HABITAT USE, POPULATION SIZE, AND NESTING ECOLOGY OF CONSERVATION PRIORITY BIRD SPECIES USING RESTORED FIELDS IN AGRICULTURAL LANDSCAPES

BY

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DISSERTATION

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ABSTRACT

Agricultural intensification has negatively affected biodiversity throughout the world. In the U.S., population declines of many early successional bird species have been linked with habitat loss due to agriculture. One conservation effort that attempts to mitigate these losses is through programs that restore habitat on private lands that were formerly used for row-crop agriculture (hereafter private land programs). These programs have been widely regarded as beneficial to wildlife at local, landscape, and regional scales. In particular, studies have shown that farmland restoration efforts are associated with population-level increases of some conservation priority bird species. However, most studies in these habitats have focused exclusively on grassland bird species and so information is lacking on the habitat use of species that occupy later successional habitat. Based on their scale, and the lack of alternative large-scale solutions, these programs likely present the best opportunity for achieving broad scale conservation goals for birds in Midwestern landscapes. Although we know these programs provide large-scale conservation benefits, we do not understand the magnitude of this contribution. Beyond habitat use and population goals, conservation efforts to attract birds are only effective if those birds successfully reproduce. Therefore, an understanding of the relationship between selected habitat features and fitness consequences are important to understanding the conservation benefit of restored habitats, especially in situations where these are more likely to be decoupled such as agriculturally fragmented landscapes. Specifically, my research focused on addressing: (1) What attributes of restored patches are associated with habitat use by conservation priority birds and how this habitat use is mediated by the surrounding landscape, particularly the prevalence of surrounding agriculture?, (2) Are current levels of farmland restoration enough to achieve population goals for declining species and, if not, what level of additional restoration would be needed to achieve them?, (3) Are the habitat selection patterns of birds in these agriculturally fragmented landscapes associated with successful reproduction, and do relationships between selection and fitness vary among spatial scales? Regarding my first question, response to habitat features at all scales varied by species but the amount of habitat restoration in the surrounding landscape generally had little influence on habitat use; in most cases the amount of surrounding agriculture did not

moderate the effect of surrounding restored habitat. For question 2, I found that current restoration efforts in Illinois may be achieving population goals for the Bell's Vireo and Willow Flycatcher, but more than 10 times the current amount of restored habitat would be required to achieve the population goal for the Field Sparrow and Northern Bobwhite. Lastly, analyses related to my third question demonstrated that nesting Bell's Vireos and Willow Flycatchers selected habitat that increased nest survival, but not number of young fledged from successful nests, in these drastically altered landscapes. Patterns between nest survival and selected habitat features were apparent at the nest-site scale, but not at the patch or landscape scales. Overall, this study demonstrates the importance of private land programs to conservation priority birds, including what features are associated with their use and how to increase habitat selection by these species, that they are providing population level increases that achieve conservation goals for some species, and that, for two species, breeding habitat selection in agriculturally fragmented landscapes was not maladaptive.

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CHAPTER 1: GENERAL INTRODUCTION

Extensive conversion of natural land cover for human use has fundamentally altered ecosystems over the past 400 years (e.g. Pimm and Askins 1995). This alteration, largely driven by agricultural land conversion, has led to an overall loss of natural habitat (Foley et al. 2011) which has negatively affected wildlife species worldwide (Karp et al. 2013). Historically, efforts to curb wildlife declines have focused on creating protected areas; however, in some regions, such as the Midwestern U.S., public land is scarce and thus consideration of private land is necessary for effective large-scale conservation. One effective mechanism for reducing the negative impact of habitat loss has been habitat restoration. In landscapes affected by widespread agriculture this is typically done through programs that restore habitat on private lands that were formerly used for row-crop agriculture (hereafter private land programs). These programs pay farmers to remove land from agricultural land use and convert it into perennial land cover (Heard et al. 2000). Although the program-specific purposes differ, they are generally aimed at reducing agricultural runoff, flood prevention, erosion control, wetland restoration, and wildlife habitat restoration. Importantly, private land programs are implemented at sufficiently large scales to have population-level impacts on conservation priority species (e.g. Herkert 2007). For example, the Conservation Reserve Program, a restored private land program in the U.S., has more than 3 million ha of former agricultural land currently enrolled (USDA 2017).

Numerous studies have demonstrated the importance of private land programs to birds (Kleijn and Sutherland 2003, Van Buskirk and Willi 2004, Perkins et al. 2011). Moreover, population trends for declining birds are positively associated with the proportion of surrounding landscape that is restored private land (Herkert 1998; 2007, Veech 2006, Dallimer et al. 2010). While important, most investigations into the effects of private land programs on birds have focused on grassland dependent species due to their precipitous declines (e.g. Fletcher and

Koford 2003, Benson et al. 2011, Broyer et al. 2017). However, these habitats may also be important to declining species that occupy later successional stages such as shrubland birds (e.g. Wentworth et al. 2010, Pabian et al. 2013). Yet, little is known about what habitat features are associated with habitat use for these species in habitat restored through private land programs. To precisely relate information to improve the design and future management of habitat restored through these programs more information is needed on the influence local-scale vegetation, patch, and landscape features has on habitat use for species that occupy both grassland and shrubland habitats. To this end, in Chapter 2 I examined what local, patch, and landscape features are associated with the density of early successional bird species in restored habitats. I predicted that bird densities would respond positively to increasing patch size and the amount of restored habitat in the surrounding landscape, and I expected the strength of the positive response to be moderated by the amount of surrounding agriculture.

While understanding the habitat use of conservation priority species could be important for improving the implementation and management of restored private land, this information could also be useful in understanding what role these habitats play in achieving broad scale conservation goals for migratory birds (e.g. Carter et al. 2000, Faaborg et al. 2010). For example, recent efforts have focused on estimating the global population size of many declining migratory species (Rosenberg and Blancher 2005). Additionally, this effort was used to establish conservation objectives for these species through setting aspirational population goals that focused on achieving pre-decline levels (Rich et al. 2004). Subsequently, these large-scale goals have been "stepped-down" to regional and state-level objectives (e.g. IDNR 2005, Potter et al. 2007). However, while important, these efforts did not identify ways to achieve these goals, though they assume breeding habitat is limiting most bird populations (e.g. Will et al. 2005, Rushing et al. 2016). Thus, habitat restoration may be the most effective mechanism to reverse population declines (Morrison et al. 1992). Yet, extensive habitat restoration may be

economically infeasible and existing restoration programs, such as agri-environmental schemes (Europe) or Farm Bill conservation programs (North America) may occur at sufficiently broad scales (e.g. van Buskirk and Willi 2004, Herkert 2007) to achieve these population goals. Based on their scale, these private land programs are contributing to achieving conservation goals but to what extent, and are the current goals realistic? Additionally, based on the current contributions, given the existing monetary investment, what additional programmatic investments would be necessary to achieve these goals? In Chapter 3, I selected a set of focal species that are dependent on scrub/shrub and grassland habitats that have experienced severe population declines (Sauer et al. 2014) with stated population goals (IDNR 2005, Potter et al. 2007). For these species, I developed spatially explicit estimates of bird abundance within former agricultural fields enrolled in a habitat reestablishment program. I used these estimates to quantify the contribution of these habitats to meeting aspirational population goals (IDNR 2005) for my focal species and, in cases where population goals were not being met, the additional amount of habitat and the approximate cost that would be required to meet the goals.

Beyond examining habitat use and population size of conservation priority species, understanding the fitness consequences of habitat selection could shed light on the ability of animals to assess habitat quality in highly modified landscapes. Specifically, natural selection should favor the ability to assess habitat quality (Jaenike and Holt 1991, Martin 1998) and that animals should choose the best available habitat (Fretwell and Lucas 1970). Yet, many studies have demonstrated non-ideal or maladaptive habitat selection in numerous species (e.g., Arlt and Pärt 2007, Magi et al. 2009) especially in habitats heavily altered by anthropogenic land use (e.g. Weldon and Haddad 2005, Gilroy et al. 2011). For example, nests located in highly fragmented and agriculturally dominated landscapes, like those of private land programs, are more likely to have greater predation and brood parasitism rates than nests in more intact landscapes (e.g. Donovan et. al. 1995, Chalfoun et al. 2002). Furthermore, restored habitat in agricultural

landscapes, while widely thought to be beneficial for birds (e.g., Herkert 2007), has also been suggested to result in fecundity rates too low to sustain populations of some species (McCoy et al. 1999). Thus, understanding habitat choice and reproductive outcomes in restored habitats could both provide new insights into understanding the adaptive nature of habitat selection in birds and provide managers with valuable information to be applied to future conservation efforts for priority species. In Chapter 4, I studied the breeding habitat selection and two reproductive metrics (nest success and fledgling production) of Bell's Vireo (*Vireo bellii*) and Willow Flycatcher (*Empidonax traillii*) across three ecologically relevant scales 1) nest site (5-m buffer around nest), 2) habitat patch, and 3) landscape (200-m buffer) using data collected in restored habitat on former farmland. I predicted that nest-site habitat selection for both species would result in greater nest survival and fledgling production. Conversely, because restored patches are embedded in agriculturally altered landscapes with increased threats of predation and brood parasitism, habitat selection for patch and landscape features may have become disassociated with fitness benefits. Chapter 2 is intended for submission to Agriculture, Ecosytems, and Environment with T. J. Benson as a coauthor, Chapter 3 is intended for the Journal of Applied Ecology with T. J. Benson, D. L. Linden, and K. W. Stodola as coauthors, and Chapter 4 is intended for the Journal of Animal Ecology with T. J. Benson as a coauthor.

LITERATURE CITED

Arlt, D., and T. Pärt. 2007. Nonideal breeding habitat selection: A mismatch between preference and fitness. Ecology 88:792–801.

Benson, T. J., J. J. Dinsmore, and W. L. Hohman. 2011. Short-term effects of burning and disking on songbird use of floodplain conservation easements. American Midland Naturalist 165:257–273.

Broyer, J. L., L. Pelloli, L. Curtet, and R. Chazal. 2017. On habitat characteristics driving meadow passerine densities in lowland hay-meadow systems in France. Agriculture, Ecosystems, and Environment 237:24–30.

Carter, M. F., Hunter, W. C., Pashley, D. N., and Rosenberg, K. V. 2000. Setting conservation priorities for landbirds in the United States: the Partners in Flight approach, Auk 117:541–548.

Chalfoun, A. D., and K. A. Schmidt. 2012. Adaptive breeding-habitat selection: Is it for the birds? Auk 129:589–599.

Donovan, T. M., F. R. Thompson, III, J. Faaborg, and J. R. Probst. 1995. Reproductive success of migratory birds in habitat sources and sinks. Conservation Biology 9:1380–1395.

Faaborg, J., R. T. Holmes, A. D. Anders, K. L. Bildstein, K. M. Dugger, S. A. Gauthreaux, Jr., P. Heglund, K. A. Hobson, A. E. Jahn, D. H. Johnson, S. C. Latta, D. J. Levey, , P. P. Marra, , C. L. Merkord, E. Nol, , S. I. Rothstein, , T. W. Sherry, T. S. Sillett, , F. R. Thompson, III, and N. Warnock. 2010 Conserving migratory land birds in the New World: Do we know enough? Ecological Applications 20:398–118.

Fletcher Jr., R. J., and R. R. Koford. 2003. Changes in breeding bird populations with habitat restoration in Northern Iowa. American Midland Naturalist 150:83 – 94.

Foley, J. A., N. Ramankutty, K. A. Brauman, E. S.Cassidy, J. S. Gerber, M. Johnston, , N. D. Mueller, C. O'Connell, D. K. Ray, P. C. West, C. Balzer, E. M. Bennett, S. R.

Carpenter, J. Hill, C. Monfreda, S. Polasky, J. Rockström, J. Sheehan, S. Siebert, G. D. Tilman, and D. P. M. Zaks. 2011. Solutions for a cultivated planet. Nature 478: 337–342.

Fretwell, S. D., and H. L. Lucas. 1970. On territorial behavior and other factors influencing habitat distribution in birds. I. Theoretical development. Acta Biotheoretica 19:16–36.

Heard, L. P., A. W. Allen, L. B. Best, S. J. Brady, W. Burger, A. J. Esser, E. Hackett, D. H. Johnson, R. L. Pederson, R. E. Reynolds, C. Rewa, M. R. Ryan, R. T. Molleur, and P. Buck. 2000. A comprehensive review of Farm Bill contributions to wildlife conservation, 1985–2000. *In*: W. L. Hohman and D. J. Halloum, eds. United States Department of Agriculture, Natural Resources Conservation Service, Wildlife Habitat Management Institute, Technical Report, USDA/NRCS/WHMI-2000.

Karp, D. S., A. J. Rominger, J. Zook, J. Ranganathan, P. R. Ehrlich, and G. C. Daily. 2012. Intensive agriculture erodes biodiversity at large scales. Ecology Letters 15:963–970.

Kleijn, D., and W. J. Sutherland. 2003. How effective are European agri-environmental schemes in conserving and promoting biodiversity. Journal Applied Ecology 40: 947–969.

Herkert, J. R. 1998. The influence of the CRP on Grasshopper Sparrow population trends in the mid-continental United States. Wildlife Society Bulletin 26:227–231.

Herkert, J. R. 2007. Conservation Reserve Program benefits on Henslow's Sparrows within the United States. Journal of Wildlife Management 71:2749–2751.

Gilroy, J. J., G. Q. A. Anderson, J. A. Vickery, P. V. Grice, and W. J. Sutherland. 2011. Identifying mismatches between habitat selection and habitat quality in a ground nesting farmland bird. Animal Conservation 14: 620–629.

Illinois Department of Natural Resources. 2005. The Illinois comprehensive wildlife conservation plan & strategy. Version 1.0. Illinois Department of Natural Resources, Springfield, IL.

Jaenike, J., and R. D. Holt. 1991. Genetic variation for habitat preference: evidence and explanations. American Naturalist 137:567–590.

Mahon, C. L., E. M. Bayne, P. Solymos, S. M. Matsuoka, M. Carlson, E. Dzus, F. K. A. Schmiegelow, and S. J. Song. 2014. Does expected future landscape condition support proposed population objectives for boreal birds? Forest Ecology and Management 312:28–39.

Mägi, M., R. Mänd, H. Tamm, E. Sisask, P. Kilgas, and V. Tilgar. 2009. Low reproductive success of Great Tits in the preferred habitat: A role of food availability. Ecoscience 16:145–157.

Martin, T. E. 1998. Are microhabitat preferences of coexisting species under selection and adaptive? Ecology 79:656–670.

McCoy, T. D., M. R. Ryan, E. W. Kurzejeski, and L. W. Burger, Jr. 1999. Conservation Reserve Program: source or sink habitat for grassland birds in Missouri? Journal of Wildlife Management 63:530–538.

Morrison, M. L., B. G. Marcot, and R. W. Mannan. 1992. Wildlife Habitat Relationships: Concepts and Applications. Madison, WI, USA: University Wisconsin Press.

Pabian, S. E., A. M. Wilson, and M. C. Brittingham. 2013. Mixed response of farmland birds to the conservation reserve enhancement program in Pennsylvania. Journal of Wildlife Management 77:616–625.

Perkins, A. J., H. E. Maggs, A. Watson, J. D. Wilson. 2011. Adaptive management and targeting of agrienvironment schemes does benefit biodiversity: a case study of the corn bunting *Emberiza calandra*. Journal Applied Ecology 48:514–522.

Pimm, S. L., R. A. Askins. 1995. Forest losses predict bird extinctions in Eastern North America. Proceedings of the National Academy of Sciences 92:9343–9347.

Potter, B. A., G. J. Soulliere, D. N. Ewert, M. G. Knutson, W. E. Thogmartin, J. S. Castrale, and M. J. Roell. 2007. Upper Mississippi River and Great Lakes Region Joint Venture landbird habitat conservation strategy. U.S. Fish and Wildlife Service, Fort Snelling, MN

Rich, T. D., C. J. Beardmore, H. Berlanga, P. J. Blancher, M. S. W. Bradstreet, G. S. Butcher, D. W. Demarest, E. H. Dunn, W. C. Hunter, E. E. Iñigo-Elias, J. A. Kennedy, A. M. Martell, A. O. Panjabi, D. N. Pashley, K. V. Rosenberg, C. M. Rustay, J. S. Wendt, and T. C. Will. 2004. Partners in Flight North American Landbird Conservation Plan. Cornell Lab of Ornithology, Ithaca, New York, NY, USA.

Rosenberg, K.V., and P. J. Blancher. 2005. Setting numerical population objectives for priority landbird species. Pages 57–67 in C. J. Ralph and T. D. Rich, editors. Bird conservation implementation and integration in the Americas: Proceedings of the Third International Partners in Flight Conference. Volume 1. USDA Forest Service General Technical Report PSW-GTR-191, Pacific Southwest Research Station, Albany, California, USA.

Rushing, C. S., T. B. Ryder, and P. P. Marra. 2016. Quantifying drivers of population dynamics for a migratory bird throughout the annual cycle. Proceedings of the Royal Society B 2016;283.

Sauer, J. R., D. K. Niven, J. E. Hines, Ziolkowski Jr., K. L. Pardiek, J. E. Fallon, & W. A. Link. 2014. The North American Breeding Bird Survey, Results and Analysis 1966 - 2013. Version 01.30.2015 USGS Patuxent Wildlife Research Center, Laurel, MD

Thogmartin, W. E., S. M. Crimmins, and J. Pearce. 2014. Prioritizing bird conservation actions in the prairie hardwood transition of the Midwestern United States. Biological Conservation 176: 212–223.

United States Department of Agriculture, Farm Service Agency. 2017. "Conservation Enrollments Monthly Summary – JUNE 2017." Available: https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/Conservation/PDF/JUN%202017%20Summary.pdf

Van Buskirk, J. and Y. Willi. 2004. Enhancement of farmland biodiversity within set-aside land. Conservation Biology 18: 987–994.

Veech, J. A. 2006. A comparison of landscapes occupied by increasing and decreasing populations of grassland birds. Conservation Biology 20:1422–1432.

Weldon, A. J., and N. M. Haddad. 2005. The effects of patch shape on Indigo Buntings: evidence for an ecological trap. Ecology 86:1422–1431.

Wentworth, K. L., M. C. Brittingham, A. M. Wilson. 2010. Conservation Reserve Enhancement Program fields: benefits for grassland and shrub-scrub species. Soil and Water Conservation 65: 50–60.

Will, T. C., J. M. Ruth, K. V. Rosenberg, D. Krueper, D. Hahn, J. Fitzgerald, R. Dettmers, and C. J. Beardmore. 2005. The five elements process: designing optimal landscapes to meet bird conservation objectives. Partners in Flight Technical Series 1. Partners in Flight:http://www.partnersinflight.org/pubs/ts/01-FiveElements.pdf>. Accessed 1 Mar 2017.

CHAPTER 2: DIFFERENTIAL EFFECTS OF LANDSCAPE COMPOSITION AND PATCH SIZE ON AVIAN HABITAT USE OF RESTORED FIELDS IN AGRICULTURALLY FRAGMENTED LANDSCAPES

ABSTRACT: Existing habitat restoration programs play an important role in restoring wildlife habitat in agriculturally fragmented landscapes. Yet, restoration programs are often implemented without identifying target species and conservation benefit could be improved if the needs of declining species were used to guide future enrollments and management of habitat created through these programs. To improve implementation and management of these programs, we examined what local, patch, and landscape features affected habitat selection by 6 conservation priority bird species. Specifically, we were interested in understanding how species density responded to patch size, surrounding restored habitat and other landscape features, and the degree to which the amount of surrounding cropland moderated the response to the amount of restored habitat. To do this, we conducted bird surveys for six species: Bell's Vireo (*Vireo bellii*), Dickcissel (*Spiza americana*), Field Sparrow (*Spizella pusilla*), Northern Bobwhite (*Colinus virginianus*), Willow Flycatcher (*Empidonax traillii*), and Yellow-billed Cuckoo (*Coccyzus americanus*) at 172 restored patches during 2012 -2015. While most species densities were greater with increased patch size, only the relationship for Dickcissels was strong. Dickcissel (+) and Field Sparrow (-) had strong relationships with the amount of surrounding restored habitat, although the amount of cropland only moderated the response of Willow Flycatcher to surrounding restored habitat; however, the relationship was opposite expectations. Additionally, we found that Bell's Vireo, Dickcissel, and Willow Flycatcher densities increased with greater amounts of grassland in the surrounding landscape. Contrary to many previous studies, our results suggest that for most of our focal species, targeting enrollments in areas with existing restored habitat and creating larger patches did not increase densities of our focal species at patches

created by these programs, although maintaining and placing them in grass-dominated landscapes will.

INTRODUCTION

Ecosystems have been significantly altered during the past 400 years due to the extensive conversion of natural land cover for human uses (e.g. Pimm and Askins 1995). Agriculture now occupies more than 38% of the world's terrestrial habitat (Clay 2004) and is considered the primary driver of habitat alteration and decreased ecosystem services (Foley et al. 2011). Landscapes affected by widespread agricultural conversion are characterized by a mosaic of urban, semi-natural, natural, and agricultural habitats with different levels of land-use intensity. This increased land-use intensity combined with concurrent reduction in natural habitat heterogeneity has had well-established negative effects on species occupying these landscapes (Karp et al. 2013). As a result, habitat loss and fragmentation has been the primary cause of population declines of species worldwide (e.g. Adams et al. 2013; Bohm et al. 2013).

In response to widespread population declines, traditional conservation efforts have focused on establishing protected areas. However, the proportion of land that can be affected by these protected areas is relatively small and effective conservation efforts will require the inclusion of private lands (Scott et al. 2001). An increasingly common conservation effort in agricultural landscapes is to restore wildlife habitat on private farms through habitat restoration programs (hereafter private land programs). These programs are currently the primary mechanism available to deliver large-scale ecological restoration to areas with widespread agriculture (Donalds and Evans 2006). Importantly, private land programs are implemented at sufficiently large scales to have population-level impacts on conservation priority species (IDNR 2005; Potter et al. 2007). For example, one of many private land programs in the U.S., the Conservation Reserve Program, has more than 3 million ha currently enrolled (USDA 2017). Habitat restoration efforts of these types of programs have been demonstrated to be important to numerous species (e.g. Evans et al. 2014; Conover et al. 2011; Batàry et al. 2011)

Birds have been observed to benefit from private land programs in the U.S. and Europe (e.g. Kleijn and Sutherland 2003; Van Buskirk and Willi 2004; Herkert 2007; Perkins et al. 2011). Indeed, evidence suggests that population trends for declining birds are positively associated with the proportion of restored habitat in the broader landscape (Herkert 1998, 2007; Veech 2006; Dallimer et al. 2010). Studies of birds using restored habitats have evaluated the importance of surrounding landscape cover with grassland species generally responding negatively to the amount of surrounding forest (e.g. Osborne and Sparling 2013) and positively to the amount of grassland (e.g. Wentworth et al. 2010) and cropland cover (e.g. Riffell et al. 2008). Additionally, the amount of restored habitat in the surrounding landscape has been observed to positively influence early successional bird species (Pabian et al. 2015; Evans et al. 2014; Wentworth et al. 2010). Yet, other studies found neutral or negative responses of early successional birds to the amount of restored habitat in the landscape (e.g. Davey et al. 2010; Wilson and Wentworth 2012; Riffell et al. 2013). These contrasting results may be the result of differences between landscape context between native and restored patches (Fletcher and Koford 2002) due to higher amounts of cropland within agriculturally dominated landscapes (e.g. Quinn et al. 2014). Indeed, the influence of farmland restored habitats on bird abundance may depend on the amount of cropland cover in the surrounding landscape (e.g. Tscharnake et al. 2005; Batàry et al. 2011). Further, Batàry et al. (2011) found that increased occupancy of bird species to private land programs are more likely to be found in simple landscapes (i.e. cropland or grassland dominated) vs. complex landscapes (i.e. mix of forest, grassland, and cropland).

Beyond landscape context, both local-scale vegetation and patch-scale features of restored habitats can affect the occupancy and abundance of birds (e.g. Batàry et al. 2007; Concepcion et al. 2008). Specifically, both local-scale vegetation characteristics (Fletcher and Koford 2002; Vander Haegen et al. 2015) and patch size affect density of bird species in restored habitat (Vanbuskirk and Willi 2004; Wentworth et al. 2010). To date, most investigations into the effects

of these features on birds using restored habitat have primarily focused on grassland birds due to their precipitous declines (e.g. Brennan et al. 2005). However, recent research has suggested these habitats are important to declining species that occupy later successional stages such as shrubland birds (e.g. Wentworth et al. 2010; Pabian et al. 2013). Yet, little is known about what influence local-scale vegetation and patch size has on habitat use for these species and improved understanding of these features for both grassland and shrubland species would provide useful information for the design and future management of habitat restored through these programs.

While important for birds, private land programs are often implemented without identifying target species intended to benefit from conservation actions (Miller and Hobbs 2007) Conservation benefits may be improved by using the needs of priority species to guide future enrollments and management of restored private land programs (Wells 2010). In Europe private land programs, called agri-environmental schemes and set asides, have included explicit goals of increasing biodiversity (e.g. Kleijn et al. 2003) or increasing populations of conservation priority species (e.g. Perkins et al. 2011). While there have been efforts to target program enrollments to benefit focal species in the U.S. (e.g. Evans et al. 2013), restored private-land programs generally focus on broader environmental goals such as improving water quality and regulating commodity prices. Nevertheless, more specific conservation goals, such as achieving state and regional population goals for priority species (e.g. IDNR 2005; Potter et al. 2007), could be realized if they were incorporated into these broad-scale habitat restoration programs.

In order to improve implementation and management of private land programs for increased conservation benefit for avian species, we examined what local, patch, and landscape features are associated with the density of early successional bird species in restored habitats. Specifically, using repeated counts in randomly selected patches restored through a large-scale restoration program, we examined the influence of vegetation structure and composition, patch size, and landscape context on density of conservation priority bird species. We predicted that bird

densities would respond positively to increasing patch size and the amount of restored habitat in the surrounding landscape, and we expected the strength of the positive response to be moderated by the amount of surrounding agriculture with greater benefits in landscapes that have higher amounts of surrounding row-crop agriculture due to the marginal habitat quality of the surrounding habitat.

