

**WINTER ECOLOGY OF RAPTORS IN COVER-CROPPED
AGROECOSYSTEMS IN WESTERN INDIANA**

by

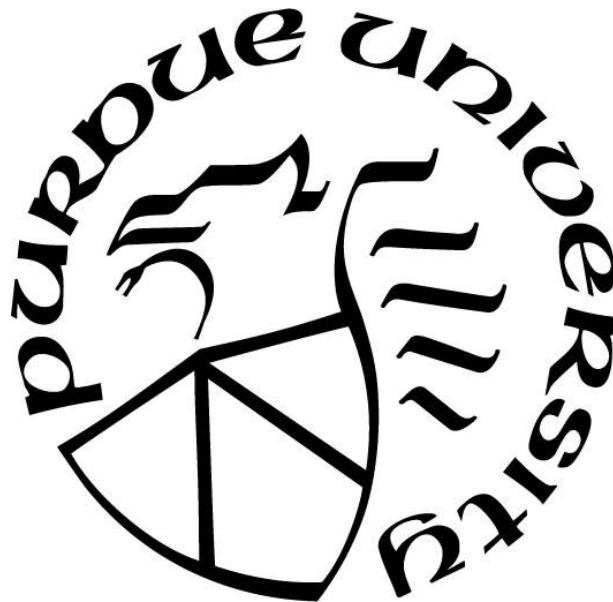
Megan Elizabeth Zagorski

A Thesis

Submitted to the Faculty of Purdue University

In Partial Fulfillment of the Requirements for the degree of

Master of Science



Department of Forestry and Natural Resources

West Lafayette, Indiana

December 2019

THE PURDUE UNIVERSITY GRADUATE SCHOOL
STATEMENT OF COMMITTEE APPROVAL

Dr. Robert K. Swihart, Chair

Department of Forestry and Natural Resources

Dr. John B. Dunning, Jr.

Department of Forestry and Natural Resources

Dr. Adam K. Janke

Iowa State University, Department of Natural Resource Ecology and Management

Approved by:

Dr. Robert G. Wagner

ACKNOWLEDGMENTS

First and foremost, I would like to thank my advisor, Dr. Rob Swihart, for his tireless support and assistance over the last 2.5 years. I would also like to thank my committee members, Drs. John B. Dunning Jr. and Adam Janke, for their input and insight. Their perspectives and questions greatly improved this work.

This thesis would not have been possible without the participation of our cooperating farmers and landowners. I greatly appreciate their willingness to allow us repeated access to their land. Mark Arvin and Drs. Steven Lima and Peter Scott pointed me to northern harrier roosts, and the staff at The Nature Conservancy's Efroymson Restoration at Kankakee Sands, Prophetstown State Park, Chinook Fish and Wildlife Area, and the Indiana Division of State Parks and Nature Preserves graciously granted me access to collect pellets.

DJ Hixon and Claudette Conigliaro, our technicians, were always enthusiastic and willing to collect data in cold, frigid winds and snow, even when things did not go as planned. The phalanx of undergraduate workers and volunteers were a great help during busy field seasons: Taylor Doetterl, Anna Downing, Laura Estrada, Shannon Flannigan, Caroline Harms, Helen Nesius, Camille Vann, and Alexis Vercauteren.

Brian Beheler helped brainstorm artificial perch design. Brian Dillman, Sebastian Ponce, Austin Decker, and Purdue Aviation helped with coordination and piloted aerial imagery flights. Local Natural Resources Conservation Services, Soil and Water Conservation District, and Indiana Conservation Cropping Systems Initiative staff facilitated and identified producers willing to partner with us.

Lastly, but certainly not least, I thank the Swihart lab for their help and input in discussing various stages of my projects. Andrea Brennan and Mayra Rodríguez-González were always ready with a literary discussion or a laugh. Finally, I never would have gotten this far without my parents. Thank you for always being there and encouraging me to keep going.

TABLE OF CONTENTS

LIST OF TABLES	6
LIST OF FIGURES	9
ABSTRACT.....	10
CHAPTER 1. INTRODUCTION	12
CHAPTER 2. RAPTOR USE OF ARTIFICIAL PERCHES IN COVER-CROPPED SOYBEAN (<i>GLYCINE MAX</i>) FIELDS OF WEST-CENTRAL INDIANA.....	14
2.1 Abstract	14
2.2 Introduction.....	14
2.3 Methods.....	17
2.3.1 Study Area	17
2.3.2 Experimental and Perch Design.....	17
2.3.3 Data Collection	17
2.3.4 Statistical Analysis.....	18
2.4 Results	19
2.5 Discussion	20
CHAPTER 3. RAPTOR WINTER HABITAT USE IN AGRICULTURAL LANDSCAPES OF WEST-CENTRAL INDIANA.....	27
3.1 Abstract	27
3.2 Introduction.....	27
3.3 Methods.....	29
3.3.1 Study Area	29
3.3.2 Data Collection	30
Transects	30
Aerial Photography.....	31
3.3.3 GIS Analysis	31
3.3.4 Statistical Analysis.....	32
3.4 Results	34
3.4.1 American Kestrels	34
3.4.2 Red-tailed Hawks.....	35
3.4.3 Rough-legged Hawks.....	36

3.4.4 Northern Harrier	37
3.5 Discussion	37
CHAPTER 4. VARIATION IN NORTHERN HARRIER (<i>CIRCUS HUDSONIUS</i>) DIETS IN WESTERN INDIANA.....	49
4.1 Abstract	49
4.2 Introduction.....	49
4.3 Methods.....	51
4.3.1 Study Area	51
4.3.2 Pellet Collection and Dissection.....	51
4.3.1. Statistical Analysis	53
4.4 Results	54
4.4.1 Influence of Counting Method.....	54
4.4.2 Variation Between Years	55
4.4.3 Variation Between Roost Sites	55
4.5 Discussion	56
CHAPTER 5. CONCLUSIONS.....	69
APPENDIX A. SUMMARY OF LANDCOVER, RAPTOR OBSERVATIONS, AND MODEL OUTPUT ALONG ROADSIDE TRANSECTS	72
APPENDIX B. COMPARISON OF METHODS FOR COUNTING THE MINIMUM NUMBER OF PREY INDIVIDUALS PER PELLET	79
REFERENCES	81

LIST OF TABLES

Table 2.1: Raptor artificial perch use for 2018 and 2019 in 17 cover-cropped soybean (<i>Glycine max</i>) fields in west-central Indiana.	24
Table 2.2: Parameter estimates from the AIC-best GLMM for raptor use of artificial perches in cover-cropped soybean (<i>Glycine max</i>) fields in west-central Indiana, USA. Factor variable coefficients are given in comparison to their reference level (Distance = 50m; Species = American kestrel; Interaction = American kestrel:Day).	25
Table 3.1: Landcover and habitat classes used in the set of models of raptor resource selection. For all features listed, explanatory variables were distance (m) from the focal landcover or habitat class. The variables considered for each species are noted (+). Species abbreviations: AMKE = American kestrel (<i>Falco sparverius</i>), RTHA = red-tailed hawk (<i>Buteo jamaicensis</i>), RLHA = rough-legged hawk (<i>B. lagopus</i>), and NOHA= northern harrier (<i>Circus hudsonius</i>). ..	42
Table 3.2: Summary of raptor species seen on 1184 km of transects driven in west-central Indiana in January-April 2018 and 2019.	43
Table 3.3: Summary of the top candidate models ($\leq 2 \Delta AICc$) and the number of models including each variable for each species. Area under the receiver operating curve (AUC) and normalized area under the precision-recall curve (AUCNPR) are reported for the top model. Abbreviations: RTHA = red-tailed hawk (<i>Buteo jamaicensis</i>), AMKE = American kestrel (<i>Falco sparverius</i>), RLHA = rough-legged hawk (<i>Buteo lagopus</i>), NOHA = northern harrier (<i>Circus hudsonius</i>), R = available points selected randomly, CR = available points constrained by hunting mode, Cover = cover-cropped fields, NoCover = non-cover-cropped fields, PermHab = permanent herbaceous habitat, Woods = woodlots, Dev = development, Uline = utility lines, Treeperim = outer perimeter of tree canopies. NA = not included in models.	44
Table 3.4: Regression coefficients (\pm Standard Error) for the top 1-3 resource selection models for each species, as determined with $\Delta AICc$. Abbreviations: RTHA = red-tailed hawk (<i>Buteo jamaicensis</i>), AMKE = American kestrel (<i>Falco sparverius</i>), RLHA = rough-legged hawk (<i>Buteo lagopus</i>), NOHA = northern harrier (<i>Circus hudsonius</i>), R = available points selected randomly, CR = available points constrained by hunting mode, Cover = cover-cropped fields, NoCover = non-cover-cropped fields, PermHab = permanent herbaceous habitat, Woods = woodlots, Dev = development, Uline = utility lines, Treeline = linear tree features, Treeperim = outer perimeter of tree canopies. Variables with a p value ≤ 0.05 are bolded and those with $0.05 < p \leq 0.1$ are italicized.	45
Table 4.1: List of prey identified in northern harrier (<i>Circus hudsonius</i>) pellets collected at 4 communal roosts in western Indiana, December through April 2018 and 2019.	61
Table 4.2: Percent occurrence of prey groups in northern harrier (<i>Circus hudsonius</i>) pellets collected December-April 2018 and 2019 at 4 roost sites in western Indiana. If $\geq \frac{1}{2}$ of the teeth were present for pellets without skulls or mandibles or where the teeth did not match the skulls and mandibles, 1 individual was counted.	62

Table 4.3: Comparison of methods of counting teeth toward the minimum number of individuals (MNI) per pellet for northern harrier (*Circus hudsonius*) pellets collected at 4 roost sites in western Indiana in 2018 and 2019. I evaluated 4 methods for counting individuals: if ≥ 1 tooth was present (any teeth), if $\geq \frac{1}{4}$ of the teeth were present, if $\geq \frac{1}{2}$ of the teeth were present, and ignoring teeth (no teeth). The proportion of the maximum number of individuals for that prey group is included in parentheses. 63

Table 4.4: Adjusted likelihood ratio tests and associated parameter estimates (\pm SE) from a multivariate Poisson generalized linear model testing whether northern harrier (*Circus hudsonius*) pellet contents changed in 2018 and 2019 at the Kankakee Sands roost, Newton County, Indiana. If $\geq \frac{1}{2}$ of the teeth were present for pellets without skulls or mandibles or where the teeth did not match the skulls and mandibles, 1 individual was counted. The baseline year was 2018; thus, positive coefficients indicate larger expected counts in 2019. Boldfaced font used for tests with $p < 0.05$, and italicized font for tests with $0.05 \leq p < 0.10$ 64

Table 4.5: Adjusted likelihood ratio tests and associated parameter estimates (\pm SE) from a multivariate Poisson generalized linear model testing whether northern harrier (*Circus hudsonius*) pellet contents changed between months and roost sites in western Indiana during 2018 and 2019. If $\geq \frac{1}{2}$ of the teeth were present for pellets without skulls or mandibles or where the teeth did not match the skulls and mandibles, 1 individual was counted. The baselines were February (Collection Month) and Chinook Mine (2019, Roost Site). Positive coefficients indicate larger expected counts in March (Collection Month) or at the specified roost site relative to the baseline. Boldfaced font used for tests with $p < 0.05$ 65

Table A1: Landcover (%) of the 18 transect routes in 2018 and 2019. Abbreviations: Cover = cover-cropped fields, NoCover = non-cover-cropped fields, PermHab = permanent herbaceous habitat, Woods = woodlots, Dev = development. 72

Table A2: P values for covariates in the top 1-3 candidate models for each species, scale, and type of available points, as determined with Δ AICc. Abbreviations: RTHA = red-tailed hawk (*Buteo jamaicensis*), AMKE = American kestrel (*Falco sparverius*), RLHA = rough-legged hawk (*Buteo lagopus*), NOHA = northern harrier (*Circus hudsonius*), R = available points selected randomly, CR = available points constrained by hunting mode, Cover = cover-cropped fields, NoCover = non-cover-cropped fields, PermHab = permanent herbaceous habitat, Woods = woodlots, Dev = development, Uline = utility lines, Treeline = linear tree features, Treeperim = outer perimeter of tree canopies. Variables with a p value ≤ 0.05 are bolded and those with $0.05 < p \leq 0.1$ are italicized. 73

Table B1: Multivariate analysis of deviance table for a negative binomial multivariate regression model of northern harrier (*Circus hudsonius*) pellets collected at 3 roosts in western Indiana in 2018 and 2019. The influence of 4 different methods of counting the minimum number of individuals (MNI) on conclusions regarding collection and roost site was assessed with 2-way interaction terms. The 4 methods of counting were: if ≥ 1 tooth was present (any teeth), if $\geq \frac{1}{4}$ of the teeth were present, if $\geq \frac{1}{2}$ of the teeth were present, and ignoring teeth (no teeth) on the statistical outcome was assessed. A colon between factors denotes an interaction. 79

Table B2: Percent occurrence of prey groups in northern harrier (*Circus hudsonius*) pellets collected December-April 2018 and 2019 at 4 roost sites in western Indiana. Percent occurrences are presented for all 4 methods of counting teeth toward the minimum number of individuals per pellet: ≥ 1 molar (any teeth), $\geq \frac{1}{4}$ of molars, $\geq \frac{1}{2}$ of molars, not including molars (no teeth).....80

LIST OF FIGURES

Figure 2.1: Interaction between ordinal day and estimated species-specific use of artificial perches in cover-cropped soybean (<i>Glycine max</i>) fields of west-central Indiana, USA.....	26
Figure 3.1: Transect routes surveyed for raptors in west-central Indiana January-April 2018 and 2019.....	48
Figure 4.1: Northern harrier (<i>Circus hudsonius</i>) roost sites in western Indiana. Pellets were collected in 2018 (n=2 roosts) and 2019 (n=3 roosts).	66
Figure 4.2: Percent occurrence of prey groups by collection month for northern harrier (<i>Circus hudsonius</i>) pellets collected at 4 roost sites in 2018 and 2019 in western Indiana.	67
Figure A1: Observed distances (m) for American kestrels (<i>Falco sparverius</i>) along transects in west-central Indiana in 2018 and 2019. Only observations of birds that could potentially be hunting (perched or hovering) are included. The distance (m) at which I set the species-specific buffer is indicated with a dotted line.	75
Figure A2: Observed distances (m) for red-tailed hawks (<i>Buteo jamaicensis</i>) along transects in west-central Indiana in 2018 and 2019. Only observations of birds that could potentially be hunting (perched) are included. The distance (m) at which I set the species-specific buffer is indicated with a dotted line.	76
Figure A3: Observed distances (m) for rough-legged hawks (<i>Buteo lagopus</i>) along transects in west-central Indiana in 2018 and 2019. Only observations of birds that could potentially be hunting (perched or hovering) are included. The distance (m) at which I set the species-specific buffer is indicated with a dotted line.	77
Figure A4: Observed distances (m) for northern harriers (<i>Circus hudsonius</i>) along transects in west-central Indiana in 2018 and 2019. Only observations of birds that could potentially be hunting (perched or coursing) are included. The distance (m) at which I set the species-specific buffer is indicated with a dotted line.	78

ABSTRACT

Intensive row-crop agriculture in the Midwestern United States has radically reduced and fragmented wildlife habitat. In agroecosystems, the distribution of foraging raptors is influenced by the availability of perches and prey abundance. Cover crops are a recent trend in agriculture that could increase the abundance and distribution of raptor prey including small mammals. However, they also benefit agricultural pest species such as voles (*Microtus*), which have damaged cover-cropped soybean (*Glycine max*) fields in Indiana.

I tested the feasibility of attracting raptors, which are natural predators of voles, to cover-cropped fields by supplying artificial perches from which to hunt. Perches were erected at 3 different distances from the field edge: 50, 125, and 200m. Raptors used 82% of the perches, and perch use was greatest at 200m. However, even at peak use, my best model predicted a low probability of overall perch use for all 3 species. Although raptor perch use by itself is unlikely to control vole populations in cover-cropped fields, artificial perches could form a valuable tool as one part of an integrated pest management system.

I also assessed the habitat use of 4 species of raptors, American kestrel (*Falco sparverius*), red-tailed hawk (*Buteo jamaicensis*), rough-legged hawk (*B. lagopus*), and northern harrier (*Circus hudsonius*), testing for selection of areas near cover-cropped fields. I constructed resource selection functions within a use-availability design to evaluate raptor habitat use with a series of weighted logistic regression models at 2 scales (transect and landscape) and using 2 definitions of available points (random and constrained by hunting method). American kestrels were the only species to be strongly associated with cover-cropped agricultural fields. Across the species, random models identified potential perches, while constrained random models identified more subtle habitat preferences not included in the random models. Modeling resource selection with constrained random availability will work best for well-studied species with discrete, easily mapped habitat features.

Finally, although northern harriers are important vole predators, they were unaffected by the artificial perches and were encountered infrequently on my transects. Therefore, I collected and dissected regurgitated pellets to analyze the diets of wintering northern harriers at communal roosts in western Indiana. Additionally, because there is no uniformly adopted method for determining the minimum number of individuals (MNI) contained in a pellet, I also assessed

whether the method used to determine MNI influenced our conclusions about pellet contents between months and roosts. Pellet contents differed between years, months, and roost sites. *Microtus* was the most commonly occurring prey group at all roosts and in both years (range: 45-73%), but was encountered less frequently than reported by other studies in the Midwest. Other important prey groups included *Peromyscus* (5-16%), *Reithrodontomys* (4-14%), *Blarina* (8-18%), and birds (7-19%). Relative to other studies, shrews and *Reithrodontomys* were more important components of harrier diets in western Indiana. The 4 methods of counting MNI produced no discernible effects on our conclusions concerning pellet contents between months or roosts. Northern harrier diets in western Indiana are highly flexible and likely reflect local prey populations. Considering teeth in addition to bones may prove beneficial for pellet-based diet studies of other diurnal raptors.

CHAPTER 1. INTRODUCTION

Agriculture is a leading cause of anthropogenically induced habitat loss and species decline (Benton et al. 2003, Green et al. 2005). In the Midwestern United States, agricultural conversion of grassland and forest has radically reduced and fragmented wildlife habitat (Kremen et al. 2002, Stanton et al. 2018). In Indiana, <1% of the historic tallgrass prairie and <25% of the pre-European forest remains (Samson and Knopf 1994, Carman 2013). Intensive row-crop agriculture, primarily of corn (*Zea mays*) and soybeans (*Glycine max*), has dominated the region for >50 years (Griffith et al. 1977). Characterized by high levels of chemical inputs and monocrop rotations of 2-3 crops planted in large fields, intensive row-crop agriculture has been linked to declines in a wide array of wildlife, ranging from native bees to grassland birds (Griffith et al. 1977, Kremen et al. 2002, Swihart and Verboom 2004, Yan and Roy 2016, Stanton et al. 2018).

In recognition of the decline of native fauna, various programs, most notably the federal Conservation Reserve Program (CRP), were established to restore habitat and ecosystem functionality to regions dominated by agriculture and have benefitted a variety of wildlife, including raptors (Riffell et al. 2008, Mushet et al. 2014, Morefield et al. 2016, Otto et al. 2018, Wilson et al. 2010).

A newer agricultural practice, planting overwinter cover crops, could provide similar or greater benefits to wildlife. Promoted as a way to improve soil health in row-crop agriculture, cover crops are non-commodity crops planted following the fall harvest to improve soil and water quality by reducing soil erosion, compaction, nutrient runoff, and suppressing weed growth (Dabney et al. 2001, Villamil et al. 2006). Although a relatively recent trend, cover crops have seen increasing adoption, particularly in Indiana, where >375,000 ha of cover crops have been planted annually since 2014 (Indiana State Department of Agriculture 2019, USDA National Agricultural Statistics Service 2019).

By providing vegetative habitat through the winter and early spring, cover-cropped fields have been associated with increased abundances for native bees and birds (Ellis and Barbercheck 2015, Wilcoxen et al. 2018). However, agricultural pest species, particularly small mammals such as voles (*Microtus*), may also benefit from the increased vegetative cover. In recent years, producers in Indiana have complained of vole damage to cover-cropped soybean (*Glycine max*)

fields, which could negatively influence adoption of an otherwise beneficial agricultural practice (Fisher et al. 2014).

Raptors and other predators have the potential to assist producers in controlling vole populations. Of the raptors found in Indiana, 5 prey upon small mammals and utilize agricultural fields: American kestrel (*Falco sparverius*), red-tailed hawk (*Buteo jamaicensis*), rough-legged hawk (*B. lagopus*), northern harrier (*Circus hudsonius*), and great horned owl (*Bubo virginianus*). However, raptors face unique challenges within agroecosystems. With the exception of the northern harrier, which hunts exclusively from flight, all of the other raptor species hunt partly or mainly from perches, which are unevenly and sparsely distributed in agricultural landscapes of the Midwest. Kestrels and rough-legged hawks will hunt by hovering, but perch hunting is more energetically efficient than hunting in flight or while hovering (Rudolph 1982, Collopy and Koplin 1983). Thus, raptors are largely limited by perch availability within row-crop agroecosystems.

I examined raptor foraging ecology within the agricultural landscape of west-central Indiana. Cover-crop adoption varies in the region, with county-level estimates ranging from 3 to 28% (Indiana State Department of Agriculture 2019). I investigated habitat use of agricultural fields by diurnal raptors and assessed whether they selected cover-cropped fields. Within cover-cropped fields, I assessed whether raptors would utilize artificial perches and thus could provide a potential top-down control of in-field small mammal populations. Finally, as northern harriers were unlikely to utilize the artificial perches, I explored northern harrier diets, particularly vole consumption, at communal roost sites in west-central Indiana.

CHAPTER 2. RAPTOR USE OF ARTIFICIAL PERCHES IN COVER-CROPPED SOYBEAN (*GLYCINE MAX*) FIELDS OF WEST-CENTRAL INDIANA

2.1 Abstract

Cover crops are an increasingly common conservation practice in intensive row-crop agriculture of the Midwestern United States and can improve wildlife habitat. However, they also benefit agricultural pest species such as voles (*Microtus*), which have damaged cover-cropped soybean (*Glycine max*) fields in Indiana. I tested the feasibility of attracting raptors, which are natural predators of voles, to cover-cropped fields by supplying artificial perches from which to hunt. I assessed raptor use of artificial perches in cover-cropped fields during the winters of 2018 and 2019. Perches were erected at 3 different distances from the field edge: 50, 125, and 200m. I modeled perch use of my 3 most common species, great horned owl (*Bubo virginianus*), red-tailed hawk (*Buteo jamaicensis*), and American kestrel (*Falco sparverius*), with a logistic generalized linear mixed model. Raptors used 82% of the perches, and perch use was greatest at 200m. However, even at peak use, my best model predicted a low probability of overall perch use for all 3 species. Artificial perches can attract raptors into large row-crop fields. Sturdier perch design, extended perch availability, and greater vole populations could increase use of perches. Although raptor perch use by itself is unlikely to control vole populations in cover-cropped fields, artificial perches could form a valuable tool as one part of an integrated pest management system.

2.2 Introduction

Proactive pest management is preferred with vertebrate pest populations, which can rapidly increase and cause high levels of damage (Wilson and Whisson 1993, Smith et al. 2003). When prey populations are low, predators have the potential to limit their vertebrate prey (Andersson and Erlinge 1977, Newsome 1990, Sinclair et al. 1990, Lin and Batzli 1995, Fargallo et al. 2009). Encouraging predators could therefore serve as one aspect of vertebrate pest control programs (Pelz 2003, Antkowiak and Hayes 2004). For raptors in particular, such efforts often entail supplementing available habitat through the addition of nest boxes, platforms and artificial

perches (Forren 1981, Hall et al. 1981, Reinert 1984, Askham 1990, Kay et al. 1994, Taylor 1996, Wolff et al. 1999, Sheffield et al. 2001, Hafidzi and Mohd 2003, Kim et al. 2003, Ojwang and Ogue 2003, Antkowiak and Hayes 2004, Witmer et al. 2008, Wong and Kross 2018). Although raptors attracted by habitat manipulation struggled to control prey at high densities (Howard et al. 1985, Wolff et al. 1999), in general raptors attracted to manipulated areas were able to reduce rodent populations and associated damage (Labuschagne et al. 2016). Moreover, in Spain, vole (*Microtus arvalis*) populations near nest boxes for barn owls (*Tyto alba*) and common kestrels (*Falco tinnunculus*) occurred at lower densities than vole populations in areas without nest boxes (Fargallo et al. 2009, Paz et al. 2012). Enhancement of raptor habitat therefore holds potential as a valuable aspect of pest management programs, particularly in environments dominated by agriculture.

Globally, vertebrate pests cause more than \$1 billion in damage to crops every year (Fagerstone 2014). Although rodent pests are commonly controlled with rodenticides, such chemicals carry a recognized risk to non-target wildlife (Colvin et al. 1988). Consequently, integrated pest management programs and other ecologically beneficial controls have grown in popularity (Pelz 2003, Antkowiak and Hayes 2004). Ranging across a wide array of agricultural products, such programs frequently incorporate predation. In Chilean pine (*Pinus radiata*) plantations, habitat modification and the addition of artificial perches resulted in reduced damage by rodents when raptor densities were high (Murua and Rodríguez 1989). In Malaysian rice (*Oryza sativa*) paddies, barn owls attracted by nest boxes reduced rodent damage (Hafidzi and Mohd 2003), as did raptors hunting from artificial perches in Kenyan corn (*Zea mays*) fields (Ojwang and Ogue 2003). Barn owl occupation of nest boxes near Israeli alfalfa (*Medicago sativa*) fields also benefited producers by increasing yield and income (Motro 2011).

