings. Following the assessment for Observer B, I asked Observer A to also interpret the ambient levels for each of the recordings only. By using the ambient noise levels from the recordings I have a relative noise, or distraction level, for each point count from each observer.

To characterize the proportion of total sound energy collected for low frequency (<1 kHz) noises for each recording, I used the Cornell Lab of Ornithology, Raven 1.3 interactive sound analysis software (http://www.birds.cornell.edu/brp/raven/raven.html). I began by selecting the portion of the sound spectrogram <1 kHz for the entire recording and determining the Energy metric (measured in dB, and calculated within Raven) for the selection. Then the entire spectrogram was selected to determine the overall Energy. The proportion of energy <1 kHz was calculated using Equation 1 for each of the 88 recordings.

Eqn 1.

$$PercentEnergy < 1kHz = 100 * 10\text{^}((\text{Band Energy in dB/10}) - (\text{Total Energy in dB})/10)$$

To test for an urbanization influence on low frequency noise, I independently regressed both built-structure density and total road length against the percent of sound energy <1 kHz. Performing both regressions is somewhat redundant because structure density and total road length were highly correlated ($R = 0.828$; $p < 0.01$). Nonetheless, there existed a potential for each to have a slightly different relationship with low frequency noise, so I chose to test both. The regressions for built-structure density and total road length were completed as both linear and quadratic relationships. The outcome with the highest R^2-value for each comparison was selected as the model with the best fit. Both structure density as a quadratic relationship ($Log_{10} + 1$ transformed; $R^2 = 0.204$; $p < 0.001$; $F_{2, 85} = 10.886$) and road length as a linear relationship (Log_{10} transformed; $R^2 = 0.302$; $p < 0.001$; $F_{1, 86} = 37.251$) significantly influenced the percent (arcsine square-root transformed) of sound energy collected <1 kHz (Figure 3.2). While each comparison described only a portion (16% or 30%) of the variability in the low frequency data, the analysis indicated that a relationship does exist and, generally, more urbanized, more traveled (i.e., vehicle traffic), sites do contain higher proportions of low frequency noise, indicating that my supposition about urbanization was valid.

In addition to noise-level measurements, at each site I collected other variables likely to affect signal attenuation and degradation, including temperature, humidity, topography, and wind speed (Simons et al. 2007). Temperature, humidity, and wind speed (all samples had wind speeds < 3 mph [4.8 km/h]) were collected using a Nielsen Kellerman Kestrel 3000 and were taken as forest floor measurements prior to the bird sampling. Topography was evaluated using a 30 m digital elevation model (DEM), from the Michigan Center for Geographic Information. The mean slope for each sample area was calculated using a GIS-based mean focal-neighborhood operation.

Expert-birder Point Counts (field)

To perform avian point counts in a standard approach (Martin et al. 1997), observers were instructed to document all birds heard or seen at the sample locations (although for this comparison I focused only on birds that were audibly detected). Using modified Michigan Breeding Bird Atlas (MiBBA) field logs (Brewer et al. 1991), the observers recorded species, abundance, behavior (e.g., singing, calling, fly-over), and estimated distance and direction (in relation to North) from the observation location. Observation distance was considered unlimited, with observations parsed into three distance classes: 0-50 m, 50-100 m, and >100 m. Since it has been suggested that even a single person moving through a bird's territory can influence singing for some species (Gutzwiller et al. 1994), each point count began after a 10-minute rest period (Martin et al. 1997), allowing time for normal singing to commence.

Acoustic-recording Point Counts (recording)

Simultaneous to the expert-birder observations, acoustic-recording point counts were completed using a combination of two CZM-180 microphones from River Forks Research Corporation (http://rfrc1998.sasktelwebhosting.com) and a Marantz PMD 660 Solid-State digital recorder. The stereo recordings were collected as 16-bit, uncompressed, non-attenuated, waveform audio format (.wav) files recorded at a frequency of 48 kHz. Mounted atop a tripod ~1.5 m above the ground, the microphones were fixed 29 cm apart. At each site the front of the tripod was oriented north such that the right channel always faced east and the left channel faced west. Additionally, the microphone's position was 1-3 m due south of the field observer.

In order to compare the sound recordings with the field observations, the recordings first needed to be interpreted. To complete the interpretations as a blind study, all samples were re-numbered prior to review. Similarly, to remove any observer bias, all digital recordings were interpreted by the same observer that performed the simultaneous field observation. To complete the interpretation the field assistants were provided with professional quality Sony MDR 7506 headphones and instructed to perform a critical listening session, identifying bird species and abundances in a similar manner that they would follow in the field. The critical listening session were used to remove potential distractions of working with the computer, and its multiple forms of bird-song visualization (e.g., song waveform and spectrogram). Additionally, the assistants were asked to note, by time stamp, any species where the identification was uncertain. Unidentified songs were evaluated using Raven by comparing song audio and song spectrograms against archives found in song libraries, for example the online Birds of North America Series (http://bna.birds.cornell.edu/bna).

Similar to the standard field observations, interpretations of sound recordings were completed on MiBBA field logs but only for birds that were heard singing or calling at the sample locations. Also as before, species, abundance, and estimated distance and direction (in relation to North) from the observation location were documented.

Observer A performed the initial interpretations using a standard PC-based audio software package (e.g., Windows Media Player, Winamp, etc.) while Observer B performed the interpretations using Raven's playback utility. While standard music software automatically splits a stereo recording into its left and right channels (which was my intent), Raven's playback feature needed an adjustment to assign the right and left playback channels to the left and right headphones, respectively. By default Raven assigns one channel to both headphones. Inadvertently, Observer B was not instructed to split the channels, so all of their recordings were interpreted in mono, such that 180 degrees of the listening field was not fully evaluated. Based on this methodological deviation, I predicted that Observer B would still be able to identify the majority of species present at a site (as bird songs will still be heard, even if the observer is not listening in the direction of the song) but the observer would be significantly hindered in attempts to replicate field-based abundances, because direction and depth perception appear severely degraded when listening in mono versus stereo. The original intent was to average the results across observers, but since the methodologies were different, results are presented for each observer independently and compared by their relative effectiveness for each of the metrics.

Method Comparison

Total species richness and total abundance for both field and recording observations were tallied for each observer. To account for variability among sites, average richness (i.e., the number of unique species identified) and abundance (i.e., total number of individuals counted) were tallied at each site, first for the field observations and then for the recording interpretations. Both average richness and average abundance for the two methods were tested for statistical difference using a two-tailed paired Student's *t*-test. In addition to testing if the sampling methods were statistically different, the percentage of sites where richness or abundance were equivalent, decreased, or increased in the recordings versus the field observations was noted.

Each of the previous comparisons, while good as general descriptors, are limited by the fact that they did not account for species identifications. For example, perfect agreement in richness could be achieved by ten species observed in the field and ten different species observed from the recording; yielding equivalent numeric results but different species compositions. To address this issue and to provide a direct comparison with the results achieved by Hobson et al. (2002), I calculated a suite of community similarity indices using Ecostat 1.0.2 (http://www.exetersoftware.com/ cat/trinity/ ecostat.html) that included: (1) percent similarity; (2) the Jaccard similarity coefficient; and (3) the Sorenson coefficient. Results for each of these indices are reported, but for analytical purposes focused on percent similarity (i.e., the species occurrence and proportional abundance) for its ease of interpretation. Because the two observers interpreted the recordings using different methods, I tested their pair-wise site-level differences in percent similarity against using a paired Student's *t*-test.

To determine if individual species influence recording effectiveness, I calculated the average number of sites where a species was observed (i.e., encounters) and average abundance per site for each species for both the field observations and then for recording interpretations. The species-level counts were not normally distributed; therefore, I tested for differences between recording and field-based abundances and encounters using a Wilcoxon signed-rank test. I also compared site-level abundance and encounter totals for each species independently. Abundance differences were tested using a Wilcoxon signed-rank test while encounter differences were tested using a McNemar test (which is commonly used in a repeated measure, binary situations; SPSS 15.0). In addition to the statistical tests, I calculated the number of species where abundances and encounters were equivalent, underestimated, or overestimated by the recordings interpretations. All recording versus field comparisons were assessed with a critical value of $\alpha = 0.05$.

Understanding the ability of observers to spatially locate sample specimens from the recordings was important for two reasons. First, like Hobson et al. (2002), I was concerned that differences in detection distances between the methods might bias the results. Second, I hypothesized that some level of spatial location perception is lost when interpreting from the recordings, and that this loss directly influences an observer's ability to accurately count the number of birds at a site. To discern spatial locating ability from the recordings, all data logs from a field-observation and its matching recording observation were directly compared, by first matching individual locations of each bird (including all documented moves of each individual), then evaluating each match-pair to determine if the recording-observed location was in a similar direction and distance class as the field-observed location. The matches were evaluated to determine if the interpreted recording observations were similar in one of four quadrants (NE, SE, SW, NW), separated by the cardinal directions. It was possible for the direction to match, to be transposed from front-to-back or left-to-right, or to have crossed diagonally (e.g., NE-to-SW). Similarly, distances were evaluated for three distances classes (0-50 m, 50-100 m, and >100 m). Distance comparisons could produce a match, or recording observations that were either closer or farther, as compared to the field observation. Totaling all possible combinations of direction and distance classes, it was possible for each comparison to fall into one of 12 agreement classes (combining the categories of distance and direction agreement). Percent agreement was evaluated for each observer independently. As with other metrics, potential species-level effects on spatial locating ability were of interest. I tallied percent agreement for the 12 spatial-locating classes, this time aggregated by species, and calculated percent agreement by observer. Then, I performed a linear regression of percent correctly located versus median song frequency for each species identified.

To describe the influence of a suite of predictor variables on percent similarity for each observer independently, I first tested all variables for normality by calculating the Kolmogorov–Smirnov goodness of fit test (SPSS 15.x, 2007). If necessary, each variable was transformed using either Log_{10} or arcsine square-root. All variables were tested for correlation and variance inflation factors (VIF) were examined to assess the possibility of collinearity among the landscape variables. Several variables (e.g., development density, total road length, ambient noise level, temperature, and combined richness) were excluded from further analysis because of high correlation and high VIF. Hence, I performed multiple regression analysis using percent similarity as the dependent variable and (a) percent low frequency (arcsine square-root transformed); (b) humidity; (c) time of day; (d) mean slope (Log_{10} transformed); (e) combined abundance; and, (f) percent correctly located (spatially) as the independent variables. Regression predictor variables were assessed with a critical value of $\alpha = 0.10$.

Results

Across all times, sites, and observers, a total of 39 forest bird species were identified. Observer A identified 35 bird species and a total of 471 individual birds in the field, and 32 species and 392 birds from the recording interpretations. Observer B identified 32 bird species and 529 individual birds in the field and 27 species from 415 birds from the recording interpretations. For the overall sampling effort, recording-based total species richness accounted for 91.4% and 83.2% of field-based total species for Observer A and Observer B, respectively. Similarly, recording-based total abundance accounted for 84.4% and 78.4% of field-based total abundance species for Observer A and Observer B, respectively.

At the site level, mean richness for Observer A based on field observation was 7.32 (± 1.9) species per site, while the recording interpretations captured an average of 6.93 (± 2.2) species per site (Table 3.1), which were no different from one another ($p = 0.166$; $t = 1.41$; $df = 43$). The recordings accounted for 94.7% of the field-based site richness on average. Only 9 of 44 (20.5%) recording-based richness estimates perfectly (i.e., 100%) matched the field-based estimates. The balance of the sites were either underestimated (21 of 44; 47.7%) or overestimated (14 of 44; 31.8%) by the recording interpretations.

Mean site abundance for Observer A based on field observations was 10.14 (± 3.0) individuals per site, while the recording interpretations captured an average of 8.91 (± 2.8) individuals per site (Table 3.1), resulting in significantly fewer birds being counted ($p = 0.005$; $t = 3.00$; $df = 43$). In this case, the recording interpretations described 87.8% of the average site-level abundances. Five of 44 (11.4%) recording-based abundance estimates perfectly matched the field-based observations. The balance of the sites were either underestimated (28 of 44; 63.6%) or overestimated (11 of 44; 25.0%) by the recording interpretations.

For observer B, average field-based richness was 7.95 (± 2.4) species per site compared to recording-based estimates which were 7.25 (± 2.6) species per site (Table 3.1), indicating a statistically significant reduction ($p = 0.022$; $t = 2.39$; $df = 43$). The recordings accounted for 91.1% of the site richness. Eleven of 44 (25.0%) recording-based richness estimates perfectly matched the field-based observations. Richness values for the balance of the sites were either underestimated (23 of 44; 52.3%) or overestimated (10 of 44; 22.7%) by the recording interpretations.

Average abundance for Observer B based on field observations was 12.02 (± 2.9) individuals per site while the recording interpretations captured an average of 9.43 (± 3.4) individuals per site (Table 3.1), resulting in significantly fewer recording-based bird counts ($p < 0.000$; $t = 6.33$; $df = 43$) as compared to field-based abundances. In this case, the recording interpretations described 78.5% of the average site-level abundances. Five of 44 (11.4%) recording abundance estimates matched the field observations. The balance of the sites were either underestimated (34 of 44; 77.2%) or overestimated (5 of 44; 11.4%) by the recording interpretations.

Accounting for differences in richness and abundance simultaneously by calculating standard community similarity indices, the recording-based interpretations have between a 61% and 82% match to the field observations, depending on the similarity index and observer assessed (Table 3.2). Percent similarity, indicated an average match of 71.8% (± 10.9%) between the methods for Observer A, and a 66.2% (± 11.8%) match for Observer B. Observer B's recording-based interpretations were significantly lower ($p = 0.034$; $t = 2.19$; $df = 43$) in percent similarity as compared to Observer A.

Investigating the species-level effects on Observer A recording interpretation effectiveness, I found that the number of sites where a species was encountered in the field versus on the recordings (8.3 ± 9.9 and 7.8 ± 10.2, respectively) was not statistically different ($p = 0.200$; $t = 1.304$; $df = 38$; Table 3.3). Conversely, average abundance of individuals from a species (11.6 ± 16.4) as sampled in the field, when compared on a paired basis, was significantly different ($p = 0.009$; $t = 2.761$; $df = 38$) from the abundance interpreted from the recordings (10.1 ± 14.6). For the 37 species identified by Observer A, the correct number of sites, where each species was encountered, was estimated from the recordings for only three species (8.1%) while the total abundance was correctly estimated for six species (16.2%). For the balance of the species, the number of encounters was underestimated for 19 (51.4%) species and overestimated for 12 (32.4%) species, while the total abundance of 23 species (62.2%) was underestimated and eight species (21.6%) were overestimated. I found that significant differences in abundance (Table 3.3) included: Red-bellied Woodpecker (*Melanerpes carolinus*; $p = 0.013$); Eastern Wood-pewee (*Contopus virens*; $p = 0.023$); Blue Jay (*Cyanocitta cristata*; $p = 0.034$); Red-eyed Vireo (*Vireo olivaceus*; $p = 0.051$); and Blue-gray Gnatcatcher (*Polioptila caerulea*; $p = 0.007$). Differences in encounter estimates were insignificant for all species.

For Observer B (Table 3.3), I found that recording-based encounters (9.0 ± 10.2), when compared on a paired basis, were significantly fewer ($p = 0.032$; $t = 2.230$; $df = 38$) than field-based encounters (8.2 ± 9.4). Similarly, average abundances of individuals from a species as sampled in the field (13.8 ± 18.2), when considered on a paired basis, were significantly different ($p = 0.002$; $t = 3.416$; $df = 38$) from the abundances interpreted from the recordings (10.6 ± 13.5). For the 32 species identified by Observer B, the correct number of sites and total abundances were estimated from the recordings for only two species (6.3%). For the balance of the species, the number of encounters was underestimated for 21 (65.6%) species and overestimated for nine (28.1%) species, while the total abundance of 22 species (68.8%) was underestimated and eight species (25.0%) were overestimated. For Observer B, significant species-level underestimates in abundance (Table 3.3) include: Blue Jay (p = 0.018); Eastern Towhee (*Pipilo erythrophthalmus*; p = 0.046); Northern Cardinal (*Cardinalis cardinalis*; p = 0.001); Red-eyed Vireo (p = 0.001); Black-capped Chickadee (*Poecile atricapillus*; p = 0.048); and Wood Thrush (*Hylocichla mustelina*; p = 0.021). Differences in encounter estimates were insignificant for all species.

