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# Avian diversity in a suburban park system: current conditions and strategies for dealing with anticipated change

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**Abstract** A growing trend towards increased urbanization emphasizes the role of suburban parks in wildlife conservation. Spatial planning aimed at maintaining biological diversity and functionality must consider how changes at landscape and more local scales will influence the biotic structure of urban areas. From May 2006 to July 2010, bird surveys were conducted in three metropolitan parks in Cleveland, Ohio, USA. Surveys were conducted with the goal of examining the effect of vegetation structure and adjacent land cover on the distribution and species richness of breeding birds within this park system. A total of 65 species were recorded throughout the study area. Avian species richness was linked to several habitat metrics, measured at both the local and landscape scale. Generally, species richness was highest at locations characterized by moderate forest cover. The proportion of canopy cover at survey sites related negatively to species richness and the density of understory vegetation showed a positive relationship with species diversity. Despite the influence of these three metrics, sensitivity analysis indicates that the density of understory vegetation is the most significant correlate to avian diversity within this suburban park system. Management actions aimed at providing habitat for the greatest diversity of breeding songbirds within the study area should allow for moderate canopy cover while retaining or improving the structural complexity of understory vegetatation.

**Keywords** Canopy cover · Forest cover · Urban birds · Understory density

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## Introduction

Through continued expansion, urban areas increasingly abut wild lands, often leading to the elimination or fragmentation of native communities. Urbanization can irreversibly replace natural habitats with novel ecosystems, often confining remnant communities to urban parks. With predictions of continued urban expansion, urban parks can serve an important role in conserving biodiversity (Niemelä 1999; Savard et al. 2000; Alvey 2006). This may be particularly true for urban bird communities as urban park systems have been identified as the land use that supports the most biological diverse and complex avian communities within urban environments (Ortega-Álvarez and MacGregor-Fors 2009).

Spatial planning aimed at maintaining biological diversity and functionality must consider how land use changes will influence the biotic structure of urban areas (Hasse and Lathrop 2003). For instance, human land use during the 19$^{th}$ century altered the structure and community composition of hardwood dominated forests in the Upper American Midwest (Rhemtulla et al. 2007). With the loss of recurrent disturbance remnant forests have shifted from a dominance of shade intolerant species to an increased presence of shade tolerant species (Burns and Honkala 1990; Lorimer et al. 1994). Such changes in plant communities can affect other ecosystem components, with several studies demonstrating relationships between vertical vegetation complexity and bird species diversity (deCalesta 1994; McShea and Rappole 2000). In urban habitats, where the presence of shrub layers has been shown to be particularly important to the variety of bird species (Gavareski 1976; Melles et al. 2003; Ausprey and Rodewald 2011), these changes may be especially influential to avian diversity.

Because urbanization can add novel ecosystem elements, at both the patch and landscape-levels, conservation of avian diversity within urban parks can poses a unique suite of challenges. Understanding the processes that foster avian diversity within urban park systems is critical to maintaining the ecological support provided by these ecosystems. The main goals of this study were to describe the current diversity and distribution of avifauna in forest patches of an urban park system in the Upper American Midwest. We compare avian diversity across a range of habitat variability, reflecting anticipated variation within this park system. Specifically, we were interested in the potential implications to avian diversity as a consequence of changes in understory density and canopy cover, both at the local- and landscape-scales. In doing so, we address the following specific questions: 1) What are the distribution and abundance patterns of avian species across this park system?, 2) What factors (plot-scale and landscape-scale) are correlated with the distribution of species within this forested urban park system?, 3) What are likely future avifaunal patterns given anticipated plot- and landscape-scale changes in these forest communities? 4) What management practices, if implemented would be likely to help sustain or increase avian diversity?

## Methods

### Study area

Study sites were located in Cleveland Metroparks, a system of 16 reservations in the metropolitan region of Cleveland, Ohio covering more than 8900 ha. Within this region, study sites were located within Rocky River (1045 ha), Hinckley (1070 ha), and Mill Stream Run (1350 ha) reservations, which lie within Cuyahoga and Medina Counties. Based on the

2010 census, populations of Cuyahoga County and Medina County are 1,280,122 and 172,332, respectively. Between 2006 and 2010, an average of 2,225,218 people made recreational visits to one of these three reservations per year (Bixler 2010).

The Cleveland metropolitan area has a humid continental climate with wet, cold winters and dry, warm summers and average temperatures ranging from 1 °C to 28 °C. Cleveland is bordered by Lake Erie to the north and the study area receives, on average 93 cm of precipitation per year. Much of the study area is characterized by steep, forested ravine systems formed along tributaries of the Rocky River. Forests are typically second growth (>70 years old) and composed of mixed-mesophytic species including sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), northern red oak (*Quercus rubra*), shagbark hickory (*Carya ovata*), and yellow-poplar (*Liriodendron tulipifera*). Soils are characterized as deep, moderately well-drained to well-drained (Ritchie and Steiger 1990).