Unfortunately, the majority of prior studies occurred in agricultural systems very different from those of the Midwestern United States. Agriculture in the Midwest has been dominated for >50 years by intensive row-crop farming, which is characterized by large fields (mean = 33 ha in Iowa, Yan and Roy 2016) planted in monocrop rotations of 2-3 crops (Griffith et al. 1977). In recent decades increasing numbers of producers have adopted ecologically beneficial farming practices, such as conservation tillage and cover crops. Cover crops in particular have seen a rise in adoption in the past decade throughout the Midwest (Conservation Technology Information Center 2017). Within Indiana alone, producers have planted more than

375,000 ha of cover crops every year since 2014 (Indiana State Department of Agriculture 2019). Throughout the region, cover crops generally are planted following the fall harvest to reduce chemical inputs and improve yield while improving soil and water quality through suppression of weed growth and reduction of soil erosion, compaction, and nutrient runoff (Dabney et al. 2001, Villamil et al. 2006). In addition to the benefits they provide to soil and water quality, cover crops also benefit wildlife by providing habitat over the winter and early spring. Cover crops increase diversity of reptiles (Carpio et al. 2017) and native bees (Ellis and Barbercheck 2015), as well as host higher bird abundance than fields without cover crops (Wilcoxon et al. 2018).

However, cover crops can also benefit agricultural pests such as voles. In recent years, producers in Indiana complained of high levels of vole damage to cover-cropped soybean (*Glycine max*) fields following wet springs and delayed planting (Fisher et al. 2014; J. Rorick, Agronomist, Conservation Cropping Systems Initiative, pers. comm.). Although many Midwestern raptors are effective vole predators, most hunt from perches, which are sparse in the region's row-crop agroecosystems. Perches in these systems are largely confined to telephone poles, woodlots, fencerows, or signs. While some raptors, like the American kestrel (*Falco sparverius*), will hunt by hovering and are not strictly constrained by perch availability, perch hunting is more energetically efficient than hunting in flight or while hovering (Rudolph 1982, Collopy and Koplin 1983). There is disagreement over which hunting technique is more successful, but in California kestrels hunted from perches more than 70% of the time (Mills 1979, Rudolph 1982, Collopy and Koplin 1983). Consequently, the placement of artificial perches within cover-cropped fields could increase easily accessible hunting habitat and encourage raptors to hunt previously unexploited field interiors.

To assess feasibility and provide recommendations to producers concerning artificial perch use and placement, I implemented a study with 3 objectives: 1) determine whether raptors would use artificial perches in intensive row-crop agricultural fields of Indiana, 2) identify which species would use artificial perches, and 3) determine if a perch's distance from the field edge influenced use by raptors.

2.3 Methods

2.3.1 Study Area

I conducted my study over 2 years in 17 agricultural fields (8 in 2018 and 9 in 2019) within a 50-km radius of West Lafayette, Indiana, USA. Agriculture in west-central Indiana is dominated by a corn-soybean rotation. Study fields ranged in size from 15.8-86.3 ha and were planted with a cereal rye (*Secale cereale*) cover crop followed by soybeans in the spring. Although producers plant a variety of cover crop species and mixes, I restricted my study to fields planted with cereal rye because this is the most commonly used cover crop in the study area (Conservation Technology Information Center 2017). Each field had a history of raptor sightings, vole damage, or both, and contained a permanent habitat feature that could serve as year-round rodent habitat, such as a grass waterway, treeline, or woodlot.

2.3.2 Experimental and Perch Design

I erected three 3-m perches (n=51) in each field. I designed the perches to be affordable and easily removed so that farmers could place them in the field following the planting of cover crops and remove them before cover crop termination in the spring. I constructed each perch by securing a 5 x 30-cm dowel rod to the top of a 2.5-cm diameter aluminum conduit using a PVC t-connector and secured each perch to a t-post with hose clamps. Total material cost was \$25 per perch. To test whether raptor use of artificial perches differed based on a perch's distance from the edge of a field, I placed perches at 3 distances (50, 125, and 200 m) from a permanent habitat feature. Perches were available to raptors in the field from 8 February to 24 April 2018, and 15 January to 16 May 2019. I removed all equipment from fields before cover crop termination in the spring. I was dependent on producer's planting schedules; hence, not all perches were in the field for the entire period.

2.3.3 Data Collection

I monitored raptor use of each perch with Reconyx PC900 Hyperfire Professional IR (Reconyx, Holmen, WI, USA) and Browning Strike Force HD XV (Browning, Birmingham AL, USA) trail cameras placed 2-5 m from the perch. To avoid offering raptors an alternative perch

site, I mounted cameras to ground-based platforms facing north at a 45° angle toward the perch. Camera images were collected from 25 February to 24 April 2018 and for the entire duration of artificial perch availability in 2019. I identified all birds that landed on perches to species, or to group (i.e., “*Buteo*”, “owl”) when identification to species was not possible due to a fogged camera lens or failed flash. I tallied perch use by calendar day for all raptors and for my 3 most common species: American kestrel, red-tailed hawk (*Buteo jamaicensis*), and great horned owl (*Bubo virginianus*). I assigned a value of 1 to days when a perch was used, and 0 to days without a perch use event. I assigned a value of NA to all days without camera imagery for a perch due to camera failure, accumulated snow, or wind damage to perches (n=392 of 3433 total camera days).

2.3.4 Statistical Analysis

I analyzed overall perch use for my 3 focal species using a logistic generalized linear mixed model with the lme4 package (Bates et al. 2015) in R version 3.6.1 (R Core Team 2019). My global model incorporated distance from field edge to perch, year, ordinal day, camera type, and species as fixed effects and study field as a random intercept. My species variable had 3 levels, 1 for each of my 3 target species. I centered ordinal day and used a quadratic term to test for seasonal trends in perch use. Thus, the global model had the form:

$$y_{perch\ use} = B_0 + B_1x_{distance} + B_2x_{year} + B_3x_{ordinal\ day} + B_4x_{ordinal\ day}^2 + B_5x_{camera} + B_6x_{species}$$

I removed any variables from my global model that did not significantly reduce deviance after testing with the Anova function from the car package (Fox and Weisberg 2019).

I also separately modeled each species’ perch use to assess whether the trends for ≥1 species were influenced by the others. My response variable was species-specific perch use, and my global model included perch distance, year, centered ordinal day, and camera type as fixed effects and study field as a random intercept. I assessed model accuracy by calculating the area under the receiver operating characteristic curve (AUC) with the ROCR package (Sing et al. 2005). Because I was primarily concerned with modeling fixed effects and used the same random effect structure for all of my models, I used the marginal AIC (mAIC) to select my best model, where the lowest mAIC value indicated the model with the best fit (Säfken et al. 2018).

2.4 Results

I observed 251 perch-use events by raptors over 3433 camera days (Table 2.1). Over 2 years, raptors used 42 of 51 (82%) artificial perches. Every field had at least one instance of artificial perch use, but use varied widely by field (range = 1-58, $\bar{x} \pm SD = 14.9 \pm 14.5$, median = 13). In 2019, raptors began to use the artificial perches within 2 weeks of placement. I was unable to assess initial perch use for 2018 due to the delay in obtaining viable images.

I identified 4 raptor species using my artificial perches: American kestrels, great horned owls, red-tailed hawks, and rough-legged hawks (*Buteo lagopus*, Table 2.1). Of the 4 species, great horned owls most frequently used the perches (Table 2.1). Perch use by rough-legged hawks was too infrequent to warrant inclusion in statistical modeling (Table 2.1). In addition to raptors, I captured imagery of 8 non-target species utilizing the perches, including American robins (*Turdus migratorius*), eastern bluebirds (*Sialia sialis*), eastern phoebes (*Sayornis phoebe*), European starlings (*Sturnus vulgaris*), brown-headed cowbirds (*Molothrus ater*), red-winged blackbirds (*Agelaius phoeniceus*), mourning doves (*Zenaidura macroura*), and American crows (*Corvus brachyrhynchos*).

When testing species-specific trends in perch use, American kestrel displayed the weakest unimodal trend in perch use throughout the season ($\chi^2_1 = 3.84$, $p = 0.06$). Since the quadratic term was nearly significant at the $\alpha = 0.05$ level, and the AIC-best models for red-tailed hawk and great horned owl supported my conclusions from the model for all species, I focused my analysis solely on the all-species model.

My AIC-best model for all species incorporated terms for centered ordinal day and its square, perch distance, species, an interaction between centered ordinal day and species, and study field, and had an AUC value of 0.83 (Table 2.2). Perches at 125 m on average were not used more (1.13, 95% CI = 0.75-1.75) than 50-m perches. However, the 200-m perches on average were used 1.71 times more (95% CI = 1.17-2.49) than the 50-m perch (Table 2.2). The random field intercept provided a modest improvement to model fit (marginal $R^2 = 0.36$, conditional $R^2 = 0.49$). The interaction between centered Julian day and species was significant, with great horned owl ($z = -5.08$, $p < 0.001$) and red-tailed hawk ($z = -3.63$, $p < 0.001$) perch use peaking in early and mid-March, respectively (Fig. 2.1). Great horned owls had a higher predicted probability of peak perch use than either red-tailed hawks or American kestrels, both of which had similar probabilities of peak perch use (Fig. 2.1).

2.5 Discussion

My results indicate that raptors will use artificial perches in row-crop fields of intensive agroecosystems. Raptors preferentially used perches located 200 m from field edges, the maximum distance into field interiors I tested. Prior to installation of artificial perches, raptor activity in my study fields was largely limited to species that hover-hunt, such as American kestrels or rough-legged hawks, or species that hunt while in flight, such as northern harriers (*Circus hudsonius*) or short-eared owls (*Asio flammeus*). Of these species, only kestrels were commonly observed in the study area (Chapter 3). Before the study, the remaining winter raptor species were largely limited to field peripheries where perch sites such as trees, utility lines, and fencerows are most common. The rapid discovery and use of my artificial perches indicates that habitat management strategies can encourage raptors. Unlike agriculture in southeast Ohio where raptors primarily hunted smaller fields with wooded borders (Bildstein 1978), the installation of artificial perches in large (>16 ha) row-crop fields greatly expanded the area that is easily hunted by all raptors, providing access to previously unexploited prey populations in field interiors.

I observed seasonal trends in perch use among my 3 focal species. These seasonal patterns may be the result of weather or interspecific variation in breeding phenology and diet. The majority of snow accumulation in my study occurred from January through March, which may have discouraged perch use by diurnal raptors like red-tailed hawks and American kestrels that predominantly rely on vision to locate and capture prey. In addition to sight, great horned owls hunt by sound, which enables them to more easily hunt snow-covered fields (Marti 1974). Temperature and snow accumulation may be especially important factors for American kestrels. Mills (1975) observed a decline in kestrels in south-central Ohio following a drop in temperature and snow accumulation. There were twice as many days with snow accumulation >2.5 cm in 2019 as compared to 2018, all of which occurred in January and February (Cifelli et al. 2005, data obtained from the Midwestern Regional Climate Center, cli-MATE, accessed 20 September 2019). During this time, I only observed great horned owls on the perches until the middle of February.

Differences in breeding phenology also may have affected perch use. Peak perch use by all 3 species coincided with their respective clutch initiation dates. Clutch initiation dates for great horned owls range from January through mid-March (Holt 1996, Artuso et al. 2013), for

red-tailed hawks from mid to late March (Preston and Beane 2009), and for American kestrels from late March through late June with a peak in late April (Smallwood and Bird 2002). Prior to clutch initiation, pairs must establish and defend their territory, and may use the perches to do so. However, once incubation begins, the female is largely tied to the nest, reducing the number of raptors available to use perches. As agricultural fields typically have lower small mammal abundances than neighboring habitat, they may not provide the most efficient foraging habitat, particularly given the increased energetic demands placed upon the male (Forren 1981, Adams and Geis 1983, Artuso et al. 2013). Raptors in Argentina (Bellocq 1988) and France (Meunier et al. 2000) preferentially hunted field borders and road verges more than agricultural fields. With higher prey abundances in open habitats that adjoin fields, foraging raptors may experience greater success rates hunting there (Forren 1981, Bellocq 1988).

Perch use by American kestrels may also have been influenced by diet. Vegetation height is an important factor influencing hunting habitat of diurnal raptors, which prefer sparser, shorter vegetation (Craighead and Craighead 1956, Bechard 1982, Bildstein and Collopy 1987, Preston 1990, Sheffield et al. 2001). Cover crop growth usually exceeds these preferences in the weeks just before spring termination. However, unlike red-tailed hawks and great horned owls, kestrels incorporate invertebrates into their diet during the spring and summer (reviewed in Sherrod 1978, Guerrero González 1991). Insects could therefore continue to attract kestrels to perches and encourage their use later into the spring, even if cover crops precluded easy foraging by red-tailed hawks and great horned owls.

Despite my success in attracting raptors to artificial perches in intensive row-crop agricultural fields, the overall probability of perch use was low and likely insufficient to appreciably reduce vole populations in fields. Low perch use may reflect low raptor densities on my study site, a common problem noted in agricultural regions of Italy and Argentina and associated with poor rodent control (Bellocq 1988, Boano and Toffoli 2002). The importance of adequate densities of raptors for rodent population control has been documented in Chilean pine plantations (Murua and Rodríguez 1989), Argentine cereal fields (Bellocq and Kravetz 1990), and Malaysian rice paddies (Hafidzi and Mohd 2003). Moreover, perch use varied widely by field in my study. It is possible that fields with higher usage fell within defended territories. Future studies should investigate the influence of perch locations within known territories on perch use.

Improvements in my design could potentially increase raptor perch use and thus predation pressure on voles. I only included 3 perches within each field; provisioning perches in higher numbers could increase perch adoption by providing alternative sites. In at least some of my fields, too few perches may have been located in areas with high rodent abundance or good hunting habitat. In 2019 in particular, several fields flooded near and around my perches, which likely had a negative impact on small mammal populations and perch use. Additional perches could enhance predation by providing greater coverage of the fields and potentially higher incidences of perch use (Howard et al. 1985, Askham 1990, Bellocq and Kravetz 1990, Kay et al. 1994, Kim et al. 2003).

Raptors typically choose the highest perches available to them, so increasing the height of my perches could also encourage greater use. American kestrels and great horned owls used 2.5 m perches in the absence of 5 m perches, but when perches of these heights were paired they preferentially used the taller perches (Hall et al. 1981), a pattern also observed at artificial perches on reclaimed strip mines in West Virginia (Forren 1981). At 3 m, my perches were on the shorter end of perch heights from other studies, where artificial perches have ranged from 1.5 m (Kim et al. 2003) to 9 m (Witmer et al. 2008). Natural perch sites can range even higher (Leyhe and Ritchison 2004), and mean perch heights are typically between 6-11 m and 5-8 m for red-tailed hawks and American kestrels, respectively (Bildstein 1978, Worm et al. 2013). Thus, despite the successful implementation of 3-m perches in other study areas (Kay et al. 1994, Wolff et al. 1999, Sheffield et al. 2001, Muñoz-Pedreros et al. 2010), increasing the height of artificial perches to 5 or 6 m may encourage additional perch use. However, in consideration of installation, maintenance, and winter weather severity, I would not recommend increasing artificial perch height beyond 6 m in the Midwest.

If producers adopted perches taller than 3 m, they would need to be sturdier than my design. I attached my perches to t-posts buried 45-60 cm, but taller perches would need to be anchored deeper. In 2019 there were several extreme freeze-thaw cycles that heaved my t-posts, reduced perch stability, and necessitated reburial in 2 fields. Additionally, utilizing poles of larger diameter or with thicker walls would improve durability. In 2019, there were 2 winter storms with winds >70 km/h that bent several perches. Using wood or thicker poles may have improved the perch's ability to withstand these winds. One potential solution to increasing artificial perch height and sturdiness would be to create permanent perch sites. Permanent

perches would not need to be removed, and could therefore be more robust than perches that would need to be installed and removed every year. Siting these perches in grass waterways would not affect crop acreage, but would enable raptors to hunt agricultural fields year-round, including during planting, when seeds and shoots are most vulnerable to vole predation (Fisher et al. 2014). Grass waterways can extend into field interiors, and are a potential source of in-field vole populations. Areas adjacent to permanent cover tend to experience higher levels of damage (White et al. 2003) and would benefit most from raptor predation. If permanent perches are not feasible for producers to implement, installing removable artificial perches immediately following the cessation of post-harvest in-field activities would also be valuable. Raptors had the biggest impact on rodents in fall and early spring in agricultural systems of Argentina (Bellocq 1988), and had the greatest impact during fall in an Illinois old-field habitat (Lin and Batzli 1995).

In addition to artificial perches, producers can encourage higher predation rates by adding nest boxes to attract breeding raptors as well as encouraging mammalian vole predators such as coyotes (*Canis latrans*), red fox (*Vulpes vulpes*), and weasels (*Mustela* sp.; Korschgen 1957, Korschgen 1959, Lin and Batzli 1995, Paz et al. 2012). Furthermore, producers could directly alter vole habitat by changing their cover-crop species and seeding rates. By selecting less palatable species (Prieur 2019) or reducing overhead cover, fields could be made less enticing to voles and easier for predators to hunt (Van Vuren and Smallwood 1996, Pelz 2003). Although unlikely to reduce high vole populations, raptor predation facilitated by provisioning of artificial perches, when combined with other management strategies such as habitat alteration and nest box installation, may assist producers in reducing vole populations and their associated damage.

Table 2.1: Raptor artificial perch use for 2018 and 2019 in 17 cover-cropped soybean (*Glycine max*) fields in west-central Indiana.

Species	2018	2019	Total
American kestrel	28	8	36
Great horned owl	69	73	142
Red-tailed hawk	19	28	47
Rough-legged hawk	5	0	5
<i>Buteo</i> sp.	0	1	1
Owl sp.	14	6	20
Total	135	116	251

Table 2.2: Parameter estimates from the AIC-best GLMM for raptor use of artificial perches in cover-cropped soybean (*Glycine max*) fields in west-central Indiana, USA. Factor variable coefficients are given in comparison to their reference level (Distance = 50m; Species = American kestrel; Interaction = American kestrel:Day).

Independent Variable	Estimate	SE	z-value	p-value
Fixed Effects				
Intercept	-5.52	0.40	-13.95	<0.001
Distance				
125 m	0.12	0.21	0.58	0.554
200 m	0.54	0.19	2.82	0.004
Ordinal Day (centered)				
Monomial term	1.55	0.36	4.24	<0.001
Quadratic term	-0.70	0.13	-5.42	<0.001
Species				
Great horned owl	2.03	0.32	6.41	<0.001
Red-tailed hawk	0.88	0.34	2.54	0.009
Interaction				
Great horned owl:Day	-2.03	0.41	-4.99	<0.001
Red-tailed hawk:Day	-1.57	0.44	-3.56	<0.001
Random Effects				
Field	Variance	SD		
	0.81	0.90		

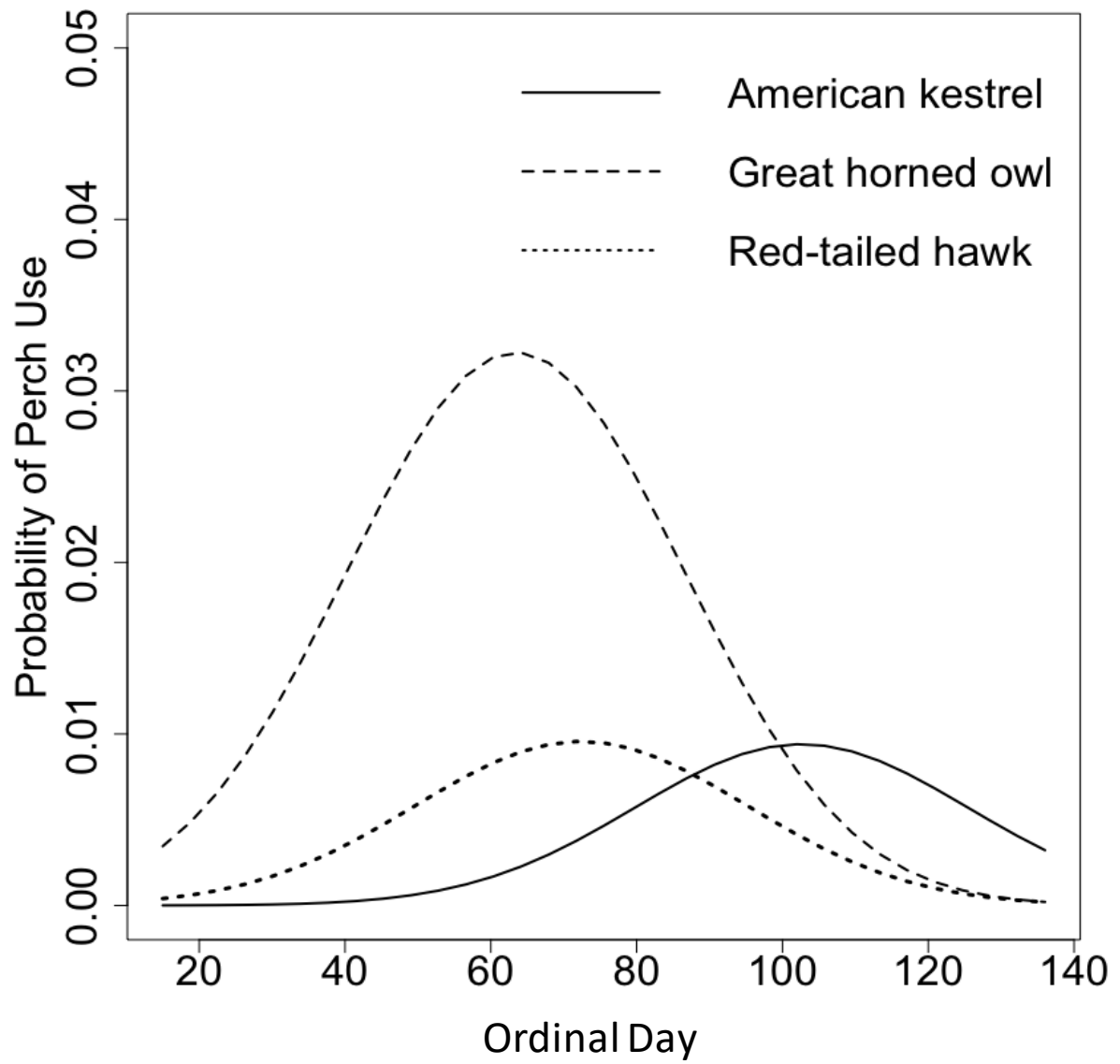


Figure 2.1: Interaction between ordinal day and estimated species-specific use of artificial perches in cover-cropped soybean (*Glycine max*) fields of west-central Indiana, USA.

CHAPTER 3. RAPTOR WINTER HABITAT USE IN AGRICULTURAL LANDSCAPES OF WEST-CENTRAL INDIANA

3.1 Abstract

The distribution of foraging raptors in agroecosystems is influenced by the availability of perches and prey abundance. Cover crops are a recent trend in areas of intensive row-crop agriculture that could increase the abundance and distribution of raptor prey including small mammals. I assessed the habitat use of 4 species of raptors, American kestrel (*Falco sparverius*), red-tailed hawk (*Buteo jamaicensis*), rough-legged hawk (*B. lagopus*), and northern harrier (*Circus hudsonius*), and specifically tested for selection of areas near cover-cropped fields. I conducted 1184 km of roadside transects in 2018 and 2019 in west-central Indiana and recorded 208 detections of my focal species. I constructed resource selection functions within a use-availability design to evaluate raptor habitat use with a series of weighted logistic regression models. For each species, I fit models at 2 scales (transect and landscape) and with 2 definitions of available points (random and constrained by hunting method). American kestrels were strongly associated with cover-cropped agricultural fields. Red-tailed hawks were strongly associated with woodlots, and rough-legged hawks were weakly associated with permanent habitat and avoided woodlots. Northern harrier models exhibited poor classification accuracy. Across the species, random models identified potential perches, whereas constrained random models identified more subtle habitat preferences not included in the random models. Modeling resource selection with constrained random availability will work best for well-studied species with discrete, easily mapped habitat features. If damage to commodity crops by rodents in cover-cropped fields is a concern, management of raptors should focus on kestrels, and could include the erection of artificial perches, nest boxes, and improving the quality of permanent hunting habitat.

3.2 Introduction

In the Midwestern United States, conversion of grassland and forest, primarily due to agriculture, has dramatically reduced and fragmented permanent habitat for wildlife (Kremen et al. 2002, Stanton et al. 2018). For instance, in Indiana, the current extent of tallgrass prairie and

forests covers <1% and < 25% of their pre-European extent respectively (Samson and Knopf 1994, Carman 2013). Intensive row-crop agriculture has dominated the region for >50 years and is characterized by high levels of chemical inputs and monocrop annual rotations of 2-3 crops planted in large fields (Griffith et al. 1977, Yan and Roy 2016). Since 1985 the federal Conservation Reserve Program (CRP) has attempted to ameliorate habitat losses in agroecosystems by taking enrolled land out of production to be managed as restored permanent cover over the course of the contract (Morefield et al. 2016).

In agroecosystems raptor distributions are influenced by the availability of perches and prey abundance, with raptor abundances typically declining as agriculture becomes more intensive (Boano and Toffoli 2002, Filloy and Bellocq 2007, Butet et al. 2010, Grande et al. 2018 and sources therein). CRP and related programs have benefited raptors and other wildlife (Best et al. 1997a, Best et al. 1997b, Riffell et al. 2008, Mushet et al. 2014, Otto et al. 2018, Wilson et al. 2010). Unfortunately, CRP enrollment has dropped in recent years, and conversion to cropland has increased (Morefield et al. 2016).