When comparing the 19 species that were observed by either observer at least 4 times in the field (an arbitrarily selected threshold), the effectiveness of the recording-based interpretations, for correctly estimating overall abundance for each species, is in disagreement for 31.5% of the species (Figure 3.3), for the two observers. That is, for nearly one-third of the species, a species' abundance is overestimated by one observer, and underestimated by the other observer. Similarly, the effectiveness of the recording-based interpretations varied between the observers, for correctly estimating the number of encounters for each species. Species encounters were in disagreement for 31.5% (6/19) of the species (Figure 3.4). Using Acadian Flycatcher (*Empidonax virescens*) to serve as an example of observer effect; we see that Observer A's results suggest that Acadian Flycatcher is documented at more sites (150%) and at a greater abundance (120%) with the recording observations, while Observer B's results suggest that the Acadian

Flycatcher is found at less sites (85.7%) and fewer individuals are counted (70.0%), using the recording equipment as compared to field observations (percents derived from Table 3.3).

Observer A was able to accurately replicate the direction of a calling or singing bird for 54.8% of the individuals counted and the distance for 75.1% of the observations, whereas Observer B was able to accurately replicate the direction of a calling or singing bird for 57.3% of the individuals counted and the distance for 66.3% of the observations (Table 3.4). Accounting for distance and direction simultaneously, Observer A was able to accurately locate individual birds 41.0% (± 23.0) of the time while Observer B demonstrated a mean accuracy of 39.0% (± 23.3). When accounting for direction and distance simultaneously, the most common errors included front-to-back inversions (25.0% and 15.2% of the samples for Observers A and B, respectively) with the distance being correct, and interpreting an individual's location to be closer than determined in the field (7.9% and 16.4% of the samples, Observers A and B respectively), while the direction was estimated correctly.

Comparisons of spatial locating ability at the species level revealed no discernable pattern, i.e., there was no observable species, or song-frequency, effect on spatial locating ability for either observer. Observer A correctly located six (Great-crested Flycatcher [*Myiarchus crinitus*], Eastern Wood-pewee, Rose-breasted Grosbeak [*Pheucticus ludovicianus*], Ovenbird [*Seiurus aurocapilla*], Black-capped Chickadee, and Hermit Thrush [*Catharus guttatus*]) of 22 species, greater than 50% of the time while Observer B correctly located three (Acadian Flycatcher, Veery [*Catharus guttatus*], and Yellow-throated Vireo [*Vireo flavifrons*]) of 24 species greater than 50% of the time. Similar to previous findings, the most common errors included front-to-back inversions and interpreting an individual's location to be closer than determined in the field.

Multiple regression results demonstrated that the suite of measured environmental variables had little influence on percent similarity between field- and recording-based observations. Only the model for Observer A was significant (adjusted $r^2 = 0.092$; $df_{1,42}$; $F = 5.334$; $p = 0.026$), with only one predictor variable (i.e., Time of Day) begin important. All potential relationships for Observer B were non-significant leading to all variables being excluded from the regression model. The relationship between % similarity and time of day for Observer A was negative such that longer surveying times decreased the similarity agreement.

Discussion

When comparing the bioacoustics recordings versus traditional point counts in human-dominated landscapes, I found that the same acoustic variables that seemed to affect recording quality likely also affected an observer when conducting a traditional point count. In addition, I found that Interpretations based on the recordings, though not in perfect agreement with field-based observations, can serve as an effective point-count mechanism. Haselmayer and Quinn (2000) suggest sound recordings perform as well or better than traditional point counts for determining species richness values. My results are consistent with the aforementioned suggestion, as the field and recording-based richness values were statistically similar. However, richness was commonly underestimated using the recording-based approach, even though agreement averaged better than 90% for both observers. In the end, I found no evidence to support my prediction that the recording method provides increased estimates of species richness as compared to the field-based method.

Since richness is generally in agreement, poorer than expected percent similarity (relative to Hobson et al. 2002), is likely related to species-specific abundance counts. Incorrect abundances could result directly from miscounting, but could also be related to misidentification of species which sound similar and were identified as different species in the field and on the recording. Since each species is not time stamped, the problem is that it is difficult to determine which species identification is correct, from the field or from the recording. Based on an assessment of species that sound similar (e.g., Rose-breasted Grosbeak, Scarlet Tanager [*Piranga olivacea*], and American Robin [*Turdus migratorius*]) and are likely to be misidentified, I compared all individuals that were missed in the recordings (as compared to the field observations) against all individuals that were missed in the field observations (as compared to the recording interpretations). I found that 4.3% and 4.5% (for Observer A and Observer B, respectively) of the individuals missed in the recordings could have resulted from misidentification. Therefore, I posit that the estimates for percent similarity are conservative and would marginally increase with a re-evaluation based on misidentifications.

In addition, while I excluded from the analyses all field observations that were denoted as visually-only detections it is possible that the ability to audibly and visually observe birds in the field, as compared to audible-only observations from the recording interpretations, falsely inflated the field measurements. The visual detection artifact has two implications. First, if visual detections did inflate an observer’s ability to detect birds in the field, again, the effectiveness of the recordings should be considered conservative with an expected increase in agreement if field observations are successfully restricted to audible-only observations. Second, if visual detections do assist (intentionally or not) in audible-only observations, then the measured decrease in recording effectiveness (compared to Hobson et al. 2002) would be expected, simply because of differences in habitats where the tests were conducted. Hobson's comparisons were conducted in mixed-hardwood forests where visual detectability would be relatively low, as compared to this study that was completed largely in woodlots surrounded by urbanization and agriculture. That is, the current study landscapes are considerable more open, increasing the possibility of visual detections and a relative reduction in agreement between the methods.

Within the study area, locations with more human influence (i.e., higher built-structure density and more roads) were typically louder (via ambient noise interpretations) and had higher percents of low frequency noise (<1 kHz). Even so, based on a regression of percent similarity on structure development-density, I found no evidence to support my prediction that higher development densities, and the consequent signal contamination, would increase the disagreement between the point-count methods. I also predicted that a higher proportion of low frequency sounds, which I have shown to correlate with human development, would negatively influence recording interpretation effectiveness. From a quantitative perspective, I found no evidence to support this prediction. The lack of a relationship between percent similarity and low frequency noise suggests that higher proportions of low frequency background noise do not negatively influence an observer's ability to perform recording-based point-counts as compared to traditional counts. Based on the strong correlation ($r = 0.71$; $p < 0.001$)

between recording- and field-based ambient noise interpretations, this result suggests that recording interpretations are subject to the same ambient noises, and similar resulting levels of distraction associated with those noises, as those based on the field-based observations.

Qualitatively, low frequency sounds severely degraded the quality of the recordings, and comparatively, those sounds influenced the recordings more significantly than listening by the human ear alone. Low frequency noises of primary concern included automotive and airplane traffic. Because the observation sites were distributed along an urban to rural gradient, many of the sites were in close proximity to roads, some of which were highways. The sensitivity of the microphone was a severe disadvantage here, compared to field-based observations. Road noise would, on occasion, flood the microphone, accounting for nearly 100% of the energy being received and leading to a phenomenon where all other sounds (e.g., bird songs) were distorted or masked. Airline traffic was limiting as well. When passing close to the sample location (which was relatively common), the airplanes caused a similar microphone flooding as experienced with road noise.

In addition to traffic-based noise, sounds created from wind, wildlife (both birds and other fauna), and to a small extent, other human activities (e.g., recreation and lawn mowing) in and around the study sites were also inhibitive. Wind had indirect effect by causing foliage rustling in the tree canopy. Wildlife, in particular squirrels and chipmunks, added additional extraneous noises to the recordings. In several cases, squirrels in close proximity to the microphones barked during an entire recording. While this seemed to have no influence on the birds singing, it added unwanted noise to the recordings. Finally, I found that noises related to other human activities (e.g., joggers, golfers, kids playing, and lawn mowers) within and around the survey sites were also inhibitive.

My prediction that abundances will be underestimated from the recording interpretations is generally supported, and follows other research results that have documented reduced abundance from recording-based point counts (Haselmayer and Quinn 2000; Cunningham et al. 2004; Simons et al. 2007). The combined recording interpretations: (1) accounted for 81% of the total abundance of birds counted; (2) underestimated abundances for 70% of individual sites; and (3) resulted in significant underestimates in the average number of birds counted per site. I predicted that this trend would result from reduced spatial locating perception when listening to the recordings through a stereo headset, and specifically, I assumed that recording interpretations would result in birds being perceived as closer with possible inversion of direction (e.g., front-to-back locating error). While I observed that reduced counts occurred for the majority of locations and species, and I have documented a relationship between spatial locating ability and percent similarity for Observer A, I found no direct tie between spatial locating ability and abundances.

The results of my analysis suggest that factors that qualitatively degrade recording interpretation (e.g., ambient noise levels and proportion of low frequency sounds) had no quantitative relationship with the effectiveness of the recorded observations for point counts (as compared to field observations) along a gradient of urbanization. It is possible that my methods for capturing ambient noise levels were insufficient. First, I used an ordinal scale because the sound level meter was not sufficiently sensitive to use a continuous dB scale; the low dB level for a disproportionate number of sites was less than 40 dB, the minimum sound level recorded with the meter. Second, when determining the energy recorded that was <1 kHz, it should be noted that a high percent of low frequency noise can be achieved in two ways. The high proportion could be the result of "strong" (i.e., many or close) low frequency noises with an abundance of singing birds, or "weak" (i.e., few or farther away) low frequency noises with few to no birds singing. My goal was to capture the highs and lows of the frequency (kHz) distribution. However, it is likely that only the background noise occurring exactly when a bird is signing could have negatively affected the song reception. If individual instances of comparing bird songs to background low frequency noise are not adequately represented, then the proportion of low frequency noise would be skewed.

Observer A was able to capture species richness, and to determine the correct species composition at each site. Even so, Observer A underestimated the number of birds both by site and by species. Therefore, a full stereo-recording playback provides a means of identifying species within a habitat patch, but will generally result in undercounting the number of individuals in the same patch. Because I was primarily concerned with identifying species using a forest patch of interest for any reason, and less concerned with overall abundances, the recording method is suitable for my continued research on the avian landscape ecology of forest birds in Southeast Michigan (Chapter 4).

Observer A's recording interpretations more closely matched the field observations compared to Observer B. Because the observers had similar birding experience, they surveyed the same set of sites, and the observer bias was removed by having each observer interpret only those recordings that were collected simultaneously with their field observations, I suspect the difference is largely due to the single microphone Observer B inadvertently used during playback (as described in the methods section).

Hobson et al. (2002) suggested that advantages of recording-based sampling include: (1) extended sampling efforts by allowing for the opportunity to perform sampling outside of the short breeding-season; (2) increased monitoring by allowing for recordings to be collected by individuals who are not expert birders; (3) control over observer variability by allowing a single, skilled observer to interpret recordings for many locations; and, (4) long-term quality control through archiving of field samples that can be re-evaluated if results are in questions. Acevedo and Villanueva-Rivera (2006) add (5) the future possibility of automated species identification of recorded samples. Based on my research, disadvantages to recording samples include reduced spatial perception while interpreting recordings, underestimation of bird abundances (supporting previous research), and increased processing time because each observation is recorded once in the field, but also requires as much or more time to evaluate the recordings in the lab. Hence, Cunningham et al. (2004) suggest that because of additional processing time, sound-recording point counts "…may not be the most effective field method in comparison with standard human observation efforts" (pg. 205). In an effort to reduce this concern, I asked the observers to listen to the recordings only once, repeating or stopping only for those species that were unknown. Average interpretation times were 13 minutes, for a 10-minute recording. While recording interpretations were not much longer than standard field observations on average, the time for data collection is nearly doubled (once to record, and once to interpret). Although, it should be noted that lab-based interpretation times do not include travel and setup, and many recordings can be interpreted in a relatively short period. Even so, there is a trade-off between the cost of spending additional time estimating birds, and the benefit of establishing permanent and consistent approaches for long term monitoring.

During the preliminary planning stages I had a concern about distance perceptibility when interpreting the recordings. My concern was that the field observer and the recording interpreter would hear birds out to different distance thresholds, therefore skewing the comparisons. Hobson et al. (2002) had a similar concern and tested distance detectability for several species out to 250 meters. They concluded that their recording interpretations came "close to approximating the human ear." To test distance perception in the study area, I compared estimated locations from field versus recording logs for individuals observed on both the recording and in the field. With a nearly 75% distance-agreement for Observer A, I determined, like Hobson et al. (2002), that stereo recording-based distances reasonably match those as determined from the field observations.

One additional concern I had is the effectiveness of a field observer for conducting a point count and colleting an entire community of species at a site. My analysis assumes that the results from traditional field-based counts provided accurate estimates of species composition, for both richness and abundance. It is possible though that, because of the time constraints, environmental conditions, and detectability issues, individuals were either missed or misidentified during field observation. Therefore, although the field observations may be incomplete, they provided the standard for the comparisons.

Conclusions

In comparing the effectiveness of CZM bioacoustics recordings versus traditional point counts of avian populations in human-dominated landscapes, interpretations based on the recordings, though not in perfect agreement with field-based observations, can serve as an effective point-count mechanism. The recording-based interpretations describe better than 90% of the species richness and better than 80% of abundance compared with traditional field observations, and provide a nearly 70% match (which is likely a conservative estimate) when accounting for richness and abundance simultaneously (via the percent similarity index). Comparing my results with those of Hobson et al. (2002), who performed a similar study in the southern boreal mixed-woods of central Canada, I found that each of the community indices demonstrate less agreement between the recording- and field-based observations. It is possible that specific factors exist in human-dominated environments that negatively influence the effectiveness of the recording method, but I did not find any. While my data analyses cannot explain why I have experienced lower community match indices, compared with Hobson, the analysis does suggest that method-comparison results are similar along the entire urbanization gradient. Therefore, I posit that using stereo-recordings for avian point-counts is equally effective at all levels of human influence found within the study area.

Table 3.1. A comparison total species richness and total abundance for each sample site between the standard field observations and recording interpretations for Observers A and B.

		Observer A				Observer B			
Site ID	Landuse	Field-Rich	Rec-Rich	Field-Abund	Rec-Abund	Field-Rich	Rec-Rich	Field-Abund	Rec-Abund
002	exurban	10	10	16	11	7	7	12	9
003	suburban	3	3	6	5	6	6	9	6
004	exurban	11	10	18	14	4	4	13	6
006	exurban	8	8	10	9	7	8	11	10
009	suburban	6	5	6	7	7	7	10	8
010	rural	4	3	6	4	8	3	11	4
012	suburban	6	4	6	5	7	5	11	7
014	suburban	6	5	7	6	5	4	7	4
015	urban	9	6	10	6	7	6	13	7
016	urban	5	4	7	7	7	3	12	5
017	urban	4	5	8	7	4	5	9	6
018	urban	11	8	16	10	6	6	11	9
020	urban	5	7	10	10	8	8	11	9
022	suburban	6	5	11	6	4	4	11	6
025	suburban	7	7	11	9	8	9	13	12
027	suburban	7	8	11	11	5	3	8	6
028	suburban	7	6	11	9	9	9	14	14
029	urban	10	11	13	11	9	8	12	11
033	urban	6	11	8	13	8	10	11	11
034	exurban	10	7	14	8	5	4	9	6
037	exurban	8	9	11	11	9	8	15	10
038	exurban	8	8	10	11	8	5	13	6
044	exurban	7	9	9	13	9	10	14	14
046	suburban	9	9	14	11	10	7	13	10
047	rural	8	6	10	8	9	7	12	11
048	suburban	6	4	9	6	8	7	11	9
051	urban	8	7	9	9	6	7	10	14
054	exurban	7	7	9	10	11	9	13	11
056	rural	7	7	9	7	11	7	15	9
057	rural	8	3	8	3	8	10	17	12
058	suburban	8	7	11	9	10	14	16	16
059	exurban	6	4	8	5	7	7	8	8
060	rural	8	9	10	12	3	5	6	7
061	rural	7	9	7	12	11	9	11	12
062	rural	11	10	17	15	9	7	14	9
064	rural	7	6	10	9	9	8	14	9
065	rural	9	5	12	6	13	14	19	18
066	rural	7	7	8	10	8	4	10	6
069	rural	9	10	12	11	9	7	12	7
070	rural	7	8	9	10	13	10	19	13
078	rural	8	6	11	7	12	12	15	17
080	rural	6	7	13	11	10	10	13	12
081	rural	6	7	7	8	5	8	7	9
083	rural	6	8	8	10	11	8	14	10
	Site Averages	7.32	6.93	10.14	8.91	7.95	7.25	12.02	9.43
	Site S.D.	1.9	2.2	3.0	2.8	2.4	2.6	2.9	3.4

Table 3.2. A comparison of the overall counts and site-based community similarity indices between the standard field observations and recording interpretations for Observers A and B.

Community Similarity	Observer A	Observer B
Number of sample comparisons	44	44
Overall species richness from field, recording	35, 32	32, 27
Overall abundance from field, recording	471, 392	529, 415
Avg. % Similarity	71.76 +/- 10.9	66.2 +/- 11.8
Avg. Jaccard Coefficient	0.70 +/- 0.1	0.61 +/- 0.2
Avg. Sorenson Coefficient	0.82 +/- 0.8	0.74 +/- 0.7

Table 3.3. A comparison of the number of sites where each species was encountered and the total abundance for each species between the standard field observations and recording interpretations for Observers A and B, sorted taxonomically.

