A MULTIDIMENSIONAL PERSPECTIVE ON WILDLIFE CONSERVATION AND MANAGEMENT: PEOPLE, POPULATIONS, AND HABITAT

By

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ABSTRACT

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As natural resource managers the decisions we make today can affect our tomorrow. Therefore, we have to have a diverse skillset to address a variety of issues to sustain our natural resources for future generations. As a manager there are three primary responsibilities 1) people, 2) habitat, and 3) populations. My dissertation is a combination of research topics that captures all three responsibilities of a natural resource manager. In Chapter 1, I assessed current natural resource students surveys to determine what motivated them to choose natural resources as a career. I analyzed their family backgrounds and the students current interest to see what motivating factors influenced their career decision. I found that students whose families participated in consumptive activities were most influential for the student in pursuing a natural resource career and that these students tended to choose natural resources as a major between the 11th and 12th grade. In Chapter 2, I investigated factors that affect trapping success of the American marten (*Martes americanus*). I wanted to determine what weather and ecological factors had an influence on trapper success. I found that more younger male marten were captured (≤2.5 years old) and harvest of marten per year varied substantially. Temperature and precipitation during the months of October, November, and December was found to have a correlation with marten harvest. North American hardwoods forest was the only existing vegetation type from LANDFIRE found significant. I also found that density of roads had an effect on trapper harvest due to accessibility into certain forested areas. Lastly, in Chapter 3 I evaluated songbird species responses to timber harvest. The extent of my dissertation research encapsulates the three

responsibilities of a natural resource manager (people, habitat, and populations) and why it is important for managers to become trained in a diverse set of skills and have a certain level of understanding of all aspects of our natural resources. Results from this research assist colleges and managers in their decision making processes of building the next generation of managers and sustaining our natural resources.

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PREFACE

Each chapter within this dissertation was drafted as a stand-alone manuscript for publication in a peer-reviewed journal. Chapter 1 is currently in press with *The Wildlife Society Bulletin*. Chapter 2 and Chapter 3 have not been submitted for publication. Due to copyright restrictions, Chapter 1 is not included in this dissertation. For chapter 1, I provide a brief summary of the article. Although I am listed as the sole author and use the pronoun I throughout this dissertation, each chapter was a collaborative effort and all associated manuscripts and chapters will include one or more co-authors when submitted for peer-review.

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INTRODUCTION

Decisions made by wildlife managers today have long-lasting effects. Wildlife management in the 21st century is highly complex (Ascher 2001; Cilliers et al.2013), requiring diverse skills for effective movement of conservation and sustainability in a positive direction. Broadly, wildlife managers have three primary responsibilities 1) people, 2) habitat, and 3) animal populations. In North America the public plays a critical, active role in wildlife conservation by providing funding (through taxation and license sales; Organ et al. 2012), interacting with public agencies that serve as wildlife trustees (Organ et al. 2012), and by voting (Kilpatrick and Walter 1997). Habitat is the foundation of wildlife population performance, and managers frequently manipulate habitats to affect populations (Morrison et al. 1992; Messmer 2009). The ultimate indicator of successful wildlife conservation and sustainable management is population performance, best expressed as long-term population growth rate (Lindenmayer 2000). Managers coordinate the actions of people, and manipulate habitats and populations to affect long-term population growth rate to meet some objective. For overabundant wildlife causing property damage, the objective is likely to reduce populations and mitigate damage (e.g., Conover 2001). For rare species, the objective is likely to increase distribution, numbers, and population growth rate (e.g., Wydeven et al. 2009). Collectively, people, habitat, and animal populations form the "three-legged stool" of wildlife management (Leopold 1987).

My dissertation is a combination of research topics that include components of the "three-legged stool" of wildlife management. An underlying theme is the connection humans have with their environments. In Chapter 1, I assessed what motivated current natural resource students to choose natural resources as a career, recognizing that younger generations in the United States may not relate to the North American Model of Wildlife Conservation. Younger generations in the United States are increasingly urbanized (Manfredo et al. 2003), often at the expense of utilitarian connections to wildlife and under-appreciation for some tools used to manage animal populations like hunting and trapping (Manfredo et al. 2003). However, younger generations have a close non-utilitarian connection to wildlife and the environment (Manfredo et al. 2003), offering a substantial conservation opportunity. This places organizations relying on hunting, trapping, and fishing license sales to implement wildlife conservation (e.g., state resource agencies) in a difficult position. On one hand, funding for the organization is tied to an increasingly outdated interest in wildlife (for example) so implementation of programs and activities must maintain or attempt to increase those interests. Conversely, those programs may alienate younger generations, potentially missing a critical opportunity to engage the broader public in conservation. Ultimately, wildlife management organizations recognize that employees must represent diverse and value public interests to remain relevant in the 21st century. In my first chapter, I analyzed family backgrounds and current interests of student enrolled in natural resource programs in the United States to understand motivating factors that influenced their apparent career decision. The premise was to lay a foundation for understanding the future employee pool responsible for implementing wildlife conservation, guide student recruiting into the profession, and offer suggestions to improve college natural resource course offerings.

Managers use harvest regulations to achieve habitat or animal population objectives and to influence public participation and interest (e.g., Riley et al. 2002; Lauber et al. 2012). Factors affecting participation and effort in wildlife harvest by the public are multi-faceted and complex in space, time, and circumstance (Riley et al. 2002; Enck 2006). For example, weather conditions (Obbard et al. 1999), state of the economy (Obbard et al. 1999), and social or cultural demographics (Miller and Vaske 2003) affect hunting participation and effort. Given that harvest regulations are a key element of many wildlife conservation programs, increased understanding of factors that motivate people to participate and be successful benefit management organizations. In Chapter 2, I investigated factors that effected trapping success of American marten (*Martes americanus*) in Michigan. I sought to determine what factors could potentially be manipulated by wildlife managers to affect harvest success. I evaluated factors directly controlled by managers (e.g., distance from maintained roads), those related to socio-economic forces beyond the management organization (e.g., pelt prices), and factors that were uncontrollable (e.g., weather). As such, this chapter contains all the elements of the "three-legged stool" of wildlife management; how trapping success (a measure of trapper involvement and effort) influenced marten populations under varying habitat conditions.

Wildlife conservation programs often include some form of habitat management. In some instances, wildlife conservation can be included in practices commonly used for resource extraction like timber harvest. In forested regions of North America, managers commonly use timber harvest purposefully to provide wildlife habitat (e.g., Linden and Roloff 2013). In other instances, timber extraction is the primary management objective but wildlife considerations are included (Blinn and Kilgore 2001; Demarais et al. 2017). One way to include wildlife in timber harvest objectives is through retention forestry, where managers retain elements of the pre-harvest forest to increase structural complexity (Fedrowitz et al. 2014; Mori and Kitagawa 2014). Retention forestry is particularly relevant in silvicultural systems like clearcutting, where managers remove all merchantable trees. Clearcutting is a common practice used on aspen (*Populus* spp.) forests in Michigan, and foresters are required to retain unharvested trees to provide wildlife habitat (Bielecki 2012). Retention of these trees comes at a cost through lost timber revenues, potentially increased safety hazards for equipment operators, and potential loss of forest regeneration. Hence, knowing that retention forestry is having a positive effect on wildlife populations is a critical information need. Otto and Roloff (2012) found that retention forestry in aspen clearcuts of Michigan had minimal effect on bird occupancy probability, and they surmised that landscape context was an important consideration. In Chapter 3, I evaluated how songbird occupancy related to structural retention in aspen clearcuts using a hierarchical model that included patch- and landscape-factors, with the goal of better understanding how landscape context affected the function of retained structures as bird habitats. Although this chapter focuses on habitat management and how it affected a population parameter (i.e., occupancy), the results inform decisions made by managers and policy-makers (i.e., people).

My dissertation research encapsulated the three responsibilities of a wildlife manager (people, habitat, and populations), highlight the importance of multi-dimensional training and experiences for managers. I also used sound sampling designs and a suite of modeling approaches to generate scientific evidence, consistent with efforts to infuse science into natural resources decision-making (Mills and Clark 2001). Results from my research offer insights into how people decide to embark on wildlife careers, how people respond to socio-economic and environmental factors to manipulate wildlife populations, and how habitat management decisions by people can influence wildlife populations.

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CHAPTER 1

NATURAL RESOURCE UNDERGRADUATE STUDENTS IN THE NEW MILLENNIUM

Historically, undergraduate college students enrolled in natural resources programs came from rural backgrounds and regularly participated in fishing, hunting, and trapping (i.e., consumptive activities). Student demographics shifted considerably over the last 30 years, with more natural resources students coming from urban backgrounds with lower levels of engagement in consumptive activities. Some stakeholders and employers are concerned that misalignment between student participation in consumptive and nonconsumptive wildlife-related activities and priorities of natural resource management authorities might result in contradicting views on consumptive activities. We sought to understand the background, participation in wildlife-related activities, and career decision-making process of undergraduate college students currently enrolled in natural resources programs in the early 21st century. We conducted an online survey of students enrolled in member universities and colleges of the National Association of University Fisheries and Wildlife Programs (NAUFWP). We examined the ways in which demographic information (including personal and family characteristics) and participation in outdoor activities shaped student; *i*) decisions to pursue natural resources training in college, and *ii*) career choices. We received 1,376 undergraduate respondents (570 males, 806 females) representing universities and colleges in 29 US states. Responding students were primarily Caucasian with the majority between 18 and 22 years old. Most identified that they regularly (>11 hours/month) spent time outdoors, and just over half acknowledged participating in hunting and fishing. Participation in hunting, fishing, and farming were lower among students identifying as female compared to those identifying as male. We also found that family participation in hunting and fishing was most influential on both student involvement in consumptive activities and their decisions to pursue natural resources careers. Students with family participation in consumptive activities made natural resources career decisions in high school compared to students whose families did not participate in consumptive activities who waited until early college. Our study indicates that 50% of college students enrolled in natural resource programs do not participate in consumptive activities. Our study also highlights the importance of family participation in consumptive activities in determining school and career outcomes, with implications for student recruitment and retention practices for natural resource academic programs and employers in the early 21^{st} century. This work is currently *in press* and will appear in the Wildlife Society Bulletin.

CHAPTER 2

FACTORS AFFECTING TRAPPING SUCCESS FOR AMERICAN MARTEN

Abstract

Wildlife managers often rely on harvest data (e.g., number of animals harvested, trapper effort) collected from trappers to make population-level inferences and establish regulations for furbearers. I used data from marten harvested across the upper peninsula of Michigan from 2001-2018 to determine if individual demographics (sex ratio and age), weather, land cover types, pelt prices, or roads played a role in successful capture of marten. I used a Bayesian geostatistical approach that accounted for spatial and temporal autocorrelation. My sample included 3,882 martens (2008 males and 825 females) with known year of harvest, gender, and harvest location to the nearest 2.6 km^2. I used PRISM and LANDFIRE to compile weather and land cover information, respectively. Relative to the global mean the harvest sample included significantly more young marten (≤2.5 years old) and fewer older marten (≥8.5 years-old), fewer females, and harvest varied substantially by year (differed from global mean 11 of 18 years). Pelt price and distance to maintained roads were not significant predictors, but I found that temperature and precipitation in October, November, and December correlated with marten harvest. Warmer temperatures in October and November, yet colder temperatures in December associated with higher marten harvests. More precipitation in October and December, but drier Novembers associated with higher marten harvests. Crown bulk density (CBD) in the 2.6 km^2 containing the harvest location generally had a positive effect on harvest; no other LANDFIRE variables were strongly influential. Harvest of marten from a section the prior year was a strong positive predictor of marten harvest. Pelt price and distance to maintained roads were not important. My results indicate that weather conditions leading up to and during the trapping season for marten

are important determinants of harvest success, offering some short-term forecasting potential. My results also indicate that the coarse resolution of harvest location limits the ability to model harvest success from readily available land cover data.

Introduction

Harvest data from game species are important sources of ecological and evolutionary information to wildlife managers (Mysterud et al. 2001; Yoccoz et al. 2002; Bonenfant et al 2003; Carranza et al. 2004; Martinez et al. 2005). Managers often use harvest data to assess population dynamics (Getz and Haight 1989; Solberg et al., 1998; Nielsen et al., 2004; Allen et al. 2018). Anglers, hunters, and trappers tend to select wildlife for particular phenotypic characteristics (Pelletier et al. 2012; Allen et al. 2018), such as large fish (JØrgensen et al. 2007), or male animals during hunting (Coltman et al. 2003; Martinez et al. 2005; Mysterud et al. 2006; Garel et al. 2007; Mysterud 2011; Pelletier et al. 2012). Preferential selection of sexually dimorphic mammals is common among ungulate hunters, but less common for certain furbearer species (Allen et al. 2018).

Selective harvests affect wildlife population sizes, age structures, and sex ratios (Markgren 1969; Crete, Taylor & Jordan 1981; Bubenik 1987; Ginsberg and Milner-Gulland 1994; Saether et al. 2003), and reduces life expectancy for selected cohorts (Langvatn and Loison 1999). For example, hunting can increase mortality and reduce life expectancy of older animals, making them rare in the population (Bubenik 1987; Courtois 1989), with ramifications for reproduction (Proaktor et al. 2007). Hunting can also play a significant role in skewing sex ratios in favor of females for some wildlife groups (e.g., ungulates; Markgren 1969; Crête et al. 1981; Bubenik 1987), potentially indirectly affecting female space use, survival, and reproductive success. For example, in the absence of sexually mature or dominant males, females may increase home range size during mating season potentially exposing them to higher risk (Claveau and Courtois 1992). Additionally, males for some species (e.g., bears (*Ursus* spp.)) may be more susceptible to harvest because of relatively larger home range sizes and risky behaviors compared to females (Mysterud et al. 2006; Bischof et al. 2009; Mysterud 2011). Ballard et al. (1991) concluded that lack of males reduces reproduction in moose (Thomason 1991, Stephenson et al. 1995), thereby decreasing fecundity, recruitment, and ultimately population growth. Low numbers of mature males can lead to sociobiological problems (Bubenik 1987; Crichton 1992), such as breakdown of dominance hierarchies (Laurian 2000). Insights into factors affecting selection of specific animals should inform harvest regulations (Mysterud et al. 2011; Allen et al. 2018).