## Study site selection and bird surveys

Two hundred potential point count locations were selected at random throughout the three reservations (Rocky River (RR), Mill Stream Run (MSR), and Hinckley (HI)). Potential survey points were first identified using geographic information systems (ArcVIEW 3.0, Environmental Systems Research Institute, Redlands, California), followed by aerial photo and ground truthing to ensure that each survey point met the following criteria: 1) each survey point was located in woodland (≥ 2 ha) with an average canopy height of≥6 m and canopy cover≥50 %; 2) survey points were at least 50 m from a forest edge, including roads and utility right-of-ways; 3) survey points were no less than 250 m apart. From the initial 200 potential point count locations 49 were selected with points spread across the three reservations (Table 1).

Point counts were carried out over five consecutive years, from 2006 to 2010 (Table 1) and conducted by volunteer observers from the Western Cuyahoga Audubon Society. Prior to field work, all observers were required to pass a rigorous 1-day training course and tested to confirm bird identification and distance estimation skills. During each breeding season, point count locations were visited 3 times during 3 specified calendar periods, with replicate surveys typically carried out by the same observer. Replicate surveys were separated by no less than 2 weeks with point counts conducted following protocols established by Hamel et al. (1996). Point counts were conducted for 10 min and occurred 0–4 h after sunrise during the breeding season.

During each survey, the number of individuals of each species heard or observed within 3 distance bands (<25 m, 26–50 m, and >50 m) was recorded. Species detected during flyovers were not included in the data set. To minimize bias associated with weather, we did not survey during rainy or windy (>15 km/h) conditions. Following data collection, abundance data was converted to presence–absence data and only those birds observed within 50 m of each survey point were considered in further analyses.

## Environmental variables

To assess patterns in avian species diversity and community structure we examined two biologically relevant spatial scales; the landscape scale and the localized habitat scale. The landscape reflected features measured within a 500 m radii centered on each point count location while localized habitat features were measured within 50 m of point count locations. Although features measured within an extent of 500 m have been considered to reflect fine scale landscape attributes (Smith et al. 2011), we adopted this level of measurement as one

**Table 1** Number following reservation name reflects number of bird survey points. Values reflect mean±1 SD. Local canopy cover reflects measurements within 50 m of survey points while landscape tree cover was measured within 500 m of survey points

| Bird surveys | | | | Vegetation surveys | | | | |
|---|---|---|---|---|---|---|---|---|
| Reservation (*n*) | Year | Survey period | *n* | Shrub density (intercepts at<2 m height) | Localcanopy cover (%) | Canopy height (m) | Landscape tree cover (%) | Edge length (km) |
| Hinckley (*15*) | 2006 | June 2 – July 8 | 10 | 4.16±2.98 | 82±22 | 27±8 | 69±16 | 61.67±39.89 |
| | 2007 | June 3 – July 7 | | | | | | |
| | 2008 | June 1 – July 12 | | | | | | |
| | 2009 | May 30 – July 7 | | | | | | |
| | 2010 | May 27 – July 11 | | | | | | |
| Mill Stream (*25*) | 2006 | June 4 – July 5 | 23 | 2.47±2.02 | 68±27 | 25±4 | 67±15 | 65.24±18.78 |
| | 2007 | May 30 – July 7 | | | | | | |
| | 2008 | May 30 – July 19 | | | | | | |
| | 2009 | June 5 – July 15 | | | | | | |
| | 2010 | May 26 – July 14 | | | | | | |
| Rocky River (*20*) | 2006 | June 2 – July 3 | 16 | 3.03±2.66 | 67±22 | 34±16 | 65±12 | 69.35±28.88 |
| | 2007 | June 2 – July 13 | | | | | | |
| | 2008 | June 8 – July 2 | | | | | | |
| | 2009 | May 29 – July 7 | | | | | | |
| | 2010 | May 16 – July 6 | | | | | | |

that typically encompassed both park boundaries and outlying habitat complexes. We defined the 50 m radius of each survey point as localized habitat because this extent was used as the cutoff distance applied to species detections used in our analysis.

The proportion of the landscape with tree cover and the length of forest edge were measured within 500 m of point count locations using aerial imagery collected during 2009 and obtained through National Agricultural Imagery Program (Digital Orthographs, 1 m true color). Using GIS (ArcMap 9.31, Environmental Systems Research Institute, Redlands, California) we quantified all forest cover, treating any point that fell within an area of forest canopy as similar. We did not differentiate types of development resulting in the lack of canopy cover (i.e., low density housing, agriculture, etc.).