			Observer A				Observer B			
			Number of Sites		Abundance		Number of Sites		Abundance	
Species	Latin Name	AOU Code	field	recording	field	recording	field	recording	field	recording
Wild Turkey	*Meleagris gallopavo*	WITU	1	0	1	0	0	0	0	0
Cooper's Hawk	*Accipiter cooperii*	COHA	2	0	2	0	0	0	0	0
Red-tailed Hawk	*Buteo jamaicensis*	RTHA	1	0	1	0	1	0	1	0
Red-shouldered Hawk	*Buteo lineatus*	RSHA	1	1	1	1	0	0	0	0
Hairy Woodpecker	*Picoides villosus*	HAWO	2	0	2	0	0	0	0	0
Downy Woodpecker	*Picoides pubescens*	DOWO	8	8	9	8	17	18	20	21
Red-bellied Woodpecker	*Melanerpes carolinus*	RBWO	23	17	28	19	18	13	25	15
Northern Flicker	*Colaptes auratus*	NOFL	3	2	3	2	1	4	1	5
Ruby-throated Hummingbird	*Archilochus colubris*	RTHU	0	0	0	0	6	9	8	10
Great Crested Flycatcher	*Myiarchus crinitus*	GCFL	7	6	7	6	5	4	5	4
Eastern Wood-Pewee	*Contopus virens*	EAWP	30	33	37	44	31	33	51	47
Acadian Flycatcher	*Empidonax virescens*	ACFL	4	6	5	6	7	6	10	7
Least Flycatcher	*Empidonax minimus*	LEFL	1	1	1	1	0	0	0	0
Blue Jay	*Cyanocitta cristata*	BLJA	16	12	22	16	17	13	27	15
American Crow	*Corvus brachyrhynchos*	AMCR	15	18	26	27	19	16	27	21
Baltimore (Northern) Oriole	*Icterus galbula*	BAOR	3	1	3	1	2	0	2	0
Eastern Towhee	*Pipilo erythrophthalmus*	EATO	1	1	1	1	7	4	8	4
Northern Cardinal	*Cardinalis cardinalis*	NOCA	35	37	69	59	34	30	71	50
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	RBGR	3	5	3	6	6	3	8	3
Scarlet Tanager	*Piranga olivacea*	SCTA	18	19	18	20	3	5	4	5
Red-eyed Vireo	*Vireo olivaceus*	REVI	36	35	65	55	35	31	64	42
Yellow-throated Vireo	*Vireo flavifrons*	YTVI	8	6	9	6	13	10	19	15

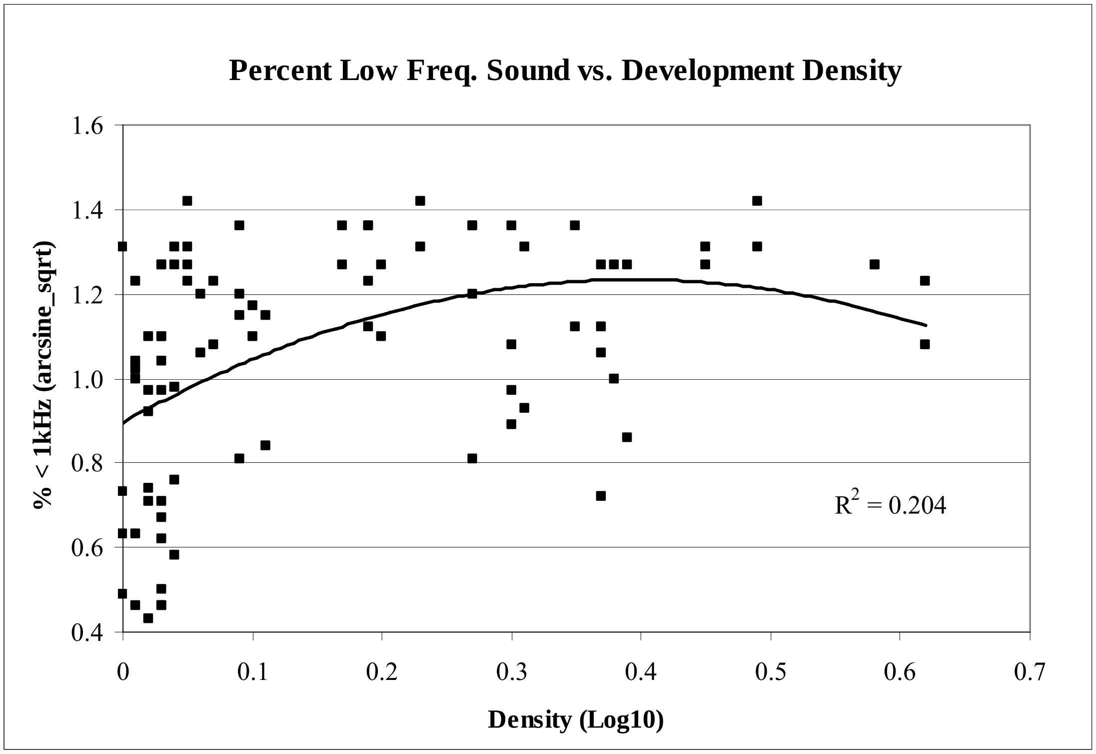
Black-throated Blue Warbler	*Dendroica caerulescens*	BTBW	1	3	1	4	2	2	2	2
Cerulean Warbler	*Dendroica cerulea*	CERW	0	0	0	0	1	3	2	4
Black-throated Green Warbler	*Dendroica virens*	BTNW	2	1	2	1	1	0	2	0
Pine Warbler	*Dendroica pinus*	PIWA	1	1	1	1	0	0	0	0
Ovenbird	*Seiurus aurocapillus*	OVEN	7	8	12	12	10	13	14	17
Hooded Warbler	*Wilsonia citrina*	HOWA	0	2	0	2	1	0	1	0
American Redstart	*Setophaga ruticilla*	AMRE	4	1	6	1	1	0	1	0
Carolina Wren	*Thryothorus ludovicianus*	CARW	2	1	3	1	2	2	2	2
House Wren	*Troglodytes aedon*	HOWR	2	0	2	0	4	5	4	5
White-breasted Nuthatch	*Sitta carolinensis*	WBNU	12	11	14	11	17	13	18	13
Tufted Titmouse	*Parus bicolor*	TUTI	26	28	35	33	23	20	35	27
Black-capped Chickadee	*Poecile atricapilla*	BCCH	9	11	15	13	25	24	45	34
Blue-gray Gnatcatcher	*Polioptila caerulea*	BGGN	13	8	17	8	4	7	5	7
Wood Thrush	*Hylocichla mustelina*	WOTH	8	7	10	7	13	12	23	15
Veery	*Catharus fuscescens*	VEER	0	1	0	1	6	5	7	6
Hermit Thrush	*Catharus guttatus*	HETH	3	3	3	3	0	0	0	0
American Robin	*Turdus migratorius*	AMRO	14	11	19	16	18	14	27	19

Table 3.4. The percent of spatial occurrences (i.e., individual bird locations, or multiple locations of same bird) that were accurately located accounting for direction and distance for each observer from the recording interpretations as compared to field observations. Direction classes include the correct direction (leftmost Yes) or direction inversions including front-to-back (F/B), left-to-right (L/R), or crossing (Cross; front-right to left-back). Distances were evaluated for three distances classes and determined to be correct (rightmost Yes) or farther or closer. Spatial n describes the average number of spatial occurrences evaluated for each sample site.

Spatial Class	Obs. A	Obs. B
Spatial n=	7.9	7.3
Yes-yes	41.0	39.0
L/R-yes	5.1	5.6
F/B-yes	25.0	15.2
Cross-yes	4.0	6.5
Yes-closer	7.9	16.4
L/R-closer	2.8	5.6
F/B-closer	3.7	5.8
Cross-closer	0.4	3.3
Yes-farther	6.0	1.9
L/R-farther	2.9	0.6
F/B-farther	0.3	0.2
Cross-farther	0.9	0.0
Dir. Correct	54.8	57.3
L/R	10.8	11.8
F/B	29.1	21.2
Cross	5.3	9.8
Dist. Correct	75.1	66.3
Closer	14.8	31.1
Farther	10.2	2.6

Figure 3.1. The study area is located in Southeast Michigan in Washtenaw County. The sample locations (dots in county inset) are focused on patches of publicly accessible woodlands. The sites are distributed along a diagonal from SE to NW, extending from the more populated Ann Arbor area to the rural northwest.

Figure 3.2. Comparison of built-structure density (Log10 transformed) and total road length (within 1 km of the sample location; Log10 transformed) versus percent low frequency noise (< 1 kHz; arcsine square-root transformed) for all 88 sample locations. Total road length explains the proportion of low frequency noise at any given site, better than development density.



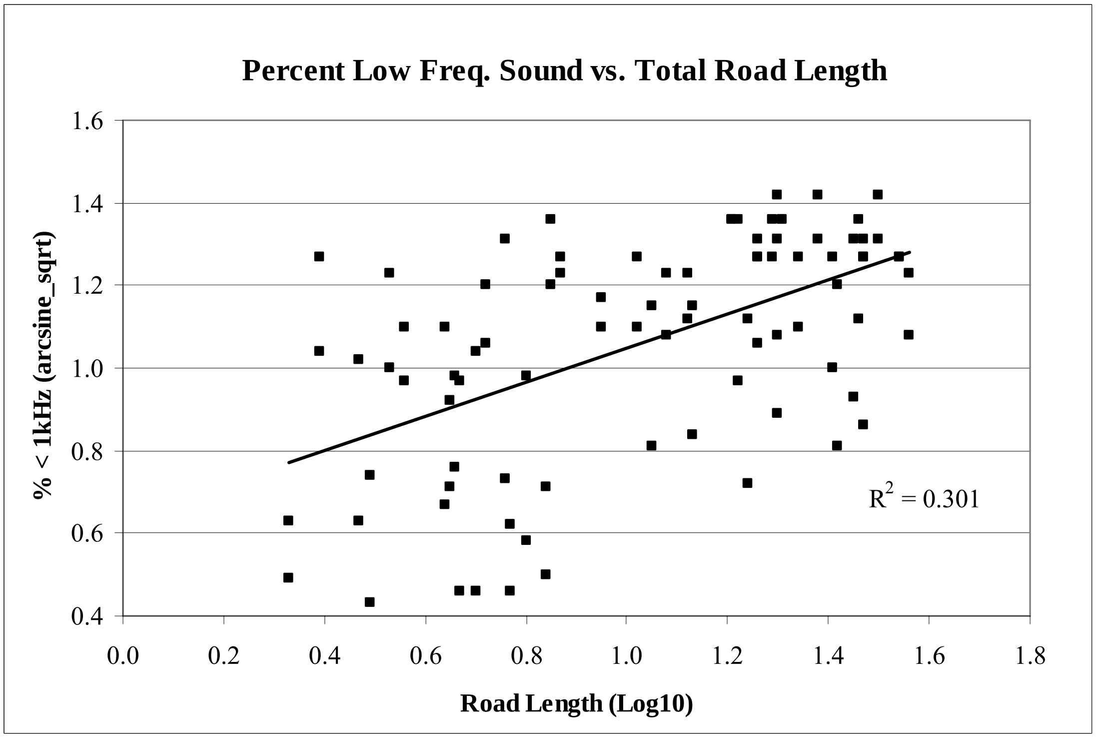
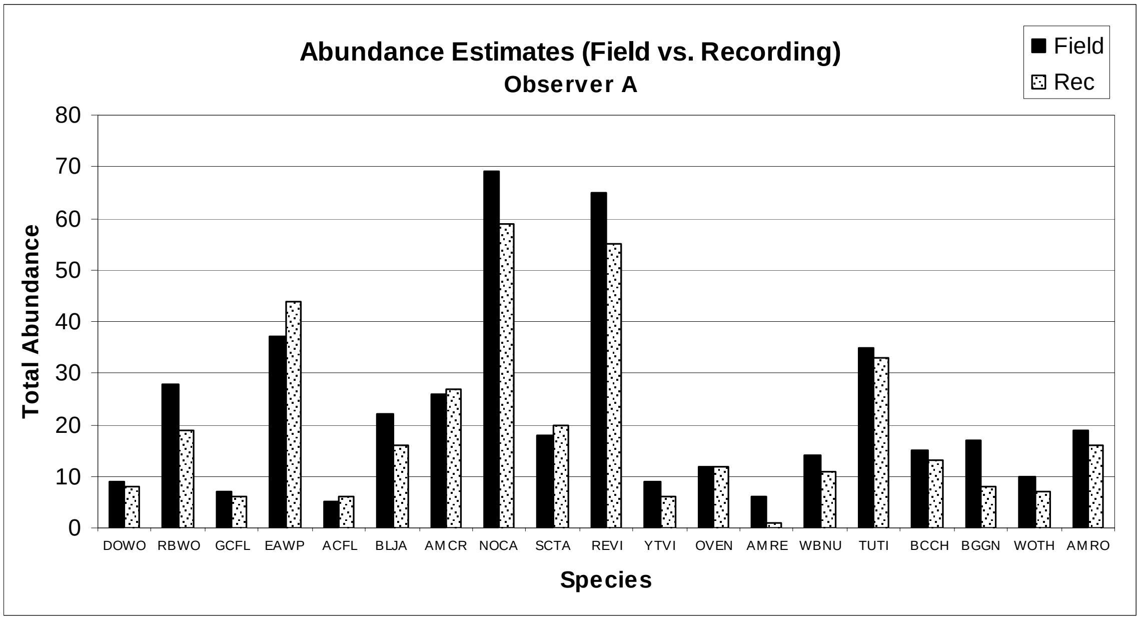


Figure 3.3. A comparison of individual species abundances, for the 19 species counted at least four times for either observer, between standard field observations and recording interpretations for Observers A and B. Refer to Table 3.3 for species code interpretations.



