

**EFFECTS OF AGRICULTURAL LAND CONVERSION AND LANDSCAPE CONNECTIVITY ON
MOVEMENT, SURVIVAL, AND ABUNDANCE OF NORTHERN BOBWHITES (COLINUS
VIRGINIANUS) IN OHIO**

THESIS

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ABSTRACT

Conversion of land for agriculture and agricultural intensification are widely recognized as two of the largest threats to biodiversity. The northern bobwhite (*Colinus virginianus*) is one of the fastest declining species in North America primarily due to farmland conversion and agricultural intensification. Bobwhites declined 4.08% annually from 1966-2013. Bobwhites in Ohio declined 6.88% per year during the same period. As a part of a larger study that aimed to assess bobwhite demographics, this study aimed to better understand how land use conversion and farmland management affect bobwhite fall dispersal, survival, and summer abundance.

Radio telemetry was used to monitor bobwhites year round in the breeding (1 Apr-30 Sep 2010 and 1 Apr-30 Sep 2012-2014) and nonbreeding (1 Oct-31 Mar 2009-2011 and 1 Oct-31 Mar 2012-2015) seasons on 4 study sites located on private lands in Ohio. Home ranges were constructed for each individual in each season ($n = 26$). Cover maps were clipped and overlaid with summer home range boundaries to quantify home range composition. The average shift from the center of the summer home range to the center of the winter home range was 515 ± 139 (SE) m. Average overlap between seasonal home ranges was 34% or 5.0 ± 1.1 (SE) ha. The model with the most support ($w_i = 0.45$) of distances moved between summer and fall winter home ranges included summer home range percent early-successional (ES) woody cover ($\beta = -3.44 \pm 1.75$ (SE)), summer home range forest edge density ($\beta = -0.017 \pm 0.005$ (SE)), mean nearest

neighbor of ES woody ($\beta = 0.0007 \pm 0.0002$ (SE)), and summer interspersion juxtaposition index of ES herbaceous habitat ($\beta = 0.005 \pm 0.002$ (SE)) with a random location effect. Though there was not a significant correlation between overwinter survival and the degree of home range shift, bobwhites shifted home ranges more in fall when summer habitat was of poorer quality for winter survival. However, this shift was on average about a half kilometer suggesting a limited dispersal ability. Bobwhites in Ohio likely need winter cover close to nesting and brood rearing habitat.

Whistle count surveys were conducted from 2008-2010 and 2012-2014 on 29 survey points across ≥ 5 of the 6 years. Landscape composition was classified around each survey location by using a 1-km buffer on aerial photographs. In 2012, ES woody vegetation was created by treating edges in 12 survey points on 2 study sites. A total of 1096 male bobwhites were detected. The detection probability for 2008-2009 and 2012-2014 was 0.503 ± 0.0274 (SE). The detection probability for 2010 was estimated to be 0.400 ± 0.0389 (SE) which was analyzed separately due to differences in recording. The pooled estimate between sites was a decline of -0.512 detections per point per year. ES woody was the only habitat type to show a significant result in response to habitat change ($\chi^2 = 4.94$, $P = 0.026$) and was positively correlated with change in abundance (1.08 ± 0.48 (SE) detections per survey point per year). These results suggest addition of treated edges may help slow population decline at a small scale.

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TABLE OF CONTENTS

Abstract.....	ii
Acknowledgments.....	iv
Vita.....	vii
List of Tables	xi
List of Figures.....	xiv
Chapter 1: Introduction.....	1
Study Design	6
Thesis Content.....	9
Literature Cited	15
Chapter 2: Fall Dispersal of Northern Bobwhites in Fragmented Agricultural Landscapes	22
Abstract	22
Introduction.....	23
Study Area.....	27
Methods.....	28

Results	32
Discussion	35
Management Implications	41
Literature Cited	52
Chapter 3: Population Level Responses of Northern Bobwhite to Land Management	
Changes in Southwest Ohio	59
Abstract	59
Introduction	60
Study Area	64
Methods	64
Results	67
Discussion	69
Management Implications	72
Literature Cited	81
Chapter 4: Management Implications	87
Literature Cited	92
Bibliography	97

LIST OF TABLES

Table 1.1. County habitat composition and average farm size in Highland and Brown Counties, Ohio in 2012	11
Table 1.2. Landcover composition and abundance of northern bobwhites on four study sites in Brown and Highland Counties, Ohio in 2008 and 2014.	12
Table 2.1. Mean breeding season home range composition of 6 habitat types for individual northern bobwhites (1 April-30 Sep 2010, 2012, 2013, and 2014, n = 26) in Brown and Highland County, Ohio.	44
Table 2.2. Model set including corrected Akaike's Information Criterion (AIC _c), difference in corrected Akaike's Information Criterion (Δ AIC _c), model weight (w_i), and number of model parameters (K) for determining the average distance moved between summer (1 April-30 September 2010-2015) to winter home ranges (1 October- 31 March 2010-2015) of northern bobwhites in southwest Ohio..	45
Table 2.3. Best mixed effects model for log of distance moved from summer (1 April-30 September 2010-2015) to winter home ranges (1 October- 31 March 2010-2015) of northern bobwhites in southwest Ohio..	46
Table 2.4. Model set including corrected Akaike's Information Criterion (AIC _c), difference in corrected Akaike's Information Criterion (Δ AIC _c), model weight (w_i), and number of model parameters (K) for determining seasonal interactions of summer (1	

April-30 September 2010-2015) home ranges on winter survival (1 October- 31 March 2010-2015) of northern bobwhites in southwest Ohio using birds with enough locations for summer and fall home ranges ($n = 21$)..... 47

Table 2.5. Summary of model averaged estimates for parameters in models within 2.00 AIC points of the top model for determining nonbreeding season survival from summer home range habitat composition of northern bobwhites on private lands in southwest Ohio, USA (2010-2015) using birds with enough locations for summer and fall home ranges ($n = 21$).. 48

Table 2.6. Model set including corrected Akaike’s Information Criterion (AIC_c), difference in corrected Akaike’s Information Criterion (ΔAIC_c), model weight (w_i), and number of model parameters (K) for determining seasonal interactions of summer (1 April-30 September 2010-2015) home ranges on winter survival (1 October- 31 March 2010-2015) of northern bobwhites in southwest Ohio using birds with enough locations to build summer home ranges that survived to the subsequent nonbreeding season ($n = 32$)..... 49

Table 2.7. Summary of model averaged estimates for parameters in models within 2.00 AIC points of the top model for determining nonbreeding season survival from summer home range habitat composition of northern bobwhites on private lands in southwest Ohio, USA (2010-2015) using birds with enough locations to build summer home ranges that survived to the subsequent nonbreeding season ($n = 32$).. 50

Table 3.1. Model set including corrected Akaike’s Information Criterion (AIC_c), difference in corrected Akaike’s Information Criterion (ΔAIC_c), and number of model

parameters (K) for determining detection probability of northern bobwhites from 2008-2009 and 2012-2014 in southwest Ohio.	77
Table 3.2. Model set including corrected Akaike’s Information Criterion (AIC_c), difference in corrected Akaike’s Information Criterion (ΔAIC_c), and number of model parameters (K) for determining detection probability of northern bobwhites in 2010 in southwest Ohio.	78
Table 3.3. Number of northern bobwhite whistle count survey points that experienced habitat gain (+) or loss (-) of six habitat categories on 4 study sites between 2008 and 2014 in Brown and Highland Counties, Ohio.	79
Table 3.4. Percent habitat change of the total land area and change in abundance on four study sites in Brown and Highland Counties, Ohio. Fee, Thurner, and Wildcat were studied from 2008-2014. Peach was studied from 2010-2014.	80

LIST OF FIGURES

Figure 1.1. Population trend map of the northern bobwhite (<i>Colinus virginianus</i>) detected by the Breeding Bird Survey 1966-2013. (http://www.mbr-pwrc.usgs.gov/bbs/tr2013/tr02890.htm)	13
Figure 1.2. Locations of four study sites used to investigate northern bobwhite population demographics and movement in Southwest Ohio 2009-2015 overlaid on a National Land Cover Database image. The inset image shows counties in Ohio where bobwhites are present at low densities (Ohio Division of Wildlife 2014).....	14
Figure 2.1. Relationship between summer home range habitat composition of northern bobwhites on distance moved from the centroid of the summer home range to the centroid of the winter home range in Brown and Highland Counties, Ohio, 2010-2015. The relationship between log distance and summer home range percent early-successional (ES) woody, forest edge density (ED), mean nearest neighbor (MNN) ES woody, and interspersed juxtaposition index (IJI) ES herbaceous are shown.....	51
Figure 3.1. Plots of competitive models estimating detection probability of whistling male northern bobwhites in Southwest Ohio 2008-2009 and 2012-2014.....	74
Figure 3.2. Plots of competitive models estimating detection probability of whistling male northern bobwhites in Southwest Ohio in 2010.....	75

Figure 3.3. Site level abundance of northern bobwhites determined from whistle count surveys conducted from 2008-2014 on 4 study sites located on private lands in Brown and Highland Counties, Ohio..... 76

CHAPTER 1: INTRODUCTION

Conversion of land for agriculture is widely recognized as the most significant of human ecosystem alterations (Matson et al. 1997, Burger et al. 2006, Rashford et al. 2012, Shaw et al. 2014). Conversion has been historically widespread and continues today (Rashford et al. 2010). Biodiversity has declined from agricultural land conversion, intensification, and subsequent fragmentation of habitat (Matson et al. 1997, Chamberlain et al. 2000, Burger et al. 2006, Riffell et al. 2008, Rashford et al. 2010, Conover et al. 2014). Agriculture dominates land use in the United States. Rural land comprises 71% of the contiguous 48 states, and cropping and grazing comprises 50% of the land (Burger et al. 2006).

Agriculture is also intensifying. Farmers are switching to high yield crops, and the use of chemical fertilizers, pesticides, irrigation, and mechanization are increasing (Matson et al. 1997, Chamberlain et al. 2000). The average field size is increasing and fencerows are declining (Vance 1976, Chamberlain et al. 2000, Conover et al. 2014). Fencerows are important because they provide wildlife habitat and cover on farms (Vance 1976, Chamberlain et al. 2000). Bird declines have been associated with intensification due to reduction in food resources, less suitable nesting habitat, and direct mortality (Chamberlain et al. 2000). Some predators are able to increase in numbers in agricultural landscapes because linear edges of crop fields provide corridors and there is a higher diversity and availability of prey (Chamberlain et al. 2000, Keuhl and Clark 2002,

Evans et al. 2004, Conover et al. 2014, Seckinger et al. 2014). Game birds specifically have been shown to be negatively affected by increased avian predators on agricultural landscapes (Chamberlain et al. 2000).

Habitat loss from agricultural intensification and land conversion has increased the amount of fragmentation by decreasing patch size, increasing patch isolation, and increasing edge (Keuhl and Clark 2002, Gehring and Swihart 2002, Boulinier et al. 2001). Habitat loss and fragmentation are intricately linked, so separating the effects on populations is difficult (Baguette et al. 2006). However, it is known that habitat loss and fragmentation can affect the abundance and distribution of species by affecting interseasonal movement ability (movement to new habitat between seasons, hereafter dispersal) (Rodriguez et al. 2001, Gobeil and Villard 2002, Vos et al. 2002, Robertson and Radford 2009). Fragmentation reduces the ability of animals to disperse by increasing the distance necessary to travel to complete dispersal, often through open habitat and can make the path more complex (Whittingham and Evans 2004, Schtickzelle et al. 2006, Bonte et al. 2012, Chaine and Clobert 2012).

Dispersal is important for populations because it enables species to reach habitats needed throughout the annual life cycle (Fies et al. 2002, Van Oort et al. 2010, Fedy et al. 2012). Dispersal is costly to the individual organism because movement makes it conspicuous to predators, costs incurred during the dispersal can make it harder to evade predators after the dispersal, and organisms do not have the advantage of knowledge of their prior habitats (Townsend et al. 2003, Bonte et al. 2012, Chaine and Clobert 2012). However, dispersal can increase individual fitness through finding more favorable habitat

for each season, avoidance of resource competition in high density areas, and by finding higher quality habitat (Perrin and Maxalov 2000, Barton et al. 2009, Van Oort et al. 2010, Bonte et al. 2012, Chaine and Clobert 2012, Fedy et al. 2012).

The northern bobwhite (*Colinus virginianus*), hereafter bobwhite, is one of the fastest declining species in North America primarily due to farmland conversion and agricultural intensification (Roseberry and Sudkamp 1998, Taylor et al. 1999, Williams et al. 2000, Cook 2004, Seckinger et al. 2006, Riddle et al. 2008, Duren et al. 2011, Bowling et al. 2014). Bobwhites declined 4.08% annually from 1966-2013 according to Breeding Bird Survey data (Fig. 1.1, Sauer et al. 2014). Bobwhites are tightly associated with agriculture and have higher numbers on areas with substantial agricultural components. They require diverse, patchy, predominately open landscapes with abundant woody edge for escape cover like that a fencerow would provide and with grasses for nesting (Roseberry and Sudkamp 1998, Riddle et al. 2014). Agricultural intensification has caused a decrease in cover diversity, an increase in patch size, and a reduction in fencerows, reducing habitat availability and quality for bobwhite (Vance 1976, Taylor et al. 1999, Chamberlain et al. 2000, Cook 2004, Flock 2006, Seckinger et al. 2006, Lohr et al. 2011, Conover et al. 2014, Riddle et al. 2014). Further, changing forest practices have led to advanced forest succession, reducing the amount of understory cover (Cook 2004, Seckinger et al. 2006, Bowling et al. 2014, Riddle et al. 2014).

Bobwhites are relatively immobile, and the majority spends their entire lives within about 1 km (Stoddard 1931, Murphy and Baskett 1952, Lewis 1954, Agee 1957, Rosene 1969, Fies et al. 2002, Townsend et al. 2003, Liberati 2013). Large scale habitat

loss and subsequent large scale fragmentation could affect bobwhite populations by reducing dispersal ability (Fies et al. 2002, Townsend et al. 2003). Though patch diversity and edge are beneficial to bobwhites (Roseberry and Sudkamp 1998, Riddle et al. 2014), expansive areas of matrix conditions like forest can significantly impede dispersal and separate populations. Townsend et al. (2003) found bobwhites that dispersed had a 1.5 times greater survival rate in summer than bobwhites that did not. They suggested dispersal may help bobwhite populations through gene flow and dispersal rescue. Increasing habitat loss and subsequent fragmentation may disrupt dispersal and impair these important processes.