During 2007 we measured vegetation structure within 50 m of each point count location. Sampling was based on protocol modified from the design of the USDA Forest Service Forest Inventory and Analysis (FIA) program (USDA Forest Service 2005) where measurements were collected from within 4 5-m subplots within the larger 50 m radius area. The number and species of live trees were counted and shrub density, canopy cover and canopy height were measured within each 5-m subplot. At each point understory tree species (those≤1.4 m high or 12 cm dbh) and dominant canopy trees were classified as shade tolerant, intermediately shade tolerant, or shade intolerant species (Burns and Honkala 1990). Shrub density and canopy cover estimates were averaged across the five subplots. Shrub density was estimated at 5 locations within each subplot using the point intercept method. The first point was located at the center of the subplot and the four additional locations were chosen randomly at locations 3 m from plot center. At each location a 2-m pole was placed vertically and the number of foliage contacts within half-meter intervals recorded. Canopy cover was measured at the center of each subplot using a concave spherical densitometer (Forestry Suppliers, Inc. Jackson, Mississippi). Shrub density and canopy cover estimates were averaged across the five subplots to obtain a single representative value for each point count location.

## Statistical analyses

Relationships among species abundance and environmental variables were analyzed by partial redundancy analysis of bird community composition (pRDA), implemented in the *vegan* package (Oksanen et al. 2010) of R (R Development Core Team 2011). pRDA analysis was used to test linear relationships between a response matrix (abundance of bird species at sampling points) and the explanatory variables (landscape forest cover, length of linear edge, local canopy cover, canopy height and shrub density), while controlling for spatial autocorrelation among co-variables, in our case sites and within years. Species matrices were Hellinger transformed to allow usage of pRDA with datasets containing many zeros (Legendre and Gallagher 2001). Only those species that were observed≥2 times, not strictly as flyovers, and at more than one survey point, were included in the pRDA analysis. The contribution of habitat variables were tested using Akaike's information criterion (using the function *ordistep* in the *vegan* package) with significant variables ($P<0.1$), which explain the greater proportion of variation in the bird community, reflected in the final model. Correlations of significant variables with the species data were presented as vector biplots with vector angles and lengths corresponding to the degree and strength of the correlations. To explore relationships between the distribution of the avian community and the shade tolerance of dominant canopy trees, a bivariate ellipse (standard deviation+95 % confidence limits) was included in the final plot, reflecting shade tolerance classification of the dominant canopy tree species at each sample point.

The Shannon-Weiner index (*H′*) was used to measure species diversity (Shannon and Weaver 1962). Species evenness was calculated as *H*’ divided by the natural log of species richness (Magurran 1988). Relationships between landscape (% forest cover) and local habitat metrics (% canopy cover, density of vegetation <2 m tall) and species diversity (*H′*) were analyzed using linear mixed-effects models implemented in the *lme4* package (Bates et al. 2011) in R (R Core Development Team 2011). The experimental unit in this analysis was the study point. Each model contained the parameter of interest as the response variable (*H′*), with landscape and local habitat metrics as fixed effects. The unique reservation, year, and survey number within year were accounted for by including them as random variables in each model. In most cases relationships between species diversity and habitat were assumed to be linear. However, because species occurrence and landscape-level habitat availability may not follow linear relationships (Betts et al. 2007; Blair and Johnson 2008), we tested for a non-linear relationship between species diversity and landscape forest cover by including a second-order polynomial. Because the different covariates entered as fixed effects were measured on several different scales, the relative influence of each covariate was assessed using z-scores where the individual covariates and response variable (species diversity) were scaled to have a mean of zero and a standard deviation of unity. The importance of each fixed effect was evaluated using a Markov Chain Monte Carlo simulation procedure to generate a highest posterior density credible interval for each fixed effect (*multcomp* package; R Core Development Team 2011). Covariate values were considered to be significantly influential if their credible interval did not include zero.

Hierarchical variance partitioning was performed using the *hier.part* package (Walsh and MacNally 2004) in R (R Core Development Team 2011) to test for independent effects of landscape tree cover, localized canopy cover, and the density of vegetation <2 m tall on species diversity and evenness. In this analysis, the nested structure of the data was ignored (Mac Nally 2000). Sensitivity analysis was used to evaluate the relative importance of landscape canopy cover, vegetation density, and local canopy cover in determining species diversity (*H′*) under future conditions. The sensitivity analysis involved model perturbations by varying all key parameters of the mixed effects model simultaneously within the observed range from this study (Table 1). The relative influence of each covariate was assessed using the covariate z-scores. Uncertainty in model parameters was addressed using bootstrap replicates where each model was run 1000 times with covariate values, generated over the observed range but sampled from uniform distributions.

## Results

### Habitat

Tree cover, measured on the landscape scale (within 500 m of survey points), ranged between 38 and 95 % but did not differ between reservations ($F_{2, 46}$=0.15, *P*=0.86), (Table 1). Canopy cover measured at the more localized scale (within 50 m of each survey point), was similar to that measured at the landscape (range 21–97 %), (Table 1). Landscape canopy cover and canopy cover measured at the localized habitat scale were not significantly correlated ($r$=0.19, $t$=1.32, df=47, $P$=0.19). Shrub density did not differ among reservations ($F_{2, 46}$=1.41, $P$=0.26), (Table 1) but was negatively correlated to local canopy cover ($r$=−0.14, $t$=−3.77, df=724, $P$<0.001). The understory vegetation at 41 % of sites was dominated by saplings of shade intolerant tree species ($z$=14.27, df=1, $P$<0.001).

