

MAKING THE MOST OF WHAT REMAINS: EXAMINING THE QUALITY OF URBAN
GRASSLANDS FOR BIRDS IN ILLINOIS

BY

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THESIS

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ABSTRACT

Grassland bird populations have experienced steep declines across North America, with especially precipitous declines in the Midwest. The primary cause of these declines is thought to be loss and fragmentation of grassland habitat. Illinois alone has lost >99% of its native prairie, and has steadily lost agricultural grasslands over the past 50 years. Despite this statewide decline in grassland availability, there are >12,000 ha of grassland in the Chicago region that may provide valuable habitat for imperiled grassland bird species. However, little is known about the value of grasslands located in an urban matrix. To investigate whether grassland birds are using these grasslands and whether they support viable populations, I examined grassland bird habitat use, reproductive success, and nest predator identities along an urban-to-rural gradient in the greater Chicago metropolitan region. Of the seven obligate grassland species I studied, only Sedge Wrens (*Cistothorus platensis*) displayed a strongly negative association to the amount of development in the landscape. I found that nest predation rates decreased in more developed landscapes, as did the probability of brood parasitism. I also found that coyotes (*Canis latrans*) and white-tailed deer (*Odocoileus virginianus*) were the dominant nest predators in this study system. This research contributes to an ongoing effort to define what factors are important for designing grassland bird conservation areas in urban landscapes.

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CHAPTER 1

GENERAL INTRODUCTION

Throughout North America, grassland bird populations have been in a steady state of decline for the past several decades (Sauer et al. 2009). These declines are largely attributed to habitat loss and fragmentation. In the Midwestern U.S., grassland habitat loss has been particularly severe, with less than 99% of original tall-grass prairie remaining (Herkert 1995). Illinois provides an excellent example of this dramatic habitat loss; <0.01% of the tall-grass prairie that once covered over half of the state remains (Iverson 1988). In addition, the agricultural grasslands that once provided nesting habitat for grassland birds throughout the Midwest have declined to their lowest point in over 100 years (Herkert et al. 1996). In Illinois, the land area of pastures and hayfields have decreased by >75% and >50%, respectively (Herkert 1994). Land cover in Illinois is now primarily dominated by row-crop agriculture, forest, and developed areas (Walk et al. 2010). As a result of these statewide habitat declines, most grassland birds have experienced a cumulative loss of 40% to 90% of their populations over the past 50 years (Walk et al. 2010).

Grassland birds breeding in Illinois are increasingly reliant on isolated grassland patches surrounded by a novel land-cover matrix. This matrix typically consists of row-crop agriculture or developed land. Although grasslands in agricultural landscapes have been extensively studied, little is known about the habitat quality of grasslands in developed landscapes and their importance to breeding grassland bird populations. Development is assumed to be hostile, but few studies have examined grassland bird abundance or reproductive success in urbanized landscapes; instead, most studies are conducted in landscapes dominated by row-crop agriculture, grassland, or trees (e.g., Ribic and Sample 2001, Fletcher and Koford 2002, Bakker

et al. 2002, Horn and Koford 2004, Grant et al. 2004, Cunningham and Johnson 2006; Winter et al. 2006, Renfrew and Ribic 2008). As development in many areas is likely to increase, understanding the relationship between landscape-level development and the ecological needs of grassland birds will become important.

Though some factors, such as increased disturbance, noise, or light pollution, may deter grassland birds from using habitat in developed areas (Shanahan et al. 2014), urban matrices may benefit birds through changes in predator communities. Previous work in urban systems has documented increased abundance of prey species in urban areas and prey switching by important nest predators, which could result in greater nest survival for birds in urban landscapes (Stracey 2011, Fischer et al. 2012). Thus, examining how predator communities change with urbanization is an important part in understanding bird population dynamics in such landscapes.

In northern Illinois, the number and size of grasslands have decreased while developed land cover and the number of people have increased by 135% and 30%, respectively, over the past 50 years (Walk et al. 2010). Despite this intensified development, a mosaic of >12,000 ha of grasslands exists in the greater Chicago metropolitan region (Audubon Chicago Region 2011). This may be valuable habitat for grassland breeding birds, particularly Bobolinks (*Dolichonyx oryzivorus*) and Savannah Sparrows (*Passerculus sandwichensis*) that are common species to northern Illinois. The extent, to which grassland birds are using these grasslands, and whether they support viable populations, remains unknown. In general, research on grassland birds in developed landscapes is lacking despite the increasing restoration and creation of grasslands in developed landscapes throughout North America. I addressed this knowledge gap by studying grassland bird habitat use, reproductive success, and nest predator communities in grasslands located along an urban-to-rural gradient in the greater Chicago metropolitan region.

THESIS ORGANIZATION

This thesis is organized into four chapters: Chapter 1 is a general introduction. Chapter 2 discusses factors influencing grassland bird density along an urban-to-rural gradient. Chapter 3 examines reproductive success of grassland birds along an urban-to-rural gradient. Chapter 4 summarizes conclusions from the previous chapters and addresses important directions for future research.

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CHAPTER 2
EFFECTS OF LANDSCAPE-LEVEL DEVELOPMENT ON GRASSLAND BIRD
HABITAT USE

ABSTRACT

For the past several decades, grassland birds have experienced steep population declines across North America. These declines have been particularly severe in the Midwest, where most grassland cover has been lost, fragmented, and surrounded by unsuitable habitat. Conservation efforts have focused on protecting large grasslands surrounded by minimal amounts of cover types that are assumed to be hostile for grassland birds, including trees and development. Though urban development is considered hostile and grasslands located in urban areas are assumed to be of poor quality, this assumption has not been tested. In heavily fragmented landscapes where habitat is limited, urban grasslands may be of significant value to grassland birds. I examined grassland bird response to development and additional landscape and habitat variables in the greater Chicago metropolitan area. In the breeding seasons of 2012 and 2013, I surveyed bird communities in 30 grassland patches representing a gradient of urbanization and patch sizes. Density of Savannah Sparrows (*Passerculus sandwichensis*) increased with amount of development, while density of Sedge Wrens (*Cistothorus platensis*) decreased. Development did not appreciably impact Bobolinks (*Dolichonyx oryzivorus*), Dickcissels (*Spiza americana*), Eastern Meadowlarks (*Sturnella magna*), Grasshopper Sparrows (*Ammodramus savannarum*), or Henslow's Sparrows (*Ammodramus henslowii*). Response to landscape-level forest cover varied by species but was not a strong predictor of density for most species, whereas patch size had a positive effect on the majority of species densities. These results indicate that conservation-

priority grassland birds are using urban grasslands and these grasslands may provide quality habitat for some species.

INTRODUCTION

Grassland birds in North America have experienced steeper population declines than any other avian group over the last three decades (Sauer et al. 2009). This trend is particularly evident in the Midwest, where populations of >75% of all grassland species declined from 1966–1993 (Herkert 1995). These declines have largely resulted from the loss and fragmentation of grassland habitat; of the 38.1 million ha of tall-grass prairie that once blanketed the core Midwest, <1% remains (Herkert 1995). Agricultural grasslands (i.e. hayfields and pastures) that once provided breeding habitat for grassland birds have declined by >50% in the Midwest and are currently at their lowest levels in >100 years (Herkert et al. 1996). These grassland declines are primarily attributed to increases in agricultural intensification and urbanization (Herkert 1994). In the Midwest, most grassland habitat has been reduced and fragmented by row-crop agriculture, resulting in small, isolated habitat patches.

The effects of habitat fragmentation, as well as some local and landscape level variables on grassland bird occupancy, abundance, and density are well documented (e.g., Herkert 1994, Winter and Faaborg 1999, Ribic and Sample 2001, Johnson and Igl 2001, Cunningham and Johnson 2006). Numerous studies have examined species response to local-level vegetation variables, resulting in relatively predictable associations to habitat characteristics (synthesized in Dechant et al. 1999). Patch size has also been shown to regularly influence occupancy or abundance of grassland species including Eastern Meadowlarks (*Sturnella magna*), Grasshopper Sparrows (*Ammodramus savannarum*), and Henslow's Sparrows (*Ammodramus henslowii*; Ribic

et al. 2009). These species, termed “area-sensitive” (Herkert 1994), avoid small patches and are observed more frequently in larger patches. Area sensitivity may be mediated by landscape context. Some have found area sensitivity to be more pronounced in landscapes with lower grass or greater woody cover (e.g., Bakker et al. 2002, Horn and Koford 2004, Renfrew and Ribic 2008).

The majority of grassland bird studies have been conducted in rural landscapes where, in addition to agricultural crops, the primary cover types are either grassland or woodland. These studies have found that grassland birds generally respond positively to grass cover and negatively to wooded cover in the landscape (Ribic and Sample 2001, Grant et al. 2004, Horn and Koford 2004, Cunningham and Johnson 2006). As a result of these findings, conservation efforts have focused on protecting large grassland patches surrounded by a matrix primarily dominated by grassland with a minimal amount of woody or developed cover (Sample and Mossman 1997). This guiding framework, referred to as the Grassland Bird Conservation Area (BCA) model, assumes that developed areas are hostile to grassland birds and that habitat located within 50 m of developed areas are of poor quality (Johnson et al. 2010). However, few studies have considered the effects of development on occurrence or abundance of grassland birds (Veech 2006), and thus the assumption that developed landscapes are hostile may incorrectly undervalue urban grasslands. As development increases, conservation of habitat in urbanized landscapes will become increasingly common. Factors influencing the value of these habitats must be assessed to inform future conservation and management decisions.

In northeastern Illinois, there are >12,000 ha of grassland that may provide valuable habitat for grassland birds (Audubon Chicago Region 2011). Most of these grasslands are located in an urban matrix, and the extent to which grassland birds are using these areas is

unknown. To assess the impact of urbanization on grassland bird abundance, I sampled 30 grassland patches of varying size across an urban to rural gradient during 2012 and 2013. I examined how amount of development surrounding a patch, as well as patch size, and other local and landscape-level variables affected the density of grassland birds.

METHODS

Study area

I sampled grassland birds in 30 sites located throughout Cook, DuPage, Kane, and McHenry Counties, Illinois (Fig. 1). All sites were located in natural areas maintained by county forest preserve districts. Sites located in Cook and DuPage counties were surrounded by greater amounts of urbanization than those located in Kane and McHenry counties (Appendix A, Table A1). Sites were dominated by grassland vegetation, and included native warm-season grasses and non-native cool-season grasses. Each site consisted of a grassland patch within a preserve, and sites were at located least 1 km apart. Patches ranged from 3.0 to 400.0 ha (mean = 159.9 ± 134.8 SD). If trees extended across $\geq 75\%$ of the grassland patch and tree-cover was ≥ 20 m wide, I considered this to be a site edge (Winter et al. 2006). I considered two- and four-lane roads with disturbed roadsides to be site edges, whereas trails or roads without disturbed roadsides were not considered edges (Winter et al. 2006). Only one site was bisected by a two-lane road with a disturbed roadside. I considered agricultural fields to constitute site edges. In the urbanized counties, lawns often comprised edges of patches.

Bird Surveys

I surveyed bird communities in the breeding seasons of 2012 and 2013 between mid-May and early July. I surveyed birds using 10 minute, unlimited radius point counts. I recorded all birds seen and heard at a point and estimated the distance of each bird from the point count center using a laser range finder. I conducted surveys between sunrise and 0930 hours CDT, and did not survey birds during heavy fog, wind, or steady drizzle or rain (Ralph et al. 1993). I systematically placed point-count stations throughout each site prior to the field season using ArcGIS (ESRI 2011) and established stations ≥ 100 m away from patch edges and ≥ 300 m from other point-count stations. I repeated bird surveys four times in 2012 and three times in 2013, as the 2012 data showed that detection probability for the majority of species decreased significantly during the fourth visit. Visits were separated by ≥ 14 days. Surveys were conducted by two observers each season that rotated among patches to minimize potential bias. Observers were trained for >1 week to ensure correct species identification and accurate distance estimation.