Bobwhites have been one of the most intensively researched species since the 1920s (Guthery 1997, Taylor et al. 1999, Flock 2006). Studies aimed at improving habitat quality have generally been unsuccessful at improving populations while studies aimed at increasing useable space have had more success (Guthery 1997). The Conservation Reserve Program (CRP) was developed in the Food Security Act of 1985 to take highly erodible and environmentally sensitive land out of agricultural production in favor of conservation practices (Stubbs 2014). The CRP subsequently added habitat for wildlife, and in the 1996 and 2002 farm bills, wildlife habitat became a specific goal of US agricultural policy (Burger et al. 2006). CRP has been documented to increase many avian populations (Riffell et al. 2008, Wilson et al. 2010, Evans et al. 2014, Stubbs 2014), notably bobwhite populations (Bowling et al. 2014, Evans et al. 2014, Riddle et al. 2014). A new practice, CP33-Habitat Buffers for Upland Birds, was implemented in 2004 to meet bobwhite recovery plans developed by the Northern Bobwhite Conservation

Initiative (NBCI). Despite the successes of the CRP in wildlife conservation, enrollment in the CRP is declining due to high commodity prices with low rental rates, and as contracts expire, many farmers are not renewing. Further, the 2014 farm bill reduces the cap of eligible acres from 32 million to 24 million by FY2018 and allows early termination of contracts after 5 years instead of 10 for non-environmentally sensitive practices (Stubbs 2014). With such large modification to these rules and the concurrent continued loss of remaining early successional habitat, research is needed to explore how loss of CRP affects wildlife populations.

A new, relatively untested management technique for grassland/shrubland associated species in agricultural landscapes is woodlot edge enhancement. This technique attempts to create minimal disturbance to farm production while creating early-successional woody habitat by hinge-cutting trees along the borders of fields (Iowa Department of Natural Resources 2001, Dailey and Hutton 2003, Hoosier Heartland Resource Conservation and Development Council 2007, Kentucky Department of Fish and Wildlife Resources 2012, McPeake and Roberg 2012). Though widely recommended by universities and agencies, this technique has relatively few published, peer reviewed, empirical assessments of efficacy (to date only Smith et al. 2014). Smith et al. (2014) reported a negative effect of woodlot edge enhancement on survival of northern bobwhites, so its effectiveness as a management tool needs to be further assessed despite its widespread recommendation.

As a part of a larger study that aimed to assess bobwhite demographics affecting population growth rates and habitat factors that affect survival, I aimed to better

understand how land use conversion and farmland management affect bobwhite fall dispersal, survival, and summer abundance. I specifically wanted to address how summer home range composition affects subsequent seasonal movement to the non-breeding season home range and if there are any carry over effects of summer home range habitat composition on winter survival. Additionally, I wanted to assess how land use change affects bobwhite abundance. I was especially interested in how implementation or retraction of conservation practices used on private farmlands affects bobwhite movement, survival, and abundance throughout the annual life cycle in order to make better management recommendations. My broader objective was to better understand how to manage farmland associated birds on private lands as habitat continues to decline from agricultural practices.

STUDY DESIGN

Study Sites

Study sites were chosen in the core of the bobwhite range in Ohio in Highland and Brown Counties (Fig. 1.2, Spinola and Gates 2008, Ohio Division of Wildlife 2014). Highland and Brown Counties are in the glaciated till plains physiogeographic region (Ohio Division of Geologic Survey 1998) and the Eastern Tall Grass Prairie Bird Conservation Region (Terhune and Palmer 2011). The gently undulating topography of the region restricts the size of row crop fields, providing better habitat for bobwhites than other parts of the state (Guthery 1997). Highland County is 74.7% cropland, 9.9% pastureland, 10.5% woodland, and 4.9% other. The average farm size in 2012 was 187

acres. Brown County is 67.5% cropland, 10.8% pastureland, 15.7% woodland, and 6.1% other. The average farm size in 2012 was 150 acres (Table 1.1, National Agricultural Statistics Services 2012). The long-term (30 yr) mean yearly temperature in the region was 11.1°C, the mean temperature during the non-breeding season (1 October-31 March) was 3.68°C, and the mean temperature during the breeding season (1 April-30 September) was 18.65°C. Long-term (30 yr) mean annual precipitation in the region was 108.10 cm, and mean annual snowfall was 45.21 cm (National Oceanic and Atmospheric Administration [NOAA] 2015).

There were four study sites located in Brown and Highland Counties: Fee and Peach fell fully within Highland County while Wildcat fell on the border of Highland and Brown Counties. Thurner was fully within Brown County. Study sites ranged in size between 400 and 1200 ha. Land cover was grouped into six general categories for the study: early-successional herbaceous, early-successional woody, forest, non-habitat, pasture/hay, and row crop (see below for details). Throughout the years of the study, land cover ranged from 9-28% woodlot cover, 38-73% agricultural cover, 4-15% grassland cover, 4-7% early-successional woody cover, 3-23% pasture/hay, and 4-7% other land uses (Table 1.2). Land use changed over time throughout the sites, primarily from expiration of CRP contracts, conversion of pasture/hay fields to row crop fields, conversion of row crop fields to pasture/hay fields, and implementation of woodlot edge feathering. Abundance estimates determined from whistle count surveys varied widely by site (see Chapter 3).

Early-successional herbaceous vegetation primarily consisted of fields enrolled in the CRP and abandoned fields. CRP fields were predominately enrolled in the CP1, CP2, CP4D, CP8A, CP33, or CP42 practices. CP1 is comprised of cool-season introduced grasses and legumes, and fields enrolled in this practice on the sites were primarily composed of fescue (*Festuca* spp.), goldenrod (*Solidago* spp.), and Queen Anne's lace (*Daucus carota*). CP2 fields are composed of warm-season native grasses. CP2 practices on the sites were predominately Indian grass (*Sorghastrum nutans*), big bluestem (*Andropogon gerardii*), switch grass (*Panicum virgatum*), and little bluestem (*Schizachyrium scoparium*). CP4D provides permanent wildlife habitat and had similar plant composition to fields enrolled in CP2. CP8A provides grassy waterways with cool season grasses. CP33 provides habitat buffers for upland birds and is specifically designed to meet NBCI's conservation goals. These buffers are comprised of native warm season grasses, legumes, and shrubs (Burger et al. 2006). CP42 provides pollinator habitat, and fields enrolled in this program are required to have at least 9 species of pollinator friendly wildflowers, legumes, and/or shrubs. Common species in CP42 fields on the sites included bee balm (*Monarda* spp.), purple coneflower (*Echinacea purpurea*), black eyed Susan (*Rudbeckia hirta*), and daisies (*Leucanthemum vulgare*). Idle fields were typically comprised of a larger woody component than CRP fields because farmers enrolled in CRP are required to remove woody components as part of their mid-contract management. Idle fields typically contained blackberry (*Rubus fruticosus*), black raspberry (*R. occidentalis*), trumpet creeper (*Campsis radicans*), poison ivy (*Toxicodendron radicans*), and Japanese honeysuckle (*Lonicera japonica*).

Early-successional woody vegetation included fencerows, ditches, forested patches < 50 m wide, and old fields with a large woody component. Common species included blackberry, black raspberry, poison ivy, trumpet creeper, multiflora rose (*Rose multiflora*), and Japanese honeysuckle. Amur honeysuckle (*Lonicera maackii*) had a small presence in all habitat types throughout the study sites.

Forest composition varied by management history and elevation. Forests in low elevations were primarily composed of black walnut (*Juglans nigra*), green ash (*Fraxinus pennsylvanica*), and American elm (*Ulmus americana*). Dry, upland ridges were primarily composed of white oak (*Quercus alba*), red oak (*Quercus rubra*), shagbark hickory (*Carya ovata*), and pignut hickory (*Carya ovata*). Bottomland forests were dominated by pin oak (*Quercus palustris*). Most forests had closed canopies with sparse understory vegetation during the winter. Forest understories and edges contained blackberry, black raspberry, and multiflora rose.

Pasture and hay fields included grass species such as fescue, orchard grass (*Dactylis glomerata*), timothy (*Phleum pratense*), and alfalfa (*Medicago sativa*). Pastures primarily held cattle but occasionally included horses. Row crop fields were primarily soybeans (*Glycine max*) and corn (*Zea mays*). A small number of fields grew winter wheat (*Triticum aestivum*) and tobacco (*Nicotiana tabacum*). Non-habitat included bodies of water, streams, roads, parking lots, buildings, and mowed yards.

THESIS CONTENT

The broad objective of my study was to contribute to understanding how agricultural land use conversion and subsequent fragmentation affects wildlife. I sought to evaluate wildlife responses to management practices on these landscapes. I used the northern bobwhite as a model organism due to its close association with farmland habitat, low seasonal movement rates, and significant declines throughout its range. I sought to determine how landscape factors affect movements, survival, and abundance of bobwhites at both the population and individual level throughout the annual life cycle. In Chapter 2 I take an individual perspective and investigate the effects of breeding season habitat on fall dispersal and subsequent nonbreeding season survival. In Chapter 3 I take a population perspective and investigate rate of decline in relation to habitat changes. The final chapter investigates management implications from my study. Chapters 2 and 3 are prepared as stand-alone manuscripts formatted following the Journal of Wildlife Management guidelines.

Table 1.1. County habitat composition and average farm size in Highland and Brown Counties, Ohio in 2012.

	Highland	Brown
Percent Cropland	74.7	67.5
Percent Pasture	9.9	10.8
Percent Woodland	10.5	15.7
Percent Other	4.9	6.1
Average farm size	187 acres	150 acres

Table 1.2. Landcover composition and abundance of northern bobwhites on four study sites in Brown and Highland Counties, Ohio in 2008 and 2014.

Site	Year	ESH^a	ESW^b	Forest	Non-habitat	Pasture/ Hay	Row Crop
Fee	2008	10.7%	4.2%	9.1%	4.0%	3.2%	68.9%
	2014	3.7%	3.5%	8.9%	4.0%	7.1%	72.7%
Peach	2010	15.3%	5.0%	28.4%	4.0%	7.1%	40.3%
	2014	7.9%	5.3%	28.3%	4.2%	16.7%	37.6%
Thurner	2008	10.3%	6.7%	16.5%	7.0%	5.4%	54.2%
	2014	10.2%	6.8%	15.9%	7.2%	4.9%	55.0%
Wildcat	2008	17.6%	3.8%	10.4%	4.1%	22.7%	41.4%
	2014	11.2%	3.8%	10.4%	4.1%	18.6%	52.0%

^aEarly-successional herbaceous

^bEarly-successional woody

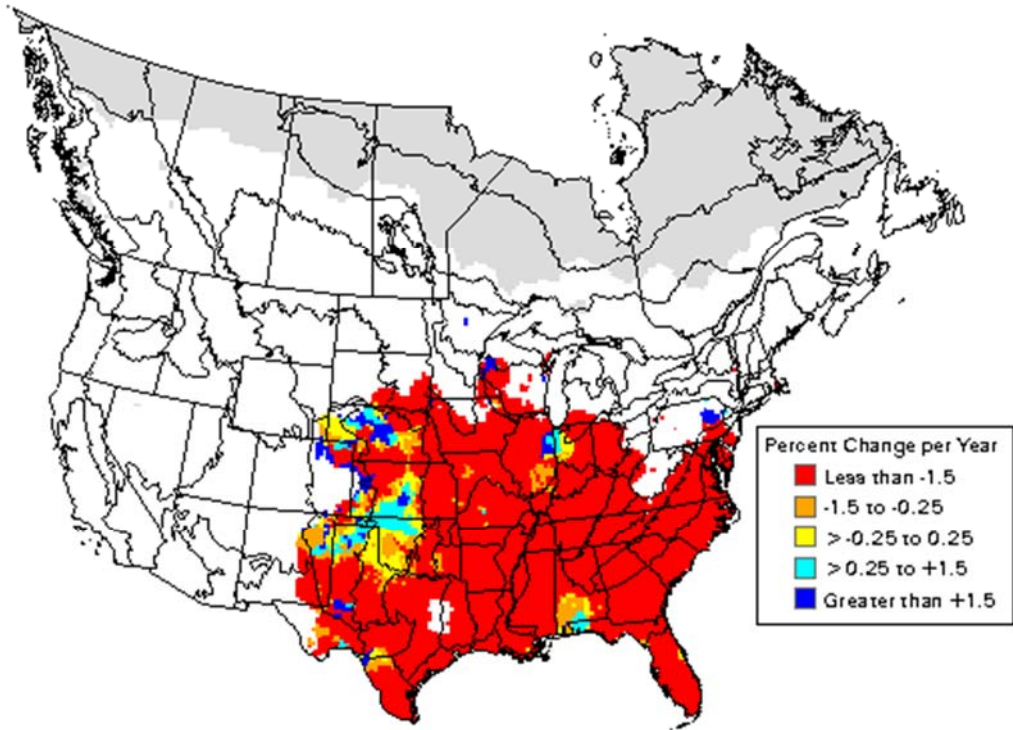


Figure 1.1. Population trend map of the northern bobwhite (*Colinus virginianus*) detected by the Breeding Bird Survey 1966-2013. (<http://www.mbr-pwrc.usgs.gov/bbs/tr2013/tr02890.htm>)

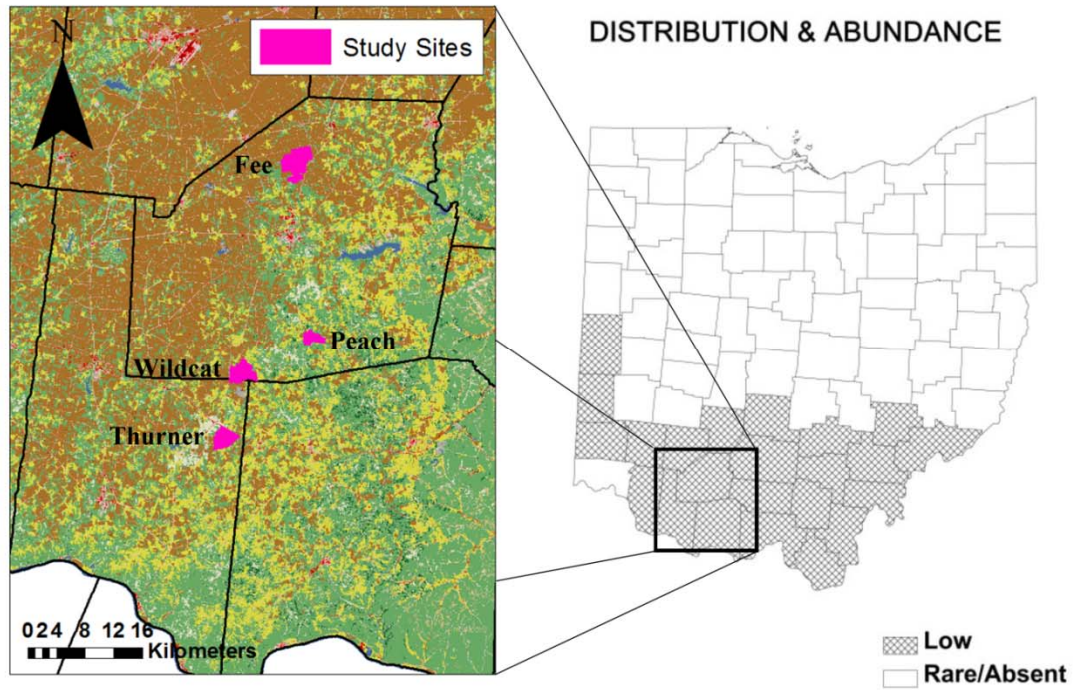


Figure 1.2. Locations of four study sites used to investigate northern bobwhite population demographics and movement in Southwest Ohio 2009-2015 overlaid on a National Land Cover Database image. The inset image shows counties in Ohio where bobwhites are present at low densities (Ohio Division of Wildlife 2014).