Cover crops may offset negative effects experienced by wildlife due to declining CRP enrollment, especially in winter. Cover crops are non-commodity crops usually planted after the fall harvest to improve soil drainage and reduce soil compaction, erosion, nutrient loss, and weed growth (Dabney et al. 2001, Villamil et al. 2006). An additional benefit of cover crops is the vegetative habitat they can provide to wildlife through the winter and early spring, with increased abundances documented for native bees and birds (Ellis and Barbercheck 2015, Wilcoxen et al. 2018). Use of cover crops has grown in the past decade, and Indiana ranks 3rd in the United States in terms of cover-crop adoption with >375,000 ha of cover crops planted annually since 2014 (Indiana State Department of Agriculture 2019, USDA National Agricultural Statistics Service 2019).

Cover crops also likely encourage use of farm fields by voles (*Microtus*) and other small mammals that are important prey for raptors (Jug et al. 2008, Fisher et al. 2014). In particular, voles use wheat (*Triticum aestivum*) and alfalfa (*Medicago sativa*) fields (Babińska-Werka 1979, Getz and Brighty 1986, Kaufman and Kaufman 1990) and are a principal prey of many raptors (Craighead and Craighead 1956, Baker and Brooks 1981). Common cover crops such as cereal rye (*Secale cereale*) provide a similar vegetative profile that may promote greater abundances of

voles relative to conventional fields (Conservation Technology Information Center 2017). Hence, cover-cropped fields may contain higher densities of small mammal prey.

I investigated winter selection of hunting habitat in west-central Indiana by four raptors that hunt in open areas of Midwestern agroecosystems (Bildstein 1978): American kestrel (*Falco sparverius*), northern harrier (*Circus hudsonius*), red-tailed hawk (*Buteo jamaicensis*), and rough-legged hawk (*B. lagopus*). My primary objective was to test whether raptors preferentially use areas in proximity to cover-cropped fields. Raptors typically forage where prey densities are highest (Baker and Brooks 1981, Preston and Beane 1996). Thus, if raptor responses are driven by prey density, raptor use should be greater in areas proximal to cover-cropped fields and other areas of high-quality habitat for prey. Alternatively, if dense vegetation discourages raptor habitat use (Craighead and Craighead 1956, Bechard 1982, Preston 1990), selection of more exposed sites with greater ease of prey capture is predicted.

The four focal species differ greatly in their modes of hunting. At one extreme, red-tailed hawks hunt primarily from perches (Preston 1990, Preston and Beane 2009), whereas northern harriers course open fields while hunting on the wing (Vukovich and Ritchison 2008, Smith et al. 2011). In between these extremes, American kestrels and rough-legged hawks hunt using perches and hovering, with the mode dependent on prey density, vegetation height, and wind (Bildstein 1978, Bildstein and Collopy 1987, Bechard and Swem 2002). Thus, a secondary objective was to evaluate whether explicitly incorporating into analyses constraints imposed by raptor hunting methods can affect conclusions regarding factors that influence site selection.

3.3 Methods

3.3.1 Study Area

I surveyed agricultural landscapes with roadside transects in a 10-county area of central Indiana (Fig. 3.1). These counties are characterized by intensive row-crop agriculture with 60-90% of land planted to either corn or soybeans (USDA National Agricultural Statistics Service Cropland Data Layer 2019). Cover-crop adoption ranged from 2 to 28% in the surveyed counties (Indiana State Department of Agriculture 2019). I did not identify cover crops to species, but the 5 most commonly planted winter-hardy cover crops in the area were cereal rye,

rapeseed (*Brassica napus*), winter wheat, annual ryegrass (*Lolium multiflorum*), and crimson clover (*Trifolium incarnatum*, Conservation Technology Information Center 2017).

3.3.2 Data Collection

Transects

I designed my transects in Google Earth Pro 2018 (Google, Mountain View, CA), attempting to maximize the surrounding agriculture landscape while also maintaining routes that were as straight as possible to facilitate aerial imaging flights. Where possible, I limited routes to secondary roads. In 2019, I repeated 13 of my 14 original transects; one transect was partially rerouted to avoid a construction zone, and one with low agricultural landcover was dropped. I added 2 routes each in counties with 11% (Pulaski) and 29% (Miami) cover crop adoption in 2017, respectively (Indiana State Department of Agriculture 2019). These additional routes were designed after consulting local NRCS offices and 2018 Sentinel-2 (ESA) and Landsat-8 satellite imagery. Transects were driven once in a given year and averaged 38 km (range: 29-47 km).

I conducted surveys by driving transects from 14 January to 8 April 2018 and 21 January to 2 April 2019. To limit temporal bias in 2019, I attempted to survey any of my repeated 2018 routes at similar dates and supplemented the 4 additional transects throughout the season. I surveyed for raptors beginning 1 hour after sunrise until routes were completed on days where wind was at or below level 4 on the Beaufort Scale and precipitation was no heavier than a light flurry or drizzle (Fuller and Mosher 1981, 1987). Other studies have ranged widely in the timing of their transects, but I elected to conduct morning surveys because I was interested in modeling habitat use at times when birds were more likely to be perched rather than soaring (Fuller and Mosher 1981, Bunn et al. 1995). I drove between 16-24 kph and had 2 observers counting raptors, with the primary observer scanning both sides of the road and the secondary (driving) observer scanning in front of the vehicle and occasionally on the driver's side (Craighead and Craighead 1956, Fuller and Mosher 1981). I counted all raptors seen along the routes, using a pair of 10x42 Leupold and Stevens binoculars (Beaverton, OR, USA) to confirm identifications. When I saw a raptor, I stopped to note the species, behavior, perching substrate, perpendicular distance (Prostaff 7i rangefinder, Nikon, Tokyo, Japan), and GPS coordinates from the road (GPSmap 78s Garmin Ltd., Olathe, KS).

Aerial Photography

Due to the ephemeral nature of cover crops, I obtained aerial imagery of cover crops planted along transects. I timed my flights to coincide with maximum cover-crop growth just before termination: 19-20 April 2018 and 24-28 April 2019. Transects were flown in a Cessna 172P Skyhawk at an altitude of 2000 m with clear skies or few clouds. A 2000-m altitude allowed me to classify landcover within 1 km on either side of transect routes. In 2019, 5 transects were flown at 1800 m due to a low cloud ceiling, but without adverse effects on landcover classification. All images were taken with a FinePix HS50EXR (Fujifilm, Tokyo, Japan) on the landscape setting from the belly of the plane.

3.3.3 GIS Analysis

Using the aerial imagery, I digitized landcover within 1 km of each transect in ArcGIS Pro (ESRI, Redlands, CA). I assigned landcover to 5 classes (Table 3.1): agricultural fields with cover crops (Cover), agricultural fields without a cover crop (NoCover), permanent habitat, including CRP strips, large grass waterways, and remnant grasslands (PermHab), woodlots (Woods), and developed areas (Dev). In addition to traditional cover crops terminated in spring, I included alfalfa and wheat in my cover crop category. These crops were not commonly planted along my transects ($\leq 1\%$), but they serve the same ecological function as cover crops by providing overhead cover to small mammals.

In addition to landcover, I digitized 3 habitat features that could serve as potential perches for raptors (Table 3.1), including linear rows of trees (Treeline), the outer canopy of trees in yards and pastures (Treeperim), and utility lines (Uline). I digitized utility lines based on Google Earth imagery; all of the remaining features were digitized from the aerial imagery. For all features, I only digitized those elements that would have been visible to observers from the road and thus available to survey for raptor presence. For example, for the canopy of trees in yards/pastures, I only digitized the portion of the canopy facing the road, and excluded any trees that would have been hidden by houses or woodlots.

To model habitat use for each of my 4 target species, I buffered each transect by the perpendicular distance at which detections for that species began to decline based on my observations: 250 m for red-tailed and rough-legged hawks, 150 m for northern harriers, and 75

m for American kestrels (Figs A1-A4). I excluded any observations beyond a species' respective buffer. For red-tailed hawks, rough-legged hawks, and American kestrels, I also excluded any individuals that were not perched or hover-hunting. Perched and hovering individuals were assumed to be actively hunting, an assumption I could not make for birds that were flying through or soaring. Northern harriers are aerial hunters, and all of the observed individuals were coursing or perched on the ground. For paired raptors, I only included the individual that was observed first.

3.3.4 Statistical Analysis

I adopted a use-availability design (Johnson et al. 2006) to develop models of resource selection by raptors as a function of distance (m) to each of the measured landcover and habitat features (Table 3.1, Conner et al. 2003). Estimates derived from logistic regression for use-availability data converge to an equivalent log-linear inhomogeneous point process model if the number of available points is sufficiently large or infinite weights are assigned to all available points (Warton and Shepherd 2010, Fithian and Hastie 2013). Following Muff et al. (2019), I standardized all explanatory variables (Table 3.1) and weighted available points by a factor of 1000 (Muff et al. 2019). When detections are rare, as in my study, the random selection of a large number of available points can improve model accuracy (Lobo and Tognelli 2011, Nad'oo and Kaňuch 2018). Thus, I selected 1000 random points for each model set; this was the sample size at which mean distances from available points to covariates tended to stabilize (Benson 2013).

I fit a series of weighted logistic regression models for each focal species in R version 3.6.1 (R Core Team 2019). First, I fit a global model to compare with my set of candidate models. I then compared all possible models containing p or fewer explanatory variables using package MuMIn (Shoemaker et al. 2018, Bartoń 2019). Due to the relatively low number of detections among all species, I guarded against overparameterizing models by fitting a maximum of $p = 4$ explanatory variables to models for red-tailed hawks ($n=72$ detections) and American kestrels ($n=43$) and a maximum of $p = 2$ variables to my models for northern harriers ($n=8$) and rough-legged hawks ($n=7$). I inspected in more detail candidate models within 2 AICc units of the top model. To assess contributions of explanatory variables to top models, I conducted

analysis of deviance using the car package (Fox and Weisberg 2019). Residual diagnostics were assessed with the DHARMA package (Hartig 2019). I evaluated model classification accuracy using area under the receiver-operating curve (AUC). AUC may yield unrealistically high assessments of model performance for use-availability data with a low proportion of detections relative to available points (Sofaer et al. 2019). Thus, I also computed normalized area under the precision-recall curve (AUC-PR) using R package PRROC (Grau et al. 2015). Normalized AUC-PR ranges from 0 (worst performance possible) to 1 (best performance possible); it offers a more robust measure of model accuracy for data with relatively few detections of use, as it adjusts for skew, can be adjusted for weighted points, and does not incorporate available (0) points that are predicted to be 0 (Boyd et al. 2012, Keilwagen et al. 2014, Sofaer et al. 2019).

To address my second objective, I assessed to what degree the manner in which available points were chosen influenced models of resource selection. Typically in use-availability or other studies that rely on detection, available points are chosen completely at random (Iturbide et al. 2015). However, detection for several of the focal raptors in my study was limited to sites with suitable perching substrates. In the context of species distribution modeling, species detections often are biased due, e.g., to greater sampling effort in more accessible areas. Although my sampling was conducted systematically on transects, the constraints imposed on my detections by perch availability presumably would not be reflected in a set of available points selected randomly from across the width of each transect. Instead, random selection of available points that reflects constraints in sampling of species detections may improve model performance (Phillips et al. 2009, Hanberry et al. 2012). Consequently, I generated 2 types of available points: completely random, and random subject to biological constraints (hereafter, constrained random). For the former type, I selected points randomly from all landcover classes within each species' respective buffer. For constrained random points, I randomly generated available points in landcover classes that could have been used by raptors and were available to be surveyed. Because red-tailed hawks, American kestrels, and rough-legged hawks hunt primarily from perches, constrained random points were selected from available perching substrates (i.e., Treeperim, Treelines, edges of woodlots, Uline, Bildstein 1978, Bildstein and Collopy 1987, Bechard and Swem 2002, Preston and Beane 2009). Because northern harriers hunt on the wing, I selected constrained random points from available open habitats (Cover, NoCover, PermHab). For each species and point type, I selected available points at each of 2 scales to mimic 2nd and

3rd-order selection (Johnson 1980): across all transects in my study landscape (1184 km), and within each transect (29 - 47 km). Thus, I considered a total of 2 types of available points x 2 scales = 4 sets of models for each species, for a total of 16 sets of models.

3.4 Results

I detected 319 raptors of 9 species over 1184 km of transects (Table 3.2). Of these, 208 belonged to my 4 target species (Table 3.2). Ninety-two percent of perched red-tailed hawks were observed in trees, and 67% of perched American kestrels were observed on utility lines. Landcover was predominantly agricultural, with most of the sampled area in fields without cover crops (range: 53.6% - 91.4%), or in fields with cover crops (range: 1.5% - 12.5%). Transects were rural, with small amounts devoted to developed areas (1.4% - 6.5%). Permanent herbaceous habitat (1.4% - 9.8%) and woodlands (0.6% - 20.9%) occurred primarily as small patches (see Table A1 for details).

Of 116 predictors included in the 16 global models, only 2 had variance inflation factors > 2 (2.4 and 3.1). Hence, multicollinearity was not a concern. For all combinations of species, scales, and types of available points, global models never received support as the AICc-best model. Indeed, in 15 of 16 model sets considered, the global model was ≥ 4 AIC units from the best model. Hence, global models were not considered further.

3.4.1 American Kestrels

For American kestrels, classification accuracy of top models was greater when available points were chosen completely at random, with AUC of 0.88-0.91 compared to AUC of 0.70 for models with available points constrained by hunting mode (Table 3.3). When available points were selected randomly at the transect scale, 5 of 163 candidate models were within 2 Δ AICc units with a collective weight of evidence of 0.44 (Table 3.3). All 5 of these models included overwhelmingly strong negative relationships with distance to utility lines and, to a lesser extent, cover-cropped fields and permanent herbaceous habitat (Table 3.4, A1). For the best model, a 1 SD increase in distance to utility lines (598 m) reduced odds of kestrel use to essentially zero, whereas 1 SD increases in distance to cover crops (1201 m) and permanent habitat (560 m) reduced odds of use to 0.48 and 0.55, respectively, relative to odds at mean values. Three of the

models also included a weak positive relationship with distance to woodlots (Table 3.3, 3.4). For available points constrained by hunting mode at the transect scale, 4 candidate models were within 2 Δ AICc units, with a collective weight of evidence of 0.37 (Table 3.3). All 4 of these models included a strong negative relationship with distance to cover-cropped and non-cover-cropped fields, and a strong positive relationship with woodlots (Table 3.3, 3.4). In the best model a 1 SD increase in distance to cover crops (1111 m) and fields without cover crops (57 m) reduced odds of use to 0.44 and 0.49, respectively.

When available points were selected randomly at the landscape scale, 5 candidate models were within 2 Δ AICc units, with a collective weight of evidence of 0.51 (Table 3.3). All 5 included strong negative relationships between detections and distance to cover-cropped fields and utility lines, with effects that were similar in magnitude to those for the corresponding model at the transect scale (Table 3.4). For available points constrained by hunting mode at the landscape scale, 10 candidate models were within 2 Δ AICc units and exhibited a collective weight of evidence of 0.61 (Table 3.3). All 10 models incorporated a strong negative relationship with distance to cover-cropped and non-cover-cropped fields, with effects similar in size to the constrained random models at the transect scale (Table 3.3, 3.4).

3.4.2 Red-tailed Hawks

For red-tailed hawks considered at both scales and for both types of available points, AICc-best models exhibited moderate classification accuracy, with AUC values of 0.69-0.77 and normalized AUC-PR values of 0.52-0.66 (Table 3.3). For available points selected randomly at the transect scale, 13 of the 163 candidate models were within 2 Δ AICc units, with a collective weight of evidence for the AICc-best model of 0.5 (Table 3.3). All 13 models included a negative relationship with distance to woodlots. A negative effect of distance to tree perimeter occurred in 11 models including the top 3, and a negative relationship with distance to permanent habitat occurred in 6 of the 13 models (Table 3.3, 3.4, A1). Relative to mean values, a 1 SD increase in distance to woods (191 m) and tree perimeter (241 m) reduced odds of use to 0.20 and 0.67. For available points constrained by hunting mode at the transect scale, 11 candidate models were within 2 Δ AICc units, with a collective weight of evidence of 0.46 (Table 3.3). All 11 top models included strong positive relationships between use and distance to utility lines,

while 9 of the top models included a strong positive relationship with development, and 8 of the top models included a strong negative relationship with woodlots (Table 3.3, 3.4). Relative to mean values, a 1 SD increase in these 3 variables (162, 115, and 566 m, respectively) for the best model resulted in odds of 1.19, 1.22, and 0.64, respectively.

For available points selected randomly at the landscape scale, 15 candidate models were within 2 Δ AICc units, with a collective weight of evidence of 0.56 (Table 3.3). A strong negative relationship with distance to woodlot was found in all 15 models. In contrast to the transect scale, a strong negative relationship with distance to development was included in 13 of the 15 models (Table 3.3, 3.4). For available points constrained by hunting mode at the landscape scale, 13 candidate models were within 2 Δ AICc units, with a collective weight of evidence of 0.53 (Table 3.3). Similar to the transect scale, all top models included a strong positive relationship with distance to utility lines, and 12 of 13 models included a strong negative relationship with distance to woodlots (Table 3.3, 3.4).

3.4.3 Rough-legged Hawks

For rough-legged hawks, classification accuracy of top models was moderate for both scales and types of available points, with AUC of 0.65-0.70 and normalized AUC-PR of 0.52-0.64 (Table 3.3). When available points were selected randomly at the transect scale, 5 of the 29 candidate models were within 2 Δ AICc units with a collective weight of evidence of 0.43 (Table 3.3). The top model included a weak negative relationship with distance to permanent herbaceous habitat (Table 3.4), and the 3rd-best model was an intercept-only model. For available points constrained by hunting mode at the transect scale, 11 candidate models were within 2 Δ AICc units and collectively had a weight of evidence of 0.67 (Table 3.3). The top 2 models both included strong positive relationships with distance to woodlots; for the top model, a 1 SD increase in distance to woods increased odds of use 1.63-fold relative to the mean distance (Table 3.4). Random available points at the landscape scale yielded 11 candidate models within 2 Δ AICc units and a collective weight of evidence of 0.68 (Table 3.3). Although both of the top 2 models included a negative relationship with distance to permanent habitat, this relationship was only strong in the 2nd-best model (Table 3.4). When available points were constrained by hunting mode at the landscape scale, 12 candidate models were within 2 Δ AICc

units with a collective weight of evidence of 0.68 (Table A1). Half of these models, including the top 2 models, included a positive relationship with development; for the top model a 1 SD increase in distance to development (135 m) increased odds of use 1.52-fold relative to the mean distance. Negative relationships with distance to permanent habitat also occurred in 3 of the top 12 models (Table 3.3, 3.4).

3.4.4 Northern Harrier

For northern harriers, classification accuracy of top models was poor for both scales and types of available points, with AUC of 0.50-0.58 and normalized AUC-PR of 0.28-0.34 (Table 3.3). Regardless of scale or type of available points, 8-9 of the 22 candidate models were within 2 Δ AICc units and exhibited collective weight of evidence of 0.56-0.66 (Table 3.3). Only distance to non-cover-cropped fields emerged as a variable consistently associated with northern harrier use, and its effect tended to be weak (Table 3.4, A1).

3.5 Discussion

Among my 4 focal species, only American kestrels exhibited selection for hunting near cover-cropped fields. Northern harriers tended to select non-cover-cropped fields, but these models had poor classification accuracy likely owing in part to the limited number of detections. Available perches and preferred non-agricultural habitats were more important than cover crops in determining the distributions of red-tailed and rough-legged hawks. In light of these results, producers could encourage further kestrel hunting of cover-cropped fields by erecting nest boxes and artificial perches as well as managing permanent habitat (Chapter 2, Toland and Elder 1987, Valdez et al. 2000). Supplementing available perches, particularly in the field interior, could increase the area kestrels can easily hunt, while attracting a breeding pair could increase predation pressure during planting and germination when voles consume seeds and new sprouts (Fisher et al. 2014, Prieur 2019). Moreover, managing permanent herbaceous habitats to increase invertebrate diversity could provide kestrels with more insect prey and increase crop pollination services (Clark et al. 2005, Marshall et al. 2006, Hopwood 2008, Garibaldi et al. 2014, Evans et al. 2016).

American kestrel distributions were driven largely by access to hunting habitat. Although American kestrels in Kentucky avoided crop fields (Andres 1994), kestrels were associated with both cover-cropped and non-cover-cropped fields in all of my top constrained availability models. Unlike Kentucky, permanent herbaceous habitat in my study area was sparse, and much of it was positioned along roads. The inclusion of permanent habitat in several of my top kestrel models was noteworthy, as roadsides can serve as important habitat for grassland invertebrates and small mammals, especially in intensively agricultural regions (Getz et al. 1978, Adams and Geis 1983, Adams 1984, Evans et al. 2016). During my survey period, cover-cropped fields had a similar vegetative profile to roadside habitats and thus could have facilitated small mammal dispersal into the fields, attracting kestrels. Untilled fields without cover crops also likely harbor prey available to kestrels. Within the surveyed counties, an estimated 26-83% of corn and 51-90% of soybean fields were untilled following the harvest and prior to my surveys (Indiana State Department of Agriculture 2019). Deer mice (*Peromyscus maniculatus*) and, to a lesser extent, prairie voles (*Microtus ochrogaster*) use reduced tillage row-crop fields within my study area during winter (Berl et al. 2017), and additional species of small mammals move into fields during the growing season (Abercrombie et al. 2017, Berl et al. 2018). As we only surveyed each transect once a year, we were unable to assess whether use of cover-cropped and non-cover-cropped agricultural fields changed over the course of the winter. Future studies should therefore investigate temporal variation in habitat use.

Unlike kestrels, few of the top red-tailed hawk models incorporated agricultural landcover. Instead, red-tailed hawks associated most closely with woodlots. The association between red-tailed hawks and woodlots in the Midwest has been well documented; woodlots are used throughout the year by both hunting and nesting hawks (Ingold 2010, Bildstein 1978, Petersen 1979, Schnell 1968). In addition to defending woodlots, red-tailed hawks may encounter increased abundances of small mammals at the border between woodlots and fields (Bildstein 1978, Cummings and Vessey 1994, García et al. 1998, Šálek et al. 2010). The potential for high prey encounter rates as well as the abundance of perches along woodlot edges likely increases red-tailed hawk hunting efficiency, constraining their habitat selection primarily toward woodlots.

Most of the best models for rough-legged hawks demonstrated an avoidance of woodlots or an association with permanent herbaceous habitat, consistent with their affinity for open areas

(Craighead and Craighead 1956, Schnell 1968, Bildstein 1978, Ingold 2010, Bechard and Swem 2002). When considering constrained availability at the landscape scale, rough-legged hawks also strongly avoided development. Over the study area, permanent herbaceous habitat generally was associated with agriculture (i.e. grass waterways or CRP land) or protected areas (i.e. remnant prairies or herbaceous wetlands) with few trees, and with the exception of farmsteads, and roads, removed from development. Model classification accuracy for rough-legged hawks was low, so my findings should be viewed as hypotheses to be tested in future work with larger sample sizes.

Scale did not greatly affect the inclusion of habitat variables into top models. One exception was observed for American kestrels, which avoided woodlots at the transect scale but not at the landscape scale (Table 3.4). As North America's smallest diurnal raptor, kestrels are susceptible to predation by other raptors, including red-tailed hawks, sharp-shinned hawks (*Accipiter striatus*) and Cooper's hawks (*Accipiter cooperii*), all of which are associated with woodlots (Smallwood and Bird 2002, Farmer et al. 2006). Among wintering kestrels in Pennsylvania, avian predators were responsible for 62% of kestrel mortality (Farmer et al. 2006). Therefore, by avoiding woodlots at a scale more closely tied to territories, kestrels may have chosen foraging sites that reduced their risk of predation.

Models based on biologically constrained "available" points tended to yield worse predictive accuracy than models using available points drawn completely at random, a pattern also observed by Cerasoli et al. (2017). Reduced classification accuracy follows logically from the inclusion of prior knowledge into selection of available points. Selection of available points from the suite of available perch sites provided a more biologically relevant comparison for species that are known to hunt predominantly from perches. By extension, available points constrained in this manner more closely aligned with detections, making it more difficult for the models to classify points accurately but enabling us to identify more subtle features that influenced selection of foraging sites for some species. In particular, the method of selection of available points influenced the category (perch or landcover) of habitat variables deemed important to red-tailed hawks and American kestrels.

For kestrels, utility lines were only included in the best models created with randomly selected available points, and the magnitude of selection for this feature was 60 (transect) and 15 (landscape) times greater than for the next most important variable in these models (cover crops,

Table 3.4). Such an outcome is unsurprising when considering prior knowledge about kestrels, as they often hunt from utility lines (Andres 1994, Ingold 2010, Bobowski et al. 2014). In my study, 37 of 43 detections were from utility lines. Thus, the inclusion in models of available points chosen randomly from all possible points confirmed my understanding of the importance of utility lines as perch sites for kestrels in agricultural regions. Importantly, models that relied on available points drawn solely from possible perch sites revealed avoidance of woodland perches and selection of perch sites near fields lacking cover crops (Table 3.4). Constraint of available points to possible perch sites thus shed light on additional features of perch site use by kestrels that would not have been evident with the more conventional (completely random) approach to modeling use-availability.

The influence of competing definitions of availability was more nuanced for red-tailed hawks. In particular, the 2 methods of generating available points yielded best models that diverged in their associations with development. In constrained availability models, red-tailed hawks avoided (transect scale) or were neutral (landscape scale) to development, whereas in random availability models they were neutral (transect) or selected (landscape) development. Thus, at both scales constrained availability resulted in a stronger signal of aversion to development. Discrepancies in expected associations with development resulted from altered distributions of distances chosen with the two methods of defining availability. Mean observed distance to development was 93m. For available sites constrained to potential perches, mean distances to development were less than mean observed distance, (69m for transect, 79m for landscape), whereas mean distances for available sites selected randomly (138m for transect, 165m for landscape) were greater than the observed mean. This example thus illustrates how the method of selection of available points can impact not only the inclusion of variables, but also the direction of association. For red-tailed hawks, the constrained availability models, by virtue of selecting from other available perches, highlighted habitat associations, particularly avoidance of utility lines that were missed by the random models.