Local vegetation measurements

I conducted vegetation sampling after the last round of point counts had been completed at a site (11 July – 5 August). I adapted our sampling design for vegetation structure and composition from the BBIRD Grassland Field Protocol (Martin et al. 1997). I sampled four plots at each point-count station, one at the point-count center, and the other three at 0, 120, and 240° from the central plot at distances of 30, 60, and 90 m from the central plot. At each plot, I used a 50 × 30 cm Daubenmire frame to assess percent cover of grass, bare ground, standing dead vegetation, litter, forbs, and woody vegetation (Daubenmire 1959). I used a Robel pole marked

at 10-cm increments to determine visual obstruction (Robel et al. 1970). I recorded the height at which 90% of the pole was obscured by vegetation from 4 m away from the pole in the four cardinal directions at a height of 1 m above the ground (Robel et al. 1970). I averaged these four readings into a single visual obstruction measurement for each sampling point. I estimated litter depth and average vegetation height using a ruler (in mm). For all vegetation measurements, I averaged the values from the four plots to generate one value for each point-count station.

Landscape variables

I quantified landscape composition surrounding each patch using ArcGIS 10.0 (ESRI 2011) and data from the 2012 National Agricultural Statistics Service Cropland Data Layer. Land cover was classified as low-density developed (impervious surfaces 20%-49%), medium-density developed (impervious surfaces 50%-79%), high-density developed (impervious surfaces 80%-100%), forested land, grass land, hay/pasture cover (hereafter referred to as hay cover) and crop land. I combined all three categories of development together to obtain a single proportion of development for each site. Using Geospatial Modelling Environment (Beyer 2012), I obtained proportions of each land cover variable in a 1600-m buffer around each site.

Statistical analyses

I restricted analyses to 7 species of obligate grassland birds: Bobolink (*Dolichonyx oryzivorus*), Dickcissel (*Spiza americana*), Eastern Meadowlark, Grasshopper Sparrow, Henslow's Sparrow, Savannah Sparrow (*Passerculus sandwichensis*) and Sedge Wren (*Cistothorus platensis*) (Vickery et al. 1999). I obtained density estimates (birds/ha) for each species using Program Distance 6.0 Release 2 (Thomas et al. 2010). I compared candidate

models for detection functions that included no covariates, as well as effects of observer, day of season, and vegetation density. I fit models using combinations of uniform, half-normal, and hazard-rate base functions with series expansions of cosine, hermite polynomial, and simple polynomial. I truncated observations for all species at 100 m and binned observations into 20 m intervals. I examined Akaike's Information Criterion (AIC) values to determine which model best fit the data (Burnham and Anderson 2002) and used goodness of fit tests to confirm overall model fit (Thomas et al. 2010).

Prior to examining candidate models of factors influencing density, I examined correlations among vegetation and landscape variables. There were strong correlations between visual obstruction and height of live vegetation ($r = 0.75$), as well as developed land and cropland within 1.6 km ($r = -0.81$). I retained cover of developed land because this was our primary interest, and removed cropland cover from subsequent analyses. I also removed height of live vegetation as visual obstruction includes information on both vegetation height and density. Patch size was not highly correlated with any of the landscape or vegetation variables.

I used general linear mixed models (SAS PROC MIXED) to analyze factors influencing density (SAS Institute 2011). To account for multiple points within sites, and the sampling of each site in two separate years I specified random effects of site and year \times site. I examined diagnostic plots to confirm that the residuals approximated a normal distribution and met the assumption of homogeneous variance. When necessary, I accounted for heterogeneous variances by including a dispersion effect in analysis (Littell et al. 2006).

I generated two model sets evaluating influences of 1) local-level (i.e. vegetation) variables and 2) combinations of the best local-level model and landscape variables (i.e. patch size, developed cover, grass cover, hay cover, and forest cover) on species density. I included

the best local-level model as an additive effect in the landscape models because habitat variables are well-established predictors of species density (e.g., Cunningham and Johnson 2006, Renfrew and Ribic 2008, Fisher and Davis 2010). Descriptions for each variable used in the models are located in Table 1. I ranked models according to AIC adjusted for small sample size (AIC_c ; Burnham and Anderson 2002). I model-averaged parameter estimates and 95% confidence intervals using the zero method (Grueber et al. 2011) to account for any model selection uncertainty (Burnham and Anderson 2002).

RESULTS

I surveyed 92 unique point count stations, resulting in a total of 364 point surveys in 2012 and 297 point surveys in 2013. Eastern Meadowlarks were the most abundant of the seven obligate grassland bird species examined in the study (22.5% of observations). Dickcissels were the second most abundant (21.9% of observations), followed by Bobolinks (19.7% of observations). Sedge Wrens and Grasshopper Sparrows were the least abundant, with each representing <7% of total observations (Table 2).

Response to vegetation variables differed by species (Appendix B). The amount of dead vegetation was the best local predictor of Bobolink and Eastern Meadowlark density, with both species responding negatively to dead vegetation. Height of dead vegetation was the best local predictor of Grasshopper Sparrows and Sedge Wrens, with Sedge Wrens responding positively to height of dead vegetation and Grasshopper Sparrows responding negatively. Savannah Sparrows displayed a negative response to litter depth, while Dickcissels displayed a negative response to visual obstruction. Henslow's Sparrows responded positively to grass cover.

In our study area, developed cover ranged from 2.4 to 76.9% (mean = $37.3 \pm 23.9\%$ SD), grass cover ranged from <0.01 to 12.2% (mean = $0.06 \pm 0.04\%$ SD), hay cover ranged from <0.01 to 24.6% (mean = $0.08 \pm 0.07\%$ SD), forest cover ranged from 2.0 to 27.9% (mean = $11.2 \pm 7.2\%$ SD), and patch size ranged from 3.0 to 400.0 ha (mean = 159.4 ± 134.7 SD). Combined models were better than local-level models for all species (Table 3). Development was in the top model for one of the seven species (Savannah Sparrows), with four species positively associated with development and three species negatively associated (Fig. 2). Confidence intervals of the development parameter did not include zero for Savannah Sparrows and Sedge Wrens, indicating a strong association with this variable (Table 4). Forested cover was in the top model for three of the seven species (Grasshopper Sparrows, Henslow's Sparrows, and Savannah Sparrows), with four species positively associated and three species negatively associated with forested cover (Fig. 3). Confidence intervals of the forest parameter did not include zero for Bobolinks and Henslow's Sparrows (Table 4). Grass cover was in the top model for two of the seven species (Bobolinks and Sedge Wrens), with four species positively associated and three species negatively associated with grass cover. Confidence intervals of the grass parameter did not include zero for Sedge Wrens (Table 4). Hay cover was in the top model for one of the seven species (Sedge Wrens), with three species positively associated and four species negatively associated with hay cover. Confidence intervals of the hay parameter did not include zero for Sedge Wrens (Table 4). Patch size was in the top model for every species, with all seven species positively associated to patch size except for Sedge Wrens (Fig. 4). Confidence intervals did not include zero for Dickcissels, Eastern Meadowlarks, and Grasshopper Sparrows (Table 4).

DISCUSSION

Of the seven focal grassland species I studied, only Sedge Wrens exhibited a strongly negative association to the amount of development in the landscape. Three species displayed a weakly positive association to development, while one species, the Savannah Sparrow, displayed a strongly positive association to amount of development. The mechanisms behind these responses are unclear. Northeast Illinois is a highly fragmented area and grassland habitat is embedded in a matrix dominated by a combination of row-crop agriculture and development. It is possible that both of these cover types are hostile to grassland birds and thus grassland birds are making decisions based on two non-preferred options, which may explain the weak responses to development. If this was the case, it might be expected that densities in my system would be lower overall than those in less-fragmented systems. However, my estimates are very similar to estimates from other studies (e.g., Horn and Koford et al. 2004, Lusnier 2004, Winter et al. 2006, Quammen 2007, Renfrew and Ribic 2008, Vogel 2011, Ellison et al. 2013), suggesting that urban grasslands are used at a similar intensity to grasslands located in systems containing greater amounts of grassland cover. While the use of density as an indicator of habitat quality is not always reliable (Bock and Jones 2004), I have found that nest predation rates in urban grasslands are lower than in rural grasslands (Chapter 3) which supports my use of density as an indicator of habitat quality. Further research is needed to determine grassland bird response to development in other regions and to elucidate the mechanisms behind species avoidance of or attraction to habitat in developed areas.

Bobolinks and Henslow's Sparrows were positively associated with forest cover, which is surprising given that several studies have documented negative effects of woody vegetation on these species (Ribic and Sample 2001, Grant et al. 2004, Quammen 2007). While at least five

studies have consistently found negative relationships between woody vegetation and occupancy or abundance of Grasshopper Sparrows, Savannah Sparrows, and Bobolinks (Bakker 2003), a few examples of positive relationships have also been documented (e.g., Murray et al. 2008). In this study, I considered that the positive relationship I observed with forest cover may have been an artifact of a relationship with another correlated variable. While I found no strong correlations between forest cover and any of my measured variables, it is possible that there may be an unmeasured variable causing this relationship. It's also possible that sites surrounded by more trees are more productive and birds are using productivity to assess habitat quality and make settlement decisions. However, I found no evidence of a relationship between forest cover and reproductive success (VLB, Chapter 3).

Associations with hay cover and grass cover were mixed, with no consistent responses among species. The majority of species did not show strong associations with these variables, and confidence intervals for parameter estimates were consistently large because of the skewed distribution of these variables (see Table 1). Grass cover is extremely limited in the greater Chicago area, with the majority of sites surveyed containing less than <5 % grass cover in a 1.6 km radius. While surrounding grass cover may have a strong influence on density in other systems, it is likely not prevalent enough in this system to have a significant effect.

As documented in other studies, larger patches resulted in greater densities for the majority of species, and appeared in top models for all species. Thus it appears that patch size plays an important role in developed landscapes just as it does in more rural landscapes. While I did not consider interactions between patch size and landscape variables in my models, preliminary investigations revealed that interactions were not important for most species.

However, there was some suggestion that patch-size effects were more pronounced for Eastern Meadowlarks in more developed landscapes.

There are other variables known to affect grassland bird density that were not included in this study, including burn history and the species composition of grasses (e.g., cool- vs. warm-season species). I lacked this information for some sites, but examined effects for the subset of sites with complete data. I found that these variables affected densities of some species, but they did not change the landscape-level relationships I observed. However, the importance of these variables for some species, as well as the importance of local-level variables included in this analysis, underscores the importance of using a hierarchical approach in modelling species density. This illustrates that both local-level and landscape-level variables are important in grassland bird habitat selection, and both scales must be considered when making management decisions.

Urban grasslands appear to provide valuable habitat for breeding grassland birds, including Henslow's Sparrows, Dickcissels, and Grasshopper Sparrows, which are listed as species of continental importance by Partners in Flight (Rich et al. 2004), and Savannah Sparrows, Bobolinks, and Eastern Meadowlarks, listed as species in need of management action in Illinois and surrounding states (PIF Science Committee 2012). Because development does not appear to deter the majority of grassland species, managers in urban landscapes should consider creating or conserving habitat for grassland birds if the opportunity arises. Bigger patches may be preferred over smaller patches because they allow for increased habitat heterogeneity and greater numbers of birds, though evidence suggests that even small patches in developed areas have high reproductive success (VLB, Chapter 3) and thus may also be useful to breeding grassland birds.

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TABLES AND FIGURES

Table 1. Descriptions of variables used to predict densities of grassland bird species in northeast Illinois, 2012-2013.

Model Set/Variable Code	Description	\bar{x}	SE	Range
Local				
VOR	Visual obstruction reading measured using a Robel pole	5.71	0.24	0.81–17.63
Dead veg. height	Height of dead vegetation within a quadrat (in cm)	22.31	2.42	0–152.25
Litter depth	Depth of litter measured using a ruler (in mm)	2.13	0.20	0–24.75
%Grass	Percent cover of grass within a quadrat	41.07	1.69	0–98.75
%Forb	Percent cover of forbs within a quadrat	35.02	1.63	0–98.75
%Bare	Percent cover of bare ground within a quadrat	5.59	0.56	0–45.00
%Litter	Percent cover of litter cover within a quadrat	12.45	0.87	0–55.00
%Dead	Percent cover of standing dead vegetation within a quadrat	3.54	0.51	0–38.75
Landscape				
Patch size	Size of patch (ha)	159.41	9.99	3–400
Developed	Percent cover of developed land within a 1.6 km radius	37.28	0.02	2.44–76.91
Grassland	Percent cover of grassland within a 1.6 km radius	0.06	0.04	0.01–12.22
Hay	Percent cover of hay and pasture within a 1.6 km radius	0.08	0.07	0.01–24.55
Forest	Percent cover of forested land within a 1.6 km radius	11.17	0.01	2.04–27.87

Table 2. Total number of observations (n), percentage of sites and points each species was observed at, and average density (birds/ha) of focal species in northeast Illinois grasslands, 2012-2013.