Little information exists on preferential selection of furbearers or environmental factors that affect composition of furbearer harvests. Activity of trappers and hunters who pursue furbearers (collectively called furharvesters) and composition of animals harvested is a complex expression of intrinsic (e.g., physical, experience, or equipment capabilities of individual trappers or hunters) and extrinsic (e.g., market prices, furbearer populations, weather) factors, complicating management. The primary reason to pursue furbearers is to sell or use the fur or other body parts (Elsken-Lacy et al. 1999; Hiller et al. 2011; Kapfer and Potts, 2012). Effort expended by furharvesters directly links to market prices (Gehrt et al. 2002; Poole 2003; Hiller et al. 2011; Elsken-Lacy et al. 1999), but maybe mediated by weather and landscape factors. In northern regions, for example, snow depth, temperature, and restricted road access hinders trapper success (Strickland 1994; Hiller et al. 2011; Kapfer and Potts 2012). Understanding spatio-temporal variation in furbearer harvests as a function of weather and landscape factors can assist managers in spatially allocating trapping effort or predicting harvest amounts.

American marten (*Martes americanus*) are high-valued furbearers trapped across North America. In some areas (e.g., northern Michigan and Wisconsin), marten populations are highly sensitive to habitat and harvest pressures (Hiller et al. 2011). Marten are habitat specialists and indicator species (Buskirk and McDonald 1989; Watt et al. 1996; Fecske et al. 2002), sensitive to amounts of mature forest cover (Buskirk and Powell 1994; Fecske et al., 2002; Cushman et al. 2011), anthropogenic activity (Davis 1983; Soukkala 1983; Hodgman et al. 1997; Robitaille and Aubry 2000), and forest cover type (Cushman et al., 2011). American marten are lean, with small body mass resulting in home ranges three times the size predicted for terrestrial carnivores based on allometrics (Lindstedt et al. 1986). Additionally, males travel farther than females (Chapin et al. 1998; Silet 2017), making males more vulnerable to harvest (Soukkala 1983; Archibald and Jessup 1984; Fortin and Cantin 1994). Little information on age specific mortality rates among sexes exists for marten (Hodgman et al. 1994; Bull and Heather 2001). Trappers catch marten with body-grip traps (e.g., conibears), foothold traps, and cage traps (Frawley 2017), resulting in opportunities for selective harvest pressures (primarily with cage traps). Factors influencing trapper efforts (Daigle et al. 1998) and success remain poorly understood by wildlife managers and should be further investigated (Ahlers 2016).

I studied spatial and temporal dimensions of harvested marten across the Upper Peninsula (UP), Michigan, USA. My goal was to understand factors affecting locations of marten harvests, assuming that these locations represent a decision by trappers on where to focus effort. I sought to understand demographics of harvested marten and weather, landscape, and pelt price as factors affecting harvests. I predicted that marten harvest would be higher in areas with less forest disturbance, in mature forest types (particularly those dominated by conifers), in areas with high canopy cover and canopy bulk density, and closer to maintained roads. I modeled these relationships for an 18-year harvest data set. My results improve understanding of marten harvest dynamics with implications for managing this valuable furbearer.

Study Area

My study area encompassed the Upper Peninsula (UP) of Michigan (approximately 42,600 km^2; Fig. 1), which is approximately 85% forested and <5% agricultural lands (United States Department of Agriculture, Natural Resources Conservation Service 2006), with 40% in public ownership (state and federal) (Skalski et al. 2011). Major forest types include aspen (*Populus* spp.), hemlock (*Tsuga canadensis*), maple (*Acer* spp.), spruce (*Picea* spp.), birch (*Betula* spp.) and fir (*Abies balsamea;* Sommers 1977, Hiller et al. 2011). Approximately 308,000 people lived in the UP in 2009 (U.S. Census Bureau 2009). Annual precipitation in the western portion generally ranges from 660 to 940 mm, and from 762 to 914 mm in the eastern portion (United States Department of Agriculture, Natural Resources Conservation Service 2006), and substantially varies in space during the trapping season (Hiller et al. 2011) from lake effect snows. Annual temperatures in the west vary from ~3 ˚C to ~7 ˚C and in the east from ~4˚C to ~6 ˚C. The western portion has greater topographic relief, with elevations ranging from 184m to 606m, whereas the eastern portion has low relief, is poorly drained, and includes peat bogs and extensive swamps (Albert 1995; Hiller et al. 2011).

Methods

Marten occur throughout the Upper Peninsula, with the majority of trapping records coming from northerly latitudes (Fig. 1). Legal trapping methods include foothold, body-gripping and cage traps (Hiller et al. 2011; Skalski et al. 2011). Once a marten is harvested trappers are required by state and Tribal regulations to have the fur sealed (Skalski 2011). Upon registration additional data are collected including harvest date, sex (estimated at the time of registration), and location (to the nearest 2.6 km^2). A tooth is also extracted for age estimation using cementum annuli (Poole et al. 1994). I obtained marten harvest records from the MDNR from 2001-2018.

Marten reduce movements during extremely low temperatures and high precipitation (Carroll 2007; Hiller et al. 2011), therefore, I calculated monthly maximum temperatures and precipitation for October, November, and December from data provided by the PRISM Climate Group (PRISM 2018). These months correspond to the early part of the Tribal trapping season (runs from October to March), and the non-Tribal season that is restricted to December. The MDNR started consistently documenting Tribal marten harvests in 2009, and since then 18% (n=523) were registered on Tribal licenses. I predicted that low monthly maximum temperatures and high precipitation would limit trap-line preparation, scouting, and correspond to reduced marten harvest.

I used rasters downloaded from the LANDFIRE Program (LANDFIRE 2010) to evaluate vegetation variables that potentially contributed to marten captures. These variables included vegetation disturbance (VDISTURB), vegetation canopy cover (EVC), canopy bulk density (CBD), vegetation height (EVH), and vegetation type (EVT) modeled and mapped at 30m resolution. Vegetation disturbance is a composite (over last 10 years) of disturbance type, severity on vegetation, and time since the disturbance occurred (LANDFIRE 2019). Vegetation canopy cover portrays the vertically projected cover of live canopy, accounting for trees, shrubs, and herbaceous cover (LANDFIRE 2019). Canopy bulk density describes the density of available canopy, defined as canopy fuel for fire per canopy volume (kg/m^3; LANDFIRE 2019). Vegetation height portrays average height of dominant vegetation, and vegetation type describes complexes of plant communities (LANDFIRE 2019). Given the temporal extent of the marten data, I used five LANDFIRE products (2001, 2008, 2010, 2012, and 2014), and matched the LANDFIRE data closest to year of marten harvest I used the zonal statistics tool in ArcGIS spatial analyst to identify the majority value for each LANDFIRE layer within the 2.6km^2 area containing the marten harvest.

Roads negatively affect marten as direct (hit by vehicles) and indirect (access by trappers) sources of mortality (Soukkala 1983; Hiller et al. 2011; Cervinka et al., 2015; Ruette et al., 2015). However, marten use certain road types for traveling and hunting (Robitaille and Aubry 2000; Silet 2017). Michigan maintains an all roads geographic framework layer that categorizes roads (Center for Shared Solutions and Technology Partnerships 2014). Using this spatial framework, I extracted the maintained roads, which describe roads that are permanent and accessible year round (maintained). I buffered maintained roads by 8m and summed the road areas within each 2.6km^2 containing a harvested marten. A buffer was important to account for roads that occurred on township section boundaries; the buffer allowed a single road to influence multiple sections.

I performed analysis using the R language version 3.4.1 for statistical computing (R Core Team 2017). To model the probability of marten harvest, I used a Bayesian geostatistical approach that accounted for spatial and temporal autocorrelation. Harvests were modeled as a Bernoulli point process such that *Z(s,t)* signifies the occurrence (1) or absence (0) of a marten harvest in a township section *s* (*s* = 1,2,3…,17276) during year t (t=2001, 2002, 2003…, 2018), with a probability of marten harvest given as $\pi_{s,t}$. Generally, this relationship is:

$$Z(s,t) \sim \text{Bernoulli}(\pi_{s,t})$$

$$logit(\pi_{s,t}) = \beta_0 + \beta X_{s,t} + \mathcal{W}_{s,t}$$

where β_0 is the intercept, $X_{s,t}$ represents a vector of fixed covariates and random effects with corresponding coefficients $\beta = (\beta_1, \dots, \beta_k)$, and $\mathcal{W}_{s,t}$ is the space-time structured effect that quantifies spatial and temporal correlation between sections (s) and across years (t) (Humphreys et al. 2017; Humphreys et al. 2019). The spatial components of $\mathcal{W}_{s,t}$ were modeled using continuous Gaussian random fields with Matèrn covariance and an Order 1 Auto-regression for time following the SPDE approach (Lindgren2011). Because the model generates a separate realization of the spatial field for each time step, it was necessary to reduce model temporal resolution by aggregating time into two year groupings or bins (i.e., time knots) (Blangiardo and Cameletti 2015). Lowering the temporal resolution enabled us to model the entire marten harvest record (2001-2018) with minimal effect to estimated spatial-temporal correlation.

I scaled and centered predictor variables, and matched monthly temperature and precipitation data to the corresponding harvest year (2001-2018). I checked for correlation among predictor variables using the colldiag function from the perturb package in R, which calculates condition indices and variance decomposition proportions (Hendricks 2019). Condition indices >30 paired with variance decomposition proportions >0.50 suggest collinearity and warrant further exploration (Belsley et al. 1980). I evaluated two model structures, with and without spatial and temporal autocorrelation, using Deviance Information Criteria (DIC) and Watanabe-Akaike Information Criteria (WAIC). We deemed model parameters significant if 95% credible intervals did not overlap zero. To help ensure identifiability and avoid over-parameterization, we enforced a zero mean constraint on all random effects (Schrodle and Held 2011, Bivand et al., 2015).

Results

Trappers registered 4,449 marten from 2001–2018; 3,412 included year and location (~190/year). Marten harvest was highest (n=353) in 2018, and lowest in 2001 (n=63). On average, <1.5 marten were harvested from the same township section within a year, indicating that captures were spread across space. Most harvested marten were young-of-the-year; 0.5 (39%; n=1,337) and 1.5 years of age (14%; n=487). The oldest marten were 12.5 years of age (0.09%; n=3). Gender was recorded for 2,833 marten; males dominated harvest (71%; n=2,008) followed by females (29%; n=825).

Landfire Factors

When summarized as majority types within 2.6 km^2 areas, I found that few categories dominated most LANDIFRE variables. I found that vegetation disturbance was not a majority characteristic of sections in the Upper Peninsula (99% of background), or where marten were harvested (99% of harvests; Table S1). Low numbers of marten were captured in sections dominated by disturbance (0.6% of captures), and almost all of these (95%) were captured >1 year after the disturbance (Table S1). Most (70%) marten were harvested in sections dominated by low CBD (0.01kg/m^3), and some (11%) in sections dominated by non-forested areas (that presumably contained forest patches; Table S2). Overall, CBD in the Upper Peninsula is low, with 34% of the sections dominated by non-forested areas (0.00 CBD), and 49% dominated by sparse overstory canopies (0.01 CBD; Table S2). At coarse resolution (i.e., 2.6 km^2), CBD is low across this landscape, likely reflecting dominance of deciduous and mixed deciduous-coniferous forests.

I found that tree cover ranging from 50 to 90% was a dominate characteristic of sections (70% of background) in the Upper Peninsula, and most marten (93%) were harvested in these areas (Table S3). Low numbers of marten (<9%) were harvested in sections dominated by shrub cover and herbaceous cover, presumably from forested or wetland patches embedded in this matrix (Table S3). Trappers caught most (98%) marten in sections dominated by taller forests (10-25 m), and this forest stature was most ubiquitous (78% of section) in the Upper Peninsula (Table S4). My height data, when combined with cover data, suggest that marten from sections dominated by shrubs tended to come from taller shrub communities, like alder (*Alnus* spp.) or scrub-shrub wetlands >10m tall.

The LANDFIRE layer contained 60 cover types in the Upper Peninsula, dominated by variations of hardwood forests (at least 63% of sections) and boreal white spruce-fir (30% of sections; Table S5). We note that the No Data class dominated 18% of sections in the Upper Peninsula (Table S5); lakes Superior, Michigan, or Huron dominated these sections. The majority of harvested marten came from sections dominated by hardwood forests (68%; Table S5). Surprisingly, low numbers of marten (13%) came from sections dominated by conifer-hardwood swamp forests, but sections dominated by this type were uncommon (17% background). Other sections where marten were trapped more frequently (i.e., >1% of the sections) were dominated by boreal acidic peatland systems (collectively 4% of sections; Table S5). At coarse resolution, our results highlight the positive relationship between hardwood forest communities and marten harvests in the Upper Peninsula.

Weather Factors

Average maximum monthly temperatures for the entire Upper Peninsula were 13.8˚C in October, 6.4 ˚C in November, and -3.3˚C in December (Table S6). On average, sections where trappers harvested marten were warmer in October and November, but colder in December than average Upper Peninsula conditions (Table S6). Average maximum precipitation was highest in October (102mm), and declined into November (65mm) and December (60mm) across the Upper

Peninsula (Table S6). Trappers harvested marten in sections that were considerably drier in October when compared to the Upper Peninsula, but precipitation amounts were comparable to the broader landscape in November and December (Table S6).

Road Factor

On average, 108 m^2/section of maintained roads occurred in sections where marten were harvested, compared to 91m^2 of maintained roads across the Upper Peninsula (Table S6).

Pelt Prices

I compiled annual marten pelt prices from a Michigan Department of Natural Resources furbearer harvest report (Frawley 2018) as a proxy for trapper effort. I assumed that effort would positively relate to pelt price. From 2001-2018, average pelt price was $46 USD for marten, with a range of $29 USD (2018) to $72 USD (2016), showing high inter-annual variability (*sd* = $13 USD).

Model Fitting

I did not find any predictor variables that were collinear (condition indices <10), so I used the complete suite of variables for modeling. The spatial and non-spatial models ranked higher than the non-spatial model (Table 1), so I used the spatial model for inference. For random effects, I found that marten age, sex, trapping year, CBD, and whether trappers harvested marten from the same section the previous year significantly influenced our model. Relative to the global mean harvest probability, harvest probability was higher for marten ≤4.5 years old and lower for marten ≥6.5 years old (Fig. S1). On average, harvest probability was lower for females (Fig. S2), and significantly fluctuated by year (Fig. S3). Harvest probability was below average in areas with low CBD, except one higher class of CBD (0.22 kg/m3) also corresponded to low harvest probability (Fig. S4). Trappers appeared to revisit the same sections where there were successfully harvested marten the prior year, as this was a strong predictor of harvest (Fig. S5). I did not find strong significant effects of vegetation disturbance, existing vegetation cover or height, or vegetation type as portrayed by majority LANDFIRE classes within a section on marten harvest.