When modeling resource selection, multiple definitions of availability potentially can yield complementary insights. However, consideration of constrained availability may not be feasible for under-studied species or those species for which discrete, biologically important habitat features are not easily identified on the landscape. To place intelligent constraints on choice of available points necessitates some knowledge of the species' biology as it relates to the

resource in question. For species with a paucity of biological information, identification of potential features on which to constrain choices of availability may not be obvious, in which case availability determined completely at random is most appropriate. Even for well-studied species, constraining availability points to biologically relevant and spatially discrete habitat features can pose a challenge when these habitat features are not easily mapped. Employing constrained availability in models of resource selection will therefore work best for well-studied species that utilize spatially distinct and biologically relevant habitat features amenable to mapping, such as with some birds (e.g., perching and nesting sites), bats (e.g., roosting structures), or amphibians (e.g., vernal pools).

Table 3.1: Landcover and habitat classes used in the set of models of raptor resource selection. For all features listed, explanatory variables were distance (m) from the focal landcover or habitat class. The variables considered for each species are noted (+). Species abbreviations: AMKE = American kestrel (*Falco sparverius*), RTHA = red-tailed hawk (*Buteo jamaicensis*), RLHA = rough-legged hawk (*B. lagopus*), and NOHA= northern harrier (*Circus hudsonius*).

Explanatory Variables		Description	Raptor Species			
			AMKE	RTHA	RLHA	NOHA
Landcover						
Cover crops	Cover	Agricultural fields planted with cover crops, alfalfa, or winter wheat	+	+	+	+
Development	Dev	Towns, buildings, large roads, etc.	+	+	+	+
No cover crops	NoCover	Agricultural fields without a cover crop, including no-till and conventional tillage fields.	+	+	+	+
Permanent habitat	PermHab	Permanent herbaceous cover, including large grass waterways, CRP strips, remnant grasslands, etc.	+	+	+	+
Woodlots	Woods	Forested lots	+	+	+	+
Habitat						
Phone lines		Telephone lines and utility wires	+	+		
Tree perimeter		The outer perimeter of the canopy of independent trees	+	+	+	
Treelines		Linear rows of trees	+	+	+	+

Table 3.2: Summary of raptor species seen on 1184 km of transects driven in west-central Indiana in January-April 2018 and 2019.

Raptor Species		2018	2019	Total	Included in model [†]
American kestrel*	<i>Falco sparverius</i>	24	36	60	43
Red-tailed hawk*	<i>Buteo jamaicensis</i>	43	88	131	72
Rough-legged hawk*	<i>Buteo lagopus</i>	5	2	7	7
Northern harrier*	<i>Circus hudsonius</i>	6	4	10	8
Turkey vulture	<i>Cathartes aura</i>	38	48	86	--
Cooper's hawk	<i>Accipiter cooperii</i>	--	9	9	--
Bald Eagle	<i>Haliaeetus leucocephalus</i>	7	2	9	--
Red-shouldered hawk	<i>Buteo lineatus</i>	--	2	2	--
Great horned owl	<i>Bubo virginianus</i>	--	1	1	--
Unidentified		2	2	4	--
Total		125	194	319	130

*Denotes focal species

[†] Number of observations included in the models for each of the 4 focal species. These values include birds that were perched (red-tailed hawk) or hovering (American kestrel and rough-legged hawk) and all northern harriers within the respective buffers from the road (American kestrel: 75 m, red-tailed hawk and rough-legged hawk: 250 m, northern harrier: 150 m).

Table 3.3: Summary of the top candidate models ($\leq 2 \Delta AIC_c$) and the number of models including each variable for each species. Area under the receiver operating curve (AUC) and normalized area under the precision-recall curve (AUCNPR) are reported for the top model. Abbreviations: RTHA = red-tailed hawk (*Buteo jamaicensis*), AMKE = American kestrel (*Falco sparverius*), RLHA = rough-legged hawk (*Buteo lagopus*), NOHA = northern harrier (*Circus hudsonius*), R = available points selected randomly, CR = available points constrained by hunting mode, Cover = cover-cropped fields, NoCover = non-cover-cropped fields, PermHab = permanent herbaceous habitat, Woods = woodlots, Dev = development, Uline = utility lines, Treeperim = outer perimeter of tree canopies. NA = not included in models.

Model			# Models ≤ 2 ΔAIC_c	AUC	AUCNPR	Number of Models Incorporating the Variable							
Species	Scale	Available				Cover	NoCover	PermHab	Woods	Dev	Uline	Treeline	Treeperim
AMKE	Transect	CR	4	0.7	0.52	4	4	1	4	0	0	0	1
		R	5	0.91	0.84	5	2	2	3	0	5	0	2
	Landscape	CR	10	0.7	0.53	10	10	3	4	0	4	0	3
		R	5	0.88	0.81	5	2	3	0	0	5	0	3
RTHA	Transect	CR	11	0.69	0.52	1	3	4	8	9	11	1	2
		R	13	0.76	0.64	2	2	6	13	4	2	1	11
	Landscape	CR	13	0.72	0.56	1	4	4	12	4	13	1	3
		R	15	0.77	0.64	1	2	6	15	13	0	3	6
RLHA	Transect	CR	11	0.68	0.53	1	1	2	7	2	NA	1	3
		R	5	0.70	0.52	0	0	3	2	0	NA	0	1
	Landscape	CR	12	0.65	0.52	1	1	3	3	7	NA	1	3
		R	11	0.70	0.64	1	1	7	2	2	NA	1	2
NOHA	Transect	CR	9	0.5	0.28	2	2	1	1	1	NA	1	NA
		R	8	0.58	0.34	2	6	1	1	1	NA	1	NA
	Landscape	CR	9	0.55	0.32	2	6	1	1	1	NA	2	NA
		R	8	0.58	0.34	2	6	1	1	1	NA	1	NA

Table 3.4: Regression coefficients (\pm Standard Error) for the top 1-3 resource selection models for each species, as determined with ΔAICc . Abbreviations: RTHA = red-tailed hawk (*Buteo jamaicensis*), AMKE = American kestrel (*Falco sparverius*), RLHA = rough-legged hawk (*Buteo lagopus*), NOHA = northern harrier (*Circus hudsonius*), R = available points selected randomly, CR = available points constrained by hunting mode, Cover = cover-cropped fields, NoCover = non-cover-cropped fields, PermHab = permanent herbaceous habitat, Woods = woodlots, Dev = development, Uline = utility lines, Treeline = linear tree features, Treeperim = outer perimeter of tree canopies. Variables with a p value ≤ 0.05 are bolded and those with $0.05 < p \leq 0.1$ are italicized.

Species	Scale	Available	Model	Cover	NoCover	PermHab	Woods	Dev	Uline	Treeline	Treeperim
AMKE	Transect	CR	1	-0.81 (0.28)	-0.71 (0.36)		0.29 (0.12)				-0.34 (0.22)
			2	-0.73 (0.28)	-0.71 (0.36)	-0.40 (0.26)	0.29 (0.11)				
			1	-0.73 (0.29)		-0.59 (0.33)	0.50 (0.20)		-43.77 (7.39)		
	Landscape	CR	1	-0.76 (0.29)	-0.79 (0.40)	-0.37 (0.25)	0.29 (0.13)				
			2	-0.83 (0.30)	-0.80 (0.40)		0.27 (0.13)				-0.30 (0.21)
			3	-0.73 (0.28)	-0.72 (0.38)				-0.62 (0.55)		
		R	1	-0.86 (0.30)	-0.38 (0.28)				-12.11 (2.10)		-0.48 (0.28)
			2	-0.78 (0.30)	-0.37 (0.28)	-0.54 (0.32)			-12.13 (2.10)		
			3	-0.77 (0.29)					-12.24 (2.10)		-0.41 (0.26)
		CR	1				-0.44 (0.25)	0.20 (0.10)	0.18 (0.05)		
			2			-0.23 (0.19)	-0.33 (0.26)	0.22 (0.10)	0.18 (0.05)		
			3			-0.32 (0.18)		0.20 (0.10)	0.20 (0.05)		
			1				-1.59 (0.43)				-0.40 (0.20)
			2			-0.31 (0.25)	-1.39 (0.45)				-0.36 (0.20)
			3				-1.51 (0.43)	-0.20 (0.20)			-0.30 (0.21)
RTHA	Transect	CR	1				-0.48 (0.26)		0.19 (0.04)		
			2				-0.54 (0.27)	0.16 (0.11)	0.19 (0.04)		
			3		-0.14 (0.12)		-0.52 (0.27)		0.22 (0.04)		
	Landscape	CR	1			-0.34 (0.26)	-1.97 (0.56)	-0.45 (0.22)			
		R	1								

RLHA	Transect	CR	2					-2.22	-0.49
			1					(0.53)	(0.22)
								0.48	
		R	2				-0.71	0.54	
			1				(0.81)	(0.22)	
							-1.58	0.46	
	Landscape	CR					(1.11)	(0.25)	
			2				-1.28		
							(1.07)		
		R	1						0.42
			2				-0.58		(0.18)
							(0.68)	0.45	(0.17)
NOHA	Transect	CR	1				-1.08		
			2				(0.91)		
							-1.50	0.48	
		R					(0.99)	(0.28)	
			1						
			2						
	Landscape	CR					-2.25		
			1				(3.97)		
			2				-4.65		
		R					(6.27)		
			1				-4.24		
			2	0.28			(6.10)		
	Transect	CR		(0.28)					
			1				-2.83		
			2				(4.23)		
		R	1				-4.24		
			2				(6.10)		
							0.28		
	Landscape	CR					(0.28)		
			1				-3.96		
			2				(5.45)		
		R	1				-3.66		
			2				(5.32)		
				0.32					

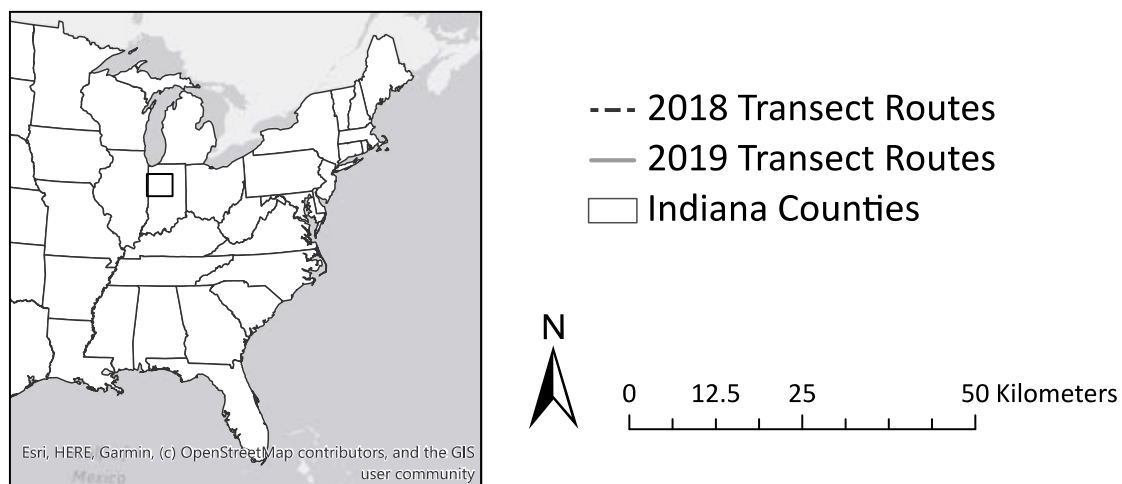
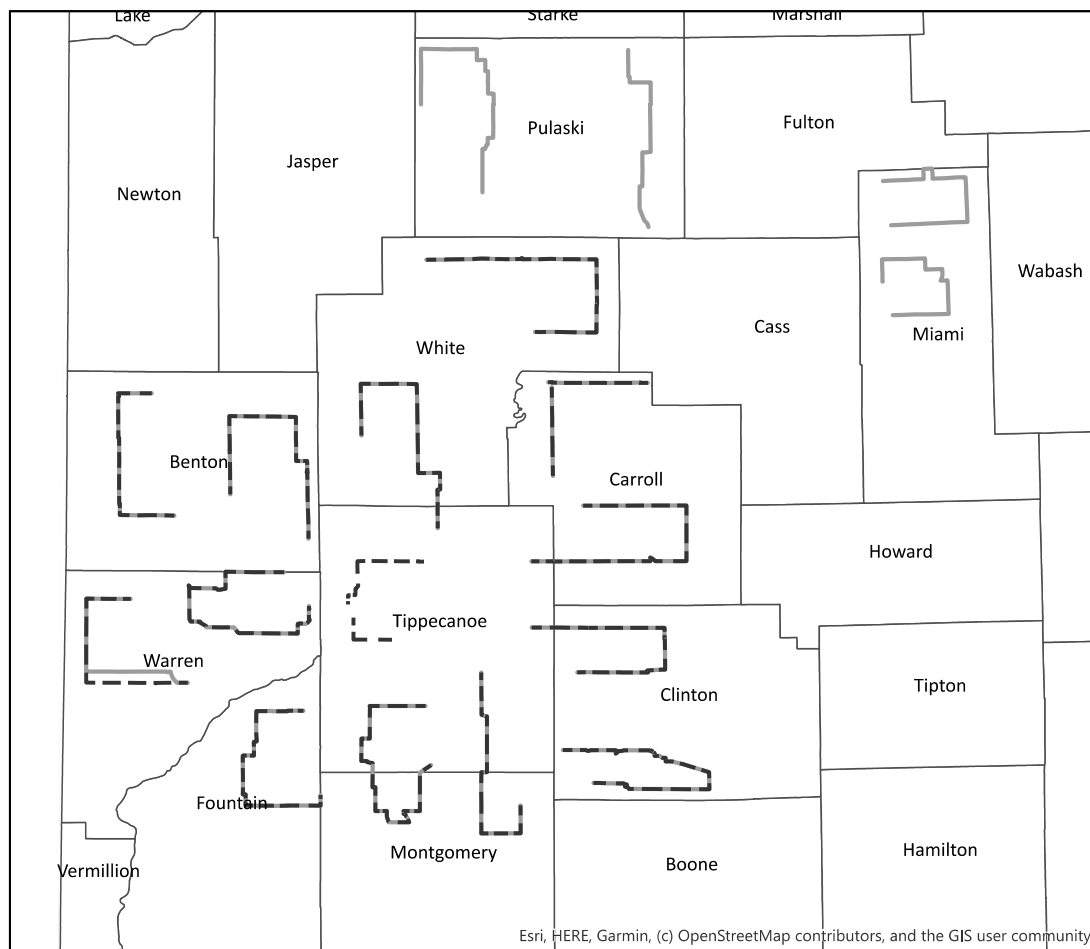


Figure 3.1: Transect routes surveyed for raptors in west-central Indiana January-April 2018 and 2019.

CHAPTER 4. VARIATION IN NORTHERN HARRIER (*CIRCUS HUDSONIUS*) DIETS IN WESTERN INDIANA

4.1 Abstract

Regurgitated pellets are a key source of dietary information about raptors. I dissected regurgitated pellets to analyze the diets of wintering northern harriers (*Circus hudsonius*) at communal roosts in western Indiana for variation between years, over the season, and between roost sites with multivariate regression. Because there is no uniformly adopted method for determining the minimum number of individuals (MNI) contained in a pellet, I also assessed whether the method used to determine MNI influenced my conclusions about pellet contents between months and roosts. I collected 821 pellets from 4 roost sites in 2018 and 2019. Pellet contents differed between years, months, and roost sites. *Microtus* was the most commonly occurring prey group at all roosts and in both years (range: 45-73%), but was encountered less frequently than reported by other studies in the Midwest. Other important prey groups included *Peromyscus* (5-16%), *Reithrodontomys* (4-14%), *Blarina* (8-18%), and birds (7-19%). Relative to other studies, shrews and *Reithrodontomys* were more important components of northern harrier diets in western Indiana. The 4 methods of counting MNI produced no discernible effects on my conclusions concerning pellet contents between months or roosts. Northern harrier diets in western Indiana are highly flexible and likely reflect local prey populations. Considering teeth in addition to bones may prove beneficial for pellet-based diet studies of other diurnal raptors.

4.2 Introduction

Diet plays a key role in raptor community ecology (Craighead and Craighead 1956, Marti 1974). Pellet analysis is one of the most common methods employed to study raptor diets, as pellets can be obtained noninvasively at roost and nest sites (Muñoz-Pedreros and Rau 2004, Marti et al. 2007). Since pellets are composed of the remains of undigested prey including bones, fur, feathers, and exoskeletons, they can efficiently answer many ecological questions regarding trophic relationships, dietary preferences, and changes to diet (Glue 1970, Muñoz-Pedreros and Rau 2004, Marti et al. 2007, Hodara and Poggio 2016). In addition, pellets can provide insight into prey communities and have been used to investigate small mammal habitat preferences,

population trends, and ranges (Wooster 1936, Avery et al. 2002, Millán de la Peña et al. 2003, McDonald et al. 2013, van Strien et al. 2015). Indeed, pellets often can offer a more accurate estimation of small mammal abundance and provide more complete records of the local small mammal assemblages than trapping (Glue 1970, Avery et al. 2002, Muñoz-Pedreros and Rau 2004, Torre et al. 2004, Avenant 2005, McDonald et al. 2013, Heisler et al. 2016).

The northern harrier (*Circus hudsonius*) is a widespread raptor that primarily consumes small mammals and birds (Sherrod 1978, Smith et al. 2011). During the past 50 years, northern harrier populations have declined in most regions across North America, largely due to habitat loss (Peterjohn and Sauer 1999, Smith et al. 2011, Sauer et al. 2013, Sauer et al. 2017, Stanton et al. 2018). Throughout much of the species' range in the Midwestern United States, intensive row-crop agriculture has replaced habitats that are important for roosting and hunting including native prairies and wetlands as well as pastures and hayfields (Dahl and Allord 1982, Serrentino 1992, Samson and Knopf 1994, Smith et al. 2011). Consequently, populations of northern harriers are listed as state endangered or threatened in 11 states including Indiana (Smith et al. 2011).

In winter, northern harriers form communal roosts, which facilitate efficient pellet collection and analysis of winter diets (Craighead and Craighead 1956, Littlefield 1970, Mumford and Danner 1974, Bildstein 1976, Bildstein 1979). Northern harrier pellets yield more information on diet than other diurnal raptors, but less information than owl pellets (Errington 1930). Pellets have been used to investigate northern harrier diets in the Midwest (Weller et al. 1955, Mumford and Danner 1974, Walk 1998). These studies examined 318, 344, and 65 pellets, respectively, and reported relative frequency of occurrence of various prey items, but no quantitative analyses of diet composition. Statistical analysis of pellet data often is limited (but see Hellström et al. 2014, Hodara and Poggio 2016 and García-Heras et al. 2017b). I used multivariate regression on 821 northern harrier pellets collected over 2 years in western Indiana to address whether northern harrier diets 1) changed over the course of a winter, 2) differed between years, or 3) differed among roost sites.

Studies of raptor pellets also vary widely in their definition of what constitutes the minimum number of individual (MNI) prey items in a pellet. Some studies required a matching set of skulls and mandibles from mammalian prey (Kirkpatrick and Conway 1947), whereas others counted the number of molars (Salamolard et al. 2000, Hellström et al. 2014). I am

unaware of any studies to systematically assess the influence of variable definitions of prey on conclusions derived from pellet analysis. Thus, a secondary objective was to conduct such an assessment for the northern harrier pellets I collected.

4.3 Methods

4.3.1 Study Area

I collected pellets from 4 communal roosts of northern harriers in western Indiana over 2 winters (Dec. 2017-April 2018 and Dec. 2018-April 2019). Intensive row-crop agriculture of corn (*Zea mays*) and soybeans (*Glycine max*) was the dominant land-use throughout the region. Of the 4 roosts, 1 (Shadeland) was a privately owned alfalfa (*Medicago sativa*) field, 2 (Kankakee Sands and Prophetstown) were restored prairies, and 1 (Chinook Mine) was a reclaimed strip mine (Fig. 4.1). The Kankakee Sands roost was managed as part of The Nature Conservancy's Efroymsen Restoration at Kankakee Sands, and the Prophetstown and Chinook Mine roosts were on property managed by the Indiana Department of Natural Resources (Prophetstown State Park and Chinook Fish and Wildlife Area, respectively). Pellets were collected from Shadeland only in the first winter, from Prophetstown and Chinook Mine only in the second winter, and from Kankakee Sands in both winters.

4.3.2 Pellet Collection and Dissection

Once a roost site was confirmed, I visited every 3 weeks, or as the weather permitted. During each visit, I collected all pellets, noted the coordinates with handheld GPS units (GPSmap 78s and GPSmap 60Cx, Garmin Ltd., Olathe, KS), and recorded the bed from which each pellet was collected and the number of pellets per bed. Beds were clearly demarcated in the field as depressions or grass platforms marked with whitewash (Mumford and Danner 1974). I excluded 5 pellets that could have been produced by short-eared owls (*Asio flammeus*) based on pellet characteristics (Holt et al. 1987). I assumed that all remaining pellets were from northern harriers based on pellet characteristics and associated whitewash (Holt et al. 1987). Although short-eared owls and northern harriers will form interspecific roosts, I did not flush any owls

during pellet collection and only observed 4 owls while searching for roost sites. Thus, the likelihood of misidentifying pellets was low (Bildstein 1976, Bildstein 1979).

I air-dried pellets for ≥ 3 weeks before manual dissections. I identified bones to the lowest possible taxonomic level based on teeth, mandibles, skulls, and feathers (Schwartz and Schwartz 2001, Elbroch 2006), referencing the Purdue University Department of Forestry and Natural Resources vertebrate collection.

I assigned a minimum number of individuals (MNI) to each pellet based on the maximum number of skulls or right and left mandibles (McDowell and Medlin 2009, McDonald et al. 2013, García-Heras et al. 2017a). When bones were absent, I assumed a pellet contained 1 individual based on the identification of guard hairs (Debelica and Thies 2009). Due to the acidity of the northern harrier digestive tract and tendency to tear prey, many skulls and mandibles were fragmented (Mumford and Danner 1974, Duke et al. 1975, Cummings et al. 1976). Consequently, I was conservative in my counts of fragmented bones.

For some pellets, teeth were the only identifiable remains ($n=47$), or did not match the species identified by skulls and mandibles ($n=113$). These teeth could represent juvenile or smaller prey whose bones were more easily digested, or they could represent teeth of prey items from prior pellets that remained in the crop and were egested separately. There is no consensus in the pellet literature concerning the best method to account for such teeth. Methods of counting MNI range from only considering matched sets of skull and mandibles (Kirkpatrick and Conway 1947) to counting molars (Salamolard et al. 2000, Hellström et al. 2014). To assess effects of variable definitions, I compared 4 different methods of incorporating teeth into my MNI values:

1. An individual of a species was present if a pellet contained ≥ 1 tooth.
2. An individual of a species was present if a pellet contained $\geq \frac{1}{4}$ of the total number of molars, i.e. the equivalent of 1 mandible.
3. An individual of a species was present if a pellet contained $\geq \frac{1}{2}$ of the total number of molars, i.e. the equivalent of 2 mandibles or 1 skull.
4. Teeth were not used to determine MNI.

4.3.1. Statistical Analysis

I calculated the percent occurrence for each prey group by dividing the number of prey within the group by all identified prey for each roost site and collection period. I excluded the Shadeland pellets from all remaining statistical analyses due to low sample size at the roost ($n=7$ pellets). Because beds with multiple pellets were assumed to reflect repeated use by the same individual, I compared the dissimilarities in contents between single-pellet beds and multiple-pellet beds to determine if they needed to be analyzed separately. Next, I ran a series of generalized linear models with a multivariate response to determine if diets changed between years, among months within a year, or between roosts. For each model, I only included data from months that were common to each of the roosts included in analysis. Separate models were fitted for each of the 4 definitions used to determine MNI. The response variable for each model was a matrix of prey species counts per pellet, which enabled us to simultaneously model variation in counts of each prey species while properly accounting for correlations between species when making inferences (Warton et al. 2012). Both Poisson and negative binomial regression models were fitted to the count data. The Poisson produced better fit as judged using Akaike Information Criterion and diagnostic plots, and was used for a majority of analyses. An exception was the comparison of methods used to determine MNI, for which I used a negative binomial model. I compared pellet composition between years only for the Kankakee Sands roost, as it was the only site where I collected pellets in both years. I compared pellet composition over the course of the winter season and between sites for the remaining 3 roosts by incorporating 2 factor variables into the model: collection month and roost site. The only exception to collection month occurred at Kankakee Sands in 2019. I visited twice in March, hence my visit on 28 March 2019 was assigned to April, as it was at the end of the month. For the collection month model, I incorporated both years of Kankakee Sands data but modeled each year as a separate roost site. To assess the effect of MNI method, I fit a model with MNI method, collection month, and roost site as main effects, with two-way interactions of MNI method x month and MNI method x roost. Strength of the interaction effects was used to test the degree to which MNI method could affect conclusions of my diet analysis. Analysis of deviance with likelihood ratio tests was used to assess the degree to which variables improved model fit (Warton et al. 2017). Multivariate regression was implemented in R package mvabund (Wang et al. 2019) in R version 3.6.1 (R Core Team 2019).