METHODS

Study sites

We examined patches restored through the Conservation Reserve Enhancement Program (CREP), a private-land program known to provide habitat for a variety of conservation priority bird species dependent on early successional habitats (Pabian et al. 2013). This is a voluntary program that provides landowners with a financial incentive to restore cropland into some type of natural perennial vegetation (both native and nonnative species). We surveyed for birds at 172 restored patches in 10 counties in central and west-central Illinois (Figure 1) with most remaining natural land cover being forest (Walk et al. 2010b). In Illinois, this program is primarily focused on the Illinois River watershed with most restored fields enrolled as permanent conservation easements. We chose focal counties in the central and western part of the state where program enrollments were most common. Sample patches all had early successional vegetative communities with structure ranging from grassland habitats consisting of warm and cool season grasses and forb cover to dense woody habitat dominated by shrubs and hardwood trees (*Quercus* spp. or naturally colonizing tree species) and were often adjacent to riparian forests, row-crop agriculture, and pastures.

Bird surveys

We sampled restored patches during the breeding seasons of 2012–2015 from mid-May to mid-July. To maximize detections of rare species, birds were surveyed using unlimited-radius

point counts, 10 minutes in duration (Ralph et al. 1995). Point counts focused on six species of conservation concern that occupy grassland or shrubland habitats and for which there were state and regional population goals (IDNR 2005; Potter et al. 2007): Bell's Vireo, Dickcissel, Field Sparrow, Northern Bobwhite, Willow Flycatcher, and Yellow-billed Cuckoo. All individuals encountered (auditory and visual) were recorded including the estimated distance to detected individuals and sex (if possible). Prior to each point count, observers used laser range finders to calibrate distances based on features within farmland-restored patches. Point count surveys were conducted between sunrise and 1100 CDT, and were not conducted during inclement weather such as rain or wind >13 mph (Ralph et al. 1993). Point count locations were chosen randomly using ArcGIS (ESRI 2012) and were > 70 m away from patch edges to ensure that we were primarily sampling individual birds using the focal patch. Bird surveys were conducted 1-4 times in 2012 and 2-3 times in 2013-2015; visits were separated by >14 days. Surveys were conducted by 3 observers in 2012 and 2013 and 4 observers in 2014 and 2015 that rotated among patches to minimize potential observer bias. Observers were trained in species identification and distance estimation prior to data collection.

Local vegetation variables

We recorded data on vegetation structure and cover from May 15 - June 20 each year using a modified BBIRD protocol (Martin et al. 1997). Sampling was conducted at the randomly selected point count location. Local scale variables were chosen on the basis of published literature and judgment of relevance to early successional birds. Specifically, within a 5-m radius we estimated cover of grass, forbs, shrubs (woody cover ≤1.5 m tall), and trees (both native and non-native) in the four cardinal directions. To provide an index of tree density at each plot we used a laser range finder to estimate the distance to the nearest tree (>1.5 m). To assess understory vegetation density, we used a Robel pole marked at 0.1 m increments and recorded the

% cover in each of four 0.5 m segments from 5 m away in the four cardinal directions at a height of 1 m above the ground (Robel et al. 1970). For all vegetation variables, we averaged values from the four cardinal directions into one value for each point-count location.

Patch variables

We quantified patch size using world imagery base maps in GIS (ESRI 2012) by measuring all contiguous habitats around each restored patch considering any forest, agricultural fields, and two and four lane roads with a defined roadside edge to be patch edges (Winter et al. 2006). When patches of similar habitat were linked with corridors 20 m or wider we considered them to be part of the same patch. All patch delineations were ground truthed for accuracy.

Landscape variables

We quantified landscape variables that we hypothesized would influence early successional bird abundance (Table 2.1) using ArcGIS 10.1 (ESRI 2012) and data from the 2014 National Agricultural Statistics Service Cropland Data Layer (NASS, CDL: Boryan et al. 2014). Land cover was classified as grassland (including grass and pasture), forest (sum of deciduous, mixed, and woody wetland forest cover types), cropland (including corn, soybean, and all other crop cover types), and restored habitat (based on USDA CRP GIS data layer centered on the patch boundary. We used the Geospatial Modeling Environment (Beyer 2012) to quantify proportions of each land-cover variable in a 200-m, 700-m, and 1200-m buffer around each patch, all of which are scales known to be influential to early successional bird abundance (Renfrew and Ribic 2008).

Statistical analyses

We examined patch-level covariates on both detection probability and bird abundance with the extended hierarchical distance sampling model of Royle (2004) using the gdistsamp function in the unmarked package in R (Fiske and Chandler 2011). Specifically, we used this function to evaluate models that describe how abundance varied as functions of covariates at local and landscape spatial scales. For the abundance portion of the model, spatial variation of the number of birds was treated as a negative binomial random variable. Importantly, this formulation allowed us to model spatial variation among point-count survey plots.

The detection portion of our model relied on the distance-sampling likelihood for point transect data (Buckland et al. 2001). For each species, we evaluated two models for the detection function including a null model and one that included year as a covariate. Additionally, we fit models evaluating combinations of uniform, exponential, half-normal, and hazard-rate base functions. Candidate models for detection function were evaluated using AIC adjusted for small sample size (AIC_c) to determine which model best fit the data (Burnham and Anderson 2002). We binned observations into 20-m intervals and right truncated observations for all species where detection probability declined to <10% (e.g. Buckland et al. 2001) which for all focal species was approximately 100 m.

Because species may respond to landscape features at different scales (Wiens 1989), prior to fitting candidate models we employed a pseudo-optimized multi-scale analysis (McGarigal et al. 2016) where we evaluated each landscape covariate separately across a range of predetermined scales and used model selection to determine which scale would be used in species landscape candidate model set (Table 2.2). Each scale was ranked according to AIC_c and computed model weights (Burnham and Anderson 2002) and the top-ranked scale was used in species candidate models. Additionally, all continuous covariates were standardized to have a zero mean and unit variance. Prior to analysis, we evaluated correlations among variables and did not include any

highly correlated variables ($|r|> 0.70$) in the same model. Year was included in all models to account for expected temporal variation due to sampling in multiple years.

To determine what local vegetation, patch and landscape scale factors influenced bird density we developed three sets of *a priori* models based on factors that we believed to be important from previous research and personal observation (Tables 2.3, 2.4, and 2.5). The first model set included local vegetation variables and the second model set included the best-ranked local vegetation variable with patch and landscape variables. We included local-scale vegetation variables as an additive effect in the landscape models because these variables are well-established predictors of density for species occupying early successional habitats (Ribic and Renfrew 2008, Fisher and Davis 2010). We established the third model set to test the hypothesis that surrounding cropland positively moderates the effect of restored habitat on the density of our focal species. This model set included four competing models: a null model, a year only model, a model with additive effects of the amount of crop and restored habitat, and a model including interactive and additive effects of the amount of crop and restored habitat in the surrounding landscape. We ranked models according to AIC_c and computed model weights (Burnham and Anderson 2002). In cases where no model was overwhelmingly supported ($w_i > 0.9$), we model averaged using the average method in the R package AICcmodavg (Mazarolle 2012) to examine effects of explanatory variables on bird density. Support for local, patch, and landscape habitat variables were evaluated using model coefficients and their 95% confidence intervals and we considered variables whose coefficients 95% CI overlapped zero to be weakly supported.

We used parametric bootstrapping to evaluate goodness-of-fit of the best model for each species. We simulated 100 data sets from our model and each time refit the model to these "perfect" data and computed a fit statistic. We then compared the value of the fit statistic for the observed data sets. For a model to fit, the observed value should not be too extreme (e.g., beyond

5th percentile of the reference distribution). We used the Freeman-Tukey fit statistic to quantify the fit of the model data set.

RESULTS

Survey patches ranged from 2.9 to 174.7 ha (mean = 35.3, SE = 2.6) from 2012-2015 resulting in 420, 573, 602, and 401point count surveys each year, respectively. Field Sparrows and Dickcissels were the most abundant (70% of observations) of the six focal bird species we examined followed by Willow Flycatcher, Bell's Vireo, Northern Bobwhite, and Yellow-billed Cuckoo (Table 2.6). Year influenced detection probability for all of our focal species and was included as a detection covariate in all models. Best-fitting key functions for detection were half-normal for Bell's Vireo and Willow Flycatcher, uniform for Northern Bobwhite and Yellow-billed Cuckoo, and exponential for Field Sparrow.

Multi-scale analysis

Our multi-scale analysis of land-cover types demonstrated that 75% of landscape variable tests indicated focal species densities were influenced by landscape variables measured at the 200-m scale (Table 2.2). However, the density of a few species were most influenced by landscape variables at broader scales including the Bell's Vireo and Yellow-billed Cuckoo which were influenced by the amount of forest and restored habitat within 1200 m of the surveyed patch, respectively (Table 2.2). Similarly, Field Sparrow, Yellow-billed Cuckoo, and Willow Flycatcher densities were most influenced by the amount of agriculture within 1200 m (Table 2.2).

Local scale

Vegetation structure and composition was highly variable among sample patches from sparse grasses to early successional forest with full canopy closure. Of the local scale variables evaluated, distance to the nearest tree was overwhelmingly supported as the best predictor for Bell's Vireo and Northern Bobwhite density (Table 2.3) with both species densities increasing as distance to nearest tree decreased. Field Sparrow and Yellow-billed Cuckoo were also strongly influenced by distance to nearest tree (Table 2.3), however for these species densities increased as distance to nearest tree increased. Percent tree cover was the highest ranked local scale variable influencing Dickcissel and Willow Flycatcher density (Table 2.3) with fewer birds in areas with increasing tree cover.

Patch scale

Bell's Vireo, Dickcissel, Northern Bobwhite, and Yellow-billed Cuckoo density was positively associated with increasing patch size (Figure 2.2), however for all but the Dickcissel 95% confidence intervals for the coefficients overlapped zero (Table 2.7). Conversely, Field Sparrow and Willow Flycatcher were more abundant in smaller patches (Figure 2.2) but the 95% CIs of the coefficients overlapped zero (Table 2.7).

Landscape context

Overall, models that included local vegetation and landscape level variables ranked better than those containing no or only local vegetation variables (Table 2.4). Restored habitat within 200 m was a strong negative predictor of Field Sparrow density (Figure 2.3) and appeared in models accounting for 75% of the Akaike weight (Table 2.4). The Bell's Vireo and Northern Bobwhite were also slightly less abundant in patches with increasing amount of restored habitat in the surrounding landscape (Figure 2.3). Conversely, Dickcissel density strongly increased with greater amounts of restored habitat in the surrounding landscape (Figure 2.3; Table 2.7).

Likewise, Willow Flycatchers and Yellow-billed Cuckoos were positively associated with restored habitat in the surrounding landscape (Figure 2.3), however the relationship was weak (95% CI of parameter estimate included zero). Bell's Vireo, Dickcissel, and Willow Flycatcher were strongly positively associated with the amount of grass in the surrounding landscape (Figure 2.5) and this variable appeared in models accounting for 91%, 99%, and 93% of the Akaike weight, respectively (Table 2.4). Similarly, Field Sparrows and Northern Bobwhite were positively associated with the amount of grass cover in the surrounding landscape however the Yellow-billed Cuckoo was slightly negatively associated with this variable (Table 2.7), however these relationships were weak. Forest cover negatively influenced Dickcissel and Northern Bobwhite densities (Figure 2.6). Conversely, Yellow-billed Cuckoo densities increased with increasing forest cover (Figure 2.6; Table 2.7). Surrounding agriculture was a positive predictor of Bell's Vireo, Dickcissel, Field Sparrow, Northern Bobwhite, and Willow Flycatcher densities; however Yellow-billed Cuckoo density was negatively associated with this variable (Table 2.7).

Agricultural context

The amount of crop cover in the surrounding landscape negatively influenced the effects of restored habitat for the Willow Flycatcher and the top model included additive and interactive effects of these variables (Table 2.5; Figure 2.4; interaction $\beta = -0.13$, SE = 0.06). For all other species except the Bell's Vireo, the top model included additive effects of crop cover and restored habitat (Table 2.5).

DISCUSSION

In this study, we assessed the importance of habitat and landscape features at multiple scales for birds using areas restored through private land programs to determine how these areas could be created or managed to target conservation of priority species. At the local scale, distance to

nearest tree and tree cover were the most important variables. Landscape features within 200 m of private land program patches were most influential for the majority of our focal species. Responses to the amount of restored habitat, patch size, and landscape composition were species specific, but for most species the amount of restored habitat within the surrounding landscape had little influence on density. Likewise, patch size generally did not influence density, suggesting that even small patches can provide suitable habitat for these species. Furthermore, landscape context was important for many of our focal species.

Given the importance of scale for habitat use in birds, we attempted to objectively choose among multiple scales which we would use in our multi-scale habitat analysis rather than following the popular convention of using a single scale to use in our landscape-level analysis (McGarigal et al. 2016). While potentially limited due to the inherent subjectivity of scale choice prior to model selection, we felt this method was the best existing alternative. Similar to Quinn et al. (2014) and Pabian et al. (2013, 2015) our results suggest that bird densities of our focal species in restored habitats were primarily influenced by land cover within a 200 m radius around patches rather than at broader scales. One possible explanation for these results could be that restored patches are embedded within largely uninhabitable agriculture- and forest-dominated matrix and therefore only landscape attributes at this scale provide cues for settlement. Ultimately, our results suggest that targeted implementation of private land programs in Illinois for early successional species should consider land uses within relatively close proximity of restored patches.

The mixed and generally neutral responses to the amount of restored habitat in the surrounding landscape were surprising given that previous studies found this was important to early successional species (e.g. Herkert 2007; Dallimer et al. 2010; Pabian et al. 2015). We found that only Dickcissel density responded positively to the amount of restored habitat in the surrounding landscape and that all other results were either equivocal or negative. As documented

in previous studies (Bakker et al. 2002; Renfrew and Ribic 2008; Pabian et al. 2013), we considered the negative relationships we observed to be the possible result of the unattractiveness of additional landscape features (i.e. other than restored habitat) surrounding restored patches. For example, the primary goal of the program we examined is to reduce agricultural runoff and improve water quality and so restored patches are frequently adjacent to riparian corridors which are typically forested (Table 2.1), however many of our focal species were negatively associated with forest cover. Thus, it appears that the amount of surrounding forest may override any effect of the amount of restored habitat in the surrounding landscape and so restored habitat may have more positive benefits in open landscapes. Alternatively, as Batáry et al. (2011) found in their review, the effect of surrounding restoration was more likely to be positive in simple landscapes dominated by agriculture (0 – 20 % semi-natural habitat) likely due to the matrix being inhospitable. However, Batáry et al.'s (2011) review was based on data from European landscapes where anthropogenic influences have a longer history of high-intensity influence than those in North America. In contrast, our study was conducted in complex landscapes ($\bar{x}$ =73.6% semi-natural habitat) which may explain why we had so few strong positive relationships to the amount of restored habitat in the surrounding landscape. Yet, we still found that the amount of surrounding cropland moderated the response of Willow Flycatcher to surrounding restored habitat (Figure 2.4) though restored habitat only had a positive effect in low-crop landscapes which is opposite of what was found in European landscapes. More work is needed in North America comparing the response of species to habitat restoration in simple vs. complex landscapes to determine if this is a generalizable phenomenon.

Although the effect of patch size was small for all of our focal species, we found evidence of increasing density of birds in larger patches for three of our species. Interestingly, of the three only the Northern Bobwhite has previously been identified as area sensitive (Brennan et al. 2014). Our responses were weak relative to results found in other studies of area sensitivity; however, in

studies where abundance is the response variable responses tend to be equivocal, and strong relationships of this phenomenon are more often detected when species occupancy is the focal response variable (e.g. Bayard and Elphick 2010). Our study focused on restored patches >3 ha, however past studies have found many early successional species occupying patches as small as 1 ha (e.g. Askins et al. 2007; Schlossberg and King 2007). In contrast, we found limited evidence that Willow Flycatcher and Field Sparrow were more abundant in smaller patches. For both grassland and shrubland bird species, most recommendations call for creating and maintaining large contiguous patches of habitat (e.g. Ribic et al. 2009; Schlossberg and King 2008). This is based on past work on grassland species which suggests that larger patches have greater density or occupancy (Ribic et al. 2009) as our results with Dickcissel corroborate; however, shrubland birds have been observed to lack area sensitivity (e.g. Shake et al. 2012). Even research on grassland birds demonstrates that these species can be productive even in small patches (e.g. Walk et al. 2010; Benson et al. 2013) and so even small patches may provide important conservation opportunities in agriculturally fragmented landscapes (e.g. Quinn et al. 2014). This is particularly important for private land programs, which often restore relatively small patches.

We found most of our focal species were negatively associated with forest cover, with exception of the Yellow-billed Cuckoo which had a positive association. The negative relationships for most shrubland species are notable given the frequent association of shrubland habitats with forested landscapes (e.g. Schlossberg and King 2008 but see Shake et al. 2012 and Quinn et al. 2014). For Yellow-billed Cuckoo the positive association with forested landscapes is in agreement with known habitat associations for this species (Hughes 2015). Similarly, we found most species were positively associated with grassland habitats, including strong positive associations for the Bell's Vireo and Willow Flycatcher (Table 2.7), which is supported by previous studies of these species (Sedgwick 2000; Kosciuch et al. 2008; Carey et al. 2009; Brennan et al. 2014; Hughes 2015). This suggests that habitat selection processes (e.g. Johnson

1980) for these species may focus on selecting breeding patches within relatively open landscapes. Overall this demonstrates that it may not be appropriate to broadly generalize that forest cover is an important feature of habitat selection for shrub-associated bird species and in some regions, such as the Midwestern U.S., we may need to think of two groups of shrubland species, those associated with forested landscapes and those that are more abundant in open landscapes.

CONCLUSION

While research with private land programs in North America has previously focused on grassland species, our results concur with Quinn et al. (2014) that private land habitat may provide important conservation opportunities for both shrubland and grassland species in agriculturally fragmented landscapes. Furthermore, shifting the focus of management efforts to include shrubland species, which require less regular management to maintain habitat quality, may provide the most conservation benefit for the lowest economic input. For example, in a separate analysis, we found that for most of our focal shrubland species (except Willow Flycatcher) densities generally increased with patch age from 4 to 16 years since habitat creation (oldest patches sampled). This is especially important for habitat created by CREP in Illinois which, as stated earlier, includes a permanent conservation stipulation for many patches enrolled in this program. In general, our results demonstrate that for most of our focal species larger habitat patches, while beneficial for some species, are not necessary and that these habitats need not be clustered to provide benefits to birds of conservation concern. Furthermore, to maximize densities of our focal species, managers should focus on restoring agricultural fields embedded in landscapes that are dominated by crop or grassland habitat (>50%) rather than forest. At the local scale, shrubland associated species will benefit from patchily planted shrubs or seedling

hardwood tree species and more grassland-dependent species will benefit from frequent mowing to reduce woody intrusion.

We urge managers and policy makers to improve conservation benefits of private land programs through targeted implementation. To do this, agency personnel could use information on bird habitat use in planning and management (e.g. Evans et al. 2014) of these programs in the U.S. as in Europe (e.g. Perkins et al. 2011). In addition to benefitting birds of conservation concern, targeted implementation could provide benefits to other wildlife species (e.g. Riffell et al. 2015) thus increasing overall biodiversity and improving ecosystem services by offsetting negative impacts of habitat fragmentation due to agriculture (Donald and Evans 2006). Furthermore, targeted implementation of these programs will provide a mechanism for measuring their conservation benefits and will help demonstrate their utility for addressing societal priorities which will justify their continued funding.

LITERATURE CITED (Formatted for Agriculture, Ecosystems, and Environment)

Adams, M.J., Miller, D.A.W., Muths, E., Corn, P.S., Campbell Grant, E.H., 2013. Trends in amphibian occupancy in the United States. PLoS ONE 8,e64347http://dx.doi.org/10.1371/journal.pone.0064347

Askins R.A., 1993. Population trends in grassland, scrub-shrub, and forest birds in eastern North America. Current Ornithology 11, 1–34.

Askins R.A., Zuckerberg, B., Novak, L., 2007. Do the size and landscape context of forest openings influence the abundance and breeding success of shrubland songbirds in southern New England? Forest Ecol. Manag. 250, 137–147.

Baker, D.J., Freeman, S.N., Grice, P.V., Siriwardena, G.M., 2012. Landscape-scale responses of birds to agrienvironment management: at test of the English Environmental Stewardship scheme. J. Appl. Ecol. 49, 871–882.

Batáry, P., Báldi, A., 2004. Evidence of an edge effect on avian nest success. Conserv. Biol. 18, 389–400.

Batáry, P., Báldi, A., Szél, G., Podlussany, A., Rozner, Erdős, S, 2007. Responses of grassland specialist and generalist beetles to management and landscape complexity Divers. Distrib. 13,196–202.

Batáry, P., Báldi, A., Kleijn, D., Tscharntke, T., 2011. Landscape moderated biodiversity effects of agri-environmental management: a meta-analysis. Proc. R. Soc. Lond.[Biol] 278, 1894–1902.

Bakker, K.K., Naugle, D.E., Higgins, K.F., 2002. Incorporating landscape attributes into models for migratory grassland bird conservation. Conserv. Biol. 16, 1638–1646.

Bender, D. J., Contreras, T.A., Fahrig, L., 1998. Habitat loss and population decline: a meta-analysis of the patch size effect. Ecology 79, 517–533.

Beyer, H.L. 2012. Geospatial Modelling Environment (Version 0.7.2.1). URL:

http://www.spatialecology.com/gme. Accessed 25 September, 2014.

Benson, T.J., Dinsmore, J.J., Hohman., W.L., 2007. Response of plants and arthropods to burning and disking of riparian habitats. J. Wildl. Manag. 71, 1949–1957.

Benson, T.J., Chiavacci, S.J., Ward, M.P. 2013. Patch size and edge proximity are useful predictors of brood parasitism but not nest survival of grassland birds. Ecol. Appl. 23, 879–887.

Böhm, M., Collen, B., Baillie, J.E.M., Bowles, P., Chanson, J., Cox, N., Hammerson, G., Hoffmann, M., Livingstone, S.R., Ram, M., Rhodin, A.G.J., Stuart, S.N., van Dijk, P.P., Young, B.E., Afuang, L.E., Aghasyan, A., Garcia, A., Aguilar, C., Ajtic, R., Akarsu, F., Alencar, L.R.V., Allison, A., Ananjeva, N., Anderson, S., Andren, C., Ariano-Sanchez, D., Arredondo, J.C., Auliya, M., Austin, C.C., Avci, A., Baker, P.J., Barreto-Lima, A.F., Barrio-Amoros, C.L., Basu, D., Bates, M.F., Batistella, A., Bauer, A., Bennett, D., Bohme, W., Broadley, D., Brown, R., Burgess, J., Captain, A., Carreira, S., Castaneda, M.D., Castro, F., Catenazzi, A., Cedeno-Vazquez, J.R., Chapple, D.G., Cheylan, M., Cisneros-Heredia, D.F., Cogalniceanu, D., Cogger, H., Corti, C., Costa, G.C., Couper, P.J., Courtney, T., Crnobrnja-Isailovic, J., Crochet, P.A., Crother, B., Cruz, F., Daltry, J.C., Daniels, R.I.R., Das, I., de Silva, A., Diesmos, A.C., Dirksen, L., Doan, T.M., Dodd, C.K., Doody, J.S., Dorcas, M.E., de Barros, J.D., Egan, V.T., El Mouden, E., Embert, D., Espinoza, R.E., Fallabrino, A., Feng, X., Feng, Z.J., Fitzgerald, L., Flores-Villela, O., Franca, F.G.R., Frost, D., Gadsden, H., Gamble, T., Ganesh, S.R., Garcia, M.A., Garcia-Perez, J.E., Gatus, J., Gaulke, M., Geniez, P., Georges, A., Gerlach, J., Goldberg, S., Gonzalez, J.C.T., Gower, D.J., Grant, T., Greenbaum, E., Grieco, C., Guo, P., Hamilton, A.M., Hare, K., Hedges, S.B., Heideman, N., Hilton-Taylor, C., Hitchmough, R., Hollingsworth, B., Hutchinson, M., Ineich, I., Iverson, J., Jaksic, F.M., Jenkins, R., Joger, U., Jose, R., Kaska, Y., Kaya, U., Keogh, J.S., Kohler, G., Kuchling, G., Kumlutas, Y.,

Kwet, A., La Marca, E., Lamar, W., Lane, A., Lardner, B., Latta, C., Latta, G., Lau, M., Lavin, P., Lawson, D., LeBreton, M., Lehr, E., Limpus, D., Lipczynski, N., Lobo, A.S., Lopez-Luna, M.A., Luiselli, L., Lukoschek, V., Lundberg, M., Lymberakis, P., Macey, R., Magnusson, W.E., Mahler, D.L., Malhotra, A., Mariaux, J., Maritz, B., Marques, O.A.V., Marquez, R., Martins, M., Masterson, G., Mateo, J.A., Mathew, R., Mathews, N., Mayer, G., McCranie, J.R., Measey, G.J., Mendoza-Quijano, F., Menegon, M., Metrailler, S., Milton, D.A., Montgomery, C., Morato, S.A.A., Mott, T., Munoz-Alonso, A., Murphy, J., Nguyen, T.Q., Nilson, G., Nogueira, C., Nunez, H., Orlov, N., Ota, H., Ottenwalder, J., Papenfuss, T., Pasachnik, S., Passos, P., Pauwels, O.S.G., Perez-Buitrago, N., Perez-Mellado, V., Pianka, E.R., Pleguezuelos, J., Pollock, C., Ponce-Campos, P., Powell, R., Pupin, F., Diaz, G.E.Q., Radder, R., Ramer, J., Rasmussen, A.R., Raxworthy, C., Reynolds, R., Richman, N., Rico, E.L., Riservato, E., Rivas, G., da Rocha, P.L.B., Rodel, M.O., Schettino, L.R., Roosenburg, W.M., Ross, J.P., Sadek, R., Sanders, K., Santos-Barrera, G., Schleich, H.H., Schmidt, B.R., Schmitz, A., Sharifi, M., Shea, G., Shi, H.T., Shine, R., Sindaco, R., Slimani, T., Somaweera, R., Spawls, S., Stafford, P., Stuebing, R., Sweet, S., Sy, E., Temple, H.J., Tognelli, M.F., Tolley, K., Tolson, P.J., Tuniyev, B., Tuniyev, S., Uzum, N., van Buurt, G., Van Sluys, M., Velasco, A., Vences, M., Vesely, M., Vinke, S., Vinke, T., Vogel, G., Vogrin, M., Vogt, R.C., Wearn, O.R., Werner, Y.L., Whiting, M.J., Wiewandt, T., Wilkinson, J., Wilson, B., Wren, S., Zamin, T., Zhou, K., Zug, G., 2013. The conservation status of the world's reptiles. Biol. Conserv. 157, 372–385.

