Social Structure and Behaviour of the Eastern Wild Turkey

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Abstract

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Wildlife translocation programs are widely employed as a strategy to reintroduce extirpated species into regions they once inhabited but no longer do. Reintroduction programs can be successful at re-establishing extirpated populations and also provide unique opportunities to study post-reintroduction population dynamics and behavioural ecology. The wild turkey (*Meleagris gallopavo*) is a forest generalist species that, prior to European colonization, inhabited much of the Carolinian zone in Ontario. This species was hunted to extirpation in the early 1900's and reintroduced in the mid-1980's through a series of wildlife trade agreements and coordinated trap and transfer efforts. Ontario's contemporary populations are seemingly thriving, with wild turkey harvest permitted in many regions of the province. However, given this species history of extirpation, understanding the size, distribution, and behavioural ecology of Ontario's reintroduced population of wild turkeys is essential to their long-term persistence in the province. We captured and radio-tagged 77 wild turkeys over four years in Peterborough, Ontario and studied their movement, sociality, and habitat preferences. My findings indicate that Ontario may contain relatively high densities of this species when compared with other parts of their range. My analyses also elucidated interesting aspects of this species habitat selection patterns within an anthropogenic landscape, in addition to novel findings surrounding wild turkey sociality and genetic structure.

Keywords: reintroduction, supplemental food source, *Meleagris gallopavo*, social structure, behaviour

Preface

I have written my dissertation in manuscript format, since Chapter 2 has been published in Avian Conservation and Ecology and Chapters 3 and 4 will be submitted for publication. Each chapter was written and formatted in the style of the journal in which it was published. All my research has been in collaboration with others. As such, I have used the plural “we” where appropriate in my dissertation. I have indicated the names and role of my collaborators on the title page of each chapter, and I have obtained permission to reprint articles from the copyright holders (Appendix A).

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Chapter 1: General Introduction

Wild Turkeys Across Their Range

The wild turkey (*Meleagris gallopavo*) is a dark, large-bodied bird native to North America. This species is highly gregarious with patterns of sociality that differ depending on the time of year (Healy 1992). Six subspecies of wild turkey have been recognized historically: the eastern (*Meleagis gallopavo silvestris*), Florida (*M. g. osceola*), Rio Grande (*M. g. intermedia*), Merriam's (*M. g. merriami*), Gould's (*M. g. mexicana*), and south Mexican (*M. g. gallopavo*; Stangel et al. 1992). However, research using amplified fragment length polymorphism (AFLP) analysis, microsatellite loci, and mitochondrial control region sequencing indicates that the eastern and Florida subspecies together form a genetically distinct group (Mock et al. 2002), and that the Gould's subspecies is the most genetically distinct (Mock et al. 2001).

The eastern subspecies occupies woodlands spanning from the hardwood forests in the northeastern USA and Canada to the oak-hickory forests in the midwestern USA and the pine-oak forests in the southeastern USA. Historically, its habitat was continuous with that of the Florida subspecies to the south and the Rio Grande subspecies to the west (Schorger 1966, Williams 1981, Stangel et al. 1992). Merriam's and Gould's subspecies inhabit ponderosa pine (*Pinus ponderosa*) and pine-oak woodlands in the southwestern USA and northern Mexico. Currently, their habitats are separated from those of other subspecies and from each other by stretches of unsuitable desert and grassland. Nevertheless, it's possible that during the late Holocene's expansion of forest habitat, the ranges of Merriam's, Gould's, and Rio Grande subspecies were once contiguous (Mock et al. 2002).

Prior to European colonization, this species was an important component of forest biodiversity and held substantial cultural significance for Indigenous groups across the continent (Aldrich 1967). For instance, for the ancient peoples of the American Southwest and Mesoamerica, wild turkeys not only provided an important source of dietary protein, but also offered cultural resources such as feathers and bones having both ritual and practical uses (Muir and Driver 2002, Munro 2006). The ecosystems in many parts of this species' range also differed prior to European settlement. For instance, one of the many pathogens transported from Europe to North America during early western colonization included chestnut blight (*Cryphonectria parasitica*), a lethal fungal pathogen affecting some species in the *Castanea* genus, including the American chestnut (*C. dentata*). Prior to the arrival of Europeans to North America, the American chestnut was a common species in the Carolinian Zone, with early reports describing unbroken forest canopies and high densities of chestnut trees (Wunz and Pack 1992). Wild turkeys have historically relied on forest resources across their range, and seeds from American beech (*Fagus grandifolia*), oak (*Quercus* spp.), and other masting trees, continue to comprise a significant proportion of their diet (Hurst 1992, Vander Haegen et al. 1989, Vangilder and Kurzejeski 1995, Yarrow 2009, Otieno and Frenette 2017). As such, when American chestnut dominated the landscape, this species likely played a much more substantial role in the life history of wild turkeys in Ontario than it does today, when it is considered rare along with many other Carolinian Zone tree species (Reid 2002, Meloche and Murphy 2006). Other Carolinian Zone species like butternut (*Juglans cinerea*) and mulberry (*Morus rubra*) have also likely diminished in importance to wild turkeys as a food source as their densities and abundances have declined over time. Because wild turkeys are a generalist species (Hurst 1992), they have been able to tolerate this shift and have likely shifted their natural diet based on resource availability, while also adapting to exploit anthropogenic food resources, like plant and animal agriculture (Kane 2003, Kane et al. 2007, Restani et al. 2009).

Agricultural and residential development of North America following European settlement resulted in large-scale forest loss. This habitat loss, paired with unregulated hunting, led to the extirpation of turkeys in many regions across North America by the early 1900s (Blakey 1941, Davis 1949, Dickson 2001). The wild turkey has been successfully reestablished across its range largely because of harvest regulation, habitat reclamation following farmland abandonment in the 1920s and 1930s and coordinated trap and transfer reintroduction (Dickson 2001).

In applied wildlife management, species reintroductions are often conducted to restore wildlife populations to regions where they once existed but were extirpated from (Seddon et al. 2007, IUCN/SSC 2013). Populations of mammals (e.g. Kenup et al. 2018, Cid et al. 2014, Van Houtan et al. 2009), birds (e.g. Elliot et al. 2001, Powlesland et al. 2006, Ortiz-Catedral et al. 2010), reptiles (e.g. Towns and Ferreira 2001, Miller et al. 2011), and fish (e.g. Harig et al. 2000, Lyon 2012) have all been restored through reintroduction projects. However, reintroduction projects generally have a low success rate (Fischer and Lindenmayer 2000, Reading et al. 2002, Lipsey and Child 2007, Seddon et al. 2007, Reading et al. 2013), and there are numerous factors to consider when reintroducing animal populations into novel environments, even when those regions are part of their historical range. For instance, releasing small numbers of individuals may lead to low genetic diversity in reintroduced populations (Stüwe and Scribner 1989, Haig et al. 1990, Leberg 1990) while increasing the genetic divergence among them (Leberg 1990, Scribner and Stüwe 1993, Leberg et al. 1994). Ecosystem composition may have changed substantially from when the species was extirpated to when reintroduction efforts are being considered, resulting in less or unsuitable habitat for the species in question. Furthermore, many reintroduced wildlife populations follow similar population growth patterns in that they tend to experience fast population growth immediately after reintroduction, followed by a period of slower growth, reaching an eventual plateau or decline (Griffith et al. 1989, Wolf et al. 1996, Loyd et al. 2009). As such, ongoing post-release monitoring of reintroduced populations is important in understanding their success, as the likelihood on long-term population sustainability may not be made clear for generations.

Because wild turkeys are a highly desirable game species in North America, their reintroduction has been prioritized. Reintroductions began as early as the 1920s in some areas (Kennamer et al. 1992) and continued into the 1980s (Bailey 1980, Kennamer 1986), leading to many contemporary populations with complex histories of regional extirpations and reintroductions involving both pen-reared (Newman 1945, Mosby 1975) and wild-caught, trapped and transferred individuals (e.g. Williams 1981, Beasom and Wilson 1992, Rhodes et al. 1995, Kennamer and Kennamer 1996). For instance, wild turkeys were extirpated from many areas in the northeastern and midwestern United States, but reintroduction programs involving thousands of birds have restored turkey populations to most of this region (Mosby 1949, 1975, Williams 1981, Kennamer and Kennamer 1996), and in the southwestern United States, the Rio Grande wild turkey underwent population declines followed by reintroductions and transplant efforts in many parts of its range (Beason and Wilson 1992). Because wild turkeys are a generalist species, they have seemingly been able to easily adapt following their reintroduction into modified landscapes and have thrived in contemporary ecosystems with substantially different species compositions than they have had historically. Today, turkey populations occupy all American states, 6 of 13 Canadian provinces or territories, and northern Mexico (Kennamer and Kennamer 1996, Tapley et al. 2001, 2007, 2011).

Wildlife translocation projects are not without cost (Phillips et al. 2003, Lindsey et al. 2005, Wakamiya and Roy 2009, VanderWerf et al. 2013), with one study citing a cost of $350 - $500 per translocated turkey (Restani et al. 2009). There have been several important stakeholders involved in wild turkey translocation projects across North America. For instance, the implementation of the Federal Aid in Wildlife Restoration Act of 1937, often referred to as the Pittman-Roberston Act, imposes an 11% tax on firearms, ammunition, and archery equipment in the United States and distributes the proceeds to state governments for wildlife projects, including translocations (Pittman-Roberston Wildlife Restoration Act 1937). The National Wild Turkey Federation (NWTF) in the United States and the Canadian Wild Turkey Federation (CWTF) in Canada represent two non-governmental stakeholders that have played pivotal roles in wild turkey reintroduction efforts. Both organizations contribute to turkey conservation projects through research grants, volunteerism, and activism, with the explicit goal of promoting and advancing wild turkey harvest opportunities.