I found that only weather fixed effects were significant (Table 2). Warmer October (β = 1.03, SD = 0.07) and November (β = 4.89, SD = 0.13) temperatures positively affected marten harvest, and colder December (β = -2.37, SD = 0.08) temperatures positively affected harvest (Table 2). The most influential month for temperature was November, followed by December (Table 2). I found that drier Octobers (β = -0.72, SD = 0.06) and Decembers (β = -0.27, SD = 0.05) positively affected marten harvest probability, but wetter Novembers were better for harvest (β = 0.67, SD = 0.05). Density of maintained roads, pelt price, and prior harvest of marten at any time prior did not affect harvest probability. (Table 2).

Discussion

Predicting harvest success for game animals is difficult because so many factors (some stochastic) affect the outcome (e.g., Johnson et al. 1997; Strickland et al. 2013). Regulations often established months or years in advance of harvest seasons are critical tools for wildlife population management, thus any ability to predict harvest outcomes is beneficial. I analyzed land cover, weather, roads, and pelt prices that potentially affected marten harvest in the Upper Peninsula of Michigan from 2001-2018. I found that weather was the most important correlate of harvest at the township section level, after accounting for age, sex, year, CBD, and whether a marten was harvest from the same section the prior year. Marten are more vulnerable to capture during years with warmer temperatures (Octobers and Novembers), but with colder December temperatures. Drier Octobers and Decembers, and wetter Novembers also positively corresponded to marten harvests. Based on the magnitude of standardized model coefficients, November and December temperatures were most influential. Changes in these annual weather patterns resulted in a strong year effect, underscoring the complexities in trying to make long-term predictions of harvest outcomes for marten. Furthermore, I found that LANDFIRE covariates summarized at the section level (i.e., majority value within a section) were generally poor predictors of marten harvests, with the exception of CBD. For predicting marten harvest outcomes, my results suggest that temperature and precipitation for the months leading up the harvest season hold promise as short-term predictors.

Generally, I found that harvest probability of female marten was lower than males (see also Strickland 1994). I also found that harvest were skewed to young marten (see also Harris et al. 1997; Ruette et al 2015), consistent with a harvest strategy that focuses on young animals and preserves breeding stock (Strickland 1994). Differences in trapping vulnerability among age and sex cohorts in *Martes* often relates to differences in home range sizes, distances traveled, and behaviors (Krohn et al. 1994, Powell 1994), with males having large home ranges and traveling further than females (Powell 1994), and juveniles traveling substantial distances during dispersal (Strickland et al. 1994).

Contrary to Hiller et al. (2011), I found a strong weather effect on where and when trappers captured marten. My observed effects varied annually, likely representing an integrated response to weather, trapping effort as mediated by weather and fur prices, and annual harvest regulations (Hiller et al. 2011). Differences in how researchers summarized weather data between studies likely explains some of the contradicting results. The Hiller et al. (2011) study summarized weather during the trapping season, which ranged from 11-15 days in December, whereas I evaluated a longer temporal extent that offered the model a broader range of values. Additionally, Hiller et al. (2011) used data from proximate weather stations, whereas I relied on interpolated values that corresponded to township sections. Together, the results suggest that weather during the trapping season is not overly influential on marten harvest, but that longer-term, monthly weather patterns matter. Longer-term weather may affect trap line scouting (e.g., patterning marten from November snows) or access to trapping areas (e.g., snow accumulation from November to December).

I hypothesized that marten would be trapped in areas with less forest disturbance, mature forest types (dominated by conifers), areas that were high in canopy cover and bulk density, and in areas that were closer to roads. At coarse resolution (i.e., majority type within 2.6 km^2) I found few remotely sensed vegetation variables that significantly explained marten harvests. However, several patterns emerged from summaries of capture counts within township sections that provide insights into the Upper Peninsula landscape. Most marten were caught in sections dominated by minimal disturbance (also see Ruggiero et al. 1994; Zielinski, Kucera 1995), tree cover ≥50% (also see Bushirk et al. 1989, Cushman et al. 2011), and with lower crown bulk densities (likely reflecting the dominance of hardwood forest types). Others demonstrated that higher crown bulk density, characteristic of conifers, positively affected marten use (Williams et al. 2007, Cushman et al. 2011, Bridger et al. 2017). Marten tend to have a strong ecological relationship with certain prey species that are associated with conifer forests (Tevis 1956; Clough 1987; Nordyke and Buskirk 1991; Pearson and Ruggiero 2001), but hardwoods and mixed hardwoods and conifer forests also support marten (Williams et al. 2007). I suspect that conifers are an important component of marten habitat in the Upper Peninsula, particularly in lowland or along riparian areas, but the coarse resolution of our land cover data failed to identify these potentially important areas.

Lastly, I hypothesized that roads would positively affect marten harvests. I failed to find a road effect on harvest probability, but caution that my analysis was limited to maintained roads. I suspect that seasonal roads play a greater role in trapper access and ultimately harvest success (Soukkala 1983; Robitaille and Aubry 2000; Hiller et al. 2011), but accurate maps of seasonal roads were unavailable. The strong weather effect on marten harvest probability likely relates to access, with drier Decembers potentially allowing vehicular access to a broader landscape.

I conducted this research using a large data set accumulated from martens registered by trappers, presumably representing marten populations across the Upper Peninsula of Michigan. However, harvest data can contain inaccuracies and potential biases (Pelletier et al. 2012). I assumed that location biases in the data were minimal because trappers reported at a coarse resolution (2.6 km^2). I also acknowledge potential shortcomings in how I used LANDFIRE data. Given the section-level resolution of harvest data, I aligned resolutions by summarizing the majority LANDFIRE class in each township section. Therefore, this approach reduced my ability to identify important, less common features or fine scale heterogeneity that may affect trapping success (e.g., conifer dominated riparian areas). For example, one township section may have 51% hardwood and 49% lowland conifer; if a marten was trapped in the lowland conifers my analysis would miss this relationship.

My results indicate that wildlife managers have minimal control over distribution of marten harvests at the township section level. Instead, harvest probability seems most related to temperature and precipitation leading up to and during the harvest season. Marten trappers understand habitat, often identifying areas occupied by marten prior to the trapping season, using this information to their advantage (Wiebe et al. 2013), if weather conditions cooperate. My data also indicate that trappers return to areas where they harvested marten the prior year, partially explaining the preponderance of younger animals in the harvest population.

APPENDIX

FIGURE & TABLE CAPTIONS

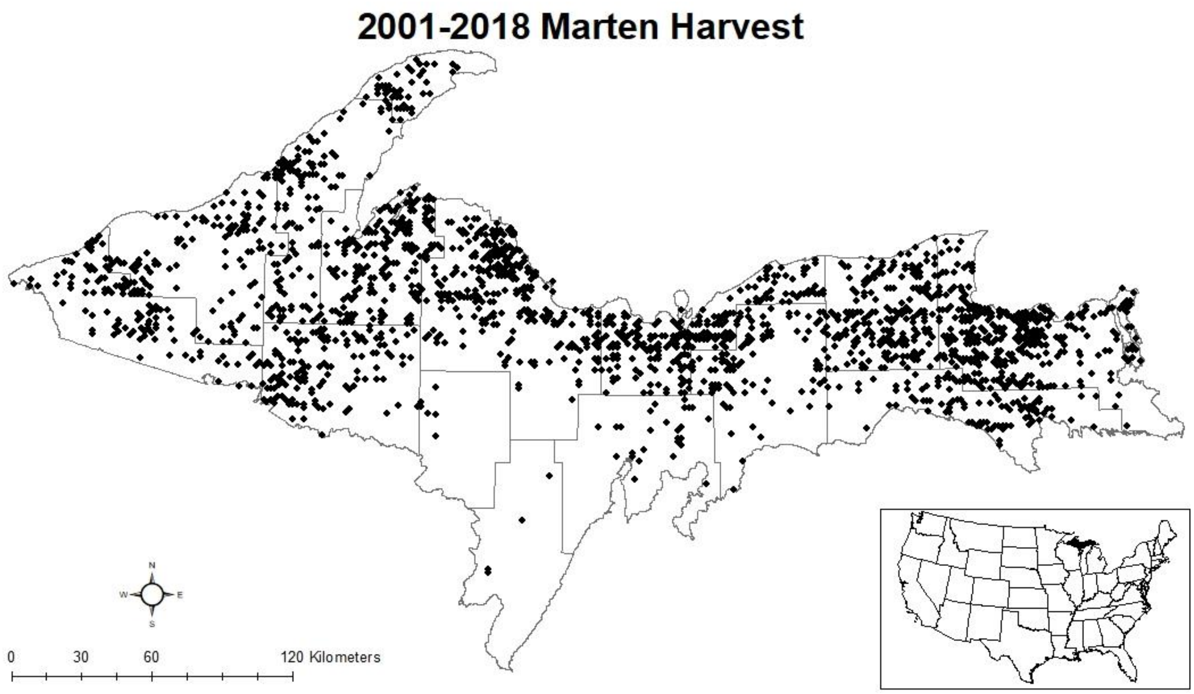

Figure 2. 1. The Upper Peninsula of Michigan, USA showing marten harvest data (recorded to the nearest 2.6 km^2) collected by the Michigan Department of Natural Resources from trappers between 2001-2018 (n=3,412).

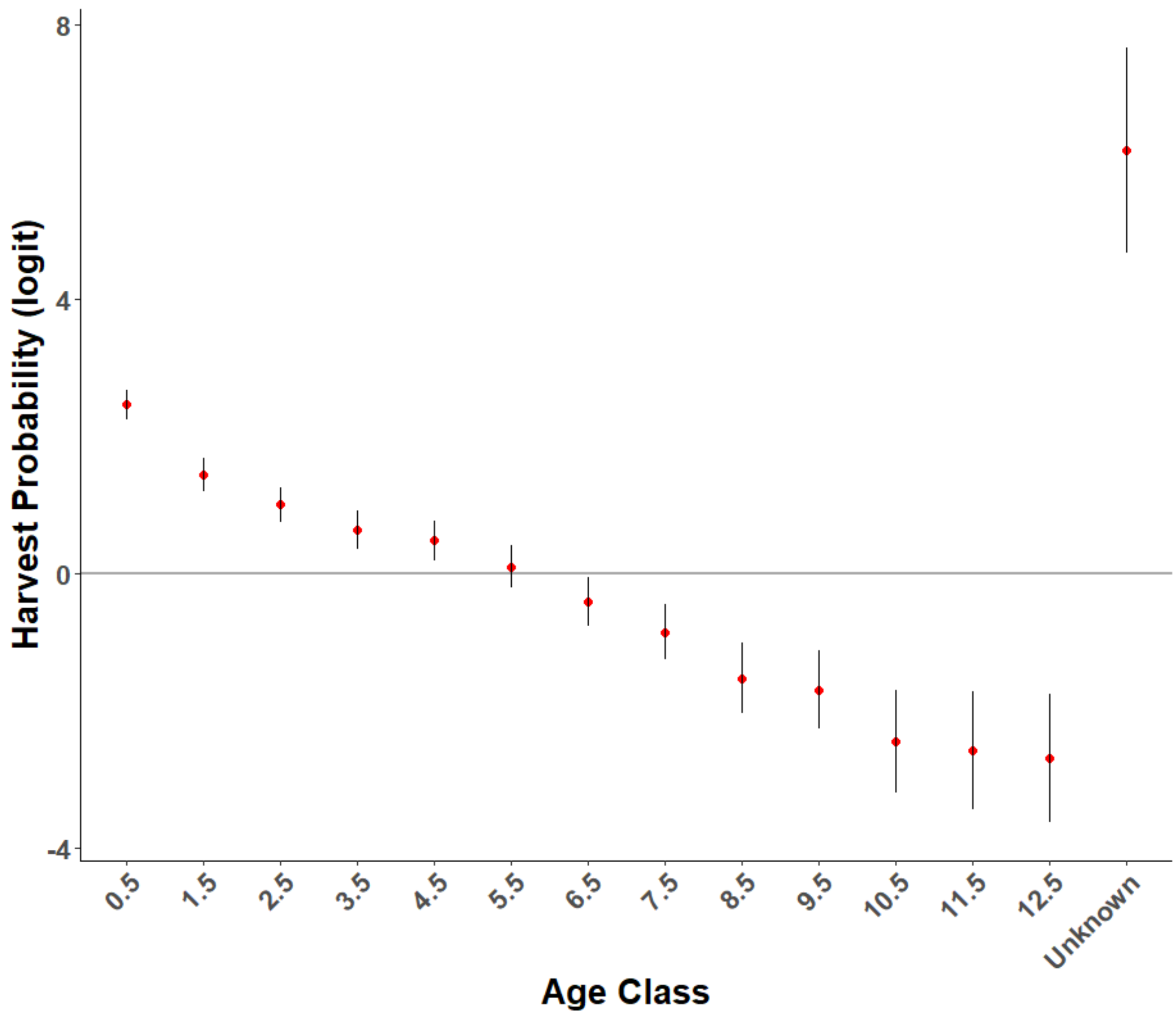

Figure 2. S1. Age at capture of marten relative to the global capture mean (horizontal line). Data collected by the Michigan Department of Natural Resources from registered marten in the Upper Peninsula, Michigan, USA between 2001-2018 (n=3,412).