4.4 Results

I collected 821 pellets from 4 roost sites in 2018 and 2019. I identified 9 prey groups: *Microtus*, *Peromyscus*, *Reithrodontomys*, *Mus*, *Blarina*, *Sorex*, *Cryptotis*, birds, and grasshoppers (Table 4.1). *Microtus* was the most common prey group among all sites and years, with all other prey groups <20% occurrence (Table 4.2). I identified 214 individual *Microtus* to species, of which 65% were *M. ochrogaster* and 35% were *M. pennsylvanicus*. Across the range of counting methods, the number of prey per pellet spanned from 1 to 8. Depending on the counting method, on average (\pm SD) there were between 2.17 ± 1.02 (counting any teeth) and 1.99 ± 1.02 (counting no teeth) prey per pellet.

I collected 55.5% (n=456) of pellets from single-pellet beds and 44.5% (n=365) from multiple-pellet beds. The highest count per bed was 15 pellets. I used a Wilcoxon rank sum test with continuity correction to test for differences in pellet contents between single- and multiple-pellet beds. If the relative frequency of single- and multiple-pellet beds differed between months for a roost site, I analyzed each month separately. The relative frequency of single- and multiple-pellet beds at Prophetstown ($\chi^2_1 = 4.0$, $p = 0.04$) and in 2019 at Kankakee Sands ($\chi^2_4 = 26.5$, $p < 0.001$) differed noticeably throughout the season; hence they were analyzed monthly. Monthly changes in relative frequency of single- and multiple-pellet beds were negligible at Chinook Mine ($\chi^2_2 = 3.7$, $p = 0.16$) and in 2018 at Kankakee Sands ($\chi^2_3 = 3.4$, $p = 0.33$); thus, I pooled pellets across all collection periods for these roosts. Of 9 site x month comparisons, prey contents differed substantially for single- versus multiple-pellet beds collected at Kankakee Sands in 2018 (Wilcoxon $W = 11440$, $p = 0.01$), April 2019 ($W = 114090$, $p < 0.001$) and in March at Prophetstown ($W=2832.5$, $p = 0.02$). However, negligible differences were noted in tests of the remaining 6 roosts x month combinations ($p = 0.12, 0.33, 0.45, 0.51, 0.84$, and 1). Hence, I pooled pellets from single- and multiple-pellet beds for subsequent analyses.

4.4.1 Influence of Counting Method

MNI counts varied with the method of counting teeth, ranging from 1781 total individuals (any teeth) to 1637 (no teeth, Table 4.3). Importantly, counting method did not affect the conclusions from multivariate tests for differences between months (likelihood ratio $X^2 = 0.9$, $p=1$) or between roosts (likelihood ratio $X^2 = 3.7$, $p=1$, Table B1). Because counting method had

virtually no influence on the overall statistical conclusions, and because estimates were quite similar for no teeth and $\geq \frac{1}{2}$ of teeth (Table 4.3), the remaining results are reported based on MNI counts including $\geq \frac{1}{2}$ of teeth. This method of counting teeth is conservative, but takes into account the biology of northern harriers, particularly their more complete digestion of osseous prey remains (Errington 1930, Duke et al. 1975) while avoiding likely double-counting of individuals.

4.4.2 Variation Between Years

Overall pellet composition differed (likelihood ratio $X^2=21.17$, $p=0.016$) between 2018 and 2019 at Kankakee Sands. Expected counts of *Peromyscus* (likelihood ratio $X^2=7.69$, $p=0.044$) and *Microtus* (likelihood ratio $X^2=6.33$, $p=0.077$) were higher on average in 2019 than in 2018 (Table 4.4). Indeed, all prey groups except *Blarina* and birds yielded positive coefficients, consistent with increased mean counts in 2019 at Kankakee Sands (Table 4.4). Due to the overall difference in pellet composition between years, I segregated the Kankakee Sands pellets by year for the following 2 analyses.

Composition of prey in pellets differed (likelihood ratio $X^2=19.26$, $p=0.039$) between February and March, even though trends were not pronounced for any prey group (Table 4.5). In 2018, *Peromyscus* at Kankakee Sands declined in late winter while *Blarina* increased in late winter and early spring (Fig. 4.2A). In 2019 at Kankakee Sands and Chinook Mine, percent occurrence of most prey groups was relatively stable, although *Microtus* increased in importance in February at Chinook Mines and birds declined after December at Kankakee Sands (Fig. 4.2B-C). At Prophetstown, birds occurred much more frequently in pellets collected in April, while *Peromyscus* and *Blarina* occurred less frequently into the early spring (Fig. 4.2D).

4.4.3 Variation Between Roost Sites

There was a strong difference (likelihood ratio $X^2=118.67$, $p=0.001$) in overall pellet composition between roost sites. Differences were particularly notable for *Microtus* (likelihood ratio $X^2=48.01$, $p=0.001$) and *Peromyscus* (likelihood ratio $X^2=31.49$, $p=0.001$, Table 4.5). Compared to Chinook Mine, pellets from Prophetstown and Kankakee Sands roosts in both winters had fewer *Microtus* and *Cryptotis*, but *Peromyscus* and *Reithrodontomys* were more

common in pellets at Kankakee Sands and birds were more common at Prophetstown (Table 4.5, Fig. 4.2A-D).

4.5 Discussion

Northern harrier diets changed over the course of the winter, between years, and between sites in west-central Indiana. My results demonstrate the adaptability of northern harriers to local prey communities in landscapes extensively modified by humans. As with other studies, *Microtus* was the most abundantly represented prey in pellets from each of my roost sites (Weller et al. 1955, Craighead and Craighead 1956, Mumford and Danner 1974, Bildstein 1978). However, occurrence of *Microtus* in my study was lower overall than levels reported in other studies; indeed, only the pellets from Chinook Mine approached previously published levels of *Microtus* in northern harrier diets from the Midwest, where voles constituted >75% of pellet contents, even in years of low vole abundance (Weller et al. 1955, Craighead and Craighead 1956, Mumford and Danner 1974, Bildstein 1978). However, in France, vole presence in Montagu's harrier (*Circus pygargus*) diets dropped to 40-50% when common vole (*Microtus arvalis*) populations were low (Salamolard et al. 2000). In the spring of 2018 and 2019, vole captures in permanent grassy habitat bordering agricultural fields were >5 times lower than trap success in a prior study (Whitaker and Mumford 2009, Prieur 2019), which suggests that vole populations in 2018-2019 were at a low phase in their population cycle over my study area.

The relatively low occurrence of voles in my pellets was offset by higher frequencies for several other small mammal prey groups. *Peromyscus*, *Reithrodontomys*, and *Blarina* occurred more frequently in my northern harrier pellets than documented by other studies in the Midwestern United States (Weller et al. 1955, Craighead and Craighead 1956, Mumford and Danner 1974, Bildstein 1978). To my knowledge, *Reithrodontomys* (8.3% occurrence) has not been reported previously in northern harrier pellets from the Midwest, although a congeneric species, the fulvous harvest mouse (*Reithrodontomys fulvescens*), has been recorded in pellets from Arkansas and Mississippi (Jackson et al. 1972, Preston 1990). Notably, shrews in my study occurred in northern harrier pellets at a level (14.6%) that was unprecedented. Percent occurrence of *Blarina* (7.9-17.8%) in pellets from my 3 primary sites was much higher than the 2.4% reported by Mumford and Danner (1974) in southeastern Indiana and exceeded levels reported elsewhere in North America (Randall 1940, Jackson et al. 1972, Bildstein 1978). At a

property <10 km from the roost, *Blarina* was one of the few species caught every year for >3 decades (Whitaker 2004). Their predictable occurrence, coupled with the high activity levels typical of shrews (Churchfield 1990), could have made *Blarina* more vulnerable to predation. Collectively, the increased prevalence of prey other than *Microtus* provides further evidence of low vole abundance in my study area. It also suggests that northern harriers can exhibit short-term functional responses to varying prey availability, in addition to the numerical responses documented previously (Phelan and Robertson 1978, Steenhof and Kochert 1988, Poulin et al. 2001).

I failed to detect southern bog lemmings (*Synaptomys cooperii*) or eastern cottontails (*Sylvilagus floridanus*) in my pellets, in contrast to other studies (Weller et al. 1955, Mumford and Danner 1974, Walk 1998). At my study sites, southern bog lemmings were likely rare, as they were encountered infrequently during small mammal surveys in Indiana (Veilleux et al. 1998, Whitaker 2004, Whitaker et al. 2007, Whitaker and Chamberlain 2011). Perhaps not surprisingly, *Synaptomys* was not reported from any of the 334 pellets collected in southeastern Indiana (Mumford and Danner 1974). My method of identifying prey may have underestimated cottontails. Cottontail skulls and mandibles would have been too large to appear in pellets, and I only identified hair from the 8% of pellets without identified skulls or mandibles. Consequently, I may have missed any rabbits that were consumed. However, rabbits are typically <5% of northern harrier diets (Randall 1940, Jackson et al. 1972), with only 1 study (Weller et al. 1955) encountering rabbits at higher (7-16%) frequencies. Estimating large prey in pellets is acknowledged as a problem in pellet studies, and could be rectified through the use of a complementary diet technique such as direct observations (Marti et al. 2007) or DNA barcoding (Valentini et al. 2009, Pompanon et al. 2012).

At Kankakee Sands, northern harrier diets changed between 2018 and 2019. *Peromyscus* and *Microtus* were the most important drivers of inter-annual variation. Both genera increased in 2019, suggesting that populations may have increased somewhat relative to 2018. However, the magnitude of change was modest, with percent occurrence over the entire season increasing by 5% and 3%, respectively (Table 4.2). Continued monitoring of northern harrier diets in conjunction with independent sampling of small mammals could yield useful information regarding the use of northern harrier pellets as an index of small mammal population trends.

Dietary changes within a winter also may have been influenced by seasonal population trends or changes in prey behavior. Despite changes in overall prey composition, most prey groups were relatively stable dietary components over the course of the winter. *Peromyscus* was an exception, as its representation in northern harrier pellets declined in late winter at Kankakee Sands in 2018 and Prophetstown in 2019, consistent with population trends generally observed for this prey group in agroecosystems of Indiana, Kansas, and Minnesota (Fig. 4.2, Fleharty 1972, Yahner 1983, Berl et al. 2017). Small mammal mortality is often highest in late winter as a result of inclement weather, reduced food availability, and predation pressure (Gottschang 1965, Sauer 1985, Swihart and Slade 1985, Lin and Batzli 1995). Additionally, small mammals in temperate regions change their activity patterns seasonally, potentially increasing their vulnerability to predation, particularly in late winter (Erkinaro 1961). Therefore, it is perhaps unsurprising that the frequency of some prey groups declined in northern harrier pellets during this period.

Differences in pellet contents between roost sites were especially strong for *Microtus* and *Peromyscus*. Although northern harriers at Kankakee Sands consumed *Microtus* least frequently among the 3 sites, they also had the highest percentages of 3 of the other prey groups: *Peromyscus*, *Reithrodontomys*, and *Blarina* (Fig. 4.2). Kankakee Sands was located on the largest of the 3 roost site properties and had the highest proportion of herbaceous cover, with the surrounding landscape predominantly in row-crop agriculture (USDA National Agricultural Statistics Service Cropland Data Layer 2019). At 2900 ha, it was within 10 km of 3 additional Indiana DNR properties totaling 4600 ha. Consequently, small mammal diversity was likely higher in the area surrounding the roost than at the other 2 sites. Indeed, over a 34-year period 72% of Indiana's mammal species were observed <10 km from the roost (Whitaker 2004). Prey groups that I observed frequently in pellets, namely *Blarina*, *Peromyscus*, and *Microtus*, were particularly abundant in the vicinity of Kankakee Sands (Whitaker 2004). Moreover, *Reithrodontomys megalotis* was first recorded in Indiana <10 km from the Kankakee Sands roost (Whitaker and Sly 1970). *R. megalotis* has since expanded throughout west-central Indiana, and its prevalence in northern harrier diets likely reflects its expanded range and abundance. Although all of my roost sites were within the range of *R. megalotis*, the Chinook Mine roost occurs at the most southerly range extent in Indiana (Leibacher and Whitaker 1998, Whitaker and Chamberlain 2011). Vertebrates near their range boundaries in west-central Indiana exhibit

lower patch occupancy (Swihart et al. 2003), which tends to be related to density in a nonlinear manner (Linden et al. 2017). Consequently, it is likely that *Reithrodontomys* populations around Chinook Fish and Wildlife Area were less abundant.

The Prophetstown roost, like Kankakee Sands, is a restored prairie surrounded by a high degree of row-crop agriculture in addition to developed areas (USDA National Agricultural Statistics Service Cropland Data Layer 2019). At 360 ha, Prophetstown State Park was my smallest site, was fragmented by roads and trails, and likely experienced the highest degree of human disturbance. The smaller size, fragmentation and human activity within the park and in the surrounding area may have reduced small mammal diversity (Swihart et al. 2006, Kellner et al. 2019) and thus contributed to lower diversity in pellet contents. Similarly, there was a moderate amount of development in the area around the Chinook Mine roost site (USDA National Agricultural Statistics Service Cropland Data Layer 2019). Chinook Mine had the highest amount of forested land and the lowest amount of cropland among the 3 sites, which may further reduce the available foraging area for northern harriers (USDA National Agricultural Statistics Service Cropland Data Layer 2019). Studies at other regional properties have found lower small mammal diversity than the Kankakee Sands region (Veilleux et al. 1998, Whitaker et al. 2007, Whitaker and Chamberlain 2011). At Goose Pond Fish and Wildlife Area, 64 km south of the Chinook Mine roost, 81% of captured small mammals were *Microtus* (Whitaker and Chamberlain 2011). The higher levels of *Microtus* and limited regional species diversity were reflected in the pellets collected at Chinook Mine.

Ultimately, my overall conclusions were unaffected by varying the inclusion rules based on number of teeth counted to compute MNI. However, only 19% of my pellets were affected when varying the inclusion rules. Future studies should investigate the consequences of including teeth in MNI estimations when a higher proportion of pellets are affected by the different counting methods and for other raptor species. Potential biases in counting method toward certain prey groups should be assessed further, perhaps with captive birds. Although the definition of MNI did not have a discernible effect on any of my model-based hypotheses or on expected counts, it is unclear whether my finding applies more broadly to other raptors. Therefore, I urge others to adopt a modeling framework that can explicitly address the potential for bias induced by different definitions of MNI, as such information is important when assessing the degree to which comparisons across studies are justified. Based on my results, I

recommend counting $\geq \frac{1}{2}$ of the teeth for northern harriers. This method is conservative, while also accounting for northern harrier biology. Studies investigating other diurnal raptors may benefit from a consideration of teeth in addition to bone content.

Table 4.1: List of prey identified in northern harrier (*Circus hudsonius*) pellets collected at 4 communal roosts in western Indiana, December through April 2018 and 2019.

Prey Group	Common Name	Scientific Name
<i>Microtus</i>	Prairie vole	<i>Microtus ochrogaster</i>
	Meadow vole	<i>Microtus pennsylvanicus</i>
<i>Peromyscus</i>	Deer mouse	<i>Peromyscus maniculatus</i>
	White-footed mouse	<i>Peromyscus leucopus</i>
<i>Reithrodontomys</i>	Western harvest mouse	<i>Reithrodontomys megalotis</i>
<i>Mus</i>	House mouse	<i>Mus musculus</i>
<i>Blarina</i>	Northern short-tailed shrew	<i>Blarina brevicauda</i>
<i>Sorex</i>	Masked shrew	<i>Sorex cinereus</i>
	Southeastern shrew	<i>Sorex longirostris</i>
<i>Cryptotis</i>	Least Shrew	<i>Cryptotis parva</i>
Birds*	Northern cardinal	<i>Cardinalis cardinalis</i>
	Eastern bluebird	<i>Sialia sialis</i>
	Common grackle	<i>Quiscalus quiscula</i>
	European Starling	<i>Sturnus vulgaris</i>
Grasshopper	---	<i>Acridimorpha</i>

*I only positively identified 4 species based on bill shape or feather color. Most birds were unidentifiable.

Table 4.2: Percent occurrence of prey groups in northern harrier (*Circus hudsonius*) pellets collected December-April 2018 and 2019 at 4 roost sites in western Indiana. If $\geq \frac{1}{2}$ of the teeth were present for pellets without skulls or mandibles or where the teeth did not match the skulls and mandibles, 1 individual was counted.

Prey Group	Roost				
	2018		Chinook Mine	2019	
	Kankakee Sands	Shadeland		Kankakee Sands	Prophetstown
<i>Microtus</i>	45.1	41.7	73.2	48.5	56.7
<i>Peromyscus</i>	11.5	16.7	4.6	16.0	5.3
<i>Reithrodontomys</i>	13.8	8.3	4.1	10.1	4.7
<i>Mus</i>	0.4	0	0.8	1.7	0
<i>Blarina</i>	17.8	33.3	7.9	13.7	12.9
<i>Sorex</i>	0.4	0	0.8	1.7	1.2
<i>Cryptotis</i>	0	0	1.9	0.2	0
Birds	10.7	0	6.8	8.0	19.3
Grasshopper	0	0	0	0.2	0
Number of Pellets	148	7	223	354	89

Table 4.3: Comparison of methods of counting teeth toward the minimum number of individuals (MNI) per pellet for northern harrier (*Circus hudsonius*) pellets collected at 4 roost sites in western Indiana in 2018 and 2019. I evaluated 4 methods for counting individuals: if ≥ 1 tooth was present (any teeth), if $\geq \frac{1}{4}$ of the teeth were present, if $\geq \frac{1}{2}$ of the teeth were present, and ignoring teeth (no teeth). The proportion of the maximum number of individuals for that prey group is included in parentheses.

Prey Group	Method			
	Any teeth	$\geq \frac{1}{4}$ teeth	$\geq \frac{1}{2}$ teeth	No teeth
<i>Microtus</i>	993 (1)	959 (0.97)	929 (0.94)	912 (0.92)
<i>Peromyscus</i>	196 (1)	185 (0.94)	178 (0.91)	173 (0.88)
<i>Reithrodontomys</i>	153 (1)	143 (0.93)	138 (0.9)	136 (0.89)
<i>Mus</i>	22 (1)	19 (0.95)	19 (0.95)	19 (0.95)
<i>Blarina</i>	226 (1)	214 (0.99)	208 (0.92)	207 (0.92)
<i>Sorex</i>	23 (1)	22 (0.99)	22 (0.99)	22 (0.99)
<i>Cryptotis</i>	12 (1)	12 (1)	12 (1)	12 (1)
Birds	156 (1)	156 (1)	156 (1)	156 (1)
Grasshopper	1 (1)	1 (1)	1 (1)	1 (1)
Total	1781 (1)	1708 (0.96)	1662 (0.93)	1637 (0.92)

Table 4.4: Adjusted likelihood ratio tests and associated parameter estimates (\pm SE) from a multivariate Poisson generalized linear model testing whether northern harrier (*Circus hudsonius*) pellet contents changed in 2018 and 2019 at the Kankakee Sands roost, Newton County, Indiana. If $\geq \frac{1}{2}$ of the teeth were present for pellets without skulls or mandibles or where the teeth did not match the skulls and mandibles, 1 individual was counted. The baseline year was 2018; thus, positive coefficients indicate larger expected counts in 2019. Boldfaced font used for tests with $p < 0.05$, and italicized font for tests with $0.05 \leq p < 0.10$.

Prey Group	χ^2	p-value	Coef (\pm SE)
<i>Microtus</i>	6.3	0.08	0.28 (0.11)
<i>Peromyscus</i>	7.69	0.04	0.59 (0.22)
<i>Reithrodontomys</i>	0.04	0.96	0.04 (0.22)
<i>Mus</i>	0.76	0.93	0.56 (0.67)
<i>Blarina</i>	0.29	0.96	-0.1 (0.19)
<i>Sorex</i>	3.51	0.34	1.59 (1.05)
<i>Cryptotis</i>	1.88	0.67	7.83 (40.87)
Birds	0.11	0.96	-0.08 (0.25)
Grasshopper	0.6	0.96	6.58 (43.12)

Table 4.5: Adjusted likelihood ratio tests and associated parameter estimates (\pm SE) from a multivariate Poisson generalized linear model testing whether northern harrier (*Circus hudsonius*) pellet contents changed between months and roost sites in western Indiana during 2018 and 2019. If $\geq \frac{1}{2}$ of the teeth were present for pellets without skulls or mandibles or where the teeth did not match the skulls and mandibles, 1 individual was counted. The baselines were February (Collection Month) and Chinook Mine (2019, Roost Site). Positive coefficients indicate larger expected counts in March (Collection Month) or at the specified roost site relative to the baseline. Boldfaced font used for tests with $p < 0.05$.

Prey Group	Collection Month			Roost Site				
				Kankakee Sands (2018)	Kankakee Sands (2019)	Prophetstown (2019)	χ^2	p
	Coef (\pm SE)	χ^2	p	Coef (\pm SE)	Coef (\pm SE)	Coef (\pm SE)		
<i>Microtus</i>	-0.03 (0.09)	0.13	0.85	-0.79 (0.14)	-0.49 (0.1)	-0.27 (0.14)	48.01	0.001
<i>Peromyscus</i>	0.03 (0.23)	1.0	0.839	0.36 (0.37)	1.28 (0.27)	0.26 (0.45)	31.49	0.001
<i>Reithrodontomys</i>	0.53 (0.3)	3.96	0.339	0.61 (0.37)	0.79 (0.31)	0.26 (0.56)	10.37	0.141
<i>Mus</i>	0.38 (0.67)	0.48	0.85	0.26 (0.92)	0.86 (0.68)	-10.85 (137.36)	5.04	0.53
<i>Blarina</i>	0.21 (0.21)	2.25	0.596	0.37 (0.28)	0.54 (0.23)	0.69 (0.3)	7.53	0.262
<i>Sorex</i>	9.33 (54.98)	6.9	0.119	-8.4 (54.98)	0.002 (0.76)	0.45 (0.91)	2.92	0.799
<i>Cryptotis</i>	1.13 (0.74)	3.19	0.432	-11.61 (104.83)	-1.83 (1.08)	-11.23 (137.36)	10.28	0.141
Birds	0.17 (0.25)	0.63	0.85	0.19 (0.32)	0.08 (0.27)	0.39 (0.35)	1.29	0.799
Grasshopper	6.36 (41.02)	0.76	0.85	0.4 (85.93)	7.9 (54.74)	-1 (147.68)	1.75	0.799

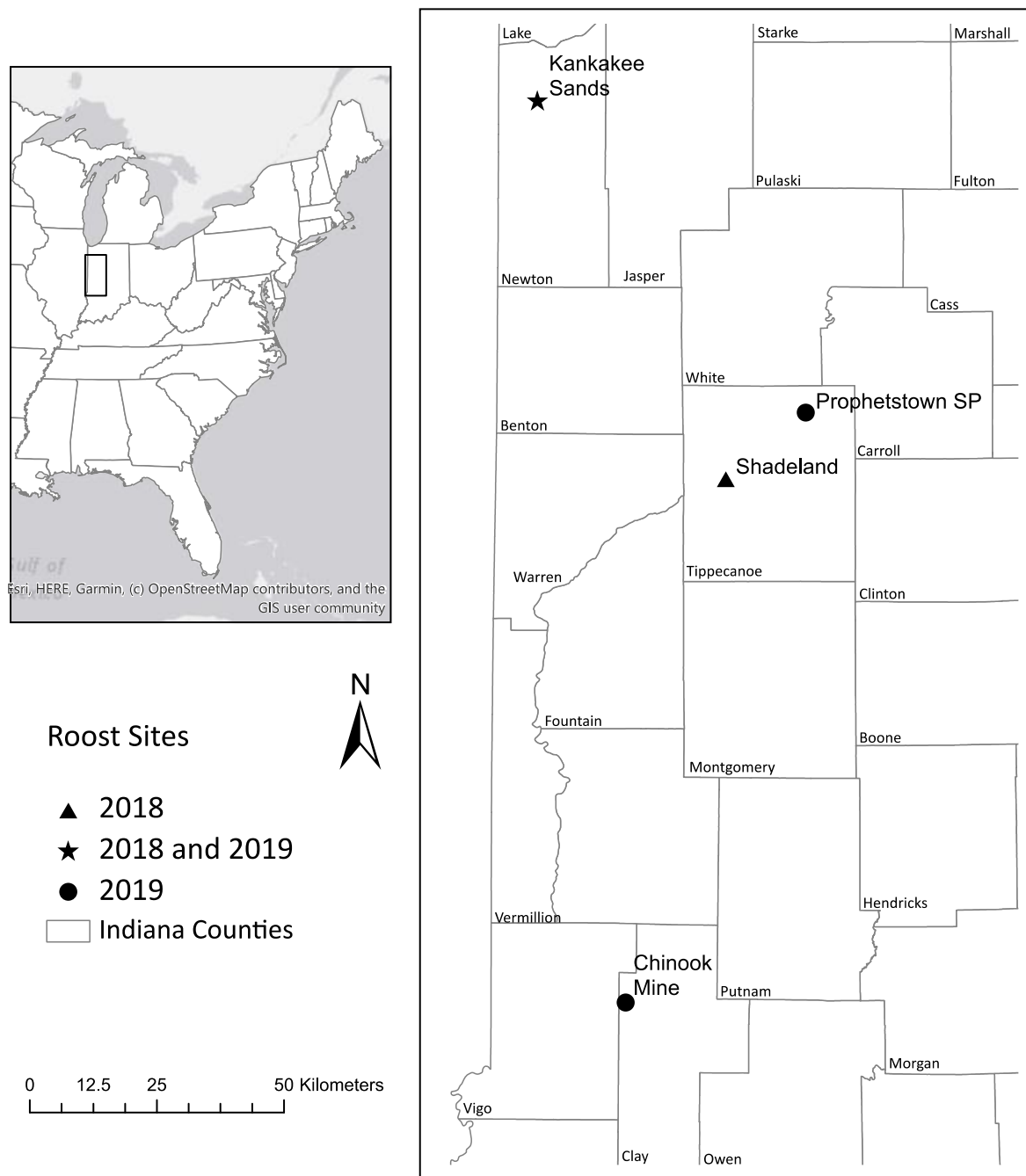
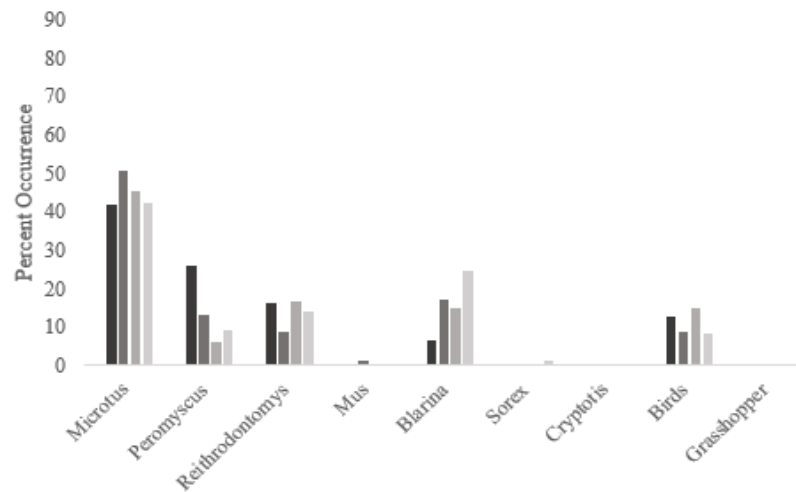


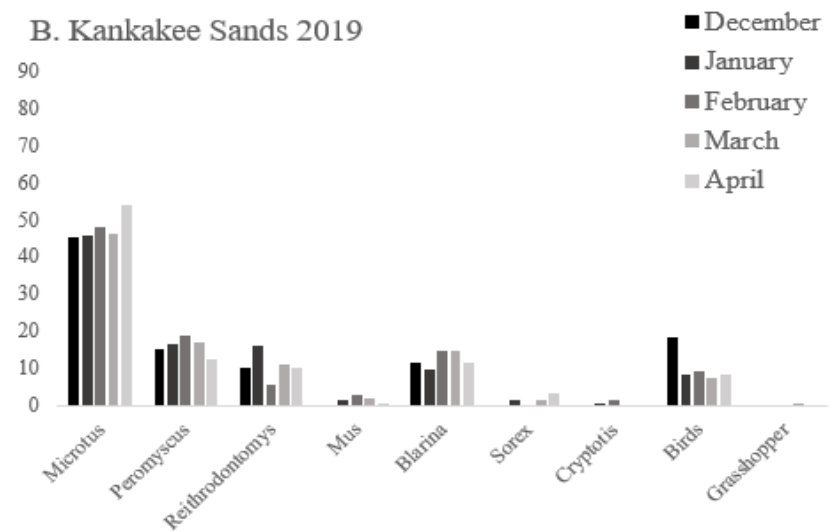
Figure 4.1: Northern harrier (*Circus hudsonius*) roost sites in western Indiana. Pellets were collected in 2018 (n=2 roosts) and 2019 (n=3 roosts).