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CHAPTER 2: FALL DISPERSAL OF NORTHERN BOBWHITES IN FRAGMENTED AGRICULTURAL LANDSCAPES

ABSTRACT

Anthropogenic land use conversion of grasslands to farms removes wildlife habitat, induces fragmentation, and reduces structural and functional connectivity of the landscape impeding animal movements. Dispersal is a fundamental process at both the individual and population level and is impeded by fragmentation. The northern bobwhite (*Colinus virginianus*) is associated with farmland habitat in large portions of its range, is relatively immobile, and is known to have small seasonal shifts in home range. Bobwhites are declining throughout their range, and their limited dispersal ability may affect survival. I studied bobwhites on private agricultural lands in Ohio from 2009-2015 year-round using radio telemetry. 621 individuals were radio-marked, but only 26 survived from the breeding season to the subsequent non-breeding season with enough locations to build seasonal home ranges due to a combination of extremely high mortality rates, transmitter failure, a lapse in the study in the winter of 2011-2012, and low tracking effort in the 2012-2013 breeding season. The mean shift from the center of the breeding season home range to the center of the non-breeding season home range was 515 ± 139 (SE) m. Average overlap area between seasonal home ranges was 5.0 ± 1.1 (SE) ha. The best model predicting distance moved was summer home range percent early-successional woody vegetation, forest edge-density, early-successional woody mean

nearest neighbor, and early-successional herbaceous interspersed juxtaposition index. I ran 2 models to examine the carry over effects of summer home range habitat composition on subsequent winter survival: one using birds that had enough locations to build winter home range to explore the effects of fall dispersal distance and one with birds without enough locations in fall to build a home range to increase my sample size to explore the effects of home range habitat composition on subsequent winter survival. The best model from the first analysis was percent early-successional herbaceous but was within 2 ΔAIC_c points of the null model. I found no direct link between survival and distance moved between seasons. The second analysis found greater interspersed of early successional woody habitat in the summer home range was correlated to greater survival in the subsequent non-breeding season. My results suggest managers should place winter habitat within a half kilometer of nesting and brood rearing habitat in order for bobwhites to reach preferred winter cover. Additionally, lack of interspersed woody cover in the breeding season may affect survival in the subsequent non-breeding season, further suggesting the importance of providing early-successional woody cover near nesting and brood rearing habitat.

INTRODUCTION

Anthropogenic land use changes are a major threat to biodiversity worldwide. Globally, 21.8% of land area has been converted to human dominated uses (Hoekstra et al. 2005). Conversion of native land cover to agriculture has been historically extensive and continues today as one of the primary threats to biodiversity (Matson et al. 1997,

Rashford et al. 2010, Rashford et al. 2012). Cultivated land worldwide has increased by 466% from 1700 to 1980 (Matson et al. 1997).

Habitat loss and fragmentation necessarily results in changes in habitat configuration and connectivity (Baguette 2012, Chaine and Clobert 2012).

Fragmentation is the subdivision of a continuous habitat patch into smaller pieces which involves the loss of original habitat, reduction in patch size, and increased isolation of patches (Andren 1994, Marzluff and Ewing 2001). Fragmentation reduces structural connectivity, or the physical arrangement of habitat, by creating areas of cleared land (gaps) between habitat patches (Robertson and Radford 2009). Reduced structural connectivity can have impacts on species abundance and distribution, and decrease functional connectivity, which is how connected pieces of the landscape are to organisms (Rodriguez et al. 2001 Gobeil and Villard 2002, Vos et al. 2002, Robertson and Radford 2009).

Interseasonal movement, or movement to new habitat between seasons (hereafter dispersal), has been documented to decrease with increasing fragmentation (Vos et al. 2002, Schtickzelle et al. 2006, Alcock 2009, Baguette et al. 2012, Bonte et al. 2012, Chaine and Clobert 2012, Fedy et al. 2012). Dispersal is a risky process for animals due to increased predation, energy use, chances of getting lost, aggression, and stress before, during, and after the dispersal event (Yoder et al. 2004, Schtickzelle et al. 2006, Bonte et al. 2012). For example, predation risk increases with dispersal due to increased movement which increases visibility to predators, increases consumption of energy which reduces the ability to detect and avoid predators, and through unfamiliarity with new

habitats causing animals to potentially move to dangerous areas (Yoder et al. 2004, Bonte et al. 2012). Fragmentation increases the risks of dispersal by increasing the distance necessary to travel to complete dispersal, often through open habitat (Whittingham and Evans 2004, Schtickzelle et al. 2006, Bonte et al. 2012, Chaine and Clobert 2012).

Dispersal is an important process both at the individual and population level (Brooker et al. 1999, Matthysen 1999, Schtickzelle et al. 2006, Fies et al. 2002, Barton et al. 2009, Van Oort et al. 2010, Bonte et al. 2012, Chaine and Clobert 2012). Individual animals can potentially find better (i.e. higher quality) habitat by relocating and avoid resource competition in high density areas (Bonte et al. 2012, Chaine and Clobert 2012, Fedy et al. 2012). Animals that require different types of habitat throughout the annual cycle for nesting, brood-rearing, and winter often must relocate to find suitable habitat (Fedy et al. 2012).

The northern bobwhite (*Colinus virginianus*) lends itself to understanding how fragmentation affects dispersal because it is a relatively immobile species (Stoddard 1932, Murphy and Baskett 1952, Lewis 1954, Agee 1957, Rosene 1969, Fies et al. 2002, Townsend et al. 2003, Liberati 2013) that occupies highly fragmented habitats throughout large portions of its range due to agricultural practices (Fies et al. 2002, Flock 2006, Janke 2011, Lohr et al. 2011, Wiley 2012, Bowling et al. 2014). Bobwhites are declining range wide at an annual rate of 4.08% (Sauer et al. 2014), and have as high as an 80% annual mortality rate in the northern portion of their range (Brennan 1991, Gutery 1997) making them a species of concern. I studied bobwhites in Ohio where populations are declining at a higher rate than the national average (6.88%, Sauer et al. 2014). Winter

survival is limiting to this population (Gates et al. 2012), so fall dispersal may be an especially important period. Birds could find more suitable wintering habitat through dispersal to improve survival, but if the landscape impedes dispersal, birds may get isolated in bad habitat patches where winter survival will be lower (Bonte et al. 2012, Chaine and Clobert 2012). Several studies have examined spring dispersal (Fies et al. 2002, Townsend et al. 2003, Cook 2004, Liberati 2013), but fall dispersal remains poorly understood.

The decision to disperse is complex and based on many factors including habitat composition, habitat configuration, predation risk, and competition (Cote et al. 2010). Early-successional woody vegetation has been found to be both preferred and limiting to winter survival (Schroeder 1985, Williams et al. 2000, Flock 2006, Janke 2011). Row crop fields are useful for brood rearing and feeding habitat and comprise a large portion of usable space in summer for bobwhites (Schroeder 1985, Liberati 2013). However, when row crop fields are harvested in the fall they become a matrix, or area of inhospitable habitat (Schroeder 1985). Percent habitat composition may affect dispersal of bobwhites in fall depending on suitability and permeability of the habitat types. Additionally, configuration may have an impact on dispersal ability. Distance between patches and interspersed nature of the landscape and individual habitat types may affect the ability to disperse between patches.

Many studies have shown that conditions in one season can affect reproduction, behavior, and survival in the following season (Saether et al. 2000, Sillett et al. 2000, Gill et al. 2001, Norris 2005, Webster and Marra 2005, Marra 2007). In an individual level

interaction, events or conditions, such as quality of habitat, in one season produce non-lethal, residual effects that carry over to influence individual fitness in the next season (Norris 2005, Marra 2007). Summer home range composition and configuration of bobwhites could potentially cause survival consequences even after dispersal due to a carry-over effect.

In a year-round study, I attempted to identify factors in summer home range composition and configuration that affect subsequent fall dispersal and survival of bobwhites. Winter survival of bobwhites is limiting in Ohio, and understanding how summer habitat affects dispersal to winter home ranges and survival can aid in bobwhite management. I predicted that summer home range composition would influence interseasonal movement and affect survival throughout the winter. I specifically predicted 1) bobwhites whose summer home ranges have larger portions of preferred winter habitat will disperse shorter distances, 2) bobwhites inhabiting more connected landscapes would be able to move more, and 3) bobwhites with better summer home range composition would have greater survival into the winter. To test these predictions, I used radio-telemetry data of bobwhites to monitor survival and build seasonal home ranges, and then classified their habitats.

STUDY AREA

I conducted this research on four private land sites in Brown and Highland Counties in southwest Ohio, USA (39°04'59", 83°39'10") between April 2010 and March 2015. Brown and Highland Counties are in the glaciated till plains physiographic

region (Ohio Division of Geologic Survey 1998). Study sites were chosen in the core of the bobwhite range in Ohio (Spinola and Gates 2008). Study sites ranged from 400 ha to 1,200 ha and ranged from 8-28% woodlot cover, 38-73% agricultural cover, and 4-11% grassland cover. (For a more detailed site description see Chapter 1, Janke 2011, Gates et al. 2012, Wiley 2012, or Liberati 2013.)

METHODS

Field Methods

Bobwhites were monitored year-round and separated into two distinct seasons: breeding (1 Apr-30 Sep 2010 and 1 Apr-30 Sep 2012-2014) and nonbreeding (1 Oct-31 Mar 2009-2011 and 1 Oct-31 Mar 2012-2015). Whistle count surveys, covey call surveys, pointing dogs, and snow tracking were used to locate bobwhites. Bobwhites were captured year-round using funnel traps baited with corn and lined with burlap for protection (Stoddard 1932). Bobwhites were also captured through targeted mist netting using 61 mm mesh, 4 shelf AFO mist nets 2.6x12 m in size with 3.05 conduit poles 1.9 cm wide (AFO Mist Nets, Manomet, MA, USA). Audio lures of female bobwhite calls were used during the breeding season, and previously radio-marked birds were used to capture coveys during the nonbreeding season. Birds were aged and sexed after capture, and most birds were fitted with a 6.6-g necklace style radio-transmitter (Advanced Telemetry Systems, Isanti, MN) weighing ≥ 165 g (transmitters were $\leq 4\%$ body mass) and an individually numbered aluminum leg band. Marked birds were released at capture sites. Capture, handling, and marking protocols were reviewed and approved by the

Animal Care and Use Committee at The Ohio State University (protocol number 2007A0228).

Birds were located year-round 3-7 times per week from 1 October 2009 - 30 September 2011 and 1 May 2012 - 30 March 2015 using handheld very high frequency (VHF) Yagi antennas until mortality, loss due to dispersal, or transmitter failure. Homing and triangulation were used from short distances (20-50 m), and diurnal tracking times were varied to observe the full range of daily activity patterns (White and Garrott 1990, Burger et al. 1995, Townsend et al. 2003, Terhune et al. 2006, Lohr et al. 2011). The cover type that individuals or coveys were located in was recorded, and locations were marked on a handheld Global Positioning System (GPS; Garmin GPS Map 76, Garmin International, Inc., Olathe, KS).

Analysis Methods

I manually digitized land cover on each study site along with areas over the boundaries for dispersed birds for each year in ArcGIS (version 10.1, ESRI, Redlands, CA) with high spatial resolution ortho-photographs (0.305 m; Ohio Statewide Imagery Program 2008). I used 6 habitat categories for analyses (early-successional herbaceous, early-successional woody, forest, non-habitat, pasture and hay fields, and row crop). Early-successional herbaceous cover was primarily fields enrolled in the Conservation Reserve Program (CRP) or old fields with predominately herbaceous cover. Early-successional woody vegetation was mainly fencerows, ditches, and portions of CRP or old fields with predominately woody vegetation. Woodlots were classified as forest when patches were >50 m wide and early-successional woody if they were <50 m wide. Pasture

and hay fields were herbaceous fields that had grazing or mowing. Non-habitat included residential areas, commercial areas, roads, water bodies, cemeteries, and mowed areas around roads. Field evaluations covering the entire area of each study site were completed annually to ground-truth digitized land cover maps, and any changes were reclassified in a new layer in ArcGIS. Changes typically occurred from crop rotation, enrollment into the CRP, change in CRP practice, conversion of CRP land back into production, or conversion between pasture/hay fields and row crop fields.

I used individuals as the sampling unit in my analyses with a random effect for covey membership for birds that joined the same covey in fall. I constructed home ranges using the Local Convex Hull (LoCoH) method in the ADEHABITATHR package in Program R (R Version 3.2.0, www.r-project.org, accessed 05 June 2015) for each individual in each season (Getz and Wilmers 2004). LoCoH is known to outperform other methods in fragmented habitats (Wiley 2013). I only analyzed individuals with ≥ 15 locations in each season and censored any birds whose home ranges did not stabilize in size with < 20 locations (Taylor et al. 1999, Singh et al. 2011).

I calculated the Euclidean distances between the centroids of summer and fall home ranges for individual bobwhites. I overlaid and clipped cover maps with 95% summer home range boundaries to quantify home range composition of the six habitat types. I used Patch Grid for ArcGIS 10.1 on a rasterized image of the clipped cover maps of summer home ranges to determine edge density (ED), mean nearest neighbor (MNN), and interspersed juxtaposition index (IJI) for each cover class and at the landscape scale (Taylor et al. 1999, Bowling et al. 2014). Landscape scale analyses performed

calculations across classes for the entire home range whereas class analyses differentiated between habitat classes in the home range layer. I log transformed distances for analysis. I used mixed effects models using the lme4 package in Program R to account for the random effects of site, sex, location, and covey membership. Location was used to incorporate spatial overlap between birds whose 95% home ranges overlapped by $\geq 25\%$ and was nested within sites. I used Akaike's Information Criterion (AIC) for model selection.