Wild Turkeys in Ontario

Before European colonization, wild turkeys were common in forests of southern Ontario as far north as Lake Simcoe (44.46° N; Ontario Ministry of Natural Resources 2007). The pressures from unregulated deforestation and hunting resulted in this species becoming extirpated from Ontario by 1909 (Alison 1976, Ontario Ministry of Natural Resources 2007). Beginning in 1984, the provincial government, Federation of Ontario Naturalists, and the Ontario Federation of Anglers and Hunters (OFAH) implemented a reintroduction program, distributing over 4400 turkeys across 275 sites in the province up until the restoration program ended in 2005 (Ontario Ministry of Natural Resources 2007). Their reintroduction was so successful that by 1987 a spring hunting season was introduced, followed by a fall hunting season in certain areas by 2008. Today, wild turkeys have been described as a reintroduction success in Ontario (Ontario Ministry of Natural Resources 2007). Following the release of a moderate number of individuals, populations have increased in size, dispersed, and remained self-sustaining since reintroduction.

Wild Turkey Management in Ontario

The most recent management plan for wild turkeys in Ontario was published in 2007 with a plan goal to “ensure the sustainable management of turkeys as important components of the biodiversity of southern Ontario, and for the continued social, cultural, and economic benefit of the people of Ontario” (Ontario Ministry of Natural Resources 2007). The plan includes five main objectives related to: (1) the continued restoration of wild turkey populations; (2) the management of wild turkeys in the Mixedwood Plains Ecozone and provision of hunting opportunities in the Boreal Shield Ecozone (Ontario Ministry of Natural Resources and Forestry 2007; Figure 1; (3) the maintenance of spring hunting opportunities and the allowance of novel fall hunting opportunities; (4) the reduction of landowner concerns about turkeys and the mitigation of actual human-turkey conflict situations through education, tools, and best management practices; and (5) the development of ecosystem-based habitat projects that will benefit wild turkeys and other native species. Since the management plan was published, fall hunting opportunities have been offered in several Wildlife Management Units (WMU) in the southeast and southwest regions of the province. However, there are not yet any hunting opportunities in the Boreal Shield Ecozone. There have also been several changes in harvest education and reporting requirements. As of 2017, hunters wanting to harvest wild turkey in Ontario no longer need to complete a separate Wild Turkey Hunting Course prior to harvesting. Information about turkey hunting regulations in Ontario is instead covered under the required Ontario Hunter Education Course and associated test (OMNRF 2023). As of 2019, all hunters that purchase or are issued a tag to hunt wild turkey must complete a Mandatory Wild Turkey Hunter Report. With the information submitted, seasonal harvest rates are estimated along with other metrics, allowing for more informed wild turkey management in the province.

The objectives described in Ontario's wild turkey management plan, in combination with the ways in which wild turkey harvest and anthropogenic development have changed in the province since the publication of the management plan in 2007, highlight the need to better understand the state of Ontario's wild turkey populations: specifically, how the population has grown and changed since reintroduction, their behavioural ecology and genetic patterns given this context, and what we can expect for the future of this species in the province given their associations with anthropogenic development and likely range shifts in response to climate change.   Due to the potential for reintroduced populations to experience boom-and-bust dynamics, long-term post-release monitoring is important for understanding growth trajectories and adaptively managing these populations (Nichols and Armstrong 2012). Furthermore, temporal variation in population size is typically much greater at the edge of a species' range than in the core. As a result, peripheral populations may be more likely to exhibit boom-and-bust cycles in response to fluctuations in abiotic conditions (Thomas et al. 1994, Curnutt et al. 1996). Wild turkeys in Ontario are both reintroduced and persisting close to their northern range edge. As such, monitoring the size, distribution, and health of Ontario's wild turkey population is essential to ensuring their long-term persistence in the province.

Thesis Objectives

The general purpose of my thesis is to further our understanding of wild turkey ecology, social structure, and behaviour, particularly for Ontario's reintroduced populations. To do so I have investigated the following research questions:

- What factors best explain the contemporary size and distribution of Ontario's reintroduced wild turkey population?
- How are anthropogenic food sources influencing wild turkey home range size and composition?
- Is there evidence of kin selection during the non-breeding season for this species?

Thesis Structure

My thesis is divided into 5 main chapters, of which chapters 2, 3, and 4 are directly associated to the main objectives. In chapter 2, I investigated what landscape factors best predict wild turkey distribution in Ontario, and modelled population size using existing density estimates from across their range. In chapter 3, I explored how anthropogenic food sources, like crops and bird feeders, influence wild turkey home range size and composition. I also explored the role that dominance rank and morphology may have in establishing utilization distributions around anthropogenic food sources. In chapter 4, I investigated wild turkey space use patterns related to conspecifics, and more specifically, genetic relatives. I tested the hypothesis that wild turkeys exhibit kin selection during the non-breeding season while foraging and roosting. Finally, chapter 5 presents a synthesis of my results and a discussion of the information gathered in previous chapters. I also provide specific management recommendations for Ontario's wild turkey populations in the light of my findings.

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Chapter 2: Combining community science and MaxEnt modelling to estimate Wild Turkey (*Meleagris gallopavo*) winter abundance and distribution

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Contributions: Baici and Bowman conceived and designed the study; Baici analyzed the data and wrote the manuscript. Bowman critically reviewed the manuscript.

Abstract

Understanding the distribution and abundance of species is a fundamental aspect of conservation biology. Species distribution models aim to predict distributions based on species observations and ecologically relevant information. To understand the contemporary distribution of wild turkeys (*Meleagris gallopavo*) in Ontario, we curated and collated Wild Turkey flock observations from eBird and iNaturalist submitted during winter 2018. We combined these with environmental predictors to build distribution models using MaxEnt and evaluated model fit using 10-fold cross validation. We also estimated total population size for this species under different modeling scenarios. The potential presence of unknown spatial bias in community science datasets is a complex problem often requiring context-specific statistical solutions. Data cleaning, sometimes referred to as thinning, filtering, or culling, is often proposed to manage this bias. As such, we tested the effect of data cleaning on model outputs and on subsequent analyses. We evaluated all models using area under the curve (AUC). We found building density to be the most important environmental variable followed by winter severity. We validated our habitat suitability estimates using fine-scale GPS data and found that data cleaning had no effect on habitat suitability estimates inside available Wild Turkey habitat or inside core-use areas, except at one site in 2012. Use of community collected data offers a

cost-efficient and collaborative method to obtain data for species distribution modeling and management. We discuss implications for wild turkey management and present potential contemporary distribution maps for this species.

Keywords: community science; eBird; habitat suitability; MaxEnt; species distribution model; wild turkey; winter

INTRODUCTION

Understanding the distribution and abundance of species is a fundamental aspect of conservation biology, allowing for informed, endangered and invasive species management (Livingston et al. 1990, Higgins et al. 1999, Gallagher et al. 2010, Guisan et al. 2013), successful reintroductions (Martínez-Meyer et al. 2006, Malone et al. 2017, Massaro et al. 2017, Riaz et al. 2020), ecosystem restoration (Mladenoff et al. 1995, 1997), population viability analyses (Akçakaya et al. 1995, Akçakaya and Atwood 1997, Roloff and Haufler 1997), and harvest management (Jonzén et al. 2001). Species distribution models (SDMs) aim to predict the distribution of a given species based on species observations and ecologically relevant environmental features (Hirzel et al. 2002). Absence data are often unavailable or unreliable, so modelling techniques that require presence-only information are becoming increasingly common in species distribution modelling (e.g. Razgour et al. 2011, Nazeri et al. 2012, Isaac et al. 2014, Tran and Vu 2020). MaxEnt, a presence-only machine learning software (Phillips et al. 2006, 2017), has become the most-commonly used program to generate SDMs as it has been demonstrated to outperform many other presence-only and presence-absence modelling techniques (Elith et al. 2006, Hernandez et al. 2006). MaxEnt estimates the distribution of a given species by finding the distribution that is closest to geographic uniformity (i.e. maximum entropy) subject to constraints provided by given environmental features at each species occurrence location (Phillips et al. 2006, 2017).

One way to curate species presence information over a large geographic area is to look to community-collected data: data collected or processed by volunteers as part of a scientific inquiry (Silvertown 2009). Community science projects can cover larger geographic and temporal scopes than traditional survey types and data derived from such projects are being increasingly used in modeling landscape-scale movements and distributions (e.g. Snäll et al. 2011, Tulloch, et al. 2013, Supp et al. 2015, Bradsworth et al. 2017, Brommer et al. 2017). In fact, it has been argued that any project seeking to collect large volumes of data over a wide geographic area can only succeed with the assistance of community members (Silvertown 2009). Such data are often collected without experimental design or set survey methods (Steen et al. 2019). As a result, these datasets may have unknown spatial bias, imprecise spatial or temporal resolutions, or result in the under- or over-reporting of species (Fitzpatrick et al. 2009, Dickinson et al. 2010, Steger et al. 2017, La Sorte et al. 2018). Because community science can yield a large volume of data, even with these limitations it can be very useful in predicting species distributions and informing management decisions (Dickinson et al. 2010, Gallagher et al. 2010, Guisan et al. 2013, La Sorte et al. 2018).

For some species, SDMs built from unfiltered community data can match or exceed the performance of SDMs built using systematically collected data (Bird et al. 2014, Isaac et al. 2014a, Steen et al. 2019, Johnston et al. 2021). However, for other species, filtering the dataset can greatly improve model performance (Boria et al. 2014, Fourcade et al. 2014, Aiello-Lammens et al. 2015, Kiedrzyński et al. 2017, Steen et al. 2019). As such, it has been suggested that significant time should be invested in determining how to best clean datasets prior to modeling (e.g. Walker and Taylor 2017, Steen et al. 2019) and that cleaning the data should be two-fold, involving (1) thinning the data to manage sampling bias; and (2) filtering the data to manage spatial bias associated with species sociality (Phillips et al. 2006, 2017, Syfert et al. 2013).

Findings from theoretical modelling indicate that cleaned datasets generally produce better models than the full dataset, and that filtering based on survey effort yields more powerful models than filtering based on surveyor knowledge (Kramer-Shadt et al. 2013, Steen et al. 2019, Johnston et al. 2021). However, in most cases, the most appropriate data cleaning practices and model specifications are species and context specific and should be determined using researcher knowledge and preliminary statistical analyses (e. g. tests for spatial-autocorrelation or prior knowledge about sociality).