Species	n	% Of Sites Observed At	% Of Points Observed At	Average Density \pm SE (birds/ha)
Bobolink (<i>Dolichonyx oryzivorus</i>)	780	76	69	0.95 \pm 0.10
Dickcissel (<i>Spiza americana</i>)	869	80	89	0.97 \pm 0.07
Eastern Meadowlark (<i>Sturnella magna</i>)	890	80	93	0.64 \pm 0.03
Grasshopper Sparrow (<i>Ammodramus savannarum</i>)	247	66	55	0.65 \pm 0.08
Henslow's Sparrow (<i>Ammodramus henslowii</i>)	361	63	60	0.95 \pm 0.10
Savannah Sparrow (<i>Passerculus sandwichensis</i>)	536	86	77	1.42 \pm 0.15
Sedge Wren (<i>Cistothorus platensis</i>)	277	70	63	0.45 \pm 0.06

Table 3. Number of parameters, ΔAIC_c values, and model weights for best local models and local and landscape combinations models. Variables include: the local model (local), developed cover (D), forested cover (F), grass cover (G), hay cover (H), and patch size (P). The best local model was based on the model with the minimum ΔAIC_c in the local-level model set (Appendix B, Table1). Models with the minimum ΔAIC_c are emboldened for each species.

Model	K	BOBO ^a		DICK		EAME		GRSP		HESP		SAVS		SEWR	
		ΔAIC_c	w_i	ΔAIC_c	w_i	ΔAIC_c	w_i	ΔAIC_c	w_i	ΔAIC_c	w_i	ΔAIC_c	w_i	ΔAIC_c	w_i
Constant	1	78.3	0.00	26.4	0.00	20.8	0.00	42.9	0.00	31.1	0.00	54.3	0.00	67.8	0.00
Local	2	53.0	0.00	2.4	0.08	14.9	0.00	24.9	0.00	5.1	0.02	38.5	0.00	24.0	0.00
Local+D	3	43.3	0.00	3.1	0.06	16.3	0.00	29.1	0.00	8.2	0.01	22.1	0.00	4.5	0.08
Local+F	3	54.5	0.00	5.1	0.02	17.7	0.00	28.4	0.00	0.9	0.20	16.3	0.00	27.3	0.00
Local+G	3	56.7	0.00	6.6	0.01	11.9	0.00	24.1	0.00	1.4	0.15	34.8	0.00	18.4	0.00
Local+H	3	37.5	0.00	5.2	0.02	17.4	0.00	26.3	0.00	4.5	0.03	26.7	0.00	6.5	0.03
Local+P	3	20.6	0.00	0.0^b	0.26	0.0^b	0.30	4.0	0.10	6.8	0.01	40.1	0.00	28.2	0.00
Local+D+F	4	43.6	0.00	4.8	0.02	18.3	0.00	32.6	0.00	1.8	0.13	55.3	0.00	8.5	0.01
Local+D+P	4	23.6	0.00	0.1	0.25	0.1	0.29	7.5	0.02	10.8	0.00	23.2	0.00	7.8	0.02
Local+F+P	4	20.4	0.00	2.7	0.07	2.9	0.07	0.0^b	0.72	0.0^b	0.31	18.5	0.00	31.6	0.00
Local+G+P	4	15.0	0.00	3.6	0.04	1.8	0.12	6.5	0.03	5.4	0.02	38.2	0.00	18.4	0.00
Local+H+P	4	13.4	0.00	3.2	0.05	2.9	0.07	8.1	0.01	7.2	0.01	28.9	0.00	10.8	0.00
Local+G+H+P	4	0.0^b	1.00	6.7	0.01	4.9	0.03	10.4	0.00	5.2	0.02	30.2	0.00	0.0^b	0.80
Local+D+F+P	5	23.0	0.00	1.8	0.11	1.9	0.12	3.5	0.12	2.7	0.08	0.0^b	1.00	5.5	0.05

^aSpecies names abbreviated according to American Ornithologists' Union four-letter code. BOBO = Bobolink; DICK = Dickcissel; EAME = Eastern Meadowlark; GRSP = Grasshopper Sparrow; HESP = Henslow's Sparrow; SAVS = Savannah Sparrow; SEWR = Sedge Wren.

^bMinimum AIC_c BOBO = 546.0; Minimum AIC_c DICK = 394.4; Minimum AIC_c EAME = 137.6; Minimum AIC_c GRSP = 462.6; Minimum AIC_c HESP = 581.4; Minimum AIC_c SAVS = 656.8; Minimum AIC_c SEWR = 369.4

Table 4. Model-averaged parameter estimates and 95% confidence intervals for landscape-level variables used to predict species density.

Model	BOBO			DICK			EAME			GRSP		
	β	95% CI		β	95% CI		β	95% CI		β	95% CI	
Intercept	0.884	0.541	1.227	0.800	0.536	1.064	0.533	0.394	0.673	0.528	0.269	0.788
Developed	0.496	-0.915	1.907	-0.853	-1.802	0.095	-0.398	-0.839	0.043	0.172	-0.260	0.605
Forest	4.350	0.492	8.207	-2.329	-5.444	0.787	-1.088	-2.530	0.355	1.542	-0.763	3.847
Grass	-1.684	-11.642	8.275	-2.636	-10.119	4.847	2.770	-0.563	6.103	1.036	-4.890	6.962
Hay	-2.697	-6.612	1.218	1.949	-1.510	5.407	0.681	-0.947	2.309	-0.791	-3.590	2.008
Patch size	0.000	-0.002	0.002	0.002	0.000	0.003	0.002	0.001	0.003	0.003	0.001	0.004
Model	HESP			SAVS			SEWR					
	β	95% CI		β	95% CI		β	95% CI				
Intercept	0.899	0.539	1.258	1.372	0.745	1.999	0.461	0.264	0.658			
Developed	0.609	-0.485	1.702	2.411	0.141	4.682	-0.744	-1.232	-0.257			
Forest	5.398	1.715	9.082	-1.587	-8.588	5.413	0.342	-1.215	1.899			
Grass	8.072	-0.162	16.306	-6.224	-22.852	10.405	3.442	0.665	6.220			
Hay	-2.659	-6.755	1.437	-6.014	-13.657	1.628	2.110	0.821	3.399			
Patch size	0.001	-0.063	0.065	0.001	-0.003	0.005	0.000	-0.001	0.000			

^aSpecies names abbreviated according to American Ornithologists' Union four-letter code. BOBO = Bobolink; DICK = Dickcissel; EAME = Eastern Meadowlark; GRSP = Grasshopper Sparrow; HESP = Henslow's Sparrow; SAVS = Savannah Sparrow; SEWR = Sedge Wren.

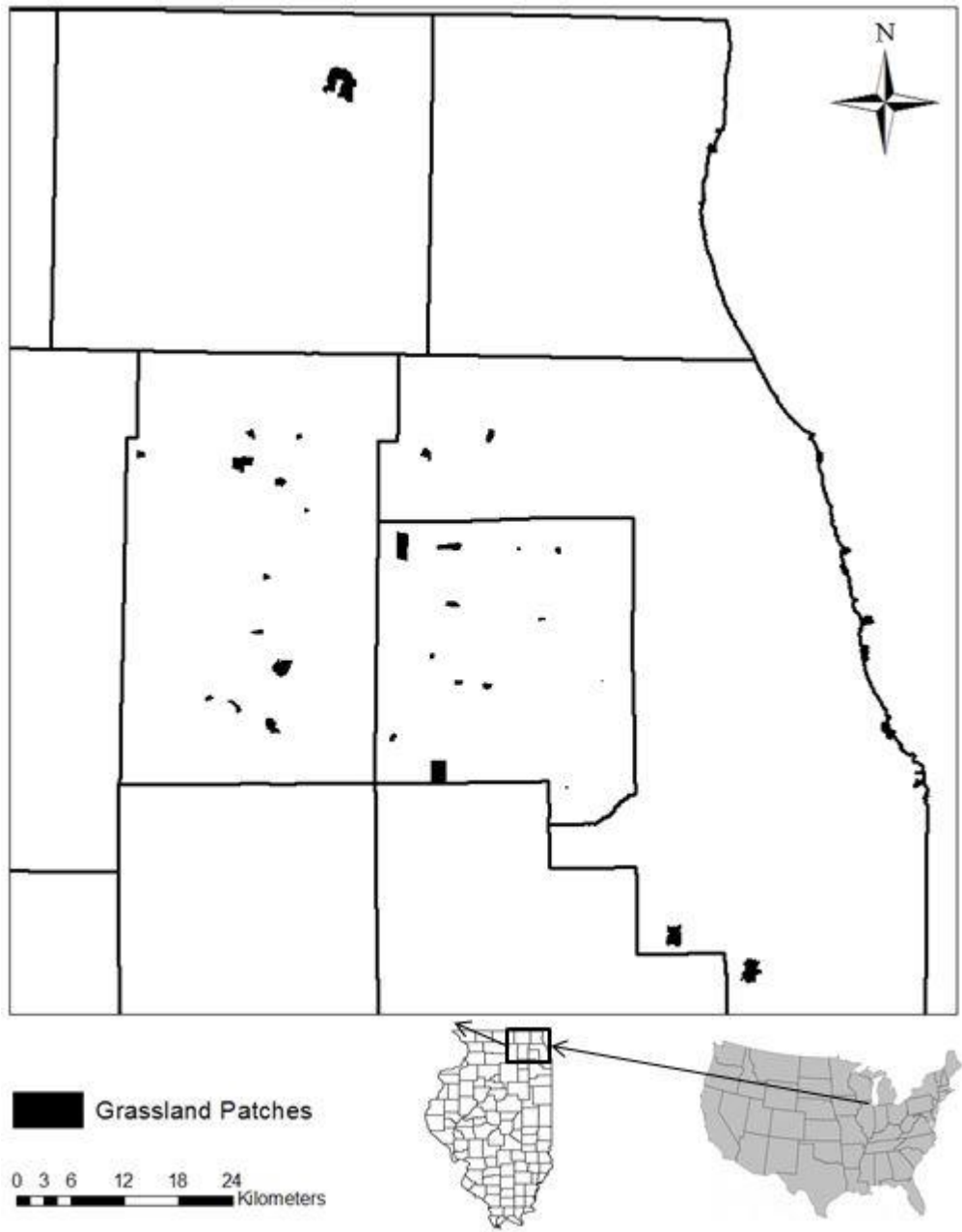


Figure 1. Map of northeastern Illinois displaying grassland patches where point counts were conducted in 2012 and 2013.