The second candidate model set investigated seasonal interactions between summer home range habitat composition and configuration and subsequent winter survival. The candidate model set included a null model with just a survival estimate and models with combinations of summer home range habitat variables individually selected by univariate logistic regression by a criteria of $p < 0.25$ (Taylor et al. 1999). I investigated if there was a correlation between dispersal distance and home range overlap on subsequent winter survival, but the variables were not significantly related ($p > 0.25$), so I did not use them for my analyses. I used the RMARK package in Program R for survival analyses using the known fates model (White and Burnham 1999). All models within 2 ΔAIC_c points of the top model were model averaged in deriving parameter estimates (Burnham and Anderson 2002). Due to the low number of bobwhites with enough locations for summer and winter home ranges, I ran the analysis using all bobwhites with enough locations for summer home ranges even if they did not have enough locations for winter home ranges and looked for seasonal interactions between summer home range

habitat composition and configuration with subsequent winter survival excluding variables that required a fall home range to compute.

RESULTS

Radio-transmitters were attached to 621 of 790 bobwhites that were captured during 2009 to 2015. Birds were not tracked in fall 2011, so these birds were excluded from analyses. With such high mortality rates, 46 of the radio-marked bobwhites survived between the breeding season and subsequent non-breeding seasons. Twenty-six of those 46 bobwhites had sufficient numbers of radio-locations to construct home ranges (7 F, 19 M). There were 3,227 locations recorded for these bobwhites, excluding mortality points that were censored from analyses due to the possibility of predators moving birds from the kill site.

Mean movement between the center of the summer home range to the center of the winter home range was 515 ± 139 (SE) m (range 64-1365 m, $n = 26$ bobwhites). Mean overlap of the nonbreeding season home range with the breeding season home range was 34% or 5.0 ± 1.1 (SE) ha (range 0-20 ha, $n = 26$ bobwhites). Mean home range size during the breeding season was 43.1 ± 6.39 (SE) ha (range 5.4-144.4 ha, $n = 26$ bobwhites) compared to 18.3 ± 2.94 (SE) ha (range 2.2-63.4 ha, $n = 26$ bobwhites) during the nonbreeding season. Breeding season home ranges were comprised of row crop (50%), early-successional herbaceous (25%), and pasture/hay (10%) habitats (Table 2.1). Nonbreeding season home ranges were dominated by row crop (39%), early-successional herbaceous (30%), and pasture/hay (10%) habitats.

The model with the most support ($w_i = 0.45$) of distances moved between summer and winter home ranges included summer home range percent early-successional woody cover ($\beta = -3.44 \pm 1.75$ (SE)), summer home range forest ED ($\beta = -0.017 \pm 0.005$ (SE)), MNN of early-successional woody ($\beta = 0.0007 \pm 0.0002$ (SE)), and summer IJI of early-successional herbaceous habitat ($\beta = 0.005 \pm 0.002$ (SE)) with a random location effect (Table 2.2, Table 2.3, Figure 2.1). A likelihood ratio test indicated the full model was an improvement on the null model ($\chi^2 = 19.71$, $P = 0.001$).

I estimated survival of 32 bobwhites (11 F, 21 M). Of the 26 birds used in the distance analyses, 4 bobwhites (15.4%) were censored by completely removing from the data set due to transmitter failure and 1 (3.8%) was censored by completely removing from the data set due to researcher induced mortality. Ten bobwhites had enough locations to build summer home ranges but insufficient numbers to build fall home ranges so were only included in the second round of survival analyses. Winter survival pooled across years for the first analysis was 0.28 ± 0.090 (SE; 95% CI = 0.14-0.48, $n = 21$). Winter survival pooled across years in the second analysis with the larger sample size was 0.211 ± 0.066 (SE; 95% CI = 0.11, 0.37, $n = 31$). For the full 32 birds, avian ($n = 7$; 21.9%), mammalian ($n = 9$; 28.1%), and undetermined ($n = 5$; 15.6%) predators were identified as probable mortality agents, and 3 mortalities had an unknown cause (9.4%). Of the probable mammalian mortalities, 1 was likely caused by a domestic cat because it was found in some bushes next to a barn with a feral cat colony. Mortality was also caused by harvest ($n = 1$, 3.1%). Seven bobwhites (21.9%) survived across the entire interval.

Sex, site, covey, and location were not associated with survival ($p > 0.25$). The model that best explained bobwhite winter survival ($w_i = 0.22$) using the birds that had enough locations for fall home ranges included percent early-successional herbaceous habitat (Table 2.4, Table 2.5). Increased percent early-successional herbaceous habitat in the breeding season home range tended towards a decline in bobwhite survival in the nonbreeding season, although the 95% confidence interval overlapped zero, and the likelihood ratio test indicated the top model was not significantly better than the null ($\chi^2 = 3.440$, $P = 0.064$). A power analysis indicated I would have needed 89 bobwhites to detect a significant effect on survival of percentage early-successional herbaceous habitat with 80% power. The second ranked model ($w_i = 0.11$) included percent row crop. Greater percentage of row crop in the summer home range tended towards higher survival in the nonbreeding season, although the 95% confidence interval overlapped zero.

The model that best explained bobwhite winter survival ($w_i = 0.31$) using all birds that survived from the breeding season to the nonbreeding season with enough locations for at least the summer home range included interspersed juxtaposition index of early successional woody habitat (Table 2.6, Table 2.7). Increased interspersed juxtaposition index of early successional habitat in the summer home range predicted increased survival throughout the non-breeding season. A likelihood ratio test indicated the top model was significantly better than the null ($\chi^2 = 4.064$, $P = 0.044$).

DISCUSSION

Despite being one of the most studied species in North America (Scott 1985), there is a large information gap on fall dispersal and covey formation in the northern bobwhite literature. The process of covey formation is important because the habitat a bird winters in can affect winter survival (Williams et al. 2000, Cook 2004, Flock 2006, Janke 2011, Lohr et al. 2011). Bobwhites are declining throughout their range, and in Ohio the rate of decline is higher than the national average (Spinola and Gates 2008). Winter survival has been shown to be limiting to the Ohio population (Gates et al. 2012). Understanding how bobwhites disperse to winter habitat through agricultural landscapes may help improve winter survival by aiding managers in understanding where to place winter habitat in relation to nesting and brood rearing habitat in order for bobwhites to be able to find and utilize it (Fedy et al. 2012).

Several studies in the 1950s (Murphey and Baskett 1952, Lewis 1954, Agee 1957) attempted to describe the distances bobwhites move during covey formation, but no studies have been published using radio telemetry data. I observed an average distance of 515 m between summer and fall home range centroids, which averaged a 5.0 ha (34%) overlap between seasonal home ranges. Bobwhites moved a relatively short distance during covey formation in comparison to the average farm size in Highland and Brown Counties (187 and 150 ha, respectively) suggesting the importance of managing for bobwhites year-round on each farm, though bobwhites typically did reside on multiple farms during the summer. Some studies documenting fall dispersal reported large movements of bobwhites in fall (Schorger 1944). One study looking at fall dispersal

reported a hunter returned a band 41 km from the summer banding site in Oklahoma (Duck 1943), but most studies that have assessed covey formation have observed average fall dispersal distances of less than 1.6 km (Stoddard 1932, Murphey and Baskett 1952, Lewis 1954, Agee 1957). Agee (1957) found an average distance of 352 meters from the last summer trap site to the first covey trap site in Missouri. Murphey and Baskett (1952) found that 14/18 quail recaptured in winter had moved less than 1.6 km from their last summer trap location. Telemetry studies have shown bobwhites to have low daily movement rates after covey formation, especially as temperature decreases (Williams et al. 2000, Janke 2011, Lohr et al. 2011).

The best model for predicting distance moved between seasons included summer home range percent early-successional woody cover, forest edge density, mean nearest neighbor of early-successional woody cover, and interspersion of early-successional herbaceous cover. Forest edge density was negatively correlated with interseasonal movements. Forest edge can provide winter cover for bobwhites in areas with little usable space like in highly agricultural landscapes (Guthery 1997). Lohr et al. (2011) found coveys in New Jersey used forest edges when shrub-scrub habitat was lacking. Janke (2011) found coveys in Ohio selected for forest in their home ranges, perhaps because of the cover the edge provides. Edge density at the landscape scale was not significant in my models. Not all edge is useful to coveys which could explain my results. Hanson and Miller (1961) found number of bobwhites and landscape edge density were not correlated. Coveys were selecting for certain edge types (pasture-row crop edge) which comprised the minority of total edge.

Mean nearest neighbor values for early-successional woody habitat was positively correlated with distance moved. Bobwhites moved further between seasons when early-successional woody patches, which are preferred for winter cover, were further apart. Large movements are dangerous, especially in landscapes with a large portion of matrix habitat (Whittingham and Evans 2004, Chaine and Clobert 2012). My study sites ranged from 41-75% agricultural cover, habitat that becomes inhospitable once crops are harvested in the fall. Crossing this habitat makes bobwhites extremely vulnerable to predation (Yoder et al. 2004, Bonte et al. 2012). Williams et al. (2000) found decreased daily movement increased survival rates. Bobwhites that select patches with shorter mean distance to nearest neighbor patches of preferred winter cover would be able to spend less time exposed to predators in the matrix. Hanson and Miller (1961) found that bobwhites in Illinois needed winter protective cover to be planted less than 90-180 m apart for optimal use. My results suggest that bobwhites will move further from their summer home ranges if winter cover patches are too far apart, perhaps to areas with closer patches that incur less cost to travel between. Errington (1933) believed that movement of quail from summer home ranges occurred to leave “food and cover impoverished” to areas better suited for winter survival.

The best model for predicting fall dispersal distance tended towards lower interseasonal movements with a higher percentage of early-successional woody cover. Early-successional woody cover has been shown to be preferred in winter and important for survival (Schroeder 1985, Williams et al. 2000, Flock 2006, Janke 2011). Birds whose summer home ranges included a larger percentage of early-successional woody cover

possibly had to move less to find suitable winter habitat. The top model also predicted higher interseasonal movements with a higher interspersion juxtaposition index of early-successional herbaceous habitat. I expected interspersion to rank in my top models because of its demonstrated importance in the literature. Baxter and Wolf (1972) and Vance (1976) both found higher bobwhite densities in areas with higher interspersion.

The top model for predicting winter survival from summer home range composition using the smaller sample size was percentage early-successional herbaceous cover, and the second best model was percent row crop, though neither was significantly better than the null. Percentage early-successional herbaceous cover in the summer home range tended towards decreased weekly survival in the winter. Early-successional habitat may decrease survival through carry over effects from stress or injury from increased numbers of predators in the breeding season (Gehring and Swihart 2002, Kuehl and Clark 2002, Whittingham and Evans 2004, Roth et al. 2008, Wilson et al. 2010). Cooper's hawks (*Accipiter cooperii*), for example, were shown to select for forest and grassland (Roth et al. 2008) and are one of the larger predators on bobwhites (Stoddard 1932, Rosene 1969). Likewise, some mammalian predators, such as red fox (*Vulpes vulpes*), have been shown to select for grasslands. Additionally, early-successional herbaceous habitat may decrease survival due to its lack of woody cover as woody cover has been shown to lead to increased survival by protection from predators (Schroeder 1985, Williams et al. 2000, Flock 2006, Janke 2011).

The null model ranked high in my candidate model set for predicting winter survival from summer home range habitat composition. I had a relatively small sample

size and automatically censored birds who survived less than 15 tracking days into the nonbreeding season due to the requirement of enough points to build a nonbreeding season home range to be included in analyses. With such high mortality rates, this severely limited my sample size and may have reduced my ability to detect an effect. My estimated power was only .25 using the reduced set of birds. Additionally, with such high overall mortality, survival may be more of a stochastic event that is hard to model (DeAngelis et al. 1993).

I conducted a second set of survival analysis with all birds with enough locations to build a summer home range which increased my sample size by 11 birds. The added birds died early in the non-breeding season and so may represent a more accurate portrait of how summer home range habitat composition affects survival. Greater interspersion of early-successional woody habitat in the summer home range predicted greater survival in winter. Woody cover has been shown to lead to increased survival by protection from predators within a season (Schroeder 1985, Williams et al. 2000, Flock 2006, Janke 2011). Stress from injury from reduced woody cover could have led to lower survival in the subsequent breeding season.

I was surprised to find no correlation between fall dispersal distance and fall survival. Dispersal is known to incur large costs to animals and increase risk of predation (Yoder et al. 2004, Bonte et al. 2012). Further, dispersal risk is known to increase in fragmented landscapes like my study sites (Schtickzelle et al. 2006, Baguette et al. 2012, Bonte et al. 2012, Chaine and Clobert 2012). Fragmentation can increase the distance necessary to travel to complete dispersal, which increases the time exposed to predators

(Schtickzelle et al. 2006, Bonte et al. 2012). By contrast, animals could potentially find higher quality habitat by relocating and increase their survival through the non-breeding season (Chaine and Clobert 2012). Townsend et al. (2003) found bobwhites that dispersed in the spring in Oklahoma had survival rates 1.5 times higher than bobwhites that did not disperse. However, distance dispersed had no effect on survival. They suggested dispersed bobwhites benefited from higher quality habitat, more frequent contact with potential mates, wider dispersion of genetic material, avoidance of population crashes, and avoidance of competition in crowded habitats. One possible explanation of why I found no correlation is because animals were killed during the dispersal process. I only included animals with ≥ 15 locations after October 1 when exploring the effect of dispersal distance on survival, so any animals that died during fall dispersal likely were not included in my analyses. Future research should attempt to get a sample size of 89 birds or more to fully explore the effects of summer home range composition, fall dispersal, and seasonal home range size contraction of subsequent winter survival.

My results suggest bobwhites will shift their home ranges more in fall if there is less suitable habitat in their summer home ranges. Though I did not find a correlation between survival through winter and the degree of home range shift, bobwhites shifted home ranges more in fall when summer habitat was of poorer quality for winter survival. In the highly agricultural landscapes of Ohio, bobwhites are most likely limited in their ability to disperse after crop harvest. Bobwhites may have some ability to find more suitable winter habitat after crop harvest but may also be limited since the average

dispersal distance was just over a half kilometer. Many studies have suggested the importance of having adequate habitat for nesting, brood rearing, and wintering in close proximity (Stoddard 1932, Hanson and Miller 1961, Schroeder 1986, Wiley 2012). Bobwhites in Ohio likely need winter cover close to nesting and brood rearing habitat due to limited dispersal ability once crops are plowed.