In practice, community data cleaning methods employed by researchers vary substantially depending on the study and the species. For instance, some researchers have filtered community data by year (Bradsworth et al. 2017) or based on variation around the mean (Supp et al. 2015), whereas others have attempted to account for pseudoreplication by removing observations within a predetermined and ecologically informed distance (Razgour et al. 2011, Bradsworth et al. 2017). Biddle et al. (2021), who studied the distribution of an avian species often considered difficult for community members to identify, filtered data based on geographic location and metrics of species identification accuracy, whereas Coxen et al. (2017) conducted filtering based on geographic distance between observations, also known as rarefication. Inconsistencies in data cleaning methodology may lead to inaccurate comparisons between studies, even when researchers are studying the same or closely related species.

The wild turkey (*Meleagris gallopavo)* is a dark, large-bodied bird native to North America. Prior to European colonization, this species was an important component of forest biodiversity and held substantial cultural significance for Indigenous groups (Aldrich 1967). Agricultural and residential development of North America resulted in large-scale forest loss. This habitat loss, paired with unregulated hunting, led to the extirpation of turkeys in many regions across North America by the early 1900s (Blakey 1941, Davis 1949). The Wild Turkey has been successfully reestablished across its range largely due to the implementation of harvest regulation, habitat reclamation following farmland abandonment in the 1920s and 1930s and coordinated trap and transfer efforts. Wild Turkeys were trapped and transferred across North America beginning in the 1950s and continuing into the 1980s (Mosby 1959, 1973, 1975, Bailey 1980, Kennamer 1986). Contemporary turkey populations occupy all American states, 6 of 13 Canadian provinces or territories, and northern Mexico (Kennamer and Kennamer 1996, Tapley et al. 2001, 2007, 2011).

Across their range, wild turkey population sizes have been estimated using helicopter surveys (Beason 1970, Thompson and Baker 1981, Kubisiak et al. 1997, Butler et al. 2007b, Butler et al. 2008), road-based surveys (Butler, et al. 2005, Erxleben et al. 2008), brood counts (Schwertner et al. 2003, Butler et al. 2007c), rural mail surveys (Ontario Government 1985, Applegate, 1997), harvest rates (Gonnerman 2021), and roost counts (Thomas et al. 1966, Butler et al. 2006). However, these traditional survey types are often costly, time intensive, and require substantial effort, all of which act as a barrier in applying these methods over large geographic areas or temporal scales. As such, most population surveys of this species conducted thus far have been limited in scope and are often unstandardized (Healy and Powell 1999, Butler 2006). In addition, many survey types provide indices of population size or growth, rather than an estimate of true population size, and may need to be repeated over subsequent years to yield informative results and understand their management implications.

Turkeys are highly visible birds, especially in autumn and winter when the birds form large flocks, and their dark plumage forms visual contrast with the white snow. Consequently, we considered wild turkeys to be a good species to use for exploring the potential for use of community science in generating large-scale maps of distribution and abundance for conservation and management. Using wild turkeys as a case study, we investigated the accuracy of SDMs derived from community science datasets for predicting species distribution and abundance. We used the abundance-suitability relationship to investigate Wild Turkey populations for a range of scenarios. Geographic regions with higher environmental suitability should also have larger populations of wild turkeys and vice-versa (de la Fuente et al. 2021, Gonnerman 2021). This hypothesis is termed the abundance–suitability relationship and states that environmental suitability derived from species distribution models should explain the spatial variation in abundance over a species' geographical range (Weber et al. 2017).

High environmental suitability does not always indicate high abundance. Indeed, density can be a misleading indicator of habitat quality (Van Horne 1983). However, wild turkey natural history suggests that there is typically a positive relationship between abundance and suitability for this species (Porter 1992, Gonnerman 2021) and a recent meta-analysis concluded that occurrence data can be a reasonable proxy for abundance, especially for vertebrates (Weber et al. 2017).

We tested the sensitivity of our analyses to data cleaning by building models and generating population size estimates using both the raw community science dataset (un-thinned) and a heavily filtered version of the same dataset in which we attempt to account for replicate flock detections and spatial bias associated with sampling. It has been suggested that, when possible, researchers should externally validate SDMs against real fine-scale spatial use data (Bradsworth et al. 2017). As such, we validated our predicted probabilities and population size estimates against traditional aerial and road surveys as well as fine-scale GPS tracking data in two regions in Ontario, Canada. We expected that community science data would provide the basis for accurate estimation of distribution and abundance, particularly for a common and easily identifiable species like the wild turkey.

METHODS

Study Area

The total land area is 908 699.33 km^2 and it contains the most populated city in Canada: Toronto (Statistics Canada 2017) (Figure 1). The estimated human population size of Ontario in 2022 was 15 034 547, equalling almost 40% of the country's total population (Statistics Canada 2022).

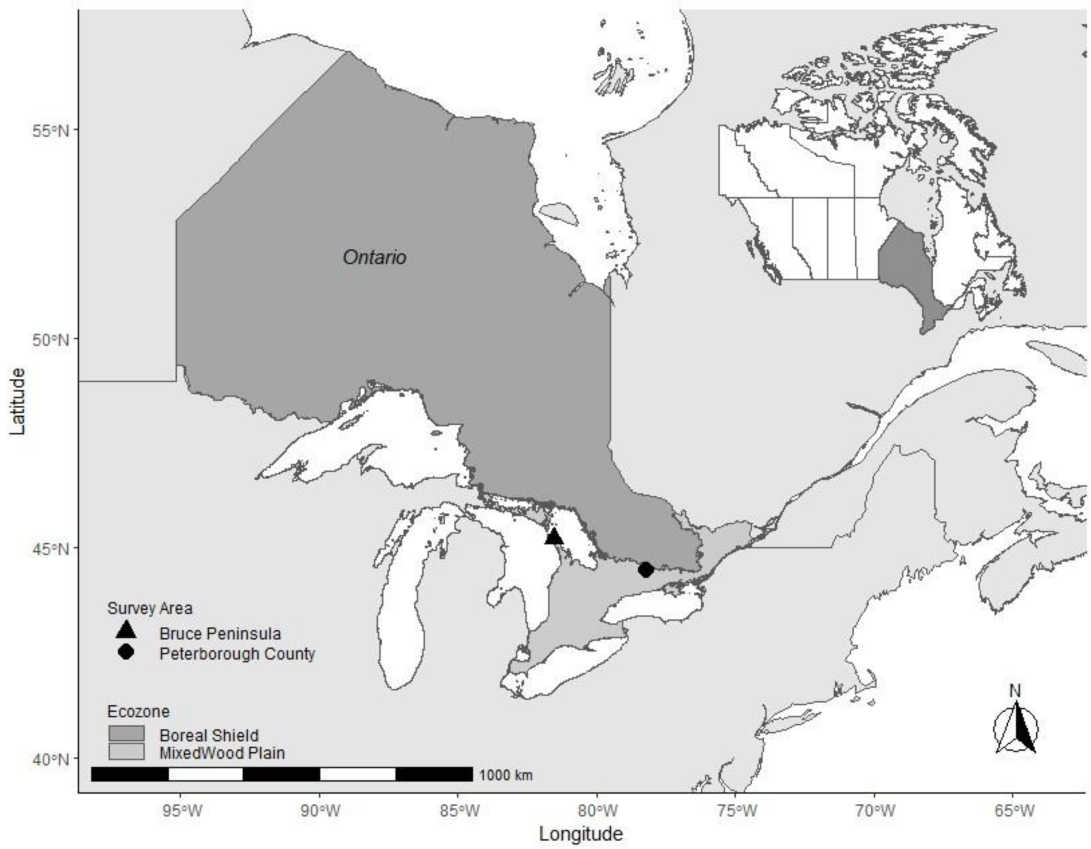

Figure 1. The area over which a wild turkey (Meleagris gallopavo) community science study was conducted in Ontario, Canada.

Ontario contains three of Canada's 15 terrestrial ecozones: the Mixedwood Plains in the south, the Boreal Shield, and Hudson Bay lowlands in the north (Crins et al. 2009). Historically, wild turkeys only inhabited the Mixedwood Plains ecozone. However, land conversion in the north has allowed Wild Turkeys to expand their range northward and inhabit regions of the Boreal Shield that have been converted for agriculture or other anthropogenic developments (OMNR 2007). Eastern wild turkeys (*Meleagris gallopavo silvestris*), the only subspecies present in Ontario, are found in eastern North American from Ontario to Florida (Fink et al. 2021). As such, Ontario encompasses the northern range edge of the Eastern wild turkey.

Ideal wild turkey habitat has been described as an even mix between primarily deciduous forest and open field (Kurzejeski and Lewis 1985). However, large contemporary turkey flocks can be found in regions that do not meet these criteria, suggesting that wild turkeys may display high adaptability to anthropogenic landscapes (OMNR 2007). For instance, although the geographic range of wild turkeys is seemingly expanding northward, this species still primarily inhabits the southern portion of the province, which is characterized by a high level of disturbance, habitat fragmentation, and agricultural development (OMNR 2007).

The wild turkey was considered extirpated from Ontario in 1909 (OMNR 2007). In 1984, efforts began to reintroduce this species to the province and from 1984 to 1987, 274 wild turkeys were trapped and transferred to Ontario from Missouri, Iowa, Michigan, New York, Vermont, New Jersey, and Tennessee (OMNR 2007). In subsequent years, to support the dispersal of this species in Ontario, over 4000 individuals resulting from the initial founder population were trapped and transferred to numerous sites throughout potentially suitable habitats in Ontario primarily within the Mixedwood Plains ecozone. The most recent estimate of Ontario's wild turkey population size, published in 2007, was 70 000 individuals. However, this estimate is a rough approximation based on annual harvest and an assumption about harvest rate (OMNRF 2007).

Data Collection and Cleaning

To compile known wild turkey observations from across Ontario, we conducted a community science campaign in which we requested that participants submit Ontario Wild Turkey observations to eBird (Sullivan et al. 2009) or iNaturalist (California Academy of Sciences 2020) from 1 December 2018 to 31 Mar 2019. eBird is a virtual tool that allows users to submit and review observations of bird species anywhere in the world. The community-collected data is then archived and freely accessible to anyone, including researchers (Sullivan et al. 2009). iNaturalist is a similar observation reporting platform to eBird however, it extends to all taxa. iNaturalist also differs from eBird in that it encourages users to submit additional media with each observation. For instance, users may upload a photograph of a plant species or an audio recording of a bird call to aid in species identification.