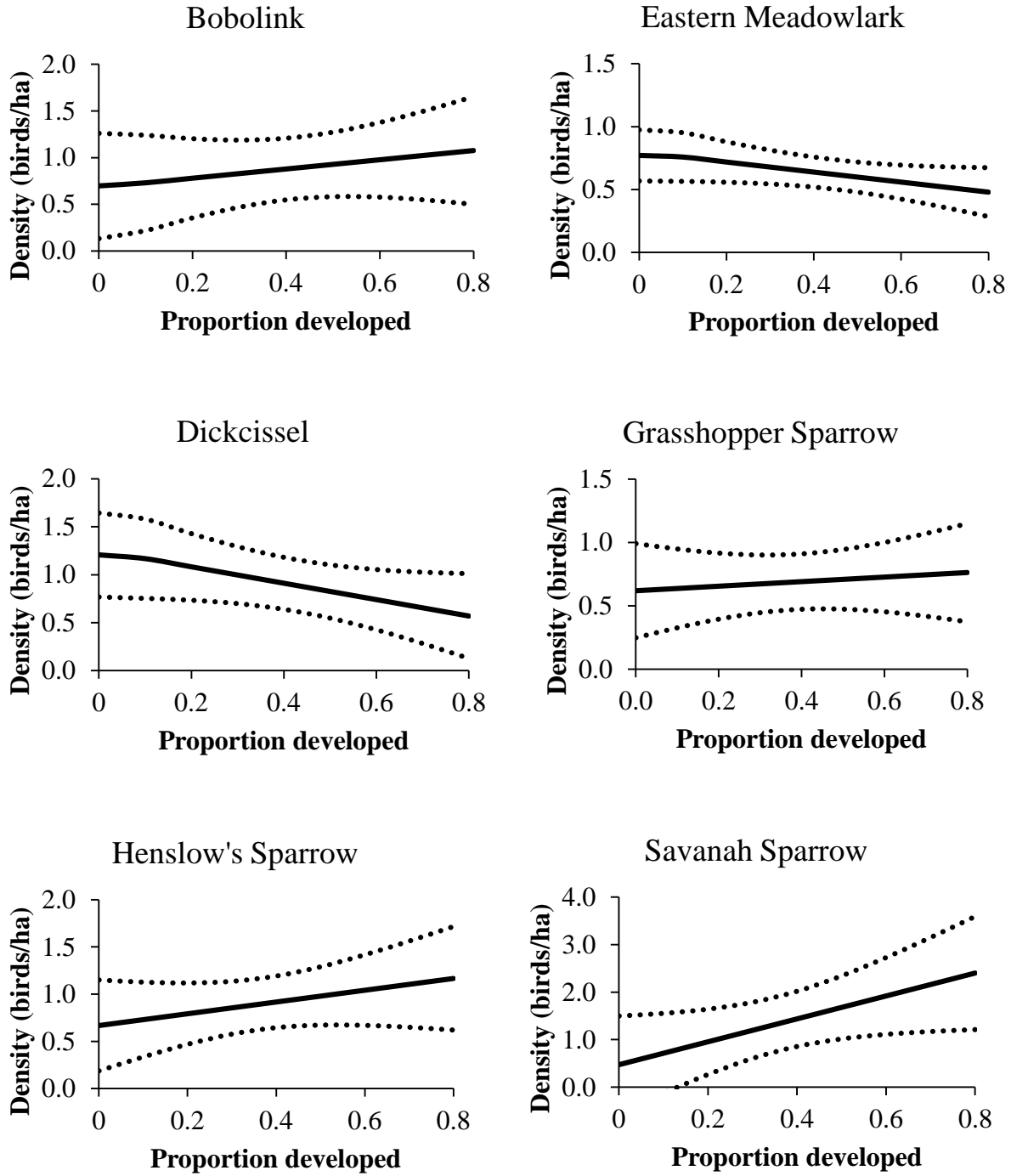


Figure 2. Model-averaged density (birds/ha) and 95% confidence intervals in relation to the proportion of developed cover in the landscape (1600 m) for the seven focal species.

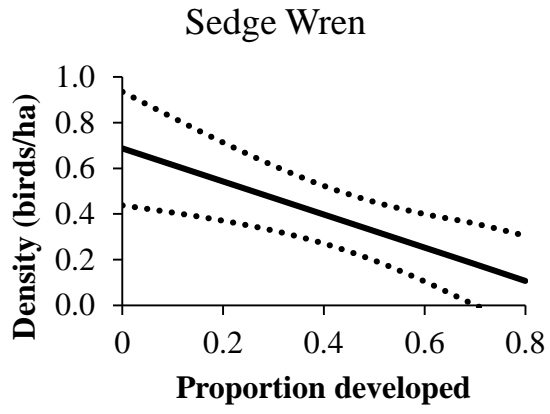


Figure 2 (cont.)

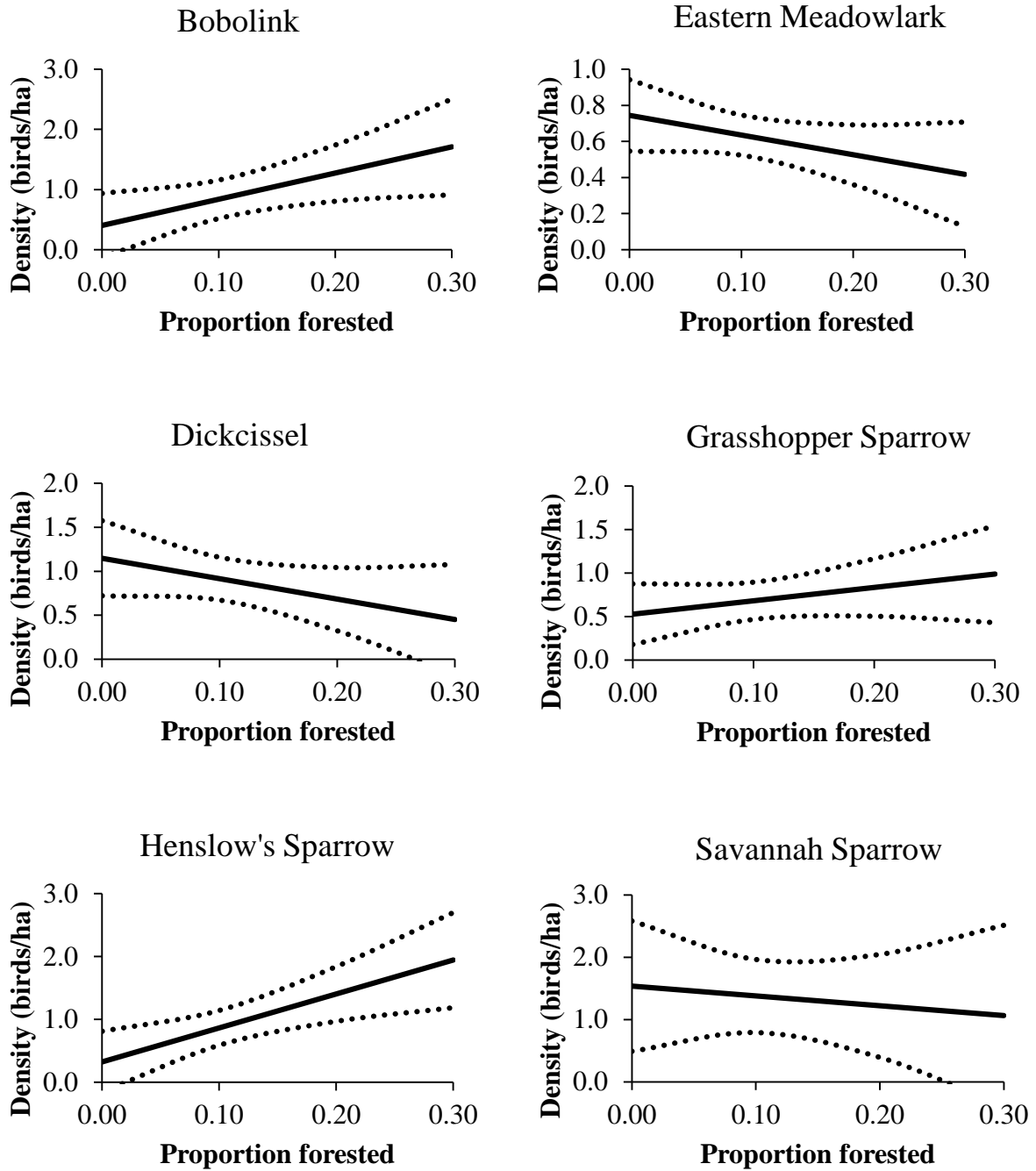


Figure 3. Model-averaged density (birds/ha) and 95% confidence intervals in relation to the proportion of forest cover in the landscape (1600 m) for the seven focal species.

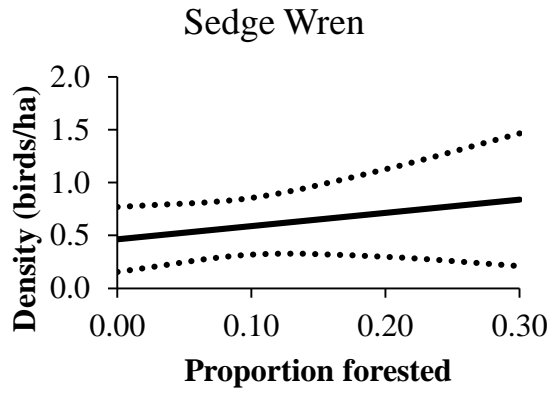


Figure 3 (cont.)

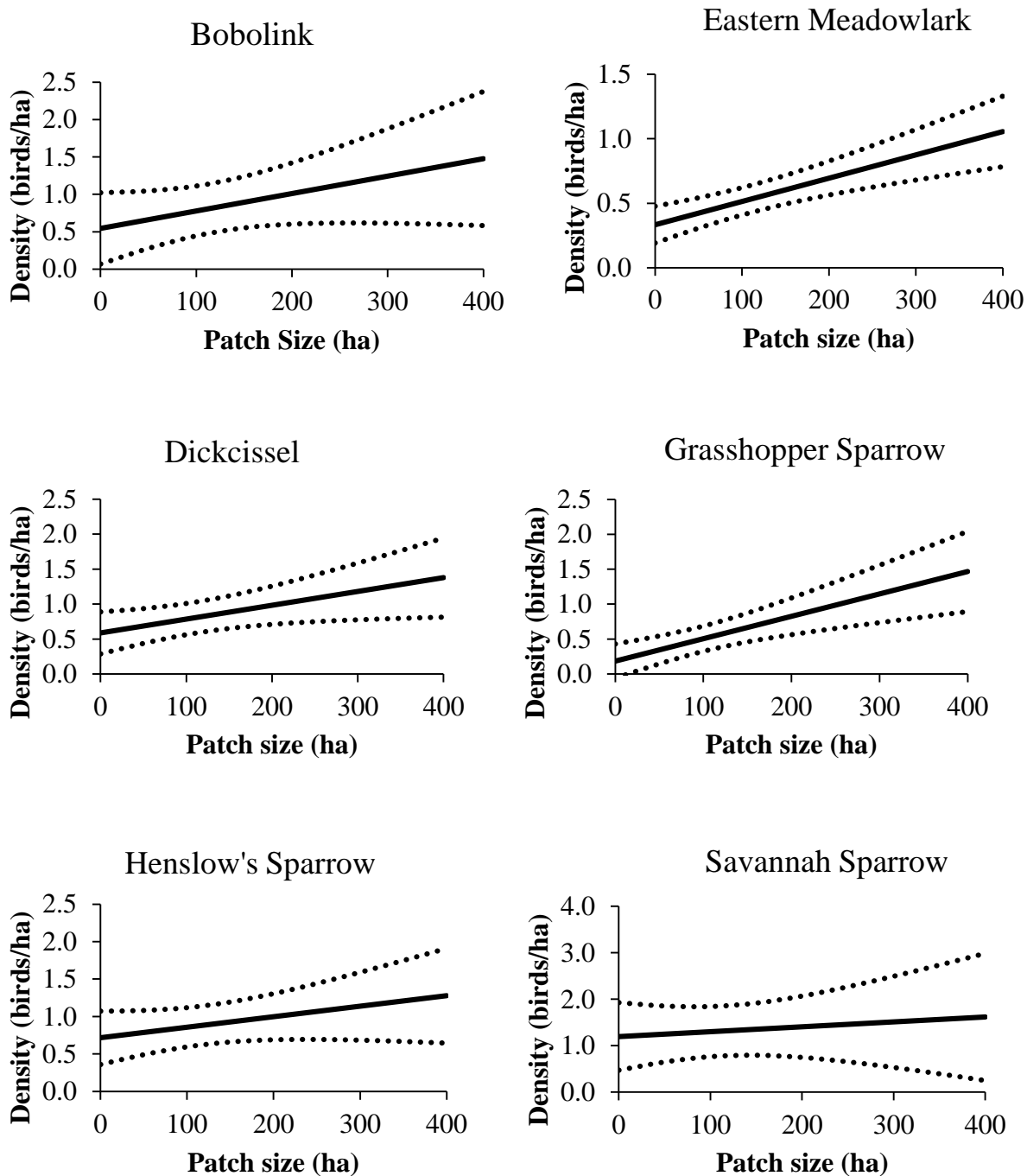


Figure 4. Model-averaged density (birds/ha) and 95% confidence intervals in relation to patch size (ha) for the seven focal species.

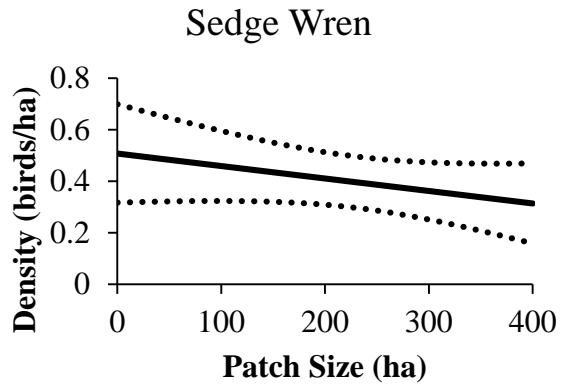


Figure 4 (cont.)