As land use conversion continues, it is important to understand how to manage wildlife in these anthropogenic, fragmented landscapes. Bobwhites are declining rapidly throughout their range despite a vast amount of research and management recommendations. Many studies have shown habitat that improves survival in winter (Williams et al. 2000, Cook 2004, Flock 2006, Janke 2011, Lohr et al. 2011), but no studies have looked at how summer home range habitat composition affects subsequent movements and survival into winter. Understanding fall dispersal and carry-over effects on survival may aid in improving winter survival. Additionally, this information can aid in improved management in highly fragmented agricultural landscapes as agriculture continues to intensify and remove wildlife habitat.

MANAGEMENT IMPLICATIONS

I demonstrated bobwhites may disperse to find more suitable habitat in fall but are probably limited in how far they can travel. I found that bobwhites move about a half kilometer between the centers of their summer home ranges to the centers of their winter home ranges in Ohio. Management efforts should place early-successional woody habitat within a half kilometer or less of nesting and brood rearing habitat due to limited

dispersal of bobwhites in fall. Techniques such as edge-feathering should be conducted near areas bobwhites inhabit in late summer such as CRP and row crop fields.

The CRP practice CP33, Habitat Buffers for Upland Birds, is one potential way to add nesting habitat directly adjacent to row crop fields used for brood rearing. CP33 is part of the Northern Bobwhite Conservation Initiative's (NBCI) species recovery plan. Farmers can enroll the first 9.4-36.6 m of their fields and plant them to native warm season grasses, legumes, and shrubs for a 10 year contract period. Farmers in return receive a \$10/acre/year sign up incentive, an annual rental payment, 50% cost-share for the cover establishment, a sign-up incentive payment of 40% of approved establishment costs, and 50% cost-share for mid-contract management (Burger et al. 2006). Farmers could then selectively cut less valuable tree species along nearby woodlot edges in order to provide quail with habitat needed for all phases of the annual cycle.

NBCI states that landowner incentives are among the most important facets of managing for northern bobwhites. The NBCI states it is necessary to provide landowners with proper financial incentives to implement habitat management. One way is to identify priority management areas and offer extra economic incentives for adopting practices in these areas and provide assistance on implementation (Terhune et al. 2011). My study sites were identified as a bobwhite management priority area by the Ohio Division of Wildlife, and landowners were asked to adopt CP33 buffers and woodlot edge enhancement. Landowners were given an increased sign-up incentive payment to encourage enrollment, and Pheasants Forever funded implementation of woodlot edge feathers. This incentive system may encourage more landowners to adopt conservation

strategies. Having willing farmers adopt multiple strategies to manage for the full annual cycle appears to be extremely important since the majority of bobwhites in Ohio did not shift seasonal home ranges by more than one or two farms.

Table 2.1. Mean breeding season home range composition of 6 habitat types for individual northern bobwhites (1 April-30 Sep 2010, 2012, 2013, and 2014, $n = 26$) in Brown and Highland County, Ohio.

Landscape metrics	Mean	SE	Minimum	Maximum
Percent of home range				
Early-successional herbaceous	25.1	3.8	1.9	63.2
Early-successional woody	4.1	0.5	1.1	11.0
Forest	5.2	1.1	0.0	17.6
Non-habitat	5.1	0.8	0.0	16.7
Pasture/hay	10.1	2.3	0.0	37.6
Row crop	50.2	3.8	20.5	83.0
Edge-density ^a				
Landscape	109.0	7.1	47.5	203.9
Early-successional herbaceous	33.9	4.2	6.6	87.5
Early-successional woody	33.2	3.5	7.4	85.6
Forest	11.0	2.3	0.0	51.4
Non-habitat	22.9	3.3	0.0	63.9
Pasture/hay	16.0	3.6	0.0	64.5
Row crop	55.3	4.9	20.0	121.9
Mean nearest neighbor ^b				
Landscape	217.5	27.0	43.3	658.9
Early-successional herbaceous	236.2	66.6	1.0	1467.5
Early-successional woody	198.4	41.6	1.0	961.4
Forest	726.5	200.5	1.0	3161.9
Non-habitat	301.8	78.7	1.0	1569.4
Pasture/hay	279.3	106.2	1.0	1908.4
Row crop	99.0	19.2	4.1	348.7
Interspersion juxtaposition index ^c				
Landscape	73.1	1.9	54.9	89.3
Early-successional herbaceous	65.2	4.8	1.0	98.2
Early-successional woody	66.2	3.6	20.1	87.8
Forest	31.1	4.9	0.0	94.6
Non-habitat	67.2	4.5	1.0	91.3
Pasture/hay	63.3	5.1	0.0	93.8
Row crop	64.1	3.8	10.8	94.4

^a Meters/hectares

^b The average distance in meters between individual classes and the mean of the class nearest neighbor distance at the landscape level

^c Approaches zero when the distribution of unique patch adjacencies becomes uneven and 100 when all patch types are equally adjacent

Table 2.2. Model set including corrected Akaike’s Information Criterion (AIC_c), difference in corrected Akaike’s Information Criterion (Δ AIC_c), model weight (w_i), and number of model parameters (K) for determining the average distance moved between summer (1 April-30 September 2010-2015) to winter home ranges (1 October- 31 March 2010-2015) of northern bobwhites in southwest Ohio.

Model ^a	K	AIC _c	Δ AIC _c	w_i
Percent ES Woody + ED Forest + MNN ES Woody + IJI ES Herbaceous + (1 Location)	7	12.49	0.00	0.45
Percent ES Woody + ED Forest + MNN ES Woody + (1 Location)	6	14.83	2.34	0.14
Percent ES Woody + ED Forest + MNN ES Woody + IJI ES Herbaceous + IJI ES Woody + (1 Location)	8	15.88	3.39	0.08
Percent ES Woody + (1 Location)	4	16.43	3.94	0.06
Percent ES Woody + IJI ES Herbaceous + S_ESWood * IJI ES Herbaceous + (1 Location)	6	17.01	4.52	0.05
Percent ES Woody + ED Forest + (1 Location)	5	17.18	4.69	0.04
Percent ES Woody + MNN ES Woody + (1 Location)	5	17.66	5.17	0.03
Percent ES Woody + IJI ES Herbaceous + (1 Location)	5	18.10	5.61	0.03
Percent ES Herbaceous + (1 Location)	4	18.31	5.82	0.02
Percent ES Woody + MNN ES Woody + IJI ES Woody + (1 Location)	6	18.54	6.05	0.02
(1 Location)	4	18.65	6.16	0.02
MNN ES Woody + (1 Location)	4	18.87	6.39	0.02
IJI ES Herbaceous + (1 Location)	3	19.07	6.58	0.02
(1 Location)	4	20.67	8.18	0.01
ED Forest + (1 Location)	4	20.90	8.41	0.01
IJI ES Woody + (1 Location)	6	20.98	8.49	0.01
Percent ES Woody + ED Forest + MNN ES Woody + IJI ES Herbaceous + IJI ES Woody + Percent ES Woody * IJI ES Herbaceous + MNN ES Woody * IJI ES Herbaceous + (1 Location)	10	24.48	11.99	0.00

^a Early-successional (ES), edge density (ED) in meters/hectare, mean nearest neighbor (MNN; the average distance in meters between patches of the same habitat type), interspersed juxtaposition index (IJI; approaches zero when the distribution of unique patch adjacencies becomes uneven and 100 when all patch types are equally adjacent)

Table 2.3. Best mixed effects model for log of distance moved from summer (1 April-30 September 2010-2015) to winter home ranges (1 October- 31 March 2010-2015) of northern bobwhites in southwest Ohio.

	Estimate	SE	χ^2	P-value
% Early-successional woody	-3.437	1.753	3.587	0.058
Forest edge-density	-0.017	0.005	11.093	0.001
Mean nearest neighbor early-successional woody	0.0007	0.0002	8.051	0.005
Interspersion juxtaposition index early-successional herbaceous	0.005	0.002	6.143	0.013

Table 2.4. Model set including corrected Akaike’s Information Criterion (AIC_c), difference in corrected Akaike’s Information Criterion (ΔAIC_c), model weight (w_i), and number of model parameters (K) for determining seasonal interactions of summer (1 April-30 September 2010-2015) home ranges on winter survival (1 October- 31 March 2010-2015) of northern bobwhites in southwest Ohio using birds with enough locations for summer and fall home ranges ($n = 21$).

Model ^a	AIC_c	ΔAIC_c	w_i	k
Percent ES herbaceous	129.15	0	0.22	2
Percent row crop	130.55	1.4	0.11	2
Null	130.57	1.42	0.11	1
Percent ES herbaceous + ED row crop	131.13	1.98	0.08	3
Percent row crop + IJI ES woody	131.58	2.44	0.07	3
ED row crop	131.67	2.52	0.06	2
IJI ES woody	131.72	2.57	0.06	2
ED ES herbaceous	131.72	2.57	0.06	2
Percent non-habitat	131.96	2.81	0.05	2
Percent ES herbaceous + ED ES woody + IJI ES woody	132.53	3.38	0.04	4
ED ES woody	132.53	3.38	0.04	2
ED Landscape	132.57	3.42	0.04	2
ED non-habitat	132.59	3.44	0.04	2

^a Early-successional (ES), edge density (ED) in meters/hectare, interspersed juxtaposition index (IJI; approaches zero when the distribution of unique patch adjacencies becomes uneven and 100 when all patch types are equally adjacent)

Table 2.5. Summary of model averaged estimates for parameters in models within 2.00 AIC points of the top model for determining nonbreeding season survival from summer home range habitat composition of northern bobwhites on private lands in southwest Ohio, USA (2010-2015) using birds with enough locations for summer and fall home ranges ($n = 21$).

Model	Estimate	SE	95% CI
Percent early-successional herbaceous	-2.50	1.35	-5.32, 0.32
Percent row crop	1.97	1.40	-0.77, 4.72
Row crop edge density	0.0027	0.01	-0.02, 0.02

Table 2.6. Model set including corrected Akaike’s Information Criterion (AIC_c), difference in corrected Akaike’s Information Criterion (ΔAIC_c), model weight (w_i), and number of model parameters (K) for determining seasonal interactions of summer (1 April-30 September 2010-2015) home ranges on winter survival (1 October- 31 March 2010-2015) of northern bobwhites in southwest Ohio using birds with enough locations to build summer home ranges that survived to the subsequent nonbreeding season ($n = 32$).

Model ^a	AIC_c	ΔAIC_c	w_i	k
IJI ES Woody	190.72	0	0.31	2
% Row Crop + IJI ES Woody	192.65	1.92	0.12	3
Null	192.77	2.04	0.12	1
% Non-habitat	193.35	2.62	0.08	2
ED Non-habitat	193.78	3.05	0.07	2
ED Row Crop	194.65	3.92	0.04	2
% ES Herbaceous + ED ES Woody + IJI ES Woody	194.71	3.98	0.04	4
% Row Crop	194.77	4.04	0.04	2
ED ES Woody	194.77	4.05	0.04	2
Landscape ED	194.79	4.06	0.04	2
% ES Herbaceous	194.79	4.06	0.04	2
ED ES Herbaceous	194.79	4.06	0.040	2
% ES Herbaceous + ED Row Crop	196.65	5.92	0.02	3

^a Early-successional (ES), edge density (ED) in meters/hectare, interspersed juxtaposition index (IJI; approaches zero when the distribution of unique patch adjacencies becomes uneven and 100 when all patch types are equally adjacent)

Table 2.7. Summary of model averaged estimates for parameters in models within 2.00 AIC points of the top model for determining nonbreeding season survival from summer home range habitat composition of northern bobwhites on private lands in southwest Ohio, USA (2010-2015) using birds with enough locations to build summer home ranges that survived to the subsequent nonbreeding season ($n = 32$).

Model ^a	Estimate	SE	95% CI
IJI ES Woody	0.023	0.011	0.002, 0.045
Percent row crop	0.34	1.06	-1.74, 2.43

^a Early-successional (ES) and interspersed juxtaposition index (IJI; approaches zero when the distribution of unique patch adjacencies becomes uneven and 100 when all patch types are equally adjacent)

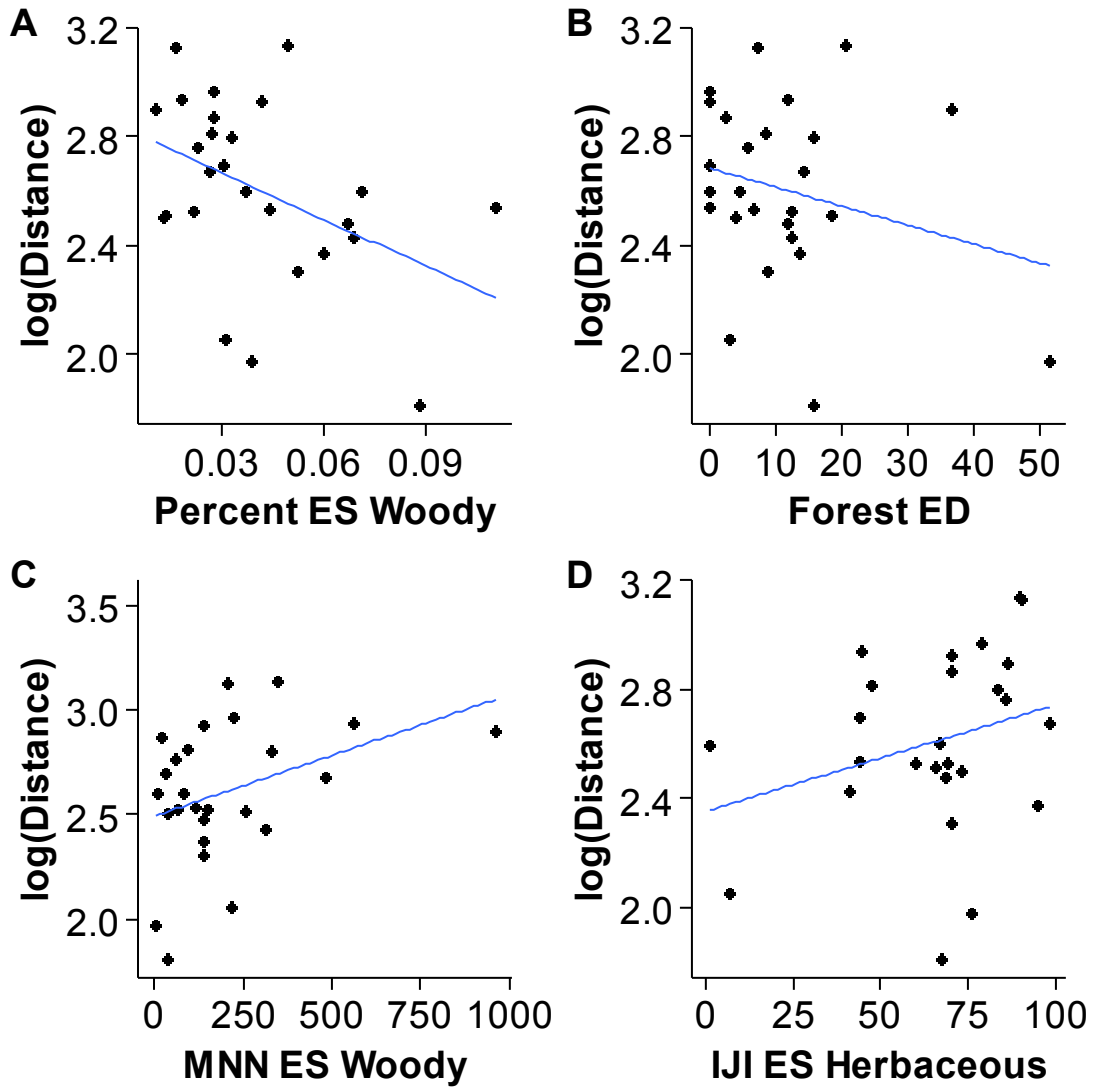


Figure 2.1. Relationship between summer home range habitat composition of northern bobwhites on distance moved from the centroid of the summer home range to the centroid of the winter home range in Brown and Highland Counties, Ohio, 2010-2015. The relationship between log distance and summer home range percent early-successional (ES) woody, forest edge density (ED), mean nearest neighbor (MNN) ES woody, and interspersed juxtaposition index (IJI) ES herbaceous are shown.