Boryan, C., Yang, Z., Di, L., Hunter, K., 2014. A new automatic stratification method for U.S. agricultural area sampling frame construction based on the Cropland Data Layer, pp. 1939-1404, May 2014, DOI: 10.1109/JSTARS.2014.2322584.

Brennan, L.A., Engstrom, R.T., Palmer, W.E., Hermann, S.M., Hurst, G.A., Burger, L.W., Hardy, C. L., 1998. Whither wildlife with- out fire? Trans. N. Am. Wildl. Nat. Res. 63, 402–414.

Brennan, L.A., Hernandez, F., Williford, D., 2014. Northern Bobwhite (*Colinus virginianus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/397doi:10.2173/bna.397

Broughton, R.K., Shore, R.F.R., Heard, M.S., Meek, W.R., Redhead, J.W., Turk, A., Pywell, R.F., 2014. Agri-environment scheme enhances small mammal diversity and abundance at the farm-scale. Agric. Ecosyst. Environ. 192, 122–129.

Buckland, S.T., Burnham, D.R., Anderson, K.P., Laake, J.L., Borchers, D.L., Thomas., L., 2001. Distance Sampling: Estimating Abundance of Biological Populations. Chapman & Hall, London, UK.

Burnham, K.P., Anderson, D.R., 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.

Carey, M., Burhans, D.E. Nelson, D.A., 2008. FIELD SPARROW (*Spizella pusilla*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/103doi:10.2173/bna.103

Chandler, R.B., King, D.I., DeStefano, S., 2009. Scrub-shrub bird habitat associations at multiple spatial scales in beaver meadows in Massachusetts. Auk 126, 186–197.

Clay, J., 2004. World Agriculture and the Environment: A Commodity-by-Commodity Guide to Impacts and Practices. Island Press, Washington, D. C.

Concepción, E., Díaz, M., Baquero, R., 2008. Effects of landscape complexity on the ecological effectiveness of agri-environment schemes. Lands. Ecol. 23, 135–148.

Conover, R.R., Dinsmore, S.J., Burger Jr., L. W., 2011. Effects of conservation practices on bird density and survval in intensive agriculture. Agric. Ecosyst. Environ.141, 126–132.

Dallimer, M., Gaston, K.J., Skinner, A.M.J., Hanley, N., Acs, A., Armworth, P.R., 2010. Field-level bird abundances are enhanced by landscape-scale agri-environment scheme uptake. Biol. Lett. 6, 643–646.

Davey, C.M., Vickery, J.A., Boatman, N.D., Chamberlain, D.E., Parry, H.R. Siriwardena, G.M., 2010. Assessing the impact of Entry Level Stewardship on lowland farmland birds in England. Ibis, 152, 459–474

Donald, P.F., Evans, A.D., 2006. Habitat connectivity and matrix restoration: the wider implications of agri-environment schemes. J. Appl. Ecol. 43, 209–218.

Evans, K.O., Burger, L.W., Riffell, S., Smith, M.D., 2013. Assessing multi-region avian benefits from strategically targeted agricultural buffers. Conserv. Biol. 28, 892–901.

Ellison, K.S., Ribic, C.A., Sample, D.W., Fawcett, M. J., Dadisman, J. D., 2013. Impacts of tree rows on grassland birds and potential nest predators: a removal experiment. PLoS ONE 8, e59151.

ESRI, 2012. ArcGIS version 10.0. Earth Systems Research Institute, Redlands, California, USA.

Fahrig, L., 1997 Relative effects of habitat loss and fragmentation on population extinction. The J. Wildl. Manag. 61, 603–610.

Fahrig, L., 2001. How much habitat is enough? Biol. Conserv. 100, 65–74.

Fisher, R.J., Davis, S.K., 2010. From Wiens to Robel: a review of grassland-bird habitat selection. J. Wildl. Manag. 74, 265–273.

Fiske, I.J., Chandler, R.B., 2011 Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. J. Stat. Softw. 43, 1–23.

Fletcher Jr., R.J., Koford, R.R., 2002. Habitat and landscape associations of breeding birds in native and restored grasslands. J. Wildl. Manag. 66, 1011–1022.

Fletcher Jr., R.J., Koford, R.R., 2003. Changes in breeding bird populations with habitat restoration in Northern Iowa. Am. Midl. Nat. 150, 83–94.

Foley, J.A., N. Ramankutty, Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, G.D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342.

Heard, L.P., Allen, A.W., Best, L.B., Brady, S.J., Burger, W., Esser, A.J., Hackett, E., Johnson, D.H., Pederson, R.L., Reynolds, R.E., Rewa, C., Ryan, M.R., Molleur, R.T., Buck, P., 2000. A comprehensive review of Farm Bill contributions to wildlife conservation, 1985–2000. *In*: W.L. Hohman and D.J. Halloum, eds. United States Department of Agriculture, Natural Resources Conservation Service, Wildlife Habitat Management Institute, Technical Report, USDA/NRCS/WHMI-2000.

Herkert, J.R., 1995. An analysis of Midwestern breeding bird population trends: 1966 – 1993. Am. Midl. Nat. 134, 41–50.

Herkert, J.R., Sample, D.W., Warner, R.E., 1996. Management of Midwestern grassland landscapes for the conservation of migratory birds. In Managing Midwestern landscapes for the conservation of Neotropical migratory birds. United States Department of Agriculture Forest Service. Technical Report GTR-NC-187, North Central Forest Experiment Station, St. Paul, Minnesota, USA.

Herkert, J.R., 1998. The influence of the CRP on Grasshopper Sparrow population trends in the mid-continental United States. Wildl. Soc. Bull. 26, 227–231.

Herkert, J.R., 2007. Conservation Reserve Program benefits on HENSLOW'S SPARROWS within the United States. J. Wildl. Manag. 71, 2749–2751.

Hiron M., Berg, Å., Eggers, S., Josefsson, J., Pärt, T., 2013 Bird diversity relates to agri-environment schemes at local and landscape level in intensive farmland. Agric. Ecosyst. Environ. 176, 9–16.

Hughes, J.M., 2015. Yellow-billed Cuckoo (*Coccyzus americanus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/418 doi:10.2173/bna.418

Hunter, W.C., Buehler, D.A., Canterbury, R.A., Confer, J.L., 2001. Conservation of disturbance-dependent birds in eastern North America. Wildl. Soc. Bull. 29, 440–455.

Gray, R.L., Teels, B.M., 2006. Wildlife and fish conservation through the Farm Bill. Wildl. Soc. Bull. 34, 906–913.

Karp, D.S., Rominger, A.J., Zook, J., Ranganathan, J., Ehrlich, P.R., Daily, G.C., 2012. Intensive agriculture erodes b-diversity at large scales. Ecology Letters 15, 963–970.

Kleijn, D., Berendse, F., Smit, R., Gilissen, N., 2001. Agri-environment schemes do not effectively protect biodiversity in Dutch agricultural landscapes. Nature 413, 723–725

Kleijn, D., Sutherland, W.J., 2003. How effective are European agri-environmental schemes in conserving and promoting biodiversity. J. Appl. Ecol. 40, 947–969.

Kasciuch, K.L., Sandercock, B.K., 2008. Cowbird removals unexpectedly increase productivity of a brood parasite and the songbird host. Ecol. Appl. 18, 537–548.

Lahti, D. C., 2001. The “edge effect on nest predation” hypothesis after 20 years. Biol. Conserv. 99, 365–374.

Lehnen, S.E., Rodewald. A.D., 2013. Daily and seasonal movements of a scrub-shrub-obligate breeder in relation to mature forest edge habitat. Forest Ecol. Manag. 305, 112–119.

Martin, T.E., Paine, C.R., Conway, C.J., Hochachka, W.M., Allen, P., Jenkins., W., 1997. BBIRD field protocol. Montana Cooperative Wildlife Research Unit, University of Montana, Missoula.

Mazerolle, M.J., 2012. Package 'AICcmodavg', version 1.25. R Foundation for Statistical Computing, Vienna. www.R-project.org

McGarigal, K., Wan, H.Y., Zeller, K.A., Timm, B.C., Cushman, S.A., 2016. Multi-scale habitat selection modeling: a review and outlook. Landsc. Ecol. 31, 1161–1175.

Miller, J.R., Hobbs, R.J., 2007. Habitat restoration – do we know what we are doing? Restor. Ecol. 15, 382–390.

Osborne, D.C., Sparling, D.W., 2013. Multi-scale associations of grassland birds in response to cost-share management of conservation reserve program fields in Illinois. J. Wildl. Manag. 77, 920–930.

Pabian, S.E., Wilson, A.M., Brittingham, M.C., 2013. Mixed response of farmland birds to the conservation reserve enhancement program in Pennsylvania. J. Wildl. Manag. 77, 616–625.

Pabian, S.E., Wilson, A.M. Klinger, S.R., Brittingham, M.C., 2015. Pennsylvania's Conservation Reserve Enhancement Program benefits Ring-Necked Pheasants but not enough to reverse declines. J. Wildl. Manag. 79, 641–646.

Perkins, A.J., Maggs, H.E., Watson, A., Wilson J.D., 2011. Adaptive management and targeting of agrienvironment schemes does benefit biodiversity: a case study of the Corn Bunting *Emberiza calandra*. J. Appl. Ecol. 48, 514–522.

Pimm, S.L., Askins, R.A., 1995. Forest losses predict bird extinctions in Eastern North America. Proc. Natl. Acad. Sci. 92, 9343–9347.

Quinn, J.E., Johnson, R.J., Brandle. J.R., 2014. Identifying opportunities for conservation in embedded in cropland anthromes. Landsc. Ecol., 29, 1811–1819.

Ralph, C.J., Sauer, J.R., Droege, S., 1993. Managing and monitoring bird populations using point counts: standards and applications. USDA Forest Service General Technical Report PSW-GTR-149. Pacific Southwest Research Station, USDA Forest Service, Albany,

California.

Ralph, C.J., Sauer, J.R., Droege. S., 1995. Monitoring bird populations by point count. USDA Forest Service General Technical Report PSW-149.

Renfrew R.B., Ribic, C.A., 2008. Multi-scale models of grassland passerine abundance in a fragmented system in Wisconsin. Landsc. Ecol., 23, 181–193.

Ribic, C.A., Guzy, M.J., Sample, D.W., 2009. Grassland bird use of remnant prairie and conservation reserve program fields in an agricultural landscape in Wisconsin. Am. Midl. Nat. 161, 110–122.

Riffell, S.K., Scognamillo, D., Burger, L.W., 2008. Effects of the Conservation Reserve Program on Northern Bobwhite and grassland birds. Environ. Monit. Assess. 146, 309–323.

Riffell, S.K., Monroe, A.P., Martin, J.A., Evans, K.O., Burger, L.W., Smith, M.D., Minderman, J., 2015. Response of non-grassland avian guilds to adjacent herbaceous field buffers: testing the configuration of targeted conservation practices in agricultural landscapes. J. Appl. Ecol. 52, 300–309.

Robel, R.J., Briggs, J.N., Dayton, A.D., Hulburt, L.C., 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. J. Range Manage. 23, 295–297.

Robinson, S.K., Thompson III, F.R., Donovan, T.M., Whitehead, D.R., Faaborg, J., 1995. Regional forest fragmentation and the nesting success of migratory birds. Science 267, 1987–1990.

Royle, J.A., Dawson, D.K., Bates, S., 2004. Modeling abundance effects in distance sampling. Ecology 85, 1591–1597.

Sauer, J.R., Hines, J.E., Fallon, J., 2011. The North American Breeding Bird Survey, results and analysis 1966–2011, ver. 07.03. 2013. USGS Patuxent Wildlife Research Center, Laurel, Maryland.

Schlossberg, S., King, D.I., 2008. Are shrubland birds edge specialists? Ecol. Appl. 18, 1325–1330.

Schwegman, J.E., 1973. Comprehensive plan for the Illinois Nature Preserve System. Part 2. The natural divisions of Illinois. Illinois Nature Preserves Commission, Rockford, IL. 32 pp.

Sedgwick, J.A., 2000. Willow Flycatcher (*Empidonax traillii*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/533doi:10.2173/bna.533

Shake, C.S., Moorman, C. E., Riddle, J.D., Burchell, M.R., 2012. Influence of patch size and shape on occupancy by scrub-shrub birds. Condor 114, 268–278.

Temple, S.A., 2002. Dickcissel (Spiza americana), The Birds of North America (P. G. Rodewald, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America: https://birdsna.org/Species-Account/bna/species/dickci DOI: 10.2173/bna.703

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005 Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. Ecol. Lett. 8, 857–874.

United States Department of Agriculture, Farm Service Agency. 2017. "Conservation Enrollments Monthly Summary – JUNE 2017." Available: https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/Conservation/PDF/JUN%202017%20Summary.pdf

Vander Haegen, W.M., Schroeder, M.A., Chang, W.Y., Knapp, S.M., 2015. Avian abundance and reproductive success in the intermountain west: local-scale response to the conservation reserve program. Wildl. Soc. Bull. 39, 276–291.

Van Buskirk, J., Willi, Y., 2004. Enhancement of farmland biodiversity within set-aside land. Conserv. Biol. 18, 987–994.

Veech, J.A., 2006. A comparison of landscapes occupied by increasing and decreasing

populations of grassland birds. Conserv. Biol. 20, 1422–1432.

Walk, J.W., Ward, M.P., Benson, T.J., Deppe, J.L., Lischka, S., Bailey, S.D., Brawn, J.D., 2010a. Illinois Birds: A Century of Change. Illinois Natural History Survey Special Publication 31.

Walk, J.W., Kershner, E.L., Benson, T.J., Warner, R.E., 2010b. Nesting success of grassland birds in small patches in an agricultural landscape. Auk 127, 328–334.

Wentworth, K.L., Brittingham, M.C., Wilson, A.M., 2010. Conservation Reserve Enhancement Program fields: benefits for grassland and shrub-scrub species. J. Soil and Water Conserv. 65, 50–60.

Weldon, A.J., Haddad, N.M., 2005. The effects of patch shape on Indigo Buntings: evidence for an ecological trap. Ecology 86, 1422–1431.

Wiens, J., 1989. Spatial scaling in ecology. Funct. Ecol. 29, 541–557.

Wells, J.V., 2010. From the last of the large to the remnants of the rare: bird conservation at an ecoregional scale. In: Trombulak, S.C. Baldwin, R.F. (Eds.), Landscape-scale Conservation Planning. Springer, London, UK; New York, NY. USA, pp. 121–137.

Wilson, A.M., Brittingham, M.C., 2012. Initial response of bird populations to conservation grasslands in southern Pennsylvania. J. of Soil and Water Cons. 67, 59–66.

Wilcove, D.S., 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66, 1211–1214.

Winter, M., Johnson, D.H., Shaffer, J.A., Donovan, T.M., Svedarsky, W.D., 2006. Patch size and landscape effects on density and nesting success of grassland birds. J. Wildl. Manag. 70, 158–172

TABLES AND FIGURES

Table 2.1 Descriptions of variables used to predict early successional bird density in Illinois, 2012–2015.

Variables	Description	$\bar{x}$	SE	Range
Local				
Tree distance	Distance (m) to the nearest tree	17.47	1.26	0 - 300
% Grass	Percent grass cover within 5 m radius	34.47	0.77	0 - 100
% Green	Percent green cover within 5 m radius	77.63	1.01	0 - 100
% Shrub	Percent shrub cover within 5 m radius	4.79	0.28	0 - 0.78
% Tree	Percent tree cover within 5 m radius	17.47	0.97	0 - 100
VOR	Visual obstruction estimate measured using Robel pole	6.99	0.16	0 - 20
Patch				
Patch	Size of patch (ha)	19.47	1.29	2.98 - 122.22
Landscape				
Crop200	Proportion of crops within 200 m	0.11	0.00	0.0 - 0.59
Grassland200	Proportion of grass within 200 m	0.40	0.01	0.0 - 0.89
Forest200	Proportion of forest within 200 m	0.40	0.01	0.0 - 0.98
Restore200	Proportion of restored habitat within 200 m	0.03	0.01	0.03 - 0.96
Crop700	Proportion of crops within 700 m	0.12	0.00	0 - 0.5
Grassland700	Proportion of grass within 700 m	0.30	0.00	0.06 - 0.89
Forest700	Proportion of forest within 700 m	0.39	0.01	0.01 - 0.87
Restore700	Proportion of restored habitat within 700 m	0.41	0.01	0.01 - 0.70
Crop1200	Proportion of crops within 1200 m	0.28	0.01	0 - 0.81
Grassland1200	Proportion of grass within 1200 m	0.25	0.00	0.06 - 0.89
Forest1200	Proportion of forest within 1200 m	0.37	0.01	0.03 - 0.81
Restore1200	Proportion of restored habitat within 1200 m	0.17	0.00	0.01 - 0.48

Table 2.2 Number of parameters, ΔAIC_c values, and model weights for 3 different landscape scales to be used in early successional bird models for four land-cover types, cropland(Crop), forest (Forest), grassland (Grass), and restored habitat (Restore) measured at 3 radii (200, 700, and 1200m) centered on the patch boundary.

Model	BEVI[a]			FISP			DICK			NOBO			WIFL			YBCU		
	K	ΔAIC_C	w_i	*K*	ΔAIC_C	w_i	*K*	ΔAIC_C	w_i	*K*	ΔAIC_C	w_i	*K*	ΔAIC_C	w_i	*K*	ΔAIC_C	w_i
Crop200	7	0.48	0.31	7	1.96	0.17	7	**0.00**	**1.00**	7	**0.00**	**0.85**	7	2.39	0.19	7	7.06	0.03
Crop700	7	1.80	0.16	7	1.98	0.17	7	10.52	0.00	7	6.21	0.04	7	3.11	0.13	7	4.91	0.08
Crop1200	7	2.02	0.14	7	1.49	0.21	7	4.89	0.00	7	4.23	0.10	7	**0.00**	**0.62**	7	**0.00**	**0.89**
Null	6	**0.00**	**0.39**	6	**0.00**	**0.45**	6	19.06	0.00	6	8.92	0.01	6	4.53	0.06	6	11.44	0.00
Forest200	7	**0.00**	**0.47**	7	**0.00**	**0.49**	7	**0.00**	**1.00**	7	**0.00**	**0.97**	7	**0.00**	**0.97**	7	**0.00**	**0.70**
Forest700	7	0.22	0.42	7	1.66	0.22	7	18.26	0.00	7	6.79	0.03	7	8.14	0.02	7	4.69	0.07
Forest1200	7	4.28	0.06	7	3.48	0.09	7	31.56	0.00	7	20.62	0.00	7	9.08	0.01	7	4.79	0.06
Null	6	4.64	0.05	6	1.79	0.20	6	51.19	0.00	6	35.56	0.00	6	21.03	0.00	6	2.84	0.17
Grass200	7	**0.00**	**0.83**	7	1.51	0.21	7	**0.00**	**1.00**	7	**0.00**	**1.00**	7	**0.00**	**1.00**	7	**0.00**	**0.38**
Grass700	7	3.26	0.16	7	1.97	0.17	7	21.37	0.00	7	11.55	0.00	7	20.26	0.00	7	2.06	0.14
Grass1200	7	10.72	0.00	7	1.91	0.17	7	22.59	0.00	7	15.25	0.00	7	24.65	0.00	7	2.16	0.13
Null	6	11.54	0.00	6	**0.00**	**0.45**	6	20.86	0.00	6	14.53	0.00	6	24.74	0.00	6	0.20	0.35
Restore200	7	5.86	0.04	7	**0.00**	**0.50**	7	**0.00**	**0.80**	7	0.33	0.30	7	**0.00**	**0.52**	7	2.46	0.11
Restore700	7	2.63	0.20	7	0.72	0.35	7	3.67	0.13	7	0.93	0.22	7	4.33	0.06	7	1.37	0.20
Restore1200	7	**0.00**	**0.73**	7	3.29	0.10	7	6.30	0.03	7	2.02	0.13	7	0.51	0.41	7	**0.00**	**0.39**
Null	6	5.93	0.04	6	4.32	0.06	6	6.32	0.03	6	**0.00**	**0.35**	6	8.47	0.01	6	0.54	0.30

[a] Species names abbreviated according to American Ornithological Society four letter code. *BEVI* Bell's Vireo, *FISP* Field Sparrow, *NOBO* Northern Bobwhite, *WIFL* Willow Flycatcher, *YBCU* Yellow-billed Cuckoo.

Table 2.3. Number of parameters, ΔAIC_c values, and model weights for models that incorporated local variables. See Table 1 for variable description.

Model	BEVI[a]			FISP			DICK			NOBO			WIFL			YBCU		
	K	ΔAIC_C	*wi*	*K*	ΔAIC_C	*wi*	*K*	ΔAIC_C	*wi*	*K*	ΔAIC_C	*wi*	*K*	ΔAIC_C	*wi*	*K*	ΔAIC_C	w_i
Tree distance	7	**0.00**	**0.94**	7	**0.00**	**0.99**	5	23.76	0.00	5	**0.00**	**1.00**	6	25.84	0.00	5	**0.00**	**1.00**
% Grass	7	16.95	0.00	7	14.13	0.00	5	47.54	0.00	5	32.19	0.00	6	31.35	0.00	5	32.19	0.00
% Green	7	16.99	0.00	7	11.67	0.00	5	51.30	0.00	5	32.60	0.00	6	17.65	0.00	5	32.60	0.00
% Shrub	7	5.58	0.06	7	11.09	0.00	5	40.79	0.00	5	32.18	0.00	6	30.42	0.00	5	32.18	0.00
% Tree	7	15.66	0.00	7	11.43	0.00	5	**0.00**	**1.00**	5	25.82	0.00	6	**0.00**	**1.00**	5	25.82	0.00
VOR	7	15.29	0.00	7	14.52	0.00	5	34.03	0.00	5	31.12	0.00	6	31.34	0.00	5	31.12	0.00
Null	6	15.10	0.00	6	13.69	0.00	4	49.61	0.00	4	32.03	0.00	5	30.29	0.00	4	32.03	0.00

[a] Species names abbreviated according to American Ornithological Society four letter code. *BEVI* Bell's Vireo, *FISP* Field Sparrow,*DICK* Dickcissel, *NOBO* Northern Bobwhite, *WIFL* Willow Flycatcher, *YBCU* Yellow-billed Cuckoo

Table 2.4 Number of parameters, ΔAIC_c values, and model weights for models that incorporated landscape variables. Variables include: the local model (Local), grassland cover within 200 m (G), patch size (P), forest cover within 200 m (F), crop cover (C), and restored habitat cover (R).

Model	BEVI[a, b]			DICK			FISP[c]			NOBO[b]			WIFL[c]			YBCU[b, c]		
Landscape	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i
Null	5	26.67	0.00	5	135.64	0.00	5	16.27	0.00	5	33.37	0.00	5	55.58	0.00	5	54.12	0.00
Local[b]	7	13.64	0.00	7	58.01	0.00	7	4.40	0.03	7	34.98	0.00	7	25.19	0.00	7	10.96	0.00
Local + P[c]	8	9.25	0.00	8	51.20	0.00	8	5.06	0.02	8	35.14	0.00	8	27.15	0.00	8	12.11	0.00
Local + P^2	8	9.25	0.00	8	51.20	0.00	8	5.06	0.02	8	35.14	0.00	8	27.15	0.00	8	12.11	0.00
Local + F[d]	8	12.33	0.00	8	36.80	0.00	8	3.19	0.05	**8**	**0.00**	**0.36**	8	6.94	0.01	8	9.42	0.00
Local + R[e]	8	13.89	0.00	8	55.03	0.00	**8**	**0.00**	**0.27**	8	37.00	0.00	8	16.85	0.00	8	10.41	0.00
Local + C[f]	8	13.55	0.00	8	27.85	0.00	8	5.40	0.02	8	26.32	0.00	8	24.15	0.00	**8**	**0.00**	**0.32**
Local + G[g]	8	2.47	0.12	8	42.55	0.00	8	5.97	0.01	8	20.38	0.00	8	2.71	0.11	8	11.53	0.00
Local + G + P	9	2.15	0.12	9	37.33	0.00	9	7.01	0.00	9	22.26	0.00	9	3.59	0.07	9	11.76	0.00
Local + F + P	9	9.63	0.00	9	24.33	0.00	9	4.70	0.02	**9**	**1.36**	**0.18**	9	8.24	0.00	9	9.11	0.00
Local + R + P	9	10.93	0.00	9	50.72	0.00	9	2.02	0.10	9	37.13	0.00	9	17.69	0.00	9	12.12	0.00
Local + C + P	9	9.56	0.00	9	18.30	0.00	9	5.91	0.01	9	25.33	0.00	9	26.11	0.00	9	0.56	0.24
Local + C + G	**9**	**0.00**	**0.35**	**9**	**0.00**	**0.99**	**9**	7.06	0.00	9	2.64	0.10	**9**	**0.00**	**0.43**	**9**	**0.37**	**0.27**
Local + R + C	9	12.46	0.00	9	16.43	0.01	**9**	**1.89**	**0.10**	9	27.51	0.00	9	9.48	0.00	**9**	**1.77**	**0.13**
Local + R + G	9	3.23	0.08	9	44.60	0.00	**9**	**1.00**	**0.16**	9	20.50	0.00	9	4.39	0.05	9	9.30	0.00
Local + F +G	9	3.89	0.06	9	29.55	0.00	9	4.43	0.03	**9**	**1.99**	**0.13**	9	2.43	0.13	9	11.31	0.00
Local + R + P + G	10	2.24	0.12	10	39.08	0.00	10	2.96	0.06	10	22.12	0.00	10	4.65	0.04	10	10.35	0.00
Local + R + P + F	10	11.13	0.00	10	24.50	0.00	10	3.09	0.06	**10**	**1.70**	**0.16**	**10**	4.87	0.04	10	8.22	0.00
Local + F + P + G	10	3.87	0.06	10	18.79	0.00	10	5.57	0.02	10	3.40	0.06	10	2.96	0.10	10	11.11	0.00

[a] *BEVI* Bell's Vireo, *FISP* Field Sparrow, *DICK* Dickcissel, *NOBO* Northern Bobwhite, *WIFL* Willow Flycatcher, *YBCU* Yellow-billed Cuckoo.
[b] Model set included restored habitat cover within 1200 m.
[c] Model set included crop cover within 1200 m.
[d] Model includes year and best fit local scale variable.