Figure 4.2: Percent occurrence of prey groups by collection month for northern harrier (*Circus hudsonius*) pellets collected at 4 roost sites in 2018 and 2019 in western Indiana.

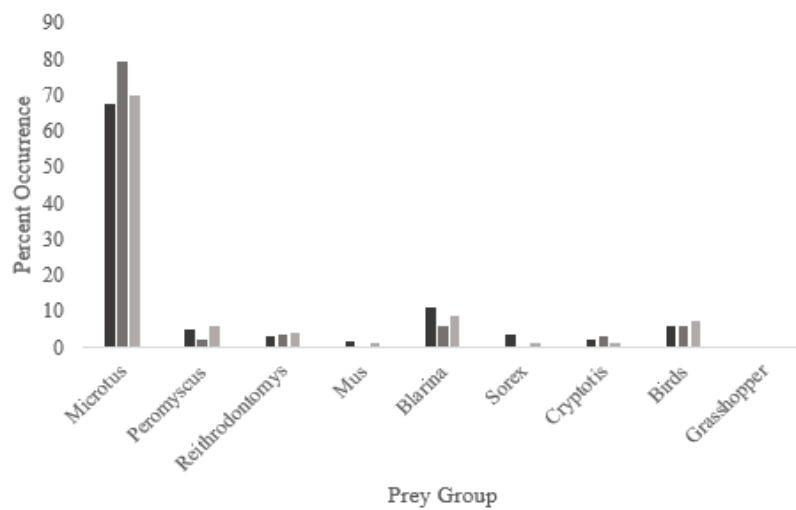
A. Kankakee Sands 2018



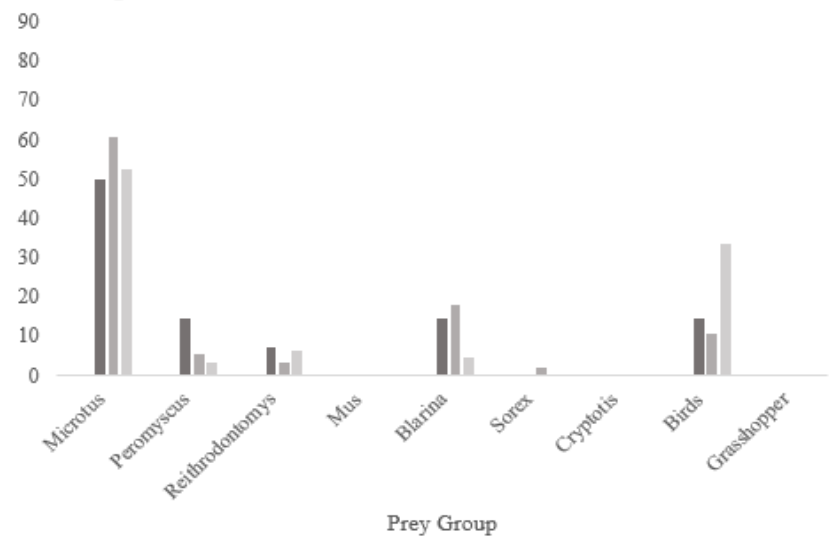
B. Kankakee Sands 2019



C. Chinook Mine



D. Prophetstown



CHAPTER 5. CONCLUSIONS

Agricultural conversion of grassland and forest has radically reduced and fragmented wildlife habitat in the Midwestern United States, including Indiana (Griffith et al. 1977, Samson and Knopf 1994, Kremen et al. 2002, Carman 2013, Stanton et al. 2018). Agricultural intensification negatively impacts wildlife, including raptors, which tend to decline as agriculture intensifies (Boano and Toffoli 2002, Filloy and Bellocq 2007, Butet et al. 2010). Recent agricultural trends, like adoption of cover crops, could benefit raptors by increasing prey abundances. Cover crops, an increasingly common conservation practice in intensive row-crop agriculture of the Midwest, can improve wildlife habitat (Ellis and Barbercheck 2015, Wilcoxon et al. 2018). However, cover crops can also benefit agricultural pests, such as voles, which can cause damage to cash crops in the spring (Fisher et al. 2014). With adoption increasing throughout the Midwest, understanding the impact of cover crops on wildlife is important to understanding the wider effects of this practice. Therefore, I explored raptor use of cover crops in an agricultural region of west-central Indiana.

Within cover-cropped soybean fields, I investigated whether supplementing available perches in the interior of fields would encourage raptors to hunt these fields. With large fields common in intensive row-crop agriculture, available perches for raptors are sparse, limiting the area that is easily hunted. In my study fields, raptors used 82% of the perches and preferentially used the perches farthest into the field interior (200 m). Perch use for each of the 3 primary species (American kestrel, red-tailed hawk, and great horned owl) varied seasonally, with great horned owl perch use peaking at the end of February and early March, red-tailed hawk perch use peaking in mid-March, and American kestrel use peaking in early April. However, despite my success in attracting raptors to artificial perches in intensive row-crop agricultural fields, the overall probability of perch use was low and likely insufficient to appreciably reduce vole populations in fields. It is unclear whether low use was driven by low vole populations, sparse availability of artificial perches, or some other factor. Future studies should investigate if taller, sturdier perches available at elevated densities and for longer periods of time increase raptor use of artificial perches, and whether raptor predation from artificial perches reduces in-field small mammal populations. Although raptor predation alone is likely insufficient to control vole populations, raptors may still be valuable components in an integrated pest management plan.

I also was interested in examining raptor responses to cover crops without the added inducement of artificial perches. I found that American kestrels were strongly associated with cover-cropped fields, whereas red-tailed hawks were strongly associated with woodlots and rough-legged hawks were weakly associated with permanent habitat and avoided woodlots. Within a resource-selection modeling framework, I also assessed how my definition of available points influenced results by testing models with randomly selected points as well as models constrained by the availability of perches for American kestrels and red-tailed hawks. For both of these species, random models identified potential perches, whereas constrained random models identified more subtle habitat selection not included in the random models or, in the case of red-tailed hawks, produced results that conflicted with the random models. Modeling constrained random availability could be useful for modeling habitat use of well-studied species with discrete, easily mapped habitat features. Producers could manage their fields for American kestrels and increase the probability of use by other raptors by adding artificial perches and nest boxes near their fields (Askham 1990).

Northern harriers consume voles in high quantities but were unaffected by artificial perches and were encountered in low numbers on my transects. To gain more insight into their potential impacts, I investigated northern harrier diets in western Indiana. Additionally, I assessed whether the method of counting the number of prey items in a pellet affected my results. As with other studies, voles were the most frequently encountered prey in harrier pellets. However, they occurred at much lower frequencies than reported by other studies in the Midwest (Weller et al. 1955, Mumford and Danner 1974, Bildstein 1978, Walk 1998). Relative to other studies, shrews and western harvest mice were more important components in harrier diets. The method of counting prey contained in a pellet produced no discernible effects on my conclusions concerning pellet contents between months or roosts. Hence, considering teeth in addition to bones may prove beneficial for pellet-based diet studies of other diurnal raptors. My results suggest that vole populations may have been at a low point in their population cycle over the 2 years surveyed, and that northern harriers responded by expanding their diet. Continued monitoring of harrier diets in combination with small mammal trapping could provide useful long-term information regarding harrier responses to vole populations in western Indiana.

Future studies should quantify the reduction in soybean yield related to vole damage as well as the economic and ecological costs associated with different methods of addressing vole

damage. Additionally, future studies should continue to monitor raptor responses to cover crops, particularly during vole population peaks, to assess the degree to which functional and numerical responses can contribute to limitation of voles. Raptor predation has potential as a valuable component of an integrated pest management plan for vole damage to soybeans. To encourage increased raptor use of cover crops, producers can erect artificial perches during the winter to encourage raptor predation of field interiors. They can also add nest boxes or platforms to attract breeding pairs and thus maintain predation pressure throughout the planting and growing season, when newly sown crops are most susceptible to vole predation (Fisher et al. 2014). Producers can also reduce pesticide use and manage vegetation in permanent habitats to increase insect abundances and discourage voles while making them more amenable to raptors (Garratt et al. 2012). Cover-cropping improves soil health and also has the potential to benefit wildlife including raptors. In turn, the intentional use of techniques to attract raptors may partially mitigate damage associated with voles in cover-cropped fields.

APPENDIX A. SUMMARY OF LANDCOVER, RAPTOR OBSERVATIONS, AND MODEL OUTPUT ALONG ROADSIDE TRANSECTS

Table A1: Landcover (%) of the 18 transect routes in 2018 and 2019. Abbreviations: Cover = cover-cropped fields, NoCover = non-cover-cropped fields, PermHab = permanent herbaceous habitat, Woods = woodlots, Dev = development.

Transect	Cover		NoCover		PermHab		Woods		Dev	
	2018	2019	2018	2019	2018	2019	2018	2019	2018	2019
1 Carroll	6.8	4.7	79.8	82.4	3.0	2.7	6.2	6.1	4.1	3.9
2 Clinton	10.5	1.5	81.7	89.8	1.4	1.7	2.6	2.9	3.6	4.0
3 Tipp-Mont	7.0	2.6	83.2	89.7	2.8	2.5	3.2	3.3	3.7	3.7
4 Warren	5.5	3.7	63.6	66.0	9.7	9.1	15.9	16.0	4.6	4.6
5 Benton	4.7	3.8	91.2	92.0	2.0	2.1	0.6	0.6	1.5	1.4
6 White	3.3	2.7	86.3	86.5	4.0	4.5	3.4	3.4	2.7	2.7
7 Clinton	6.5	2.0	69.8	73.9	2.5	3.2	14.1	14.8	6.0	5.9
8 Tipp-Mont	6.9	6.4	73.8	74.0	4.7	5.4	9.9	9.7	4.5	4.3
9 Fountain	10.2	7.6	76.6	79.2	4.7	4.8	5.8	5.8	2.6	2.5
10 Warren	4.7	3.6	83.3	84.3	2.0	2.3	6.9	7.1	3.0	2.7
11 Benton	3.5	7.4	91.4	87.0	1.9	2.4	1.2	1.2	2.0	2.0
12 Tippecanoe	3.1	--	63.8	--	9.4	--	13.6	--	9.0	--
13 Carroll	9.8	2.9	68.9	72.4	3.7	4.4	14.3	16.4	3.0	3.5
14 White	7.1	3.6	70.8	74.0	5.3	5.6	10.9	10.8	5.5	5.5
15 Pulaski	--	5.3	--	63.1	--	6.0	--	20.9	--	4.2
16 Miami	--	9.3	--	60.9	--	9.8	--	15.0	--	4.3
17 Pulaski	--	8.1	--	70.7	--	4.2	--	11.4	--	5.2
18 Miami	--	12.5	--	53.6	--	6.8	--	19.9	--	6.5

Table A2: P values for covariates in the top 1-3 candidate models for each species, scale, and type of available points, as determined with ΔAICc . Abbreviations: RTHA = red-tailed hawk (*Buteo jamaicensis*), AMKE = American kestrel (*Falco sparverius*), RLHA = rough-legged hawk (*Buteo lagopus*), NOHA = northern harrier (*Circus hudsonius*), R = available points selected randomly, CR = available points constrained by hunting mode, Cover = cover-cropped fields, NoCover = non-cover-cropped fields, PermHab = permanent herbaceous habitat, Woods = woodlots, Dev = development, Uline = utility lines, Treeline = linear tree features, Treeperim = outer perimeter of tree canopies. Variables with a p value ≤ 0.05 are bolded and those with $0.05 < p \leq 0.1$ are italicized.

Species	Scale	Available	Model	AICc	ΔAICc	Weight	Variables							
							Cover	NoCover	PermHab	Woods	Dev	Uline	Treeline	Treeperim
AMKE	Transect	CR	1	1263.9	0.00	0.14	<0.001	0.008		0.037				0.076
			2	1264.3	0.38	0.11	0.002	0.008	0.096	0.032				
		R	1	1172.8	0.00	0.16	0.002		0.05	0.025		<0.001		
	Landscape	CR	1	1264.5	0.00	0.09	0.002	0.006	0.104	0.055				
			2	1264.7	0.20	0.08	<0.001	0.006		0.081				0.118
			3	1265.1	0.54	0.07	0.001	0.01				0.132		
		R	1	1191.1	0.00	0.14	<0.001	0.086				<0.001		0.043
			2	1191.5	0.39	0.12	0.002	0.088	0.054			<0.001		
			3	1192.0	0.95	0.09	0.001					<0.001		0.073
RTHA	Transect	CR	1	2123.9	0.00	0.07				0.031	0.060	0.005		
			2	2124.3	0.44	0.06			0.211	0.134	0.044	0.005		
			3	2124.6	0.68	0.05			0.046		0.061	0.002		
		R	1	2104.1	0.00	0.07				<0.001				0.027
			2	2104.2	0.16	0.07			0.175	<0.001				0.049
			3	2105.0	0.88	0.05				<0.001	0.290			0.117
	Landscape	CR	1	2119.0	0.00	0.07				0.023		<0.001		
			2	2119.1	0.18	0.06				0.013	0.178	<0.001		
			3	2119.4	0.44	0.06		0.211		0.016		<0.001		
RLHA	Transect	R	1	2086.7	0.00	0.06			0.153	<0.001	0.026			
			2	2086.8	0.04	0.06				<0.001	0.014			
		CR	1	208.2	0.00	0.12				0.073				
			2	208.9	0.69	0.08			0.250	0.043				
			1	208.8	0.00	0.12			0.058	0.124				
		Landscape	2	209.1	0.37	0.1			0.130					
	CR		1	208.0	0.00	0.11						0.065		
			2	208.9	0.92	0.07			0.299		0.047			
		R	1	209.1	0.00	0.1			0.130					
2	209.2		0.07	0.1			0.049	0.164						
3	209.4		0.29	0.09		0.211		0.016		<0.001				
NOHA	Transect	CR	1	239.0	0.00	0.13								
			2	239.6	0.52	0.1			0.224					
		R	1	237.8	0.00	0.16			0.459					
			2	239.0	1.12	0.09	0.347	0.116						
	Landscape	CR	1	238.7	0.00	0.12			0.125					
			2	239.0	0.35	0.1	0.347	0.116						
		R	1	238.0	0.00	0.15			0.468					
			2	238.8	0.74	0.1	0.261	0.126						

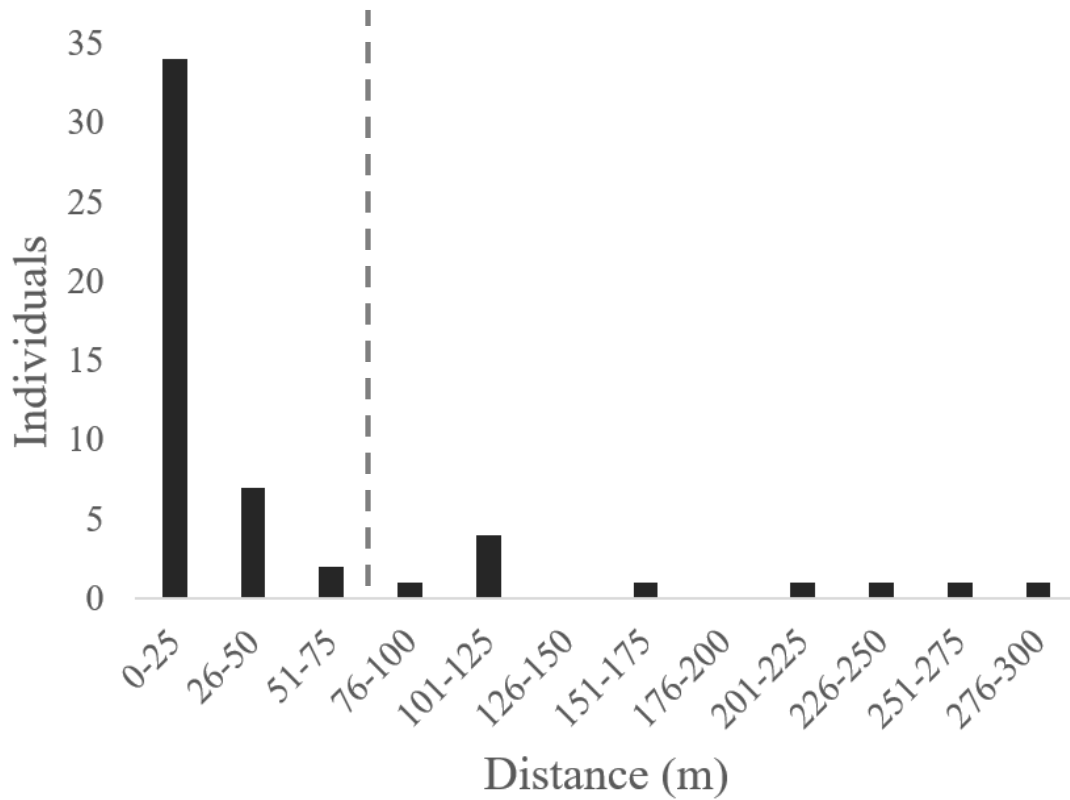


Figure A1: Observed distances (m) for American kestrels (*Falco sparverius*) along transects in west-central Indiana in 2018 and 2019. Only observations of birds that could potentially be hunting (perched or hovering) are included. The distance (m) at which I set the species-specific buffer is indicated with a dotted line.

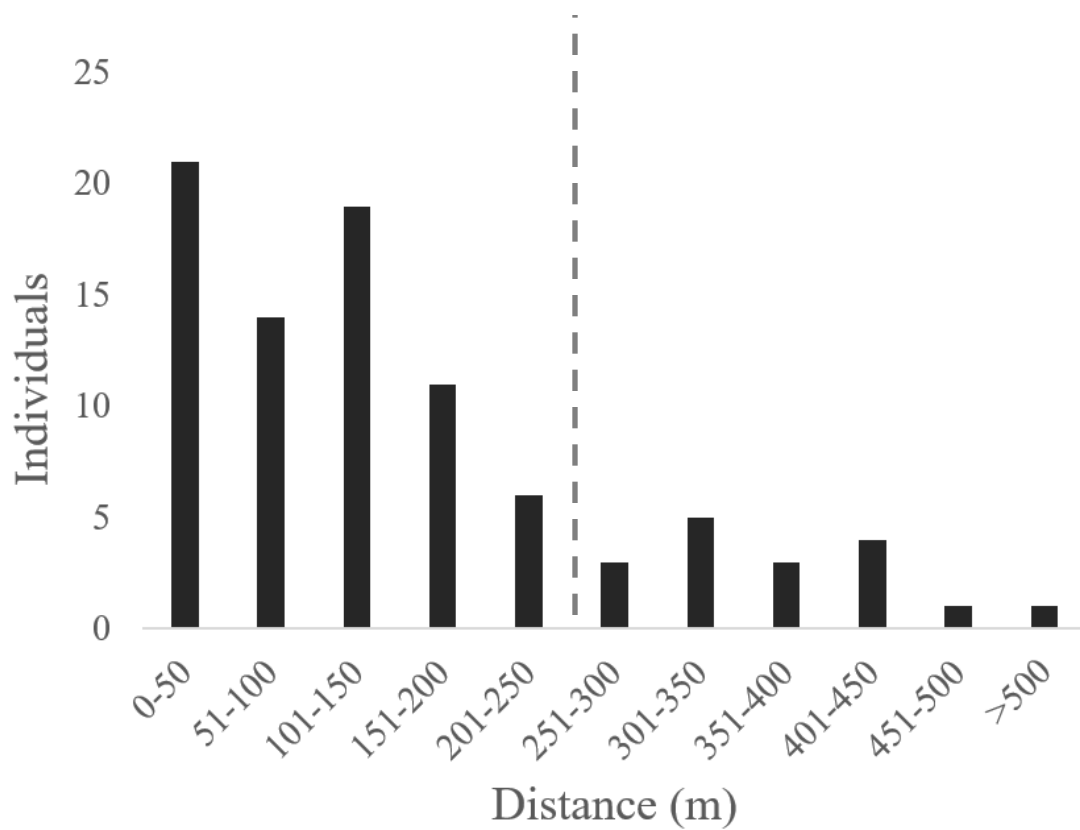


Figure A2: Observed distances (m) for red-tailed hawks (*Buteo jamaicensis*) along transects in west-central Indiana in 2018 and 2019. Only observations of birds that could potentially be hunting (perched) are included. The distance (m) at which I set the species-specific buffer is indicated with a dotted line.

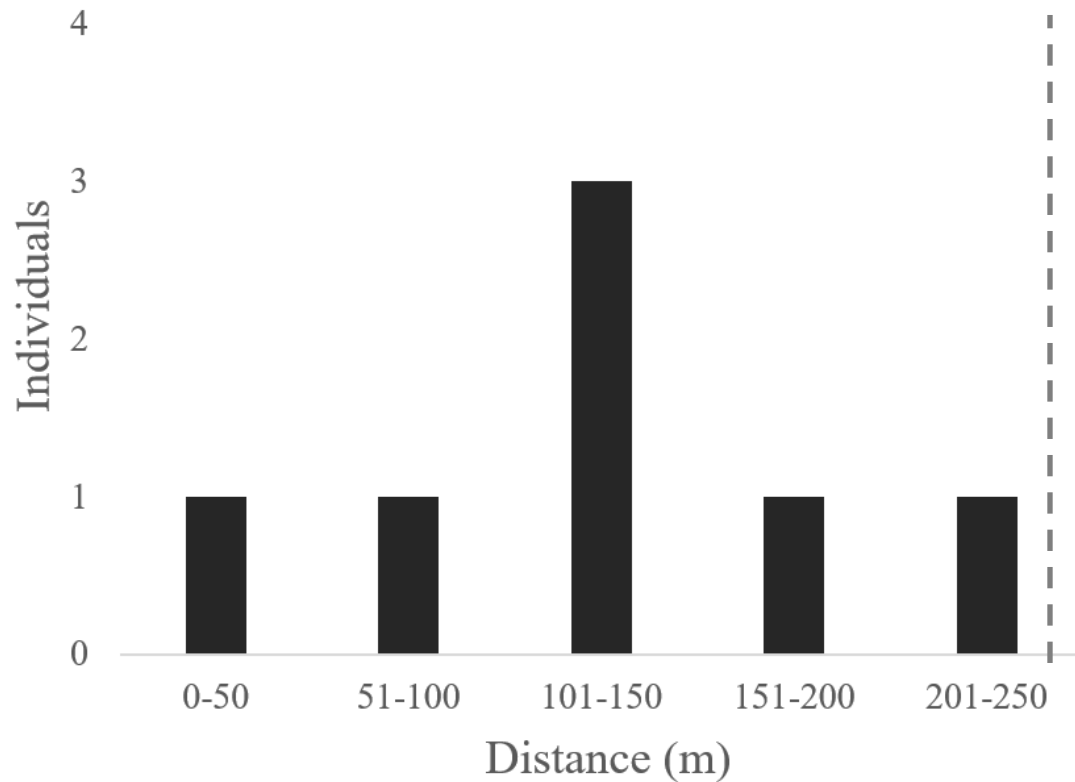


Figure A3: Observed distances (m) for rough-legged hawks (*Buteo lagopus*) along transects in west-central Indiana in 2018 and 2019. Only observations of birds that could potentially be hunting (perched or hovering) are included. The distance (m) at which I set the species-specific buffer is indicated with a dotted line.