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CHAPTER 3: POPULATION LEVEL RESPONSES OF NORTHERN BOBWHITE TO LAND MANAGEMENT CHANGES IN SOUTHWEST OHIO

ABSTRACT

Agricultural land use conversion is one of the primary threats to biodiversity worldwide. The Conservation Reserve Program (CRP) is one tool used in the United States to provide habitat to farmland wildlife that otherwise have little habitat. Woodlot edge enhancement is another method land managers are implementing to manage for farmland wildlife as agricultural intensification and forest succession continue to remove early-successional wildlife habitat. The northern bobwhite (*Colinus virginianus*) is a rapidly declining bird associated with farmland habitat that is often managed for using CRP and woodlot edge enhancement. I studied northern bobwhites from 2008-2010 and 2012-2014 in Brown and Highland Counties, Ohio to document the effects of loss of CRP contracts and concurrent gain in row crop habitat and the effects of woodlot edge feathering on rate of decline in abundance using whistle count surveys. The rate of decline slowed with implementation of woodlot edge feathering but was not significantly correlated with any other predictor variables. My results suggest woodlot edge feathering may slow the rate of population decline around managed areas.

INTRODUCTION

Land conversion to agriculture is one of the primary threats to ecosystems and biodiversity worldwide (Rashford et al. 2012, Shaw et al. 2014). Agricultural land conversion has been historically extensive and continues today (Rashford et al. 2012). Worldwide, 21.8% of land area has been converted to human dominated uses (Hoekstra et al. 2005). The amount of land used for agriculture increased 466% from 1700-1980 (Matson et al. 1997). Not only has the amount of land used for agriculture increased, agriculture has intensified. Biodiversity has been shown to decline with agricultural intensification (Best et al. 1997, Matson et al. 1997, Chamberlain et al. 2000, Smith et al. 2005, Shaw et al. 2014). There has been an increase in mechanization, a steady shift to larger farms with row crops replacing small grains, and a reduction in uncultivated areas within farms such as fencerows and grassy field edges. These changes have resulted in a decrease in farmland habitat heterogeneity (Best et al. 1997, Chamberlain et al. 2000, Smith et al. 2005). Agricultural intensification can cause a decline in cover available for wildlife, a decline in food, a decrease in nesting habitat, can cause direct mortality, and is associated with an increase in predators (Chamberlain et al. 2000). In addition to agricultural intensification, many farmland bird species face habitat loss due to advanced forest succession. Fire suppression following World War II and farm abandonment has caused an increase in second-growth forests with sparse understories leaving little habitat for early-successional species (Rosene 1969, Trani et al. 2001).

Agricultural intensification causes habitat loss and fragmentation which affects the movement of wildlife (Matson et al. 1997). Fragmentation has been shown to

decrease species richness, abundance, occurrence, and reproductive success (Askins and Philbrick 1987, Boulinier et al. 1998, Boulinier et al. 2001, Donovan and Flather 2002).

Island biogeography theory suggests more isolated fragments should hold fewer species and have lower population sizes than larger, less isolated patches (Gotelli 1995).

Fragmentation can act on abundance and distribution by reducing functional connectivity, or the degree to which the landscape facilitates or impedes movement and dispersal (Gobeil and Villard 2002; Robertson and Radford 2009; Rodriguez et al. 2001). Reduced dispersal can cause extinction of subpopulations, inbreeding depression, and increased kin competition, all with negative population effects (Barton et al. 2009, Van Oort et al. 2010, Bonte et al. 2012).

The Conservation Reserve Program (CRP) was implemented in the Food Security Act of 1985 to lower commodity supplies and raise prices and to take highly erodible and environmentally sensitive land (areas such as highly erodible land, stream banks, and field margins) out of production. Land is planted with more natural forms of cover such as native grasses. Farmers are given financial compensation to offset the cost of retiring the land and are bound by a 10 to 15 year contract. Wildlife benefited from the addition of habitat, and wildlife conservation was later added as an objective (Best et al. 1997, Riffell et al. 2008, Conover et al. 2014, Stubbs 2014). However, the number of enrolled acres in the CRP has declined by 17.1 million acres since 2007 due primarily to high commodity crop prices and declining interest in retiring land from production. Contract expiration threatens wildlife that depends on habitat created through this program (Riffell et al. 2008, Stubbs 2014). Further, the Agricultural Act of 2014 reduced the cap on CRP

acres from 32 million to 24 million by FY 2015. It also allows termination of contracts after 5 years as long as the land is not enrolled in an environmentally sensitive practice (Stubbs 2014). Environmental benefits, such as wildlife population increases, gained from CRP are expected to be lost or reduced as land is returned to agricultural production (Stubbs 2014), and research is needed to quantify the effects.

Woodlot edge feathering is another management technique being recommended to enhance wildlife habitat in agricultural settings (i.e. Iowa Department of Natural Resources 2001, Dailey and Hutton 2003, Hoosier Heartland Resource Conservation and Development Council 2007, Kentucky Department of Fish and Wildlife Resources 2012, McPeake and Roberg 2012, Liberati et al. 2013). In this technique, early-successional woody habitat is created by hinge-cutting trees, or cutting about 2/3 of the way into the trunk and bending the tree parallel to the ground, along the edge of a forest. Recent research suggests woodlot edge feathering creates habitat that could be suitable for early-successional species (Brooks 2015), but no research has been conducted that documents a benefit to wildlife to date.

The northern bobwhite (*Colinus virginianus*), hereafter bobwhite, is one of the fastest declining species in North America which has been widely attributed to land use conversion, agricultural intensification, and forest succession (Stoddard 1931, Rosene 1969, Vance 1976, Brennan 1991, Seckinger et al. 2006, Veech 2006). Management recommendations to enhance bobwhite habitat commonly include addition of CRP and woodlot edge feathering. Bobwhites have been documented to increase in abundance with addition of CRP (Best et al. 1997, Riddle et al. 2008, Riffle et al. 2008, Bowling et

al. 2014, Evans et al. 2014, Lohr et al. 2014); however, no studies to date have been published empirically examining the loss of CRP on wildlife populations.

I sought to assess the effects of declining CRP and addition of woodlot edge feathering on farmland bird conservation. The bobwhite lends itself to my objectives because it is widely documented to benefit from addition of CRP and is seen as a beneficiary of woodlot edge feathering. Bobwhites have low dispersal rates and require nesting habitat, brood rearing habitat, and wintering habitat in close proximity (Stoddard 1931, Rosene 1969, Schroeder 1985). Small amounts of habitat can cause low dispersal rates, trapping birds in small habitat islands (Seckinger et al. 2006, Riddle et al. 2008, Bowling et al. 2014). I expected the loss of CRP to increase the rate of decline, not only from habitat loss but from declines in connectivity due to fragmentation's documented negative effects on bobwhite populations (Duren et al. 2011, Lohr et al. 2011). I expected addition of feathered edges to slow the rate of decline by increasing the amount of edge and improving habitat quality (Hanson and Miller 1961, Schroeder 1985, Guthery and Bingham 1992, Roseberry and Sudkamp 1998, Duren et al. 2011, Janke 2011).

I examined the effects of habitat change on the rate of change in abundance of bobwhites in southwest Ohio with annual whistle count surveys from 2008-2010 and 2012-2014. My objectives were 1) to describe the rate of change of northern bobwhites in southwest Ohio and 2) to determine how habitat change affects the rate of change in abundance. I predicted populations would decline on my sites, mirroring range wide and state wide trends. I further predicted that loss of early-successional herbaceous habitat, as would be expected with expiration of CRP contracts, would cause a faster rate of decline

in abundance. I also predicted that addition of early-successional woody vegetation by woodlot edge feathering would slow the rate of decline in abundance. My goals were to assess rates of decline, describe how habitat manipulations can affect local abundance, and inform managers of best practices to enhance bobwhite populations.

STUDY AREA

I conducted my research on four private land sites in Brown and Highland Counties in southwest Ohio, USA (39°04'59", 83°39'10") between May 2008 and July 2014. Brown and Highland Counties are in the glaciated till plains physiographic region (Ohio Division of Geologic Survey 1998). Study sites were chosen in the core of the bobwhite range in Ohio (Spinola and Gates 2008). Study sites ranged from 400 ha to 1,200 ha and ranged from 8-28% woodlot cover, 38-73% agricultural cover, and 4-11% grassland cover. (For a more detailed site description see Chapter 1, Janke 2011, Gates et al. 2012, Wiley 2012, or Liberati 2013.)

METHODS

Northern bobwhite surveys

A total of 47 unique points were surveyed across the Fee, Thurner, and Wildcat study sites from 2008-10 and 2012-2014, but only 22 survey points were surveyed across ≥ 5 of the 6 years of the study. The Peach study site was added in 2010. Nine points were surveyed on Peach in 2010, but 2 points were dropped from the 2012-2014 surveys. Points were excluded over time based on changing landowner permission and bird

densities. Sites had between 2 and 12 points that were consistently surveyed. All survey locations were ≥ 250 m apart (Gates et al. 2012).

Point counts were conducted between 0800 and 1100 hours from 01 May to 18 July to coincide with the breeding season in Ohio. Each point was surveyed every 4-7 days. Each unique male bobwhite seen or heard during a 6-minute time interval was recorded (Norton et al. 1961, Burger et al. 2006). The distance to each singing male was estimated up to 500 m on an aerial photograph with pre-measured distance rings created in ArcMap 10.1 (Environmental Systems Research Institute, Redlands, CA) to allow estimation of detection probabilities using distance based methods (Burger et al. 2006, Rusk et al. 2007, Bowling et al. 2014). Time of detection of singing males was recorded. Wind speed, noise level, and cloud cover was estimated for each survey (Burger et al. 2006). Birds detected on the border of adjacent points were only recorded inside one of the points.

Landscape Composition Analysis

Land cover was manually digitized on each study site for each year in ArcMap 10.1 with high spatial resolution ortho-photographs (0.305 m; Ohio Statewide Imagery Program 2008). I used 6 habitat categories for analyses (early-successional herbaceous, early-successional woody, forest, non-habitat, pasture/hay fields, and row crop fields). For details on how habitat was classified see Chapter 1, Janke 2011, Gates et al. 2012, Wiley 2012, or Liberati 2013. In 2012, early-successional woody vegetation was created by feathering woodlot edges in some survey points on the Fee ($n = 7$) and Peach ($n = 5$) study sites (Brooks 2015). In order to increase the amount of usable space, edges were

treated in areas with little usable space (see Wiley 2012 for details on usable space on my study sites). Cover was mapped yearly by ground-truthing, and any changes were reclassified in a new layer in ArcGIS.

I described landscape composition around each survey location by using a 1-km buffer which was the distance of the survey listening radii (Bowling et al. 2014). I used Patch Grid for ArcMap 10.1 on a rasterized image of survey point buffers to determine edge density (ED), mean nearest neighbor (MNN), and interspersed juxtaposition index (IJI) for each cover class and at the landscape scale (Taylor et al. 1999, Bowling et al. 2014). Landscape scale analyses performed calculations across classes for the entire 1 km buffer area whereas class analyses differentiated between habitat classes in the 1 km buffer area. I calculated percent composition of each habitat type within 1-km buffers (Bowling et al. 2014). Additionally, I classified percent composition of each habitat type at the site level.

Abundance Analysis

I used the Distance package in Program R (R Version 3.2.0, www.r-project.org, accessed 05 June 2015) to estimate male bobwhite abundance and density (Buckland et al. 1993, Guthery 1988, Somershoe et al. 2006, Risk et al. 2007, Thomas et al. 2010, Bowling et al. 2014). I truncated observations to 500-m surrounding survey locations. For 2008-2009 and 2012-2014 I binned observations into 10 50-m distance intervals due to evidence of rounding to favored distances to dampen measurement error (Thomas et al. 2010). I pooled data across years to increase the number of detections in determining detection probability (Somershoe et al. 2006, Bowling et al. 2014). I analyzed the 2010

survey data separately because detections were recorded in three bins (0-100, 100-250, and 250-500) rather than at more precise distances like the other years.

I analyzed models with every combination of key function (half-normal, hazard-rate, and uniform) and adjustments (cosine, Hermite polynomial, and simple polynomial). I selected the best model using Akaike's Information Criterion (AIC_c), chi-square model-fit statistics, and visual inspection of the detection probability plots (Buckland et al. 1993, Somershoe et al. 2006, Thomas et al. 2010, Bowling et al. 2014).

Abundance was calculated for each year for each site in Distance. These estimates were used to calculate the rate of change in abundance. A slope comparison test was used in the *smatr* package in Program R to look for differences in rates of change by site. Abundance was calculated for each year for each point in Distance. Percent change in abundance between 2008 and 2014 was calculated for each point to use in further analyses. Change in ED, MNN, and IJI was calculated for each habitat variable in the 500 m buffer around each point by subtracting estimates of 2014 from 2008 habitat estimates. Percent change between 2008 and 2014 in the percent composition of habitat types in each point was calculated. Mixed effects models were used in the *lme4* package in Program R to assess the effects of habitat change on rate of change in abundance (Askins and Philbrick 1987, Shaw et al. 2014). Site was used as a random effect in building models.