Table 2.5 Number of parameters, ΔAIC_c values, and model weights for models that incorporated surrounding crop cover and restored habitat. Variables include: crop cover (C), and/or restored habitat cover (R).

Model	BEVI[a,b]			DICK			FISP[c]			NOBO[b]			WIFL[c]			YBCU[b,c]	
Landscape	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i	K	ΔAIC_C	w_i
Null	**5**	**0.00**	**0.57**	5	66.76	0	5	0.32	0.33	5	5.89	0.03	5	24.82	0	5	21.8	0
Year	6	2.04	0.21	6	26.33	0	6	2.28	0.12	6	6.66	0.02	6	25.31	0	6	9.83	0.01
Year + R + C	9	2.99	0.13	9	**0.00**	**0.51**	9	**0.00**	**0.39**	9	**0.00**	**0.61**	9	2.97	0.18	9	**0.00**	**0.71**
Year + R × C	8	3.63	0.09	8	0.11	0.49	8	1.76	0.16	8	1.17	0.34	**8**	**0.00**	**0.81**	8	1.86	0.28

[a] Species names abbreviated according to American Ornithological Society four letter code. *BEVI* Bell's Vireo, *FISP* Field Sparrow, *DICK* Dickcissel, *NOBO* Northern Bobwhite, *WIFL* Willow Flycatcher, *YBCU* Yellow-billed Cuckoo.
[b] Model set included restored habitat cover within 1200 m.
[c] Model set included crop cover within 1200 m.

Table 2.6 Total numbers of observations and average density (birds/ha) of each focal species in west-central Illinois, 2012–2015.

Species	2012	2013	2014	2015	*n*	Average density±SE (birds/ha)
Bell's Vireo	64	119	103	108	394	0.10 ± 0.01
Dickcissel	500	391	458	146	1495	1.79 ± 0.18
Field Sparrow	595	714	706	548	2563	2.84 ± 0.39
Northern Bobwhite	118	93	61	71	343	0.73 ± 0.16
Willow Flycatcher	144	197	177	203	721	0.39 ± 0.07
Yellow-billed Cuckoo	48	49	142	63	292	1.83 ± 0.64

Table 2.7 Model-averaged parameter estimates and 95% confidence intervals for standardized landscape-scale variables used to predict species density.

Model	BEVI[a]			DICK			FISP			NOBO			WIFL			YBCU		
	β	95 % CI		β	95 % CI		β	95 % CI		β	95 % CI		β	95 % CI		β	95 % CI	
Intercept	0.12	0.10	0.14	0.19	-0.01	0.38	2.89	2.19	3.81	0.90	0.59	1.38	0.50	0.34	0.74	2.08	1.04	4.15
Crop[b]	**0.18**[c]	**0.01**	**0.36**	**0.42**	**0.29**	**0.55**	0.02	-0.06	0.11	**0.36**	**0.20**	**0.52**	**0.13**	**0.01**	**0.25**	**-0.26**	**-0.04**	**-0.11**
Forest	-0.06	-0.25	0.12	**-0.31**	**-0.45**	**-0.18**	0.07	-0.03	0.18	**-0.61**	**-0.83**	**-0.39**	-0.17	-0.37	0.03	**0.15**	**0.00**	**0.29**
Grass	**0.35**	**0.14**	**0.56**	**0.36**	**0.23**	**0.49**	0.05	-0.06	0.16	0.18	-0.34	0.70	**0.30**	**0.14**	**0.47**	-0.08	-0.20	0.04
Restored	-0.14	-0.36	0.09	**0.24**	**0.11**	**0.38**	**-0.11**	**-0.20**	**-0.02**	-0.13	-0.31	0.06	0.09	-0.09	0.27	0.04	-0.09	0.17
Patch size	0.15	-0.03	0.32	**0.21**	**0.09**	**0.33**	-0.02	-0.11	0.07	0.08	-0.10	0.26	-0.07	-0.19	0.04	0.08	-0.05	0.20

[a] Species names abbreviated according to American Ornithological Society four letter code. *BEVI* Bell's Vireo, *FISP* Field Sparrow, *DICK* Dickcissel, *NOBO* Northern Bobwhite, *WIFL* Willow Flycatcher, *YBCU* Yellow-billed Cuckoo.

[b] See table 4 for variable descriptions.

[c] Bold coefficients indicate parameter estimates with confidence intervals that do not overlap zero.

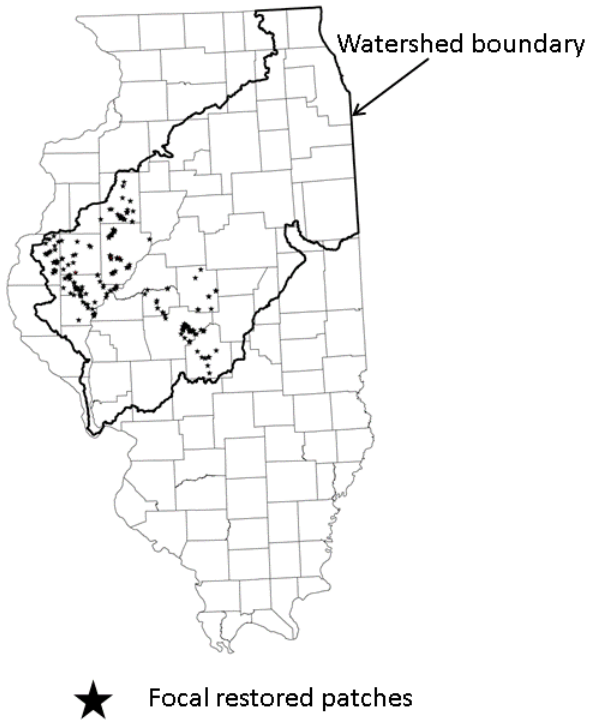

Figure 2.1 Map of Illinois displaying restored patches where point counts were conducted in 2012–2015.

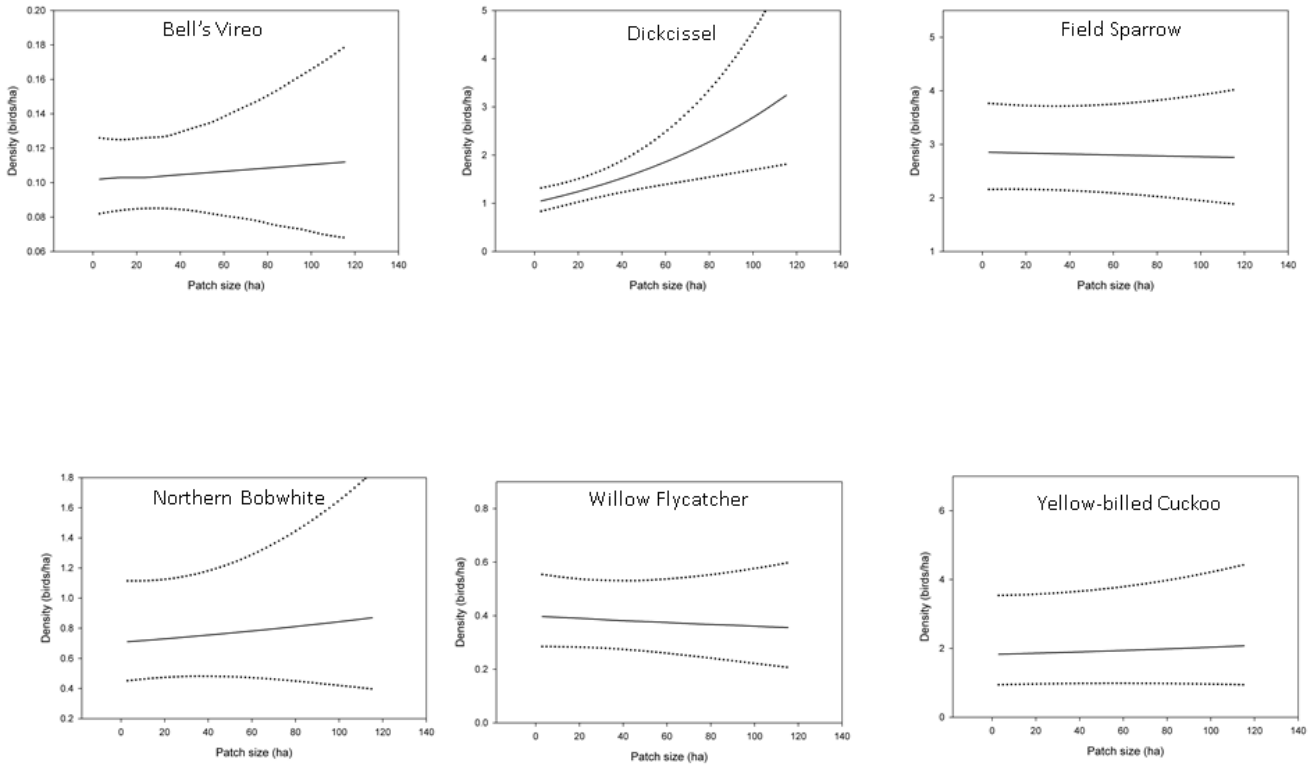

Figure 2.2 Model-averaged density (birds/ha) and 95% confidence intervals in relation to patch size (ha) for the six focal species.

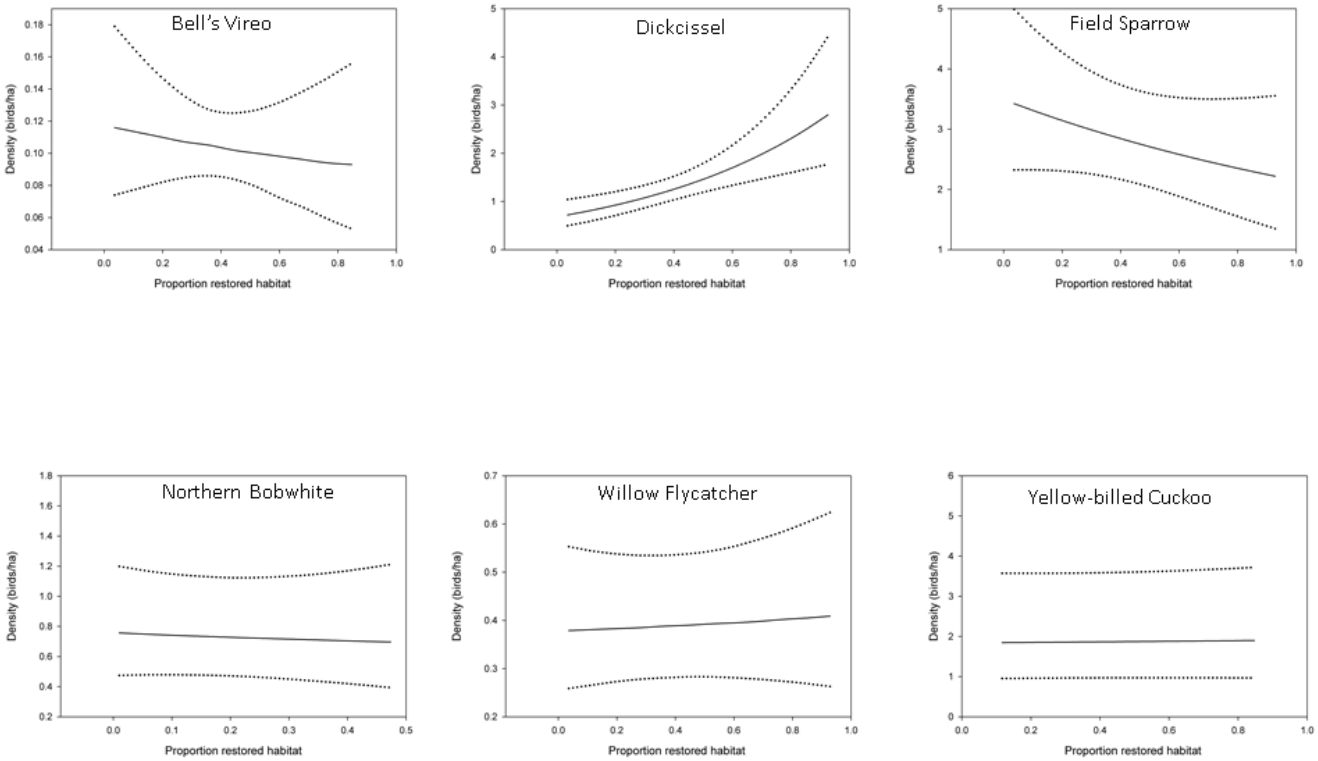

Figure 2.3 Model-averaged density (birds/ha) and 95% confidence intervals in relation to the proportion of restored habitat within 200 m for all species except Bell's Vireo which was in relation to restored habitat within 1200 m.

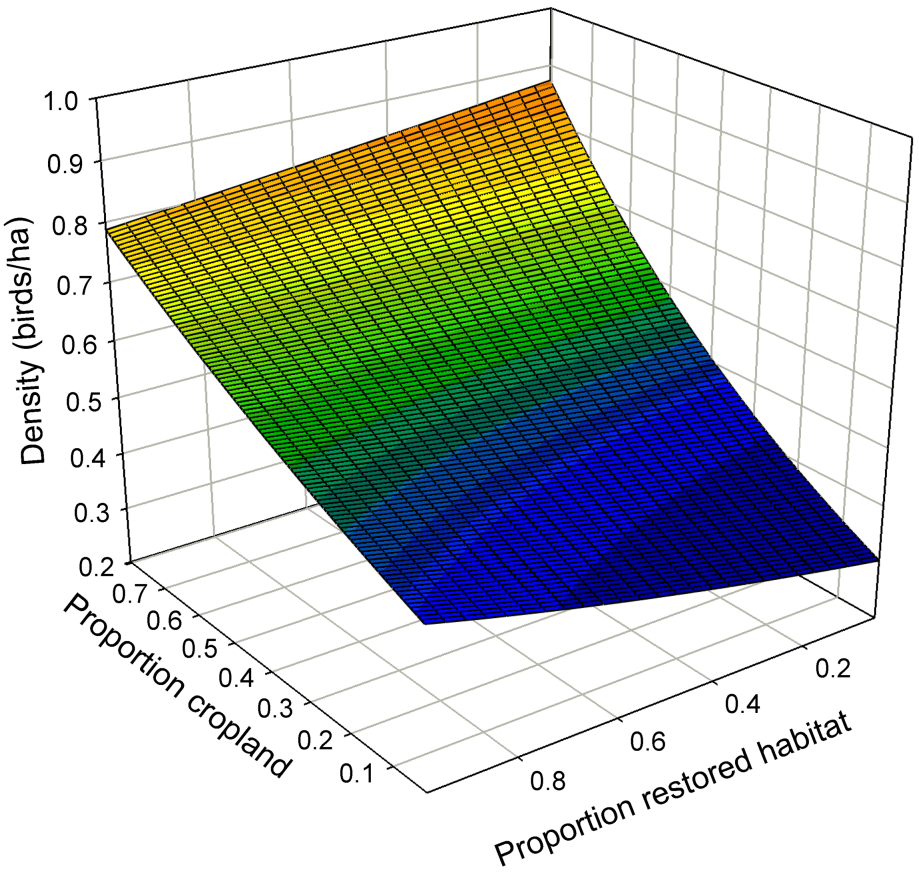

Figure 2.4 Model-averaged density (birds/ha) as a function of proportion cropland within 1200 m and restored habitat within 200 m for the Willow Flycatcher.

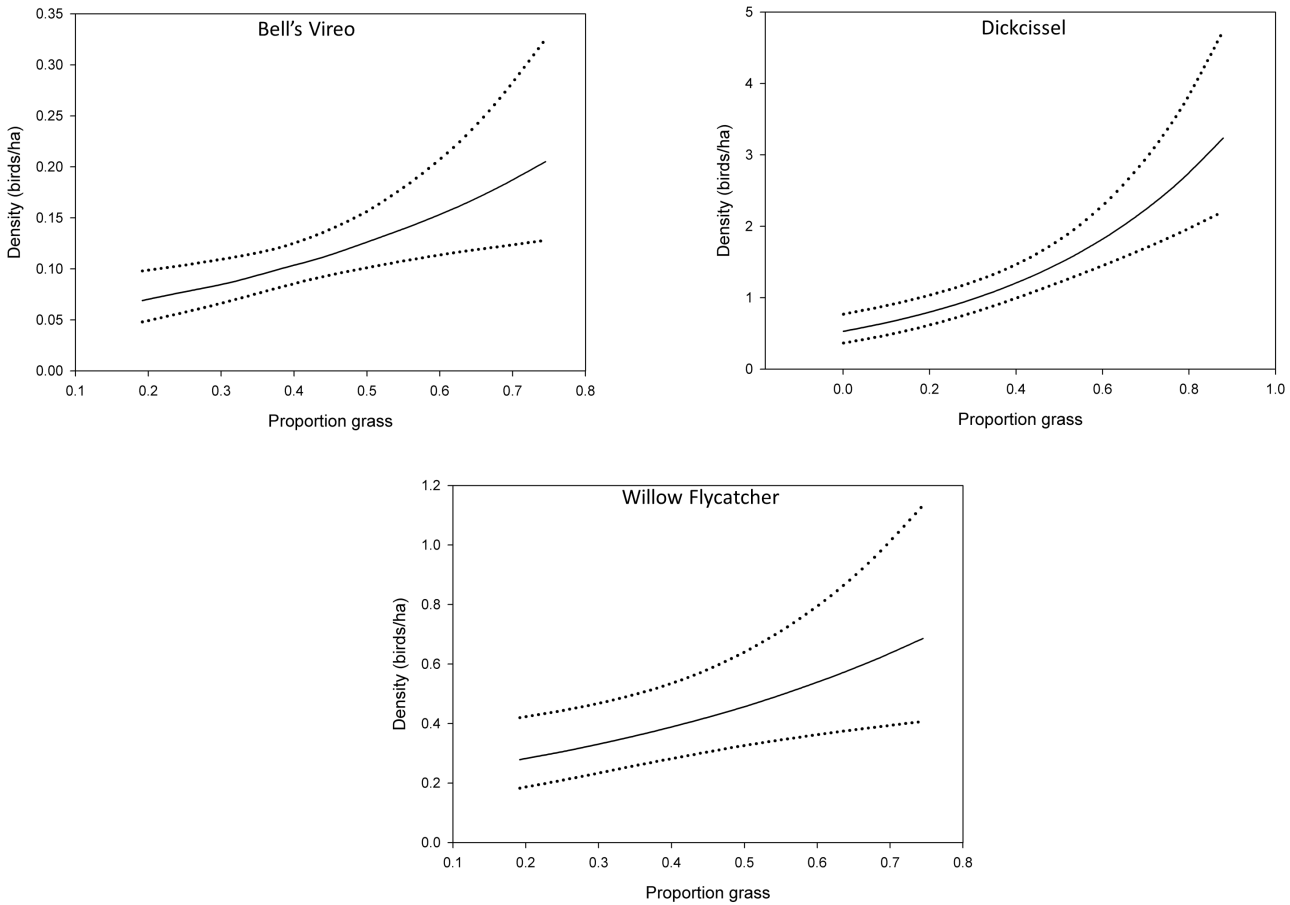

Figure 2.5 Model-averaged density (birds/ha) and 95% confidence intervals in relation to the proportion of grass cover within 200 m for three focal species. Only species with strong (coefficient confidence interval did not overlap zero) relationships shown.

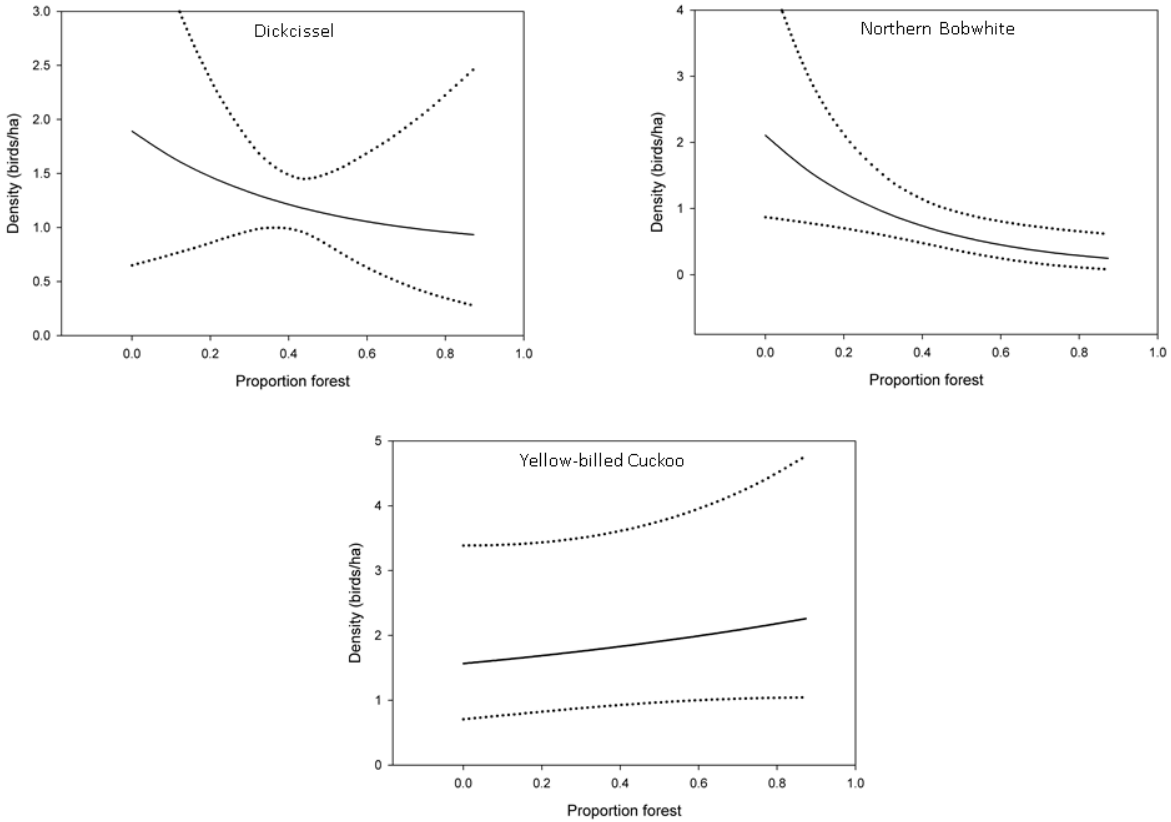

Figure 2.6 Model-averaged density (birds/ha) and 95% confidence intervals in relation to the proportion of forest cover within 200 m for three focal species. Only species with strong (coefficient confidence interval did not overlap zero) relationships shown.

CHAPTER 3: ARE AVIAN POPULATION TARGETS TOO OPTIMISTIC?

ABSTRACT: Agricultural intensification has negatively affected biodiversity throughout the world. In the U.S., population declines of many early successional bird species have been linked with habitat loss due to agriculture. In light of this, recent efforts have attempted to prioritize conservation actions at continental scales. Importantly, these efforts have used existing data to estimate the global population size for declining species and set future population goals. Though these efforts consider broad scales for conservation efforts and provide population targets, they lack explicit plans to achieve those goals. One way to achieve these goals is through the process of habitat restoration. In particular, voluntary private-land programs that restore habitat in agricultural landscapes may be implemented at a sufficiently large scale to achieve population goals. In order to understand how private-land restoration programs can be used to achieve population goals for declining species we examined the population-level impacts of the Conservation Reserve Enhancement Program, a program that has created more than 57,000 hectares of restored habitat in Illinois, for four conservation priority species, the Bell's Vireo, Field Sparrow, Northern Bobwhite, and Willow Flycatcher. We surveyed 172 randomly chosen restored fields in 10 counties in central and west-central Illinois during the 2012-2015 breeding seasons. We found that current restoration efforts in Illinois may be achieving state population goals for the Bell's Vireo and are close to achieving goals for the Willow Flycatcher, but more than 10 times the current amount of restored habitat would be required to achieve the Field Sparrow and Northern Bobwhite population goals. Our results suggest that achieving population goals for some species may be unrealistic.

INTRODUCTION

Ecosystem alteration due to anthropogenic land use has driven widespread loss of biodiversity (e.g. Pimm and Askins1995). One of the primary causes of this biodiversity loss has been increased land use for agricultural production which now makes up more than 38% of terrestrial land cover worldwide (Clay 2004). In response, conservation efforts have focused on creating protected areas to conserve biodiversity (e.g. Pullin 2002). While important, these efforts tend to be limited in geographical extent, and effective conservation efforts are needed at broader scales such as state, regional, or continental (Millspaugh and Thompson 2009).

Widespread agricultural conversion has led to population declines in many wildlife species (e.g. Reidsma et al. 2006). For example, populations of many early successional bird species have declined by between 50 - 100% since the start of the North American Breeding Bird Survey in 1966 (Sauer et al. 2014). In light of this, recent efforts have attempted to prioritize conservation actions for migratory birds (e.g. Carter et al. 2000, Faaborg et al. 2010). One aspect of this effort was to estimate global population size of many declining migratory species at continental scales using a temporally and spatially extensive data set (Rosenberg and Blancher 2005). This effort helped prioritize North American conservation for birds by estimating both current and historical population sizes. Estimates were used to establish conservation objectives such as setting aspirational population goals that aim to achieve pre-decline levels over the next 30 years (Rich et al. 2004). The intention behind this approach has been to tie conservation actions to measureable population goals, and accordingly, these large-scale goals have been "stepped-down" to regional and state-level objectives (e.g. IDNR 2005, Potter et al. 2007). Although these efforts consider broad scales for conservation efforts and provide population targets, they lack explicit plans to achieve those goals (Wells 2010).