CHAPTER 3

NEST SURVIVAL AND BROOD PARASITISM OF GRASSLAND BIRDS ON AN URBAN-TO-RURAL GRADIENT

ABSTRACT

The large expanses of tallgrass prairie that once dominated the Midwestern U.S. have been lost and fragmented as a result of agricultural intensification. Structurally diverse pastures and hayfields have also been replaced by monocultures of corn and soybeans. These habitat declines have caused corresponding decreases in grassland bird populations. To stem these declines, conservation efforts have centered on conserving large patches of grassland in landscapes containing minimal amounts of woody vegetation and development. Because of the perceived negative influence of development, habitat located in developed landscapes has been assumed to be of poor quality. However, this assumption has yet to be tested and may wrongly devalue grasslands located in an urban matrix. To examine the influence of development on habitat quality for grassland birds, I studied grassland bird nest predation and brood parasitism in patches of varying size along an urbanization gradient in northeastern Illinois. Because nest predation is the primary cause of reproductive failure and a potentially limiting factor for grassland bird populations, I used miniature video cameras to identify predators at a subset of nests. From 2012 to 2013, I monitored 432 nests of 16 grassland bird species. Nest predation rates decreased with urbanization in the landscape as did probability of brood parasitism. I filmed 38 nests and documented 19 predation events. I found that coyotes (*Canis latrans*), white-tailed deer (*Odocoileus virginianus*), and thirteen-lined ground squirrels (*Ictidomys*

tridecemlineatus) were important nest predators in my study system. My results suggest that grasslands located in urban landscapes provide valuable habitat for breeding grassland birds.

INTRODUCTION

Grassland bird populations have steadily declined throughout North America for several decades (Sauer et al. 2009). These declines have been particularly severe in the Midwest where many types of grasslands have been lost, fragmented, and surrounded by unsuitable habitat (Herkert 1995). The loss and alteration of grasslands is primarily attributed to increases in agricultural intensification (Warner 1994). Structurally diverse hayfields and pastures that once existed throughout the Midwest have also been replaced by monocultures of corn and soybeans, further reducing habitat for grassland birds (Warner 1994).

To curtail grassland bird population declines, conservation efforts have focused on protecting large core grassland patches surrounded by a matrix with substantial grassland cover and minimal woody vegetation (Sample and Mossman 1997). These principles have been conceptually formalized under the “grassland bird conservation area” (BCA) model. However, numerous studies testing the traditional BCA assumptions have found that grassland birds respond inconsistently to variation in patch size and landscape composition (Davis et al. 2006, Winter et al. 2006), and nest survival is not consistently associated with these factors (Walk et al. 2010, Benson et al. 2013). More recently, the BCA model has been modified to account for other land cover types, including urban land cover (Johnson et al. 2010). Urban land cover is assumed to be hostile to grassland birds because of increased human activity and anthropogenic noise, and habitat located in urban landscapes is subsequently assumed to be of poor quality.

While many principles of the BCA have been examined, the assumption that urban land cover is hostile to grassland birds remains untested.

Despite the lack of studies on grassland bird nest survival in urban landscapes, work in forests suggests greater nest survival in urban compared to rural landscapes (Rodewald et al. 2011). These differences in survival rates are generally attributed to differences in nest predation rates (Rodewald et al. 2011), possibly because of landscape-mediated differences in the predator community (e.g., Chalfoun et al. 2002, Rodewald et al. 2011). To improve understanding of nest survival dynamics, it is important to have knowledge of which predators are depredating nests (Benson et al. 2010). As development continues to alter landscapes, the need to determine the relationships among grassland birds, predator communities, and urbanization becomes particularly important (Ribic et al. 2009). Ultimately, a more informed understanding of predator-prey dynamics in these systems will provide insight into the value of urban grasslands.

To investigate whether developed landscapes provide poor-quality habitat for grassland birds, I examined reproductive success in nine grasslands situated along a rural to urban gradient in northeast Illinois, a region with a significant number of grassland patches in a highly urbanized matrix. Specifically, I examined how landscape-level variables, including amount of developed land, and patch size affected nest predation rates and brood parasitism by Brown-headed Cowbirds (*Molothrus ater*). Additionally, I monitored a subset of nests with small video cameras to determine which predators were responsible for nest failure.

METHODS

Study area

I searched for and monitored grassland bird nests in nine study sites near Chicago in northeastern Illinois (Fig. 5). I focused on five sites in 2012. I replaced three of these sites in 2013 because burning and mowing in spring 2013 made the sites less suitable for video monitoring of nests (i.e. sparser and shorter vegetation). I also added a site in 2013, monitoring nests at a total of six sites. Sites were located in Cook, DuPage, Kane, and McHenry Counties (Appendix C). Cook and DuPage counties are highly urbanized while Kane and McHenry counties are comparatively more rural. All sites were located on natural areas maintained by the county forest preserve districts and consisted of a grassland patch within a preserve. Patches ranged from 23 to 242 ha (mean = 97.7 ± 69.3 ha SD). If trees extended across at least 75% of the grassland patch and tree cover was at least 20 m wide, I considered this to be a site edge (Winter et al. 2006). I considered agricultural fields and two- and four-lane roads with disturbed roadsides to be site edges, whereas trails or roads without disturbed roadsides were not considered edges (Winter et al. 2006). In the urbanized counties, lawns often constituted edges of patches.

Nest searching and monitoring

I searched for nests in grassland patches from May to July in 2012 and 2013. Nest searching was typically conducted between 09:30 and 16:00 CDT. Search effort was consistent among sites and included a combination of systematic rope dragging and haphazard walking. Upon locating a nest, I recorded GPS coordinates and placed flagging at least 5 m away from the nest. I recorded nest contents, including presence of cowbird eggs or nestlings, every three days

until the nest fledged or failed, and I was cautious to minimize disturbance at the nest site. I considered a nest active if at least one egg was present and the female was observed in or directly around the nest, and I considered a nest successful if at least one young fledged. If I no longer observed the nestlings in the nest prior to the expected fledge date, and there were obvious signs of predation such as egg shells, feathers, or dead nestlings, then I considered the nest to have failed. If fledging could not be confirmed, I eliminated the last nest check from analysis.

Video monitoring

I placed small video cameras at a subset of nests to determine which predators were responsible for nest failure. Camera systems consisted of one small security camera, a digital video recorder, and a deep cycle battery. The battery and digital video recorder, connected to the camera with a 10-m cable, were housed in a weatherproof plastic bin (see Cox et al. 2012 for details). I mounted cameras on dowels and placed them at least 0.3 m away from the nest. I camouflaged the plastic bin, camera, and dowel using green and brown spray paint, and I concealed cameras with surrounding vegetation. I only placed cameras on nests that birds had started incubating, and I reviewed footage within one hour of camera installation to ensure that the bird returned to the nest. If the bird had not returned, I removed the camera system.

Landscape metrics

I quantified landscape composition surrounding each site in ArcGIS 10.0 (ESRI 2011) using data from the 2012 National Agricultural Statistics Service Cropland Data Layer. Land cover was classified according to the categories of low-density developed (impervious surfaces 20%-49%), medium-density developed (impervious surfaces 50%-79%), high-density developed

(impervious surfaces 80%-100%), forested land, and cropland. For this analysis, I combined all three categories of development together to obtain a single proportion of development for each site. Using Geospatial Modelling Environment (Beyer 2012), I obtained proportions of each land cover variable in buffers of 800 m, 1600 m, and 2500 m around each site. I used multiple buffer sizes because of uncertainty about the most appropriate scale for use in the nest predation analysis. I also measured the distance from each nest to the nearest patch edge using ArcGIS (ESRI 2011).

Statistical analyses

I modeled daily predation probability using the logistic-exposure method in SAS PROC GENMOD (Shaffer 2004). In this analysis I included only nests that were successful or depredated. I excluded nests that failed due to other causes (e.g., mowing, weather, etc.). I used an information theoretic approach to determine which variables had the most influence on daily predation rate (DPR). These variables included nest stage (incubation or nestling), day of season, a quadratic effect of day of season, patch size (ha), distance to nearest edge (m) and landscape cover (proportion cropland, development, and forested). Because the linear effect of day of season better explained nest predation probability than the quadratic effect, I only retained the former in subsequent models. I conducted preliminary analyses to determine which buffer size was most appropriate, and incorporated only models with landscape cover within 2.5 km in subsequent models. I examined correlations among variables and found strong correlations between patch size and crops ($r = 0.83$), patch size and forest ($r = -0.79$), and crops and forest ($r = -0.67$), thus these variables were not used together in the same models. I developed candidate models using combinations of variables that I thought may influence nest predation likelihood.

This resulted in a suite of 13 different models, including a constant model of nest predation probability. I also examined a model containing an effect of species identity, as well as a model differentiating ground nesters from above-ground nesters, but these models were dropped because there was little support for DPR differences among species. I used logistic regression to examine predictors of brood parasitism and incorporated the same explanatory variables used in the nest predation analysis in the brood parasitism model set with the exception of nest stage. I only included species that are known hosts for Brown-Headed Cowbird eggs in the parasitism analysis. I ranked candidate models using Akaike's Information Criterion adjusted for small sample size (AIC_c) and calculated model weights (w_i) (Burnham and Anderson 2002). Because there was model selection uncertainty, I used model averaging for parameters and predicted values and their 95% confidence intervals (Burnham and Anderson 2002).

RESULTS

I found and monitored 432 nests of 16 species in 2012 ($n = 215$) and 2013 ($n = 217$). Red-winged Blackbirds (*Agelaius phoeniceus*) ($n = 263$) and Dickcissels (*Spiza americana*) ($n = 86$) were the most commonly found nests in both years (Table 5). Savannah Sparrows (*Passerculus sandwichensis*), Song Sparrows (*Melospiza melodia*), and Bobolinks (*Dolichonyx oryzivorus*) also comprised a significant proportion of our sample ($n = 46$). Brood parasitism by Brown-headed Cowbirds occurred in <11% of nests ($n = 45$) for both years combined and 13% of parasitized nests were later abandoned ($n = 6$). In our nest predation analysis, I only included passerine nests that survived or failed due to predation, which limited the sample size to 351 nests (resulting in 3,969 exposure days). Other causes of nest failure included abandonment ($n = 40$), weather ($n = 10$), mowing by forest preserve district employees ($n = 10$), and nests tipping

over ($n = 5$). Nests that I was unable to find again were also not included in analysis ($n = 3$). I only included nests of passerines in our analysis because length of incubation and nestling periods was similar among species in this group ($n = 13$ non-passerine nests).

Nest predation

The model that best explained nest predation probability was the additive effect of the amount of developed land in a 2500-m buffer and patch size (Table 6). The top three models accounted for >99% of the weight, with patch size found in all of the top models. DPR was negatively related to amount of development within 2500 m ($\beta = -0.628$, $SE = 0.316$, 95% CI = -0.008 , -1.247) and patch size ($\beta = -0.005$, $SE = 0.001$, 95% CI = -0.003 , -0.007). DPR was positively related to distance from the edge of a patch, though 95% confidence intervals included zero ($\beta = 0.001$, $SE = 0.001$, 95% CI = -0.003 , 0.001). The model containing proportion of forest cover ranked higher than the constant predation model but had little support ($\beta = 2.162$, $SE = 1.301$, 95% CI = -4.727 , 0.403). Models with proportion of cropland, day of season, and nest stage had less support than the constant model.

Extrapolating daily survival rate out into a 23 day nesting cycle (the average for Red-winged Blackbirds and Dickcissels), overall nest success was approximately 24% ($SE = 0.4\%$). In the smallest and largest patches, overall nest success was approximately 14% and 46%, respectively. Overall nest success in the most rural and most developed site was approximately 19% and 32%, respectively.

Brood parasitism

The model that best explained brood parasitism was the additive effect of the amount of developed land in a 2500-m buffer and patch size (Table 7). The top three models accounted for 76% of the weight, with development found in all three models. Probability of parasitism was negatively related to development ($\beta = -2.862$, SE = 0.967, 95% CI = -4.757, -0.967), patch size ($\beta = -0.007$, SE = 0.003, 95% CI = -0.013, -0.001), and distance to edge ($\beta = -0.002$, SE = 0.002, 95% CI = -0.006, 0.002).