RESULTS

A total of 1096 whistling male bobwhites were detected during the 6-year study. The detection probability for 2008-2009 and 2012-2014 was 0.503 ± 0.0274 (SE), based

on the model that best fit the 2008-2009 and 2012-2014 data which was the hazard-rate key function with cosine adjustments (Table 3.1, Fig. 3.1). The detection probability for 2010 was estimated to be 0.400 ± 0.0389 (SE) based on the model that best fit the 2010 data which was the uniform key function with cosine adjustments (Table 3.2, Fig. 3.2). Abundance declined on all sites during the study period (Fig. 3.3). The pooled estimate between sites was a decline of -0.512 detections per point per year (95% CI -0.826, -0.343, $n = 22$). I ran the model for rate of change without the 2010 data in case the different recording methods and detection model affected the results. The model excluding the 2010 data predicted a decline of -0.651 detections per point per year (95% CI -0.856, -0.411, $n = 18$). Slope did not differ between sites ($P = 0.192$, $n = 22$). One point slightly increased in abundance, 5 points were unoccupied in the first and last years of the survey, and 23 points declined in abundance.

Throughout the study period, early-successional woody vegetation was intentionally added in survey points on the Fee ($n = 7$) and Peach ($n = 5$) study sites in 2012, but the Fee site also lost a woody CRP contract in 2012, reducing the amount of early-successional woody vegetation on 3 survey points (Table 3.3, Table 3.4). Grassy CRP was also converted into row crop fields in 2012, affecting 7 survey points distributed throughout all 4 sites. One row crop field was converted into pasture affecting 1 point. Six survey points lost pasture habitat from conversion into row crop. Eleven survey points gained row crop habitat during the study period.

I tested significance of habitat change at the point level in all categories assessed in predicting change in abundance, but early successional woody was the only category to

show a significant result in response to habitat change ($\chi^2 = 4.9425$, $P = 0.0262$).

Addition of early-successional woody vegetation predicted a rate of increase in abundance of 1.077 ± 0.481 (SE) detections per survey point per year.

DISCUSSION

I documented a decline in northern bobwhite abundance on all sites and in most points during the study period. Both models including and excluding the 2010 data predicted the same direction of decline, but the model excluding the 2010 data predicted an even larger rate of decline. Since bobwhites are declining so rapidly, a 2 year gap in the middle of the data likely caused a steeper slope due to exclusion of the years of intermediate abundance. This mirrors range wide trends of bobwhites which declined 4.08% annually from 1966-2013 according to Breeding Bird Survey data. Bobwhites in Ohio decreased at an even higher rate of 6.88% per year during the same period (Sauer et al. 2014). Statewide whistle count surveys conducted by the Ohio Division of Wildlife showed a 94% decline in abundance from 1984-2014, and the number of whistling males detected dropped by 50% between 2013 and 2014 (Ohio Division of Wildlife 2014).

I wanted to test woodlot edge feathering's efficacy in preserving bobwhite populations because of its increasing prevalence as a management tool for bobwhites (i.e. Iowa Department of Natural Resources 2001, Dailey and Hutton 2003, Hoosier Heartland Resource Conservation and Development Council 2007, Kentucky Department of Fish and Wildlife Resources 2012, McPeake and Roberg 2012, Liberati et al. 2013) despite only one empirical study being published at the time of this writing. This study showed a

decline in survival of bobwhites whose home ranges were closer to treated edges (Smith et al. 2014). Woodlot edge feathering spurs growth of early-successional woody vegetation which provides visual obstruction from predators and helps with thermoregulation (Forrester et al. 1998, Smith 2014, Brooks 2015). Because winter survival has been shown to be limiting in Ohio, and woody cover is both preferred and sparse in agricultural dominated landscapes (Janke 2011), adding woody cover could lead to increased winter survival, helping in population preservation (Williams et al. 2000, Janke 2011, Janke and Gates 2013). Treated edges were placed in areas with little usable space where abundance was already low to try to increase the total usable space on the study area (Wiley 2012). I saw a slight positive effect on rate of decline in abundance in survey points with treated edges. My results suggest addition of treated edges may help slow population decline at least at a small scale. However, radio-collared birds on my study sites during the same time period were rarely documented using treated edges in both the breeding and nonbreeding season, even when the treated edges fell within their home ranges (Brooks 2015). Birds were monitored ≥ 3 -7 times per week at variable times each day to sample the full range of diurnal activities. This suggests limited use of treated edges, but perhaps I would have seen greater use with more constant monitoring. It is possible that the edges are used primarily as escape cover (Williams et al. 2000). Bobwhites may forage in nearby habitat and move into the treated edge when a predator is detected. Future studies could monitor use of treated edges with motion activated cameras or by looking for tracks in snow inside the edges.

The lack of an effect of loss of CRP on rate of decline is surprising. Many studies have documented the positive effects of CRP on bobwhite abundance, density, and occupancy (Seckinger et al. 2006, Riddle et al. 2008, Riffel et al. 2008, Bowling et al. 2014, Conover et al. 2014, Evans et al. 2014), so I expected loss to have a negative effect. One possible reason I did not detect an effect is the limited sample size. Only 7 points experienced loss of CRP during the study period and these were distributed over the 4 study sites. Additionally, conversion of grass to row crop fields may not have shown an effect on summer breeding male abundance because row crop fields were still permeable to bobwhites in summer when surveys were conducted (Hanson and Miller 1961, Vance 1976, Schroeder 1985, Guthery 1997, Wiley 2012). Furthermore, nesting habitat loss may require several breeding seasons to take effect (Chamberlain et al. 2000, Duren et al. 2014). Bobwhite responses may need to be monitored for a longer time period to see an effect. Landscape context has been shown to be important in effectiveness of CRP at increasing bobwhite abundance (Riddle et al. 2008, Evans et al. 2014). Though I could not test for statistical significance of larger landscape scale habitat loss, I did note that the site with the largest decline in abundance (Wildcat) had 10% of its land area converted from CRP and pasture/hay into row crops during the study period which was the largest loss of grassland habitat to row crops on any of the sites.

I expected interspersed/juxtaposition, edge density, and patch distance to be important in point level abundance due to the documented benefit of edge and interspersed on bobwhite abundance and documented negative effects of increasing patch distance (fragmentation) (Hanson and Miller 1961, Baxter and Wolfe 1972,

Schroeder 1985, Guthery and Bingham 1992, Guthery 1997, Roseberry and Sudkamp. 1998, Riddle et al. 2008, Duren 2011, Janke 2011, Lohr et al. 2011, Bowling et al. 2014). I was surprised to find no effect. I looked at how point level habitat change during the study period affected rate of change in abundance, so perhaps spatial scale or sample size caused me to not find an effect.

As the land area converted to agriculture increases and agricultural intensification increases, it is necessary to develop effective conservation practices for wildlife that depend on early-successional habitat within agricultural landscapes. Woodlot edge feathering may help temporarily slow the decline of bobwhites at a small scale, but more research is needed on how to best place treated edges and how they affect other species of concern. The effectiveness of CRP at increasing bird populations has been thoroughly demonstrated in the literature (Best et al. 1997, Riffell et al. 2008, Conover et al. 2014), but its concurrent loss as contracts expire is relatively undocumented. As CRP contracts continue to expire without replacement and the cap on the number of acres declines (Stubbs 2014), it is increasingly important to understand how loss of CRP will affect wildlife populations. Despite decades of research, bobwhite populations continue to decline range wide at a rapid rate, so understanding how to effectively manage them on working agricultural lands is imperative.

MANAGEMENT IMPLICATIONS

Implementation of edge feathering slowed the rate of decline in bobwhite abundance in the points the treated edges fell within. Though widely recommended, this

is the first evidence that woodlot edge feathering may help slow local population declines. More research is needed on the optimum placement and effects on species of concern. I did not find a significant effect on rate of population change for the first 3 years after the loss of CRP despite many studies showing its addition increases local abundance. Future research should investigate the effects of loss at a larger spatial scale over a larger time period to further assess how loss of CRP affects the rate of decline in bobwhites.

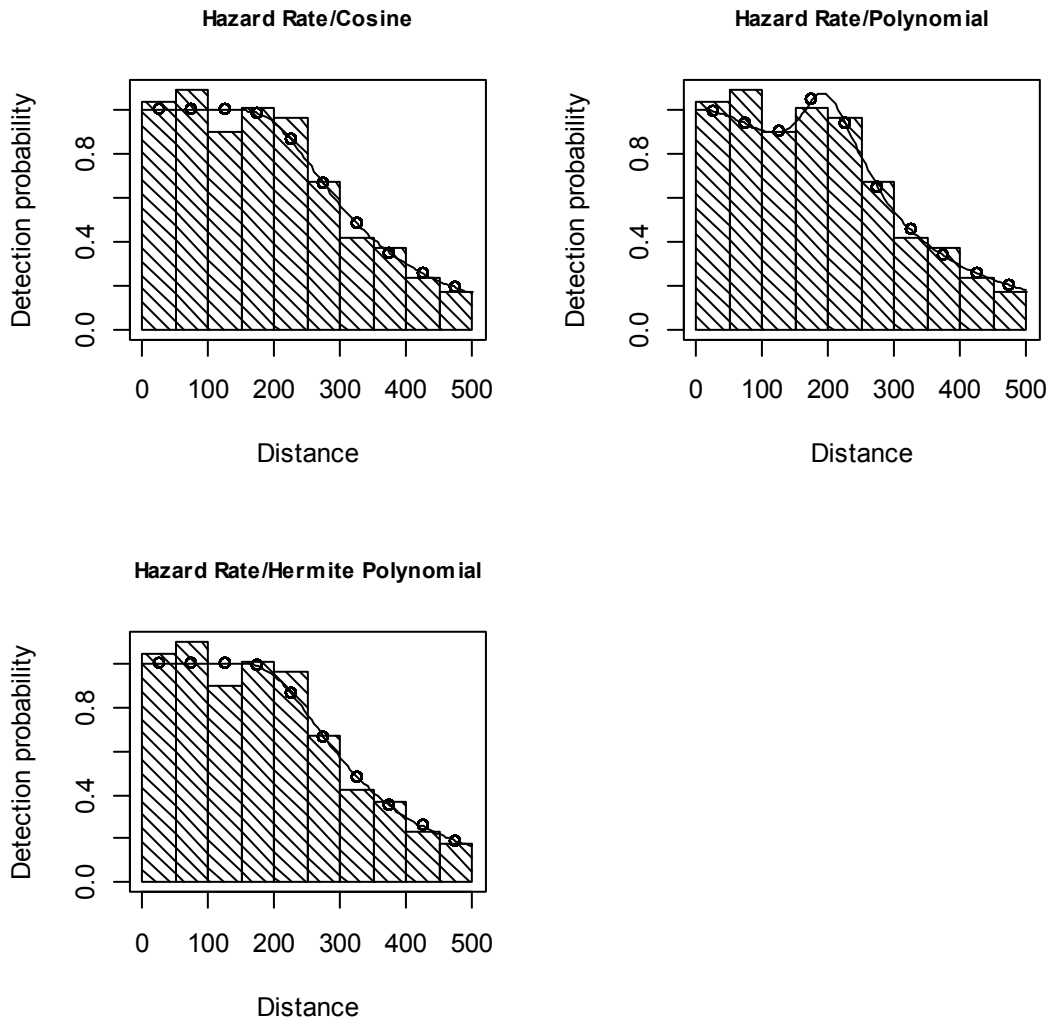


Figure 3.1. Plots of competitive models estimating detection probability of whistling male northern bobwhites in Southwest Ohio 2008-2009 and 2012-2014.

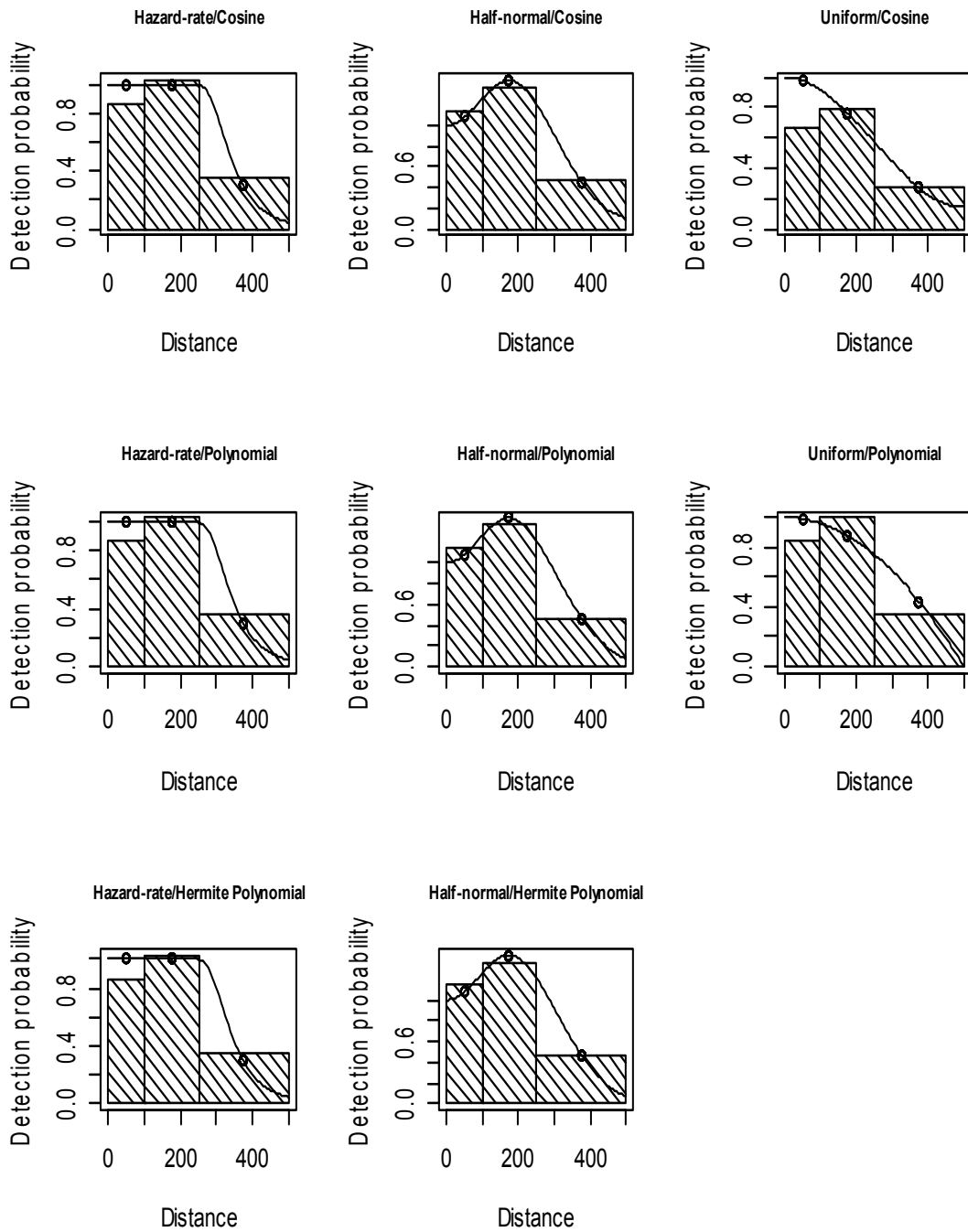


Figure 3.2. Plots of competitive models estimating detection probability of whistling male northern bobwhites in Southwest Ohio in 2010.