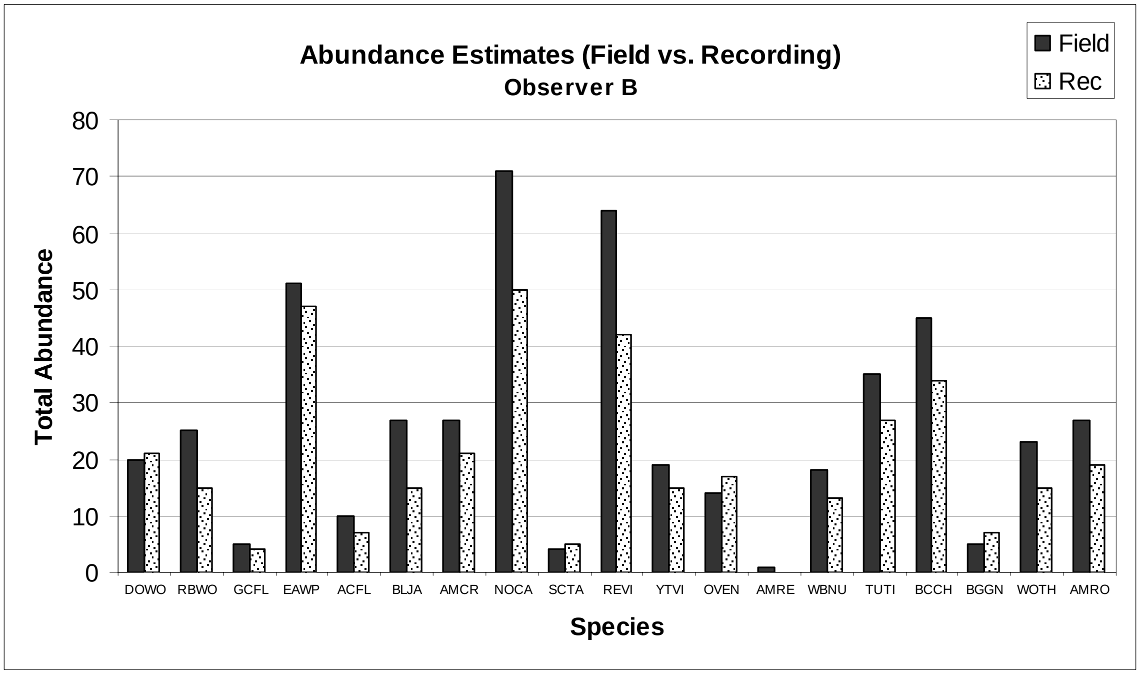
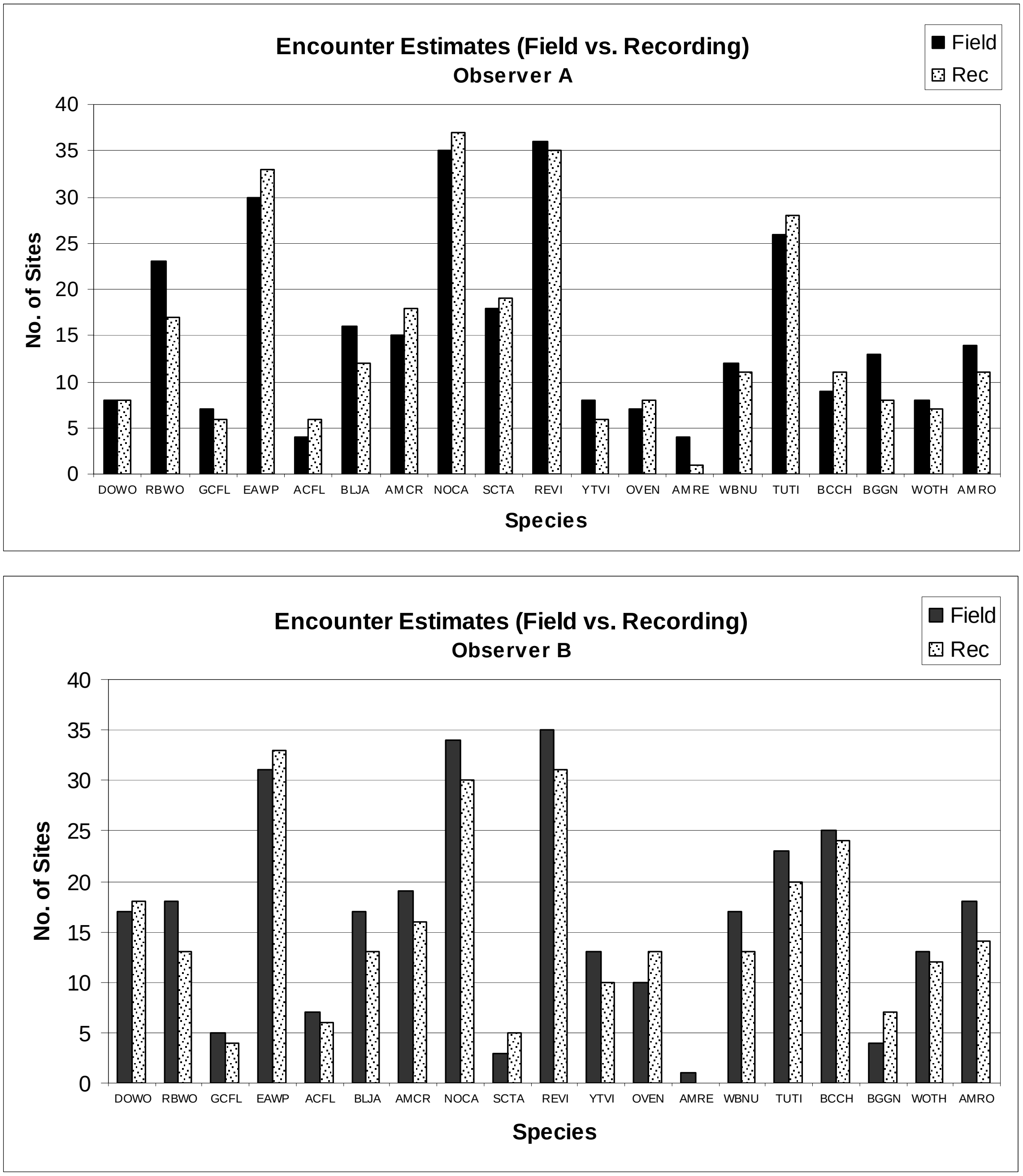


Figure 3.4. A comparison of individual species encounters (number of sites), for the 19 species counted (i.e., total abundance) at least four times for either observer, between standard field observations and recording interpretations for Observers A and B. Refer to Table 3.3 for species code interpretations.



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CHAPTER 4

MATRIX AND PATCH-SCALE INFLUENCES ON FOREST BIRDS IN SET-ASIDE WOODLOTS AT THE URBAN-RURAL INTERFACE

Introduction

According to Vitousek et al. (1997), humans have influenced nearly half of all landscapes on earth. A large portion of this influence results both directly and indirectly from urbanization. With urbanization comes land conversion (Brown et al. 2005), hydrological system disruption (Arnold and Gibbons 1996; Girling and Kellet 2002), and alterations of habitats and species compositions (Collinge 1996; McKinney 2002). Each of these effects may occur in landscapes via natural disturbances, but with time, one would expect perturbed systems to once again move toward a climax state. Urbanization as a disturbance is especially devastating, partially because of it permanence (Marzluff and Ewing 2001). Residential developments, shopping malls, and parking lots are unlikely to be re-established as natural habitats in the foreseeable future.

At the turn of the twentieth century, over 5% of the land in the United States (US) was classified as urban or built-up (McKinney 2002; Brown et al. 2005). At five percent, the US has more urban or built-up land, than land found in national parks, state parks, and Nature Conservancy preserves, combined (McKinney 2002). Much of the increase in urban land-use increase can be attributed to a growing population, but even in some areas where population is declining (e.g., Metropolitan Detroit) land continues to be developed at astonishing rates. Urbanization is the expansion of urban land uses including commercial and industrial, but also residential (Brown et al. 2005). Therefore, urbanization is not limited to core urban centers with high population and development densities, and high levels of impervious surface. For example, McKinney (2002) reports that in Missouri a large portion of urbanization is related to suburban growth where an estimated 54,600 hectares of land is in the form of residential yards, which is three-times the area occupied by Missouri State Parks. Urbanization also occurs at exurban and rural densities where residential developments are expanding and commonly intermixed with relatively large habitat remnants (Brown et al. 2005; Radeloff et al. 2005; Lepczyk et al. 2007).

Urban and built-up lands may be considered only a small portion of contemporary landscapes, both in the US and globally, but with population growth leading to increased urbanization [60% of the population is predicted to live in urban areas by 2030, as compared to 49% in 2005 (United Nations Secretariat 2006)], the proportion of urban lands will necessarily increase in the future. With urbanization likely to continue in some form in perpetuity (Marzluff and Ewing 2001), we have an obligation to offer science that can inform future landscape-development polices, leading to more ecologically sensitive development trends and patterns. To accomplish this task we must focus on the ecosystems that are in greatest threat of direct transition from natural to an urban state, those landscapes at the urban-rural interface (i.e., the leading edge of urbanization where multiple, interacting networks come together the link urban and rural areas (Browder 2002); hereafter referred to as interface). Landscapes at the interface are diverse, they do not conform to a simple dichotomy of urbanized or pristine, and they are not simply habitat and non-habitat (Blair 2004). Urbanization at the interface (i.e., urban sprawl), creates landscape mosaics that are complex (Blair 2004) both temporally and spatially, and commonly introduces contrasting mixes of land covers (e.g., impervious surfaces and tree cover).

In order to preserve natural settings within the interface and elsewhere at varying scales, it is common for policy-makers to set aside (i.e., preserve) suburban woodlots, wildlife refuges, national parks, and other natural areas. These set-aside locations are commonly intended to be refuges from direct human uses (e.g., urban land conversion, forestry, and agriculture), although most are still managed in some way and are indirectly affected (e.g., recreation, climate change, and isolation). The protected areas serve as pseudo-natural settings where humans can recreate and where other animals can meet their life-history needs. In relation to the former, it should be noted that recreational uses as seemingly non-harmful as the presence of recreational trails can affect some species (Miller et al. 2003). In regards to the latter, many argue (Collinge 1996; Donnelly and Marzluff 2004; Stratford and Robinson 2005) that large, contiguous set-asides are necessary to properly manage and promote long-term biodiversity, but there also exists a literature that supports the idea that where large protected areas are impractical (Janzen 1983) many small habitat fragments have important conservation value (Simberloff 1982; Simberloff and Abele 1982; Loman and Vonschantz 1991). Both kinds of habitats likely matter; the larger, more contiguous patches can serve as sources for regional species pools, while smaller patches can serve as “stepping stones” (Ricketts 2001) between the larger patches, post-breeding habitats, migratory stop-over sites, or simply as temporary dispersal endpoints (Faaborg 2002). Depending on the surrounding matrix (i.e., the composition and configuration of the surrounding landscape), it is also possible that smaller patches could serve the entire life-history needs of many species, including native or exotic, and resident or migrant species, even in urbanized environments.

When dealing with habitat islands in complex terrestrial landscapes like we find at the interface, the matrix may be integral in supporting species existence (Collinge 1996; Jokimaki and Huhta 1996; Fahrig 2001; Ricketts 2001; Brotons et al. 2003; Friesen et al. 2005). On the one hand, if matrix habitats are highly contrasting with focal habitats, resistance to movement may be high and significantly influence the effective isolation of a patch (Ricketts 2001). On the other hand, it may be possible that even the smallest, adjacent habitat fragments work in combination with habitat preserves to emulate, or serve as surrogates for much larger, core-area habitats. A similar argument is suggested by Ahlering and Faaborg (2006), when they support the thought that the amount of habitat area surrounding a study site could minimize the severity of local responses to patch size. The idea of developed landscapes sustaining interior habitat species was supported by recent study of Barred Owls (Harrold 2003; Mason 2004) in the urbanized landscapes of Charlotte, N.C., US, where they found that “old suburban neighborhoods in fact are an old growth forest, at least as far as the barred owls are concerned.”

In determining matrix influences on habitat usability, Rodewald (2003 pg. 588) states that an important question remains “Which is more important, the amount of forest or the type of land use?” Furthermore, Rodewald notes that knowledge about interactions between ecological communities and the matrix remains limited (Miller et al. 2001). Therefore, my goal was to test the influence of habitats and urbanization within the matrix on bird communities in set-aside forest patches. To accomplish this goal, I examined the occurrence of forest birds in forest patches located within the human-dominated urban-rural interface. I began from the hypothesis that a forest patch's location (and inherently the characteristics of the patch and the location) influences what avian species occur within the patch. Several predictions focused on forest-bird species richness including: (1) that there would be a negative relationship between total forest species richness and the level of urbanization surrounding the patch, but that this relationship will be buffered by the tree-cover area at a site; and, (2) that total tree-cover area in the landscape (accounting for adjacent, matrix-based habitat fragments) would be more influential on total forest species richness than the area of tree-cover in the focal patch alone.

Methods

Study Area

Site selection focused on forest patches along a gradient of housing densities within Washtenaw County, located in Southeast Michigan, US (Figure 4.1). Sample locations were selected within publicly accessible conservation and recreation lands that were surrounded by diverse land covers and distributed across a gradient of development densities (urban, suburban, exurban, rural, and natural) interpreted from aerial photography (method described below). To represent the distribution of development densities in the region, I defined natural development densities as 0.0 units (i.e., built structures)/hectare (ha); rural development densities as an average of ≤ 0.25 units/ha (i.e., ≤ 0.10 units/acre); exurban densities as 0.82 – 0.24 units/ha; suburban densities as 3.29 – 0.81 units/ha; and urban densities as > 3.29 units per ha (i.e., > 1.33 units/acre).

To select sites, I identified all publicly accessible conservation and recreation lands (data provided by Ducks Unlimited Great Lakes Atlantic Regional Office) in Washtenaw County (241 identified properties). Using 30 m resolution Integrated Forest Monitoring, Assessment, and Prescription (IFMAP) land-cover data (Zhao et al. 2007) from the Michigan Department of Natural Resources that were reclassified as binary forest (yes/no); the 241 sites were subset to include all sites with ≥3 hectares (ha) of forest land, yielding a total of 130 properties. Three hectares was selected because it is roughly equivalent to the area of a 100 m radius sample site, which was arbitrarily selected as an appropriate sampling distance for conducting avian point counts. From the 130 properties, 85 were determined to have a total of 157 tree-cover patches with 3-ha of contiguous forest. Using GIS, each of the patches was manually inspected as potential sampling locations by passing a 100 m radius circle across the landscapes using 1998, 1m color-infrared imagery (Michigan Center for Geographic Information, 2006). Final site selection was based on field inspections of the 85 locations, which reduced the sample size to 44 locations that were accessible and still contained forest patches. These 44 patches were distributed almost equally along the gradient (5 urban, 10 suburban, 9 exurban, 12 rural, and 8 natural). Sample locations were placed 100 m in from the edge of each woodlot to reduce roadside noise that can be a problem in surveys such as the North American Breeding Birds Survey (BBS; Bart et al. 1995; Keller and Scallan 1999). Sample locations were recorded using a recreational global positioning system (GPS; Garmin Map12) with an accuracy of ±10 m.

Bird Data

A total of 176, 10-minute forest-patch point counts were completed over two field seasons between June 3 to June 25, 2006, and May 21 to June 27, 2007. Four recordings were completed in each of the 44 sites (hence, a total of 176 observation occurrences), two during each field season. For 2006, acoustic-recording point counts were completed using a combination of two CZM-180 microphones (creating a stereo recording) from River Forks Research Corporation (http://rfrc1998.sasktelwebhosting.com) and a Marantz PMD 660 Solid-State digital recorder. Depending on the observer, the recording point counts were, on average: 91.4% or 83.2% as effective as traditional point counts for assessing species richness; and, 71.8% or 66.2% similar when assessing species and abundance simultaneously (Chapter 3). Two acoustic recordings were completed at each site in 2006, one between June 3rd and June 13th with the second between June 14th and June 25th. In 2007, skilled and experienced birders were available, and since traditional point counts proved more effective overall for sampling species in the study area, birders were the method of choice, and were employed for the 2007 surveys. Similar to 2006, two sampling sessions occurred in 2007, the first session was between May 21st and May 29th with the second round completed between June 19th and June 27th.

Each sample (including 2006 and 2007) began between 5:40 a.m. and 9:58 a.m. E.S.T. (sunrise ranged between 5:55 a.m. and 6:08 a.m.). The observations were divided into two time groups, 5:30-7:45 (Time A), and 7:45-10:00 (Time B), in an attempt to observe different species singing at different times of the morning over the breeding season. Although the sampling objective was to survey each plot during each time period within the breeding season, this was not always logistically possible. Specifically, in each year 33 sites were observed in both times (i.e., A and B), 10 sites were observed twice in time A, and one site was observed twice in time B. Sampling required nearly 400 miles of travel to survey each of the sites once, within the daily time constraints. The order of sampling was solely determined based on logistics. During the second round, sampling order was determined by a combination of logistical considerations and whether or not the site had been surveyed in the early time period during the first round.

During each point count, observers were instructed to document all birds heard or seen at the sample locations. Using modified Michigan Breeding Bird Atlas (MiBBA) field logs (Brewer et al. 1991) as a basis for record-keeping, the following variables were documented at each location: species, abundance, behavior (e.g., singing, calling, fly-over), and estimated distance and direction (in relation to North) from the observation location. Observation distance was considered unlimited, although observations were placed into three distance classes: 0-50m, 50-100m, and >100m. Since it has been suggested that even a single person moving through a bird's territory can influence singing for some species (Gutzwiller et al. 1994), each point count began after a 10-minute rest period (Martin et al. 1997), allowing time for normal singing to commence. While point counts documented all species, to match the audible-only detectability of bioacoustic sampling during 2006, data analysis of all final point counts was limited to birds that were observed audibly, and to birds that use woodland as primary or secondary habitat (Brewer et al. 1991).

Factors including temperature, humidity, topography, and wind speed (Simons et al. 2007), and also sounds associated with urbanization, are likely to attenuate or degrade bird song transmission; additionally, vocalization frequency (i.e., count) is species specific. Each of these factors influences detectability. Hence, there exists a reduced likelihood that any one survey will capture all species at a given study location, in a single sample. Therefore, I chose to use a single observation, from any of the four samples, as an indication that a site is being used by a given species. To assess the frequency of each species at each site, I calculated a commonness index, which is simply the percent of samples (out of four) where a species was observed at a single site.

Landscape Data

To characterize each of the 44 study locations, data were collected for an 800 m diameter landscape that was selected to correspond to the BBS sampling extent (Sauer et al. (2004); Figure 4.2). Landscapes were divided into two components, a focal patch (within which bird samples and vegetation characteristics were collected; described below), and the matrix (i.e., all areas exterior to the focal patch but within the 800 m extent). Tree cover was interpreted within each of the landscapes from 2005 high-resolution (0.15 m) color aerial photography. Interpretations were completed within ArcGIS 9.2 (ESRI, 2007) by digitizing tree-cover canopy boundaries at a scale of 1:2000 using a linear minimum mapping unit of approximately 5 m. Working at this scale allowed for the delineation of both large woodlots and individual trees, which were separated from the woodlots or interspersed among built structures. Tree-cover interpretations were subset based on the two levels of analysis. Patch tree-cover area described the total area of the focal patch, while matrix tree-cover area included all tree cover for the entire 800 m landscape, but excluded the area within the focal patch. All final tree cover delineations were converted to 1 m rasters for landscape pattern analysis.