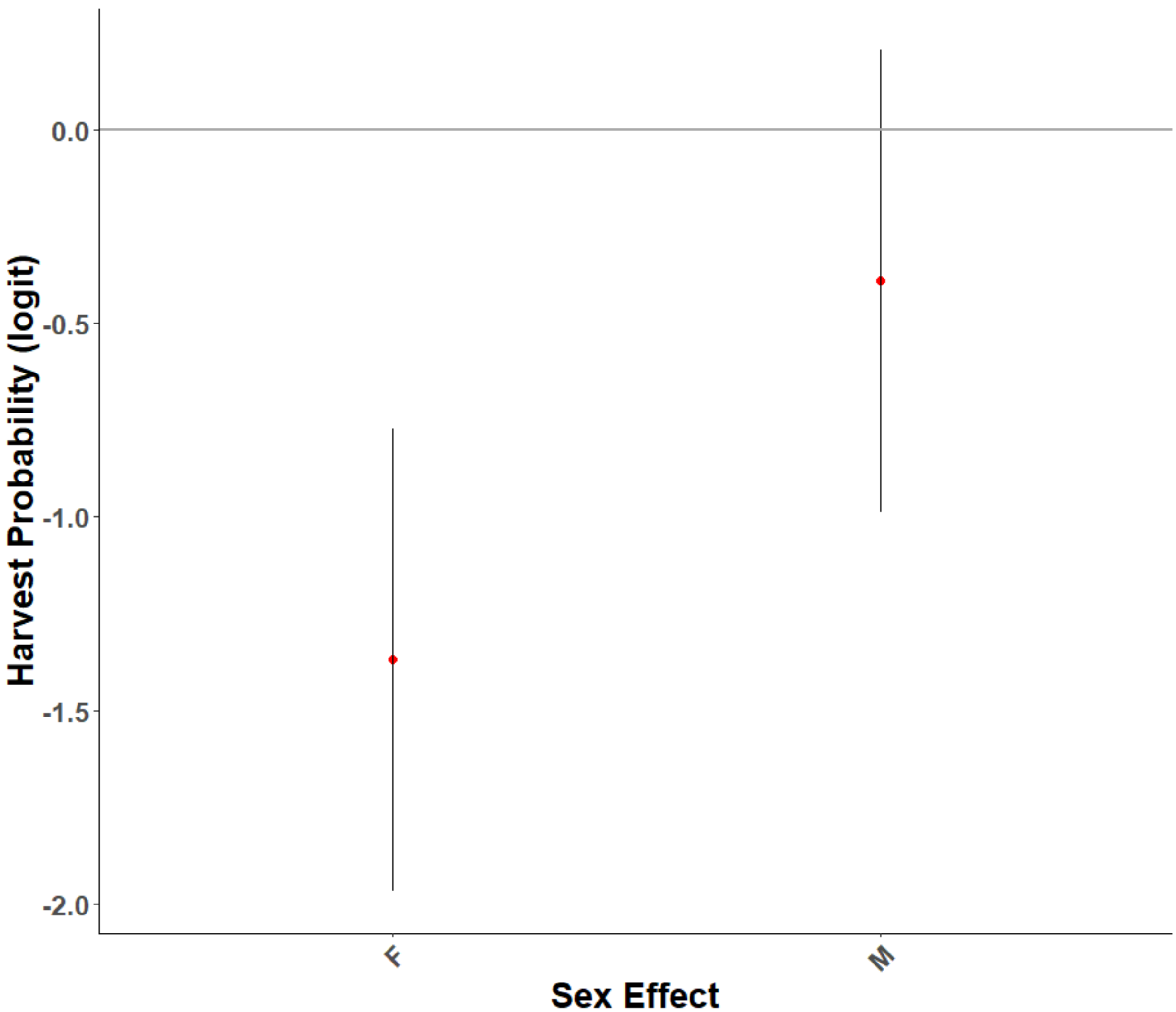

Figure 2. S2. Sex of captured marten relative to the global capture mean (horizontal line). Data collected by the Michigan Department of Natural Resources from registered marten in the Upper Peninsula, Michigan, USA between 2001-2018 (n=3,412).

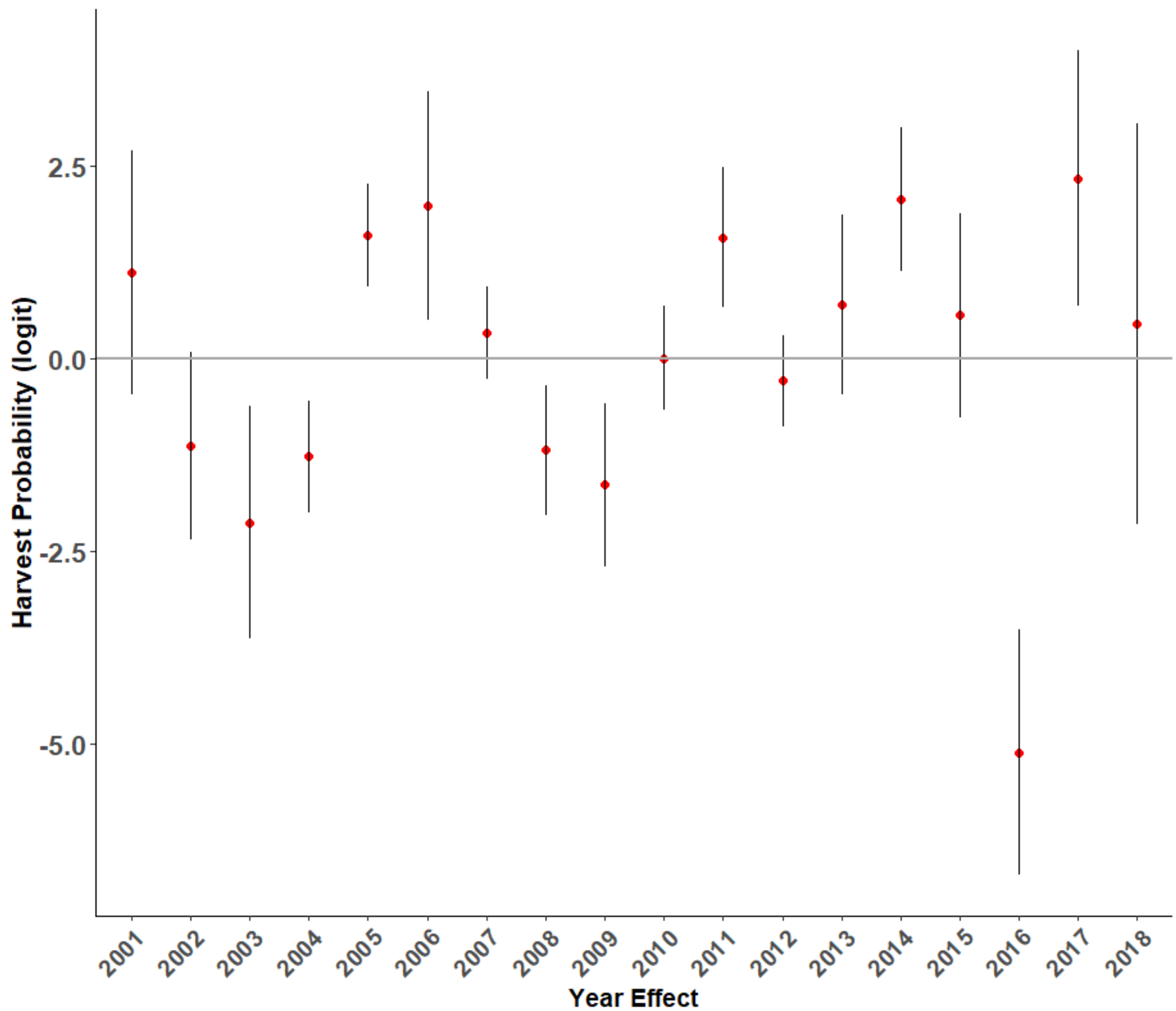

Figure 2. S3. Year of marten harvest relative to the global mean (horizontal line). Data collected by the Michigan Department of Natural Resources from registered marten trappers in the Upper Peninsula, Michigan, USA captured between 2001-2018 (n=3,412).

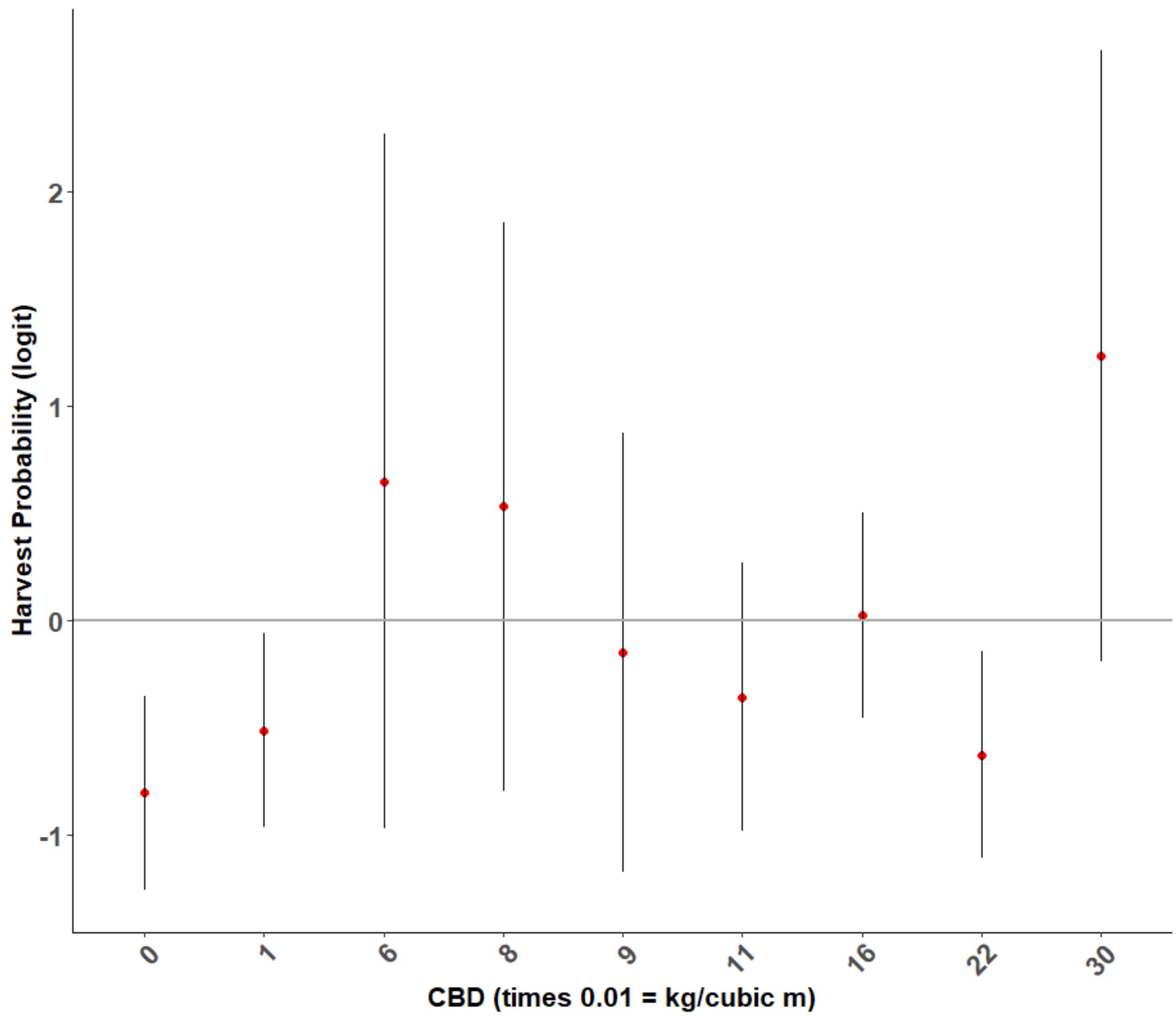

Figure 2. S4. Crown Bulk Density (CBD) and marten harvest probability relative to the global mean (horizontal line). Data collected by the Michigan Department of Natural Resources from registered marten in the Upper Peninsula, Michigan, USA (n=3,412). To convert CBD class code to kg/m3, multiply by 0.01.

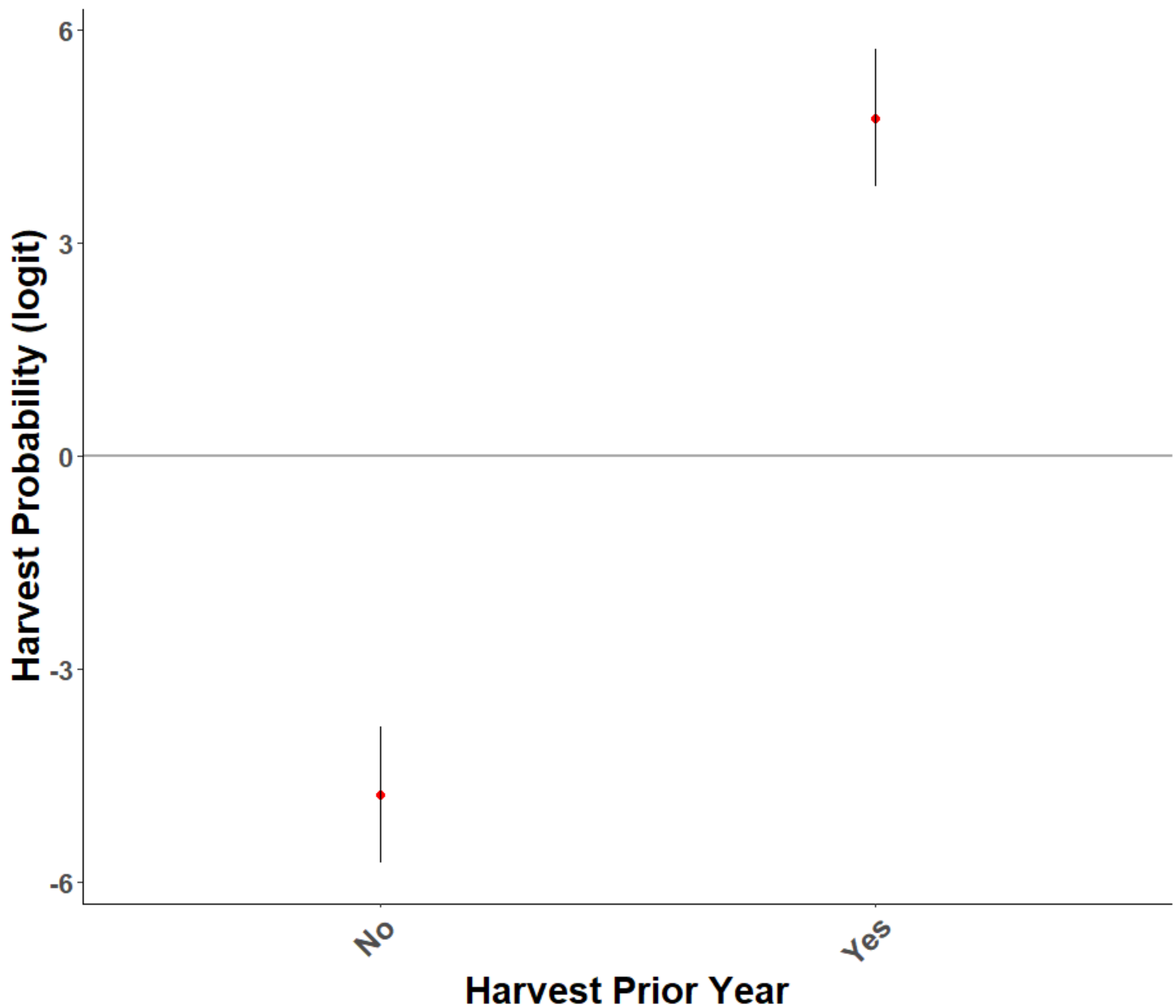

Figure 2. S5. Influence of prior year harvest from the same section on marten harvest probability. Data collected by the Michigan Department of Natural Resources from registered marten in the Upper Peninsula, Michigan, USA between 2001-2018 (n=3,412).