To advertise our project and encourage wild turkey observation submissions to each platform, we made an informative poster with a summary of important details about the project and instructions regarding how to submit observations (Appendix B, Figure A3.1). This poster was circulated to field naturalist groups across the province and was shared on multiple social media platforms. If participants sought more information about the project, they were encouraged to visit a website we produced with multiple pages presenting further details about the project and updates about our findings.

We ran this campaign during Ontario's winter because turkeys are dark-coloured and are thus more visible in the winter months against the contrasting white snow. Turkeys also congregate into large flocks during the winter (Healy 1992), allowing observers to easily detect multiple individuals at one time. We chose to use eBird (Sullivan et al. 2009) and iNaturalist (California Academy of Sciences 2020), as both reporting platforms are free, well-known, and already widely used (Silvertown 2009). The popularity of these two platforms also allowed us to include observations from individuals that may not have been familiar with our community science campaign because all wild turkey observations submitted within the reporting period were included in our analysis.

Removing Replicates and Thinning the Data

To examine how thinning the data may impact SDM performance in this case, we ran models with two versions of our community collected dataset: one containing all observations collected during the study period and another heavily cleaned version of the dataset that we (1) filtered for replicate observations and (2) thinned to account for potential underlying spatial bias. This resulted in a raw, un-thinned version of the dataset (n = 5846) and a heavily filtered and thinned version of the same dataset (n = 492).

To generate the thinned dataset we first identified and removed potential replicate observations. During the winter, wild turkeys congregate and move through the landscape in large flocks and individuals are rarely observed alone (Healy 1992). As such, during this time of year, individual home ranges can be used to approximate the size and distribution of whole flock home ranges. Thus, to identify potential replicate observations of flocks, we first estimated the average winter home range size of wild turkeys in two regions of the province: the Bruce Peninsula and Peterborough County (Figure 1) with data derived from individuals that had been GPS-tagged for research projects during 2011-2012 on the Bruce Peninsula and during 2017-2019 in Peterborough County. Birds were captured at baited locations using a rocket net (Grubb 1988) following methods outlined by Niedzielski and Bowman (2014, 2016). Processing entailed weighing, sexing, collecting a blood sample for DNA extraction, and GPS-transmitter (Model PP-VHF-3600L, Lotek, Newmarket, Canada) attachment. The schedule at which GPS locations were recorded varied depending on the capture year and location and ranged from once per hour to once every 4.25 hours.

We estimated wild turkey winter home range by calculating 95% kernel home range polygons using GPS points collected during the winter for 45 individuals (24 M, 21 F). Kernels were estimated using the adehabitatHR package in R Studio (Calenge 2006). We selected bandwidth values using an ad-hoc approach in which we reduced bandwidth values from 100% percent of the reference bandwidth (246.95 in Peterborough County and 144.77 on the Bruce Peninsula) to 10%, by 10% decrements, and then selecting the lowest bandwidth that resulted in the same number of polygons as the reference bandwidth. Individuals who were tracked for less than 13 days, or approximately 15% of the season, were excluded from the analysis (Niedzielski and Bowman 2016). Across both sites and both sexes, we estimated a mean winter home range size (95% kernel) of 1.8 km^2, ranging from 0.06 km^2 to 6.58 km^2, with a standard deviation of 1.45 (Appendix B, Table A2.1).

Observation clusters were identified using the leader algorithm from the leaderCluster package in R Studio (Arnold 2014). This clustering algorithm allows users to set the approximate radius of clusters rather than the desired number of clusters (Arnold 2014). This is useful when attempting to assign observations to clusters based on geographic distance. We selected a radius of 756 m because this value results in circular clusters with approximately the same area as the average wild turkey mean winter home range (1.8 km^2; see Appendix B, Table A2.1). For each cluster, only one observation was retained. Observation locations were then converted to the centroid of each cluster. This differed from the un-thinned dataset in which observation locations were kept at the same location as reported by participants.

In addition to filtering for replicate observations, we also thinned this dataset, using the spThin package (Aiello-Lammens et al. 2019) in R Studio (R Core Team 2013, Version

2022.12.0, Build 353), to achieve an approximate uniform density of 1 flock per 10km^2. This was done to account for potential unknown spatial bias in the dataset, e.g., observer density. (Aiello-Lammens et al. 2015).

Multiple users may sometimes submit identical information. For instance, eBird users may submit shared checklists: lists submitted by multiple users that contain identical observations. We did not intentionally remove shared eBird checklists. However, if the observations shared identical GPS locations for the observed individuals, then the observations would have been identified as replicate observations in geographic space and removed. We did not remove observations associated with incidental checklists (Sullivan et al. 2009) but we did compare the effect of removing observations associated with checklists that exceeded 5 hours, 5 kilometres in length, or both, as is recommended under eBird's best practices (Strimas-Mackey et al. 2020). Upon comparing models built with datasets including observations from long (spatially and temporally) checklists with models built without these observations, we found no substantive effect on any model estimates including AUC, variable contributions, and threshold values. As such, we chose to include observations associated with long (spatially or temporally) checklists in our analyses.

Species Distribution Modelling

To estimate the distribution of wild turkeys in Ontario, we collected and analysed six environmental variables and modeled their relationship to community-collected wild turkey observations. We modeled this relationship using MaxEnt, a machine learning process that determines the spatial probability distribution of a species based on presence-only records and relevant environmental variables (Phillips et al. 2006, 2017). Winter is the season that most limits wild turkey abundance at their northern range edge (Niedzielski and Bowman 2014, Gonnerman 2021), and the time when turkeys are the most visible due to their dark colour and flocking behaviour. Therefore, we focused on estimating the distribution and abundance of Wild Turkeys in winter. We evaluated winter habitat suitability across the wild turkey range in Ontario (Phillips and Dudik 2008).

We selected environmental variables on the basis of turkey ecology and relationships among variables. For instance, turkeys are a generalist, forest-dependent species (Healy 1992, Porter 1992). Wild Turkeys are also heavily reliant on supplemental food sources, particularly during the winter months when their natural food sources are less available (Vander Haegen et al. 1989, Roberts et al. 1995, Paisley et al. 1996, Nguyen et al. 2003, Kane et al. 2007, Restani et al. 2009). Thus, we included both forest and agricultural land cover in our model. Using ArcMap (ESRI 2011), we created the forest and agriculture variables by combining land cover classes within the OMNR Provincial Land Cover Dataset (2000). Cells representing coniferous forest and deciduous forest were combined and exported as a new data layer to create our forest variable, and cells representing pasture and cropland were combined and exported to create our agriculture variable. Both variables were converted to binary, categorical variables prior to modeling; cells that contain the respective landscape cover type (1) and cells that do not (0).

In addition to plant and animal agricultural operations, bird feeders may also represent a winter food source for wild turkeys in Ontario (Niedzielski and Bowman 2014). Thus, two anthropogenic variables were included to represent human development as a proxy for human-maintained supplemental food sources: road density (OMNR 2010 - 2013) and building density (OMNR 1977 - 2014). Wild turkey presence may also have a negative association with road density, as individuals closer to roads may be more likely to experience increased road mortality as a confounding effect of hunting pressure (Holbrook and Vaughan 1985) and/or increased predation (Thogmartin and Schaeffer 2000). Road density was estimated by calculating the cumulative road length, in kilometres, per pixel. Building density was estimated by calculating the number of buildings per pixel.

It is known that wild turkeys are affected by deep, powdered snow in that it restricts their movement across the landscape during the winter months, reducing their ability to forage and evade predators (Austin and DeGraff 1975, Wunz and Hayden 1975, Porter 1977, Porter 1980, Roberts et al. 1995, Wright et al. 1996, Nguyen et al. 2003, Kane et al. 2007, Gonnerman 2021). There has also been one reported instance of a wild turkey dying from frostbite in Ontario (MacDonald et al. 2016). Thus, to represent differences in winter severity across their range, minimum winter temperature and average snow depth were included in the model. Climate data was retrieved from Environment and Climate Change Canada weather stations in Ontario using the weathercan R package (LaZerte and Albers 2018). We interpolated values between the stations using kriging (ESRI 2011). Kriging is a form of spatial interpolation that uses mathematical formulas to estimate a continuous surface of values. Kriging assumes that there is a structural component present (in this case, observations) and that the local trend varies from one location to another (ESRI 2011).

To standardize the cell size and spatial extent across layers, each layer was rasterized, and values were assigned to 2.27 x 2.27km square pixels (5.15km^2) covering the extent of the survey area (ESRI 2011). This cell size was selected for multiple reasons. First, to allow for multiple flock observations to occur within the same cell because we estimated a mean winter home range size (95% kernel) of 1.8 km^2. Furthermore, all data layers were resampled to a resolution of 5.15km^2 as this value allowed for multiple wild turkey flock observations to occur within the same cell for the un-thinned dataset prior to MaxEnt modelling and allowed for the standardizing of cell sizes across data layers of different spatial scales. We then converted all layers to data format *.asc for import into MaxEnt.

In MaxEnt, model restrictions are applied as feature types. Hinge features represent piece-wise linear functions in that they behave like linear functions with thresholds allowing the steepness and direction of the linear relationship to differ below and above each threshold (Phillips 2017). As such, hinge features tend to make linear and threshold features redundant (Elith et al. 2010). The default setting in MaxEnt is to employ auto features which allows the software to tune parameters based on model performance (Phillips & Dudík 2008). To reduce the potential of overfitting, our final models included linear, quadratic, product, and threshold feature types, but not hinge (Phillips & Dudík 2008, Elith et al. 2010).

Prior to setting the user-specified parameters in MaxEnt, we used the “ENMeval” R package to identify the most appropriate regularization parameter value for our dataset (Muscarella et al. 2014). Forty-eight models with combinations of restrictions (feature types) and regularization multipliers were compared to select the most appropriate multiplier value. The regularization multiplier is a parameter that helps to prevent model over-complexity and/or over-fitting (Elith et al. 2010). The MaxEnt default value is centered at 1.0. A regularization parameter less than 1.0 will produce estimates with a more localized output distribution with a closer fit to the presence records provided. A regularization parameter greater than 1.0 will produce estimates with a less localized prediction (Phillips 2017). We developed models with multiplier values increasing from 0.5 to 4 by increments of 0.5).