Landscape spatial pattern metrics were calculated for tree-cover patches using Fragstats 3.3 (McGarigal et al. 2002). Metrics were chosen to capture matrix-level horizontal landscape factors thought to be important to birds, including habitat area (Turner 1989; Burke and Nol 2000) and core habitat area (Burke and Nol 2000); the number of patches (Boulinier et al. 1998; Donovan and Flather 2002) and amount of edge (Burke and Nol 2000; Fahrig 2001), which are commonly used as surrogates for fragmentation; and, spatial isolation (Bellamy et al. 2003; Ahlering and Faaborg 2006). Specific forest, class-level metrics included total class area, total core area, and focal-patch area; the number of patches; and, mean proximity to the nearest neighbor of the same class.

In addition to land-cover metrics, matrix-level land use was determined for each landscape by interpreting built-structure densities. Structures (e.g., houses and commercial buildings) were identified on the photographs using a working scale of 1:3000. Overall built-structure density (i.e., density within each 800 m landscape) was used for sampling stratification (i.e., urban, suburban, exurban, rural, and natural). For comparing the effects of matrix development patterns on species richness, built-structure density for each landscape was calculated by dividing the total number of structures by the total matrix area.

Patch-level characteristics were determined by performing on-site vegetation surveys. One transect was completed for each survey location. Whether transects ran north-south or east-west was determined by the location of the access point for each individual forest patch, e.g., if the access point entered from the east then the transect was run east-west. The final length of each full transect was 200 m, but on several occasions

I was not able to complete a full transect (discussed further, below). At 10 m intervals along each transect, vertical vegetation structure was estimated. Similar to MacArthur and MacArthur (1961), three vertical categories were used for categorization including 0-1 m, 1-8 m, and > 8 m (MacArthur and MacArthur used 0-2', 2-25', and > 25'). Canopy closure was also recorded at each 10 m stop. For the instances where the canopy was either completely closed or open, it was recorded as such. In all intermediate cases, the canopy was estimated using a GRS Densiometer. I followed a winner take all rule in this case, where ≥ 50% of the densiometer viewable area needed to be covered to indicate canopy present, otherwise if < 50% of the densitometer was covered, the canopy was considered open or absent. Along each transect, all trees ≥ 3 cm diameter at breast height (dbh), that were within 1.5 m of the transect line, were measured and recorded as deciduous, coniferous, or snag.

As previously described, sample locations were established by pacing approximately 100 m in from the edge of a forest patch. After surveying the transects, it became obvious that not every location ended up a full 100 m in from the edge. In a number of cases (12 of 49) transects were shorter than 200 m (e.g., 3 sites 20 m shorter, the rest < 6 m shorter) with the last few meters crossing an open road or into a residential lawn. In one instance the transect line crossed an impassable river causing the transect to be shortened by 20 m. There were several cases where transects were offset from their original path to work around mid-forest, standing water wetlands (n = 2), or changed bearing, for a fenced property boundary (n = 2).

Statistical Analyses

Since the sites were selected based on strata of development density, and were generally spatially clustered, the bird counts might be spatially autocorrelated and thus violate the assumption of statistical independence (Smith 1994; Villard et al. 1999). Spatial autocorrelation was tested by calculating Moran's I index on residual values for site-based richness and for each of the 16 individual species (described below). For all models, the selected set of landscape descriptors (i.e., within-patch and matrix variables) captured the spatial variation within the samples and removed spatial autocorrelation (i.e., Moran's I near zero, with a $p \geq 0.10$) from the model residuals (Smith 1994; Pidgeon et al. 2007).

I tested all variables for normality by calculating the Kolmogorov–Smirnov goodness of fit test (SPSS 15.0, 2007). Where necessary (i.e., data were not normal) variables were transformed using either square-root (e.g., development density and deciduous tree cover percent) or arcsine square-root transforms (e.g., percent mid-story vegetation and percent over-story vegetation). To assess redundancy in the calculated vegetation and landscape metrics (Riitters et al. 1995), Pearson's correlation coefficients were calculated for all pairs of variables. Variable pairs with correlations ≥ 0.4 were evaluated, with one variable removed from consideration. After highly correlated variables were removed, the final variable set describing each focal patch included: tree-cover area; tree density; percent ground cover; percent mid-story vegetation; percent over-story vegetation; and, the proportion of a site that was deciduous tree cover.

An assessment of matrix variables showed that the majority of variables were highly correlated (Pearson's $r > 0.4$) with tree-cover area. Therefore, for further analysis, the calculated suite of spatial pattern metrics was reduced down to tree-cover area, alone. Aside from the high correlations with other variables, the use of this single variable to describe the influence of matrix tree cover on forest species richness is justified by previous research that has shown that area alone (as compared to more complex spatial configurations) is the most important aspect of habitat (Blake and Karr 1987; Tilghman 1987; McGarigal and McComb 1995; Moilanen and Hanski 1998; Trzcinski et al. 1999; Burke and Nol 2000; Crooks et al. 2004). Resulting matrix variables included: matrix tree-cover area, development density; and, interaction term of total tree-cover area and density index. Final variable sets (for both levels of analysis) were tested again for correlations and variance inflation factors (VIF) were examined to assess the possibility of collinearity among the landscape variables.

To quantify the relationships between forest patch characteristics and surrounding matrix content and species richness I performed stepwise-multiple regression. To describe the relationship between the landscape metrics and species occurrence, I performed backward-conditional logistic regressions. The multiple and logistic regressions were completed using a block (i.e. group) configuration. Block one contained only patch variables, while block two contained only matrix variables. Blocks were used to first determine if patch characteristics influenced species richness or occurrence, and then to discern the added influence of matrix variables. For completeness models including only matrix characteristics were also run. To determine if a model was viable and to assess the relative fit of the patch-only, matrix-only, and patch-by-matrix logistic models Hosmer-Lemeshow tests (H-L Test) were also calculated, which is “a goodness-of-fit test of the null hypothesis that the model adequately fits the data" (SPSS 15.0). If the H-L Test was not significant for a species, then the presence-absence of that species was considered indiscernible. For species models were the H-L Test was significant, I selected the model with the lowest -2 Log likelihood where all predictor variables were significant (at $\alpha \leq 0.10$). Finally, for logistic models where -2 Log likelihood was the lowest, but one-to-many of the predictor variables were not significant (at $\alpha \leq 0.10$), forced-entry logistic regressions were re-calculated with the non-significant variables excluded.

Results

Sample landscapes averaged 1.2 (± 1.7 [S.D.]) built structures/ha of land, and 27.6 (± 9.2) ha of tree-cover. Landscape-scale development-density increased from zero structures/ha for natural settings to 5.0 (± 1.5) structures/ha in urban environments, while total tree-cover area generally decreased with urbanization, from 37.5 (± 4.1) ha for natural, to 18.9 (± 3.6) ha for urban areas. The matrix for each sample landscape contained 7.3 (± 4.8) ha of tree-cover intermixed with 1.7 (± 2.3) built structures/ha of matrix (Table 4.1).

Focal forest patches averaged 20.3 (± 8.6) ha of tree cover. The patches contained 760.8 (± 213.1) trees/ha that were: 87.6% (± 11.1) deciduous; 3.0% (± 7.5) coniferous; and, 9.4% (± 7.6) snags (Table 4.2). Vertical structure sampling demonstrated that the patches were 88.2% (± 8.5) closed canopy and composed of 55.1% (± 17.4) ground cover; 80.6% (± 13.1) mid-story vegetation; and, 82.1% (± 18.4) high-story vegetation. In addition, there was an average of 1.3 (± 1.2) hiking trails within each patch.

Across all sites and sampling seasons a total of 43 forest-associated species accounting for 1,962 individuals were observed. Based upon migratory strategy, these 43 species included 21 Neotropical migrants, 19 residents, and three short-distance migrants (Table 4.3). Similarly, based upon habitat requirements, 11 species were interior or area sensitive, 13 were edge-preferring, and 19 were considered ubiquitous and commonly occur within both interior and edge habitats. Among the 43 species, five [American Crow (*Corvus brachyrhynchos*), Northern Cardinal, (*Cardinalis cardinalis*), Tufted Titmouse (*Baeolophus bicolor*), Eastern Wood-Pewee (*Contopus virens*), and Red-eyed Vireo (*Vireo olivaceus*)] were exceptionally common at most sites (Table 4.4).

Stepwise multiple regression of species richness on both patch and matrix characteristics, independently and simultaneously, suggests that focal patch characteristics alone described the largest portion of total species richness. While vegetation >8m explained less than 14% (adjusted $r^2 = 0.131$; $df_{1,43}$, F = 7.454, p = 0.009) of variance species richness, and focal-patch tree-cover area explained 22% (adjusted $r^2 = 0.220$; $df_{1,43}$, F = 13.112, p = 0.001). Combined, vegetation > 8 m (p = 0.044) and focal-patch tree-cover area (p = 0.004) explained nearly 28% (adjusted $r^2 = 0.277$; $df_{2,42}$, F = 9.245, p < 0.001) of the observed species richness.

Stepwise multiple regressions of Neotropical and resident species richness (short-distance migrants were not assessed because of low counts and non-normality) on both patch and matrix characteristics suggests that response to the measured variables is guild specific. Neotropical migrant richness, like overall richness, was positively associated with vegetation > 8 m (p = 0.048) and focal-patch tree-cover area (p = 0.001). In addition, Neotropical migrants were positively influenced by the amount of matrix tree-cover area surrounding each focal patch ($p < 0.100$). These three variables explained nearly 40% (adjusted $r^2 = 0.390$; $df_{2,43}$, $F = 10.170$, $p < 0.001$) of the variance in observed migrant-species richness. Resident species, however, were not related to tree-cover area. Instead resident species were best described by the percentage of deciduous tree cover at each site (adjusted $r^2 = 0.144$; $df_{1,43}$, $F = 8.216$, $p = 0.006$). Interior or area-sensitive species were particularly of interest in this study. Like overall Neotropical migrant richness, the interior-sensitive subset was significantly associated with both patch and matrix characteristics. The interior-sensitive, Neotropical migrant species were significantly associated with patch tree-cover area ($p < 0.001$) and matrix tree-cover area ($p < 0.001$). These two variables explained just under 50% (adjusted $r^2 = 0.497$; $df_{2,43}$, $F = 22.230$, $p < 0.001$) of the observed Neotropical interior, or area-sensitive migrant-species richness (Figure 4.3).

Logistic regressions of the 16 species where models could be fit indicated that both patch and matrix characteristics were important for species presence, but that the response is species specific. At $\alpha \leq 0.10$, each of the patch variables significantly contributed to the occurrence of at least one species, with each variable influencing a different number of species. The patch variables and the number of species they significantly influenced include: focal patch tree-cover area (8 species); tree density per hectare (3); vegetation <1 m (1); vegetation <1 m and <8 m (4); vegetation >8 m high (3); and, percent deciduous tree cover (3) (Table 4.5). For eight species (Downy Woodpecker [*Picoides pubescens*], Red-bellied Woodpecker [*Melanerpes carolinus*], Ruby-throated Hummingbird [*Archilochus colubris*], American Crow [*Corvus brachyrhynchos*], Scarlet Tanager [*Piranga olivacea*], Yellow-throated Vireo [*Vireo flavifrons*], Ovenbird [*Seiurus *aurocapilla*], and White-breasted Nuthatch [*Sitta carolinensis*]), patch characteristics alone best described presence-absence (Table 4.5).

For three species (Carolina Wren [*Thryothorus ludovicianus*], House Wren [*Troglodytes aedon*], and Veery [*Catharus fuscescens*]), matrix characteristics alone best described presence-absence (Table 4.5). Six of the 16 species (37.5%) were significantly influenced by the addition of matrix tree-cover area in the models, with all but one of the relationships being positive. Four species showed a significant relationship with structure density. Carolina Wren and House Wren were both positively influenced by the level of development density and Wood Thrush and Veery were negatively influenced. The interaction between matrix tree-cover area and development density was not significant for any species. Finally, a comparison of model fit for patch or matrix characteristics alone versus the addition of matrix characters, suggest that for five of the species modeled, the combination of patch and matrix metrics increased model fit (Table 4.5).

Discussion

Although several patch- and matrix-level factors are important to individual species and overall richness, forest-patch size was the predominate predictor of overall forest species richness. Therefore, my prediction that total forest species richness would be influenced by the surrounding matrix composition was not supported. Although on a species basis, including matrix tree-cover area, that is, including tree-cover areas external to the forest patch contributed to the occurrence of nearly 40% of the species modeled. For those species tied to the area of tree cover at a site (10/16 overall), only two were independently associated with the area of tree-cover in the matrix, all others were either related to tree cover at both scales (n = 4), or related to focal patch tree cover only (n = 4). In addition, the presence/absence of area-sensitive Neotropical migrant species were strongly influenced by focal-patch tree-cover area (+) and matrix tree-cover area (+). This relationship suggests that the amount of matrix tree cover surrounding woodlots, parks, and other preserved set-asides may play a critical role in supporting area-sensitive Neotropical migrant species in urbanizing environments. Hence, my prediction that surrounding matrix composition would influence species occurrence is supported for some forest species, especially Neotropical migrants. Therefore, conservation strategies for interior-sensitive birds, and Neotropical migrants specifically, must extend beyond the boundaries of preserved set-asides (Villard et al. 1995).

I found that total species richness ranged from 11 to 22 species across the 44 sampled forest patches, and, while a significant negative relationship existed between richness and matrix development density, this surrounding development explained only 7.3% of the richness variance among forest patches. Conversely, focal patch tree-cover area had a significant relationship with forest species richness. A weak relationship between patch richness and matrix density, coupled with a strong response between richness and focal tree-cover area for the same landscapes (many of which are moderately to heavily developed), supports the idea that built structures surrounding forest patches do not completely eliminate the conservation value of patches in developed settings. This idea is further bolstered by the fact that I regularly observed several forest-interior species in suburban and urban contexts and that Mortberg and Wallentinus (2000) observed seven Swedish red-listed (i.e., conservation concern) species breeding in natural areas near city centers. These findings suggest that decreases in forest-patch species richness may not linked to urbanization in the matrix per se, but to the loss of supporting tree-based habitat that can be (but is not always) associated with development, and are at least partially contrary to studies that suggest a direct negative influence of urbanization surrounding forest patches (Friesen et al. 1995; Rottenborn 1999; Rodewald 2003 citing unpublished data). The described contradiction supports the notion that development in the matrix surrounding forest patches should not be considered monotypically negative.

Of particular interest for the matrix influence comparison was the distribution of Neotropical migrants across the study sites because they are commonly thought to be heavily influenced by urbanization and habitat fragmentation (Friesen et al. 1995; Villard et al. 1995). While my results also find a relationship between Neotropical migrants and urbanization, the relationship was only significant when considered independently of the focal-patch tree-cover area, the proportion of tall trees, and amount of matrix tree cover surrounding a site. When all potentially contributing variables were considered simultaneously, the level of surrounding urbanization had no predictive value.

An additional variation in the overall richness and species-level regression models that I tested was altering the block structure so that matrix variables were considered first, and patch characteristics were included second. This alteration had no affect on the multiple regression and strengthens the argument that patch characteristics are the best predictors of overall species richness out of the suite of metrics assessed. At the species occurrence level, altering the block structure influenced the results for only one species, Wood Thrush (*Hylocichla mustelina*), and exposed an interesting relationship between patch and matrix characteristics. For this species, if patch variables were considered first, only one matrix characteristic (development density) was significant, but, if matrix characteristics were considered first, then two patch variables also contributed significantly to the overall logistic regression. While only a product of variable interactions in a statistical model, this relationship potentially suggests that, for some species, even when patch characteristics are appropriate for a species, if the matrix is not supportive in some way, then those birds may be absent from the patch altogether. The position that matrix factors may override within-patch characteristics for some species is supported by Saunders et al.(1991 Pgs. 18-19) who contend that "the dynamics of remnant areas are predominately driven by factors arising out of the surrounding landscape."

Notably, there are several caveats to this study. For example, site selection, mapping, and subsequent land cover interpretations were completed with data at several resolutions. Two potential issues arise out of the mix of data resolutions. First, because the smallest forest patches were just 3 ha in size, the accuracy of the GPS could have changed the survey location sufficiently enough that characteristics external to the actual study site were interpreted as a part of the focal area. Second, using higher resolution photography for tree-cover delineations (as compared to sampling design) exposed areas of the study sites that were not actually tree covered, but might have been a forest opening, leading to a situation where each focal patch was not 100% contiguous tree cover. Lastly, the high-resolution imagery, which spanned the extent of the study region, was collected over several months from leaf-off to nearly full-growth conditions. This inconsistency within the images increased interpretation difficulty and likely introduced error into the process.