Since the establishment of population targets, independent efforts (e.g. Mahon et al. 2014, Thogmartin et al. 2014) have focused on identifying ways to achieve these goals, largely focusing on increasing the amount of breeding habitat. This is based on the assumption that habitat availability on the breeding grounds is the main factor limiting bird

populations (e.g. Will et al. 2005, Rushing et al. 2016). Indeed, previous work suggests that increasing the amount of habitat would be the most effective way to achieve population goals (Thogmartin et al. 2014). Given this, habitat restoration may be the most effective mechanism to reverse population declines (Morrison et al. 1992). For economic reasons, buying and restoring large amounts of land to natural ecosystems may be infeasible. However, existing restoration programs, such as agri-environmental schemes (Europe) or Farm Bill conservation programs (North America), that pay farmers to restore habitat in agricultural landscapes (hereafter private land programs) may occur at sufficiently broad scales to achieve these population goals (e.g. van Buskirk and Willi 2004; Herkert 2007). For example, in North America the Conservation Reserve Program and Wetlands Reserve Program have led to the establishment of 3 million ha of potentially beneficial wildlife habitat where there was previously agricultural land (USDA 2017). Private land programs are generally designed for multiple purposes, including decreased commodity production, improvements in water quality, and creation of wildlife habitat (Hohman and Holloum 2000, Gray and Teels 2006). Importantly, these programs have been shown to affect population trends of conservation priority wildlife species (e.g. Van Buskirk and Willi 2004, Herkert 2007, Perkins et al. 2011).

Research has demonstrated that priority wildlife species are using areas restored by private land programs (e.g. Van Buskirk and Willi 2004, Herkert 2007). Additionally, we know that patches restored through these programs have greater abundances of species than the crop fields they replaced (e.g. Best et al. 1997). We also know that they have been shown to have comparable abundances to natural (non-restored) habitats (e.g. Fletcher and Koford 2002, Ribic et al. 2009). Given their scale, we assume that private land programs are contributing to achieving conservation goals yet we do not understand the magnitude of this contribution. Thus, we sought to understand the potential of restored habitats to provide state level population benefits to bird species of conservation concern and what additional programmatic investments would be necessary to achieve these goals. To do this, we selected a set of focal species representing scrub/shrub and grassland birds with stated population goals (IDNR 2005, Potter et al. 2007) due to precipitous population declines (Sauer et al.

2014). For these species, we developed spatially explicit estimates of bird abundance within patches enrolled in a private land program. We used these estimates to quantify the contribution of these habitats to meeting population targets within Illinois (IDNR 2005) for our focal species and, in cases where population goals were not being met, what additional amount of habitat and the approximate cost would be required to meet the goals.

METHODS

Study species

We focused on avian species of concern that had state level population goals (IDNR 2005) and for which we had sufficient data (>30 detections) for analyses (Table 3.1). The 4 focal species, which have declined between >50 % since the start of the North American Breeding Bird Survey in 1966, were Bell's Vireo (*Vireo bellii bellii*), Field Sparrow (*Spizella pusilla*), Northern Bobwhite (*Colinus virginianus*), and Willow Flycatcher (*Empidonax trailli trailli*) (Table 3.1).

Study sites

One private land program that provides a unique opportunity to understand the potential of habitat restoration for achieving population targets is the Conservation Reserve Enhancement Program (CREP). This program was initiated in 1996 within the Conservation Reserve Program (CRP) to allow North American states to address specific environmental priorities. CREP is a voluntary program that gives landowners a financial incentive to convert cropland into perennial vegetation. Within CREP, states were able to set their priorities and the structure of the program within that state. In 1998, Illinois entered into CREP with the explicit goals of reducing erosion and agricultural runoff, and increasing habitat for native wildlife. This program initially started in the Illinois River watershed and later in the Kaskaskia River watershed. Importantly, more than 57,000 ha in Illinois have been enrolled in CREP, almost half of which will become permanent conservation easements (IDNR 2012).

We surveyed for birds at 172 randomly chosen restored patches in central and west-central Illinois. Restored patches were >3 ha and randomly selected from among four conservation practice (CP) types, CP22 (riparian buffer; 26% of sampled fields), CP23 (wetland restoration; 37%), CP3A (hardwood tree plantings; 21%), and CP4D (permanent wildlife habitat; 16%) which were the most common practices (comprising 79% of all enrolled CREP fields). Importantly, sample patches, even though designated under different administrative CPs, were generally visually indistinguishable from one another based on habitat structure, and preliminary analysis demonstrated that CP was not an important predictor of density for our focal species (*B. Reiley unpublished*). Restored patches were chosen in 10 focal counties in the central and western part of the state where enrollments of this program were most common. Sample counties fell into two clusters, one centered on Macomb, IL and the other around Springfield, IL (Fig.3.1). The Macomb cluster consisted of Knox, McDonough, Fulton, Hancock, Brown, and Schuyler counties, which are generally in the Western Forest-Prairie natural division (Schwegman 1973) that was historically characterized by prairie, with forests associated with tributaries of the Illinois River. The Springfield cluster (hereafter Springfield) included Sangamon, Menard, Logan, and Christian counties, which fall within the Grand Prairie natural division (Schwegman 1973) and is characterized by deep loess soil deposits. Restored patches in this area were frequently adjacent to floodplain forest of the lower Sangamon River and row-crop agriculture. Patches in this study were converted from row crops into natural vegetation between 1999 and 2007; and sample patches contained early successional vegetative communities with structure ranging from grassland habitats consisting of warm and cool season grasses and forb cover, to dense scrub-shrub habitat dominated by shrubs and hardwood trees (planted oaks (*Quercus* spp.) or naturally colonizing species). Farmland restored patches in both regions were frequently adjacent to riparian forests, row-crop agriculture, and pastures.

Bird surveys

Patches were sampled from mid-May to mid-July of 2012 – 2015. To maximize visual and auditory detections of rare species, birds we surveyed using unlimited-radius point counts, 10 minutes in duration (Ralph et. al.1995). We recorded all focal species detections including the estimated distance to detected individuals and sex (if possible). Observers used laser range finders prior to each survey to calibrate distances based on features within patches. Surveys were conducted between sunrise and 1100 CDT, and not conducted during inclement weather (i.e. rain or wind (Beaufort scale > 4) Ralph et al. 1993). Between 1 and 3 point-count locations ($\bar{x}$ =1.09, SE = 0.02) were chosen randomly within each patch using ArcGIS (ESRI 2012) and were > 70 m away from patch edges and > 300 m from other point count locations. Bird surveys were conducted 1-4 times per season and visits were separated by >14 days. Surveys were conducted by 3 observers in 2012 and 2013 and 4 observers in 2014 and 2015; for each year of the study observers were rotated among patches to minimize potential bias.

Landscape data

Based on previous work we know that the density of birds in these restored patches was influenced by surrounding land cover (Chapter 2). Specifically, the most important landscape variables for our focal species were proportion grass (+), forest (-), and restored habitat (-) in the surrounding landscape. To extrapolate to unsampled patches enrolled in CREP, we quantified the proportion of these variables for all CREP patches within our sampling CP categories using polyisect function in the Geospatial Modelling Environment (Beyer 2012) in ArcGIS (ESRI 2011), using digital orthophotos (30 m resolution), data from the 2014 National Agricultural Service Cropland Data Layer (NASS, CDL: Boryan et al. 2014) and a 2012 NRCS Conservation Reserve Program (CRP) Data Layer. Unsampled patches included 79,578 ha among our four CP types: CP22 (18% of enrolled CREP patches) CP23 (28% of enrolled CREP patches), CP3A (3% of enrolled CREP patches), and CP4D (30% of enrolled CREP patches). Land-cover classes included the amount of forest,

grassland, and restored habitat within a 200-m radius of each restored patch, the scale found to be most important for these species in previous analysis (Chapter 2).

Statistical analyses

We used the extended hierarchical distance sampling model of Royle (2004) to estimate abundance while accounting for imperfect detection using the gdistsamp function in the unmarked package in R (Fiske and Chandler 2011). We evaluated models that described how abundance varied as a function of landscape covariates. For the abundance portion of the model, spatial variation of the number of Bell's Vireo, Field Sparrow, Northern Bobwhite, and Willow Flycatcher was treated as a negative binomial random variable. Year was included in all models to account for expected temporal variation in abundance among four separate years. To estimate population size, we included the best-ranked landscape variables for each species from Chapter 2 (Table 2.2). A more detailed description of model development can be found in Chapter 2.

The detection probability component of our model relied on the distance-sampling likelihood for point transect data (Buckland et al. 2001). We evaluated multiple models for the detection function including a constant model with no covariates and year. We fit models evaluating combinations of uniform, exponential, half-normal, and hazard-rate base functions. Candidate models for detection function were evaluated using Akaike's Information Criteria (AIC_c) values to determine which model best fit the data (Burnham and Anderson 2002). We binned observations into five 20-m distance classes and right truncated observations at 100 m for all species because that was approximately where detection probability declined to <10%.

We used parametric bootstrapping to evaluate goodness-of-fit of the best model for each species. We simulated 100 data sets from our model and each time refit the model to these "perfect" data and computed a fit statistic. We then compared the value of the fit statistic for the observed data sets. We used Freeman-Tukey fit statistic to quantify the fit of

the model data set. We interpreted values above the 5th percentile of the reference distribution as indicative of model fit.

We used the best-ranked model (based on AIC_c) for each focal species to predict density of each species in all enrolled Illinois CREP patches within our sample CPs using the species-appropriate landscape metric for each patch to generate population estimates for each species. We accounted for uncertainty by randomly selecting a density value from a normal distribution based on the predicted value (mean) and standard error, then multiplying that density value by patch size to generate an expected abundance of male birds. Because the resulting values for expected abundance are unrealistic in providing fractional numbers of individuals, we used this expected value to generate an integer value for each patch using a Poisson distribution. We iterated this process 1000 times and used the mean as our population estimate with the 2.5% and 97.5% percentiles representing an approximate 95% confidence interval around the estimate. We then compared these estimates to the established aspirational population goals for these species (Table 3.2). Importantly, we assumed that our estimates were in addition to the birds used to generate the state (IDNR 2005) estimates given that the data used to generate those estimates (1990 – 1999) were collected almost entirely prior to the start of the CREP in Illinois (1998; i.e. there was no overlap between our estimate the statewide estimate). We also assumed, based on trend analysis data obtained from the BBS (Sauer et al. 2014), that baseline populations of our focal species (i.e., outside of CREP) were not increasing or decreasing compared to the time period used to generate the statewide population estimates.

For species where estimates did not meet or exceed population goals, we used a simulation approach to estimate the amount of additional restored habitat that would be needed to achieve those population goals. We simulated 1,000 landscapes starting with the current amount of restored habitat within our four sampled CPs (32,000 ha), and incrementally added 1,000 ha until the average population size was approximately equal to the population goal. For each iteration of the simulation, we randomly drew patches (with replacement) from the existing population of CREP patches in the Illinois River watershed

within the four sampled CPs we examined. For each iteration, we used the best-fitting landscape model for each species to estimate population size. We calculated population size using the methods detailed above. To estimate the approximate annual cost that would be needed to achieve population goals, we multiplied the total hectares needed to achieve the population goal by the average yearly federal CREP rental rate of $212.30/ha (IDNR 2016). While this doesn't fully account for costs paid to landowners that have enrolled in permanent easements in Illinois because the state makes additional payments to landowners to extend contracts beyond typical 10 to15-year enrollment periods, we felt using this rate was more broadly representative of the potential cost of restoration within the Midwestern U.S.

RESULTS

We conducted bird surveys at 172 restored patches from 2012-2015 resulting in 401-602 point-count surveys each year. Sample patches ranged from 2.9 to 174.7 ha (mean= 35.3, SE=2.6). Field Sparrows were the most abundant of the four focal species included in our analysis (64%), followed by Willow Flycatcher (18%), Bell's Vireo (10%), and Northern Bobwhite (8%) for a total of 4,021 observations. Year was included as a covariate on detection probability for all four focal species. Best-fitting key functions for detection were half-normal for Bell's Vireo and Willow Flycatcher, uniform for Northern Bobwhite, and exponential for Field Sparrow. Bell's Vireo and Willow Flycatcher were strongly positively associated with the amount of grass in the surrounding landscape (Tables 3.3, 3.4). Restored habitat within 200 m was a strong negative predictor of Field Sparrow density (Tables 3.3, 3.4). Forest cover negatively influenced Northern Bobwhite and Willow Flycatcher densities (Tables 3.3, 3.4).

Landscape data

Grass cover within a 200-m radius around patches used for point count sampling ranged from 0 – 89% ($\bar{x}$ = 40%, S.E. = 1%), forest cover ranged from 0 – 98% ($\bar{x}$ = 40%, S.E. = 1%), and total restored cover ranged from 3 – 97% ($\bar{x}$ = 41%, S.E. = 1%). For

unsampled patches, grass cover ranged from 0 – 97% ($\bar{x}$ = 33%, S.E. = 1%), forest cover ranged from 0 – 100% ($\bar{x}$ = 35%, S.E. = 1%), and restored cover ranged from 0 – 98% ($\bar{x}$ = 37%, S.E. = 1%).

Current population estimates

Based on our abundance models, total population sizes of birds occupying restored patches within our sample CPs for the Bell's Vireo was 4,418 (2.5th and 97.5th percentiles: 4,395, 4,545), Field Sparrow was 88,579 (2.5th and 97.5th percentiles: 87,781, 89,398), Northern Bobwhite was 19,919 (2.5th and 97.5th percentiles: 19,575, 20,283), and Willow Flycatcher was 15,640 (2.5th and 97.5th percentiles: 15,367, 15,929), which was 110, 33, 6, and 92.0% of statewide population goals, respectively (Table 3.2).

Landscape simulation estimates

Based on our abundance models, simulations indicated that to achieve Field Sparrow, Northern Bobwhite, and Willow Flycatcher aspirational population goals farmland restored habitat would need to be increased by approximately 67,700, 341,000, and 3,000 ha, respectively. Cost estimates, based on average rental rate of $212.30/ha (IDNR 2016), indicated that it would cost an additional $14.4 million, $72.4 million and $640,000 per year in additional expenditures on just land rental (not including habitat establishment or management) to meet the population goals for Field Sparrow, Northern Bobwhite, and Willow Flycatcher, respectively (Table 3.2).

DISCUSSION

Species recovery plans generally assume that with enough habitat, populations of conservation priority species will increase to historic levels (e.g. Rich et al. 2004). We demonstrated, for two of our focal species, the broad-scale restoration provided by Illinois CREP appeared to be sufficient for meeting population goals. Yet, for other species, our simulation approach showed that meeting those goals, while theoretically possible, would

require adding significantly more habitat to the landscape. Importantly, a better understanding of differences among these species may provide insight into our disparate results.

For one conservation priority species, Bell's Vireo, current restoration efforts appear to achieve the statewide aspirational population goal (Table 3.1). Similarly, population estimates for the Willow Flycatcher indicate that current restoration is achieving about 92% of the statewide goal (Table 3.1). The apparent success in meeting goals for these two species likely reflects the modest goals that were set for these species. These goals reflect the historically small estimated population size of the Bell's Vireo and the Willow Flycatcher, which have had only moderate declines within the state. Importantly, because these species prefer early successional habitats (e.g. Kus et al.2007, Sedgwick 2000), patches enrolled in CREP likely have ideal management regimes (i. e. planted hardwood trees) and successional trajectories, and are situated in landscapes that are well suited for them (i.e. shrub or young tree dominated habitat with surrounding grassy land cover). Similar benefits may not be possible, even with modest goals, for species, such as grassland birds, that require habitats that require regular management because natural successional processes would reduce suitability within a few years.

Conversely, our results for the Field Sparrow and Northern Bobwhite demonstrated that restored habitats are supporting populations that are less than one third of the statewide goal. This likely reflects ambitious population goals that result from historical precipitous declines. Importantly, we know that densities of Northern Bobwhite within CREP patches are far below estimates from previous studies (Brennan et al. 2014, Chapter 1). This suggests that in the absence of management actions targeted specifically for this species, habitat creation through private land programs alone may not provide the most effective option for increasing populations of this species. However, densities of Field Sparrows observed in this study (Chapter 1) are well above average compared to what was documented in previous studies (e.g. Carey et al. 2008) suggesting these patches are providing suitable habitat. Yet, even with above average densities for Field Sparrows, patches restored through this program are not providing enough individuals to meet population goals. Thus, it appears that for conservation

priority species tolerant of a broad gradient of early successional habitat with limited distributions and moderate population goals, current private land restoration efforts can be important for achieving national, regional, and state population goals. Conversely, for species that require frequent habitat disturbance and those with precipitous declines and consequently large population goals, achieving these conservation goals may require substantial increases in the amount of restored habitat.

Given that the majority of conservation priority species are found on private lands (e.g. Thogmartin and Rohweder 2008), these lands must play an important role in reversing population declines and achieving aspirational population goals. Importantly, this study has shown that if private land programs are to be used as a mechanism for achieving population goals for the Willow Flycatcher, Field Sparrow, and Northern Bobwhite, then the amount of restored habitat would need to increase at a cost up to 1 to 3 times the current $26.3 million budget for this program (IDNR 2015). Our results suggest that meeting aspirational goals for all focal species would require the restoration of approximately 1% of approximately 27 million hectares of Illinois farmland. However, while possible, given the overall 30% decline in private land program habitat in the U.S. between 2007-2013 (*B.Reiley unpublished data*), the potential for negative societal views on decreasing crop production, and significant monetary losses for farmers, it seems unlikely this could become a reality for private land programs. Importantly, the recent losses of habitats restored through private land programs is alarming given the importance of these habitats to conservation priority species.

Alternatively, if increasing the amount of habitat restored through private land programs to achieve population goals is too economically or politically optimistic, what about the possibility of managing existing conservation lands to achieve them? For example, managing existing public lands to attract greater densities of conservation priority species? Unfortunately, based on our estimates of expected land area required to achieve goals for our focal species, the limited amount of state owned public land in Illinois (approximately 171,000 ha) and other Midwestern states likely makes achieving goals using only public land impossible. Additionally, could existing farmland habitat restoration be improved to attract

greater densities of conservation priority species? One option would be to implement targeted habitat management schemes aimed at improving habitat quality for focal species on existing restored patches. In our study, many restored patches contained low densities of our focal species. Although the reasons for low density are probably varied, management actions taken during initial establishment of vegetation at restored patches (e.g., cool-season grass plantings) or lack of habitat management after establishment could be leading to unattractive vegetation structure for these species. For example, similar to the findings of Dimmick et al. (2002) and Riffell et al. (2008), we observed that many CREP patches were dominated by monocultures of invasive grasses mixed with sparse woody vegetation which tended to be dominated by generalist rather than conservation-priority bird species. (B. Reiley unpublished data). If establishment and subsequent management were targeted toward focal species, then private land programs would likely have greater population impacts for conservation priority species. Indeed, management provisions established within the CRP (Conservation Reserve Program) to improve habitat structure for Northern Bobwhite in Illinois doubled the density of this species in study patches (Osborn et al. 2012). Assuming this best-case scenario of doubling species densities, it may be possible to meet population goals for most species, including those with ambitious goals, through management alone. However, doubling densities of most species is unlikely, and it may be that a combination of management of existing lands and increased restoration is needed. Additionally, although less costly than restoration, even costs for managing lands to improve quality may be prohibitive. Beyond the cost, management provisions are not standard practice for many private-lands programs, and broad-scale implementation of targeted management would require adding these provisions to existing and future contracts.

Importantly, our population estimates represent a snapshot in time and population benefits for many species are likely to be ephemeral. In part, these gains would be limited by the length of the private land contracts which are typically 10-15 years after which many patches go back into agricultural production. Additionally, population gains are likely to change with succession; to maintain populations in the face of successional changes,

management is necessary. For example, there are groups of species dependent on herbaceous areas with little woody vegetation, others dependent on a combination of herbaceous and woody vegetation, and some that are found in greater abundance as a woody canopy begins to develop (e.g. Schlossberg and King 2009). However, in many cases our focal species have a wider successional threshold than previously found. For example, Field Sparrows have been found to be associated with grassy habitats with sparse shrub cover (e.g. Peterjohn and Rice 1991), and to decrease in density after 10 years (Carey et al. 2008), yet this species increased at our sample patches as patch age increased even up to 15 years since habitat restoration (*B. Reiley unpublished data*). Similarly, our data on both Bell's Vireo and Northern Bobwhite suggest these species can occupy much later seral stages than previously thought, and benefits of these habitats in the absence of disturbance may be longer than expected based on previous research. Nevertheless, without disturbance, as these areas age they will eventually more closely resemble forests and benefits to early successional species will decline.

Establishing population goals (Rich et al. 2004; Potter et al. 2007) was an important step for landbird conservation within North America. While an important first step, this approach has limitations. For example, the estimation methods do not take into account species-habitat relationships or breeding habitat availability (Mahon et al. 2014). As a result, these goals fail to account for what may be possible given current land use, making the goal of doubling many species populations impossible. Methods that use species habitat relationships and account for habitat availability (e.g. Mahon et al. 2014; Thogmartin et al. 2014) have provided insights into what level of conservation is possible given current and future conditions. Recent advances in remote sensing such as LiDAR may offer greater promise for helping to better quantify habitat metrics that would make goals more realistic.

Based on our analysis, private land programs can be important for achieving broad-scale population goals for early successional species. More attention on large-scale benefits of such programs is needed to further elucidate what these habitats are contributing toward achieving national population goals. Our results demonstrate that while population goals can be met for some species with current amounts of private land program habitat, targeting these

programs toward at-risk species could produce more pronounced benefits. In particular, species that occupy shrublands appear to benefit most from these programs in our study area and focusing on these species would likely provide the most benefit for the least amount of expenditure. While the current study focused primarily on shrubland birds, private-lands programs appear to be crucial for grassland birds (e.g. Herkert 2007) and future studies should examine current and potential contributions of large scale private land programs to grassland bird population goals.

LITERATURE CITED (Formatted for Journal of Applied Ecology)

Best, L.B., Campa III, H., Kemp, K.E., Robel, R.J., Ryan, M.R., Savidge, J.A., Weeks Jr., H.P., & Winterstein, S.R. (1997). Bird abundance and nesting in CRP fields and cropland in the Midwest: a regional approach. Wildlife Society Bulletin 25, 864–877.

Beyer, H. (2012) Geospatial Modelling Environment (Version 0.7.2.1). URL: http://www.spatialecology.com/gme. Accessed 25 September, 2014.

Boryan, C., Z. Yang, L.Di, & Hunter, K. (2014) A new automatic stratification method for U.S. agricultural area sampling frame construction based on the Cropland Data Layer, IEEE Journal of Selected Topics in Applied Earth Observations and Remote Sensing 7, 1939-1404.

Brennan, L.A., Hernandez, F., & Williford, D. (2014) Northern Bobwhite (*Colinus virginianus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/397doi:10.2173/bna.397

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., & Thomas, L. (2001) Introduction to distance sampling: estimating abundance of biological populations. Oxford University Press, Oxford, U.K.

Burnham, K.P., & Anderson, D.R. (2002) Model selection and multimodel inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.

Carey, M., Burhans, D.E., & Nelson, D.A. (2008) Field Sparrow (*Spizella pusilla*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/103doi:10.2173/bna.103

Carter, M.F., Hunter, W.C., Pashley, D.N., and Rosenberg, K.V. (2000) Setting conservation priorities for landbirds in the United States: the Partners in Flight approach. Auk, 117, 541–548.

Clay, J. (2004) World Agriculture and the Environment: A Commodity-by-Commodity Guide to Impacts and Practices. Island Press, Washington, DC, USA.

Davey, C.M., Vickery, J.A., Boatman, N.D., Chamberlain, D.E., Parry, H.R., & Siriwardena, G.M. (2010) Assessing the impact of entry-level stewardship on lowland farmland birds in England. Ibis, 152, 459–474.

Dimmick, R.W., M.J. Gudlin, & D.F. McKenzie (2002) The Northern Bobwhite conservation initiative. Miscellaneous publication of the Southeastern Association of Fish and Wildlife Agencies, Columbia, South Carolina, USA.

ESRI (2012) ArcGIS version 10.1. Earth Systems Research Institute, Redlands, California, USA.

Evans, K.O., Burger, L.W., Riffell, S., Smith, M.D. (2014) Assessing multiregion avian benefits from strategically targeted agricultural buffers. Conservation Biology, 28, 892–901.

Faaborg, J, Holmes, R.T., Anders, A.D., Bildstein, K.L., Dugger, K.M., Gauthreaux, Jr., S.A., Heglund, P., Hobson, K.A., Jahn, A.E., Johnson, D.H., Latta, S.C., Levey, D.J., Marra, P.P., Merkord, C.L., Nol, E., Rothstein, S.I., Sherry, T.W., Sillett, T.S., Thompson III, F.R., & Warnock, N. (2010) Conserving migratory land birds in the New World: Do we know enough? Ecological Applications, 20, 398–118.

Fiske, I., & Chandler, R. (2010) unmarked: Models for Data from Unmarked Animals. R package version 12–2. CRAN website. Available: http://cran.r-project.org/web/packages/unmarked. Accessed 2016 Aug 7.

Fletcher Jr., R.J., & Koford, R.R. (2003) Changes in breeding bird populations with habitat restoration in Northern Iowa. American Midland Naturalist, 150, 83–94.

Hohman, W.L., & Halloum, D.J. (eds.). 2000. A comprehensive review of Farm Bill contributions to wildlife conservation, 1985– 2000. U.S. Department of Agriculture, Natural Resources Conservation Service, Wildlife Habitat Management Institute, Washington, DC, USA. USDA/NRCS/WHMI-2000.

Herkert, J.R. (2007) Conservation Reserve Program benefits on Henslow's Sparrows within the United States. Journal of Wildlife Management, 71, 2749–2751.