Akaike Information Criterion scores corrected for sample size (AICc) were generated by ENMeval for all models, with the lowest score indicating the model with the highest maximum likelihood estimate. This analysis indicated that a regularization multiplier of 1.5 would be most appropriate to model wild turkey distribution as this value yielded the lowest AICc (Isaac et al. 2014b).

The climate variables “snow depth” and “minimum seasonal temperature” were highly correlated (R^2 = -0.92; R Core Team 2017). Research shows that wild turkey movement is limited during the winter by both deep snow and cold temperatures resulting in higher mortality from starvation and predation (Austin and DeGraff 1975, Wunz and Hayden 1975, Porter 1977, Porter 1980, Roberts et al. 1995, Wright et al. 1996, Nguyen et al. 2003, Kane et al. 2007). As such, two global models were generated: (1) minimum temperature as a climate variable and (2) snow depth as a climate variable. Both global models were run using the thinned dataset and the un-thinned dataset. All models were run using 10-fold cross-validation, which allowed us to calculate summary statistics. By running the models using 10-fold cross-validation, we have parsed the larger dataset into 10 subsets and calculated summary statistics, such as mean AUC from the outputs. For each model, we also examined variable contributes and habitat suitability threshold values. For each of the 10 runs, 90% of the dataset was used to train the model, and 10% was used to test the model.

To evaluate each model’s goodness-of-fit, we referred to the Area Under the Curve (AUC) of the Receiver Operator Characteristics (ROC; Pearson et al. 2006, Phillips et al. 2006, 2017).

Area under the curve measures the ability of a probabilistic model output to correctly distinguish presence from random locations (Phillips et al. 2006, 2017). When assessing the importance of each of the variables in the models, we selected the jackknife method: a resampling technique that provides an estimate of variance by analyzing subsamples of size (n – *1*) obtained by omitting one observation during each run (Wu 1986). This method is effectively sampling without replacement.

The MaxEnt software, version 3.4.4., (Phillips et al. 2006, 2017) generated a raster layer with pixels representing the probability of presence from 0 – 1, following a user-specified conditional log-log (clog-log) transformation. It is important to note that in models relying on presence-only records, such as MaxEnt, the probability of presence refers to the probability of wild turkey presence given the observations used by MaxEnt's algorithm and their association with the layers included. As such, from MaxEnt's probability of presence, we can only infer habitat suitability rather than true probability of a species' presence in an area.

Population Size Estimation

To account for uncertainty in our modelling process and in the dataset, we generated a range of potential population sizes using our thinned (n = 492) and un-thinned (n = 5846) datasets (Steen et al. 2019) under different habitat suitability threshold scenarios (Liu et al. 2016) and under different population density scenarios.

To distinguish between suitable and unsuitable habitat, we used four habitat suitability threshold values, two thresholds identified by MaxEnt: minimum training presence (MTP) and 10^{th} percentile training presence, and two thresholds assigned post-hoc based on a visual assessment of the spread of the data (Appendix B, Figure A3.1). The MTP threshold finds the lowest predicted suitability value for an occurrence point and is thus the least conservative threshold value resulting in the largest predicted range. The 10th percentile training presence omits all regions with habitat suitability estimates lower than the suitability values for the lowest 10% of occurrence records (Phillips et al. 2006, 2017). This is a more conservative value and is commonly used in SDM studies (e.g. Raes et al. 2009, Rebelo and Jones 2010). Prior to generating population size estimates, raster cells with habitat suitability values less than the defined threshold for each scenario were removed from the dataset resulting in four potential geographic ranges of wild turkeys in Ontario.

To estimate potential densities of turkeys in Ontario, we first calculated the mean density of Wild Turkeys reported for each U. S. state by Erikson et al. (2014). Researchers compiled estimates of total population size by surveying state wildlife agencies (Table 3 in Erikson et al. 2014). For each state, researchers also reported the estimated occupied range of Wild Turkeys (Table 4 in Erikson et al. 2014; Appendix B, Table A2.5). We divided the estimated total population size by the estimated occupied range for each state then calculated the mean and standard deviation. To generate values representative of low- and high-density estimates, we subtracted and added one standard deviation to the mean, respectively (2.07 +/- 0.817).

To examine the sensitivity of our analysis to various user-specific parameters, we also generated estimates using un-thinned data and an inflated mean density of 4.02: the mean density of wild turkeys in Alabama and the highest density reported in the United States (Erikson et al. 2014). We used a standard deviation of 2.07 to generate low- and high-density estimates around the inflated mean.

For each habitat suitability threshold scenario, the suitability values were binned to identify the proportion of values contained within each bin and to examine the overall distribution of the data. Using a "for loop" in R (R Core Team 2017), we generated a range of population size estimates by selecting 1000 habitat suitability values proportional to the underlying distribution of the data. For instance, the SDM built using thinned data predicts that 21.4% of the habitat suitability values fall between 0.7 and 0.79. Thus, we randomly selected 214 (21.4% of 1000) samples from this bin.

We then assigned these values as thresholds between low- and high-quality habitat and categorized raster cells below this value as likely containing habitat suitable for low-density populations (1.25 turkeys per km^2), and raster cells above this value as likely containing habitat suitable for high-density populations (2.89 turkeys per km^2). We then calculated the geographic area of the low- and high-density habitat and multiplied the areas by our estimated low and high turkey densities to generate a distribution of potential population size estimates from which we could calculate summary statistics.

Validation of SDM Against Fine-Scale Tracking Data

It has been suggested that more researchers should consider external validation of species distribution models against real, fine-scale spatial data (Bradsworth et al. 2017). Thus, we attempted to validate the results of our SDM against wild turkey GPS tracking data collected within two regions in Ontario: Peterborough County and the Bruce Peninsula (Figure 1).

Peterborough County is a region in south-central Ontario, characterized by mixed forest, plant, and animal agricultural operations. The Bruce Peninsula also contains a mix of land cover types but is characterized primarily by coniferous forest, mixed forest, fields, and deciduous forest (Niedzielski and Bowman 2016). The dominant industries in both regions are agriculture and tourism. The main crops grown in Peterborough County are soybean, corn, and winter wheat. Pasture and hay fields are also significant components of the landscape to support animal agriculture. Animal agriculture is the predominant type of farming in the Bruce Peninsula in the form of small-scale beef farming (Niedzielski and Bowman 2016). Both regions are located nearby to publicly accessible provincial parks: Kawartha Highlands Provincial Park in the northern half of Peterborough County and Bruce Peninsula National Park on the north-western tip of the Bruce Peninsula.

Using GPS data collected in the Bruce Peninsula (Niedzielski and Bowman 2014) and Peterborough County, we compared the mean habitat suitability within four hierarchical levels of wild turkey space use: (1) annual available habitat (a 100% minimum convex polygon (MCP) around pooled wild turkey GPS locations); (2) winter available habitat (a 100% MCP around pooled Wild Turkey winter GPS locations, (3) wild turkey winter home ranges (95% kernel home range polygons); and (4) wild turkey winter core use areas (50% kernel home range polygons). See supplemental material in Appendix B, Figure A3.3 and Koen et al. 2014. We calculated the 50% kernel home range polygons using the same percent bandwidth as when calculating the 95% kernel home range polygons. That is, the bandwidth selected using our ad-hoc approach. We defined winter from December 21 to March 20, the calendar winter season in North America, as calendar seasons align with seasonal changes in turkey ecology and behaviour (Kurzejeski et al 1987, Badyaev et al. 1996, Humberg et al. 2009, Niedzielski and Bowman 2014). Wild turkeys were captured and fitted with transmitters in the winter in both Ontario study areas. As such, we defined each year from December 21 (the first day of winter) to December

20 (the last day of fall) the following year. For instance, year one begins on 21 December 2016 and ends on 20 December 2017.

To the 100% MCPs representing annual available and winter available habitat, we added buffers based on the annual and winter wild turkey home range sizes. We estimated the mean annual and mean winter wild turkey home range size using 95% kernels and then calculated the diameter of a circle equivalent in area to the average home range size for each time-period. We then added this value, in meters, as a buffer to the respective available habitat MCPs (Appendix B, Table A2.8).

Using t-tests, we explored whether thinning the data had a significant effect on habitat suitability values inside wild turkey home ranges and core use areas. If our habitat suitability models accurately predict wild turkey probability of presence, then we expected that suitability predictions would be higher in known areas of wild turkey space use than in surrounding areas.

Validation of Density Estimates Against Survey Data

In addition to validating our results against fine-scale GPS tracking data, we also compared our provincial density estimates against density estimates derived for our Peterborough County and Bruce Peninsula study sites using systematic survey techniques.

Peterborough County

We employed three wild turkey survey methods during the winter of 2017 (1 December 2017 to 31 March 2018) to detect and estimate wild turkey density in a 422 km^2 region within a smaller survey region in Peterborough County (Figure 1). Within this region, we employed an aerial survey, a community science survey, and an opportunistic road survey. For a complete description of each survey method and data processing, including replicate detection, see Appendix B1.

We investigated the accuracy of our model-predicted density estimates by comparing them with an estimate we derived for the survey region by combining population counts across survey types while also accounting for replicate observations.

<u>Bruce Peninsula</u>

A comprehensive survey was conducted on the Bruce Peninsula in 2012 during which time most roads along the peninsula were driven numerous times throughout the season, and the locations of all observed turkey flocks were recorded. This included a turkey capture and telemetry program. The Bruce Peninsula is a geographic region on the eastern shore of Lake Huron dividing the lake from Georgian Bay (Figure 1). The peninsula lends itself to a road survey as a census technique for this species because the roads provide good visibility of the agricultural land and operations on the peninsula: an important resource for contemporary wild turkeys (Vander Haegen et al. 1989, Roberts et al. 1995, Paisley et al. 1996, Nguyen et al 2003, Kane et al. 2007, Restani et al. 2009).