Table 2.1. Deviance Information Criteria (DIC) and Watanabe-Akaike Information Criteria (WAIC) for models predicting likelihood of harvesting a marten in a township section of the Upper Peninsula of Michigan, 2001-2018. One model included a spatial random effect term (Spatial) and one did not (No Spatial). *k* = effective number of parameters.

Model	DIC	k_{DIC}	WAIC	k_{WAIC}
Spatial	5,411.71	95.16	5,397.06	77.58
No Spatial	5,416.92	92.33	5,403.20	75.71

Table 2.2. Parameter estimates (standard deviation (SD)) and 95% credible intervals for fixed effects used to predict the likelihood of trapping a marten in a township section in the Upper Peninsula of Michigan, 2001 – 2018.

Variable[a]	Estimate (SD)	95% Credible Intervals
Temperatures		
October	1.03 (0.07)	0.90 – 1.16
November	4.89 (0.13)	4.64 – 5.15
December	-2.37 (0.08)	-2.52 – -2.23
Precipitation		
October	-0.72 (0.06)	-0.85 – -0.61
November	0.67 (0.05)	0.58 – 0.77
December	-0.27 (0.05)	-0.37 – -0.18
Maintained Roads	0.04 (0.03)	-0.03 – 0.10
Pelt Price	-0.04 (0.03)	-0.10 – 0.03
Prior Harvest	-0.01 (0.01)	-0.02 – 0.01

[a] Temperature and precipitation represent mean maximum monthly values, derived from PRISM. Roads represent those not maintained throughout the year (seasonal) and those maintained (e.g., plowed). Prior harvest indicates whether a trapper harvested a marten in the same section at any time previously.

Table 2. S1. Vegetation disturbance (VDISTURB) category as a majority cover type in sections where trappers harvested marten, and comparable background data points used for modeling. Sections represent counts, and percent represents percentage of the total (calculated from total number of registered marten (n=3,412) and background locations (n=24,570).

LANDFIRE Vegetation Disturbance		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
0	No disturbance	3,361	99	24,595	99
111	Fire, Low, 1 yr ago	0	0	4	<1
112	Fire, Low, 2-5 yrs ago	1	<1	18	<1
113	Fire, Low, 6-10 yrs ago	0	0	10	<1
121	Fire, Medium, 1 yr ago	1	<1	2	<1
122	Fire, Medium, 2-5 yrs ago	1	<1	16	<1
123	Fire, Medium, 6-10 yrs ago	0	0	7	<1
132	Fire, High, 2-5 yrs ago	3	<1	11	0
133	Fire, High, 6-10 yrs ago	2	<1	7	0
212	Mechanical Add, Low, 2-5 yrs ago	0	0	8	<1
213	Mechanical Add, Low, 6-10 yrs ago	3	<1	19	<1
311	Mechanical Remove, Low, 1 yr ago	0	0	2	<1
312	Mechanical Remove, Low, 2-5 yrs ago	8	<1	24	<1
313	Mechanical Remove, Low, 6-10 yrs ago	0	0	4	<1

Table 2. S1 (cont'd)

LANDFIRE					
Vegetation Disturbance		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
322	Mechanical Remove, Medium, 2-5 yrs ago	1	<1	11	<1
323	Mechanical Removal, Medium, 6-10 yrs ago	0	0	5	0
332	Mechanical Remove, High, 2-5 yrs ago	1	<1	7	<1

Table 2. S2. The majority canopy bulk density (CBD) within a section where trappers harvested marten, and comparable background data points used for modeling. Sections represent counts, and percent represents percentage of the total (calculated from total number of registered marten (n=3,412) and background locations (n=24,570).

LANDFIRE Canopy Bulk Density		**Marten**		**Background**	
Code	**Kg/m^3**	**Sections**	**Percent**	**Sections**	**Percent**
0	0.00	379	11	8,431	34
1	0.01	2,375	70	12,216	49
6	0.06	2	<1	0	0
8	0.08	5	<1	13	<1
9	0.09	10	<1	78	<1
11	0.11	60	2	541	2
16	0.16	306	9	2,021	9
22	0.22	231	7	1,450	5
30	0.30	14	<1	0	0

Table 2. S3. The majority existing vegetation cover class (EVC) in sections where trappers harvested marten, and comparable background data points used for modeling. Sections represent counts, and percent represents percentage of the total (calculated from total number of registered marten (n=3,412) and background locations (n=24,570).

LANDFIRE Vegetation Cover Code	**Type**	**Marten**		**Background**	
		Sections	**Percent**	**Sections**	**Percent**
0	None	7	<1	4643	19
11	Open water	38	1	430	2
16	Developed-upland herbaceous	6	<1	18	<1
17	Developed-upland shrubland	0	0	7	<1
22	Developed-low intensity	1	<1	9	<1
23	Developed - medium intensity	0	0	18	<1
25	Developed roads	3	<1	94	<1
31	Barren	0	0	72	<1
32	Quarries-strip mines-gravel pits	0	0	47	<1
64	NASS-row crop	0	0	27	<1
65	NASS-close grown crop	1	<1	75	<1
66	Nass-pasture and hayland	1	<1	33	<1
81	Pasture/hay	2	<1	12	<1
82	Cultivated crops	16	<1	171	<1

Table 2. S3. (cont'd)

LANDFIRE					
Vegetation Cover		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
95	Herbaceous wetlands	19	<1	253	1
100	Sparse vegetation canopy	0	0	9	<1
101	Tree cover >= 10 and < 20%	0	0	12	<1
102	Tree cover >=20 and <30%	2	<1	37	<1
103	Tree cover >=30 and <40%	29	<1	168	<1
104	Tree cover >=40 and <50%	5	<1	79	<1
105	Tree cover >=50 and <60%	119	4	1087	5
106	Tree cover >=60 and <70%	489	14	4164	17
107	Tree cover >=70 and <80%	2139	63	10805	42
108	Tree cover >=80 and <90%	389	12	1583	6
111	Shrub cover >=10 and <20%	55	2	359	1
112	Shrub cover >=20 and <30%	4	<1	4	<1
113	Shrub cover >=30 and <40%	3	<1	34	<1
114	Shrub cover >= 40 and <50%	0	0	29	<1
115	Shrub cover >=50 and <60%	2	<1	50	<1
121	Herb cover >=10 and <20%	17	<1	91	<1
122	Herb cover >=20 and <30%	8	<1	96	<1

Table 2. S3. (cont'd)

LANDFIRE Vegetation Cover		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
123	Herb cover >=30 and <40%	0	0	4	<1
124	Herb cover >=40 and <50%	9	<1	96	<1
125	Herb cover >=50 and <60%	18	<1	134	<1

Table 2. S4. Majority existing vegetation height (EVH) in sections where trappers harvested marten, and comparable background data points used for modeling. Sections represent counts, and percent represents percentage of the total (calculated from total number of registered marten (n=3,412) and background locations (n=24,570).

LANDFIRE Vegetation Height		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
0	No Data	6	<1	4626	19
11	Open water	11	<1	237	<1
16	Developed-upland herbaceous	2	<1	0	<1
25	Developed-roads	0	0	63	<1
31	Barren	0	0	72	<1
32	Quarries-strip mines-gravel pits	0	0	36	<1
64	NASS-row crop	0	0	9	0
65	NASS-close grown crop	0	0	14	<1
81	Pasture/hay	0	0	4	<1
82	Cultivated crops	1	<1	36	<1
95	Herbaceous wetlands	6	<1	67	<1
101	Herb height 0 to 0.05 meters	5	<1	32	<1
102	Herb height 0.5 to 1.0 meters	18	<1	118	<1
103	Herb height > 1.0 meter	0	0	2	<1

Table 2. S4. (cont'd)

LANDFIRE Vegetation Height		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
104	Shrub height 0 to 0.5 meters	12	<1	108	<1
107	Shrub height >3.0 meters	11	<1	77	<1
108	Forest height 0 to 5 meters	4	<1	12	<1
110	Forest height 10 to 25 meters	3306	98	19237	78

Table 2. S5. Majority existing vegetation type (EVT) in sections where trappers harvested marten, and comparable background data points used for modeling. Sections represent counts, and percent represents percentage of the total (calculated from total number of registered marten (n=3,412) and background locations (n=24,570).

LANDFIRE Vegetation Type		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
0	No data	6	<1	4626	18
11	Open water	12	<1	189	<1
16	Developed-upland herbaceous	2	<1	9	<1
25	Developed-roads	0	0	27	0
31	Barren	0	0	36	<1
32	Quarries-strip mines-gravel pits	0	0	27	<1
64	Agricultural- row crop	0	0	9	0
65	Agricultural-grown crop	0	0	5	0
66	Agricultural-fallow/idle cropland	1	<1	24	<1

Table 2. S5. (cont'd)

LANDFIRE					
Vegetation Type		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
81	Agricultural-pasture and hay	2	<1	4	<1
82	Agricultural-cultivated crops and irrigated agriculture	6	<1	118	<1
95	Caribbean	9	0	144	1
2191	Recently logged-herb and grass cover	17	<1	101	<1
2195	Recently burned-herb and grass cover	5	<1	10	<1
2198	No data	0	0	5	0
2301	Boreal aspen-birch forest	1	<1	27	<1
2302	Laurentian-Acadian North American hardwoods forest	860	25	5650	22
2344	Boreal jack pine-black spruce forest	21	1	180	1
2362	Laurentian-Acadian northern pine forest	4	<1	27	<1
2365	Boreal white spruce-fir forest	36	1	469	30

Table 2. S5. (cont'd)

LANDFIRE					
Vegetation Type		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
2366	Laurentian-Acadian pine-hemlock forest	45	1	153	<1
2407	Laurentian pine barrens	2	<1	18	<1
2444	Eastern boreal floodplain	3	<1	9	<1
2466	Great Lakes wooded dune and swale	0	0	9	<1
2475	Laurentian-Acadian floodplain systems	11	<1	36	<1
2477	Boreal acidic peatland systems	72	2	386	2
2481	Laurentian-Acadian alkaline conifer-hardwood swamp	169	5	2200	8
2492	Great Lakes coastal marsh systems	0	0	8	<1
2534	Managed tree plantation-northern and central hardwood	21	<1	182	<1
3191	Recently logged-herb and grass cover	7	<1	31	<1
3195	Recently burned-herb and grass cover	1	<1	13	0

Table 2. S5. (cont'd)

LANDFIRE					
Vegetation Type		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
3240	Laurentian-Acadian hardwood forest	28	1	61	<1
3241	Laurentian-Acadian pine-hemlock-hardwood forest	4	<1	0	<1
3242	Laurentian- oak barrens	0	0	9	0
3245	Boreal white spruce-fir-hardwood forest	0	0	27	<1
3272	Eastern boreal floodplain shrubland	1	<1	0	0
3277	Laurentian-Acadian floodplain shrubland	1	<1	0	0
3279	Boreal acidic peatland shrubland	13	<1	99	<1
3281	Laurentian-Acadian alkaline conifer-hardwood swamp shrubland	3	<1	63	<1
3285	Laurentian-Acadian shrub wetlands	14	<1	144	1
3292	Open water	36	1	216	1
3294	Barren	0	0	36	<1

Table 2. S5. (cont'd)

LANDFIRE					
Vegetation Type		Marten		Background	
Code	Type	Sections	Percent	Sections	Percent
3494	Laurentian-Acadian forested wetlands	2	<1	0	0
3295	Quarries-strip mines-gravel pits	0	0	20	<1
3299	Developed-roads	1	<1	36	<1
3301	Boreal aspen-birch forest	3	<1	36	<1
3302	North American hardwoods forest	1438	43	5969	24
3344	Boreal jack pine-black spruce forest	50	1	224	<1
3362	Laurentian-Acadian northern pine forest	1	<1	9	<1
3365	Boreal white Spruce-fir forest	58	2	250	1
3366	Laurentian-Acadian pine-hemlock forest	5	<1	36	<1
3407	Laurentian pine barrens	21	<1	36	<1
3444	Eastern boreal floodplain woodland	3	<1	9	0

Table 2. S5. (cont'd)

LANDFIRE					
Vegetation Type		**Marten**		**Background**	
Code	**Type**	**Sections**	**Percent**	**Sections**	**Percent**
3466	Great Lakes wooded dune and swale	0	0	9	<1
3475	Laurentian-Acadian floodplain forest	5	<1	36	<1
3477	Boreal acidic peatland forest	76	2	234	1
3481	Laurentian-Acadian alkaline conifer hardwood forest swamp	263	8	2016	8
3534	Managed tree plantation-northern and central hardwood and conifer plantation group	20	1	151	1
3909	Eastern cool temperate urban shrubland	0	0	7	<1
3974	Eastern cool temperate row crop	0	0	4	<1
3975	Eastern cool temperate close grown crop	3	<1	52	<1
3977	Eastern cool temperate pasture and hayland	20	1	229	1

Table 2. S6. Average (standard error) and range of monthly maximum temperatures and monthly maximum precipitation for October, November, and December based on PRISM data, and average (standard error) and range of road densities (based on All Roads layer; Center for Shard Solutions and Technology Partnerships 2014) for the Upper Peninsula, Michigan from 2001-2018.

		Marten		**Background**	
Category	**Variable**[a]	**$\bar{x}$ (SE)**	**Range**	**$\bar{x}$ (SE)**	**Range**
Temperature (˚C)	Oct	13.8 (0.03)	9.7, 17.8	12.3 (0.01)	6.1, 19.2
	Nov	6.4 (0.04)	2.0, 12.6	4.6 (0.02)	-3.3, 11.3
	Dec	-3.3 (0.07)	-10.1, 4.0	-2.0 (0.02)	-11.0, 5.7
Precipitation (mm)	Oct	65 (.71)	6, 200	102 (.27)	22, 285
	Nov	71 (.24)	32, 133	65 (.20)	5, 203
	Dec	59 (.36)	18, 162	60 (.16)	4, 165
Roads (m^2)	Maintained	108 (2.84)	0, 1,038	91 (1.10)	0, 1,731

[a]Temp = average monthly maximum temperature, Precip = average monthly maximum precipitation, Seasonal = density of unmaintained roads, Maintained = density of roads maintained throughout the year (Center for Shared Solutions and Technology Partnerships 2004).