Gray, R.L., & Teels, B.M. (2006) Wildlife and fish conservation through the Farm Bill. Wildlife Society Bulletin, 34, 906–913.

Kus, B, Hopp, S.L., Johnson, R.R., & Brown, B. (2010) Vireo (*Vireo bellii*), The Birds of North America online (A. Poole, ed.) Ithaca: Cornell lab of Ornithology; Retrieved from Birds of North America online:http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/035 doi.org/10.2173/bna.35

Illinois Department of Natural Resources (2005) The Illinois comprehensive wildlife conservation plan & strategy. Version 1.0. Illinois Department of Natural Resources, Springfield, IL.

Illinois Department of Natural Resources (2015) The Illinois Conservation Reserve Enhancement Program Annual Report. Springfield, IL.

Mahon, C.L. Bayne, E.M., Solymos, P., Matsuoka, S.M., Carlson, M., Dzus, E., Schmiegelow, F.K.A. & Song, S. J. (2014) Does expected future landscape condition support proposed population objectives for boreal birds? Forest Ecology and Management, 312, 28–39.

Millspaugh, J.J., & Thompson, F.R., III (2009) Models for Planning Wildlife Conservation in Large Landscapes, Academic Press.

Morrison, M.L., Marcot, B.G., & Mannan, R.W. (1992) Wildlife Habitat Relationships: Concepts and Applications. Madison, WI, USA: University Wisconsin Press.

Osborne, D.C., Sparling, D.W., & Hopkins, R.L. (2012) Influence of conservation reserve program mid-contract management and landscape composition on Northern Bobwhite in tall fescue monocultures. Journal of Wildlife Management, 76, 566–574.

Perkins, A.J., Maggs, H.E., Watson, A., Wilson J.D. (2011) Adaptive management and targeting of agri-environment schemes does benefit biodiversity: a case study of the Corn Bunting *Emberzia calandra*. Journal of Applied Ecology, 48, 514–522.

Peterjohn, B.G., & Rice, D.L. (1991) The Ohio Breeding Bird Atlas. Ohio Dep. Natural Resources, Columbus, Ohio, USA.

Pimm, S.L., & Askins, R.A. (1995) Forest losses predict bird extinctions in Eastern North America. Proceedings of the National Academy of Sciences, 92, 9343–9347.

Potter, B.A. Soulliere, G.J., Ewert, D.N., Knutson, M.G., Thogmartin, W.E., Castrale, J.S., and Roell, M.J. (2007) Upper Mississippi River and Great Lakes Region Joint Venture landbird habitat conservation strategy. U.S. Fish and Wildlife Service, Fort Snelling, MN.

Pullin, A.S. (2002) Conservation Biology. Cambridge University Press, Cambridge, UK.

Ralph, C.J., Sauer, J.R., & Droege, S. (1993) Managing and monitoring bird populations using point counts: standards and applications. USDA Forest Service General Technical Report PSW-GTR-149. Pacific Southwest Research Station, USDA Forest Service, Albany, California.

Ralph, C.J., Sauer, J.R., & Droege, S. (1995) Monitoring bird populations by point count. USDA Forest Service General Technical Report PSW-149.

Reidsma, P. Tekelenburg, T., van den Berg, M., Alkemade, R., (2006) Impacts of land use change on biodiversity: an assessment of agricultural biodiversity in the European Union. Agriculture, Ecosystems & Environment, 114, 86–102.

Ribic, C.A., Guzy, M.J., & Sample, D.W. (2009) Grassland bird use of remnant prairie and conservation reserve program fields in an agricultural landscape in Wisconsin. American Midland Naturalist, 161, 110–122.

Rich, T.D. Beardmore, C.J., Berlanga, H., Blancher, P.J., Bradstreet, M.S. W., Butcher, G.S., Demarest, D.W., Dunn, E.H., Hunter, W.C., Iñigo-Elias, E.E., Kennedy, J.A., Martell, A.M., Panjabi, A.O., Pashley, D.N., Rosenberg, K.V., Rustay, C.M., Wendt, J.S., & Will, T.C. (2004) Partners in Flight North American Landbird Conservation Plan. Cornell Lab of Ornithology, Ithaca, New York, NY, USA.

Riffell, S.K., Monroe, A.P., Martin, Evans, K.O., Burger, L.W., Smith, M.D., & Minderman, J. 2015. Response of non-grassland avian guilds to adjacent herbaceous field buffers: testing the configuration of targeted conservation practices in agricultural landscapes. Journal of Applied Ecology, 52, 300–309.

Rosenberg, K.V., & Blancher, P.J. (2005) Setting numerical population objectives for priority landbird species. Pages 57–67 in C. J. Ralph and T. D. Rich, editors. Bird conservation implementation and integration in the Americas: Proceedings of the Third International Partners in Flight Conference. Volume 1. USDA Forest Service General Technical Report PSW-GTR-191, Pacific Southwest Research Station, Albany, California, USA.

Royle, J.A., Dawson, D.K., & Bates, S. (2004) Modeling abundance effects in distance sampling. Ecology, 85, 1591–1597.

Rushing, C.S., Ryder, T.B., & Marra, P.P. (2016) Quantifying drivers of population dynamics for a migratory birds throughout the annual cycle. Proceedings of the Royal Society B, 283.

Sauer, J.R., Niven, D.K., Hines, J.E., Ziolkowski Jr., Pardiek, K.L., Fallon, J.E., & Link, W. A. (2014) The North American Breeding Bird Survey, Results and Analysis 1966 - 2013. Version 01.30.2015 USGS Patuxent Wildlife Research Center, Laurel, MD

Sedgwick, J.A. (2000) Willow Flycatcher (*Empidonax traillii*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/533doi:10.2173/bna.533

Schwegman, J.E. (1973) Comprehensive plan for the Illinois Nature Preserve System. Part 2. The natural divisions of Illinois. Illinois Nature Preserves Commission, Rockford, IL. 32 pp.

Thogmartin, W.E., & Rohweder, J.J. (2008) Conservation opportunity assessment for rare birds in the Midwestern United States: a private lands imperative. Proceedings of the fourth international partners in flight conference: tundra to tropics T. D. Rich, C.

Arizmendi, D. Demerest, & C. Thompson, Editors. McAllen, Texas, USA. Pp.419–425.

Thogmartin, W.E., Crimmins, S.M., & Pearce, J. (2014) Prioritizing bird conservation actions in the prairie hardwood transition of the Midwestern United States. Biological Conservation, 176, 212–223.

United States Department of Agriculture, Farm Service Agency (2017) "Conservation Enrollments Monthly Summary – JUNE 2017." Available: https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/Conservation/PDF/JUN%202017%20Summary.pdf

Van Buskirk, J., & Willi, Y. (2004) Enhancement of farmland biodiversity within set-aside land. Conservation Biology, 18, 987–994.

Wells, J.V. (2010) From the last of the large to the remnants of the rare: bird conservation at an ecoregional scale. (Eds. Trombulak, S.C. Baldwin, R.F.), Landscape-scale Conservation Planning. Springer, London, UK; New York, NY. USA, pp. 121 – 137.

Will, T.C., Ruth, J.M., Rosenberg, K.V., Krueper, D., Hahn, D., Fitzgerald, J., Dettmers, R., & Beardmore, C.J. (2005) The five elements process: designing optimal landscapes to meet bird conservation objectives. Partners in Flight Technical Series 1. Partners in Flight:http://www.partnersinflight.org/pubs/ts/01-FiveElements.pdf>. Accessed 1 Mar 2017.

TABLES AND FIGURES

Table 3.1 State population goals and desired population increase for each focal species.

Species	Estimate[1]	Desired % Increase	Desired Population Increase
Bell's Vireo	4,000	100	+ 4,000
Field Sparrow	270,000	100	+ 270,000
Northern Bobwhite	320,000	100	+ 320,000
Willow Flycatcher	31,000	50	+ 17,000

[1]Estimated population size using Breeding Bird Survey data from 1990-1999 based on Rich et al. 2004 and IDNR Wildlife Action Plan (2005).

Table 3.2 Estimated population contribution of restored habitats toward meeting population goals for each focal species.

Species	Current Estimate[1]	Simulated Estimate[2]	Habitat Needed[3]
Bell's Vireo	4,418[4]	-	-
Field Sparrow	88,579[5]	271,831[8]	67,700
Northern Bobwhite	19,919[6]	320,183[9]	341,000
Willow Flycatcher	15,640[7]	17,167[10]	3,000

[1] All fields within conservation practices within original sample criteria

[2]Average estimated population size based on the amount of habitat needed to achieve species goal.

[3]Amount of habitat needed to achieve species goal.

[4]2.5thPercentile = 4,395, 97.5th Percentile = 4,545

[5]2.5th Percentile =87,781, 97.5th Percentile = 89,398

[6]2.5th Percentile =19,575, 97.5th Percentile = 20,283

[7]2.5th Percentile =15,346, 97.5th Percentile = 15,902

[8] 2.5th Percentile =270,627, 97.5th Percentile = 273,022

[9]2.5th Percentile =318,110, 97.5th Percentile = 322,370

[10]2.5th Percentile =16,837, 97.5th Percentile = 17,615

Table 3.3 Best-fitting hierarchical models of focal species density (λ) and detection probability (*p*), including estimates, used to calculate population size of focal species of concern in restored habitat.

Species	Model	Detection Probability (within 100 m)	Average density±SE[a]
Bell's Vireo	λ (Year+Grass) *p*(Year)	74.1 ± 3.75	0.103 ± 0.010
Field Sparrow	λ (Year+Restored habitat) *p*(Year)	50.6 ± 2.07	2.836 ± 0.393
Northern Bobwhite	λ (Year+Forest) *p*(Year)	1	0.734 ± 0.161
Willow Flycatcher	λ (Year+Forest+Grass) *p*(Year)	51.6 ± 2.10	0.392 ± 0.065

[a]Birds/ha

Table 3.4 Parameter estimates and standard errors from the models used to calculate population size of focal species of concern in restored habitat. Variables include: grassland cover within 200 m (Grass), forest cover within 200 m (Forest), and restored habitat cover (Restored habitat).

Model	BEVI[1]		FISP[2]		NOBO[3]		WIFL[4]	
	β	SE	β	SE	β	SE	β	SE
Intercept	-2.19	0.10	1.05	0.14	-0.31	0.22	-0.81	0.18
Year	0.01	0.10	0.03	0.06	-0.12	0.09	-0.11	0.08
Forest					-0.62	0.10	-0.15	0.09
Grass	0.35	0.09					0.23	0.09
Restored habitat			-0.10	0.04				

[1]Bell's Vireo (*Vireo bellii bellii*)

[2] Field Sparrow (*Spizella pusilla*)

[3] Northern Bobwhite (*Colinus virginianus*)

[4] Willow Flycatcher (*Empidonax trailii traillii*)

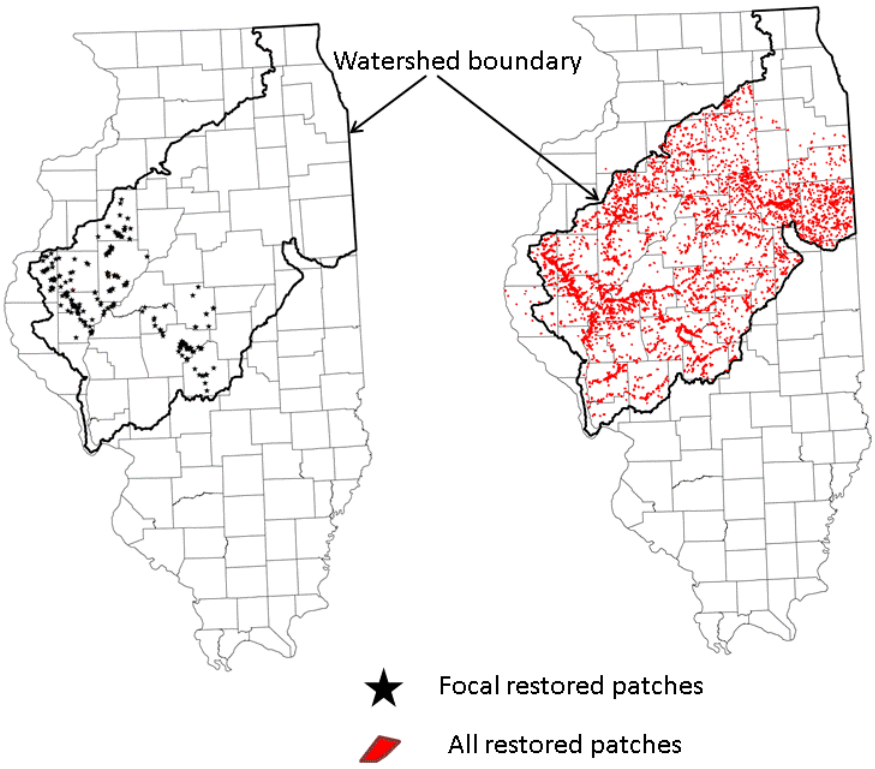

Figure 3.1 Map of Illinois River watershed including focal restored patches and all patches restored through the Conservation Reserve Enhancement Program in Illinois watershed.

CHAPTER 4: AVIAN FITNESS CONSEQUENCES MATCH HABITAT SELECTION AT THE NEST SITE BUT NOT THE LANDSCAPE SCALE IN AGRICULTURALLY FRAGMENTED LANDSCAPES

ABSTRACT: When choosing between habitats of heterogeneous quality, birds should choose the best available habitat. Yet, studies show that individuals fail to choose habitats that maximize their fitness especially in drastically altered landscapes. Most studies have focused on selection at single scales, often using a single measure of fitness. However, links between habitat selection and fitness may vary depending on the spatial scale and measure of fitness, especially in situations where agricultural land use has altered the surrounding landscape. Here, we examined multi-scale habitat selection and fitness measures of the Bell's Vireo and Willow Flycatcher using data collected in restored agricultural fields where we expected extensive alteration of the surrounding landscape would lead to mismatches between selection and fitness outcomes. We found evidence for selection of nest sites with dense understory, larger patches, and increasing grassland cover in the surrounding landscape. Fitness outcomes mostly aligned with species habitat selection particularly at the nest and landscape scales, though the relationship was strongest at the nest scale. Yet, we observed mismatches where nest survival was lower in preferred landscapes with more surrounding grassland cover and fledgling production was lower at nest sites with dense understory cover. We speculate that observed mismatches may be influenced by anthropogenic habitat fragmentation and may represent reproductive tradeoffs. Our results demonstrate that individuals of both species can accurately select habitat that increases fitness in drastically altered landscapes but the connection between habitat selection decisions and reproductive consequences varies depending on the scale and measure of fitness.

INTRODUCTION

Understanding why animals choose particular habitats is important for understanding how they assess habitat quality. This is especially true for migratory species whose mobility provides access to an array of potential habitat patches over vast spatial scales. For migratory individuals, choosing where to breed has important reproductive implications (e.g. Holmes et al. 1996, Misenhelter and Rotenberry 2000, Kolbe and Janzen 2002) as patches differ in quality and habitat quality within patches is rarely homogeneous. Natural selection should favor the ability to assess habitat quality (Jaenike and Holt 1991, Martin 1998) and animals should choose the best available habitat (Fretwell and Lucas 1970). Consequently, we would expect clear links between habitat selection and resulting fitness consequences.

Despite the expected links between habitat selection and fitness outcomes, individuals often fail to choose habitats that maximize their fitness potential (e.g. Woodward et al. 2001, Chalfoun and Martin 2007, Germain and Arcese 2014). While studies have found positive relationships between habitat preference and fitness outcomes (e.g. Martin 1998, Perot and Villard 2009, McKellar et al. 2015) numerous studies have documented non-ideal or maladaptive habitat preference (e.g., Arlt and Pärt 2007, Magi et al. 2009). Explanations for non-ideal habitat preference have been varied and include lack of available cues at the time of selection (Orians and Wittenberger 1991), selection cues not linked to habitat quality (Schlaepfer et al. 2002), conflicting choices between cues and mate choice (Kokko and Sutherland 2001), and site fidelity (Pulliam and Danielson 1991). Maladaptive habitat preference has frequently been explained to result from anthropogenic disturbance whereby traditional indicators of habitat quality become unreliable (e.g., Bock and Jones 2004, Weldon and Haddad 1999, Misenhelter and Rotenberry 2000).

To accurately estimate reproductive consequences of habitat selection decisions, studies must focus on biologically relevant spatial scales (Chalfoun and Martin 2007). Given that predation is the primary cause of failure for songbird nests (Ricklefs 1969, Martin 1993, Cox et al. 2014), evaluations of adaptive habitat selection for these species have generally focused on the influence of habitat structure immediately surrounding nests (Martin 1998,

Misenhelter and Rotenberry 2000). At this scale habitat features such as understory density can affect fitness outcomes by affecting the probability of nest discovery by predators (Martin 1998) and, as a result, birds may choose breeding locations with thicker understory vegetation to minimize the risk of predation (Martin 1993). At the nest site, microclimate and food availability may also influence the number of fledglings produced per successful nest (e.g. Lloyd and Martin 2004). Beyond the nest site, broad-scale features may influence the distribution and abundance of predators and brood parasites. For example, nests in smaller patches may be more vulnerable to generalist nest predators (e.g., Rush and Stutchberry 2008), and may face a greater threat of parasitism by Brown-headed Cowbirds (*Molothrus ater*; Hoover et al. 1995, Rush and Stutchberry 2008, Benson et al. 2013). Moreover, nest predation is spatially heterogeneous and dependent on landscape context (Tewksbury et al. 2006). For example, nests located in highly fragmented and agriculturally dominated landscapes may have greater predation and brood parasitism rates than nests in more intact landscapes (e.g. Donovan et. al. 1995, Chalfoun et al. 2002). Additionally, the number of young fledged from successful nests can be affected by food abundance and habitat features that affect foraging efficiency (e.g. Pärt 2001), both of which may be influenced by attributes of the breeding patch.

Multi-scale studies that document both habitat choice and resulting components of reproductive performance are rare (Martin 1998, Clark and Shutler 1999, Lloyd and Martin 2005), and this type of research is particularly needed in habitats embedded in agriculturally fragmented landscapes where there is a high potential for mismatches between preference and reproductive consequences (e.g. Weldon and Haddad 2005, Gilroy et al. 2011). Restored habitats in agricultural landscapes, while widely thought to be beneficial for birds (e.g., Herkert et al. 2007), have also been suggested to have fecundity rates too low to sustain populations of some species (McCoy et al. 1999). Thus, understanding habitat choice and reproductive outcomes in these restored habitats could both provide new insights into the adaptive nature of habitat selection in birds and provide managers with valuable information to be applied to future conservation efforts for priority species.

Here we studied the breeding habitat selection and two reproductive metrics including nest success and number of young fledged per successful nest (hereafter fledgling production) of two species of conservation concern. We did this across three ecologically relevant scales 1) nest site (5-m buffer around nest), 2) habitat patch, and 3) landscape (200-m buffer) using data collected in restored habitat on former farmland where we expected a high potential for mismatches between habitat preferences and fitness consequences. Specifically, based on previous work (e.g. Kus et al 2007, Sedgwick 2000, Chapter 2) we expected habitat selection at the nest-site scale would be related to habitat features associated with increased cover at the nest, species would be associated with increased patch size, and that species would be positively associated with surrounding grassland cover. Relative to fitness consequences of habitat selection, given that previous studies (e.g. Chalfoun and Schmidt 2012) found the highest probability of finding evidence for adaptive habitat selection was at the nest scale and long standing associations between species and vegetation around the nest site, we expected that nest-site habitat selection for both species would result in greater nest survival and fledgling production. Conversely, because the alteration of landscapes and creation of patchy landscapes due to agricultural fragmentation is relatively novel, as measured on evolutionary time scales, we expected that habitat selection for patch and landscape features should lead to either neutral or maladaptive selection.

METHODS

Study system

Our focal species were the Bell's Vireo (*Vireo bellii bellii*) and Willow Flycatcher (*Empidonax traillii traillii*), both species of conservation concern due to recent declines. The Bell's Vireo (hereafter vireo) is a single-brooded, small, insectivorous Neotropical migrant that breeds in dense understory vegetation associated with shrubland habitats (Kus et al. 2010). Widespread declines of this species are thought to be the result of habitat loss and degradation, high rates of nest predation, and nest parasitism by Brown-headed Cowbirds (Budnik et al. 2000).

The Willow Flycatcher (hereafter flycatcher) is a single-brooded, insectivorous Neotropical migratory species that breeds in shrubby habitats associated with riparian areas from Maine to North Dakota (Sedgwick 2000). In Illinois, this species is associated with willow thickets, upland shrub areas, and roadside shrub edges (Graber et al. 1974). The causes of declines for this species are unknown but thought to be the result of habitat loss and brood parasitism by Brown-headed Cowbirds (Sedgwick 2000).

Data collection took place within the Illinois River watershed in central Illinois, USA (40° 15' N, - 90° 28') in 6 counties (Christian, Fulton, Logan, McDonough Sangamon, and Schuyler). Sample patches (n=172; Chapter 2) were randomly drawn from a group of former agricultural fields restored through a farmland restoration program. Sample patches ranged from 2.9 to 174.7 ha ($\bar{x}$ =35.3, SE=2.6). Patches were dominated by early successional vegetative communities (restoration efforts began in 1999), with structure ranging from herbaceous areas dominated by grasses and forbs to dense scrub-shrub habitat dominated by shrubs and young hardwood trees. Patches were frequently adjacent to riparian forests and row-crop agriculture.

Habitat selection

To determine vireo and flycatcher patch and landscape habitat selection, patches were sampled using standard point-count methods during the breeding seasons (mid-May to mid-July) of 2012–2015. Surveys consisted of unlimited-radius point counts, 10 minutes in duration (Ralph et al. 1995). Prior to data collection, observers calibrated distances using a laser range finder based on surrounding features. Surveys were conducted between sunrise and 1100 CDT and were not conducted in inclement weather (rain, wind >13mph) (Ralph et al. 1993). We used ArcGIS (ESRI 2012) to randomly choose point count locations. Point count locations were >70 m away from patch edges. We conducted point count surveys between 1-4 times in 2012 and 3 times in 2013-2015; visits were separated by >14 days. Surveys were conducted by 3 observers in 2012 and 2013 and 4 observers in 2014 and 2015.

We rotated observers among patches in all years to minimize potential observer bias and trained observers in species identification and distance estimation prior to data collection.

To examine habitat use at the nest-scale and study the reproductive consequences of habitat selection, we searched for nests at a subset of our sample patches. Nest-sampling patches were chosen from among the point-count patches based on the presence of our focal species (from point-count surveys) and through the use of additional survey transects to identify territories of focal species. Survey transects consisted of both passive listening and active playback using male vireo and flycatcher vocalizations; these systematic surveys were conducted along a grid of transects at 100-m intervals within each nest sampling patch. The number of sample points per patch varied depending on patch area and shape. Once per season, observers listened for focal species passively for 2 minutes and then used active playback for 2 minutes with a two speaker CD player set to a volume such that observers could hear broadcasts up to distances of 70 m in moderately dense vegetation with clear atmospheric conditions. All occupied locations and the number of individuals heard singing were documented with GPS coordinates and were used to guide subsequent nest searches.

Nest searches were conducted from May–August 2013–15 at 14 focal patches, 7 had only vireos, 2 had only flycatchers, and 5 had both species. Focal patches ranged from 5.4 to 81.4 ha ($\bar{x}$=28.9 ha), and were separated by >4 km. All vireo and flycatcher territories identified within each patch were searched for nests systematically and by using behavioral cues. Nest searches were conducted at all occupied territories every 1-3 days until an active nest was found. We recorded GPS coordinates for each nest and monitored nests at 1-4 day intervals to determine status (i.e., active or inactive based on presence of viable eggs or young). We attempted to minimize disturbance to the surrounding vegetation by using different paths to approach and leave the nest site on each visit (Martin and Geupel 1993). Territories where a nest was deemed inactive or failed were subsequently searched throughout the breeding season to monitor additional nesting attempts for each pair. While we did not band individuals, we felt confident our renest searches were focused on the same male's territories due to the relatively wide spacing and small size of our focal species territories

(Kus et al. 2010, Sedgwick 2000) as well as finding few undocumented (<3% of total nests per year) nests during these additional systematic nest searches.

Nest scale variables

After the termination of nesting each year, we recorded data on vegetation structure at both nest sites (starting around July 15) and a paired random location within a 160-m radius of each nest site using a modified BBIRD protocol (Martin et al. 1997). We chose this radius based on average territory size of both species (Kus et al. 2010, Sedgwick et al. 2000). Habitat variables were chosen based on previous studies or presumed relevance to the nesting ecology of early successional birds and search efficiency of predators. Specifically, within a 5-m radius we estimated the percent cover of grass, forbs, shrubs, and trees within quadrants oriented based on the four cardinal directions. To assess understory vegetation density we used a Robel pole (Robel et al. 1970) marked at 0.1 m increments and recorded the percent cover of all vegetation in each of four 0.5 m segments from 5 m away in the four cardinal directions at a height of 1 m above the ground. For each vegetation variable, we averaged values from the four cardinal directions into one value.

Patch and landscape scale variables

At both the patch and landscape scales, we used density as a measure of habitat selection of our focal species. Variables were chosen based on previous studies, presumed relevance to the nesting ecology of early successional birds and the predators that affect them. Specifically, because nests may have higher nest predation and parasitism rates in smaller patches with higher edge to area ratio (Lahti 2001, Benson et al. 2013, respectively) we quantified patch size by measuring all continuous habitat, considering roads with two or more lanes with disturbed roadsides or habitat transitions (i.e. grassy field to forest interface, etc.) as edges that delineated a patch border. When patches of similar habitat were linked with corridors ≥20 m wide we considered them to be part of the same patch. All patch delineations were ground truthed for accuracy.

Landscape composition around each nest and sample patch was quantified using data from the 2014 National Agricultural Statistics Service Cropland Data Layer (NASS, CDL: Boryan et al. 2014) in ArcGIS 10.1 (ESRI 2012). We used the Geospatial Modeling Environment (Beyer 2012) to quantify proportions of each cover type within a 200-m buffer around each nest site which was previously demonstrated to be an important scale for early successional birds (Pabian et al. 2013, Quinn et al. 2014) and for our focal species (Chapter 2).