### Bird community

A total of 65 species were detected during the 5 years of this study (Table 2). Of these, 20 % have experienced regional population declines over the last 30 years with an additional 21 % characterized with unknown or highly variable population trends (Table 2; Partners in Flight 2005).

Of the 65 species detected, 40 species were detected≥2 times, were not detected strictly as flyovers, and were subsequently included in the pRDA analysis (Table 2). Avian community structure was evident in the pRDA with the first two components, explaining 81 % of variation in the avian community (Fig. 1). Of the five habitat metrics considered (Table 1), the *ordistep* procedure selected only landscape forest cover, local canopy cover, and shrub density for entrance into the final model. In general, community composition ranged from species associated with a dense forest canopy to those associated with a more open canopy and dense understory (Fig. 1, from left to right). Several species depicted furthest from the centroid included Song Sparrow (*Melospiza melodia*) and Gray Catbird (*Dumetella carolinensis*), species associated with denser understory (Fig. 1). Few species occurred in a highly forested landscape with dense canopy cover at local scales (Fig. 1). One species associated with this environment was the Red-eyed Vireo (*Vireo olivaceus*), (Fig. 1). The majority of species occurred centrally along the gradient between increased cover within 2 m of ground and forest cover (Fig. 1). At more than half of all sampling locations (65 %) dominant canopy trees included both shade tolerant and intolerant species. Remaining locations reflected canopy trees characterized by wholly shade tolerant (12 %) or intolerant species (22 %). Ellipses, reflecting the shade tolerance of the dominant canopy trees, overlapped considerably, indicating similar avian communities despite differences in canopy tree species (Fig. 1).

At the survey point species diversity ranged from 4 to 29 species. The regression model fitted to these data indicated that species richness was low in areas with the least canopy cover, increased as the canopy cover gradient increased, and peaked in areas characterized with 60–70 % forest cover. At the survey point, species richness related negatively with canopy cover but positively with understory density (Table 3). Species evenness showed a positive response to understory density (Table 3).

Hierarchical partitioning of the explained variance indicated that the independent effect of density of vegetation <2 m tall is the most important factor in determining avian diversity. Overall, the explanatory power of landscape tree cover and localized canopy cover were low (Fig. 2). The influence of the density of understory vegetation on species richness was 2.5 times higher than that of local canopy cover (Table 3), indicating that this characteristic significantly adds to the importance of the forests in determining the diversity of the avian community. The sensitivity analysis corroborated this result with species richness most closely tied to understory structure (Fig. 3). The combined influence of changes in landscape forest cover and local canopy cover had little influence on species richness (Fig. 3, panel a). In contrast, change in canopy cover had less influence on species richness than understory structure (Fig. 3, panel b).

## Discussion

Studies report non-linear patterns between species diversity and landscape development, where species diversity can peak at intermediate levels of development (Blair 1996;

**Table 2** Species recorded in urban forests presented by the range of habitat measurements (min−max) recorded at point count locations where detected. Single values reflect species detected at only one location

| Species | Alpha Code | Density of vegetation < 2 m above ground (# of foliage hits) | Local canopy cover (%) | Landscape tree cover (% of landscape) | North American population trend * |
|---|---|---|---|---|---|
| Acadian Flycatcher† (*Empidonax virescens*) | ACFL | (0.2–6) | (23–96) | (42–92) | 3 |
| Alder Flycatcher (*Empidonax alnorum*) | ALFL | 3.4 | 79 | 56 | 2 |
| American Crow (*Corvus brachyrhynchos*) | AMCR | (0.2–9.6) | (23–97) | (42–92) | 1 |
| American Goldfinch† (*Carduelis tristis*) | AMGO | (0.2–9.6) | (21–96) | (38–92) | 2 |
| American Redstart† (*Setophaga ruticilla*) | AMRE | (0.2–8.6) | (23–93) | (42–82) | 2 |
| American Robin† (*Turdus migratorius*) | AMRO | (0.2–9.6) | (21–96) | (38–95) | 2 |
| Baltimore Oriole† (*Icterus galbula*) | BAOR | (0.2–9.6) | (21–96) | (38–92) | 4 |
| Black-capped Chickadee† (*Poecile atricapillus*) | BCCH | (0.2–9.6) | (21–97) | (38–95) | 1 |
| Barred Owl (*Strix varia*) | BAOW | 0.8 | 78 | 79 | 3 |
| Belted Kingfisher (*Ceryle alcyon*) | BEKI | (0.2–6.2) | (41–81) | (52–82) | 4 |
| Blue-gray Gnatcatcher† (*Polioptila caerulea*) | BGGN | (0.2–6.2) | (21–96) | (43–82) | 3 |
| Blue-headed Vireo (*Vireo solitarius*) | BHVI | 5.6 | 97 | 82 | 1 |
| Blue Jay† (*Cyanocitta cristata*) | BLJA | (0.2–8.6) | (21–97) | (38–95) | 2 |
| Black-and-white Warbler (*Mniotilta varia*) | BWWA | (1.6–9.6) | (23–92) | (45–69) | 1 |
| Brown-headed Cowbird† (*Molothrus ater*) | BHCO | (0.2–8) | (23–97) | (42–82) | 4 |
| Carolina Wren† (*Thryothorus ludovicianus*) | CAWR | (0.2–8) | (21–96) | (42–88) | 1 |
| Cedar Waxwing† (*Bombycilla cedrorum*) | CEDW | (0.2–6) | (41–93) | (38–88) | 1 |
| Cerulean Warbler (*Setophaga cerulea*) | CERW | (0.2–4.4) | (32–92) | (38–70) | 3 |
| Chipping Sparrow† (*Spizella passerina*) | CHSP | (0.2–8.6) | (41–96) | (38–73) | 2 |
| Common Grackle† (*Quiscalus quiscula*) | COGR | (0.2–9.6) | (23–97) | (38–95) | 5 |
| Cooper's Hawk (*Accipiter cooperii*) | COHA | (2.2–3.4) | (76–90) | (73–82) | 1 |
| Common Yellowthroat† (*Geothlypis trichas*) | COYE | (0.2–9.6) | (23–96) | (43–82) | 2 |
| Downy Woodpecker† (*Picoides pubescens*) | DOWO | (0.2–8.6) | (21–97) | (38–95) | 2 |