Nest predators

I video-monitored 38 nests, primarily Red-winged Blackbirds ($n = 27$) and Dickcissels ($n = 9$), and documented 19 predation events. In four cases the camera was knocked over before predation occurred and I was unable to identify the predator responsible. I observed one nest that was depredated by both a Brown-headed Cowbird and a Thirteen-lined ground squirrel (*Ictidomys tridecemlineatus*) and I considered these separate predation events. I recorded another nest where a North American Racer (*Coluber constrictor*) force-fledged all of the nestlings but was unable to capture any of the nestlings. White-tailed deer (*Odocoileus virginianus*) ($n = 4$) and coyotes (*Canis latrans*) ($n = 4$) were most frequently documented as nest predators, though weasels (*Mustela sp.*), striped skunks (*Mephitis mephitis*), and avian predators were also recorded (Table 8). Across the urban to rural gradient, predation events by a weasel and a Brown-headed Cowbird were only documented in the more rural sites, while predation by an unidentified hawk was only documented in a more urban site. Deer and coyotes were documented in almost equal numbers in both urban and rural sites.

DISCUSSION

I found that the amount of development in the landscape influenced nest predation rates as well as brood parasitism. Contrary to the assumptions about the negative influence of developed land surrounding grasslands, nests in urban landscapes had lower nest predation rates than nests in more rural landscapes (Fig. 6). Moreover, brood parasitism was also less frequent in more urbanized landscapes (Fig. 7). Point-count data from the same grassland patches shows that urban grasslands also support greater densities of some species of grassland birds, further supporting the importance of these urban grasslands (see Chapter 2). These results challenge the previously untested assumption that developed landscapes are hostile to grassland birds, and show that urban grasslands in northeastern Illinois actually have lower nest predation rates (and thus greater nest survival) than grasslands surrounded primarily by row-crop agriculture. In fact, the overall nest survival rate of 24% in this study is very similar to estimates reported in previous studies (generally ranging from 19% to 30%), while the estimate of 32% success in the most urban grassland is greater than many reported estimates (Benson et al. 2013).

Because nest predation was greater in rural landscapes, as has been found in some other studies (e.g., Rodewald et al. 2011), one might initially assume that rural sites harbor more nest predators than urban sites, resulting in the lower nest success I observed in the rural sites. However, predator abundance is often greater in urban areas (Chamberlain et al. 2009). This disconnect between predation rates and predator abundance in urban settings, termed the “urban nest predator paradox”, may have several potential explanations (Shochat et al. 2006, Stracey 2011) including a loss of important predators in urban areas, and prey-switching or specialization by urban predators (Stracey 2011).

While video-monitoring a subset of nests, most predators that I documented appeared in both urban and rural landscapes. Even though my limited sample size precludes me from making substantive conclusions regarding predator abundance and activity in relation to the urban nest predator paradox, it is noteworthy that I only observed cowbird and weasel predation events in the rural sites, and a concurrent study on shrublands has found raccoons (*Procyon lotor*) depredating nests in rural but not urban forest preserves (S.J. Chiavacci, University of Illinois, unpublished data). In addition, I observed that brood parasitism by cowbirds was greater in rural areas compared to urban areas.

Evidence from radio-telemetry and mark-recapture studies suggests, however, that raccoons, as well as striped skunks and coyotes, are frequent residents of natural areas in urban landscapes (Gehrt 2004). Consistent with a tenet of the urban nest predator paradox, greater population densities of raccoons and coyotes have been observed in urban compared to rural sites in metropolitan Chicago (Prange et al. 2003, Gehrt 2004). However, raccoons are frequently observed exploiting artificial food resources and high raccoon density in urban areas is often attributed to the abundance of these resources (Prange et al. 2004). Though coyotes appear to be less reliant on human-provided subsidies (Morey et al. 2007), it is possible that their main prey in urban areas—small rodents—are more abundant because of the increased subsidies. Thus coyotes and other predators may specialize on food sources other than bird nests, and this could explain the lower nest predation rates observed in urban landscapes (Rodewald et al. 2011, Fischer et al. 2012). Beyond the abundance or activity of any given predator species, higher nest predation rates in rural grasslands may result from having a more diverse predator community in these areas (e.g., McKinney 2008). Increased predator diversity in rural landscapes may result in a greater probability a nest is discovered and depredated in these areas.

Beyond landscape-level effects of development, I also found evidence for an effect of patch size on nest predation and brood parasitism. Nest predation decreased (Fig. 8), as did brood parasitism (Fig. 9), with patch size, supporting the notion that bigger patches are better than smaller patches for grassland birds. However, most studies that have examined the relationship between patch size and nest survival have failed to find a relationship, and patch size does not appear to be a generalizable predictor of nest survival for grassland birds (Benson et al. 2013). The inverse relationship between brood parasitism and patch size, on the other hand, does appear to be consistent among studies (Benson et al. 2013). In this study, one particularly large patch (219 ha) was driving the observed relationship for nest predation, and the relationship did not remain across the remaining patch sizes (23–125 ha) when this large patch was removed from the analysis. The relationship between patch size and brood parasitism, however, was still apparent after the removal of this large patch from analysis. Nonetheless, the relationship between nest predation likelihood and patch size may reflect a real effect of large patches, and I recommend that further research be done to elucidate the effects of patch size in other developed landscapes.

I also observed slightly lower predation rates for nests closer to patch edges (Fig. 10). While nest predation is often assumed to be greater near edges, previous research has not led to generalizable patterns (i.e. positive or negative effects of edges), and is likely to depend on the predators present in the study system (Benson et al. 2013). Other studies reporting greater nest survival near edges have attributed this to the prevalence of thirteen-lined ground squirrels, an interior grassland nest predator (Grant et al. 2006, Ribic et al. 2012). Based on my camera data, ground squirrels are nest predators in this study system and they may be driving the trend of lower predation rates near edges. I also observed that the probability of brood parasitism slightly

increased with proximity to patch edges (Fig. 11), a pattern which appears to be relatively consistent across grassland bird studies (Benson et al. 2013). Future research on movements of nest predators, such as ground squirrels, using radiotelemetry will help clarify the relationship between predators, edges, and nest predation.

This research shows that grasslands in urban areas provide valuable breeding habitat for grassland birds. Few studies have considered landscape-level predictors of grassland bird nest predation, and none of these studies have been in an urbanized landscape. This general lack of focus on landscape-level variables is surprising given the known influence of landscape context on predator community assemblages. As development continues to expand, understanding the links between landscape context, predator communities, and nest survival will continue to become important for birds breeding in grasslands as well as other cover types embedded in these urban landscapes. Currently, little is known about how processes operate in these areas. The results of my study contradict the recommendations put forth for grassland bird conservation areas (BCAs) with respect to developed areas (Johnson et al. 2010). This suggests that developed land should not necessarily be viewed as hostile when designating BCAs, and that the typical BCA model may not be applicable in urban areas and new guidelines should be considered in these landscapes.

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TABLES AND FIGURES

Table 5. Number of nests, number of sites nests found in, number and % of successful nests, number and % of depredated nests, and number and % of parasitized nests of all species found in northeast Illinois grasslands, 2012-2013.

Species	# of Nests	# Sites Found Nesting In	# Successful	%	# Depredated	%	# Parasitized	%
American Robin (<i>Turdus migratorius</i>)	1	1	0	0	1	100	1	100
American Woodcock (<i>Scolopax minor</i>)	1	1	0	0	1	100	0	0
Blue-winged Teal (<i>Anas discors</i>)	1	1	0	0	1	100	0	0
Bobolink (<i>Dolichonyx oryzivorus</i>)	12	4	4	33.3	8	66.7	0	0
Common Yellowthroat (<i>Geothlypis trichas</i>)	10	5	3	30.0	5	50.0	2	20.0
Dickcissel (<i>Spiza americana</i>)	86	6	19	22.1	49	56.9	14	16.3
Eastern Meadowlark (<i>Sturnella magna</i>)	7	3	3	42.9	3	42.9	0	0
Field Sparrow (<i>Spizella pusilla</i>)	1	1	0	0	1	100	1	100
Grasshopper Sparrow (<i>Ammodramus savannarum</i>)	3	3	2	66.7	1	33.3	1	33.3
Henslow's Sparrow (<i>Ammodramus henslowii</i>)	1	1	0	0	1	100	0	0
Killdeer (<i>Charadrius vociferous</i>)	2	2	0	0	2	100	0	0
Mallard (<i>Anas platyrhynchos</i>)	9	7	1	11.1	5	55.6	0	0
Red-winged Blackbird (<i>Agelaius phoeniceus</i>)	263	9	69	26.2	152	57.8	22	8.4
Savannah Sparrow (<i>Passerculus sandwichensis</i>)	19	3	8	42.1	11	57.9	0	0
Song Sparrow (<i>Melospiza melodia</i>)	15	8	3	20.0	10	66.7	4	26.7
Vesper Sparrow (<i>Pooecetes gramineus</i>)	1	1	0	0	1	100	0	0

Table 6. Model selection results for the 13 candidate models considered in predicting nest predation in northeast Illinois grasslands, 2012-2013.

Model	K	ΔAIC_c	-2LogL	w_i
development (2.5 km) + patch size	3	0 ^a	1217.24	0.43
distance to edge (m) + patch size	3	0.13	1217.37	0.41
patch size	2	2.00	1221.24	0.16
forest (2.5 km)	2	15.67	1234.91	0.00
constant survival	1	16.34	1237.59	0.00
forest (2.5 km) + distance to edge (m)	3	16.42	1233.66	0.00
distance to edge (m)	2	16.48	1235.72	0.00
day of season	2	16.78	1236.02	0.00
crops (2.5 km) + distance to edge (m)	3	16.84	1234.08	0.00
stage	2	18.17	1237.41	0.00
development (2.5 km) + distance to edge (m)	3	18.23	1235.47	0.00
development (2.5 km)	2	18.29	1237.53	0.00
crops (2.5 km)	2	18.29	1237.53	0.00

^aMinimum $AIC_c = 1223.25$

Table 7. Model selection results for the 12 candidate models considered in predicting probability of brood parasitism in northeast Illinois grasslands, 2012-2013.

Model	K	ΔAIC_c	$-2\text{Log}L$	w_i
development (2.5 km) + patch size	3	0 ^a	256.35	0.60
development (2.5 km)	2	3.72	260.07	0.09
development (2.5 km) + distance to edge (m)	3	4.29	258.60	0.07
crops (2.5 km)	2	4.91	261.26	0.05
constant survival	1	5.14	263.52	0.05
crops (2.5 km) + distance to edge (m)	3	5.55	259.85	0.04
distance to edge (m)	2	6.40	262.75	0.02
day of season	2	6.74	263.09	0.02
forest (2.5 km)	2	6.79	263.14	0.02
patch size	2	6.80	263.15	0.02
forest (2.5) + distance to edge (m)	3	7.76	262.06	0.01
distance to edge (m) + patch size	3	8.33	262.64	0.01

^aMinimum $AIC_c = 262.42$

Table 8. Species documented depredating camera-monitored nests in northeast Illinois grasslands, 2012-2013.