In addition to mixed resolution issues, the results might have been improved with the inclusion of several additional factors in the study design. First, site selection could have focused more specifically on holding forest patch-area constant so I could directly compare the influence of matrix content on within-patch characteristics. Then, I could have accounted for age of patch and matrix tree-cover, since different bird species are known to be associated with varying seral stages. Lastly, I could have limited the analysis to sites that were largely focused on residential development, and excluded sites with commercial-like components. Commercial, institutional, or industrial developments can have much different density to impervious surface ratios, which could underestimate the influence of built-structure density on species richness or occurrence, at any given site. An additional point to consider is that species presence and therefore richness values were possibly underrepresented because I used bioacoustics for one of the sample years. The possibility that this uncertainty influenced the model results is reduced since I required only a single observation at a site for species inclusion into the statistical model.

Conclusions

Forest-patch area is the predominate contributor to a site's overall forest bird species richness, but the area comprised of trees in the surrounding matrix also influences the ability of the patches to support many forest-obligate species, especially Neotropical migrants. The relationship that I observed between overall richness and focal-patch tree-cover area, and Neotropical migrants and the addition of matrix tree-cover, suggests that decreases in species richness are not linked to urbanization surrounding the patch per se, but to the loss of tree-cover habitat that is typically associated with development.

Furthermore, Neotropical migrants that require forest interior habitat can be found in areas that are not always large, continuous habitats, but also in areas where the total-tree cover area at a site (a combination of patch and matrix tree-cover area) may serve as a surrogate habitat for these species. This relationship suggests that the amount of matrix tree cover surrounding woodlots, parks, and other preserved set-asides may play a critical role in supporting area-sensitive neo-tropical migrant species in urbanizing environments.

From an avian conservation and land-use planning perspective, this research supports a strategy that gives priority to creating large set-asides to provide for the needs of interior or area-sensitive species instead of focusing on species diversity alone (Robbins et al. 1989). Establishment of large preserves is not practical for all landscapes, especially those that are largely, or soon to be, urbanized. While it has been suggested that the conservation value of small woodlots in urban settings may be minimal (Stratford and Robinson 2005), the results of this study suggest that when including matrix habitats external to a focal patch, the habitats include species commonly thought to be in decline (Neotropical migrants). Therefore, in developed environments, several small set-asides, in conjunction with surrounding matrix habitats can provide a significant conservation value. While no park is an island, and as parks' decrease in size the matrix has increasingly more influence (Janzen 1983), it is important to recognize that development will continue in some form in perpetuity (Marzluff and Ewing 2001). Therefore, more effort should be directed toward understanding the supporting role that habitats in urbanized environments can provide (Rosenzweig 2003).

The conclusion that matrix habitats surrounding public set-asides positively influences the ability of those patches for supporting some species, suggests that habitats on private land, even in more densely developed areas, should be considered in conservation planning. These results suggest that it is imperative for land-use planners to evaluate the affects of surrounding landscapes for potential ecological consequences or benefits of any given proposed development. Even better, if planners evaluated the potential effects of each given proposed development as part of a regional conservation strategy, the life history needs of all species of forest avifauna can be met as urbanization continues its advance (Fitzgerald et al. 2005).

Table 4.1. Average metric values (± standard deviation) for each built-structure density class and total for landscapes, and matrices. Non-transformed values are presented including: total sample area built-structure density (Units/ha); total tree-cover area (Total Tree Area); the number of patches (NP); total core area (TCA); mean proximity to the nearest neighbor of the same class (PROX_MN); total matrix-only tree-cover area (Matrix_Tcov); and, matrix sample area built-structure density (Matrix Density).

	Landscape Variables					Matrix Variables	
Class	**Units/ha**	**Total Tree Area (Ha)**	**NP**	**TCA (Ha)**	**PROX_MN**	**Matrix_Tcov**	**Matrix Density**
Natural	0.0 (± 0.0)	37.5 (± 4.1)	18.1 (± 16.4)	3.7 (± 2.6)	2485.2 (± 1438.0)	11.5 (± 4.5)	0.0 (± 0.0)
Rural	0.1 (± 0.1)	31.1 (± 9.5)	25.9 (± 17.1)	2.8 (± 3.4)	1426.1 (± 1355.4)	6.5 (± 5.2)	0.3 (± 0.4)
Exurban	0.5 (± 0.2)	24.6 (± 8.2)	81.4 (± 39.0)	2.6 (± 2.8)	1022.2 (± 1150.4)	6.1 (± 5.1)	0.8 (± 0.4)
Suburban	2.0 (± 0.6)	21.6 (± 5.1)	109.9 (± 57.6)	1.5 (± 1.8)	383.5 (± 422.3)	6.4 (± 3.8)	3.2 (± 1.4)
Urban	5.0 (± 1.5)	18.9 (± 3.6)	259.0 (± 79.0)	0.7 (± 0.7)	235.1 (± 133.8)	6.9 (± 2.4)	6.5 (± 1.8)
All	1.2 (± 1.7)	27.6 (± 9.2)	80.4 (± 84.0)	2.6 (± 2.8)	1182.7 (± 1289.7)	7.3 (± 4.8)	1.7 (± 2.3)

Table 4.2. Average values (± standard deviation) for patch variables for each built-structure density class and total for all sites. Non-transformed values presented include: focal-patch tree-cover area (Focal-patch Area); total tree density (Trees/ha); the proportion of a site that was deciduous tree cover (%Decid); diameter at breast height (Decid DBH); the percent of conifer trees, snags, and closed canopy (%Conifer, %Snag, and %Canopy, respectfully); percent ground cover (%Veg<1m); percent mid-story vegetation (%Veg1-8m); percent over-story vegetation (%Veg>8m); and the number of recreational trails counted at each site (#Trails).

Site Class	Focal-patch Area (ha)	Trees/ha	%Decid	Decid DBH	%Conifer	%Snag	%Canopy	%Veg < 1m	%Veg 1-8m	%Veg > 8m	# trails
Natural	26.0 (± 3.0)	693.5 (± 154.9)	86.4 (± 9.9)	18.8 (± 2.9)	2.3 (± 2.8)	11.3 (± 9.5)	91.6 (± 8.3)	54.9 (± 13.9)	78.3 (± 7.2)	92.7 (± 4.4)	1.6 (± 1.3)
Rural	24.9 (± 9.4)	821.7 (± 257.4)	84.7 (± 15.0)	18.8 (± 4.7)	5.4 (± 12.6)	9.8 (± 5.9)	83.7 (± 8.9)	55.3 (± 16.3)	80.1 (± 17.3)	79.4 (± 18.6)	0.4 (± 0.8)
Exurban	18.5 (± 6.1)	754.2 (± 116.9)	93.1 (± 4.4)	19.0 (± 1.7)	0.7 (± 2.0)	6.3 (± 3.9)	92.0 (± 7.9)	56.8 (± 17.8)	79.5 (± 11.6)	92.0 (± 12.8)	1.2 (± 1.1)
Suburban	14.9 (± 7.9)	779.0 (± 282.7)	84.6 (± 12.0)	17.5 (± 4.6)	3.3 (± 6.1)	12.2 (± 10.9)	83.3 (± 5.8)	54.4 (± 18.9)	78.7 (± 15.3)	64.8 (± 23.9)	1.6 (± 1.1)
Urban	11.8 (± 4.8)	773.3 (± 151.7)	90.5 (± 8.8)	18.3 (± 1.0)	2.7 (± 6.1)	6.8 (± 5.7)	95.2 (± 3.4)	47.6 (± 23.6)	90.5 (± 7.5)	84.8 (± 5.2)	2.4 (± 0.9)
All	20.3 (± 8.6)	760.8 (± 213.1)	87.6 (± 11.2)	18.8 (± 3.8)	3.0 (± 7.5)	9.4 (± 7.6)	88.2 (± 8.5)	55.1 (± 17.4)	80.6 (± 13.1)	82.1 (±18.4)	1.3 (± 1.2)

Table 4.3. All the species observed in this study, listed taxonomically by AOU code. Also reported is each species primary, and if present, secondary habitat; migration status; and whether each species is considered interior (or area) sensitive (I), has a preference for edges (E), or is ubiquitous (U), and commonly occurs in both interior and edge habitats.

Code	**Common Name**	**Species Name**	***Habitat**	****Migration**	****Int. vs Edge**
WITU	Wild Turkey	*Meleagris gallopavo*	Wood	Resident	U
COHA	Cooper's Hawk	*Accipiter cooperii*	Wood/Open	Resident	U
RTHA	Red-tailed Hawk	*Buteo jamaicensis*	Wood/Open	Resident	U
RSHA	Red-shouldered Hawk	*Buteo lineatus*	Wetland/Wood	Resident	I
HAWO	Hairy Woodpecker	*Picoides villosus*	Wood	Resident	U
DOWO	Downy Woodpecker	*Picoides pubescens*	Wood	Resident	U
PIWO	Pileated Woodpecker	*Dryocopus pileatus*	Wood	Resident	U
RBWO	Red-bellied Woodpecker	*Melanerpes carolinus*	Wood	Resident	U
NOFL	Northern Flicker	*Colaptes auratus*	Wood/Open	Resident	E
RTHU	Ruby-throated Hummingbird	*Archilochus colubris*	Wood	Neo-tropical	U
GCFL	Great Crested Flycatcher	*Myiarchus crinitus*	Wood	Neo-tropical	E
EAWP	Eastern Wood-Pewee	*Contopus virens*	Wood	Neo-tropical	E
ACFL	Acadian Flycatcher	*Empidonax virescens*	Wood	Neo-tropical	I
LEFL	Least Flycatcher	*Empidonax minimus*	Open/Wood	Neo-tropical	U
BLJA	Blue Jay	*Cyanocitta cristata*	Wood/Urban	Resident	E
AMCR	American Crow	*Corvus brachyrhynchos*	Open/Wood	Resident	E
BAOR	Baltimore Oriole	*Icterus galbula*	Wood/Open	Neo-tropical	E
EATO	Eastern Towhee	*Pipilo erythrophthalmus*	Wood/Shrub	Resident	E
NOCA	Northern Cardinal	*Cardinalis cardinalis*	Wood/Shrub	Resident	U
RBGR	Rose-breasted Grosbeak	*Pheucticus ludovicianus*	Wood	Neo-tropical	E
SCTA	Scarlet Tanager	*Piranga olivacea*	Wood	Neo-tropical	I
REVI	Red-eyed Vireo	*Vireo olivaceus*	Wood	Neo-tropical	U
YTVI	Yellow-throated Vireo	*Vireo flavifrons*	Wood/Wetland	Neo-tropical	E
NOPA	Northern Parula	*Parula americana*	Wood	Neo-tropical	I
BTBW	Black-throated Blue Warbler	*Dendroica caerulescens*	Wood	Neo-tropical	I
CERW	Cerulean Warbler	*Dendroica cerulea*	Wood	Neo-tropical	I
BTNW	Black-throated Green Warbler	*Dendroica virens*	Wood	Neo-tropical	U
PIWA	Pine Warbler	*Dendroica pinus*	Wood	Short-distance	I
OVEN	Ovenbird	*Seiurus aurocapilla*	Wood	Neo-tropical	I
HOWA	Hooded Warbler	*Wilsonia citrina*	Wood	Neo-tropical	I
AMRE	American Redstart	*Setophaga ruticilla*	Wood	Neo-tropical	I
CARW	Carolina Wren	*Thryothorus ludovicianus*	Wood/Shrub	Resident	U
HOWR	House Wren	*Troglodytes aedon*	Wood/Shrub	Neo-tropical	E
WBNU	White-breasted Nuthatch	*Sitta carolinensis*	Wood	Resident	E
TUTI	Tufted Titmouse	*Baeolophus bicolor*	Wood	Resident	U
BCCH	Black-capped Chickadee	*Poecile atricapillus*	Wood	Resident	U
GCKI	Golden-crowned Kinglet	*Regulus satrapa*	Wood	Short-distance	U
BGGN	Blue-gray Gnatcatcher	*Polioptila caerulea*	Wood	Neo-tropical	U
WOTH	Wood Thrush	*Hylocichla mustelina*	Wood	Neo-tropical	U
VEER	Veery	*Catharus fuscescens*	Wood	Neo-tropical	I
HETH	Hermit Thrush	*Catharus guttatus*	Wood	Short-distance	U
AMRO	American Robin	*Turdus migratorius*	Urban/Wood	Resident	E
EABL	Eastern Bluebird	*Sialia sialis*	Open/Wood	Resident	E

*Source: Brewer et al. 1991 (i.e., Michigan Breeding Bird Atlas I)

**Source: Birds of North America online series (http://bna.birds.cornell.edu/bna)

Table 4.4. All the species observed in this study by built-structure density class, listed taxonomically. The percentage of sites in each class where a species was observed at least once during the four surveys is indicated by the number of pluses (+): blank = no observations; 1 to 24% of the sites (+); 25 to 49% (++); 50 to 74% (+++); 75 to 99% (++++); and, 100% (+++++). Species observations that are outlined (hollow box), are those species with relatively high site fidelity. These species were observed at the same site for, on average, > 50% of the sites visits.

Species	Natural	Rural	Exurban	Suburban	Urban
Wild Turkey	++				
Cooper's Hawk		+			++
Red-tailed Hawk			+	+	
Red-shouldered Hawk				+	
Hairy Woodpecker	+	+		++	
Downy Woodpecker	+++	++++	+++++	++++	++++
Pileated Woodpecker		+			
Red-bellied Woodpecker	+++++	+++++	$\boxed{+++++}$	+++	$\boxed{+++}$
Northern Flicker	+	++	+	+	
Ruby-throated Hummingbird	++	+++	+++	++	+
Great Crested Flycatcher	++	++	++++	+++	+++
Eastern Wood-Pewee	$\boxed{+++++}$	$\boxed{+++++}$	$\boxed{+++++}$	$\boxed{++++}$	$\boxed{++++}$
Acadian Flycatcher	$\boxed{+++}$	++	+	+	
Least Flycatcher				+	
Blue Jay	++++	++++	++++	$\boxed{++++}$	$\boxed{++++}$
American Crow	$\boxed{+++++}$	$\boxed{+++}$	$\boxed{++++}$	$\boxed{+++}$	$\boxed{+++++}$
Baltimore Oriole	+	+	+	+	+
Eastern Towhee	+++	++		+++	
Northern Cardinal	$\boxed{+++++}$	$\boxed{+++++}$	$\boxed{+++++}$	$\boxed{+++++}$	$\boxed{+++++}$
Rose-breasted Grosbeak	++	++++	+++	++	+++
Scarlet Tanager	++++	+++	++++	+++	+
Red-eyed Vireo	$\boxed{+++++}$	$\boxed{+++++}$	$\boxed{+++++}$	$\boxed{+++++}$	$\boxed{+++++}$
Yellow-throated Vireo	+++++	+++	+++	++	+++
Northern Parula		+	+		
Black-throated Blue Warbler	++	+			
Cerulean Warbler		+			
Black-throated Green Warbler	+	+	+		
Pine Warbler			+		
Ovenbird	$\boxed{++++}$	$\boxed{+++}$	+	+	++
Hooded Warbler			+		
American Redstart	+++	++	+	+	++
Carolina Wren		+		+	+++
House Wren		+	+	++	+++
White-breasted Nuthatch	++++	++++	$\boxed{++++}$	++++	++++
Tufted Titmouse	$\boxed{+++++}$	$\boxed{++++}$	$\boxed{+++++}$	$\boxed{++++}$	$\boxed{+++++}$
Black-capped Chickadee	++++	++++	+++++	++++	+++++
Golden-crowned Kinglet		+			
Blue-gray Gnatcatcher	++	+++	+++	+++	++
Wood Thrush	++++	+++	+++	$\boxed{+++}$	
Veery	+++	+			
Hermit Thrush	+		++		
American Robin	+	++++	$\boxed{+++}$	++++	++++
Eastern Bluebird	+		+		

Table 4.5. Logistic regression results for the 16 species where models could be fit. Logistic regressions were completed for patch-only, matrix-only, and patch x matrix configurations for each species. The best-fit model, where all predictor variables were significant (at $\alpha \leq 0.10$), are presented in bold. Model fit results (Table 4.5a) and predictor-variable associations (Table 4.5b) can be linked using the ID number at the far left of the table. Refer to Table 4.3 for species codes.