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CHAPTER 3

MULTISCALE RELATIONSHIPS OF SONGBIRD OCCUPANCY IN A MANAGED FOREST LANDSCAPE

Abstract

Impacts of forest management on biodiversity is a major conservation concern. In response, forest managers implement practices designed to mitigate negative impacts of some timber harvest techniques, like clearcutting. These practices include protection of waterbodies and unique ecosystems, protection of uncommon species, and retention of forest structure elements. Oftentimes, information on effectiveness of these practices is lacking, yet managers increasingly must defend implementation. I evaluated the effects of stand- and landscape-level variables across varying levels of structural retention on songbird occupancy in clearcut aspen stands in the northwestern Lower Peninsula, Michigan. Field technicians surveyed birds during the breeding season of 2010 and 2011 at 250 sites. I built 5 candidate occupancy models for 8 common species. My candidate models included stand- and landscape-level covariates; stand-level covariates included clearcut age, canopy cover of structural retention, understory woody stem density, and distance to the nearest plot. Landscape-level variables included amount of 5 LANDFIRE cover types surrounding survey points; Riparian, Short Conifer (0-5m tall), Tall Conifer (>5m tall), Short Hardwood, and Tall Hardwood. I calculated the amount of each cover type within 150, 250, and 400 m buffers from plot center. I found that landscape level variables poorly described occupancy probability for mature forest birds, and that occupancy of only red-eyed vireo negatively related to a plot level covariate (i.e., woody stem density, mostly in the understory). Conversely, occupancy probability for some birds associated with dense understories or riparian areas related to landscape covariates, including American redstart,

chestnut-sided warbler, and Nashville warbler. Eastern towhee occupancy probability associated with a plot level covariate; age of the clearcut containing the plot. I found highly variable predictors of bird occupancy among species even within guilds, highlighting challenges associated with identifying widespread prescriptions that account for all species. My research highlights the importance of assessing stand- and landscape-level factors when quantifying effectiveness of wildlife conservation practices like structural retention.

Introduction

For decades, concerns regarding the role of forest management in declines of songbird populations across North America have persisted (Tittler et al. 2001; Griesser et al. 2007; Moore et al. 2010). Clearcutting (a silviculture prescription) is one forest management technique resulting in significant changes to bird habitats. Clearcutting is an effective management tool for encouraging early successional habitat and for increasing habitat heterogeneity (Annand and Thompson 1997; Costello et al. 2000; King and DeGraaf 2000; Thompson and DeGraaf 2001; Franklin et al. 2002; DeGraaf and Yamasaki, 2003), but results in loss of older forest habitats (Wallendorf et al. 2007). Some birds are disturbance-sensitive species (DeGraaf and Yamasaki 2003), meaning that occupancy, abundance, and demographics are readily impacted by forest management. Given current focus on conservation of biodiversity in managed forest landscapes (e.g., Sustainable Forestry Initiative 2010; Forest Stewardship Council 2012), improved understanding of how birds respond to varying silvicultural prescriptions and landscape context is important.

Clearcutting effects on birds varies by species, regional landscape characteristics, size of clearcut openings, and forest type (Thompson and Curran 1995; Wallendorf et al. 2007). To ameliorate the negative effects of clearcutting on mature forest dependent birds, retention of unharvested forest patches or individual structures (e.g., green trees, snags) is suggested (reviewed by Demarais et al. 2017). The idea is that these remnant forest structures, if embedded in a larger matrix of managed lands, could function as refuges allowing species sensitive to disturbance to persist until forest regeneration occurs (Thomas 2007). A problem with retained structures is that patterning rarely emulates natural patch-disturbance patterns, instead representing anthropogenic-imposed patch structure on the landscape (Freedman et al. 1994). For example in eastern boreal mixed-wood forests conversions of large-scale mature mixed-wood forests to deciduous forests (Carleton and McLellan 1994), shorter timber rotations (that truncate successional trajectories; Spies and others 1994; Gauthier et al. 1996), and timber stand improvement practices (that simplify forest structure; Wallendorf et al. 2007) resulted in anthropogenic patchiness and vegetation structure and species composition that threaten some bird communities (Drapeau et al. 2000).

Responses of birds to structural retention vary depending on patch sizes, clearcut sizes, condition of surrounding landscapes, and age of the surrounding clearcut forest. Forest structure changes rapidly within the first 10-15 years post-clearcutting in most managed ecosystems (DeGraaf and Yamasaki 2003). In landscapes dominated by older forests and small clearcut sizes, retention effects on songbirds are minimal, with age of the surrounding regenerating forest a primary determinant of the bird community (e.g., Otto and Roloff 2012). In landscapes dominated by intensively managed younger forests with larger retention patch sizes, bird responses to retention are more pronounced (e.g., Linden et al. 2012). These studies collectively indicate that landscape context has a strong influence on function of retention patches as bird habitat (Aubry et al. 1999; Tittler et al. 2001).

Aspen (*Populus* spp.) is an important component of timber production in Michigan. Aspen are highly invasive after disturbances and can aggressively reproduce under certain circumstances (Haeussler 2004) From 1900 to 1966 aspen increased from 117,400 ha to 1.7 million ha (Dickmann and Leefers 2016). From 1966 to 1993, aspen forests in Michigan declined by an estimated 600,000 ha, largely because some landowners allowed aspen to convert to shade tolerant species (Dickmann and Leefers 2016). Managers primarily use clearcutting and subsequent natural regeneration to manage aspen, with prescriptions on state-owned lands requiring retention of green trees to provide wildlife habitat (Bielecki et al. 2006).

My goal was to quantify the roles that retention patches and landscape context play in bird use of clearcut aspen forests. Previously, Otto and Roloff (2012) found minimal support for retention effects on bird occupancy dynamics in clearcut aspen so I predicted that landscape context would better describe bird occupancy. Furthermore, I predicted that birds would respond to different landscape-level components depending on general habitat affinities, with later successional species more dependent on older forest amount around the clearcuts. Lastly, I predicted that landscape context closer to clearcuts would have a greater effect on songbird use than more distal contexts. My results provide species-specific guidance for managing bird assemblages in aspen clearcuts, as influenced by three elements under direct management control: 1) structural retention, 2) clearcut age, and 3) landscape context.

Methods

Study Area

My study used data collected from Otto and Roloff (2012), and the study area description is consistent with that publication. Field data were collected in the northwestern Lower Peninsula of Michigan, USA, from 2010-2011. The last glaciation strongly influenced topography and soils in this area, with survey sites located on glacial outwash-plains with sandy porous soils, and on more fertile ice-contact zones and moraines (Albert 1995). All sampling occurred on state-owned forest lands managed for production of aspen (*Populus* spp). Clearcut stands ranged from 1 to 15 years post-harvest, and each received a green-tree retention prescription (Bielecki et al. 2006). These prescriptions included low (<3%), medium (3-10%), and high (>10%) retention of canopy cover or basal area, generally of species that reflected preharvest stand conditions (but uncommon or mast-producing species were favored; Bielecki et al. 2006). Species that occurred in retention patches included oak (*Quercus* spp.), white pine (*Pinus strobus*). red maple (*Acer rubrum*), black cherry (*Prunus serotine*), American beech (*Feagus grandifolia*), and aspen. Understories were dominated by woody plants that included aspen, red maple, blackberry (*Rubus* spp.), black cherry, downy serviceberry (*Amelanchier arborea*), witch-hazel (*Hamamelis virginiana*), American beech, and ironwood (*Ostrya carpinifolia*). Average high temperatures range from -3.3°C in January to 25.6°C in July, and monthly precipitation ranges from 3.7cm (February) to 10.0cm in September (U.S. Climate Data 2020). Snowfall mostly occurs from November through March, averaging 65.0cm per month (U.S. Climate Data 2020).

Site Selection

I used study sites selected by Otto and Roloff (2012), consisting of state-owned aspen stands >8 ha in size, between 1 and 15 years post-harvest. Otto and Roloff (2012) used a Geographic Information System (GIS; ArcGIS 9.1; Environmental Systems Research Institute, Redlands, CA) to superimpose a 60 x 60m lattice over each aspen stand, excluding lattice cells that intersected or included unharvested forest edge, active logging roads, off-road recreational vehicle trails, or wetlands visible from aerial imagery (2005 National Agricultural Imagery Program (NAIP); Michigan Department of Information Technology, 2007). The remaining lattice cells represented candidates for bird sampling. Otto and Roloff (2012) digitized retention canopy cover for the areas encompassed by candidate cells and then randomly selected cells that represented 3 levels of structural retention (<3%, 4-10%, >10% canopy cover) and 3 levels of regenerating clearcut age (1-4, 5-9, 10-15 years post-harvest).

Bird Sampling

Otto and Roloff (2012) surveyed birds at the center of each selected lattice. Bird sampling occurred in 2010 and 2011. At each site location, an individual observer conducted a 9-min point-count, sub-divided into 3,3min sub-counts (Otto and Roloff 2012). Point counts were conducted 30 mins after sunrise and no later than 3.5 hours after sunrise on days with appropriate weather conditions (i.e., no rain, no strong winds; Otto and Roloff 2012). Songbirds seen or heard within a 50-m sampling radius of the point-count center were recorded (Otto and Roloff 2012), with the 50m sampling distance reducing the likelihood of sampling birds in adjacent stands (i.e., reduce edge effects; Otto and Roloff 2012). Observers were trained in estimating songbird detection distance prior to the start of the study. Initial site visits occurred in late-May, then every 10-14 days thereafter. This sampling method continued until all surveyors visited all sites at least 4 times within a 2-month period within the breeding season (late-May to mid-July; Otto and Roloff 2012).

Habitat Covariates

I identified two levels of habitat covariates for analysis: plot and landscape. Tree crowns of structural retention in each harvested aspen stand were digitized from NAIP imagery (1998-2010, depending on stand age) in ArcGIS, and I used percent of retained canopy cover within 50-m of bird survey points as a plot-level covariate (Otto and Roloff 2012). I also included age of the clearcut as a plot-level covariate given the importance of that variable in previously describing occupancy dynamics of birds from these data (Otto and Roloff 2012). Density of trees >1m tall was calculated for each plot along 4 transects (4-m x 25-m) radiating outward in cardinal directions from plot center. I checked these covariates for collinearity using condition index (≥30; Belsey et al. 1980) and variance decomposition proportions (≥50%; Belsey et al. 1980) in R (R Core Team 2019), package perturb version 3.4.1 (Gates et al. 2019).

For landscape-level variables, I used GIS layers available from the 2012 LANDFIRE Program (LANDFIRE 2012a, b). I first extracted raster cells labeled as forests, and then assigned each to conifer (included conifer-hardwood mixes), hardwoods, and riparian using the EVT_PHYS field in the existing vegetation type (EVT) layer (LANDFIRE 2012a). Subsequently, I identified short (0-5m) and tall (>5m) forests from the existing vegetation height (EVH) layer (LANDFIRE 2012b) and merged the two queries together to produce 5 landscape-level variables: Riparian, Short Conifer, Tall Conifer, Short Hardwood, and Tall Hardwood. I calculated percent of each variable within 150, 250, and 400 m of plot center. The 400m radii (~50 ha circle) encompasses multiple breeding territories for focal bird species.

Model Generation and Analysis

I selected 8 of the more common forest bird species from Otto and Roloff (2012) for modeling that represented varying habitat requirements. Ovenbird (*Seiurus aurocapilla*), red-eyed vireo (*Vireo olivaceus*), and rose-breasted grosbeak (*Pheucticus ludovicianus*) represented mature forest species (Brewer et al. 1992), whereas indigo bunting (*Passerina cyanea*), eastern towhee (*Pipilo erythrophthalmus*), Nashville warbler (*Leiothlypis ruficapilla*), chestnut-sided warbler (*Setophaga pensylvanica*), and American redstart (*Setophaga ruticilla*) represented species with affinity for denser forest understories and riparian areas (Brewer et al. 1992). I generated detection histories for each species consistent with the format needed for a dynamic occupancy model (MacKenzie et al. 2006). For example, the detection history for 2 site locations was:

Site 1: 100 111 000 101

Site 2: 010 000 110 000

where "1" indicated that ≥1 bird was detected during a 3 min survey and "0" means not detected. Each group of 3 numbers represents a 9-min point count where sites were assumed closed to changes in occupancy (i.e., unoccupied sites did not become occupied, or occupied sites did not become unoccupied; Otto and Roloff 2012). The 9-min point count constituted a primary-period within a dynamic occupancy model, whereas the 3 sub-counts were temporal sampling replicates used for estimating detection probability (MacKenzie et al. 2006).

For each bird species, I first determined which spatial extent and landscape-level covariate best explained occupancy. Landscape covariates in my analyses were highly collinear by definition (e.g., if the value of one variable was high in a buffer, the other variables had to be low), so I constructed 15 univariate candidate models and used Akaike Information Criteria (AIC) to rank those models. I subsequently used the landscape-level covariate and spatial extent in the top-ranking model in subsequent models that included stand- and landscape-level covariates to describe bird occupancy.

I constructed 5 candidate models for each bird species that included an intercept-only model (i.e., Null Model), models based only on stand or landscape level covariates (Stand and Landscape models, respectively), and a global model that included stand and landscape covariates (Global Model). A distance to nearest plot neighbor was included in all models to account for spatial autocorrelation in plots close to each other (e.g., some plots occurred in the same clearcut stand). Lastly, I included a global model without the spatial term (Global Model No Space). For detection, colonization and extinction components of the dynamic occupancy model I used intercept-only terms. Otto and Roloff (2012) previously found this was a reasonable approach for the detection parameter, and inference on colonization or extinction was not part of my objective. For each species I ranked occupancy models using AIC, and model averaged parameter estimates for competing models (≤2 ΔAIC). I evaluated 95% confidence intervals from model averaged parameter estimates and those not including 0 were deemed significant correlates of occupancy. I used model averaging and 95% confidence intervals as described for the occupancy model to determine parameter significance. I plotted relationships of significant occupancy parameters by holding other covariates at mean values.

Results

Field crews surveyed 250 sites; 157 sites in 2010, 93 different sites in 2011. Each stratum contained samples with more old (10-15 years since harvest) and high (>10% cover; 64 plots) plots compared to the other groupings (Table 1). The least sampled stratum was old-aged (10-15 years) and mid-retention (4-10% cover; 10 plots). Imbalance in the sample design reflects field implementation of retention practices (more likely at low and high amounts) and random lattice placement. Stratifying the sample achieved representation across a broad range of covariates as continuous predictors in bird models. Retained cover across all sites ranged from 0-100% and stand ages from 1-15 years. Regenerating stem densities ranged from 27 to 1,775/100m^2. None of these variables exhibited collinearity.