Species	# Events
Coyote (<i>Canis latrans</i>)	4
White-tailed deer (<i>Odocoileus virginianus</i>)	4
Thirteen-lined ground squirrel (<i>Ictidomys tridecemlineatus</i>)	3
Striped skunk (<i>Mephitis mephitis</i>)	2
Brown-headed Cowbird (<i>Molothrus ater</i>)	1
North American Racer (<i>Coluber constrictor</i>)	2
Weasel sp. (<i>Mustela sp.</i>)	1
Hawk sp.	1
Small mammal sp.	1
Total	19

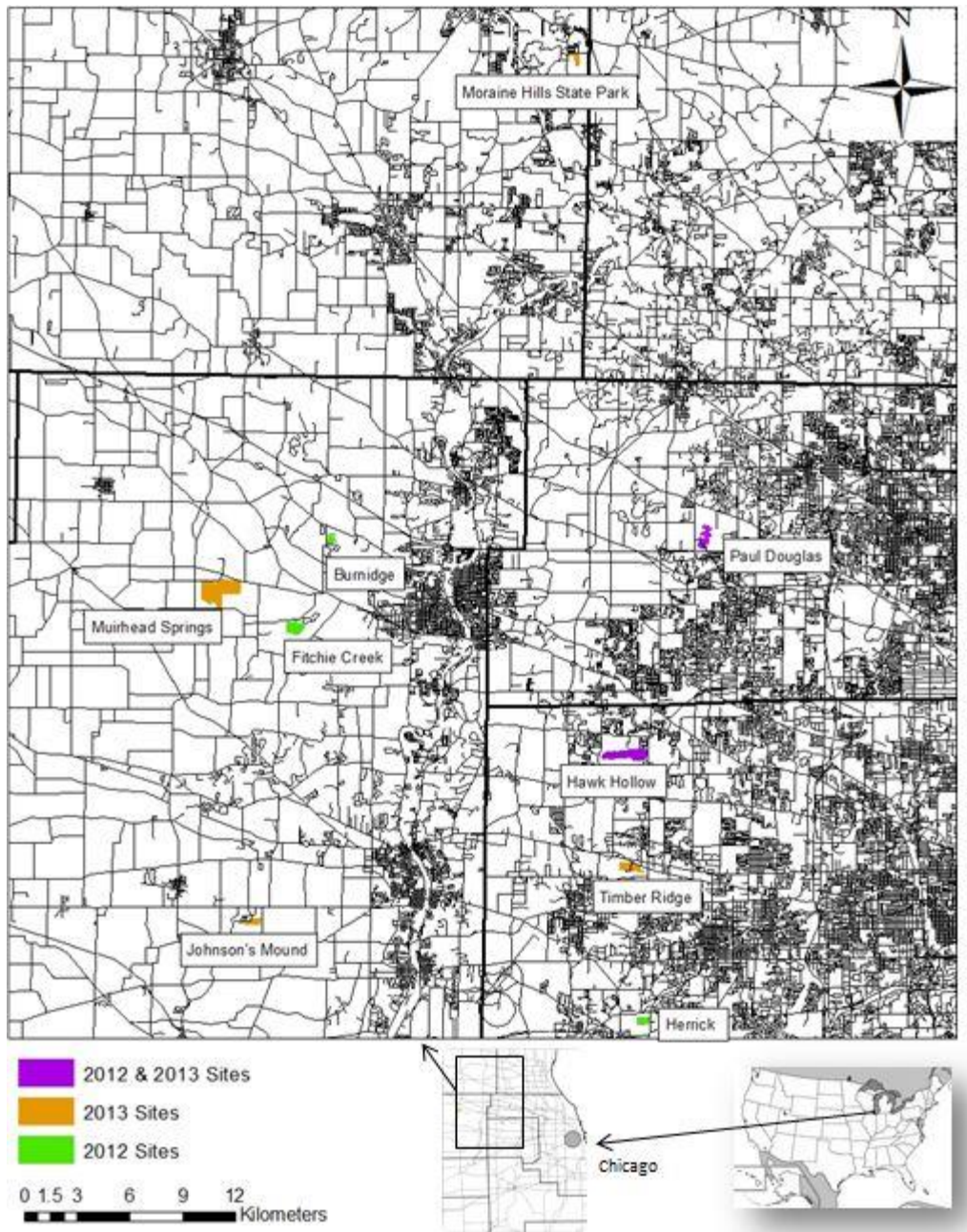


Figure 5. Map of northeastern Illinois displaying sites searched for grassland bird nests in 2012 and 2013. Thin black lines indicate roads and thick black lines indicate county borders. Map adapted from Gehrt et al. (2013).

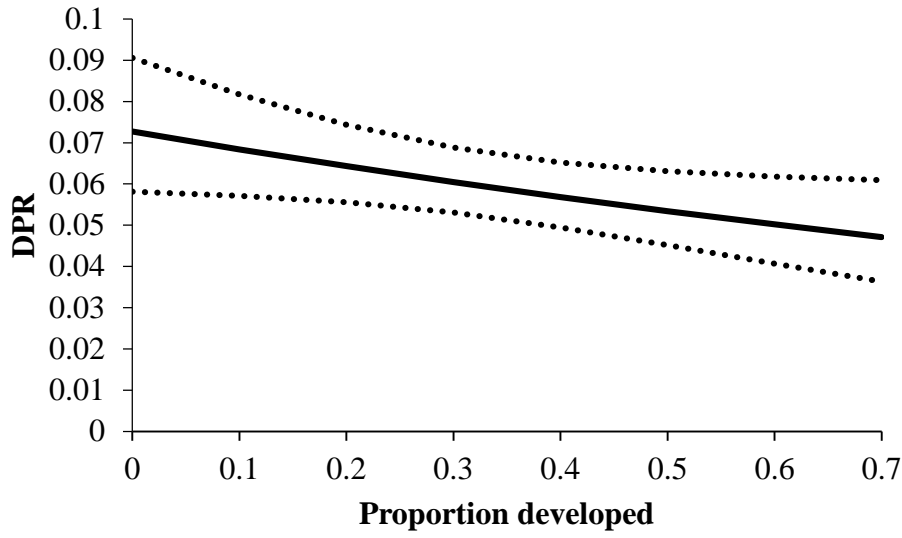


Figure 6. Model-averaged daily predation rate (DPR; \pm 95% CI) for grassland bird nests in northeast Illinois grasslands from 2012 to 2013 as a function of proportion of development in a 2500-m radius.

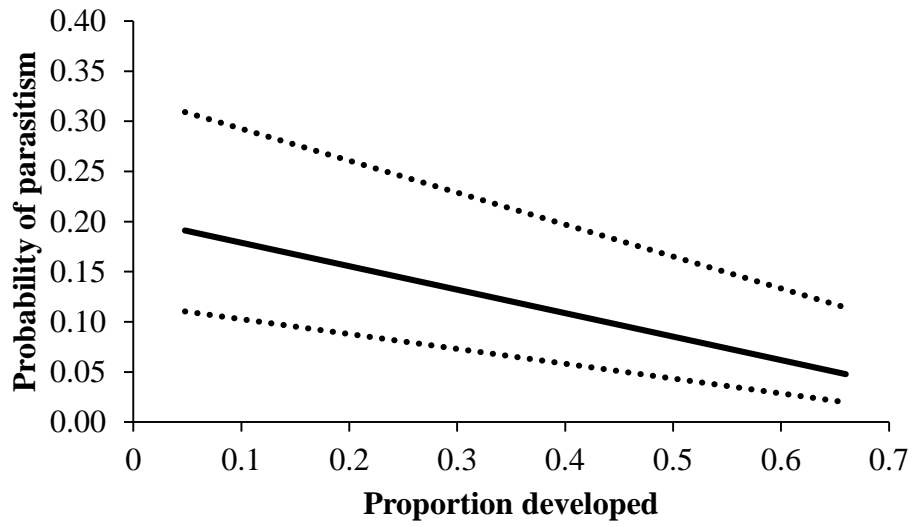


Figure 7. Probability of brood parasitism as a function of development in a 2500-m radius in northeast Illinois grasslands from 2012 to 2013.

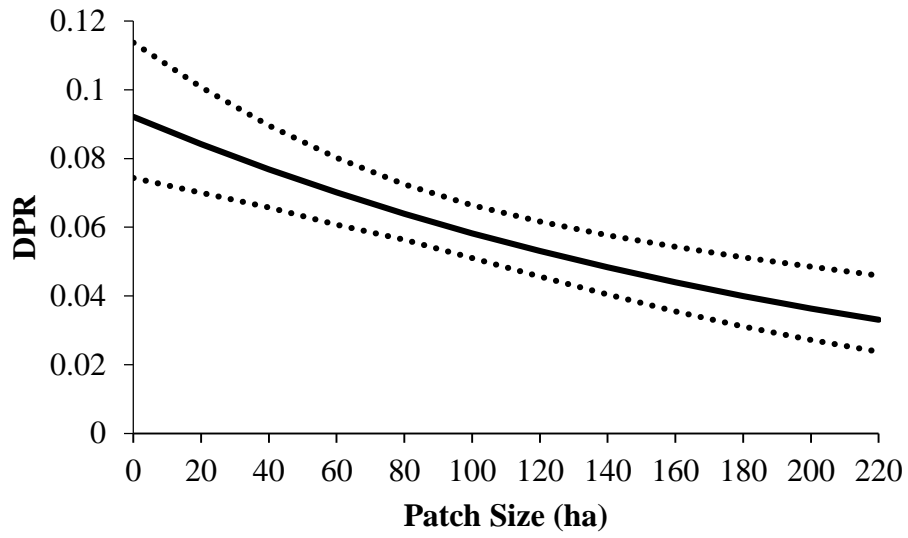


Figure 8. Model-averaged daily predation rate (DPR; \pm 95% CI) for grassland bird nests in northeast Illinois grasslands from 2012 to 2013 as a function of patch size (ha).

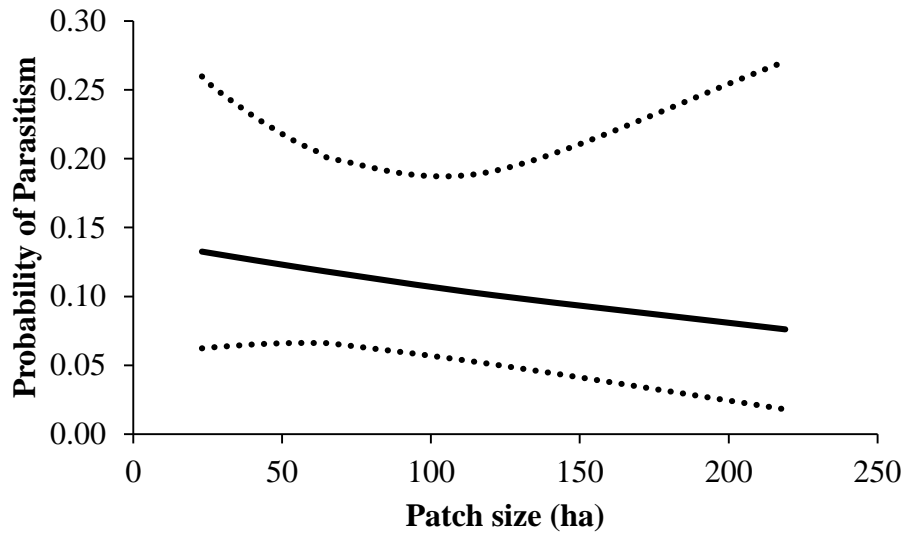


Figure 9. Probability of brood parasitism as a function of patch size (ha) in northeast Illinois grasslands from 2012 to 2013.

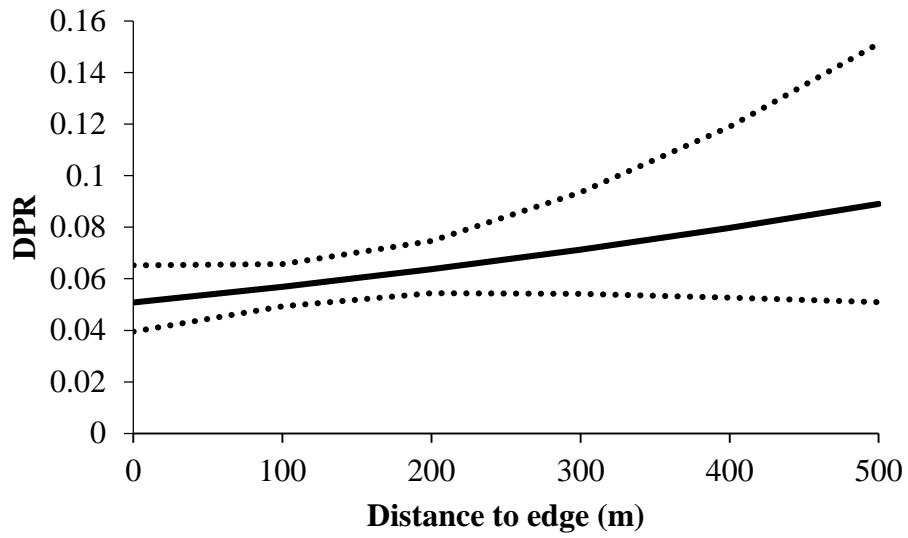


Figure 10. Model-averaged daily predation rate (DPR; \pm 95% CI) for grassland bird nests in northeast Illinois grasslands from 2012 to 2013 as a function of distance to nearest edge.