**Table 2** (continued)

| Species | Alpha Code | Density of vegetation < 2 m above ground (# of foliage hits) | Local canopy cover (%) | Landscape tree cover (% of landscape) | North American population trend * |
|---|---|---|---|---|---|
| Eastern Bluebird (*Sialia sialis*) | EABL | 4.4 | 56 | 59 | 1 |
| Eastern Kingbird (*Tyrannus tyrannus*) | EKKI | 1.6 | 85 | 38 | 4 |
| Eastern Phoebe (*Sayornis phoebe*) | EAPH | (0.6–0.6) | (24–92) | (65–92) | 2 |
| Eastern Towhee† (*Pipilo erythrophthalmus*) | EATO | (0.2–8.6) | (23–96) | (42–82) | 5 |
| Eastern Wood-Pewee† (*Contopus virens*) | EAWP | (0.2–7.8) | (21–96) | (42–95) | 4 |
| European Starling (*Sturnus vulgaris*) | EUST | (1.6–6) | (74–87) | (38–81) | 4 |
| Field Sparrow† (*Spizella pusilla*) | FISP | (1.6–8.6) | (23–92) | (42–69) | 5 |
| Great Crested Flycatcher† (*Myiarchus crinitus*) | GCFL | (0.2–9.6) | (21–97) | (38–95) | 3 |
| Gray Catbird† (*Dumetella carolinensis*) | GRCA | (0.2–9.6) | (21–94) | (42–90) | 3 |
| Hairy Woodpecker† (*Picoides villosus*) | HAWO | (0.2–8) | (21–97) | (42–90) | 1 |
| House Finch (*Carpodacus mexicanus*) | HOFI | (0.2–1.6) | (41–85) | (38–52) | 1 |
| Hooded Warbler† (*Setophaga citrina*) | HOWA | (0.2–9.6) | (23–97) | (42–92) | 1 |
| House Wren† (*Troglodytes aedon*) | HOWR | (0.2–9.6) | (21–96) | (42–90) | 3 |
| Indigo Bunting† (*Passerina cyanea*) | INBU | (0.2–8.6) | (21–96) | (43–82) | 2 |
| Louisiana Waterthrush (*Parkesia motacilla*) | LOWA | 0.6 | 24 | 92 | 3 |
| Magnolia Warbler (*Setophaga magnolia*) | MAWA | 3.4 | 96 | 67 | 2 |
| Northern Cardinal† (*Cardinalis cardinalis*) | NOCA | (0.2–9.6) | (21–97) | (38–95) | 1 |
| Northern Flicker† (*Colaptes auratus*) | NOFL | (0.2–8.6) | (21–96) | (42–88) | 5 |
| Northern Rough-winged Swallow (*Stelgidopteryx serripennis*) | NRWS | (0.6–3.6) | (41–96) | (60–73) | 3 |
| Ovenbird† (*Seiurus aurocapilla*) | OVEN | (0.2–5.6) | (23–97) | (45–92) | 2 |
| Pine Warbler (*Setophaga pinus*) | PIWA | 2.2 | 90 | 82 | 1 |
| Pileated Woodpecker† (*Dryocopus pileatus*) | PIWO | (0.2–8.6) | (21–97) | (43–92) | 1 |