Again, we investigated the accuracy of our province-wide density estimates with an estimate of turkey density calculated for a 1140km^2 region on the Bruce Peninsula (Figure 1).

RESULTS

Community-Collected Data

We received 5846 observations of wild turkeys in Ontario from 1 December 2018 to 31 March 2019. The vast majority, 97%, were submitted through eBird, with only 2% submitted through iNaturalist, and 1% received through direct email (Figure 2a). After filtering to account for potential replicate observations and spatial bias, our thinned dataset included 492 flock locations (Figure 2b).

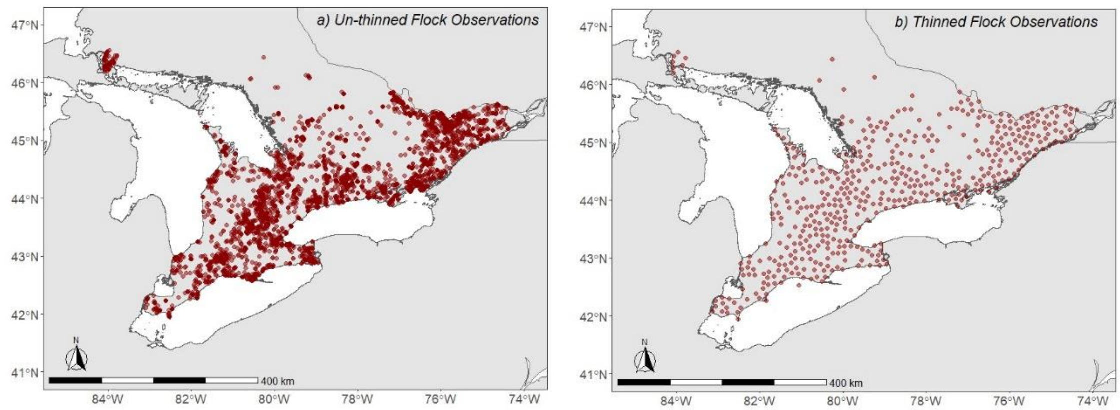

Figure 2. Wild turkey (Meleagris gallopavo) flock observations collected through our community science campaign: a) un-thinned observations (n = 5846), b) thinned observations, (n = 492). See eBird basic dataset 2019, California Academy of Sciences 2020.

Species Distribution Models: Un-Thinned Data

Model 1 (climate variable: minimum winter temperature), which employed linear, quadratic, product, and threshold feature types with a regularization value of 1.5, yielded a mean test AUC of 0.73 (0.01). Model 2 (climate variable: snow depth), which employed the same feature types and regularization value as Model 1, also yielded a mean test AUC of 0.73 (0.01). For both models, variable jackknifing indicated that building density had the strongest contribution to the model (model 1: 67.70%, model 2: 58.24%). For model 1, minimum annual temperature had the next strongest contribution to the model (27.55%), and for model 2, snow depth had the second strongest contribution (38.10%). See Table 1.

Table 1. A summary of each covariate's contribution to models 1 through 4 in order of importance. Values represent means across 10 replicates using 10-fold cross validation. Standard deviation is stated in parentheses. Note: AUC = area under the curve.

Covariate	Variable contribution to model 1	Variable contribution to model 2	Variable contribution to model 3	Variable contribution to model 4
Building density	67.72 (0.93)	58.25 (1.67)	84.74 (2.59)	78.34 (3.37)
Snow depth	n/a	38.10 (1.56)	n/a	18.09 (3.8)
Minimum temperature	27.55 (0.57)	n/a	11.04 (2.59)	n/a
Road density	2.33 (0.42)	2.33 (0.35)	1.29 (0.37)	1.24 (0.34)
Agriculture (presence/absence)	1.72 (0.70)	0.74 (0.45)	2.04 (1.33)	2.01 (1.7)
Forest cover (presence/absence)	0.66 (0.73)	0.58 (0.34)	0.89 (0.83)	0.3 (0.6)
Mean training AUC	0.74 (0.001)	0.74 (0.001)	0.74 (0.003)	0.74 (0.004)
Mean test AUC	0.73 (0.01)	0.73 (0.01)	0.72 (0.03)	0.72 (0.03)

For model 1, habitat suitability trended in a positive direction as the minimum temperature increased, with a steep decrease in suitability when minimum temperatures dropped below -13° C (Figure 3a). For model 2, the habitat suitability decreased with snow accumulation beginning at a snow depth of approximately 30 cm (Figure 4a). For both models, suitability and building density were positively correlated until an apparent density threshold of approximately 250 buildings per pixel when the slope began to flatten (Figures 3b and 4b).

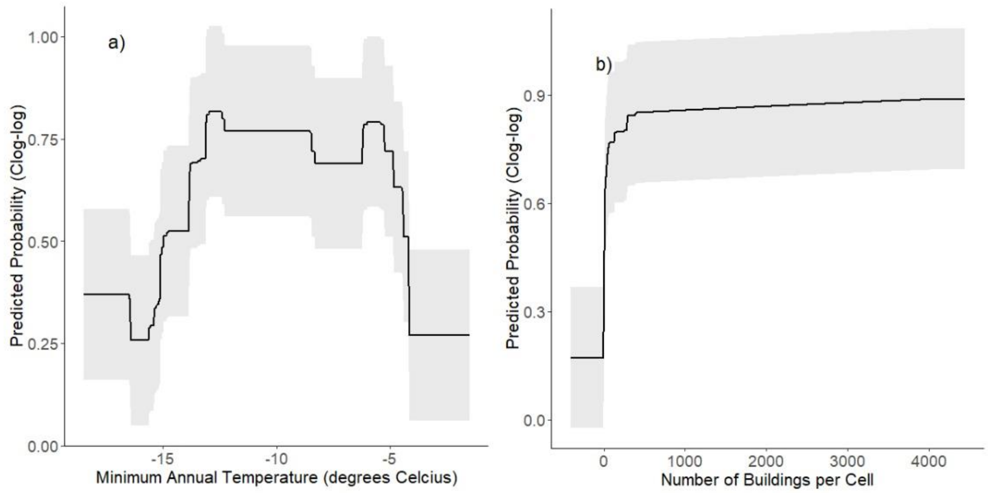

Figure 3. Response curves of a) minimum temperature and b) building density under model 1 (un-thinned data). Figures depict the conditional log-log output on the y axes and the respective predictor variables on the x axes. Gray regions represent +/- 1 standard deviation. The curves show how the predicted probability of presence changes as each variable is varied, keeping all other variables in the model at their mean value.

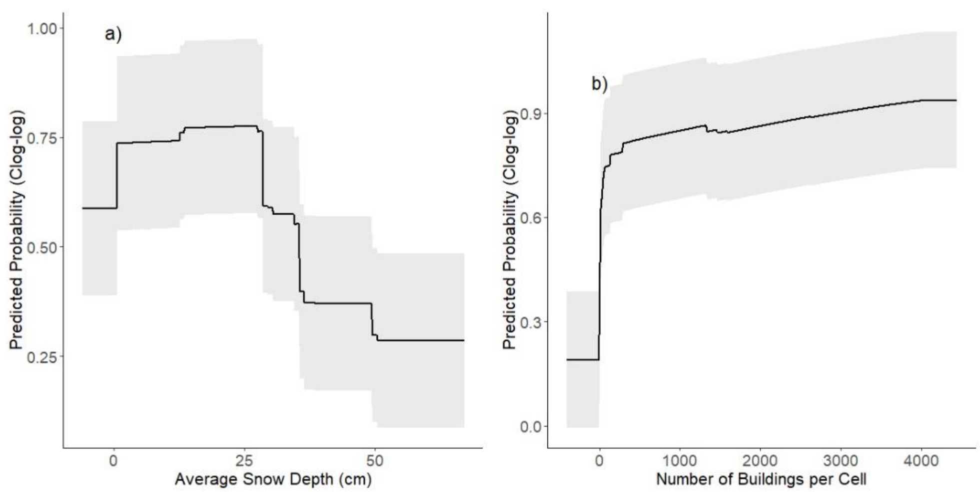

Figure. 4. Response curves of a) snow depth and b) building density under model 2 (un-thinned data). Figures depict the logistic output on the y axes and the respective predictor variables on the x axes. Gray regions represent +/- 1 standard deviation. The curves show how the predicted probability of presence changes as each variable is varied, keeping all other variables in the model at their mean value.

Species Distribution Models: Thinned Data

Models 3 and 4 were run with identical parameters to models 1 and 2 but used the thinned subset of the original data. Model 3 (climate variable: minimum winter temperature) yielded a mean training AUC of 0.72 (0.03). Model 4 (climate variable: snow depth) also yielded a mean training AUC of 0.72 (0.03). For both models, variable jackknifing indicated that building density had the strongest contribution to the model (model 3: 84.74%, model 4: 78.34%). For model 3, lowest annual temperature had the next strongest contribution to the model (11.04%), and for model 4, snow depth had the second strongest contribution (18.09%). See Table 1.

For model 3, there appeared to be no relationship between habitat suitability and minimum annual temperature until the temperature dropped below -14° C at which point the suitability values also begin to decrease (Figure 5a). For model 4, the habitat suitability decreased as snow depth increased beginning at a snow depth of approximately 10 cm (Figure 6a): 20% lower than was predicted using un-thinned data. For both models, the suitability values increased linearly as building density increased until an apparent density threshold of 1500 buildings per pixel at which point the suitability dropped from 0.92 to 0.55 (Figures 5b and 6b). Following this threshold, building density and habitat suitability were again positively linearly related.