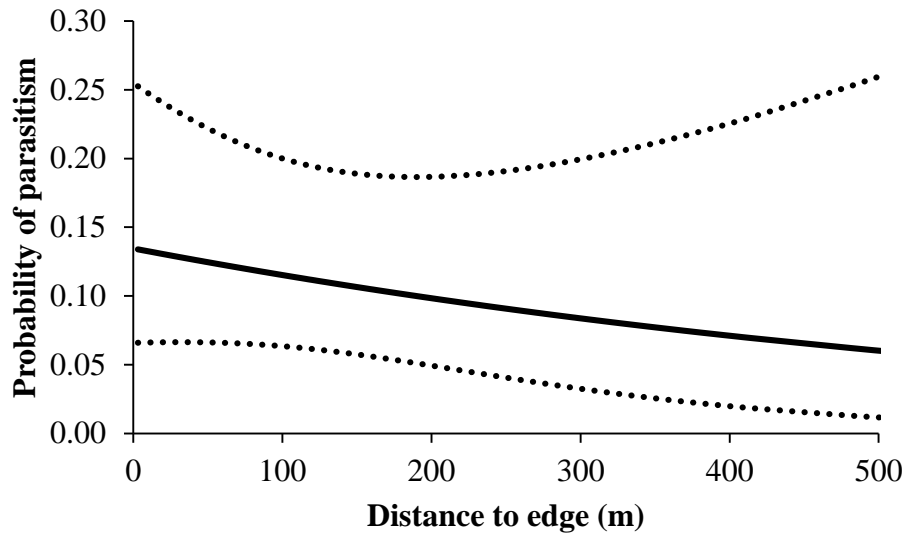


Figure 11. Probability of brood parasitism as a function of distance to nearest edge in northeast Illinois grasslands from 2012 to 2013.

CHAPTER 4

SUMMARY

As urbanization increases, understanding how this relatively novel land-cover type influences bird habitat use and demography becomes increasingly important (Ribic et al. 2009). Developed landscapes often contain natural areas, including grasslands, but the value of these areas for avian species is assumed to be poor (Johnson et al. 2010). The primary purpose of this research was to determine how development influences habitat use and reproductive success of grassland birds. The main finding from Chapter 2 was that amount of development in the landscape was positively associated with Savannah Sparrow (*Passerculus sandwichensis*) density, negatively associated with Sedge Wren (*Cistothorus platensis*) density, and was not strongly associated with the density of Bobolinks (*Dolichonyx oryzivorus*), Dickcissels (*Spiza americana*), Eastern Meadowlarks (*Sturnella magna*), or Henslow's Sparrows (*Ammodramus henslowii*). Patch size was a more important predictor of density than developed cover, with almost every species responding positively to patch size. Habitat structure and composition had a strong influence on density, and landscape variables consistently improved upon local-level models for all species.

The main finding from Chapter 3 was that landscape-level development was positively associated with reproductive success of grassland birds, with lower nest predation and brood parasitism rates in more urban landscapes. I also found a significant effect of patch size on nest survival, although this may have been influenced by one particularly large patch. Based on the camera data, I found that white-tailed deer (*Odocoileus virginianus*), coyotes (*Canis latrans*), and thirteen-lined ground squirrels (*Ictidomys tridecemlineatus*) were important nest predators in this system.

The density estimates and nest survival rates I found were similar to estimates from grassland bird studies located in less fragmented systems with greater amounts of landscape-level grassland cover, suggesting that urban grasslands are important and productive areas for breeding grassland birds in Illinois. Overall, these results suggest that urban grasslands provide quality habitat for grassland birds. In developed landscapes, managers should consider restoring or maintaining grassland habitat for declining grassland bird populations. Though conserving large patches may be preferred over small patches, this may be a difficult and costly task in developed areas. Thus, conserving small patches, which appear to have rates of reproductive success similar to larger patches, may also be beneficial to breeding grassland birds and economically more feasible.

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APPENDIX A

Table A1. Patch size and land-cover characteristics for the 30 grassland patches where point counts were conducted in 2012 and 2013.

County/Site	Patch Size (ha)	Developed Cover 1.6 km (%)	Tree Cover 1.6 km (%)	Grass Cover 1.6 km (%)	Hay Cover 1.6 km (%)
Cook county					
Bartel Grassland	360	47.13	16.85	8.01	2.32
Orland Grassland	285	68.86	4.09	3.01	1.61
Poplar Creek	611	33.02	27.87	8.42	8.98
Paul Douglas	63	44.25	21.55	4.90	2.89
DuPage county					
Blackwell	29	35.52	21.62	6.70	2.94
Churchill	10	62.67	5.76	0.44	0.14
Danada	38	54.13	13.68	4.23	2.67
Hawk Hollow	125	67.03	7.59	7.75	5.36
Herrick Lake	32	35.55	25.48	4.03	2.37
Mayslake	3	67.61	4.79	0.08	0.14
Meacham Grove	5	74.80	5.84	0.27	0.13
Night Heron Marsh	28	76.91	2.22	1.69	1.08
Oldfield Oaks	3.39	57.32	14.89	1.94	1.36
Pratts Wayne Woods	400	36.11	12.88	12.24	7.81
Songbird Slough	31	71.02	4.08	0.20	0.17
Springbrook Prairie	343	67.95	5.07	8.94	3.03
Timber Ridge	50	52.21	14.7	2.07	2.81
Kane county					
Aurora West	138	33.17	10.25	11.16	7.65
Burlington Prairie	52	2.45	5.02	1.24	11.16
Burnidge	30	27.97	18.64	4.48	15.56
Campton	24	32.78	15.57	4.63	14.11
Dick Young	250	15.66	9.15	9.20	14.14
Fitchie Creek	69	15.90	7.73	4.17	24.18
Hannaford	51	20.19	13.85	8.46	8.39
Johnson's Mound	39	6.72	8.28	7.01	11.81
Muirhead Springs	242	3.28	2.05	1.58	12.18
Otter Creek	9	35.23	7.88	4.88	11.47
Pingree Grove	55	14.81	5.39	4.38	12.05
Sauer Family Prairie Kame	29	5.43	5.93	10.22	10.61
McHenry					
Glacial Park	305	6.43	21.64	2.08	24.55

APPENDIX B

Table B1. Number of parameters, ΔAIC_c , and model weights for local-level models. Models with the minimum ΔAIC_c are emboldened for each species.

Model	K	BOBO ^a		DICK		EAME		GRSP	
		ΔAIC_c	w_i	ΔAIC_c	w_i	ΔAIC_c	w_i	ΔAIC_c	w_i
Constant	1	25.3	0.00	24.0	0.00	5.9	0.03	18.0	0.00
VOR	2	18.9	0.00	0^b	1.00	6.6	0.02	19.6	0.00
Dead veg. height	2	8.5	0.01	26.0	0.00	0.7	0.35	0^b	0.99
Litter depth	2	21.2	0.00	25.1	0.00	5.4	0.03	9.7	0.01
%Grass	2	14.3	0.00	26.6	0.00	5.0	0.04	20.8	0.00
%Forb	2	4.40	0.09	26.1	0.00	9.9	0.00	19.8	0.00
%Bare	2	23.6	0.00	23.6	0.00	9.8	0.00	15.7	0.00
%Litter	2	5.4	0.06	23.6	0.00	5.4	0.03	21.4	0.00
%Dead	2	0^b	0.84	24.1	0.00	0^b	0.49	20.8	0.00

Model	K	HESP		SAVS		SEWR	
		ΔAIC_c	w_i	ΔAIC_c	w_i	ΔAIC_c	w_i
Constant	1	11.3	0.00	15.8	0.00	43.8	0.00
VOR	2	13.3	0.00	9.0	0.01	47.8	0.00
Dead veg. height	2	12.0	0.00	13.6	0.00	0^b	1.00
Litter depth	2	13.4	0.00	0^b	0.98	47.9	0.00
%Grass	2	0^b	0.96	12.5	0.00	29.3	0.00
%Forb	2	7.1	0.03	11.7	0.00	34.3	0.00
%Bare	2	10.9	0.00	24.3	0.00	46.6	0.00
%Litter	2	13.4	0.00	16.4	0.00	39.9	0.00
%Dead	2	11.2	0.00	11.4	0.00	29.3	0.00

^aSpecies names abbreviated according to American Ornithologists' Union four-letter code. BOBO = Bobolink; DICK = Dickcissel; EAME = Eastern Meadowlark; GRSP = Grasshopper Sparrow; HESP = Henslow's Sparrow; SAVS = Savannah Sparrow; SEWR = Sedge Wren.

^bMinimum AIC_c BOBO = 599.0; Minimum AIC_c DICK = 386.4; Minimum AIC_c EAME = 152.5; Minimum AIC_c GRSP = 487.5; Minimum AIC_c HESP = 601.2; Minimum AIC_c SAVS = 695.3; Minimum AIC_c SEWR = 393.4

Table B2. Parameter estimates and 95% confidence intervals for local-level variables used to predict species density.

Parameter	BOBO			DICK			EAME			GRSP		
	β	95% CI		β	95% CI		β	95% CI		β	95% CI	
VOR	-0.041	-0.094	0.011	-0.081	-0.109	-0.052	0.015	-0.002	0.031	-0.027	-0.073	0.020
Dead veg. height	-0.005	-0.009	-0.002	0.002	-0.002	0.006	-0.001	-0.003	0.000	-0.006	-0.008	-0.003
Litter depth	0.007	-0.001	0.014	0.002	-0.026	0.031	0.004	0.002	0.006	0.006	-0.001	0.012
%Grass	0.005	-0.003	0.013	-0.002	-0.006	0.003	0.003	0.000	0.005	0.002	-0.004	0.008
%Forb	-0.001	-0.010	0.009	-0.004	-0.009	0.002	-0.001	-0.003	0.002	-0.005	-0.012	0.002
%Bare	0.003	-0.034	0.040	0.003	-0.014	0.020	0.002	-0.005	0.009	0.021	-0.002	0.044
%Litter	-0.012	-0.023	0.000	0.008	0.000	0.017	-0.004	-0.008	0.000	0.005	-0.007	0.017
%Dead	-0.002	-0.013	0.009	0.016	-0.001	0.034	-0.006	-0.011	-0.001	-0.003	-0.022	0.015
		HESP			SAVS			SEWR				
Parameter	β	95% CI		β	95% CI		β	95% CI				
VOR	0.011	-0.049	0.072	-0.012	-0.077	0.052	-0.006	-0.044	0.032			
Dead veg. height	-0.005	-0.009	0.000	-0.003	-0.009	0.003	0.008	0.003	0.013			
Litter depth	0.018	0.010	0.026	0.012	0.003	0.021	-0.002	-0.006	0.002			
%Grass	0.017	0.008	0.025	0.006	-0.005	0.016	0.004	-0.001	0.010			
%Forb	-0.011	-0.020	-0.003	-0.001	-0.013	0.011	-0.005	-0.010	0.000			
%Bare	-0.018	-0.042	0.007	0.004	-0.035	0.044	-0.005	-0.020	0.010			
%Litter	-0.004	-0.019	0.011	-0.012	-0.035	0.010	-0.003	-0.011	0.005			
%Dead	-0.008	-0.022	0.005	-0.007	-0.028	0.014	0.042	0.010	0.074			

^aSpecies names abbreviated according to American Ornithologists' Union four-letter code. BOBO = Bobolink; DICK = Dickcissel; EAME = Eastern Meadowlark; GRSP = Grasshopper Sparrow; HESP = Henslow's Sparrow; SAVS = Savannah Sparrow; SEWR = Sedge Wren.

APPENDIX C

Table C.1 Patch size and proportions for landscape variables considered in analysis of nest predation and brood parasitism for each study site in northeast Illinois.

County/Site	Patch Size (ha)	Developed Cover 2.5 km (%)	Tree Cover 2.5 km (%)	Row-crop Cover 2.5 km (%)
Cook				
Paul Douglas	63	57.50	12.30	1.40
Dupage				
Hawk Hollow	125	66.20	7.80	1.40
Herrick Lake	32	46.20	19.5	3.40
Timber Ridge	50	59.29	10.08	2.10
Kane				
Muirhead	242	4.79	3.50	68.20
Burnidge	42	22.05	14.30	30.20
Fitchie Creek	69	17.50	8.10	29.70
Johnson's Mound	39	11.79	5.03	54.90
McHenry				
Moraine Hills State Park	23	13.5	23.80	18.50