**Table 2** (continued)

| Species | Alpha Code | Density of vegetation < 2 m above ground (# of foliage hits) | Local canopy cover (%) | Landscape tree cover (% of landscape) | North American population trend * |
|---|---|---|---|---|---|
| Prothonotary Warbler (*Protonotaria citrea*) | PROW | (0.6–2.6) | (24–92) | (49–92) | 3 |
| Red-breasted Nuthatch (*Sitta canadensis*) | RGBR | (1.2–6) | (32–93) | (45–90) | 1 |
| Red-bellied Woodpecker† (*Melanerpes carolinus*) | RBWO | (0.2–9.6) | (21–97) | (38–92) | 1 |
| Red-eyed Vireo† (*Vireo olivaceus*) | REVI | (0.2–9.6) | (21–97) | (42–95) | 1 |
| Red-shouldered Hawk (*Buteo lineatus*) | RSHA | (0.2–6) | (21–92) | (38–73) | 3 |
| Ruby-throated Hummingbird (*Archilochus colubris*) | RTHU | (0.2–9.6) | (21–96) | (42–88) | 1 |
| Red-winged Blackbird† (*Agelaius phoeniceus*) | RWBL | (1.4–9.6) | (24–96) | (38–82) | 4 |
| Rose-breasted Grosbeak† (*Pheucticus ludovicianus*) | RBGR | (0.2–9.6) | (21–97) | (42–90) | 3 |
| Scarlet Tanager† (*Piranga olivacea*) | SCTA | (0.2–7.8) | (21–96) | (42–92) | 2 |
| Song Sparrow† (*Melospiza melodia*) | SOSP | (0.2–9.6) | (21–96) | (42–82) | 4 |
| Tufted Titmouse† (*Baeolophus bicolor*) | TUTI | (0.2–8.6) | (23–97) | (42–92) | 1 |
| Veery (*Catharus fuscescens*) | VEER | (1–9.6) | (45–93) | (42–82) | 2 |
| Warbling Vireo† (*Vireo gilvus*) | WAVI | (0.2–6.2) | (23–97) | (52–82) | 1 |
| White-breasted Nuthatch† (*Sitta carolinensis*) | WBNU | (0.2–8.6) | (21–97) | (38–95) | 1 |
| Willow Flycatcher (*Empidonax traillii*) | WIFL | (4.4–5.6) | (87–97) | (68–82) | 2 |
| Wild Turkey (*Meleagris gallopavo*) | WITU | 8.6 | 78 | 43 | 1 |
| Wood Thrush† (*Hylocichla mustelina*) | WOTH | (0.2–7.8) | (21–97) | (42–90) | 4 |
| Yellow-billed Cuckoo† (*Coccyzus americanus*) | YBCU | (0.4–7.8) | (32–96) | (45–90) | 3 |
| Yellow-throated Vireo (*Vireo flavifrons*) | YTVI | (4.4–6) | (24–97) | (60–88) | 3 |
| Yellow Warbler† (*Setophaga petechia*) | YWAR | (0.2–8.6) | (23–96) | (43–88) | 2 |

*For each species, we show the 30-year (ending in 2001) population trend for the Lower Great Lakes Region (Partners in Flight 2005): (1) ≥50 % increase, (2) 15–49 % increase, (3) highly variable or unknown, (4) 15–49 % decrease, (5) ≥50 % decrease

†Species included in the pRDA analysis

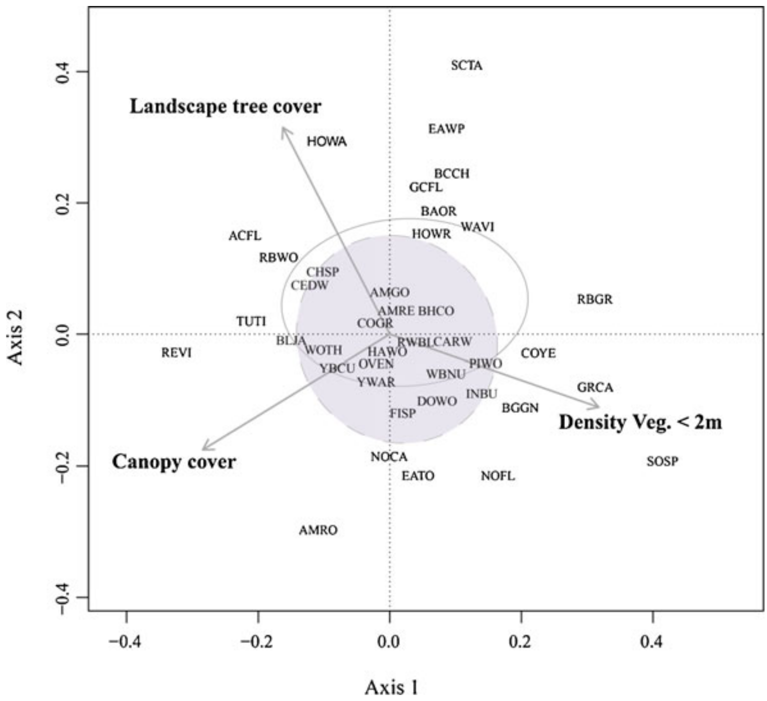


**Fig. 1** Species community analysis by pRDA depicting 40 species in the environmental space of the first two canonical axes. The three best supported explanatory variables are in bold letters. Longer arrows illustrate a higher correlation of the species with one of the main explanatory variables. Four letter alpha codes denoting species names appear in Table 2. Grey filled ellipse encompasses 95 % confidence area of habitat communities dominated by shade intolerant canopy tree species. Open ellipse reflects canopy dominated by shade tolerant species