				Final-Model Fit								
ID	Species	Model setup	# sites	Chi-square	df	Model p-value	-2 log likelihood	Nagelkerke R2	Hosmer-Lemeshow	Predicted 0	Predicted 1	%Correct
1	ACFL	Patch Only	13	13.061	3	0.005	40.351	0.365	0.820	90.3	46.2	77.3
2	ACFL	Patch in Block 1	13	17.811	4	0.001	35.602	0.474	0.453	93.5	69.2	86.4
3	**ACFL**	**w/ Sig. Var. Only**	**13**	**15.614**	**3**	**0.001**	**37.799**	**0.425**	**0.546**	**90.3**	**61.5**	**81.8**
4	ACFL	Matrix Only	13	5.990	1	0.014	47.423	0.181	0.315	90.3	38.5	75.0
5	**AMCR**	**Patch Only**	**35**	**4.661**	**1**	**0.031**	**39.923**	**0.158**	**0.821**	**22.2**	**100.0**	**84.1**
6	AMCR	Patch in Block 1	35	4.661	1	0.031	39.923	0.158	0.821	22.2	100.0	84.1
7	AMCR	Matrix Only	35	--	--	--	--	--	--	--	--	--
8	AMRO	Patch Only	29	17.464	5	0.004	39.000	0.453	0.242	60.0	82.8	75.0
9	AMRO	Patch in Block 1	29	20.552	6	0.002	35.912	0.516	0.145	60.0	89.7	79.5
10	**AMRO**	**w/ Sig. Var. Only**	**29**	**17.954**	**5**	**0.003**	**38.510**	**0.463**	**0.386**	**66.7**	**89.7**	**81.8**
11	AMRO	Matrix Only	29	--	--	--	--	--	--	--	--	--
12	AMRE	Patch Only	12	--	--	--	--	--	--	--	--	--
13	**AMRE**	**Patch in Block 1**	12	**8.604**	**2**	**0.014**	**42.963**	**0.257**	**0.273**	**93.8**	**33.3**	**77.3**
14	AMRE	Matrix Only	12	4.381	1	0.036	47.183	0.137	0.563	93.8	25.0	75.0
15	CARW	Patch Only	5	5.264	1	0.022	25.893	0.222	0.883	100.0	0.0	88.6
16	CARW	Patch in Block 1	5	8.616	2	0.013	22.541	0.335	0.477	100.0	40.0	93.2
17	**CARW**	**Matrix Only**	**5**	**9.804**	**2**	**0.007**	**21.352**	**0.394**	**0.060**	**100.0**	**40.0**	**93.2**
18	**DOWO**	**Patch Only**	**34**	**3.293**	**1**	**0.070**	**43.872**	**0.110**	**0.453**	**10.0**	**100.0**	**79.5**
19	DOWO	Patch in Block 1	34	3.293	1	0.070	43.872	0.110	0.453	10.0	100.0	79.5

20	DOWO	Matrix Only	34	--	--	--	--	--	--	--	--	--
21	EATO	Patch Only	13	11.392	3	0.010	42.021	0.324	0.480	90.3	46.2	77.3
22	**EATO**	**Patch in Block 1**	**13**	**14.939**	**4**	**0.005**	**38.474**	**0.410**	**0.384**	**90.3**	**53.8**	**79.5**
23	EATO	Matrix Only	13	--	--	--	--	--	--	--	--	--
24	HOWR	Patch Only	10	--	--	--	--	--	--	--	--	--
25	HOWR	Patch in Block 1	10	9.163	1	0.002	38.002	0.286	0.508	91.2	50.0	81.8
26	**HOWR**	**Matrix Only**	**10**	**9.163**	**1**	**0.002**	**38.002**	**0.286**	**0.508**	**91.2**	**50.0**	**81.8**
27	OVEN	Patch Only	18	13.437	2	0.001	46.097	0.355	0.730	69.2	72.2	70.5
28	**OVEN**	**w/ Sig. Var. Only**	**18**	**10.493**	**1**	**0.001**	**49.041**	**0.286**	**0.373**	**76.9**	**61.1**	**70.5**
29	OVEN	Patch in Block 1	18	13.437	2	0.001	46.097	0.355	0.730	69.2	72.2	70.5
30	OVEN	Matrix Only	18	--	--	--	--	--	--	--	--	--
31	**RBWO**	**Patch Only**	**38**	**15.565**	**3**	**0.001**	**19.486**	**0.543**	**0.509**	**50.0**	**100.0**	**93.2**
32	RBWO	Patch in Block 1	38	19.992	4	0.001	15.059	0.665	0.999	50.0	97.4	90.9
33	RBWO	Matrix Only	38	8.658	1	0.003	26.393	0.325	0.404	16.7	97.4	86.4
34	RTHU	Patch Only	21	20.314	5	0.001	40.592	0.493	0.576	73.9	76.2	75.0
35	**RTHU**	**w/ Sig. Var. Only**	**21**	**13.156**	**2**	**0.001**	**47.750**	**0.345**	**0.838**	**65.2**	**76.2**	**70.5**
36	RTHU	Patch in Block 1	21	20.314	5	0.001	40.592	0.493	0.576	73.9	76.2	75.0
37	RTHU	Matrix Only	21	--	--	--	--	--	--	--	--	--
38	**SCTA**	**Patch Only**	**29**	**9.838**	**1**	**0.002**	**46.626**	**0.277**	**0.062**	**53.3**	**89.7**	**77.3**
39	SCTA	Patch in Block 1	29	9.838	1	0.002	46.626	0.277	0.062	53.3	89.7	77.3
40	SCTA	Matrix Only	29	4.596	1	0.032	51.868	0.137	0.104	40.0	89.7	72.7
41	VEER	Patch Only	6	4.626	2	0.099	30.425	0.182	0.235	100.0	0.0	86.4
42	VEER	Patch in Block 1	6	15.101	4	0.004	19.950	0.529	0.842	97.4	50.0	90.9
43	**VEER**	**Matrix Only**	**6**	**14.218**	**2**	**0.001**	**20.833**	**0.503**	**0.453**	**94.7**	**33.3**	**96.4**
44	WBNU	Patch Only	35	11.602	3	0.009	32.982	0.364	0.610	33.3	97.1	84.1

45	**WBNU**	**w/ Sig. Var. Only**	**35**	**4.683**	**1**	**0.030**	**39.901**	**0.159**	**0.423**	**11.1**	**97.1**	**79.5**
46	WBNU	Patch in Block 1	35	12.315	3	0.006	32.269	0.383	0.124	33.3	91.4	79.5
47	WBNU	Matrix Only	35	--	--	--	--	--	--	--	--	--
48	WOTH	Patch Only	23	--	--	--	--	--	--	--	--	--
49	WOTH	Patch in Block 1	23	8.669	1	0.003	52.237	0.239	0.290	57.1	78.3	68.2
50	WOTH	Matrix Only	23	8.669	1	0.003	52.237	0.239	0.290	57.1	78.3	68.2
51	**WOTH**	**Matrix in Block 1**	**23**	**19.687**	**3**	**< 0.000**	**41.219**	**0.481**	**0.069**	**71.4**	**78.3**	**75.0**
52	**YTVI**	**Patch Only**	**27**	**3.668**	**1**	**0.055**	**55.036**	**0.109**	**0.516**	**29.4**	**88.9**	**65.9**
53	YTVI	Patch in Block 1	27	3.668	1	0.055	55.036	0.109	0.516	29.4	88.9	65.9
54	YTVI	Matrix Only	27	--	--	--	--	--	--	--	--	--

Table 4.5b (Table 4.5a continued). The gray boxes indicate predictor-variables not included in a particular regression

	Within-Patch Variables ((direction) p-value)						Matrix Variables ((direction) p-value)		
ID	Patch Tcov	Trees_HA	Veg<1m	Veg18m_ AS_SQRT	Veg>8m_ AS_SQRT	Decidp_ SQRT	Matrix Tcov	Matrix Density	Tcov x Density
1	(-) p = 0.050	(-) p = 0.069	--	--	--	(-) p = 0.018	▨	▨	▨
2	(-) p = 0.035	(-) p = 0.166	--	--	(+) p = 0.156	(-) p = 0.028	(+) p = 0.037	--	--
3	**(-) p = 0.029**	**--**	--	--	--	**(-) p = 0.041**	**(+) p = 0.019**	--	--
4	▨	▨	▨	▨	▨	▨	--	(-) p = 0.035	--
5	--	**(-) p = 0.052**	--	--	--	--	▨	▨	▨
6	--	(-) p = 0.052	--	--	--	--	--	--	--
7	▨	▨	▨	▨	▨	▨	--	--	--
8	(-) p = 0.077	(+) p = 0.024	--	(-) p = 0.048	(-) p = 0.097	(+) p = 0.031	▨	▨	▨
9	(-) p = 0.043	(+) p = 0.025	--	(-) p = 0.043	(-) p = 0.127	(+) p = 0.045	(-) p = 0.098	--	--
10	**(-) p = 0.021**	**(+) p = 0.037**	--	**(-) p = 0.100**	--	**(+) p = 0.100**	**(-) p = 0.074**	--	--

11							--	--	--
12	--	--	--	--	--	--			
13	**(+) p = 0.052**	--	--	--	--	--	**(+) p = 0.024**	--	--
14							(+) p = 0.044	--	--
15	(-) p = 0.041	--	--	--	--	--			
16	(-) p = 0.161	--	--	--	--	--	--	(+) p = 0.087	--
17							**(+) p = 0.080**	**(+) p = 0.017**	--
18	--	--	--	--	--	**(+) p = 0.079**			
19	--	--	--	--	--	(+) p = 0.080	--	--	--
20							--	--	--
21	(+) p = 0.058	--	--	(-) p = 0.055	(-) p = 0.016	--			
22	**(+) p = 0.058**	--	--	**(-) p = 0.046**	**(-) p = 0.011**	--	**(+) p = 0.074**	--	--
23							--	--	--
24	--	--	--	--	--	--			
25	--	--	--	--	--	--	--	(+) p = 0.007	--
26							--	**(+) p = 0.007**	--
27	(+) p = 0.017	--	--	--	(+) p = 0.114	--			
28	**(+) p = 0.008**	--	--	--	--	--			
29	(+) p = 0.017	--	--	--	(+) p = 0.114	--	--	--	--
30							--	--	--
31	**(+) p = 0.019**	--	**(-) p = 0.084**	**(-) p = 0.050**	--	--			
32	(+) p = 0.098	--	(-) p = 0.157	(-) p = 0.148	--	--	--	(-) p = 0.131	--
33							--	(-) p = 0.011	--
34	(+) p = 0.034	(+) p = 0.116	(+) p = 0.051	(-) p = 0.020	--	(+) p = 0.120			
35	**(+) p = 0.033**	--	--	**(-) p = 0.018**	--	--			

36	(+) p = 0.034	(+) p = 0.116	(+) p = 0.051	(-) p = 0.020	--	(+) p = 0.120	--	--	--
37							--	--	--
38	**(+) p = 0.006**	--	--	--	--	--			
39	(+) p = 0.006	--	--	--	--	--	--	--	--
40							--	(-) p = 0.040	--
41	--	--	--	--	(+) p = 0.086	(-) p = 0.115			
42	--	--	--	--	(+) p = 0.568	(-) p = 0.383	(+) p = 0.120	(-) p = 0.155	--
43							**(+) p = 0.076**	**(-) p = 0.090**	--
44	(+) p = 0.183	(-) p = 0.071	--	--	(+) p = 0.106	--			
45	--	**(-) p = 0.052**	--	--	--	--			
46	--	(-) p = 0.047	--	--	(+) p = 0.026	--	(+) p = 0.111	--	--
47							--	--	--
48	--	--	--	--	--	--			
49	--	--	--	--	--	--	--	(-) p = 0.009	--
50							--	(-) p = 0.009	--
51	--	--	--	--	**(-) p = 0.010**	**(+) p = 0.022**	--	**(-) p = 0.002**	--
52	--	--	--	--	**(+) p = 0.071**	--			
53	--	--	--	--	(+) p = 0.071	--	--	--	--
54							--	--	--

Figure 4.1. The study area is located in Southeast Michigan in Washtenaw County. The sample locations (dots in county inset) are focused on patches of publicly accessible woodlands. The sites are distributed along a diagonal from SE to NW, extending from the more populated Ann Arbor area to the rural northwest.

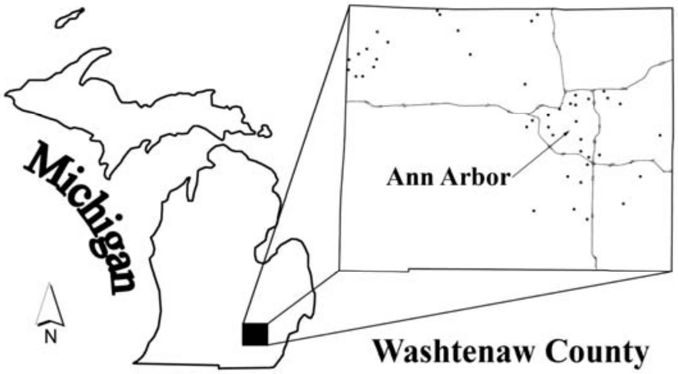
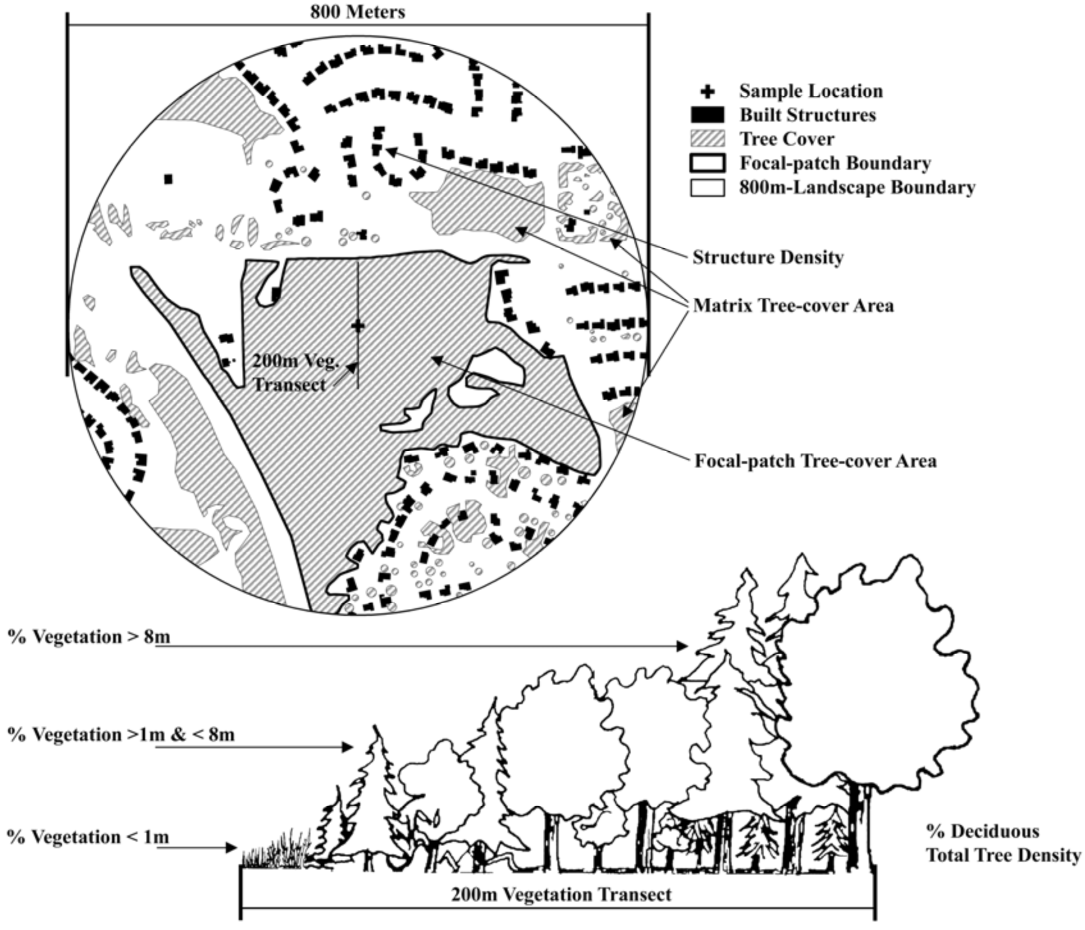
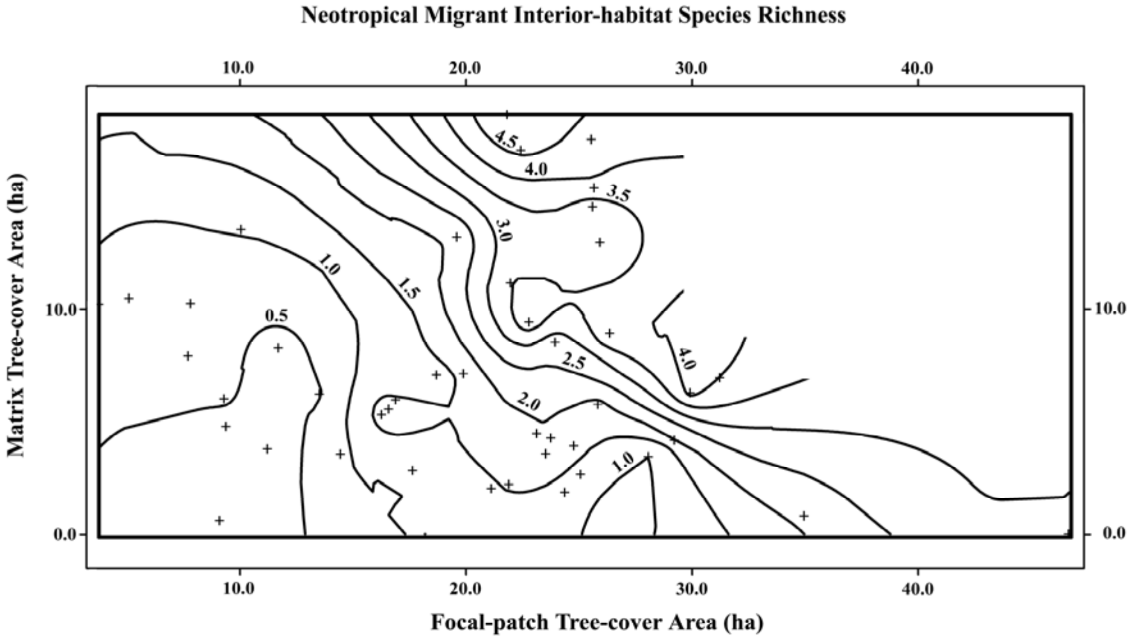


Figure 4.2. Landscapes were evaluated at two levels. At the patch level, both horizontal and vertical metrics were assessed. The final variable-set that was entered into the regression models included focal-patch tree-cover area; percent deciduous tree-cover; total tree density; and three vertical vegetation classes: <1m, between 1 and 8m, and >8m. At the matrix scale, only horizontal matrix characteristics were measured. The final variable-set included total matrix tree-cover area and built-structure density.