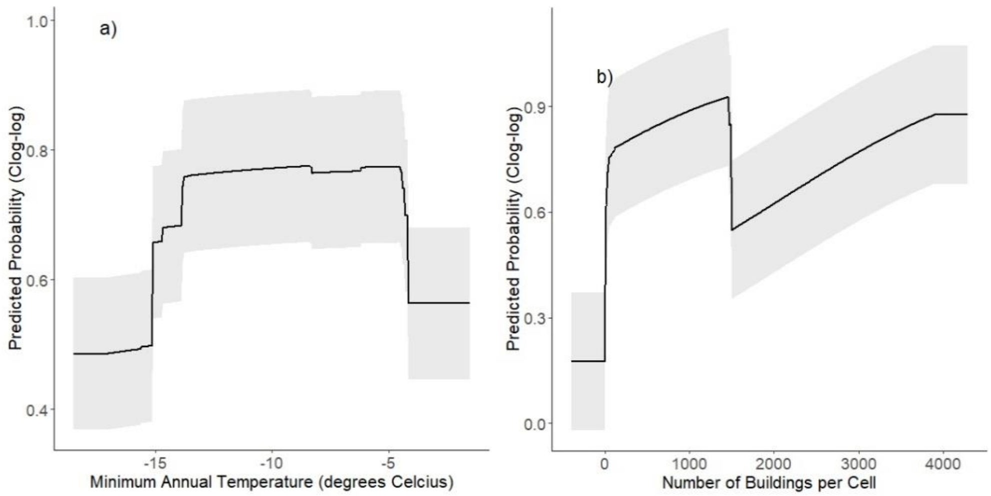

Figure 5. Response curves of a) minimum temperature and b) building density under model 3 (thinned data). Figures depict the logistic output on the y axes and the respective predictor variables on the x axes. Gray regions represent +/- 1 standard deviation. The curves show how the predicted probability of presence changes as each variable is varied, keeping all other variables in the model at their mean value.

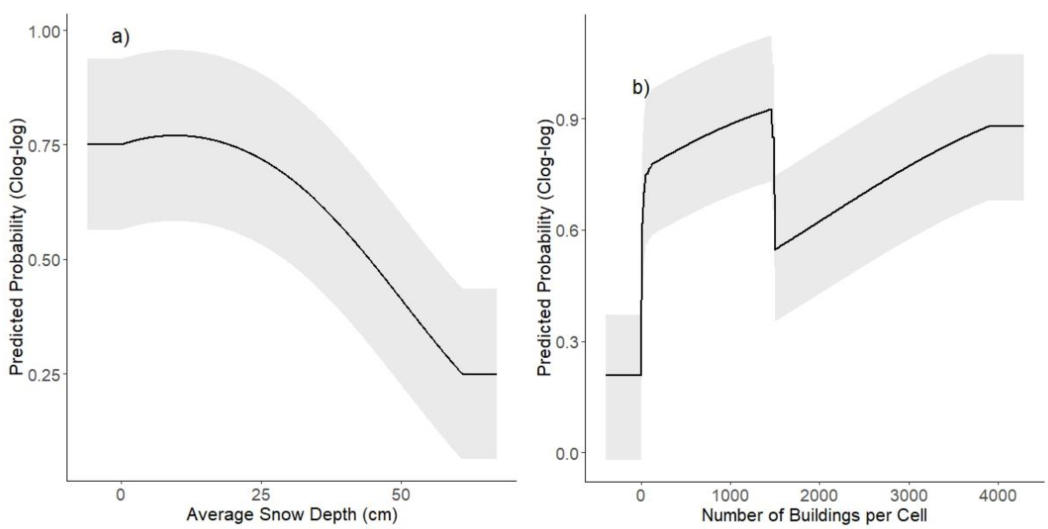

Figure 6. Response curves of a) snow depth and b) building density under model 4 (thinned data). Figures depict the logistic output on the y axes and the respective predictor variables on the x axes. Gray regions represent +/- 1 standard deviation. The curves show how the predicted probability of presence changes as each variable is varied, keeping all other variables in the model at their mean value.

Contemporary Range Map and Density Estimates

The models with temperature as the climate variable (1 and 3) reported almost identical AUC scores to models with snow depth as the climate variable (2 and 4). There is more empirical evidence to support the relationship between snow depth and winter turkey survival (Austin and DeGraff 1975, Wunz and Hayden 1975, Porter 1977, Porter 1980, Roberts et al. 1995, Wright et al. 1996, Nguyen et al. 2003, Kane et al. 2007, Gonnerman 2021) than there is to support the relationship between temperature and winter turkey survival (MacDonald et al. 2016). Therefore, we chose to estimate current wild turkey density and distribution based on habitat suitability predictions from models 2 and 4, using snow depth as the climate variable (Figures 7 and 8).

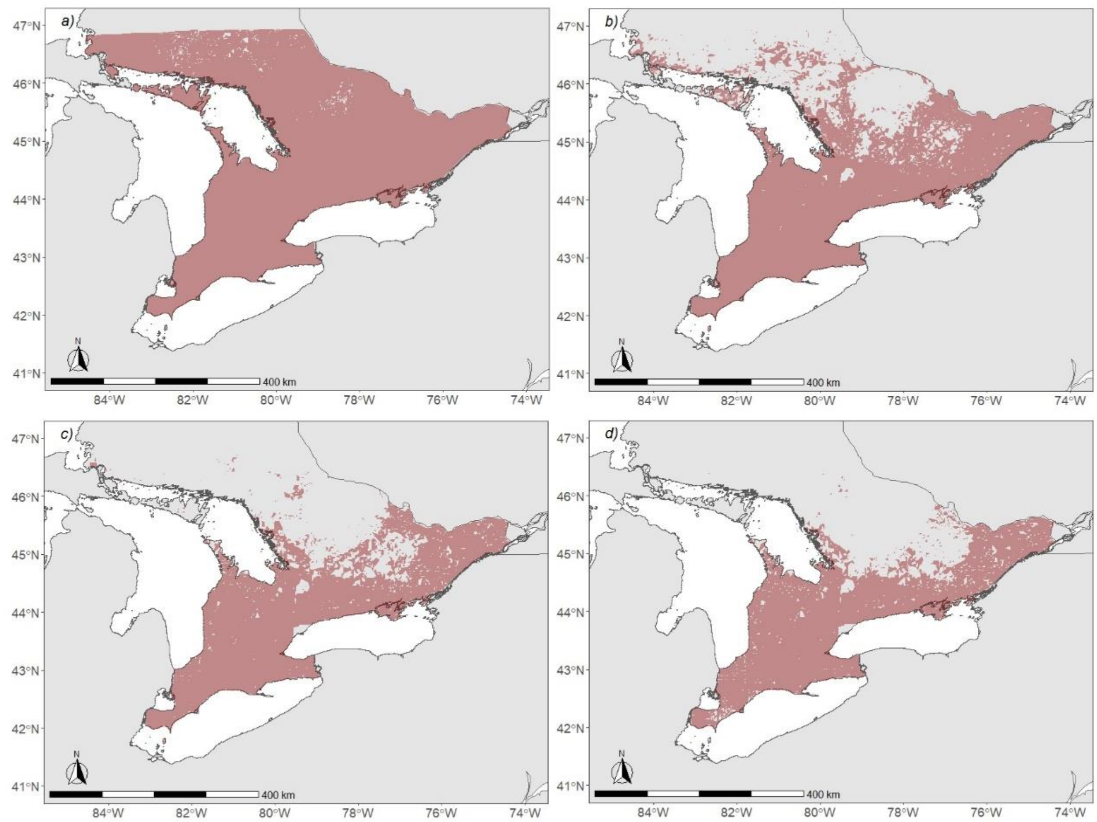

Figure 7. Potential ranges of suitable wild turkey (Meleagris gallopavo) habitat in Ontario, estimated using un-thinned data, under different habitat suitability threshold scenarios. Cells containing values less than the identified threshold value are excluded from the range. Top left: a) minimum training presence (0.02), top right: b) post-hoc 1 (0.12), bottom left: c) post-hoc 2 (0.32), bottom right: d) 10th percentile training presence (0.49). Thresholds are expressed on a conditional log-log (clog-log) scale.

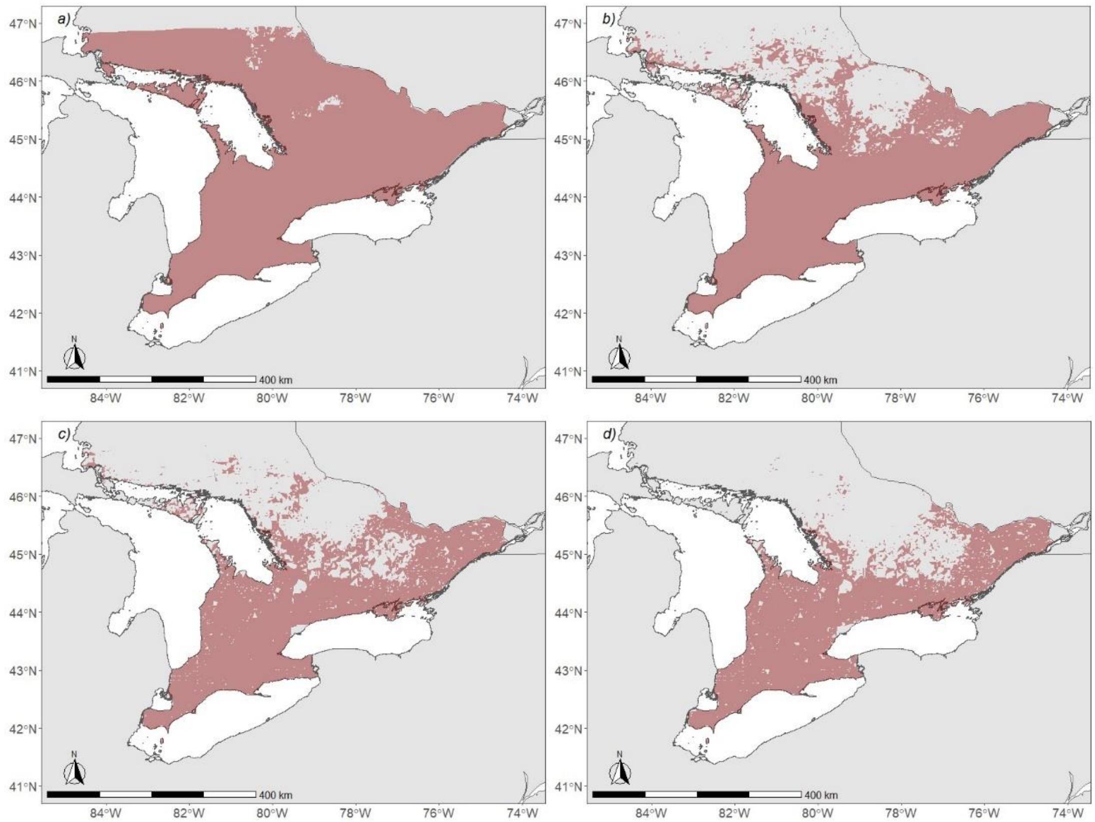

Figure 8. Potential ranges of suitable wild turkey (Meleagris gallopavo) habitat in Ontario, estimated using thinned data, under different habitat suitability threshold scenarios. Cells containing values less than the identified threshold value are removed from the range. Top left: a) minimum training presence (0.05), top right: b) post-hoc 1 (0.12), bottom left: c) post-hoc 2 (0.32), bottom right: d) 10th percentile training presence (0.51). Thresholds are expressed on a conditional log-log (clog-log) scale.