Reproductive consequences

We examined the influence of nest-site, patch, and landscape variables on two measures of avian fitness: daily nest survival (Shaffer 2004) and fledgling production. Nest survival helps differentiate complete failure from nests that fledged at least one young, whereas fledgling production can help determine the influence of factors other than predation (e.g., microclimate, food availability).

Statistical analyses

To determine vireo and flycatcher nest-site selection, we generalized linear mixed models with a binomial distribution and logit link function (SAS PROC GLIMMIX; SAS Institute 2010). Due to the close proximity of nests and random locations and because year may have an effect on vegetation variables, we considered models with nest and year as random effects. In this analysis, nest presence was the response variable and understory density, grass cover, forb cover, and shrub cover were predictor variables. We evaluated single-variable models and selected the highest-ranked model using Akaike's Information Criterion adjusted for small sample sizes (AIC_c) (Burnham and Anderson 2002). We assessed support for nest-site habitat variables using model coefficients and their 95% confidence intervals and considered variables that were supported by AIC_c, but for which confidence intervals of coefficients that overlapped zero to be weakly supported.

To determine vireo and flycatcher patch and landscape habitat preference we used the extended hierarchical distance sampling model of Royle et al. (2004) using the gdistsamp function in the unmarked package in R (Fiske and Chandler 2011) which can be used to model local scale covariates for both detection probability and density. Specifically, we used this function to evaluate models that describe how abundance varied as functions of covariates at patch and landscape spatial scales. For the abundance portion of the model, spatial variation of the number of birds was treated as a negative binomial random variable. To account for the sampling of each patch in four separate years we included year as a covariate for abundance in each model. Detection probability in our model relied on the distance-sampling likelihood for point transect data (Buckland et al. 2001). We fit models evaluating combinations of uniform, exponential, half-normal, and hazard-rate base functions for detection, and evaluated year as a covariate. Candidate models for detection function were evaluated using AIC (Burnham and Anderson 2002). We right truncated where detection probability was <10% (e.g. Buckland et al. 2001) and as a result we only included observations within 100 m, binned into 20 m intervals

To evaluate patch and landscape preference, we developed two sets of *a priori* models, one for patch scale and another for landscape scale (Table 4.1). We ranked models according to AIC_c and computed model weights (Burnham and Anderson 2002). In cases where no model was overwhelmingly supported ($w_i > 0.9$), we used model averaging using the natural average method in the R package AICcmodavg (Mazarolle 2012) to examine effects of explanatory variables on bird abundance. In addition to ranking candidate models using AIC_c, we assessed model fit using a likelihood ratio test between the global and intercept-only models, and used the ratio of the chi-squared statistic to degrees of freedom to evaluate evidence of overdispersion ($\hat{c} \leq 1.2$). Support for patch and landscape habitat variables were evaluated using model coefficients and their 95% confidence intervals. We considered variables that were supported by AIC_c, but for which confidence intervals of coefficients that overlapped zero to be weakly supported.

To understand if habitat selection was indicative of habitat quality at the nest-site,

patch, and landscape scale we examined vireo and flycatcher nest survival using the logistic exposure method (Shaffer 2004) (SAS PROC GLIMMIX; SAS Institute 2010). To account for potential non-independence due to repeated sampling of patches within and among years we included year and patch name as random effects but later dropped these random effects as they did not improve models. We evaluated nest survival by generating four sets of models: temporal and biological, nest-site, patch, and landscape. Although not to the main focus of our study, we dealt with potential nuisance variables by including temporal and biological models for nest survival including year, day of year, year × day of year, parasitism status, nest stage × year, and nest stage × day of year + year. Because temporal and biological factors reflect important sources of variation that may affect the influence of habitat or landscape features, we evaluated these models first using AIC_c and included the best-fit variables in subsequent nest survival analyses. We also examined the influence of nest scale vegetation, patch and landscape variables on the number of young fledged from successful nests using generalized linear mixed models (SAS PROC GLIMMIX; SAS Institute 2010) using a Poisson distribution and used a similar modeling approach described above but excluding nest stage from our temporal and biological models. We again included year and patch name as a random effect to account for potential non-independence due to repeated sampling of patches but dropped these terms as they did not improve models. For both nest survival and fledgling production analyses we tested our predictions using a final model for each scale including the best fit nuisance variable along with the best fit nest site, patch, and landscape variables and generated model coefficients using that model. As above, we assessed support for fitness outcomes relative to species habitat selection using model coefficients and their 95% confidence intervals. We considered variables that were supported by AIC_c, but for which confidence intervals of coefficients that overlapped zero to be weakly supported.

RESULTS

From 2013 to 2015, we found 572 vireo nests (505 with eggs or young) and 204 flycatcher nests (188 with eggs or young). Of these, 174 (34%) vireo and 106 (56%) flycatcher nests

fledged young ($\bar{x}$ = 2.89, S.E. = 0.08, $\bar{x}$ = 2.86, S.E. =0.09, young per successful nest, respectively). Among nests that fledged young, 3.4% of vireo nests fledged 1.2 Brown-headed Cowbird chicks per nest and 1.8% of flycatcher nests fledged 1.0 Brown-headed Cowbird chick per nest. We collected habitat data at paired nest-site and random locations for 572 vireo nests and 132 flycatcher nests. Out of the 505 active vireo nests, we had complete nest-site habitat, patch, and landscape composition data for 466 nests for 1,808 exposure days and included those in subsequent nest survival analyses. We had complete nest-site habitat, patch, and landscape composition data for all active flycatcher nests for 746 exposure days and included those in subsequent survival analysis. Brown-headed Cowbird parasitism rates were 31.4% (n = 159) for vireo nests, and 16.5% (n = 31) for flycatcher nests.

Habitat selection

At the nest-site scale, both vireos and flycatchers selected nest sites with greater density of understory vegetation ($\bar{x}$ = 0.81, S.E. = 0.003) compared to unused but available surrounding habitat ($\bar{x}$ = 0.49, S.E. = 0.02) (Tables 4.1, 4.2). At the patch scale, vireo density increased with increasing patch size (range = 2.9 to 174.7 ha; $\bar{x}$ = 35.3, S.E. = 2.6) with this variable appearing in models with combined AIC_c weight of 60% (Tables 4.2, 4.3) and we found no evidence of selection for patch size (range = 12.6 to 81.4 ha; $\bar{x}$ = 37.5, S.E. = 9.6) by flycatchers (Table 4.2) At the landscape scale, density of both vireos and flycatchers was greater with increasing amounts of grassland habitat (range = 21 to 64 %; $\bar{x}$ = 51%, S.E.=0.04, for both species) in the surrounding landscape with this variable appearing in models with combined AIC_c weight of 78% and 87%, respectively (Tables 4.2, 4.3).

Nest survival

The best fitting temporal and biological model for vireo nest survival incorporated a negative effect of parasitism status (Tables 4.4, 4.5) and the estimated daily nest survival of parasitized nests was 0.923, SE=0.01 (0.16 probability of surviving to fledging; assuming 24-

day nesting period) and unparasitized nests was 0.965, SE=0.003 (0.44 probability of surviving to fledging; assuming a 24-day nesting period). Daily survival rate of vireo nests was positively related to understory density (Fig. 4.1; Tables 4.4, 4.7). There was no relationship between nest survival and patch size (Fig. 4.2; Tables 4.4, 4.7). Nest survival was negatively associated with the proportion of grass cover within 200 m (Fig. 4.3) however the 95% confidence interval of the coefficient overlapped zero (Tables 4.4, 4.7).

No temporal or biological model for flycatcher nest survival fit better than the constant-survival model (Table 4.5) with an estimated daily nest survival of 0.980 SE=0.003 (0.63 probability of surviving to fledging; assuming 24-day nesting period) At the nest scale, flycatcher daily nest survival rate was positively associated with understory density (Fig. 4.1; Tables 4.4, 4.7). Similarly, at the patch and landscape scales, daily nest survival of flycatchers was positively related to patch size and the proportion of grass cover within 200 m (Figs. 4.2, 4.3) although these were weak relationships (Tables 4.4, 4.7).

Fledgling production

The best fitting temporal or biological model for the number of vireos fledging from successful nests incorporated a negative effect of parasitism status (Tables 4.4, 4.6). The mean number of vireo fledglings for unparasitized nests (n=166) was 2.97 (SE=0.08) and 1.38 (SE=0.42) fledglings per parasitized nest (n=8). There was a negative relationship between fledgling production per nest and understory density (Fig. 4.1) however confidence intervals overlapped zero (Tables 4.4, 4.7). Patch size and the amount of grassland within 200 m were both negatively associated with fledgling production (Figs. 4.2, 4.3) however the relationship was weak (Tables 4.4, 4.7).

The number of flycatchers fledged per successful nest decreased as understory density increased (Fig.4.1; Tables 4.4, 4.7) and production was greater as the amount of grassland habitat increased in the surrounding landscape (Fig. 4.3) however the relationships were weak (Tables 4.4, 4.7).

DISCUSSION

Investigating the relationship between habitat selection decisions and resulting fitness is important for understanding how individuals select habitats and for evaluating habitat quality for the conservation of declining species. Thus, our study sought to understand the multiscale habitat selection of two declining avian species breeding in patches embedded in agriculturally fragmented landscapes. We found clear but differing breeding habitat selection by our two focal species at all spatial scales examined. Despite breeding in agriculturally fragmented landscapes with high potential for maladaptive habitat selection, we found that both species had similar reproductive success to previous studies (e.g. Budnik et al. 2000, Sedgwick and Knopf 2000). Importantly, our analyses demonstrated that the choice of increased understory at nest sites for both species resulted in greater nest survival but that fledgling production results were equivocal. Furthermore, our results showed no strong relationships between selection for patch or landscape features and measures of fitness.

Despite finding evidence for multiscale habitat selection, the only feature that improved fitness, specifically nest survival, was selection for increasing understory density. This is unsurprising given that predation is the most important cause of nest failure (Ricklefs 1969). Indeed, nest predation rates have been found to be influenced by nest-site vegetation (e.g. Martin 1993, 1998, Chalfoun and Martin 2007) with generally reduced depredation rates with more concealment. Similarly, nest survival of our focal species was generally driven by nest predation, and concealment also played a role in reducing parasitism rates, especially for the vireo which had a 28% decrease in nest success when parasitized (*B.Reiley unpublished*). While high parasitism rates are common for this species (Budnik et al. 2000, Kus et al. 2010) and they often abandon parasitized nests and renest to avoid fledging cowbird young (Kosciuch et al. 2006), we found reduced parasitism rates with increased understory density. Overall, our results demonstrate that nest concealment is not only important for reducing predation but also parasitism rates.

Although we found that increased understory density was associated with both selection of nest sites and increased nest survival for both species, there was no concurrent increase in fledgling production. Indeed, selection for habitat features at all scales yielded no benefit to fledgling production from successful nests. This may be because fledgling production is primarily influenced by food availability and microclimate (e.g. Rotenberry and Wiens 1998, Martin et al. 2016). As a result, it may be that there were not sufficient differences in food resources or microclimate among nest sites for both species in our study system and their strategy for nest-site selection is primarily focused on reducing predation risk. Interestingly, we found fledgling production from successful vireo nests was most affected by brood parasitism with parasitized nests producing 1.6 fewer fledglings per successful nest. While this affected only a small proportion of successful nests for this species (4%), it could have population-level effects when combined with high rates of nest failure due to both higher predation in parasitized nests and parasitism-related nest abandonment.

Selection for increasing grassland cover by both species may be based on a hierarchical process as suggested by Johnson (1980) whereby choices at different scales may represent tradeoffs between alternative life history traits such as adult survival and fitness. For example, second order selection (i.e. choice of landscape to breed) may focus on finding a landscape with appropriate resources and potential nesting habitats (e.g. Fuller 2012) whereas finer scale selection may focus on finding a safe nest site that minimizes predation risk (e.g. Thomson et al. 2006). Selection for grassland cover may have evolved under historical conditions and may be based on innate behaviors (e.g. Clark and Shutler 1999) such that individual habitat selection decisions may be formed by evolutionary fixed habitat preferences (Chalfoun and Schmidt 2012) that historically provided an adaptive advantage. However, in agriculturally fragmented landscapes where nest parasites and predators are concentrated (e.g. Batáry and Baldi 2004), choice of these features may no longer provide a strong adaptive advantage. While neither of our focal species selected grassland cover maladaptively (e.g. Kokko and Sutherland 2001, Schlaepfer et al. 2002, Robertson and Hutto

2006), we found it provided no fitness benefit. This may be because selection for surrounding grassland cover may provide a fitness benefit not evaluated in this study such as pairing success (e.g. Habib et al. 2007), extra-pair paternity (e.g. Biagolini et al. 2017), adult or fledgling survival (e.g. Bayne et al. 2002, Streby et al. 2014, respectively), or could be the result of conspecific attraction (Ward and Schlossberg 2004).

Interestingly, while vireos selected larger patches, flycatchers did not; however we found that nest survival improved as patch size increased for flycatchers (though not a strong relationship) but not for vireos. This difference may be related to Brown-Headed Cowbird parasitism rates. Post hoc analysis suggested that as patch size increased probability of parasitism increased for vireos, but decreased for flycatchers. Generally, flycatcher nests were less likely to be parasitized in larger patches. While flycatcher nests were not heavily parasitized by traditional standards, the parasitism rates observed in our study (13%) was high compared to the 8–10% observed in previous studies (Friedman 1963, Walkinshaw 1966, Berger 1967, McCabe 1991). Response of flycatchers to parasitism, however, was variable. Similar to observations by Budnik et al. (2000), most parasitized nests were either abandoned and in a few cases, similar to Sedgwick and Knopf (1988) and Whitfield (1990), we witnessed flooring, where an additional nest is placed over the cowbird egg if laid prior to clutch initiation (*B. Reiley unpublished*) which may explain why we saw higher nest success for this species in larger patches given that parasitism rates are generally higher in smaller patches (Benson et al. 2013). Surprisingly, fledgling production per nest was comparable for both species relative to patch size. These results may provide insights into population regulation for these species. Given the lack of relationship, it appears that for both species there was similar reproductive performance across all sites providing evidence for a case of ideal free model (Fretwell and Lucas 1970) which predicts that as the density increases within a habitat there are fewer high quality breeding sites to occupy and thus reproductive output is similar among habitats of differing quality.

Beyond broader implications for understanding relationships between habitat selection and reproductive performance, these results have implications for conservation and management for our focal species. Shrubland birds have declined at national, regional, and state levels (Herkert, 1995, Sauer et al. 2011). The extent of these declines for our focal species has led to them being listed as species of conservation concern (USFWS 2008). To attract increased numbers of both species, land managers should focus on creating or maintaining grassy patches with patches of thick shrubs (e.g., *Cornus* or *Salix* spp.) or young hardwood trees embedded in landscapes with increased grassland cover. Importantly, to reduce the threat of cowbird parasitism for both species managers should avoid creating habitat patches with increasing cropland cover. To increase nest success and fledgling production for both species, managers should provide increased opportunities for safe nests sites by maintaining dense vegetation.

To summarize, our study demonstrates that for two species, fine-scale habitat selection decisions appear to match fitness outcomes whereas habitat selection at broader scales did not appear to provide fitness benefits. Importantly, even though larger patches and increased grass cover surrounding patches did not lead to enhanced reproductive output, these large-scale features were associated with increased density of nesting birds. When combined with suitably dense nesting habitat, larger patches in landscapes with significant grass cover will produce more fledglings per unit area than smaller patches in less grassy landscapes which could help inform conservation decisions that aid in the recovery of these at-risk species. While the current study focused on limited measures of fitness, inclusion of a broader set of fitness measures such as adult survival (e.g. Chalfoun and Schmidt 2012), fledgling survival (e.g. Streby et al. 2014), or obtaining data on individual traits that indicate individual quality, such as lifetime reproductive success (e.g. Germain and Arcese 2014), may elucidate whether habitat preferences that appear neutral, non-ideal, or maladaptive based on one measure of fitness may actually be adaptive based on an alternative measure. Doing so would help better understand how to identify what specific habitat features, including the appropriate

scale, are important for demographic processes and could lead to improved management to maintain or improve habitat for declining species.

LITERATURE CITED (Formatted for Journal of Animal Ecology)

Arlt, D., & Pärt, T. (2007) Nonideal breeding habitat selection: A mismatch between preference and fitness. *Ecology*, 88, 792–801.

Batáry, P., & Báldi, A. (2004) Evidence of an edge effect on avian nest success. *Conservation Biology*, 18, 389–400.

Bayne E. M., & Hobson K. A. (2002) Apparent survival of male ovenbirds in fragmented and forested boreal landscapes. *Ecology*, 83, 1307–1316.

Benson, T. J., Chiavacci, S. J., & Ward, M. P. (2013) Patch size and edge proximity are useful predictors of brood parasitism but not nest survival of grassland birds. *Ecological Applications*, 23, 879–887.

Berger, A. J. (1967) Traill's Flycatcher in Washtenaw County, Michigan. *Jack-Pine Warbler*, 45, 117–123.

Beyer, H. L. (2012) Geospatial Modelling Environment (Version 0.7.2.1). URL: http://www.spatialecology.com/gme. Accessed 25 September, 2014.

Biagolini, C., Westneat, D. F., & Francisco, M. R. (2017) Does habitat structural complexity influence the frequency of extra pair paternity in birds? *Behavioral Ecology and Sociobiology*, 71, 101.

Bock, C. E., & Jones, Z. F. (2004) Avian habitat evaluation: should counting birds count? *Frontiers in Ecology and the Environment*, 2, 403–410.

Boryan, C., Yang, Z., Di, L., & Hunter, K. (2014) A new automatic stratification method for U.S. agricultural area sampling frame construction based on the Cropland Data Layer. pp. 1939–1404, May 2014, DOI: 10.1109/JSTARS.2014.2322584.

Buckland, S. T., Burnham, D. R, Anderson, K. P., Laake, J. L. Borchers, D. L., & Thomas, L. (2001) Distance Sampling: Estimating Abundance of Biological Populations. Chapman & Hall, London.

Burnham, K. P., & Anderson, D. R. (2002) Model selection and multimodel inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.

Budnik, J. M., Ryan, M. R., & Thompson III, F. R. (2000) Demography of Bell's Vireos in Missouri grassland-shrub habitats. *Auk*, 117, 925–935.

Chalfoun, A. D., Thompson III, F. R., & Ratnswamy, M. J. (2002) Nest predators and fragmentation: a review and meta-analysis. *Conservation Biology*, 16, 306–318.

Chalfoun, A. D., & Martin, T. E. (2007) Assessments of habitat preferences and quality depend on spatial scale and metrics of fitness. *Journal of Applied Ecology*, 44, 983–992.

Chalfoun, A. D., & Schmidt, K. A. (2012) Adaptive breeding-habitat selection: Is it for the birds? *Auk*, 129, 589–599.

Clark, R. G., & Shutler, D. (1999) Avian habitat selection: pattern from process in nest-site use by ducks? *Ecology*, 80, 272–287.

Donovan, T. M., Thompson III, F. R., Faaborg, J., & Probst, J. R. (1995) Reproductive success of migratory birds in habitat sources and sinks. *Conservation Biology*, 9, 1380–1395.

ESRI. 2012. ArcGIS version 10.1. Earth Systems Research Institute, Redlands, California, USA.

Fiske, I. J., & Chandler, R. B. (2011) Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. *Journal of Statistical Software*, 43, 1–23.

Fretwell, S. D., & Lucas, H. L. (1970) On territorial behavior and other factors influencing habitat distribution in birds. I. Theoretical development. *Acta Biotheoretica*, 19, 16–36.

Friedman, H. (1963) Host relations of the parasitic cowbirds. Vol. 233. Washington: U.S. National Museum Bulletin.

Fuller R. J. (2012) Birds and habitat: relationships in changing landscapes. New York: Cambridge University Press.

Germain, R. R., & Arcese, P. (2014) Distinguishing individual quality from habitat preference and quality in a territorial passerine. *Ecology*, 95, 436–445.

Gilroy, J. J., Anderson, G. Q. A., Vickery, J.A., Grice, P. V., & Sutherland, W. J. (2011)

Identifying mismatches between habitat selection and habitat quality in a ground nesting farmland bird. *Animal Conservation*, 14, 620–629.

Graber, R. R., Graber, J. W., & Kirk, E. L. (1974) Illinois birds: Tyrannidae. Illinois Natural History Survey Biological Notes no. 86.

Habib L., Bayne E. M., & Boutin, S. (2007) Chronic industrial noise affects pairing success and age structure of Ovenbirds (*Seiurus aurocapilla*). *Journal of Applied Ecology*, 44, 176–184.

Herkert, J. R. (1995) An analysis of Midwestern breeding bird population trends: 1966 – 1993. *American Midland Naturalist*, 134, 41–50.

Herkert, J. R. (2007) Conservation Reserve Program benefits on Henslow's Sparrows within the United States. *Journal of Wildlife Management*, 71, 2749–2751.

Holmes, R. T., Marra, P. P., & Sherry, T. W. (1996) Habitat-specific demography of breeding black-throated blue warblers (*Dendroica caerulescens*): implications for population dynamics. *Journal of Animal Ecology,* 71, 183–195.

Hoover, J. P., Brittingham, M. C., & Goodrich, L. J. (1995) Effects of forest patch size on nesting success of Wood Thrushes. *Auk*, 112, 146–155.

Jaenike, J., & Holt, R. D. (1991) Genetic variation for habitat preference: evidence and explanations. *American Naturalist,* 137, 567–590.

Johnson D. H. (1980) The comparison of usage and availability measurements for evaluating resource preference. *Ecology*, 61, 65–71.

Kokko, H., & Sutherland, W. J. (2001) Ecological traps in changing environments: ecological and evolutionary consequences of a behaviorally mediated Allee effect. *Evolutionary Ecology Research*, 3, 537–551.

Kolbe, J. J., & Janzen, F. J. (2002) Impact of nest-site selection on nest success and nest temperature in natural disturbed habitats. *Ecology*, 83, 269–281.

Kus, B., Hopp, S. L., Johnson, R. R., & Brown, B. T. (2010) Vireo (*Vireo bellii*), The Birds of North America online (A. Poole, ed.) Ithaca: Cornell lab of Ornithology; Retreived

from Birds of North America
online:http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/035
doi.org/10.2173/bna.35

Kosciuch, K. L., Parker, T. H., & Sandercock, B. K. (2006) Nest desertion by a cowbird host: an antiparasite behavior or a response to egg loss? *Behavioral Ecology*, 17, 917–924.

Lahti, D.C. 2001.The "edge effect on nest predation" hypothesis after 20 years. *Biological Conservation*, 99, 365–374.

Lloyd, J. D., & Martin, T. E. (2005) Reproductive success of Chestnut-collared Longspurs in native and exotic grasslands. *Condor*, 107, 363–374.

Mägi, M., Mänd, R., Tamm, H., Sisask, E., Kilgas, P., & Tilgar, V. (2009) Low reproductive success of Great Tits in the preferred habitat: A role of food availability. *Ecoscience*, 16, 145–157.

Martin, T. E. (1993) Nest predation and nest sites: New perspectives on old patterns. *BioScience*, 43, 523–532.

Martin T. E. (1998) Are microhabitat preferences of coexisting species under selection and adaptive? *Ecology*, 79, 656–670.

Martin, T. E., & Geupel, G. R. (1993) Nest monitoring plots: methods for locating nests and monitoring success. *Journal of Field Ornithology*, 64, 507–519.

Martin, T. E., & Roper, J. J. (1988) Nest predation and nest site selection of a western population of the Hermit Thrush. *Condor*, 90, 51–57.

Martin, T. E., Paine, C. R., Conway, C. J., Hochachka, W. M., Allen, P., & Jenkins, W. (1997) BBIRD field protocol. Montana Cooperative Wildlife Research Unit, University of Montana, Missoula.

Martin, T. E., Boyce, A. J., Fierro-Calderon, K., Mitchell, A. E., Armstad, C. E., Mouton, J., & Soudi, E. E. B. (2017) Enclosed nests may provide greater thermal than nest predation benefits compared with open nests across latitudes. *Functional Ecology*, 31, 1231–1240.

Mazerolle, M. J. (2012) Package 'AICcmodavg', version 1.25. R Foundation for Statistical Computing, Vienna. www.R-project.org

McCabe, R. A. (1991) The little green bird: ecology of the Willow Flycatcher. Madison, WI: Rusty Rock Press.

McConnell, M. D., Monroe, A. P., Burger, L. W., & Martin, J. A. (2017) Timing of nest vegetation may obscure adaptive significance of nest-site characteristics: A simulation study. *Ecology and Evolution*, 7, 1259–1270.

McCoy, T. D., Ryan, M. R., Kurzejeski, E. W., & Burger Jr., L. W. (1999) Conservation Reserve Program: source or sink habitat for grassland birds in Missouri? *Journal of Wildlife Management*, 63, 530–538.

McKellar, A. E., Kesler, D. C., & Walters, J. R. (2015) Resource selection reflects fitness associations for an endangered bird in restored habitat. *Animal Conservation*, 19, 131–138.

Misenhelter M. D., & Rotenberry, J. T. (2000) Choices and consequences of habitat occupancy and nest site selection in Sage Sparrows. *Ecology*, 81, 2892–2901.

Orians, G. H., & Wittenberger, J. F. (1991) Spatial and temporal scales in habitat selection. *American Naturalist*, 137, S29–S49.

Pabian, S. E., Wilson, A. M., & Brittingham, M. C. (2013) Mixed response of farmland birds to the conservation reserve enhancement program in Pennsylvania. *Journal of Wildlife Management*, 77, 616–625.

Pärt, T. (2001) Experimental evidence of environmental effects on age-specific reproductive success: the importance of resource quality. *Proceedings of the Royal Society of London B*, 268, 2267–2271.

Perot, A., & Villard, M. A. (2009) Putting density back into the habitat quality equation: case study of an open nesting forest bird. *Conservation Biology*, 23, 1550–1557.

Pulliam, H. R., & Danielson, B. J. (1991) Sources, sinks, and habitat selection: a landscape perspective on population dynamics. *American Naturalist*, 137, 50–66.