Succesion gradient adapted from http://www.dnr.state.mi.us/publications/.../Landowners_Guide/.../forest_succession.gif

Figure 4.3. A contour-surface model was calculated to visualize the multiple regression results for significant predictors of Neotropical migrant, interior forest-species, richness. Focal-patch tree-cover area and matrix tree-cover area explain 50% of the aforementioned species richness. Pluses (+) indicate the combination of focal-patch and matrix tree-cover for each of the 44 sample locations. Contour lines designate richness values at 0.5 increments.



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CHAPTER 5

CONCLUSIONS

Summary

The pages in this dissertation describe the: (1) static-relationships and change-relationships between breeding birds and associated landscapes of Southeast Michigan; (2) effectiveness of a remote sensing technique (bioacoustics recording) for monitoring forest birds in urbanized environments; and, (3) influence of matrix characteristics on the ability of forest set-aside patches to support forest-obligate birds. Each of the chapters answers a question, or questions, related to birds in an urbanizing world.

In Chapter 2, I asked the question: how do amounts and changes in the amounts of land cover affect avian communities at the urban-rural interface, at the local scale in Southeast Michigan? In addressing this question I explored the effects of the major land-cover changes in the region: agricultural abandonment, increased urbanization, and afforestation. Some bird guilds (e.g., grassland birds) have suffered significant declines associated with habitat loss while other guilds (e.g., woodland birds) have not responded to marked increases in habitat. The finding that forest birds were not increasing in concert with increased tree cover inspired a more detailed analysis of how relatively expansive urbanization is affecting the ability of remnant and preserved habitats to support forest birds. To perform this more detailed analysis required a shift in both landscape and subject scale, and also required the adoption of a method to collected avian point counts in the absence of seasoned birders.

Chapter 3 is a minor digression from the main storyline, i.e., birds in an urbanizing world, in order to evaluate the effectiveness of proven bioacoustics equipment in an unproven environment. Results from Chapter 3 adds a technique for studying birds in urbanized environments, and therefore makes a contribution to the larger avian sampling literature while providing an evaluation of the method by which samples were collected for Chapter 4. In comparing the effectiveness of CZM bioacoustic recordings versus traditional point counts, I found that the interpretations based on the recordings, though not in perfect agreement with field-based observations, can serve as an effective point-count mechanism in an urbanized environment. The recording-based interpretations described, on average, better than 90% of the species richness and better than 80% of abundance compared with traditional field observations, and provide a nearly 70% match when accounting for richness and abundance simultaneously (via the percent similarity index). The analysis suggests that method-comparison results are similar along the entire urbanization gradient, and, therefore, stereo-recordings for avian point-counts are equally effective at all levels of human influence found within our study area.

In Chapter 4, I used a multi-scale approach combining fieldwork and geospatial analysis to understand how a forest patch's location influences what avian species occur within the patch. In testing several hypotheses I found that focal-patch area is the primary indicator to a site's overall forest species richness, but that the addition of matrix tree-cover area influences the ability of the patches to hold many forest-obligate species, especially neo-tropical migrants. Specifically, nearly 50% of neo-tropical migrant richness can be explained by focal patch and matrix tree-cover area combined. These relationships suggest that (1) decreases in species richness are not linked to urbanization per se, but to the loss of habitat that is typically associated with development (i.e., no development density effect); and (2) that the amount of matrix tree cover surrounding woodlots, parks, and other preserved set-asides may play a critical role in supporting area-sensitive species in urbanizing environments.

Research Contributions

This dissertation research makes several contributions to the field of avian landscape ecology. Little is known about how human landscape change has influenced avian species over longer periods of time, outside of space-for-time substitution (SFTS) techniques. My goal (in Chapter 2) was to understand how both amounts and changes in land cover have affected avian communities at the urban-rural interface at the local scale. In addressing this goal, I found a substantial difference in results obtained from long-term ecological studies at single sites as compared to studies following the space-for-time substitution (SFTS) methodology. The ability to document how any single landscape is changing, along with how species populations are changing, is critical to understanding how landscape change is influencing habitat-based population dynamics. Therefore, SFTS techniques alone are seemingly limited in their ability to provide answers related to the functional dynamics of avian populations. This research calls for continued research conducted in a long-term manner.

Limiting factors in the study of avian populations include the availability of skilled observers, inconsistency among observers, and repeatable and verifiable samples. The use of bioacoustic recordings offers an opportunity to alleviate each of the aforementioned concerns. While the utility of recordings as a replacement for expert birding in remote forests had been confirmed previously, this research (Chapter 3) adds understanding that use of bioacoustic recordings is a valid method of sampling outside of remote locations, including areas considered urbanized. Furthermore, I found that recordings are subject to the same ambient noises, resulting in similar levels of distraction, associated with field-based observations. Based on these results, bird-habitat studies at the urban-rural interface could: (1) employ bioacoustic recordings in lieu of skilled observers; (2) alleviate concerns of inconsistency since all recorded observations could be interpreted by a single skilled observer; and (3) provide an opportunity for a study to be repeated or verified by accessing archived re-interpretable recordings, which would not be possible with tradition paper logs.

In studying the effects of urbanization on forest birds, my research (Chapter 4) supports the commonly held notion that total habitat area is the single most important factor for most avian species. My research also supports the more novel idea that similar-type habitats external to a focal patch offer an opportunity to simulate much larger habitat patches, for many species. Therefore, with proper land-use planning, several small set-asides in developed environments that are designed in conjunction with surrounding matrix habitats can provide a significant conservation value for many forest-obligate species.

Limitations and Weaknesses

While the use of BBS data (in Chapter 2) allows for a relatively long-term change-analysis, BBS data are limited for several reasons. First, since the BBS surveys are roadside surveys and detection distances are limited, and my landscapes are largely within an agricultural context, many kinds of species (e.g., wetland and forest) will necessarily be missed, or at least underrepresented, leading to an inaccurate count or trend estimation for the region. For example, in my study, it is difficult to discern bird-habitat relationships or changes with forest birds because they are not counted frequently enough. I suspect the low forest-bird counts result because BBS stops are not stratified by cover type (partially due roadside sampling), and therefore, in landscapes in an agricultural context, forests would be underrepresented, leading to a reduced forest-bird count. Another limitation with using BBS data in my study design was the lack of data for every site in each decade analyzed. This absence of data was partially because of gaps in BBS coverage and partially because the study landscapes were limited to BBS stops which intersected with previously selected townships (via Project SLUCE). Instead of being able to observe changes for single sites across all four decades (which was my intent), the study was limited to observing changes across single paired-decades over the study duration. While value remains in this approach, the analysis became 10-year comparisons across multiple decades, as compared to a single long-term study.

For the analysis presented in Chapter 2, there were not enough spatial or temporal samples to provide a robust statistical analysis; and, many of the bird guilds suffered from inadequate counts forcing a multi-year aggregated guild analysis without the ability to differentiate species or migration strategies. The inability to perform a species-level analysis is especially important based on the results in Chapter 4 - some species' occurrence is better described by focal-patch characteristics, while others are better described by landscape characteristics. Aggregating the species to guilds, while appropriate in this case, limits the resolution of the study, the potential of the results, and the ability to discern actual bird-habitat relationships. In the end, the study did achieve its goal of providing a general description of bird change and bird-habitat relationships that provided the development of new hypotheses for additional testing (in Chapter 4). Even so, it should be noted that without the inclusion of information about population demographics (e.g., fecundity and predation rates), questions of why or how habitat and habitat change is related to population numbers remain unanswered. This phenomenon is not unique to my study; most bird-habitat studies remain correlation based like mine. What we need are bird-habitat studies of individual demographics and population or community measures addressed simultaneously (Marzluff et al. 2001). In doing so, general population trends can be linked to both habitat changes related to urbanization, and to functional or causal dynamics.

There are several approaches that could have been pursued to strengthen Chapter 3. First, while multiple observers were used, any given sample and subsequent comparison was between a recording interpretation and a field observation by a single observer, therefore, observer differences potentially exist. Adding multiple observers simultaneously would have allowed for a comparison of each observer's results for the same site and sample, and provided the possibility for observer effects to be directly addressed. Another limitation in relation to single observer interpretations was the potential for misidentification of species. Each observer listened to, and interpreted, each recording only once; but if each recording had been listened to multiple times, or by multiple reviewers, correct species identification could be verified. This aspect had no effect on the method comparison, because field and recording observations were completed by the same observer. The possibility for species misidentification could present itself in Chapter 4, where half of the samples were based on recordings. If species interpretations were inaccurate, the calculated species habitat relationships could have been misrepresented; although, the same potential for misidentification remains with the traditional point counts (in this study and in general).

The most significant limitation to the recording comparison is that the two observers used different methods for recording interpretations (i.e., stereo versus mono playback). While the difference was potentially fruitful (a systematic decrease in ability was noted for the restricted observer), the inadvertent method change detracted from the proposed comparison. In effect, my reliable sample size was halved, and caused an analysis by individual instead of using a pooled sample (increasing n), with the observer as an explanatory variable. In addition, the ability to compare the influence of noise levels on recording effectiveness was hindered by a sound level meter with an inadequately low decibel-level range. Many sites were quieter than the meter could measure, therefore the range of ambient noise levels at each site was not fully represented. Like above, this aspect had no bearing on the test of the method effectiveness, but possibly influenced the test of an explanatory variable related to a primary hypothesis (i.e., urban noise will negatively affect the recordings).

Limiting factors for Chapter 4 result directly from the study design. I suspect the analysis would be more robust if I had included several additional factors. For example, by altering my study extent, site selection could have focused more specifically on holding forest patch-area constant so I could directly compare the influence of matrix content on within-patch characteristics. Then, I could have accounted for age of patch and matrix tree-cover, since different bird species are known to be associated with varying seral stages. Lastly, I could have limited the analysis to sites that were largely focused on residential development, and excluded sites with commercial-like components. Commercial, institutional, or industrial developments can have much different density to impervious surface ratios, which could underestimate the influence of structure density on species richness or occurrence, at any given site. An additional point to consider is that species presence, and therefore richness values, were possibly underrepresented because I used bioacoustics for one of the sample years. The recordings were better than 83% as effective as standard point counts when observing species richness alone, which leaves a nearly 17% margin of uncertainty in the first year of sampling. The possibility that this uncertainty influenced the model results is reduced since I required only a single observation at a site for species inclusion into the statistical model.

The analysis in Chapter 4 subscribed to the space-for-time substitution (SFTS) methodology (Pickett 1989). The SFTS approach is potentially limiting because it can hide subtle trends in data and also does not allow for the exploration of uncommon events (e.g., destructive weather) that may influence species-habitat relationships in any given year. By removing time as a variable, as I did in this chapter, factors that could be essential to understanding species dynamics may be avoided altogether, e.g., the historical persistence of any given species or the effects of landscape change at a single location through time. The ability to document how any single landscape is changing, along with how species are changing, is critical to understanding how landscape change is influencing habitat-based population dynamics (Schrott et al. 2005). Therefore, the results in this chapter are limited in their ability to provide answers related to the functional dynamics of systems, but are more appropriate for generating future testable hypothesis and questions (Pickett 1989).

Needed Research - Next Steps

In continuing to explore the responses of forest-birds within urbanizing environments, one logical next step, which is a derivative of my presented research and is largely lacking in the literature, is to develop an experimental design that incorporates residential development processes into the analysis. That is, to work with local governments, land-use planners, and developers to include pending residential developments into a study design. The idea is to create baseline ecological assessments of focal and surrounding parcels prior to development, and then monitor the response of flora and fauna to the disturbance as well as subsequent changes through time. Such a design would initially offer insight based on a space-for-time substitution technique and, as years passed, the accumulation of long-term data would allow for the incorporation of additional affects (e.g., individual and population demographics, climatic variability, time lags, and neighborhood effects) to provide species-level population trends resulting from functional bird-habitat relationships.

Ultimately, the goal is to develop a longitudinal study that maps and then monitors the combined effects of climate- and human-induced land-cover change on the fidelity of migration routes, stopover sites, and breeding and wintering locations. The technology for such a program exists today, but is limited by the weight of GPS-based telemetry transmitters mounted on individual birds because of mass restrictions (the transmitter must be less than 4% of body weight to limit behavioral modifications). Opportunities exist for collaboration with members of the geospatial engineering community to develop satellite-based tracking equipment that is light enough to fit on small birds allowing for the mapping of their winter-to-breeding-grounds connectivity. This research would: (1) continue a much needed effort to explain the ecological effects of human-induced terrestrial landscape change; (2) offer a validation mechanism for research focusing on contemporary migration-tracking technologies (e.g., Stable Isotopes, NEXRAD Radar); (3) afford those engaging in mark and recapture techniques (i.e., bird banding) a higher probability of locating individual migrant birds at opposite ends of their migration routes; and, (4) allow avian landscape ecologists to legitimately address the question of whether landscape changes on breeding or wintering grounds has more influence on population numbers of migratory species.

Finally, existing literature (Slabbekoorn and Peet 2003; Slabbekoorn and den Boer-Visser 2006) suggests that some birds behaviorally adapt (e.g., modification of song pitch) to maintain their fitness in urban environments. Therefore, along with continuing the theme of ecological effects of urbanization, an additional next step is to examine the behavioral responses (e.g., song alteration) that may allow for maintenance of fitness levels, and therefore permit typical interior-habitat species to exist in urbanized environments. Potential questions to investigate include, which birds, and at what level of urbanization is a behavioral modification experienced?

Implications and Comment

Together, these chapters suggest that human dominance, through landscape conversion at the urban-rural interface, changes not only the appearance of the landscape, but also the ecological functions that landscapes can support. Avian populations, being habitat specific, are directly influenced by land-use choices. From an avian conservation perspective then, in a region dominated by private ownership and human development, the goal should be to understand where the habitat thresholds are (including composition, configuration, and rates of change) so that urbanization can be guided down a more ecologically sensitive path. To accomplish this, additional long-term research to elucidate bird-habitat dynamics in urbanizing environments, following a gradient paradigm, is needed.

While additional long-term study is needed in urbanizing environments (as noted above), ecological studies at the urban-rural interface (i.e., the front of urbanization in many landscapes) can be less appealing because they: (1) do not focus on exotic species in far-away lands; (2) they do not focus on charismatic mega fauna; and (3), for generations ecologists have suggested that ecosystems should be studied in the absence of humans. For me, each of the aforementioned factors are less important than the fact that we have a system (comprised of landscapes at the urban-rural interface) that is currently under a major disturbance regime that is likely to continue indefinitely, and this is also one of the systems that we know the least about, ecologically speaking.

Like Rozenweig (2003), I believe that the coexistence of human settlements and natural landscape features at the same location is crucial. Natural features support wildlife, but they also support humans through essential ecosystem services, for example carbon fixation, oxygen production, hydrological flow regulation, prevention of soil erosion, food production, and recreation. Natural landscapes and the vital services they provide continue to be threatened by what urban ecologists call a “ratcheting effect;” that is, for each successive generation growing up with nature displaced, the idea of what is natural is negatively changing. If residential areas from the urban centers to rural developments continue to displace natural features, we will soon have an entire generation thinking that the lack of woodlots, fields, and wetlands is normal - a clear and significant detriment to the possibility of a sustainable future. We have to find a way for humans and natural features to co-exist. I believe a major key is the integration of science and policy. If we can provide science that helps policy-makers and residents alike to understand the affects of their land uses on ecological systems, then we have opened the door to one aspect of a sustainable future. My hope is that by studying birds in developing landscapes I can open this door.

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