McKinney ML Lockwood 1999; Blair 2004). In our study area, we found that species richness followed this non-linear pattern peaking in landscapes characterized by 60 to 70 % forest cover. This increase in species richness towards moderate levels of forest cover likely reflects the addition of widely distributed generalist species, those found throughout this suburban landscape (Table 2). With increased forest cover, species composition changed from including "urban-adapted" species such as Black-capped Chickadee (*Poecile atricapillus*), Northern Cardinal (*Cardinalis cardinalis*) and House Finch (*Carpodacus mexicanus*) to forest specialists such as Scarlet Tanager (*Piranga olivacea*), Hooded Warbler (*Setophaga citrina*) and Rose-breasted Grosbeak (*Pheucticus ludovicianus*), (Fig. 1). The majority of species detected however were found over a broad range of forest covers (Table 2) and therefore could be considered generalist species. The homogenization of the avian community towards greater representation by generalist species is a common occurrence within disturbed habitats (McKinney and Lockwood 1999; Devictor et al. 2008) whereas generalist species may be less influenced by landscape characteristics, using a variety of habitat types within the landscape matrix (Mitchell et al. 2001).

For the majority of species, our results suggest that the diversity of local habitat features such as understory density and canopy cover may be more influential than landscape scale metrics. For many birds, suitable breeding habitat can include

**Table 3** Results of the habitat analysis for z-scores of covariates influencing avian species diversity and evenness

| | Coefficient | Standard error | 95 % CI |
|---|---|---|---|
| Species diversity (*H′*) | | | |
| Intercept | 1.25 | 0.11 | (0.82–1.72) |
| Landscape tree cover | 0.67 | 0.15 | (0.37–0.98) |
| Landscape tree cover ^2 | −0.68 | 0.15 | (−0.97–−0.38) |
| Local canopy cover | −0.04 | 0.02 | (−0.08–−0.01) |
| Density veg <2 m | 0.10 | 0.02 | (0.06–0.13) |
| Species evenness | | | |
| Intercept | 0.75 | 0.08 | (0.61–0.87) |
| Landscape tree cover | 0.12 | 0.07 | (−0.01–0.25) |
| Landscape tree cover ^2 | −0.10 | 0.06 | (−0.24–0.02) |
| Local canopy cover | 0.00 | 0.01 | (−0.02–0.02) |
| Density veg<2 m | 0.02 | 0.01 | (0.00–0.03) |

transitional ecotones such as those associated with canopy gaps, habitat features once common in the uneven-aged forests of the Midwestern United States (Abrams 2003). While transitional habitats can influence avian assemblage, they are also linked to demographic processes within avian communities (Mitchell et al. 2006; Croci et al. 2008; Rush and Stutchbury 2008; Ausprey and Rodewald 2011). Thus, a particular caveat of this study is that species richness and evenness do not provide evidence that a particular habitat can support long-term persistence (Johnson 2007).

Although we did not directly account for edge effects in our analysis and cannot disregard the potential influence of edge (whether positive or negative) on demographic processes, our results do show that understory density, whether attributable to edge, canopy gap or other factor, was the major habitat component affecting avian diversity within these suburban forests. Because local habitat appears important within these systems, management practices focused on the forest stand may be the appropriate scale to increase biodiversity within this urban park system. The ecological significance of forest canopy gap formation should be considered, especially in systems where natural agents of gap formation have been reduced or removed (Greenberg and Lanham 2001; Forsman et al. 2010). Because closed canopy conditions can limit natural regeneration and stem density actions that restore the conditions fostering regeneration, the creation of forest gaps can benefit understory density and diversity (Runkle 1990). This may be particularly true for late successional forest communities, such as those in this study which are dominated by shade intolerant understory and overstory components (Gravel et al. 2010). This management option must be undertaken with appreciation for the potential effects of disturbance in promoting species composition and ecosystem change (Hausman et al. 2010). Benefits of artificial gap creation may be fully realized only if proximate factors such as invasion by exotic plant species (Hausman et al. 2010) can be avoided and deer browsing can be maintained at moderate levels (Royo et al. 2010). Because deer browsing can account for considerable spatial variability in the abundance and diversity of bird populations (DeGraaf et al. 1991; deCalesta 1994; Allombert et al. 2005),