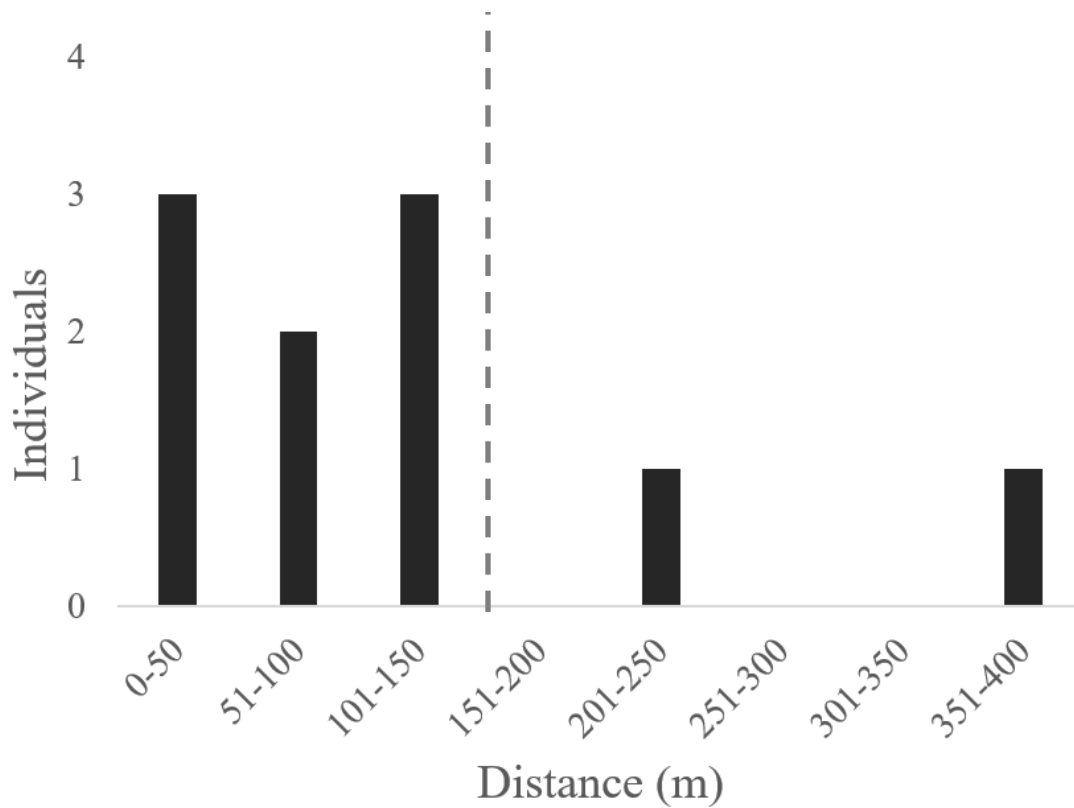


Figure A4: Observed distances (m) for northern harriers (*Circus hudsonius*) along transects in west-central Indiana in 2018 and 2019. Only observations of birds that could potentially be hunting (perched or coursing) are included. The distance (m) at which I set the species-specific buffer is indicated with a dotted line.

APPENDIX B. COMPARISON OF METHODS FOR COUNTING THE MINIMUM NUMBER OF PREY INDIVIDUALS PER PELLET

Table B1: Multivariate analysis of deviance table for a negative binomial multivariate regression model of northern harrier (*Circus hudsonius*) pellets collected at 3 roosts in western Indiana in 2018 and 2019. The influence of 4 different methods of counting the minimum number of individuals (MNI) on conclusions regarding collection and roost site was assessed with 2-way interaction terms. The 4 methods of counting were: if ≥ 1 tooth was present (any teeth), if $\geq \frac{1}{4}$ of the teeth were present, if $\geq \frac{1}{2}$ of the teeth were present, and ignoring teeth (no teeth) on the statistical outcome was assessed. A colon between factors denotes an interaction.

Factor	Residual d.f	d.f.	χ^2	Pr(> χ^2)
Intercept	1979			
Count Method	1976	3	5.1	1
Collection Month	1975	1	70.1	0.001
Roost Site	1972	3	449.1	0.001
Count Method: Collection	1969	3	0.9	1
Count Method: Roost Site	1960	9	3.7	1

Table B2: Percent occurrence of prey groups in northern harrier (*Circus hudsonius*) pellets collected December-April 2018 and 2019 at 4 roost sites in western Indiana. Percent occurrences are presented for all 4 methods of counting teeth toward the minimum number of individuals per pellet: ≥ 1 molar (any teeth), $\geq \frac{1}{4}$ of molars, $\geq \frac{1}{2}$ of molars, not including molars (no teeth).

Year	Roost	# of pellets	Percent Occurrence								Birds	Grasshopper
			<i>Microtus</i>	<i>Peromyscus</i>	<i>Reithrodontomys</i>	<i>Mus</i>	<i>Blarina</i>	<i>Sorex</i>	<i>Cryptotis</i>			
2018	Shadeland	7										
	Any teeth		41.7	16.7	8.3	0	33.3	0	0	0	0	
	≥ ¼ teeth		41.7	16.7	8.3	0	33.3	0	0	0	0	
	≥ ½ teeth		41.7	16.7	8.3	0	33.3	0	0	0	0	
	No teeth		41.7	16.7	8.3	0	33.3	0	0	0	0	
	Kankakee Sands	148										
	Any teeth		47.1	12.0	13.0	0.4	17.8	0.4	0	9.4	0	
	≥ ¼ teeth		46.6	11.1	13.4	0.4	17.6	0.4	0	10.3	0	
2019	≥ ½ teeth		45.1	11.5	13.8	0.4	17.8	0.4	0	10.7	0	
	No teeth		44.0	10.4	14.5	0.4	18.7	0.4	0	11.6	0	
	Kankakee Sands	354										
	Any teeth		48.7	15.7	12.0	1.5	12.5	1.5	0.3	8.1	0.1	
	≥ ¼ teeth		49.1	15.6	11.5	1.5	12.5	1.5	0.3	8.4	0.1	
	≥ ½ teeth		48.5	16.0	10.1	1.7	13.7	1.7	0.2	8.0	0.2	
	No teeth		47.6	16.1	11.2	1.5	12.9	1.5	0.3	9.0	0.1	
	Chinook Mine	223										
	Any teeth		72.9	5.3	3.5	1.4	10	1.6	2.0	6.5	0	
	≥ ¼ teeth		73.5	5.1	3.6	1.0	9.5	1.6	2.0	6.7	0	
	≥ ½ teeth		73.2	4.6	4.1	0.8	7.9	0.8	1.9	6.8	0	
	No teeth		74.0	4.9	3.7	1.0	9.0	1.6	2.0	6.8	0	
	Prophetstown	89										
	Any teeth		56.3	5.7	5.1	0	13.1	1.1	0	18.8	0	
	≥ ¼ teeth		57.0	5.8	4.6	0	12.6	1.1	0	19.0	0	
	≥ ½ teeth		56.7	5.3	4.7	0	12.9	1.2	0	19.3	0	
No teeth		57.1	4.7	4.7	0	12.9	1.2	0	19.4	0		

REFERENCES

- Abercrombie, S. A., J. L. Berl, E. A. Flaherty, and R. K. Swihart. 2017. Seasonal foraging by forest mice enhances loss of weed seeds from crop-field edges. *Northeastern Naturalist* 24:5-17.
- Adams, L. W. 1984. Small mammal use of an interstate highway median strip. *Journal of Applied Ecology* 21:175–178.
- Adams, L. W., and A. D. Geis. 1983. Effects of roads on small mammals. *Journal of Applied Ecology* 20: 403-413.
- Andersson, M., and S. Erlinge. 1977. Influence of predation on rodent populations. *Oikos* 29: 591-597.
- Andres, B. A. 1994. Density and habitat use of hawks wintering at the bluegrass region of Kentucky. *The Kentucky Warbler* 70:57–63.
- Antkowiak, K., and T. Hayes. 2004. Rodent pest control through the reintroduction of an extirpated raptor species. *Endangered Species Update* 21:124–127.
- Artuso, C., C. S. Houston, D. G. Smith, and C. Rohner. 2013. Great horned owl (*Bubo virginianus*). *Birds of North America*. 2.0. Cornell Lab of Ornithology.
- Askham, L. R. 1990. Effect of artificial perches and nests in attracting raptors to orchards. Pages 144–148 *in*. *Proceedings of the Fourteenth Vertebrate Pest Conference*. University of California Davis Press, Sacramento, CA.
- Avenant, N. L. 2005. Barn owl pellets: A useful tool for monitoring small mammal communities? *Belgian Journal of Zoology* 135:39–43.
- Avery, D. M., G. Avery, and A. Roberts. 2002. A contribution from barn owl pellets to known micromammalian distributions in KwaZulu-Natal, South Africa. *African Zoology* 37:131–140.
- Babińska-Werka, J. 1979. Effects of common vole on alfalfa crop. *Acta Theriologica* 24:281–297.
- Baker, J. A., and R. J. Brooks. 1981. Distribution patterns of raptors in relation to density of meadow voles. *The Condor* 83:42–47.
- Bartoń, K. 2019. MuMIn: Multi-model inference. R package version 1.43.6. <https://CRAN.R-project.org/package=MuMIn>.
- Bates, D., M. Maechler, B. Bolker, S. Walker. 2015. Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, 67:1-48. doi:10.18637/jss.v067.i01.

- Bechard, M. J. 1982. Effect of vegetative cover on foraging site selection by Swainson's Hawk. *The Condor* 84:153–159.
- Bechard, M. J., and T. R. Swem. 2002. Rough-legged hawk (*Buteo lagopus*). *Birds of North America*. 2.0. Cornell Lab of Ornithology.
- Belloq, M. I. 1988. Predación de roedores por aves en ecosistemas agrarias. Dissertation, Universidad de Buenos Aires, Buenos Aires, Argentina. [In Spanish.]
- Belloq, M. I., and F. O. Kravetz. 1990. Practical and theoretical implications of perch use for avian predators on rodent populations. *Ecosur* 16:61–67.
- Benson, J. F. 2013. Improving rigour and efficiency of use-availability habitat selection analyses with systematic estimation of availability. *Methods in Ecology and Evolution* 4:244–251.
- Benton, T. G., J. A. Vickery, and J. D. Wilson. 2003. Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology and Evolution* 18:182–188.
- Berl, J. L., E. A. Flaherty, B. J. Danielson, K. F. Kellner, and R. K. Swihart. 2017. Winter ecology of prairie deer mice (*Peromyscus maniculatus bairdii*) in cultivated habitats: Implications for agricultural ecosystem services. *Agriculture, Ecosystems, and Environment* 249:130–136.
- Berl, J. L., K. F. Kellner, E. A. Flaherty, and R. K. Swihart. 2018. Spatial variation in density of white-footed mice along edges in fragmented habitat. *American Midland Naturalist* 179:38–50.
- Best, L. B., H. Campa III, K. E. Kemp, R. J. Robel, M. R. Ryan, J. A. Savidge, H. P. Weeks Jr., and S. R. Winterstein. 1997a. Avian abundance in CRP and crop fields during winter in the Midwest. *The American Midland Naturalist* 139:311–324.
- Best, L. B., H. Campa III, K. E. Kemp, R. J. Robel, M. R. Ryan, J. A. Savidge, H. P. Weeks Jr., and S. R. Winterstein. 1997b. Bird abundance and nesting in CRP fields and cropland in the Midwest: a regional approach. *Wildlife Society Bulletin* 25:864–877.
- Bildstein, K. L. 1976. Behavior of wintering Northern Harriers (*Circus cyaneus hudsonius*) and other raptorial birds at communal roosts in south central Ohio. Master's, The Ohio State University, Columbus, Ohio.
- Bildstein, K. L. 1978. Behavioral ecology of red-tailed hawks (*Buteo jamaicensis*), rough-legged hawks (*B. lagopus*), northern harriers (*Circus cyaneus*), American kestrels (*Falco sparverius*) and other raptorial birds wintering in south central Ohio. Dissertation, The Ohio State University, Columbus, Ohio.
- Bildstein, K. L. 1979. Fluctuations in the number of Northern Harriers (*Circus cyaneus hudsonius*) at communal roosts in south central Ohio. *Journal of Raptor Research* 13:40–46.

- Bildstein, K. L., and M. W. Collopy. 1987. Hunting behavior of Eurasian (*Falco tinnunculus*) and American Kestrels (*Falco sparverius*): A review. Pages 66–82 in. The Ancestral Kestrel. Raptor Research Foundation, Inc. and Macdonald Raptor Research Centre of McGill University.
- Boano, G., and R. Toffoli. 2002. A line transect survey of wintering raptors in the western Po plain of northern Italy. *Journal of Raptor Research* 36:128–135.
- Bobowski, M. M., V. Rolland, and T. S. Risch. 2014. Abundance and distribution of overwintering Red-tailed hawks and American kestrels in an agricultural landscape in northeastern Arkansas. *Journal of Raptor Research* 48:273–279.
- Boyd, K., V. Santos Costa, J. Davis, and C. D. Page. 2012. Unachievable region in precision-recall space and its effect on empirical evaluation. *Proceedings of the 29th International Conference on Machine Learning*. Edinburgh, Scotland.
- Bunn, A. G., W. Klein, and K. L. Bildstein. 1995. Time-of-day effects on the numbers and behavior of non-breeding raptors seen on roadside surveys in eastern Pennsylvania. *Journal of Field Ornithology* 66:544–552.
- Butet, A., N. Michel, Y. Rantier, V. Comor, L. Hubert-Moy, J. Nabucet, and Y. Delettre. 2010. Responses of common buzzard (*Buteo buteo*) and Eurasian kestrel (*Falco tinnunculus*) to land use changes in agricultural landscapes of western France. *Agriculture, Ecosystems and Environment* 138: 152-159.
- Carman, S. F. 2013. Indiana forest management history and practices. Pages 12-23 in R.K. Swihart, M. Saunders, R. A. Kalb, G. S. Haulton, C. H. Michler, editors. *The Hardwood Ecosystem Experiment: a framework for studying responses to forest management*. USDA Forest Service Northern Research Station, Newtown Square, Pennsylvania.
- Carpio, A. J., J. Castro, V. Mingo, and F. S. Tortosa. 2017. Herbaceous cover enhances the squamate reptile community in woody crops. *Journal for Nature Conservation* 37:31–38.
- Cerasoli, F., M. Iannella, P. D'Alessandro, and M. Biondi. 2017. Comparing pseudo-absences generation techniques in Boosted Regression Trees models for conservation purposes: A case study on amphibians in a protected area. *PLoS One* 12:e0187589.
- Churchfield, S. 1990. *The natural history of shrews*. Ithaca, New York, USA, Cornell University Press.
- Cifelli, R., N. Doeksen, P. Kennedy, L. D. Carey, S. A. Rutledge, C. Gimmestad, and T. Depue. 2005. The Community Collaborative Rain, Hail, and Snow Network: Informal education for scientists and citizens. *Bulletin of the American Meteorological Society* 86:1069–1077.

- Clark, R. G., C. Boutin, B. Jobin, D. J. Forsyth, D. Shutler, J. Y. Leeson, O. Olfert, and A. G. Thomas. 2005. Living on the edge: Field boundary habitats, biodiversity and agriculture. Pages 113–133 in A. G. Thomas, editor. Field boundary habitats: Implications for weed, insect and disease management. Volume 1. Topics in Canadian Weed Science, Canadian Weed Science Society - Société canadienne de malherbologie, Sainte-Anne-de-Bellevue, Québec.
- Collopy, M. W., and J. R. Koplin. 1983. Diet, capture success, and mode of hunting by female American kestrels in winter. *The Condor* 85:369–371.
- Colvin, B. A., P. L. Hegdahl, and W. B. Jackson. 1988. Review of non-target hazards associated with rodenticide use in the USA. *EPPO Bulletin* 18:301–308.
- Conner, L. M., M. D. Smith, and L. W. Burger. 2003. A comparison of distance-based and classification-based analyses of habitat use. *Ecology* 84:526–531.
- Conservation Technology Information Center. 2017. Report of the 2016-17 National Cover Crop Survey. Joint publication of the Conservation Technology Information Center, the North Central Region Sustainable Agriculture Research and Education Program, and the American Seed Trade Association. West Lafayette, IN.
- Craighead, J. J., and F. C. Craighead. 1956. *Hawks, Owls and Wildlife*. The Stackpole Company, Harrisburg, PA.
- Cummings, J. H., G. E. Duke, and A. A. Jegers. 1976. Corrosion of bone by solutions simulating raptor gastric juice. *Journal of Raptor Research* 10:55–57.
- Cummings, J. R., and S. H. Vessey. 1994. Agricultural influences on movement patterns of white-footed mice (*Peromyscus leucopus*). *The American Midland Naturalist* 132:209–218.
- Dabney, S. M., J. A. Delgado, and D. W. Reeves. 2001. Using winter cover crops to improve soil and water quality. *Communications in Soil Science and Plant Analysis* 32:1221–1250.
- Dahl, T. E., and G. J. Allord. 1982. History of wetlands in the conterminous United States. Pages 19–26 in National water summary on wetland resources. United States Geological Survey Water Supply Paper 2425, United States Geological Survey.
- Debelica, A., and M. L. Thies. 2009. Atlas and key to the hair of terrestrial Texas mammals. Special publication of The Museum of Texas Tech University, Number 55, Lubbock, USA.
- Duke, G. E., A. A. Jegers, G. Loff, and O. A. Evanson. 1975. Gastric digestion in some raptors. *Comparative Biochemistry and Physiology* 50:649–656.
- Elbroch, M. 2006. *Animal skulls: A guide to North American species*. Stackpole, Mechanicsburg, Pennsylvania, USA.

- Ellis, K. E., and M. E. Barbercheck. 2015. Management of overwintering cover crops influences floral resources and visitation by native bees. *Environmental Entomology* 44:999–1010.
- Erkinaro, E. 1961. The seasonal change of the activity of *Microtus agrestis*. *Oikos* 12:157–163.
- Errington, P. L. 1930. The pellet analysis method of raptor food habits study. *The Condor* 32:292–296.
- Evans, T. R., M. J. Mahoney, E. D. Cashatt, G. de Snoo, and C. J. M. Musters. 2016. Enhancement of linear agricultural areas to provide invertebrates as potential food for breeding birds. *Land* 5.
- Fagerstone, K. A. 2014. Wildlife damage management: changes over the last 40 years and a look at the future. Pages 3–11 in R. M. Timm and J. M. O'Brien, editors. *Proceedings of the 26th Vertebrate Pest Conference*. University of California, Davis, Davis, California.
- Fargallo, J. A., J. Padilla-Martínez, J. Viñuela, G. Blanco, I. Torre, P. Vergara, and L. De Neve. 2009. Kestrel-prey dynamic in a Mediterranean region: The effect of generalist predation and climatic factors. *PLoS ONE* 4:e4311.
- Farmer, G. C., K. McCarty, S. Robertson, B. Robertson, and K. L. Bildstein. 2006. Suspected predation by Accipiters on radio-tracked American kestrels (*Falco sparverius*) in eastern Pennsylvania, U.S.A. *Journal of Raptor Research* 40:294–297.
- Fillooy, J., and M. I. Bellocq. 2007. Respuesta de las aves rapaces al uso de la tierra: un enfoque regional. *Hornero* 22:131-140 [in Spanish].
- Fisher, B., V. Shelton, and T. Bailey. 2014. Vole control. Agronomy “Crib” Notes, USDA NRCS.
- Fithian, W., and T. Hastie. 2013. Finite-sample equivalence in statistical models for presence-only data. *The Annals of Applied Statistics* 7:1917-1939.
- Fleharty, E. D. 1972. Some aspects of small mammal ecology in a Kansas remnant prairie. Pages 97–103 in. *Proceedings of the Midwest Prairie Conference*. Volume 2. Madison, Wisconsin.
- Forren, J. D. 1981. Artificial perch use by raptors on reclaimed surface mines in West Virginia. Thesis, West Virginia University, Morgantown, WV, USA.
- Fox, J. and S. Weisberg 2019. An {R} companion to applied regression, Third Edition. Thousand Oaks CA: Sage. <https://socialsciences.mcmaster.ca/jfox/Books/Companion/>.
- Fuller, M. R., and J. A. Mosher. 1981. Methods of detecting and counting raptors: A review. Pages 235–246 in C. J. Ralph and J. M. Scott, editors. *Estimating Numbers of Terrestrial Birds*. Studies in Avian Biology 6, Cooper Ornithological Society (Allen Press, Inc.), Lawrence, KS.

- Fuller, M. R., and J. A. Mosher. 1987. Raptor survey techniques. Pages 37–65 in B. A. Giron Pendleton, B. A. Millsap, K. W. Cline, and D. M. Bird, editors. Raptor management techniques. National Wildlife Federation, Washington D.C.
- García, F. J., M. Díaz, J. M. de Alba, C. L. Alonso, R. Carbonell, M. L. de Carrión, C. Monedero, and T. Santos. 1998. Edge effects and patterns of winter abundance of wood mice *Apodemus sylvaticus* in Spanish fragmented forests. *Acta Theriologica* 43:255–262.
- García-Heras, M.-S., F. Mougeot, B. Arroyo, G. Avery, M. Avery, and R. E. Simmons. 2017a. Is the black harrier *Circus maurus* a specialist predator? Assessing the diet of a threatened raptor species endemic to southern Africa. *Ostrich* 88:1–8.
- García-Heras, M.-S., F. Mougeot, R. E. Simmons, and B. Arroyo. 2017b. Regional and temporal variation in diet and provisioning rates suggest weather limits prey availability for an endangered raptor. *Ibis* 159:567–579.
- Garibaldi, L. A., L. G. Carvalheiro, S. D. Leonhardt, M. A. Aizen, B. R. Blaauw, R. Isaacs, M. Kuhlmann, D. Kleijn, D. Klein, C. Kremen, L. Morandin, J. Scheper, and R. Winfree. 2014. From research to action: enhancing crop yield through wild pollinators. *Frontiers in Ecology and Evolution* 12:439–447.
- Garratt, C. M., J. Minderman, and M. J. Whittingham. 2012. Should we stay or should we go now? What happens to small mammals when grass is mown, and the implications for birds of prey. *Annales Zoologici Fennici* 49:113–122.
- Getz, L. L., and E. Brighty. 1986. Potential effects of small mammals in high-intensity agricultural systems in east-central Illinois, USA. *Agriculture, Ecosystems, and Environment* 15:39–50.
- Getz, L. L., F. R. Cole, and D. L. Gates. 1978. Interstate roadsides as dispersal routes for *Microtus pennsylvanicus*. *Journal of Mammalogy* 59:208–212.
- Glue, D. E. 1970. Avian predator pellet analysis and the mammalogist. *Mammal Review* 1:53–62.
- Gottschang, J.L. 1965. Winter populations of small mammals in old fields of southwest Ohio. *Journal of Mammalogy* 46:44–52.
- Grande, J. M., P. M. Orozco-Valor, M. S. Liébana, and J. H. Sarasola. 2018. Birds of prey in agricultural landscapes: The role of agricultural expansion and intensification. Pages 449–464 in J. H. Sarasola, J. M. Grande, and J. J. Negro, editors. *Birds of prey: Biology and conservation in the XXI century*. Springer, Cham, Switzerland.
- Grau, J., I. Grosse, and J. Keilwagen. 2015. PRROC: computing and visualizing precision-recall and receiver operating characteristic curves in R. *Bioinformatics* 31:2595–2597. R package version 1.3.1.

- Green, R. E., S. J. Cornell, J. P. W. Scharlemann, and A. Balmford. 2005. Farming and the fate of wild nature. *Science* 307:550–555.
- Griffith, D. R., J. V. Mannering, and W. C. Moldenhaur. 1977. Conservation tillage in the eastern Corn Belt. *Journal of Soil and Water Conservation* 32:20–28.
- Guerrero González, F. M. 1991. Prey of breeding American kestrels in central Wisconsin during 1991. *The Passenger Pigeon* 53:219–222.
- Hafidzi, M. N., and N. Mohd. 2003. The use of the barn owl, *Tyto alba*, to suppress rat damage in rice fields in Malaysia. Pages 274–276 in G. R. Singleton, L. A. Hinds, C. J. Krebs, and D. M. Spratt, editors. *Rats, mice and people: rodent biology and management*. ACIAR Monographs 96, Australian Centre for International Agricultural Research, Canberra, Australia.
- Hall, T. R., W. E. Howard, and R. E. Marsh. 1981. Raptor use of artificial perches. *Wildlife Society Bulletin* 9:296–298.
- Hanberry, B. B., H. S. He, and B. J. Palik. 2012. Pseudoabsence generation strategies for species distribution models. *PLoS ONE* 7:1–12.
- Hartig, F. 2019. DHARMA: Residual diagnostics for hierarchical (multi-level/mixed) regression models. R package version 0.2.4. <https://CRAN.R-project.org/package=DHARMA>.
- Heisler, L. M., C. M. Somers, and R. G. Poulin. 2016. Owl pellets: a more effective alternative to conventional trapping for broad-scale studies of small mammal communities. *Methods in Ecology and Evolution* 7:96–103.
- Hellström, P., J. Nyström, and A. Angerbjörn. 2014. Functional responses of the rough-legged buzzard in a multi-prey system. *Oecologia* 174:1241–1254.
- Hodara, K., and S. L. Poggio. 2016. Frogs taste nice when there are few mice: Do dietary shifts in barn owls result from rapid farming intensification? *Agriculture, Ecosystems, and Environment* 230:42–46.
- Holt, D. W., L. J. Lyon, and R. Hale. 1987. Techniques for differentiating pellets of short-eared owls and northern harriers. *The Condor* 89:929–931.
- Holt, J. B., Jr., 1996. A banding study of Cincinnati area great horned owls. *Journal of Raptor Research* 30:194–197.
- Hopwood, J. L. 2008. The contribution of roadside grassland restorations to native bee conservation. *Biological Conservation* 141:2632–2640.
- Howard, W. E., R. E. Marsh, and C. W. Corbett. 1985. Raptor perches: Their influence on crop protection. *Acta Zoologica Fennica* 173:191–192.