Areas within buffers around bird survey points were primarily (>52% on average) tall hardwood forest, followed by riparian and tall conifer forests (>7%; Table 2). Thus, on average recent timber harvests were uncommon in surrounding forest areas, indicating a landscape context dominated by taller forests. However, short forests (up to 52%) surrounded some points (Table 2). Across the spatial extents I evaluated, cover type percentages were relatively invariant, except for tall hardwoods that ranged from 0-100% (Table 2).

For evaluating landscape variables across varying buffer sizes, I failed to find a clear top-ranking model (AIC_{wt} ranged from 0.09 – 0.49); most bird species except American redstart and chestnut-sided warbler had competing models (i.e., <2ΔAIC) indicating that buffer size and landscape covariates did not greatly differentiate in explaining occupancy probability (Tables S1−S8). I found that the top ranked landscape variable for ovenbirds, red-eyed vireos, and, chestnut-sided warblers was cover of tall conifer forest within 250m of the survey point (250m for ovenbird, 150m for red-eyed vireos and chestnut-sided warblers; Tables S1, S2, S3). Riparian cover was the most influential landscape variable for indigo bunting, Nashville warbler, American redstart, and rose-breasted grosbeak (Tables S4, S5, S6, S7). Riparian was most influential at 150m for indigo buntings (Table S4), 250m for Nashville warblers (Table S5, and 400m for American redstarts and rose-breasted grosbeaks (Tables S6, S7) The only species apparently affected by landscape-level hardwoods was eastern towhee at 400m radii (Table S8).

For combined stand- and landscape-level models for mature forest species, I found that the Null model received the most support (AIC_{wt} = 0.53; Table 3) for ovenbirds, indicating that none of the variables I explored described occupancy probability better than random. For red-eyed vireos I found that the Landscape model (AIC_{wt} = 0.41) competed with the Global No Space AIC_{wt} = 0.25) and Null models (AIC_{wt} = 0.21; Table 4). Model averaging indicated that only plot-level stem density was significant (β = -1.64, 95% CI = -0.003, -7.651), with occupancy probability declining as understory stem density increased. For rose-breasted grosbeaks I found that the Landscape model (AIC_{wt} = 0.53) competed with the Null model

(AIC_{wt} = 0.35; Table 5). I failed to find any significant parameters for rose-breasted grosbeak occupancy. Collectively, none of the landscape variables significantly correlated with occupancy probability for mature forest bird species, and only plot-level stem density was important to red-eyed vireos.

For bird species associated with denser forest understories or riparian areas, I found that the Null model ranked highest for indigo buntings (AIC_{wt} = 0.48; Table 6). For American redstart occupancy, I found that the Landscape model ranked highest (AIC_{wt} = 0.68), with no competing models (Table 7). I found that only the riparian 400m parameter was significant (β = -2.81, 95% CI = -5.265455, -0.373560), with occupancy declining as the amount of riparian habitat within 400m increased. Similarly, the Landscape model ranked highest for chestnut-sided warblers (AIC_{wt} = 0.71; Table 8), with no competing models (Table 7). For rufous-sided towhee occupancy, I found the Global No Space model (AIC_{wt} = 0.37) competed with the Stand (AIC_{wt} = 0.27) and Null models (AIC_{wt} = 0.17; Table 9). After model averaging, I found that age of clearcut containing the survey plot was significant (β = 0.048, 95% CI = 0.009, 0.087), indicating that towhee occupancy increased as clearcut age increased (up to 15 years). For Nashville warbler occupancy, I found that the Global No Spatial model (AIC_{wt} = 0.44) competed with Landscape (AIC_{wt} = 0.38) and Global (AIC_{wt} = 0.16) models (Table 10). After model averaging, I found that amount of riparian cover type within 250m was significant (β =2.38, 95% CI = 0.635, 4.13201), indicating the occupancy probability increased as riparian cover type increased.

Discussion

I quantified the relative contributions of stand- (e.g., amount of retention, understory stem density, age of the surrounding clearcut) and landscape-level (e.g., proportion of surrounding area in different forest structures) attributes on bird occupancy probability in aspen clearcuts across the northwestern Lower Peninsula of Michigan. I found more supporting evidence for occupancy models based on landscape-level variables compared to stand-based models, but evidence varied by guild. Occupancy probability of birds associated with mature forest conditions did not relate to landscape-level factors, whereas birds associated with dense understories and riparian areas did. Collectively, my results indicate that landscape-level variables are more influential on occupancy than stand-level variables for early-seral birds using clearcuts with retention in northern Michigan. This finding contradicts my prediction that landscape context would have a greater effect on birds associated with mature forest conditions.

Characteristics of the matrix surrounding forest patches are important parts of bird habitat assessments (Saab 1999, Renjifo 2001, Brotons et al. 2003, Heartsill-Scalley and Aide 2003). In my study, significant landscape variables included amount of riparian cover type within 400m (for American redstarts, negative relationship) and 250m (Nashville warblers, positive relationship) of survey plots. Riparian types in my study area are highly varied structurally, including dense thickets of alder and scrub-shrub. American redstarts breed in moist, deciduous second-growth woodlands with abundant shrubs, often situated with thickets (Brewer et al. 1992). Compared to occupying a recent aspen clearcut, the negative association with riparian areas is intuitive as these areas are often occupied (Brewer et al. 1992). My results suggest that abundant riparian cover type around aspen clearcuts decreases occupancy probability of the clearcuts by redstarts, potentially because redstarts are selecting riparian instead. Nashville warblers generally breed in second-growth woodlands with shrubby undergrowth (Brewer et al. 1992). In contrast to American redstarts, my results indicated that riparian cover type around clearcuts increases occupancy probability of the clearcut, potentially because more suitable forest structure occurs in clearcuts.

I predicted that landscape context closer to survey points would be more influential on bird occupancy than plot-level variables. This prediction held true for 3 of 8 bird species, but none of the landscape variables were significant at the smallest spatial extent (i.e., 150m). My study area generally consists of a forested matrix, with interspersed agriculture and recent clearcuts. Lack of a consistently strong landscape-level signal in bird occupancy in my study may relate to the relatively low level of large-scale disturbance. The influence of landscape context on bird assemblages increases as use of surrounding lands intensifies (Austen et al. 2001, Martin et al. 2006). Similar to my results, Edenius and Sjöberg (2006) found no matrix effect on bird species richness in a landscape dominated by forest.

For conservation of birds in managed forest landscapes, managers can leave unharvested reserves (i.e., structural retention) or use selective cuts to retain general stand structure (Tittler, 2001). Structural retention undoubtedly affects wildlife use of timber harvest areas (Verner et al. 1986, DeGraaf et al. 1998, Hagan and Meehan 2002, Linden et al. 2012, Otto and Roloff 2012). I found no support for retention affecting bird occupancy in relatively small (~8 ha) clearcuts embedded in a forest-dominated matrix. My landscape context was dominated by tall forest conditions, with <10% in recently harvested (or short) status and small harvest unit sizes. In contrast, bird response to stand-level retention in landscapes from industrial forest ownerships in the Pacific Northwest showed stronger effects, particularly in areas with larger clearcut sizes and overall younger forest structure across the landscape (Linden et al. 2012).

I caution that my data represent a limited number of bird species that potentially use clearcuts with retention. I restricted analysis to species that required an intercept-only detection parameter (Otto and Roloff 2012), and those that were relatively ubiquitous and represented varying habitat requirements. Furthermore, my choice of landscape variables including categories derived from LANDFIRE and extent of radii evaluated undoubtedly affected results. It is plausible that I selected spatial extents that did not coincide with important landscape patterns (Mitchell et al. 2001), or vegetation classifications too coarse to coincide with bird habitat selection (Roloff et al. 2009).

As global demands for wood products continue to increase, wildlife conservation in managed forest landscapes will depend on effective practices aimed at retaining habitat elements as part of operational forest management. The concept of structural retention as a means for mitigating the negative effects of timber harvest is well established (reviewed by Demarais et al. 2017), but effectiveness monitoring is lacking. As forest managers balance demands for increased timber production with activities that are viewed as a cost (e.g., structural retention), effectiveness monitoring becomes critical to defend conservation activities. My results indicate that an important component of monitoring effectiveness of structural retention for bird conservation (and potentially other taxon) includes consideration of landscape context.

APPENDIX

FIGURES & TABLES

Table 3.1. Number of plots sampled by retention canopy cover (in 50m radius plot) and clearcut age in northwest Lower Peninsula of Michigan, 2010 and 2011.

	Retained Canopy Cover (%)			
Clearcut Age (yrs)	<3	4-10	>10	Row Total
1-4	25	16	30	71
5-9	19	20	35	64
10-15	31	10	64	105
Column Total	75	46	129	

Table 3.2. Average (SE) and range (Min, Max) of cover (%) of landscape variables surrounding bird survey points at 3 radii in northwest Lower Peninsula of Michigan, 2010 and 2011.

	Radii (m)					
	150		250		400	
Landscape Variable[a]	Mean (SE)	Range	Mean (SE)	Range	Mean (SE)	Range
Short Conifer Forest	2 (0.45)	0,47	2 (0.25)	0, 27	2 (0.20)	0, 18
Tall Conifer Forest	8 (0.89)	0, 98	9 (0.82)	0, 63	11 (0.81)	0, 66
Short Hardwood Forest	6 (0.69)	0, 52	4 (0.44)	0, 44	3 (0.28)	0, 26
Tall Hardwood Forest	68 (1.73)	0, 100	56 (1.56)	0, 99	53 (1.54)	0, 96
Riparian	10 (1.16)	0, 97	11 (1.12)	0, 84	13 (1.24)	0, 82

[a] Derived from 2012 LANDFIRE Program (LANDFIRE 2012a, b) by first extracting grid cells labeled as forests, and then assigning each to conifer (included conifer-hardwood mixes), hardwoods, and riparian using the EVT_PHYS field in the existing vegetation type (EVT) layer (LANDFIRE 2012a). Subsequently, we identified short (0-5m) and tall (>5m) forests from the existing vegetation height (EVH) layer (LANDFIRE 2012b) and merged the two queries together.

Table 3.3. Candidate models for ovenbird occupancy at points in the northwestern Lower Peninsula of Michigan, 2010-2011. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Occupancy				
Model[a]	AIC	ΔAIC	AIC_{wt}	k
Null	2009.53	0.00	0.53	5
Landscape	2010.89	1.36	0.27	6
Stand	2012.48	2.95	0.12	8
Global	2013.96	4.43	0.06	9
Global (No Space)	2015.22	5.69	0.03	8

[a] Null = only distance to nearest neighbor as a parameter; Landscape = top-ranked landscape-level variable (Supplement A); Stand = years since timber harvest, percent cover of retention, and understory stem density; Global = Landscape + Stand; Global (No Space) = Global model without the nearest neighbor term.

Table 3.4. Candidate models for red-eyed vireo occupancy at points in the northwestern Lower Peninsula of Michigan, 2010-2011. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Occupancy				
Model[a]	AIC	ΔAIC	AIC_{wt}	k
Landscape	1468.44	0.00	0.41	6
Global (No Space)	1469.47	1.03	0.25	8
Null	1469.79	1.35	0.21	5
Global	1471.21	2.76	0.10	9
Stand	1473.64	5.20	0.03	8

[a] Null = only distance to nearest neighbor as a parameter; Landscape = top-ranked landscape-level variable (Supplement A); Stand = years since timber harvest, percent cover of retention, and understory stem density; Global = Landscape + Stand; Global (No Space) = Global model without the nearest neighbor term.

Table 3.5. Candidate models for Rose-breasted grosbeak occupancy at points in the northwestern Lower Peninsula of Michigan, 2010-2011. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Occupancy				
Model[a]	AIC	ΔAIC	AIC_{wt}	k
Landscape	1989.58	0.00	0.53	6
Null	1990.41	0.83	0.35	5
Global	1994.26	4.68	0.05	9
Global (No Space)	1994.99	5.41	0.04	8
Stand	1995.66	6.08	0.03	8

[a] Null = only distance to nearest neighbor as a parameter; Landscape = top-ranked landscape-level variable (Supplement A); Stand = years since timber harvest, percent cover of retention, and understory stem density; Global = Landscape + Stand; Global (No Space) = Global model without the nearest neighbor term.

Table 3.6. Candidate models for indigo bunting occupancy at points in the northwestern Lower Peninsula of Michigan, 2010-2011. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Occupancy				
Model[a]	AIC	ΔAIC	AIC_{wt}	k
Null	1901.30	0.00	0.48	5
Landscape	1901.83	0.53	0.37	6
Stand	1905.55	4.25	0.06	8
Global (No Space)	1905.87	4.57	0.05	8
Global	1906.32	5.02	0.04	9

[a] Null = only distance to nearest neighbor as a parameter; Landscape = top-ranked landscape-level variable (Supplement A); Stand = years since timber harvest, percent cover of retention, and understory stem density; Global = Landscape + Stand; Global (No Space) = Global model without the nearest neighbor term.

Table 3.7. Candidate models for American redstart occupancy at points in the northwestern Lower Peninsula of Michigan, 2010-2011. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Occupancy				
Model[a]	AIC	ΔAIC	AIC_{wt}	k
Landscape	1353.96	0.00	0.68	6
Global (No Space)	1356.56	2.60	0.19	8
Null	1358.46	4.50	0.07	5
Global	1359.02	5.06	0.05	9
Stand	1364.07	10.11	0.00	8

[a] Null = only distance to nearest neighbor as a parameter; Landscape = top-ranked landscape-level variable (Supplement A); Stand = years since timber harvest, percent cover of retention, and understory stem density; Global = Landscape + Stand; Global (No Space) = Global model without the nearest neighbor term.

Table 3.8. Candidate models for chestnut-sided warbler occupancy at points in the northwestern Lower Peninsula of Michigan, 2010-2011. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Occupancy				
Model[a]	AIC	ΔAIC	AIC_{wt}	k
Landscape	1417.38	0.00	0.71	6
Null	1419.70	2.32	0.22	5
Global	1423.35	5.97	0.04	9
Global (No Space)	1424.00	6.62	0.03	8
Stand	1425.78	8.40	0.01	8

[a] Null = only distance to nearest neighbor as a parameter; Landscape = top-ranked landscape-level variable (Supplement A); Stand = years since timber harvest, percent cover of retention, and understory stem density; Global = Landscape + Stand; Global (No Space) = Global model without the nearest neighbor term.