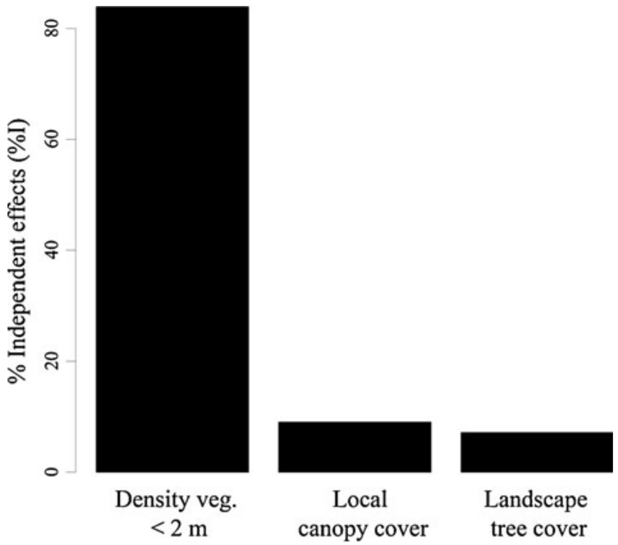


**Fig. 2** Relative independent effects of landscape tree cover, localized canopy cover and density of vegetation within 2 m of the ground on the diversity of avifauna expressed as the percentage of total explained variance

managers working to restore or bolstering avian diversity must consider the role of deer herbivory when evaluating management options.

## Management recommendations

Conservation efforts aimed at supporting biological diversity within urban park systems often present unique challenges. Understanding the influence of local and landscape-level habitat attributes on biodiversity in urban parks should be considered in urban planning. Given the current conditions within our study area, we believe avian diversity can be supported or enriched by implementing the following recommendations:

1. Conserve forested landscapes but focus restoration efforts on localized habitat management: Avian conservation strategies reflect tradeoffs in supporting generalist and specialist species, often requiring the balance of landscape conservation (which can often extend beyond park boundaries) and implementing smaller scale habitat management. Within our study area, species diversity was most influenced by understory structure measured at the local scale. Although landscapes with moderate forest cover should be conserved whenever possible, focus should be given to conserving and restoring local habitat components such as understory structure when resources for management are limited. Understory structure will benefit from supplemental plantings, deer population management, and the formation of canopy gaps.
2. Allow for the formation of canopy gaps: temper management strategies aimed at providing uniform habitat, including canopy cover and reduced landscape heterogeneity. Allowing for natural disturbance such as the formation of canopy gaps can have a critical positive impact on biodiversity.
2. Ecosystem monitoring and management: Monitoring a suite of metrics including understory structure, canopy cover and deer populations will provide valuable information

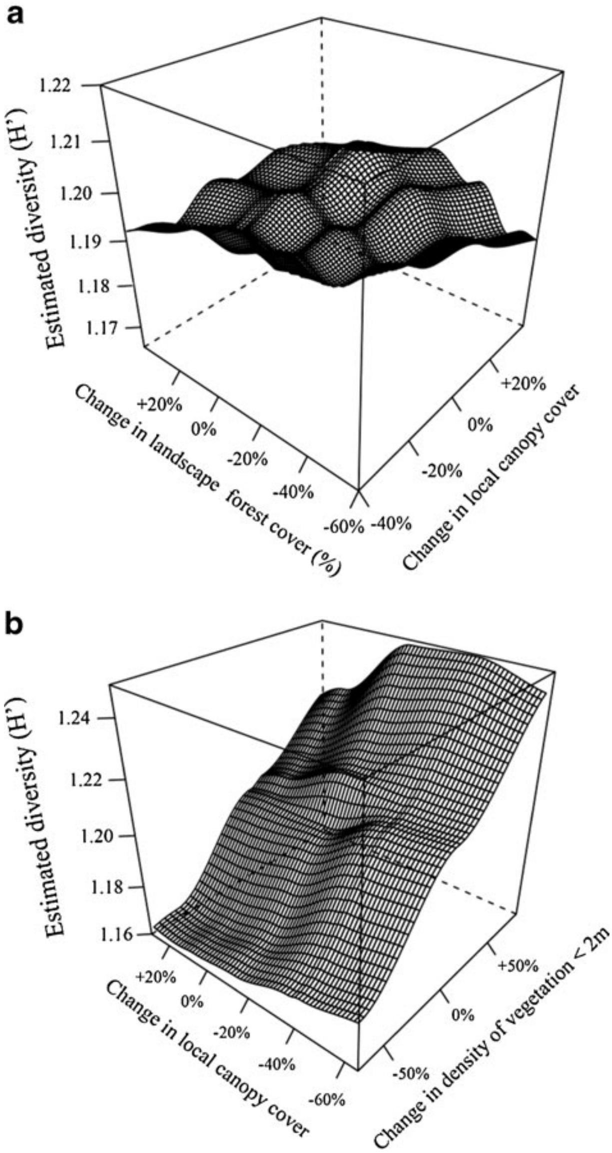


**Fig. 3** Predictions of species diversity ($H'$) response to averaged changes in landscape tree cover and local canopy cover (panel **a**) and landscape tree cover and the density of vegetation within 2 m of the ground (panel **b**). X and Y axis values reflect change in indicated habitat measurement relative to mean

for conserving communities. However, incorporating a broader monitoring scheme that includes the collection of information on rates of change of some system state variables, such as demographics and species gains or losses can provide information invaluable to ecosystem management.

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