- Indiana State Department of Agriculture. 2019. Cover crop and tillage transect data. <<https://www.in.gov/isda/2383.htm>>. Accessed 24 September 2019.
- Ingold, D. J. 2010. Abundance and habitat use of winter raptors on a reclaimed surface mine in southeastern Ohio. *Ohio Journal of Science* 110:70–76.
- Iturbide, M., J. Bedia, S. Herrera, O. del Hierro, M. Pinto, and J.M. Gutiérrez. 2015. A framework for species distribution modeling with improved pseudo-absence generation. *Ecological Modelling* 312:166-174.
- Jackson, J. A., M. W. Roberts, and J. E. Wright. 1972. The winter roosts and food habits of marsh hawks in east central Mississippi. *The Mississippi Kite* 2:25–32.
- Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. *Ecology* 61:65-71.
- Johnson, C. J., S. E. Nielsen, E. H. Merrill, T. L. McDonald, and M. S. Boyce. 2006. Resource selection functions based on use-availability data: theoretical motivation and evaluation methods. *Journal of Wildlife Management* 70:347-357.
- Jug, D., M. Brmez, M. Ivezić, B. Stipesevic, and M. Sosic. 2008. Effect of different tillage systems on populations of common voles (*Microtus arvalis* Pallas, 1778). *Cereal Research Communications* 36:923–926.
- Kaufman, D. W., and G. A. Kaufman. 1990. Small mammals of wheat fields and fallow wheat fields in north-central Kansas. *Transactions of the Kansas Academy of Science* 93:28–37.
- Kay, B. J., L. E. Twigg, T. J. Korn, and H. I. Nicol. 1994. The use of artificial perches to increase predation on house mice (*Mus domesticus*) by raptors. *Wildlife Research* 21:95–106.
- Keilwagen, J., I. Grosse, and J. Grau. 2014. Area under precision-recall curves for weighted and unweighted data. *PLOS ONE* 9(3): e92209. doi:10.1371/journal.pone.0092209.
- Kellner, K. F., J. E. Duchamp, and R. K. Swihart. 2019. Niche breadth and vertebrate sensitivity to habitat modification: signals from multiple taxa across replicated landscapes. *Biodiversity and Conservation* 28:2647-2667.
- Kim, D. H., F. Chávez-Ramírez, and R. D. Slack. 2003. Effects of artificial perches and interspecific interactions on patch use by wintering raptors. *Canadian Journal of Zoology* 81:2038–2047.
- Kirkpatrick, C. M., and C. H. Conway. 1947. The winter foods of some Indiana owls. *The American Midland Naturalist* 38:755–766.
- Korschgen, L. J. 1957. Food habits of the coyote in Missouri. *The Journal of Wildlife Management* 21:424–435.

- Korschgen, L. J. 1959. Food habits of the red fox in Missouri. *The Journal of Wildlife Management* 23:168–176.
- Kremen, C., N. M. Williams, and R. W. Thorp. 2002. Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Science* 99:16812–16816.
- Labuschagne, L., L. H. Swanepoel, P. J. Taylor, S. R. Belmain, and M. Keith. 2016. Are avian predators effective biological control agents for rodent pest management in agricultural systems? *Biological Control* 101:94–102.
- Leibacher, B., and J. O. Whitaker Jr. 1998. Distribution of the western harvest mouse, *Reithrodontomys megalotis*, in Indiana. *Proceedings of the Indiana Academy of Science* 107:167–170.
- Leyhe, J. E., and G. Ritchison. 2004. Perch sites and hunting behavior of Red-tailed Hawks (*Buteo jamaicensis*). *Journal of Raptor Research* 38:19–25.
- Lin, Y. K., and G. O. Batzli. 1995. Predation on voles: an experimental approach. *Journal of Mammalogy* 76:1003–1012.
- Linden, D. W., A. K. Fuller, J. A. Royle, and M. P. Hare. 2017. Examining the occupancy-density relationship for a low-density carnivore. *Journal of Applied Ecology* 54:2043–2052.
- Littlefield, C. D. 1970. A marsh hawk roost in Texas. *The Condor* 72:245.
- Lobo, J. M., and M. F. Tognelli. 2011. Exploring the effects of quantity and location of pseudo-absences and sampling biases on the performance of distribution models with limited point occurrence data. *Journal for Nature Conservation* 19:1–7.
- Marshall, E. J. P., T. M. West, and D. Kleijn. 2006. Impacts of an agri-environment field margin prescription on the flora and fauna of arable farmland in different landscapes. *Agriculture, Ecosystems, and Environment* 113:36–44.
- Marti, C. D. 1974. Feeding ecology of four sympatric owls. *The Condor* 76:45–61.
- Marti, C. D., M. Bechard, and F. M. Jaksic. 2007. Food habits. Pages 129–152 in D. M. Bird, K. L. Bildstein, editors and D. R. Barber and A. Zimmerman, assistant editors. *Raptor research and management techniques*. Hancock House Publishers Ltd., Blaine, Washington.
- McDonald, K., S. Burnett, and W. Robinson. 2013. Utility of owl pellets for monitoring threatened mammal communities: An Australian case study. *Wildlife Research* 40:685–697.

- McDowell, M. C., and G. C. Medlin. 2009. Using the diet of the barn owl (*Tyto alba*) as an indicator of small vertebrate abundance in the Channel Country, south-western Queensland. *Australian Mammalogy* 31:75–80.
- Meunier, F. D., C. Verheyden, and P. Jouventin. 2000. Use of roadsides by diurnal raptors in agricultural landscapes. *Biological Conservation* 92:291–298.
- Millán de la Peña, N., A. Butet, Y. Delettre, G. Paillat, P. Morant, L. Le Du, and F. Burel. 2003. Response of the small mammal community to changes in western French agricultural landscapes. *Landscape Ecology* 18:265–278.
- Mills, G. S. 1975. A winter population study of the American Kestrel in Central Ohio. *The Wilson Bulletin* 87:241–247.
- Mills, G. S. 1979. Foraging patterns of kestrels and shrikes and their relation to an optimal foraging model. Dissertation, The University of Arizona, Tucson, AZ.
- Morefield, P. E., S. D. LeDuc, C. M. Clark, and R. Iovanna. 2016. Grasslands, wetlands, and agriculture: the fate of land expiring from the Conservation Reserve Program in the Midwestern United States. *Environmental Research Letters* 11.
- Motro, Y. 2011. Economic evaluation of biological rodent control using barn owls *Tyto alba* in alfalfa. Pages 79–80 in 8th European Vertebrate Pest Management Conference. 26-30 September, 2011, Berlin, Germany.
- Muff, S., J. Signer, and J. Fieberg. 2019. Accounting for individual-specific variation in habitat-selection studies: Efficient estimation of mixed-effects models using Bayesian or frequentist computation. *Journal of Animal Ecology* 00:1–13. <https://doi.org/10.1111/1365-2656.13087>
- Mumford, R. E., and C. R. Danner. 1974. An Indiana marsh hawk roost. *Indiana Audubon Quarterly* 52:96–98.
- Muñoz-Pedreros, A., C. Gil, J. Yáñez, and J. R. Rau. 2010. Raptor habitat management and its implication on the biological control of the Hantavirus. *European Journal of Wildlife Research* 56:703–715.
- Muñoz-Pedreros, A., and J. Rau. 2004. Estudio de egagrópilas en aves rapaces. Pages 265–279 in A. Muñoz-Pedreros, J. Rau, and J. Yáñez, editors. *Aves rapaces de Chile*. CEA Ediciones. [in Spanish]
- Murua, R., and J. Rodriguez. 1989. An integrated control system for rodents in pine plantations in central Chile. *Journal of Applied Ecology* 26:81–88.
- Mushet, D. M., J. L. Neau, and N. H. Euliss Jr. 2014. Modeling effects of conservation grassland losses on amphibian habitat. *Biological Conservation* 174:93–100.

- Nad'oo, L., and P. Kaňuch. 2018. Why sampling ratio matters: Logistic regression and studies of habitat use. *PLoS ONE* 13:e0200742.
- Newsome, A. 1990. The control of vertebrate pests by vertebrate predators. *Trends in Ecology and Evolution* 5:187–191.
- Ojwang, D. O., and N. O. Ogiye. 2003. Testing a biological control program for rodent management in a maize cropping system in Kenya. Pages 251–253 in G. R. Singleton, L. A. Hinds, C. J. Krebs, and D. M. Spratt, editors. *Rats, mice and people: rodent biology and management*. ACIAR Monographs 96, Australian Centre for International Agricultural Research, Canberra, Australia.
- Otto, C. R. V., H. Zheng, A. L. Gallant, R. Iovanna, B. L. Carlson, M. D. Smart, and S. Hyberg. 2018. Past role and future outlook of the Conservation Reserve Program for supporting honey bees in the Great Plains. *Proceedings of the National Academy of Science* 115:7629–7634.
- Paz, A., D. Jareño, L. Arroyo, J. Viñuela, B. Arroyo, F. Mougeot, J. J. Luque-Larena, and J. A. Fargallo. 2012. Avian predators as a biological control system of common vole (*Microtus arvalis*) populations in north-western Spain: experimental set-up and preliminary results. *Pest Management Science* 69:444–450.
- Pelz, H. J. 2003. Current approaches towards environmentally benign prevention of vole damage in Europe. Pages 233–237 in G. R. Singleton, L. A. Hinds, C. J. Krebs, and D. M. Spratt, editors. *Rats, mice and people: rodent biology and management*. ACIAR Monographs 96, Australian Centre for International Agricultural Research, Canberra, Australia.
- Peterjohn, B. G., and J. R. Sauer. 1999. Population status of North American grassland birds from the North American Breeding Bird Survey, 1966-1996. *Studies in Avian Biology* 19:27–44.
- Petersen, L. 1979. Ecology of great horned owls and red-tailed hawks in southeastern Wisconsin. Technical Bulletin, Department of Natural Resources, Madison, Wisconsin.
- Phelan, F. J. S., and R. J. Robertson. 1978. Predatory responses of a raptor guild to changes in prey density. *Canadian Journal of Zoology* 56:2565–2572.
- Phillips, S. J., M. Dudik, J. Elith, C. H. Graham, A. Lehmann, J. Leathwick, and S. Ferrier. 2009. Sample selection bias and presence-only distribution models: implications for background and pseudo-absence data. *Ecological Applications* 19:181–197.
- Pompanon, F., B. E. Deagle, W. O. C. Symondson, D. S. Brown, S. N. Jarman, and P. Taberlet. 2012. Who is eating what: Diet assessment using next generation sequencing. *Molecular Ecology* 21:1931–1950.
- Poulin, R. G., T. I. Wellicome, and L. D. Todd. 2001. Synchronous and delayed numerical responses of a predatory bird community to a vole outbreak on the Canadian prairies. *Journal of Raptor Research* 35:288–295.

- Preston, C. R. 1990. Distribution of raptor foraging in relation to prey biomass and habitat structure. *The Condor* 92:107–112.
- Preston, C. R., and R. D. Beane. 1996. Occurrence and distribution of diurnal raptors in relation to human activity and other factors at Rocky Mountain Arsenal, Colorado. Pages 365–374 in D. M. Bird, D. E. Varland, and J. J. Negro, editors. *Raptors in human landscapes: adaptations to built and cultivated environments*. Academic Press, London.
- Preston, C. R., and R. D. Beane. 2009. Red-tailed hawk (*Buteo jamaicensis*). *Birds of North America*. 2.0. Cornell Lab of Ornithology.
- Prieur, A. 2019. Vole damage to cover-cropped soybeans: Exploring options of biological and cultural control. Thesis. Purdue University, West Lafayette, IN, USA.
- R Core Team 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <https://www.R-project.org/>.
- Randall, P. E. 1940. Seasonal food habits of the marsh hawk in Pennsylvania. *The Wilson Bulletin* 52:165–172.
- Reinert, S. E. 1984. Use of introduced perches by raptors: experimental results and management implications. *Journal of Raptor Research* 18:25–29.
- Riffell, S. D., D. Scognamillo, and L. W. Burger. 2008. Effects of the Conservation Reserve Program on northern bobwhite and grassland birds. *Environmental Monitoring and Assessment* 146:309–323.
- Rudolph, S. G. 1982. Foraging strategies of American Kestrels during breeding. *Ecology* 63:1268–1276.
- Säfken, B., D. Rügamer, T. Kneib, and S. Greven. 2018. Conditional model selection in mixed-effects models with caic4. arXiv preprint arXiv:1803.05664.
- Salamolard, M., A. Butet, A. Leroux, and V. Bretagnolle. 2000. Responses of an avian predator to variations in prey density at a temperate latitude. *Ecology* 81:2428–2441.
- Šálek, M., J. Kreisinger, F. Sedláček, and T. Albrecht. 2010. Do prey densities determine preferences of mammalian predators for habitat edges in an agricultural landscape? *Landscape and Urban Planning* 98:86–91.
- Samson, F., and F. Knopf. 1994. Prairie conservation in North America. *BioScience* 44:418–421.
- Sauer, J. R. 1985. Mortality associated with severe weather in a northern population of cotton rats. *American Midland Naturalist* 113:188–189.
- Sauer, J. R., W. A. Link, J. E. Fallon, K. L. Pardieck, and D. J. Ziolkowski Jr. 2013. The North American Breeding Bird Survey 1966–2011: summary analysis and species accounts. *North American Fauna* 79:1–32.

- Sauer, J. R., K. L. Pardieck, D. J. Jr. Ziolkowski, A. C. Smith, M.-A. R. Hudson, V. Rodriguez, H. Berlanga, D. K. Niven, and W. A. Link. 2017. The first 50 years of the North American Breeding Bird Survey. *The Condor* 119:576–593.
- Schnell, G. D. 1968. Differential habitat utilization by wintering rough-legged and red-tailed hawks. *The Condor* 70:373–377.
- Schwartz, C., and Schwartz, E. 2001. *The wild mammals of Missouri*. University of Missouri Press and Missouri Department of Conservation, Columbia.
- Serrentino, P. 1992. Northern harrier. *Circus cyaneus*. Pages 89–117 in K.J. Schneider and D.M. Pence, editors. *Migratory nongame birds of management concern in the northeast*. Newton Corner, MA: U.S. Fish and Wildlife Service.
- Sheffield, L. M., J. R. Crait, W. D. Edge, and G. Wang. 2001. Response of American kestrels and gray-tailed voles to vegetation height and supplemental perches. *Canadian Journal of Zoology* 79:380–385.
- Sherrod, S. K. 1978. Diets of North American Falconiformes. *Journal of Raptor Research* 12:49–121.
- Shoemaker, K. T., L. J. Heffelfinger, N. J. Jackson, M. E. Blum, T. Wasley, and K. M. Stewart. 2018. A machine-learning approach for extending classical wildlife resource selection analyses. *Ecology and Evolution* 8:3556–3569.
- Sinclair, A. R. E., P. D. Olsen, and T. D. Redhead. 1990. Can predators regulate small mammal populations? Evidence from house mouse outbreaks in Australia. *Oikos* 59:382–392.
- Sing T., Sander O., Beerenwinkel N., Lengauer T. 2005. ROCR: visualizing classifier performance in R. *Bioinformatics* 21:7881. <http://rocr.bioinf.mpi-sb.mpg.de>.
- Smallwood, J. A., and D. M. Bird. 2002. American kestrel (*Falco sparverius*). *Birds of North America*. 2.0. Cornell Lab of Ornithology.
- Smith, K. G., S. R. Wittenberg, R. B. Macwhirter, and K. L. Bildstein. 2011. Hen/northern harrier (*Circus cyaneus/hudsonius*). *Birds of North America*. 2.0. Cornell Lab of Ornithology.
- Smith, M., B. Dyer, A. Brodie, W. Hunt, and L. Staples. 2003. Overcoming rat infestation in Australian sugarcane crops using an integrated approach that includes new rodenticide technology. Pages 238–241 in G. R. Singleton, L. A. Hinds, C. J. Krebs, and D. M. Spratt, editors. *Rats, mice and people: rodent biology and management*. ACIAR Monographs 96, Australian Centre for International Agricultural Research, Canberra, Australia.
- Sofaer, H. R., J. A. Hoeting, and C. S. Jarnevich. 2019. The area under the precision-recall curve as a performance metric for rare binary events. *Methods in Ecology and Evolution* 10:565–577.

- Stanton, R. L., C. A. Morrissey, and R. G. Clark. 2018. Analysis of trends and agricultural drivers of farmland bird declines in North America: A review. *Agriculture, Ecosystems, and Environment* 254:244–254.
- Steenhof, K., and M. N. Kochert. 1988. Dietary responses of three raptor species to changing prey densities in a natural environment. *Journal of Animal Ecology* 57:37–48.
- Swihart, R. K., T. M. Gehring, M. B. Kolozsvary, and T. E. Nupp. 2003. Responses of “resistant” vertebrates to habitat loss and fragmentation: the importance of niche breadth and range boundaries. *Diversity and Distributions* 9:1-18.
- Swihart, R. K., J. J. Lusk, J. E. Duchamp, C. E. Rizkalla, and J. E. Moore. 2006. The roles of landscape context, niche breadth, and range boundaries in predicting species responses to habitat alteration. *Diversity and Distributions* 12:277-287.
- Swihart, R. K., and N. A. Slade. 1985. Seasonal use of brush piles by the hispid cotton rat (*Sigmodon hispidus*). *Journal of Mammalogy* 66:577-580.
- Swihart, R. K., and J. Verboom. 2004. Using ecologically scaled landscape indices to assess biodiversity consequences of land-use decisions. Pages 81-101 in R. K. Swihart and J. E. Moore, editors. *Conserving biodiversity in agricultural landscapes*. Purdue University Press.
- Taylor, S. 1996. Rodent damage control in commercial forestry in the Natal Midlands, South Africa. Thesis, University of Natal, Pietermaritzburg, South Africa.
- Toland, B. R., and W. H. Elder. 1987. Influence of nest-box placement and density on abundance and productivity of American Kestrels in central Missouri. *The Wilson Bulletin* 99:712–717.
- Torre, I., A. Arrizabalaga, and C. Flaquer. 2004. Three methods for assessing richness and composition of small mammal communities. *Journal of Mammalogy* 85:524–530.
- USDA National Agricultural Statistics Service. 2019. U.S. Census of Agriculture: United States summary and state data. Volume 1. Report AC-17-A-51. USDA: Washington D.C.
- USDA National Agricultural Statistics Service Cropland Data Layer. 2019. 2018 Indiana Cropland Data Layer [Online]. Available at <https://nassgeodata.gmu.edu/CropScape/> Accessed 1 October 2019. USDA-NASS, Washington, DC.
- Valdez, U., S. Robertson, B. Robertson, and K.L. Bildstein. 2000. Nestbox use by American kestrels (*Falco sparverius*) and European starlings (*Sturnus vulgaris*) in eastern Pennsylvania. *Pennsylvania Birds* 14:150-153.
- Valentini, A., F. Pompanon, and P. Taberlet. 2009. DNA barcoding for ecologists. *Trends in Ecology and Evolution* 24:110-117.

- van Strien, A. J., D. L. Bekker, M. J. J. La Haye, and T. van der Meij. 2015. Trends in small mammals derived from owl pellet data using occupancy modelling. *Mammalian Biology* 80:340–346.
- Van Vuren, D., and K. S. Smallwood. 1996. Ecological management of vertebrate pests in agricultural systems. *Biological Agriculture and Horticulture* 13:39–62.
- Veilleux, J. P., J. O. Whitaker, Jr., and E. A. Vincent. 1998. Mammals of the Newport Chemical Depot, Vermillion County, Indiana. *Proceedings of the Indiana Academy of Science* 107:91–104.
- Villamil, M. B., G.A. Bollero, R.G. Darmody, F.W. Simmons, and D.G. Bullock. 2006. No-till corn/soybean systems including winter cover crops: Effects on soil properties. *Soil Science Society of America Journal* 70:1936–1944.
- Vukovich, M., and G. Ritchison. 2008. Foraging behavior of Short-eared Owls and Northern Harriers on a reclaimed surface mine in Kentucky. *Southeastern Naturalist* 7:1–10.
- Walk, J. W. 1998. Winter roost sites of northern harriers and short-eared owls on Illinois grasslands. *Journal of Raptor Research* 32:116–119.
- Wang, Y., U. Naumann, D. Eddelbuettel, J. Wilshire and D. Warton. 2019. mvabund: Statistical methods for analysing multivariate abundance data. R package version 4.0.1. <https://CRAN.R-project.org/package=mvabund>.
- Warton, D. I., and L. C. Shepherd. 2010. Poisson point process models solve the “pseudo-absence problem” for presence-only data in ecology. *Annals of Applied Statistics* 4:1383–1402.
- Warton D. I., S. Wright, and Y. Wang. 2012. Distance-based multivariate analyses confound location and dispersion effects. *Methods in Ecology and Evolution* 3:89–101.
- Warton D. I., L. Thibaut, Y. A. Wang. 2017. The PIT-trap - A "model-free" bootstrap procedure for inference about regression models with discrete, multivariate responses. *PLoS One* 12:e0181790.
- Weller, M. W., I. C. Adams, Jr., and B. J. Rose. 1955. Winter roosts of marsh hawks and short-eared owls in central Missouri. *The Wilson Bulletin* 67:189–193.
- Whitaker, J. O., Jr., 2004. Mammals of Willow Slough Fish and Wildlife Area, Newton County, Indiana. *Proceedings of the Indiana Academy of Science* 113:87–93.
- Whitaker, J. O., Jr., and A. K. Chamberlain. 2011. Mammals of the Goose Pond Fish and Wildlife Area, Greene County, Indiana. *Proceedings of the Indiana Academy of Science* 119:153–157.
- Whitaker, J. O., Jr., and R. E. Mumford. 2009. *Mammals of Indiana*. Second edition. Indiana University Press, Bloomington, IN, USA.

- Whitaker, J. O., Jr., and G.R. Sly. 1970. First record of *Reithrodontomys megalotis* in Indiana. *Journal of Mammalogy* 51:381.
- Whitaker, J. O., Jr., B. L. Walters, and S. Johnson. 2007. Terrestrial mammals of the Naval Support Activity (NSA Crane) in Martin County, and of the Crane division of the Naval Surface Warfare Center (Glendora site) in Sullivan County, Indiana. *Proceedings of the Indiana Academy of Science* 116:84–89.
- White, J. G., J. Wilson, and K. Korskins. 2003. Habitat manipulation and rodent damage control: reducing rodent damage in Australian macadamia orchards. Pages 213–216 in G. R. Singleton, L. A. Hinds, C. J. Krebs, and D. M. Spratt, editors. *Rats, mice and people: rodent biology and management*. ACIAR Monographs 96, Australian Centre for International Agricultural Research, Canberra, Australia.
- Wilcoxon, C. A., J. W. Walk, and M. P. Ward. 2018. Use of cover crop fields by migratory and resident birds. *Agriculture, Ecosystems, and Environment* 252:42–50.
- Wilson, A., M. Brittingham, and G. Grove. 2010. Association of wintering raptors with Conservation Reserve Enhancement Program grasslands in Pennsylvania. *Journal of Field Ornithology* 81:361–372.
- Wilson, J., and D. Whisson. 1993. The management of rodents in north Queensland cane fields. Centre for Biological Population Management, School of Life Science, Queensland University of Technology, Brisbane, Australia.
- Witmer, G., M. Pipas, P. Burke, D. Rouse, D. Dees, and K. Mancini. 2008. Raptor use of artificial perches at natural areas, City of Fort Collins, Colorado. *The Prairie Naturalist* 40:37–42.
- Wolff, J. O., T. Fox, R. R. Skillen, and G. Wang. 1999. The effects of supplemental perch sites on avian predation and demography of vole populations. *Canadian Journal of Zoology* 77:535–541.
- Wong, E. L., and S. M. Kross. 2018. Effects of perch location on wintering raptor use of artificial perches in a California vineyard. *Journal of Raptor Research* 52:250–256.
- Wooster, L. D. 1936. The contents of owl pellets as indicators of habitat preferences of small mammals. *Transactions of the Kansas Academy of Science* 39:395–397.
- Worm, A. J., M. M. Bobowski, and T. S. Risch. 2013. Perch-type characteristics of overwintering red-tailed hawks (*Buteo jamaicensis*) and American kestrels (*Falco sparverius*). *Journal of the Arkansas Academy of Science* 67:159–162.
- Yahner, R. H. 1983. Population dynamics of small mammals in farmstead shelterbelts. *Journal of Mammalogy* 64:380–386.
- Yan, L., and D. P. Roy. 2016. Conterminous United States crop field size quantification from multi-temporal Landsat data. *Remote Sensing of Environment* 172:67–86.