Quinn, J. E., Johnson, R. J., & Brandle, J. R. (2014) Identifying opportunities for conservation

in embedded in cropland anthromes. *Landscape Ecology*, 29, 1811–1819.

Ralph, C. J., Sauer, J. R., & Droege, S. (1993) Managing and monitoring bird populations using point counts: standards and applications. USDA Forest Service General Technical Report PSW-GTR-149. Pacific Southwest Research Station, USDA Forest Service, Albany, CA, USA.

Ralph, C. J., Sauer, J. R., & Droege, S. (1995) Monitoring bird populations by point count. USDA Forest Service General Technical Report PSW-149.

Ricklefs, R. E. (1969) An analysis of nesting mortality in birds. *Smithsonian Contributions to Zoology*, 9, 1–48.

Robel, R. J., Briggs, J. N., Dutton, A. D., & Hurlburt, L. C. (1970) Relationships between visual obstruction measurements and weight of grassland vegetation. *Journal of Range Management*, 23, 295–297.

Robertson, B. A., & Hutto, R. L. (2006) A framework for understanding ecological traps and an evaluation of existing evidence. *Ecology*, 87, 1075–1085.

Rotenberry, J. T., & Wiens, J. A. (1998) Foraging patch selection by shrubsteppe sparrows. *Ecology*, 79, 1160–1173.

Royle, J. A., Dawson, D. K., & Bates, S. (2004) Modeling abundance effects in distance sampling. *Ecology*, 85, 1591–1597.

Rush, S. A., & Stutchbury, B. J. M. (2008) Survival of fledgling Hooded Warblers (*Wilsonia citrina*) in small and large forest fragments. *Auk*, 125, 183–191.

SAS Institute. (2010) SAS/STAT 9.3 user's guide. SAS Institute, Cary, North Carolina, USA.

Sauer, J. R., Hines, J. E., & Fallon, J. (2011) The North American Breeding Bird Survey, results and analysis 1966–2011, ver. 07.03. 2013. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

Schlaepfer, M. A., Runge, M. C., & Sherman, P. W. (2002) Ecological and evolutionary traps. *Trends in Ecology and Evolution*, 17, 474–480.

Sedgwick, J. A. (2000) Willow Flycatcher (Empidonax traillii), The Birds of North America (P. G. Rodewald, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the *Birds*

of North America: https://birdsna.org/Species-Account/bna/species/wilfly DOI: 10.2173/bna.53

Sedgwick, J. A., & Knopf, F. L. (1988) A high incidence of Brown-headed Cowbird parasitism of Willow Flycatchers. *Condor*, 90, 253–256.

Shaffer, T. L. (2004) A unified approach to analyzing nest success. *Auk*, 121, 526–540.

Streby, H. M., Refsnider, J. M., & Anderson, D. E. (2014) Redefining reproductive success in songbirds: moving beyond the nest success paradigm. *Auk*, 131,718–726.

Tewksbury, J. J., Garner, L., Garner, S., Lloyd, J. D., Saab, V., & Martin, T. E. (2006) Tests of landscape influence: nest predation and brood parasitism in fragmented ecosystems. *Ecology*, 87, 759–768.

Thomson, R. L., Forsman, J. T., Sarda´-Palomera, F., & Mönkeönnen, M. (2006) Fear factor: prey habitat selection and its consequences in a predation risk landscape. *Ecography*, 29, 507–514.

United States Fish and Wildlife Service. (2008) Birds of Conservation Concern 2008. United States Department of the Interior, Fish and Wildlife Service, Division of Migratory Bird Management, Arlington, VA.

Walkinshaw, L. H. (1961) The effect of parasitism by the Brown-headed Cowbird on *Empidonax* flycatchers in Michigan. *Auk*, 78, 266-268.

Ward M. P., & Schlossberg S. (2004) Conspecific attraction and the conservation of territorial songbirds. *Conservation Biology*, 18, 519–525.

Weldon, A. J., & Haddad, N. M. (2005) The effects of patch shape on Indigo Buntings: evidence for an ecological trap. *Ecology*, 86, 1422–1431.

Whitfield, M. J. (1990) Willow Flycatcher reproductive response to Brown-headed Cowbird parasitism. Master's Thesis, California State University, Chico, California, USA.

Woodward, A. A., Fink, A. D., and Thompson III, F. R. (2001) Edge effects and ecological traps: effects on shrubland birds in Missouri. *Journal of Wildlife Management*, 65, 668–675.

TABLES AND FIGURES

Table 4.1 Model selection results for nest-site selection of Bell's Vireos (*n*=572) and Willow Flycatchers (*n*=132) in restored farmland habitats in western Illinois, USA, 2012–2015.

Model	K	AIC*c*	ΔAIC*c*	w_i
Bell's Vireo				
% Understory density	**3**	**1227.83**	**0.00**	**1.00**
% Shrub	3	1565.25	337.42	0.00
% Grass	3	1583.62	355.79	0.00
Constant	3	1589.31	361.48	0.00
% Forb	3	1590.99	363.16	0.00
Willow Flycatcher				
% Understory density	**3**	**319.71**	**0.00**	**1.00**
Constant	3	370.75	51.04	0.00
% Forb	3	371.59	51.88	0.00
% Grass	3	372.41	52.70	0.00
% Shrub	3	372.76	53.05	0.00

Table 4.2 Model-averaged parameter estimates (β) and 95% confidence limits for best fit nest site, patch, and landscape habitat selection variables for Bell's Vireo and Willow Flycatcher in restored farmland habitats in western Illinois, USA, 2012–2015.

	Bell's Vireo			Willow Flycatcher		
Parameter	β	LCL	UCL	β	LCL	UCL
Nest site selection						
% Understory density	0.006	0.002	0.010	0.011	-0.001	0.023
Landscape selection						
% Grass within 200 m	0.370	0.154	0.586	0.340	0.164	0.516
Patch preference						
Patch size	0.200	0.030	0.380	0.030	-0.080	0.150

Table 4.3 Number of parameters, ΔAIC_c values, and model weights for all patch and landscape models used to evaluate habitat selection for Bell's Vireo and Willow Flycatcher in restored farmland patches (n=172) in western Illinois, USA, 2012–2015.

	Bell's Vireo				Willow Flycatcher			
Model	K	AICc	ΔAIC_C	w_i	K	AIC_c	ΔAIC_c	w_i
Landscape models								
Constant	5	3276.92	10.05	0.00	5	4585.13	25.38	0.00
Year	6	3278.95	12.09	0.00	6	4585.62	25.87	0.00
% Grassland within 200 m	**7**	**3266.87**	**0.00**	**0.46**	7	4560.56	0.82	0.30
% Forest within 200 m	7	3278.10	11.23	0.00	7	4564.37	4.62	0.04
% Restored habitat within 200 m	7	3278.85	11.98	0.00	7	4576.91	17.17	0.00
% Forest + % grass	8	3268.48	1.61	0.21	**8**	**4559.75**	**0.00**	**0.45**
% Restored habitat + % forest	8	3277.88	11.01	0.00	8	4563.11	3.37	0.08
% Restored habitat + % grass	8	3267.61	0.74	0.32	8	4562.36	2.61	0.12
Patch Models								
Constant	5	3276.92	1.39	0.30	**5**	**4585.13**	**0.00**	**0.47**
Year	6	3278.95	3.42	0.11	6	4585.62	0.49	0.37
Patch size	**7**	**3275.53**	**0.00**	**0.60**	7	4587.30	2.17	0.16

Table 4.4 Model-averaged parameter estimates (β) and 95% confidence limits, for best fit biological and habitat variables from logistic exposure models of nest survival and number of young fledged from successful nests (fledgling production) from Bell's Vireo and Willow Flycatcher nests at restored farmland habitats in western Illinois, USA, 2012–2015.

	Nest survival						Fledgling production					
	Bell's Vireo			Willow Flycatcher			Bell's Vireo			Willow Flycatcher		
Parameter	β	LCL	UCL	β	LCL	UCL	β	LCL	UCL	β	LCL	UCL
Temporal and biological models												
Intercept				3.980	3.725	4.235				1.049	0.936	1.163
Parasitism[a]	-0.770	-1.358	-0.182				0.769	0.181	1.357			
Habitat models												
% Understory density	**0.072**	**0.064**	**0.079**	**0.072**	**0.020**	**0.380**	0.000	-0.002	0.003	-0.003	-0.010	0.004
Patch size	0.000	-0.004	0.004	0.006	-0.004	0.016	-0.002	-0.005	0.001	-0.001	-0.004	0.004
% Grass within 200 m	-0.030	-0.069	0.009	0.620	-0.811	2.051	-0.020	-0.451	0.412	0.035	-0.506	0.576

[a]Probability of Brown-headed Cowbird parasitism.

Table 4.5 Model selection results of temporal and biological factors related to nest survival of Bell's Vireo and Willow Flycatcher nests at restored farmland habitats in western Illinois, USA, 2012–2015.

Models	K	AIC*c*	ΔAIC*c*	*wi*
Bell's Vireo				
Parasitism[a]	**3**	**1491.49**	**0.00**	**1.00**
Nest Stage × Day of Year + Year	8	1527.99	36.50	0.00
Nest Stage + Year	7	1529.34	37.85	0.00
Day of Year	2	1534.80	43.31	0.00
Year × Day of Year	5	1535.20	43.71	0.00
Constant	1	1541.68	50.19	0.00
Year	4	1542.15	50.66	0.00
Willow Flycatcher				
Constant	**1**	**429.63**	**0.00**	**0.38**
Day of Year	2	429.91	0.28	0.33
Parasitism	3	431.63	2.00	0.14
Year	4	432.93	3.30	0.07
Year × Day of Year	5	433.38	3.75	0.06
Nest Stage + Year	7	436.93	7.30	0.01
Nest Stage × Day of Year + Year	8	437.22	7.59	0.01

[a]Probability of Brown-headed Cowbird parasitism.

Table 4.6. Model selection results of temporal and biological factors related to per nest fledgling production of Bell's Vireo and Willow Flycatcher nests at restored farmland habitats in western Illinois, USA, 2012–2015.

Models	K	AIC*c*	ΔAIC*c*	*wi*
Bell's Vireo				
Parasitism[a]	**3**	**570.68**	**0.00**	**0.97**
Constant	1	578.93	8.25	0.02
Day of Year	2	579.29	8.61	0.01
Year × Day of Year	5	583.00	12.32	0.00
Willow Flycatcher				
Constant	**1**	**334.72**	**0.00**	**0.46**
Day of Year	2	336.32	1.60	0.21
Parasitism	3	336.70	1.98	0.17
Year × Day of Year	5	339.35	4.63	0.05

[a]Probability of Brown-headed Cowbird parasitism.

Table 4.7 Summary of results from models of nest survival and fledgling production comparing habitat selection and fitness outcomes from Bell's Vireo and Willow Flycatcher nests at restored farmland habitats in western Illinois, USA, 2012–2015.

	Bell's Vireo		Willow Flycatcher	
	Nest survival	Fledgling production	Nest survival	Fledgling production
Understory density	*Strong match*	Neutral	*Strong match*	Neutral
Patch size	Neutral	Weak mismatch		Neutral
% Grass within 200m	Weak mismatch	Weak mismatch	Weak match	Weak match

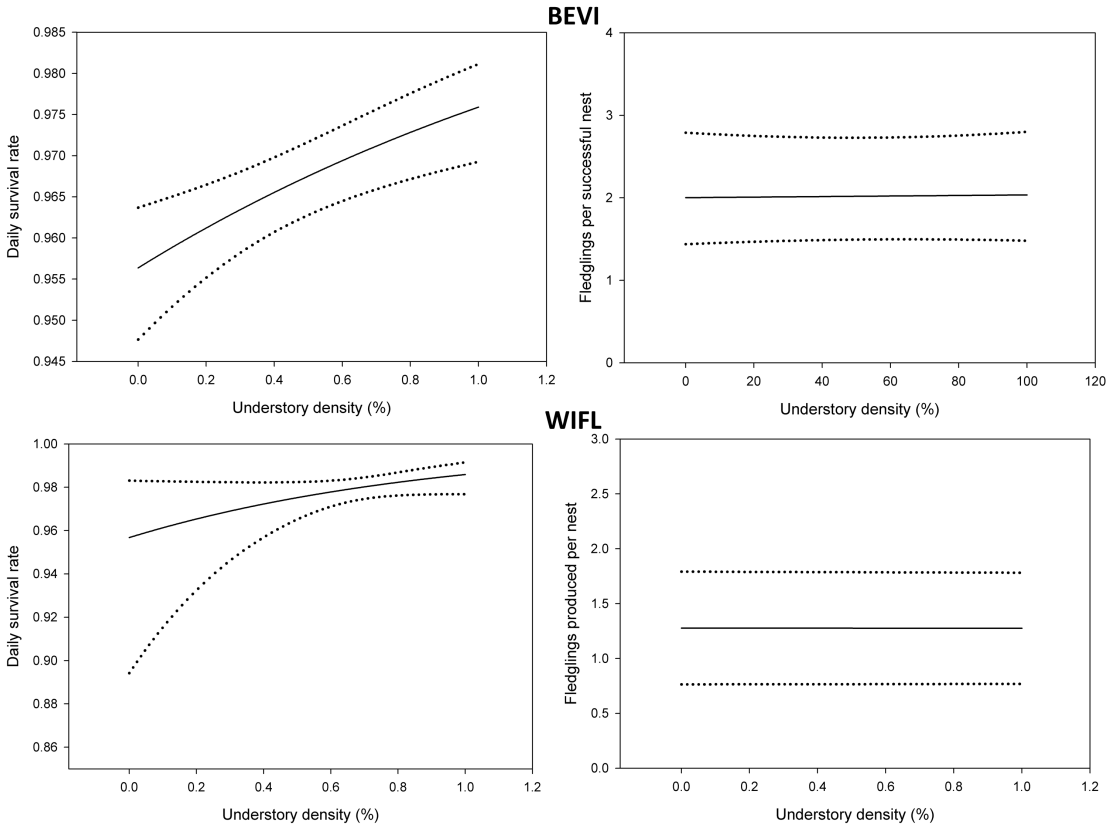

Figure 4.1 Predicted daily survival rate and numbers of fledglings produced per successful nest (with 95% confidence intervals) for Bell's Vireo and Willow Flycatcher nests relative to % understory density within 5 m of a nest.

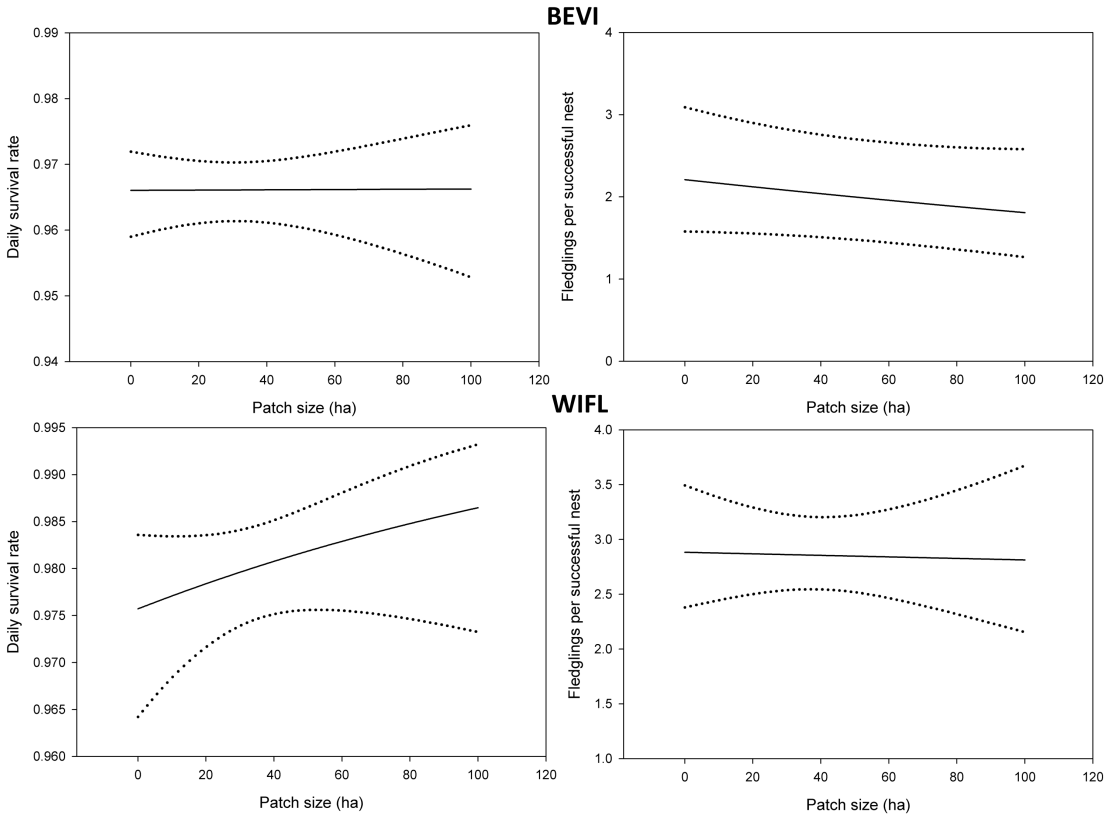

Figure 4.2 Predicted daily survival rate and numbers of fledglings produced per nest and 95% confidence intervals of Bell's Vireo and Willow Flycatcher nests relative to patch size (ha).

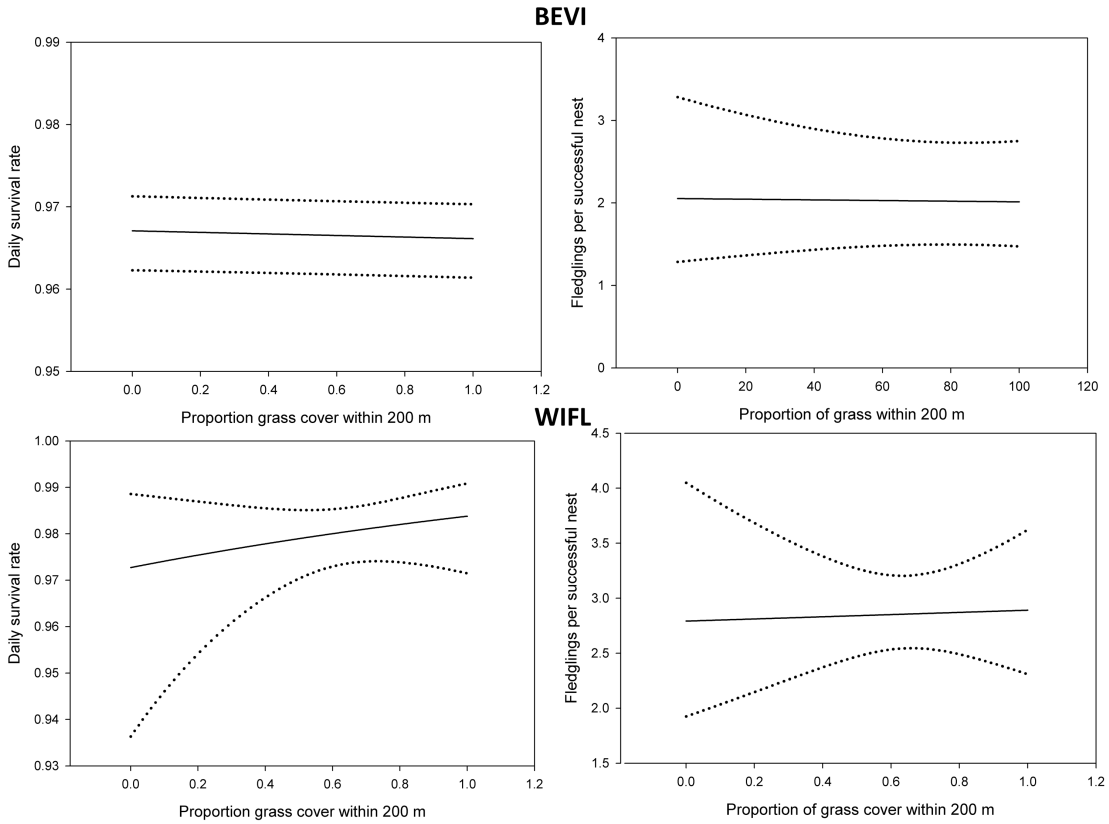

Figure 4.3 Predicted daily survival rate and numbers of fledglings produced per nest and 95% confidence intervals of Bell's Vireo and Willow Flycatcher nests relative to % grass cover within 200 m of a nest.

CHAPTER 5: SUMMARY

Programs that restore habitat on private lands (hereafter private land programs) play an important role in avian conservation within agriculturally fragmented landscapes. Yet, these programs are often implemented without specific conservation goals in mind and they could provide increased benefits if the needs of priority species are used to guide future enrollments and manage existing habitat. While the habitat needs of grassland species have received considerable attention we know much less about the needs of other groups, such as shrubland birds. Thus, the aim of Chapter 2 was to identify what local, patch, and landscape features were associated with habitat use of both declining grassland and shrubland avian species. With the help of technicians, I conducted 1,996-point count surveys over 4 years at 172 patches. Distance to the nearest tree and tree cover were the most important local scale variables with Bell's Vireo (*Vireo bellii bellii*) and Northern Bobwhite (*Colinus virginianus*) densities increasing as distance to nearest tree decreased and Field Sparrow (*Spizella pusilla*) and Yellow-billed Cuckoo (*Coccyzus americanus*) had the opposite relationship while Dickcissel (*Spiza americana*), and Willow Flycatcher (*Empidonax trailli trailli*), densities were negatively associated with increasing tree cover. This suggests that, shrubland species such as Bell's Vireo, Northern Bobwhite, and Yellow-billed Cuckoo will benefit from patchily planted shrubs or young hardwood trees and more grass associated such as Field Sparrow, Dickcissel, and Willow Flycatcher will benefit from regular management such as mowing or burning to reduce woody intrusion. Generally, landscape features within a 200-m radius around patches had the most influence on species densities. Only the Dickcissel had a strong positive relationship with increasing patch size and as a result this species will benefit from creation of large patches with increasing amounts of restored habitat in the surrounding landscape. On the other hand, creation of habitats of any size regardless of nearby restored habitat will benefit the other focal species. My results indicated that to maximize densities for all of our focal species, managers should focus on restoring agricultural fields embedded in crop or grass dominated landscapes. Based on my results, habitat restored through private land programs may provide an important conservation for both grassland and shrubland

bird species. Moreover, shifting focus to creating and managing restored private land habitats for these species will provide the highest conservation benefit for the least amount of economic output.

Ecosystem alteration has caused the decline of wildlife populations worldwide. In the U.S., avian declines have led to continental scale conservation efforts. As part of these conservation efforts natural resource professionals set population objectives to help achieve historic population levels that existed prior to widespread declines. These population objectives were then stepped down to regional and state levels. While important, the effort to set population goals provided no clear path to achieve these goals. For birds that depend on early successional habitats, programs that restore wildlife habitat on private lands likely present the best opportunity for meeting population goals. For example, the Conservation Reserve Program provides habitat on more than 3 million hectares, and no other habitat restoration effort comes close to this level of landscape-level impact. In Chapter 3, I examined the contribution of the Conservation Reserve Enhancement Program (CREP), an existing habitat reestablishment program, to state population goals for priority early successional species and if goals were not being met, what amount of habitat would be needed to achieve the goals. My population estimates indicate that CREP in Illinois is likely contributing enough, or nearly enough, birds to meet population targets for the Bell's Vireo (110% of goal) and the Willow Flycatcher (92% of goal). On the other hand, my estimates suggest that CREP provides additional habitat that helps to meet are < 33 and 7% of population goals for Field Sparrow and Northern Bobwhite, respectively. Furthermore, these results suggest that these habitats are playing an important role in achieving goals for species with historically low population sizes, but a more concerted effort will be required to achieve existing goals for species that have declined substantially. My simulations suggest that goals for all species could be met through the restoration of approximately 1% of Illinois farmland at an annual cost of approximately $76 million dollars; however, to maintain these populations restored habitat would require regular habitat management which would increase costs far beyond the additional money needed to meet the population goals.

While information on habitat use of conservation priority birds can help us understand how to create and manage habitat created by private land programs, this information could also be used to understand how species assess habitat quality. When habitat quality is heterogeneous, birds should choose to breed in the best available habitat, and natural selection should favor individuals that can accurately assess habitat quality. In light of this, we should expect to see clear links between observed habitat selection and fitness. Yet, because these reestablished habitats exist as discrete patches surrounded by agriculturally fragmented landscapes, traditional cues for assessing habitat quality may be altered. Indeed, explanations for mismatched selection and fitness are often attributed to changes in traditional cues due to anthropogenic disturbance which may affect nest predation and brood parasitism rates. Furthermore, because nest predation and brood parasitism rates are affected by habitat features immediately surrounding a nest and may be landscape dependent, evaluating links between habitat selection decisions and fitness may depend on the observed spatial scale. Thus, in Chapter 4 I examined the breeding habitat selection of two avian species of conservation concern and two measures of fitness (nest survival and number of young fledged from successful nests) at the nest, patch, and landscape scale. With the help of technicians, I found 505 active Bell's Vireo nests and 188 active Willow Flycatcher nests over three years. My results indicated that there was a positive relationship among habitat selection, nest survival, and vegetation density. In contrast, habitat selection with respect to patch size and surrounding grassland did not lead to increased nest survival or production of fledglings from successful nests. Results at the nest scale were not unexpected given most studies that have found concordance between selection and fitness outcomes at this scale due to the role habitat features surrounding a nest play in concealing nests from potential predators. Selection for larger patches and increased grass cover surrounding patches did not lead to enhanced reproductive output which may be due to relatively recent, on evolutionary time scales, changes in landscapes due to agricultural fragmentation however selection for these features may also provide an unmeasured fitness benefit not evaluated in my study. Importantly, I found no evidence of maladaptive habitat selection by either species at any scale in these agriculturally fragmented landscapes.

For managers, these findings suggest that creating or maintaining large patches in grassy landscapes with dense understory vegetation will both attract higher densities of both species and likely ensure greater reproductive output for occupying breeding pairs.