Table 3.9. Candidate models for eastern towhee occupancy at points in the northwestern Lower Peninsula of Michigan, 2010-2011. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Occupancy				
Model[a]	AIC	ΔAIC	AIC_{wt}	k
Global (No Space)	1590.95	0.00	0.37	8
Stand	1591.57	0.61	0.27	8
Null	1592.58	1.62	0.17	5
Global	1593.29	2.33	0.12	9
Landscape	1594.22	3.26	0.07	6

[a] Null = only distance to nearest neighbor as a parameter; Landscape = top-ranked landscape-level variable (Supplement A); Stand = years since timber harvest, percent cover of retention, and understory stem density; Global = Landscape + Stand; Global (No Space) = Global model without the nearest neighbor term.

Table 3.10. Candidate models for Nashville warbler occupancy at points in the northwestern Lower Peninsula of Michigan, 2010-2011. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Occupancy				
Model[a]	AIC	ΔAIC	AIC_{wt}	k
Global (No Space)	1947.08	0.00	0.44	8
Landscape	1947.40	0.31	0.38	6
Global	1949.08	1.99	0.16	9
Null	1953.95	6.87	0.01	5
Stand	1955.20	8.12	0.01	8

[a] Null = only distance to nearest neighbor as a parameter; Landscape = top-ranked landscape-level variable (Supplement A); Stand = years since timber harvest, percent cover of retention, and understory stem density; Global = Landscape + Stand; Global (No Space) = Global model without the nearest neighbor term.

SUPPLEMENTAL TABLES

Table 3. S1. Candidate univariate landscape-level models for ovenbirds surveyed in 2010-2011, northwest Lower Peninsula of Michigan. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Variable, Radii	AIC_c	ΔAIC_c	AIC_{wt}	k
Tall Conifers, 250m	2012.69	0.00	0.11	5
Tall Conifers, 400m	2012.69	0.27	0.10	5
Riparian, 250	2013.55	0.86	0.07	5
Short Hardwoods, 400m	2013.72	1.03	0.07	5
Tall Hardwoods, 400m	2013.80	1.11	0.06	5
Short Conifers, 250m	2013.82	1.13	0.06	5
Riparian, 400m	2013.84	1.15	0.06	5
Short Hardwoods, 150m	2013.91	1.22	0.06	5
Riparian, 150m	2014.03	1.34	0.06	5
Short Conifers, 150m	2014.03	1.34	0.06	5
Tall Conifers, 150m	2014.03	1.34	0.06	5
Tall Hardwoods, 250m	2014.03	1.34	0.06	5
Short Hardwoods, 250m	2014.07	1.38	0.06	5
Tall Hardwood, 150m	2014.09	1.40	0.06	5
Short Conifer, 400m	2014.10	1.41	0.06	5

Table 3. S2. Candidate univariate landscape-level models for red-eyed vireos surveyed in 2010-2011, northwest Lower Peninsula of Michigan. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Variable, Radii	AIC_c	ΔAIC_c	AIC_{wt}	k
Tall Conifers, 150m	1466.67	0.00	0.21	5
Tall Conifer, 250m	1467.27	0.60	0.16	5
Short Conifer, 150m	1468.26	1.59	0.09	5
Tall Conifers, 400m	1468.78	2.10	0.07	5
Short Conifers, 250m	1468.79	2.12	0.07	5
Short Conifer, 400m	1469.37	2.70	0.05	5
Short Hardwood, 250m	1469.69	3.02	0.05	5
Short Hardwood, 400m	1469.85	3.18	0.04	5
Tall Hardwood, 250m	1470.08	3.41	0.04	5
Riparian, 400m	1470.14	3.47	0.04	5
Riparian, 250m	1470.18	3.51	0.04	5
Tall Hardwood, 150m	1470.19	3.52	0.04	5
Short Hardwood, 150m	1470.21	3.54	0.04	5
Riparian, 150m	1470.22	3.55	0.04	5
Tall Hardwood, 400m	1470.24	3.57	0.04	5

Table 3. S3. Candidate univariate landscape-level models for chestnut-sided warblers surveyed in 2010-2011, northwest Lower Peninsula of Michigan. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Variable, Radii	AIC_c	ΔAIC_c	AIC_{wt}	k
Tall Conifers, 150m	1418.28	0.00	0.19	5
Tall Conifers, 400m	1418.35	0.07	0.19	5
Tall Hardwood, 150m	1419.75	1.48	0.09	5
Tall Hardwood, 400m	1420.31	2.03	0.07	5
Tall Conifers, 250m	1420.49	2.21	0.06	5
Short Hardwood, 250m	1420.72	2.45	0.06	5
Tall Hardwood, 250m	1420.83	2.55	0.05	5
Short Hardwood, 150m	1421.20	2.92	0.04	5
Riparian, 400m	1421.30	3.02	0.04	5
Riparian, 250m	1421.63	3.35	0.04	5
Riparian, 150m	1421.75	3.47	0.03	5
Short Conifers, 250m	1421.82	3.55	0.03	5
Short Conifer, 150m	1421.85	3.57	0.03	5
Short Conifers, 400m	1421.93	3.66	0.03	5
Short Hardwood, 400m	1422.06	3.79	0.03	5

Table 3. S4. Candidate univariate landscape-level models for indigo buntings surveyed in 2010-2011, northwest Lower Peninsula of Michigan. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Variable, Radii	AIC_c	ΔAIC_c	AIC_{wt}	k
Riparian, 150m	1901.39	0.00	0.11	5
Short Hardwood, 250m	1901.73	0.34	0.09	5
Riparian, 400m	1902.15	0.76	0.08	5
Tall Hardwood150m	1902.15	0.76	0.08	5
Tall Hardwood, 400m	1902.31	0.93	0.07	5
Short Hardwood, 150m	1902.32	0.93	0.07	5
Tall Hardwood, 250m	1902.38	0.99	0.07	5
Short Hardwood, 400m	1902.46	1.08	0.06	5
Riparian, 250m	1902.59	1.20	0.06	5
Tall Conifers, 400m	1902.80	1.42	0.05	5
Tall Conifers, 150m	1902.89	1.50	0.05	5
Short Conifers, 150m	1902.93	1.54	0.05	5
Tall Conifers, 250m	1902.95	1.56	0.05	5
Short Conifers, 400m	1902.95	1.57	0.05	5
Short Conifers, 250m	1902.96	1.57	0.05	5

Table 3. S5. Candidate univariate landscape-level models for Nashville warblers surveyed in 2010-2011, northwest Lower Peninsula of Michigan. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Variable, Radii	AIC_c	ΔAIC_c	AIC_{wt}	k
Riparian, 250m	1945.62	0.00	0.49	5
Riparian, 400m	1947.55	1.93	0.19	5
Tall Conifers, 400m	1948.39	2.77	0.12	5
Riparian, 150m	1950.90	5.28	0.03	5
Tall Conifers, 250m	1951.15	5.54	0.03	5
Tall Conifers, 150m	1951.21	5.59	0.03	5
Short Hardwood, 400m	1951.27	5.65	0.03	5
Short Hardwood, 250m	1952.63	7.01	0.01	5
Short Conifers, 400m	1952.88	7.26	0.01	5
Short Conifers, 250m	1953.31	7.69	0.01	5
Tall Hardwood, 400m	1953.43	7.82	0.01	5
Short Conifers, 150m	1953.67	8.05	0.01	5
Tall Hardwood, 150m	1954.07	8.45	0.01	5
Short Hardwood, 150m	1954.11	8.49	0.01	5
Tall Hardwood, 250m	1954.14	8.52	0.01	5

Table 3. S6. Candidate univariate landscape-level models for American redstarts surveyed in 2010-2011, northwest Lower Peninsula of Michigan. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Variable, Radii	AIC_c	ΔAIC_c	AIC_{wt}	k
Riparian, 400m	1351.69	0.00	0.37	5
Tall Hardwood, 400m	1353.03	1.34	0.19	5
Riparian, 250m	1354.33	2.64	0.10	5
Short Conifers, 400m	1355.61	3.92	0.05	5
Short Hardwood, 400m	1355.66	3.98	0.05	5
Riparian, 150m	1355.69	4.01	0.05	5
Tall Conifers, 150m	1356.97	5.29	0.03	5
Short Hardwood, 250m	1357.01	5.32	0.03	5
Short Hardwood, 150m	1357.26	5.57	0.02	5
Short Conifers, 150m	1357.28	5.59	0.02	5
Tall Conifers, 250m	1357.37	5.68	0.02	5
Short Conifers, 250m	1357.39	5.71	0.02	5
Tall Hardwood, 250m	1357.77	6.09	0.02	5
Tall Hardwood, 150m	1357.84	6.15	0.02	5
Tall Conifers, 400m	1357.96	6.27	0.02	5

Table 3. S7. Candidate univariate landscape-level models for rose-breasted grosbeaks surveyed in 2010-2011, northwest Lower Peninsula of Michigan. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Variable, Radii	AIC_c	ΔAIC_c	AIC_{wt}	k
Riparian, 400m	1990.36	0.00	0.20	5
Riparian, 250m	1991.14	0.79	0.14	5
Riparian, 150m	1992.51	2.16	0.07	5
Tall Conifers, 150m	1992.54	2.18	0.07	5
Short Conifers, 400m	1992.70	2.34	0.06	5
Tall Hardwood, 400m	1992.85	2.50	0.06	5
Short Conifers, 150m	1993.14	2.78	0.05	5
Tall Hardwood, 150m	1993.22	2.86	0.05	5
Tall Hardwood, 250m	1993.29	2.93	0.05	5
Short Hardwood, 400m	1993.29	2.93	0.05	5
Short Hardwood, 150m	1993.41	3.06	0.04	5
Tall Conifers, 250m	1993.60	3.24	0.04	5
Short Conifers, 250m	1993.65	3.30	0.04	5
Tall Conifers, 400m	1993.66	3.30	0.04	5
Short Hardwood, 250m	1993.66	3.31	0.04	5

Table3. S8. Candidate univariate landscape-level models for eastern towhees surveyed in 2010-2011, northwest Lower Peninsula of Michigan. AIC = Akaike Information Criteria, ΔAIC = difference in AIC value from top model, AIC_{wt} = weight of evidence, and k = number of model parameters.

Variable, Radii	AIC_c	ΔAIC_c	AIC_{wt}	k
Short Hardwood, 400m	1591.88	0.00	0.09	5
Short Conifers, 400m	1591.95	0.07	0.08	5
Riparian, 400m	1592.14	0.26	0.08	5
Short Hardwood, 250m	1592.21	0.34	0.07	5
Tall Conifers, 150m	1592.24	0.36	0.07	5
Short Conifer, 150m	1592.40	0.52	0.07	5
Tall Hardwood, 400m	1592.49	0.62	0.06	5
Riparian, 250m	1592.52	0.64	0.06	5
Tall Hardwood, 150m	1592.54	0.66	0.06	5
Short Conifer, 250m	1592.55	0.67	0.06	5
Tall Conifers, 250m	1592.61	0.74	0.06	5
Tall Conifers, 400m	1592.64	0.76	0.06	5
Tall Hardwood, 250m	1592.65	0.77	0.06	5
Riparian, 150m	1592.65	0.77	0.06	5
Short Hardwood, 150m	1592.65	0.77	0.06	5

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CONCLUSION

My research offers insights into factors affecting student decisions to pursue a wildlife career, how socio-economic and environmental factors affect wildlife harvest (in my case, American marten), and how habitat management decisions by people can influence wildlife populations. Collectively, my dissertation chapters represent the major components of wildlife management in North America; people, wildlife populations, and wildlife habitat (Decker et al. 2012). Each of these individual research endeavors sought to add practical knowledge to wildlife management.

In Chapter 1, I assessed 21st century students to gain a better understanding of what motivated them to choose natural resources as a career. The perception of current generations of students (e.g., Millennials, Generation Z) is that friends, social media, and other outside influences play an important role in decision-making. I showed that family strongly affects interests in consumptive activities (like hunting, trapping) and decisions on natural resource careers for current natural resources students. I also found that students with consumptive family backgrounds and current consumptive interests decided on natural resources career earlier then students without those backgrounds or interests. My results also indicated that about 50% of natural resources students were not currently interested in consumptive activities, suggesting that natural resource values among the next generation of workers is diversifying. I make the argument that student training in consumptive and non-consumptive values is important for current college curricula as both values will shape the future of wildlife management.

In Chapter 2, I assessed factors potentially affecting harvest of American marten in Michigan's Upper Peninsula. Managers manipulate harvest with regulations, yet predicting harvest success is difficult because of complexities associated with individual trapper effort and personal circumstances. I sought to model focal areas of marten harvest from land cover, roads, pelt price, and weather as a means to forecast annual harvest outcomes. I found no evidence that land cover (within the constraints of how I modeled it), proximity to maintained roads, or pelt price could be used to predict marten harvest. Rather, weather conditions preceding and during the December harvest season were most influential on harvest outcomes. I posited that combinations of temperature and precipitation in October through December affected scouting and trap-line preparation, access to trapping areas, movements and vulnerability of marten, and ultimately where trappers focused effort. My results indicate short-term forecasting potential based on weather that may help anticipate harvest outcomes and ultimately regulation setting.

Lastly, in Chapter 3 I explored stand- and landscape-level variables influencing songbird occupancy in clearcut forests. I posited that landscape-level variables would be more important for describing occupancy that stand-level variables, given that previous research failed to find strong stand-level effects (Otto and Roloff 2012). I failed to find consistent effects for birds associated with mature forest conditions, but for some birds associated with dense, shrubby undergrowth and riparian areas I found landscape effects. Lack of a consistent response to stand- or landscape-level variables likely related to condition of the matrix environment; predominately forested. I posit that birds affected by relatively small-scale disturbances (~8 ha in this study) in heavily forested matrices temporarily displace, occupying adjacent areas until the clearcut regains suitable forest structure for occupancy. Even practices for retaining forest structure in clearcut areas does not appear to improve occupancy (e.g., see also Otto and Roloff 2012) of clearcuts by birds. My results indicate that in landscapes dominated by forests few consistent, reliable predictors of bird occupancy exist (within the domain of variables I assessed), though I found some species-specific exemptions. Furthermore, my results indicate that general prescriptions for entire bird communities may not exist, instead highlighting the importance of species-specific approaches.

Overall, my research highlighted various topics that wildlife managers currently face. I provide information that can aid managers in understanding these topics further. My research integrated across people, wildlife populations, and wildlife habitat; focuses on the three pillars that has guided wildlife management for almost 100 years.

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