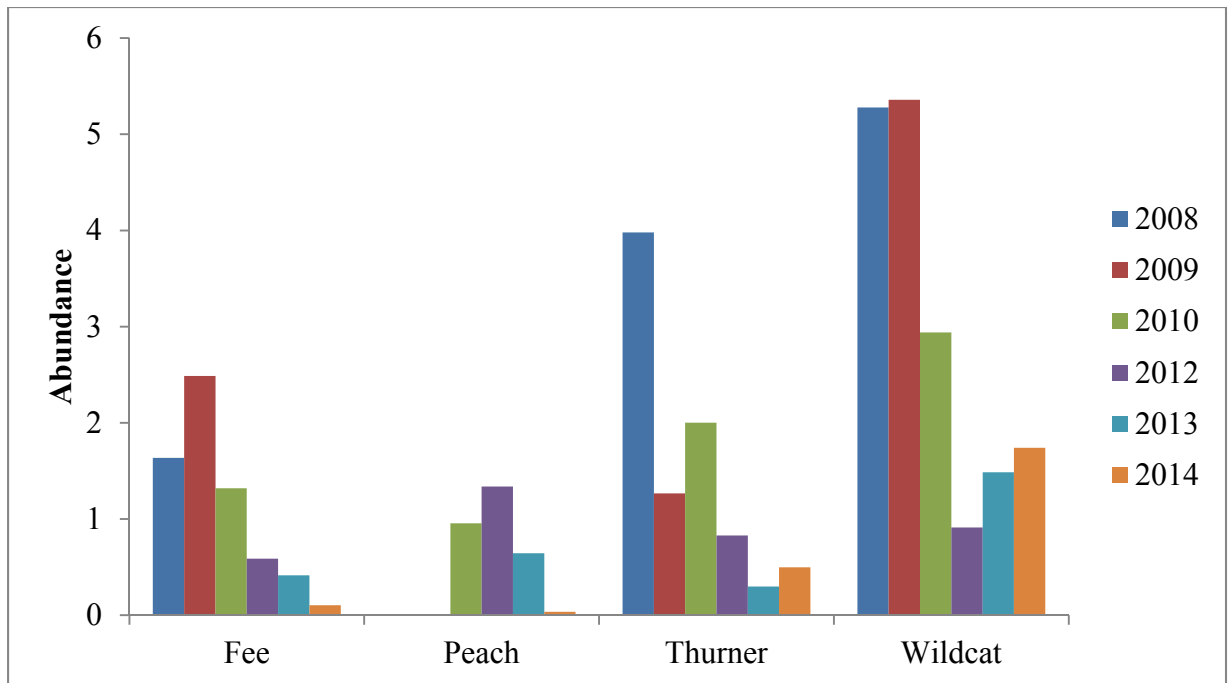


Figure 3.3. Site level abundance of northern bobwhites determined from whistle count surveys conducted from 2008-2014 on 4 study sites located on private lands in Brown and Highland Counties, Ohio.

Table 3.1. Model set including corrected Akaike’s Information Criterion (AIC_c), difference in corrected Akaike’s Information Criterion (ΔAIC_c), and number of model parameters (K) for determining detection probability of northern bobwhites from 2008-2009 and 2012-2014 in southwest Ohio.

Key function	Adjustment	AIC_c	ΔAIC_c	K
Hazard rate	Hermite polynomial	4042.986	0	2
Hazard rate	Polynomial	4043.222	0.236	3
Hazard rate	Cosine	4044.403	1.417	2
Uniform	Cosine	4048.67	5.684	1
Uniform	Polynomial	4051.064	8.078	1
Half normal	Cosine	4051.178	8.192	1
Half normal	Hermite polynomial	4052.743	9.757	1
Half normal	Polynomial	4053.146	10.16	1
Uniform	Hermite polynomial	4335.299	292.313	1

Table 3.2. Model set including corrected Akaike’s Information Criterion (AIC_c), difference in corrected Akaike’s Information Criterion (ΔAIC_c), and number of model parameters (K) for determining detection probability of northern bobwhites in 2010 in southwest Ohio.

Key Function	Adjustment	AIC	ΔAIC_c	K
Uniform	Cosine	322.667	0	1
Uniform	Polynomial	323.301	0.634	1
Hazard-rate	Cosine	323.814	1.147	2
Half-normal	Cosine	323.814	1.147	2
Hazard-rate	Polynomial	323.814	1.147	2
Half-normal	Polynomial	323.814	1.147	2
Hazard-rate	Hermite polynomial	323.814	1.147	2
Uniform	Hermite polynomial	364.284	41.617	1

Table 3.3. Number of northern bobwhite whistle count survey points that experienced habitat gain (+) or loss (-) of six habitat categories on 4 study sites between 2008 and 2014 in Brown and Highland Counties, Ohio.

Site		Early-successional herbaceous	Early-successional woody	Forest	Non-habitat	Pasture/Hay	Row Crop
Fee	(+)		4				4
	(-)	4	3				
Peach	(+)		4			1	2
	(-)	1				1	1
Turner	(+)						1
	(-)	1				1	
Wildcat	(+)						4
	(-)	1				4	

Table 3.4. Percent habitat change of the total land area and change in abundance on four study sites in Brown and Highland Counties, Ohio. Fee, Thurner, and Wildcat were studied from 2008-2014. Peach was studied from 2010-2014.

Site	Δ Abundance ^a	ESH ^b	ESW ^c	Forest	Non-habitat	Pasture/ Hay	Row Crop
Fee	-1.53	-6.96%	-0.68%	-0.16%	0.02%	3.96%	3.82%
Peach	-0.92	-7.36%	0.28%	-0.11%	0.25%	9.65%	-2.72%
Thurner	-3.48	-0.04%	0.03%	-0.58%	-0.58%	0.20%	0.88%
Wildcat	-3.54	-6.44%	0%	0%	0%	-4.11%	10.50%

^aChange in abundance from 2008-2014 for Fee, Thurner and Wildcat, and change in abundance from 2010-2014 for Peach

^bEarly-successional herbaceous

^cEarly-successional woody

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CHAPTER 4: MANAGEMENT IMPLICATIONS

The northern bobwhite is one of the fastest declining species in North America primarily from habitat loss (Roseberry and Sudkamp 1998, Taylor et al. 1999, Williams et al. 2000, Cook 2004, Seckinger et al. 2006, Riddle et al. 2008, Duren et al. 2011, Bowling et al. 2014). Bobwhites were able to expand their historical range as Europeans began to move west and converted existing land to agriculture (Rosene 1969, Brennan 1991). Historical farming practices provided ideal habitat for bobwhite in the form of food and cover. Farms consisted of small, diverse patches with abundant fence rows providing food, nesting habitat, and woody cover in close proximity (Stoddard 1932).

Though farming enabled range expansion, the bobwhite's recent decline is largely attributed to agricultural intensification and changes in forestry since World War II (Brennan 1991, Taylor et al. 1999, Williams et al. 2000, Cook 2004, Flock 2006, Riddle et al. 2008, Duren et al. 2011, Lohr et al. 2011). Agricultural intensification has caused a decrease in cover diversity, an increase in patch size, and a reduction in fencerows, reducing habitat availability and quality for bobwhite (Vance 1976, Taylor et al. 1999, Chamberlain et al. 2000, Cook 2004, Flock 2006, Seckinger et al. 2006, Lohr et al. 2011, Conover et al. 2014, Riddle et al. 2014). Further, changing forest practices have led to advanced forest succession, reducing the amount of understory cover (Cook 2004, Seckinger et al. 2006, Bowling et al. 2014, Riddle et al. 2014).

Declines continue despite the bobwhite being one of the most intensively researched species since the 1920s (Brennan 1991, Guthery 1997, Taylor et al. 1999, Flock 2006). Studies aimed at improving habitat quality have generally been unsuccessful at improving populations while studies aimed at increasing useable space have had more success (Guthery 1997). The Northern Bobwhite Conservation Initiative (NBCI) states it is necessary to manage bobwhites on private, working lands. NBCI believes it is necessary to offer assistance to landowners with habitat management and to provide good economic incentives through programs like the Conservation Reserve Program (CRP) (Palmer et al. 2011). Nationally, 65.29% of lands are privately owned, while in Ohio over 97% of land is privately owned (National Wilderness Institute 1995). Effective bobwhite management will have to take place on private, working lands.

Facilitating dispersal through agricultural lands is important because it can allow bobwhites to avoid extinction of subpopulations, find higher quality habitat, avoid resource competition in high density areas, increase gene flow, and reduce kin competition (Perrin and Maxalov 2000, Fies et al. 2002, Townsend et al. 2003, Barton et al. 2009, Van Oort et al. 2010, Bonte et al. 2012, Chaine and Clobert 2012). Fall dispersal is important because the habitat a bird winters in can affect winter survival (Williams et al. 2000, Cook 2004, Flock 2006, Janke 2011, Lohr et al. 2011, Fedy 2012), and winter survival has been shown to be limiting to bobwhites in Ohio (Gates et al. 2012).

Bobwhites in this study moved an average distance of 515 m between summer and fall home range centroids. Forest edge density was negatively correlated with

interseasonal movements, perhaps because forest edge can provide winter cover for bobwhites in areas with little usable space like in highly agricultural landscapes (Guthery 1997, Janke 2011, Lohr et al. 2011). Bobwhites moved further between seasons when preferred winter cover patches were further apart. Bobwhites that select patches with shorter mean nearest neighbor patches of preferred winter cover would be able to spend less time exposed to predators in the matrix (Whittingham and Evans 2004, Chaine and Clobert 2012).

These results suggest bobwhites will shift their home ranges more in fall if there is less suitable habitat in their summer home ranges. Though I did not find a significant correlation between survival through winter and the degree of home range shift, I did find that the shift increased habitat attributes that have been shown to increase survival. In the highly agricultural landscapes of Ohio, bobwhites are most likely limited in their ability to disperse after crop harvest when the majority of the landscape becomes impermeable. My results suggest bobwhites may have some ability to find more suitable winter habitat after crop harvest but also suggests a limitation since average dispersal distance was just over a half kilometer. Many studies have suggested the importance of having adequate habitat for nesting, brood rearing, and wintering in close proximity (Stoddard 1932, Hanson and Miller 1961, Schroeder 1986, Wiley 2012). Bobwhites in Ohio likely need winter cover close to nesting and brood rearing habitat due to limited dispersal ability once crops are harvested. Management efforts should place early-successional woody habitat such as woodlot edge feathering within a half kilometer or less of nesting (CRP) and brood rearing habitat due to limited dispersal of bobwhites in fall.

The CRP practice CP33 Habitat Buffers for Upland Birds is specifically designed to meet the NBCI's conservation goals for bobwhites. It is a continuous enrollment program that allows farmers to only enroll a 9.1-36.6 m strip on any border(s) of their fields. Farmers receive a sign up incentive, an annual rental payment, a cost-share for cover establishment, a practice incentive payment, and a cost-share for mid contract management (Burger et al. 2006). Buffers have been shown to increase bobwhite abundance and occupancy (Bowling et al. 2014, Riddle et al. 2014). Buffers can provide grassland nesting habitat for bobwhites right next to crop fields used for brood rearing. Farmers could then implement woodlot edge enhancement in nearby areas, providing all habitat requirements within an area bobwhites are likely to be able to disperse in fall while causing minimal loss of valuable field space.

Woodlot edge feathering is a widely recommended management tool for bobwhites to help conserve populations (i.e. Iowa Department of Natural Resources 2001, Dailey and Hutton 2003, Hoosier Heartland Resource Conservation and Development Council 2007, Kentucky Department of Fish and Wildlife Resources 2012, McPeake and Roberg 2012, Liberati et al. 2013) but remains relatively empirically untested. Woodlot edge feathering was implemented on 12 of 29 survey points in 2012 to test its effects on rates of decline in abundance. Treated edges were placed in areas with little usable space where abundance was already low to try to increase the total usable space on the study area (Wiley 2012). The majority of treated edges were 30 by 9 m, but they ranged from 15 by 9 m to 60 by 9 m depending on the specific landowner's needs. Addition of early-successional woody vegetation was positively correlated with change in

abundance and predicted a rate of increase in abundance of 1.077 ± 0.481 (SE) ($\chi_1^2 = 4.9425$, $P = 0.0262$) detections per survey point per year. These results suggest addition of treated edges may help slow population decline at least at a small scale. However, radio-collared birds on my study sites during the same time period were rarely documented using treated edges in both the breeding and nonbreeding season, even when the treated edges fell within their home ranges (Brooks 2015).

Woodlot edge enhancement is an attractive management tool because it provides little interference with working lands and immediately creates ground level woody cover for wildlife. Farmers can selectively fell trees at the border of crop fields along a mature woodlot edge. Less valuable tree species can be chosen so as to not have a financial burden if farmers intend to harvest trees in the future. There are no current published studies documenting the effects of woodlot edge enhancement past 3 years after implementation. More research is needed to see how vegetation changes over the course of a longer time period and how to keep habitat optimal over time.

As the area of native land cover, especially grasslands, converted to agriculture increases and agricultural intensification increases, it is necessary to develop effective conservation practices for wildlife that depend on early-successional habitat within agricultural landscapes. Management strategies are needed that can be implemented on working, private farms with little independent cost to the landowner. Seasonal movements are fundamental to population sustainability, and relatively immobile, gallinaceous birds such as the northern bobwhite need habitat for the entire annual cycle within a short area. My study suggested bobwhites need wintering habitat perhaps 500 m

or less from brood rearing habitat. Woodlot edge feathering is a relatively non-invasive technique for adding early-successional woody habitat to farmlands adjacent to brood rearing habitat. Less than 1% of my study area was converted to early-successional woody habitat, but my study showed woodlot edge feathering may help temporarily slow the decline of bobwhites at a small scale. More research is needed on how to best place treated edges and how they affect other species of concern. As CRP contracts continue to expire without replacement and the cap on the number of acres declines (Stubbs 2014), it is important to understand how loss of this habitat affects farmland birds. I did not find a significant effect, but more research is needed as the acreage of CRP continues to decline due to its demonstrated importance in preserving bobwhite populations. Despite decades of research, bobwhite populations continue to decline range wide at a rapid rate, so building an effective management strategy for private, working lands is imperative.

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