When using un-thinned data and assuming low and high density estimates of 1.261 and 2.896, respectively, our population size estimates ranged from 0 to 40, 000 individuals, with mean population size estimates ranging from 22, 444.24 to 31, 898.36 depending on the threshold scenario (Table 2). When using thinned data, our population size estimates again ranged from 0 to 40, 000 individuals, with mean population size estimates ranging from 24, 190.6 to 30, 216.02 depending on the threshold scenario (Table 2).

Using un-thinned data and inflating the mean density to 4.02 unsurprisingly resulted in much higher population size estimates ranging from 0 to approximately 60, 000 individuals and a mean population size estimate of 51, 821.78 (Table 2).

Table 2. Mean population size estimates and range size estimates, using un-thinned and thinned data, under four different habitat suitability threshold scenarios. Threshold values represent divisions between regions estimated to be suitable or unsuitable, as predicted by model 2: the model built using un-thinned data and model 4: the model built using thinned data. Range size estimates were calculated by multiplying the number of cells remaining after removing values below the specific threshold by the size of each cell (2.27 km²). Values in parentheses represent 1 standard deviation.

Threshold	Mean population size estimate	Range size estimate (km^2)
Un-thinned data:		
Mean Minimum Training Presence: 0.02	31 898.36 (18 831.75)	83 518.67
Post-Hoc 1: 0.12	22 444.24 (13 409.33)	56 093.04
Post-Hoc 2: 0.32	25 615.47 (13 564.95)	45 016.99
Mean 10th Percentile Training Presence: 0.49	28 134.09 (11 997.35)	40 083.97
Mean	27 023.04 (14 450.84)	57 283.72
Thinned data:		
Mean Minimum Training Presence: 0.04	30 216.02 (16 154.93)	83 545.91
Post-Hoc 1: 0.12	24 190.6 (14 076.15)	58 447.18
Post-Hoc 2: 0.32	26 839.97 (13 758)	47 977.26
Mean 10th Percentile Training Presence: 0.5	29 681.26 (12 304.98)	41 700.31
Mean	27 731.96 (17 567)	57 917.66

Validation of SDM Against Fine-Scale Tracking Data

Mean habitat suitability of available habitat (winter and annual) was higher than inside core-use areas and home ranges in Peterborough County in both 2018 and 2019 (Appendix B, Figure A3.4). In the Bruce Peninsula in 2012, we observed the opposite pattern with higher mean habitat suitability estimates inside core-use areas and home ranges than in available habitat (Appendix B, Figure A3.4).

To account for regional and annual differences, we explored the effect of thinning on habitat suitability estimates for each region and each year separately. The results of our t-test indicate that, in Peterborough County, thinning had no significant effect on the mean habitat suitability values inside known turkey home ranges in 2017, 2018, or 2019 ($P > 0.05$; Appendix B, Table A2.7). Thinning also had no significant effect on habitat suitability values inside core use areas, winter available habitat, or annual available habitat in Peterborough County (Appendix B, Table A2.7). Our results were consistent on the Bruce Peninsula, with thinning having no significant effect on the mean habitat suitability inside Wild Turkey home ranges, core use areas, winter available habitat, or annual available habitat in 2012 ($P > 0.05$; Appendix B, Table A2.7).

Validation of Density Estimates Against Survey Data

In Peterborough County, using a combination of three different survey types, we detected 905 turkeys, resulting in an estimated density of 2.14 individuals per km^2 within our 422 km^2 survey area. We detected 53 flocks resulting in a flock density of 0.12 flocks per km^2 with a mean flock size of 16.78 individuals and a standard deviation of 14.1. The aerial survey detected 56.6% of unique flock observations, the community science survey detected 22.6%, and the opportunistic road survey detected 20.8%. On the Bruce Peninsula, the census detected 228 individuals across 10 flocks, resulting in a density estimate of 0.2 turkeys per km^2 within the 1140 km^2 survey area.

Our SDM built with un-thinned data predicted habitat suitability values inside the Peterborough County survey area ranging from 0.44 to 0.63 (mean = 0.69, SD = 0.07; Appendix

B, Figure A3.1). Habitat suitability values derived with un-thinned data were much lower in the Bruce Peninsula survey area, ranging from 0.09 to 0.63 (mean = 0.40, SD = 0.22; Figure 1; Niedzielski and Bowman 2016). Our SDM built with thinned data predicted a slightly higher mean value in the Peterborough County survey area (mean = 0.71, SD = 0.06), but a slightly lower mean value in the Bruce Peninsula survey area (mean = 0.36, SD = 0.22) than the SDM built with un-thinned data.

DISCUSSION

We found that community science in combination with MaxEnt modeling could be used to estimate the size and distribution of populations. We found that our SDMs were not sensitive to data thinning because the mean probability of predicted presence did not differ significantly when MaxEnt models were built using thinned versus un-thinned data, nor did the range in population size estimates. We also found that range size estimates derived from SDMs using thinned data were only slightly larger than range size estimates derived from SDMs using un-thinned data (Table 2). Our study provides some support for using community science in building species distribution models, which might be helpful given that community science has substantial potential as a cost-effective alternative to systematic sampling across large geographic areas (Pimm et al. 2014).

Our population size estimates were unsurprisingly most sensitive to the mean population density value we used to set the low- and high- turkey densities. Population size estimates were much higher when mean turkey density was set to an inflated value of 4.02: the density of wild turkeys reported in Alabama and the highest reported across their range.

It is important to note that in maximum entropy modelling, all observations are weighted equally, although the reported flock size in our dataset ranged from 1 to 200 individuals, with a mean of 14.14. In the winter in Ontario, wild turkeys form large social groups and remain in these groups during daily behaviours such as foraging and roosting (Korschgen 1967, Healy 1992; J. Baici, *personal observation*). Turkeys are rarely observed alone for extended periods of time during this time of year. Most observations (92.1% of the un-thinned dataset) were of more than one individual, with 20.1% of observations reporting more than 20 individuals. Only 409 observations (6.9% of the un-thinned dataset) reported one individual. However, in these instances it is likely that a larger group was simply out-of-sight because wild turkeys often seek cover in woodlots or behind tall vegetation.

Habitat suitability estimates inside core-use areas and home ranges were higher than in available habitat in the Bruce Peninsula region whereas the opposite pattern was observed in Peterborough County (Appendix B, Figure A3.4). Furthermore, our results predicted lower mean habitat suitability in the Bruce Peninsula region than in Peterborough County overall regardless of the modeling scenario. This indicates that there may be evidence of habitat selection occurring within available habitat in the Bruce Peninsula where conditions are harsher and habitat suitability is predicted to be lower overall. In southern populations, where habitat suitability values are higher, wild turkeys may not be selecting habitat based on resource availability because resources are more abundant and homogeneous in this region of the province. Furthermore, the regional population surveys estimate that the wild turkey population density in Peterborough County is over 10 times the population density in the Bruce Peninsula region.

It is important to note that the Bruce Peninsula population survey was conducted in 2012, while the Peterborough County surveys were conducted in 2018. The climate data included in our models was also collected in 2018, while forest and agricultural cover were mapped in 2000, roads were mapped in 2013, and buildings were mapped in 2014. We validated our predicted probabilities against annual home range estimates for 2017, 2018 and 2019 even though the climate variables may not have necessarily reflected conditions in 2017 or 2019. The available turkey habitat in both regions was estimated using all available GPS data for the region, regardless of the year it was collected. In reality, the strength and direction of the effect of each of our predictor variables is likely to differ between years. We have modeled the broad-scale relationships between wild turkey occurrences and each of our predictor variables, but more work should be done to better understand the inter-annual fluctuations of these dynamic relationships. In addition, we echo Bradsworth et al. (2017) in stating the importance of externally validating SDM predictions using fine-scale tracking data because relatively few examples of this exist (Fonderflick et al. 2015, Bradsworth et al. 2017, Coxen et al. 2017).

We exported the results of our SDM following a conditional log-log transformation (clog-log), as this process is derived from the interpretation of MaxEnt as an inhomogeneous poisson process (IPP), which has stronger theoretical justification and allows for more robust interpretations of predicted probabilities (Aarts et al. 2012, Fithian and Hastic 2013, Renner and Warton 2013, Phillips et al. 2017). An assumption of the IPP is that each observation is independent. Our un-thinned dataset violates this assumption because observations that are close together in geographic space may be replicate observations of the same flock or may be observations of unique individuals that belong to the same flock and are thus influenced by one another's behaviour. However, one of the goals of this study was to determine the effect of data cleaning on model outputs. We determined that, in this case, data cleaning to address unknown sampling and spatial bias did not substantially affect model outputs or subsequent analyses.

Our habitat suitability models indicate that, in Ontario during the winter, turkeys appear to be positively associated with buildings and negatively associated with deep snow and cold temperatures. These results remain consistent when modeling with thinned and un-thinned data. It is important to note that because our predicted habitat suitability values were derived using observations collected during the winter, our estimates represent those of winter population size and distribution only. We also note however, that wild turkeys do not make a seasonal migration, and we expect large-scale distribution and abundance patterns to be quite similar among seasons.

A known challenge associated with employing community science data to estimate species distributions is the unknown spatial bias associated with the data. For instance, in our dataset, regions with a high density of buildings (e.g. residential neighbourhoods, city centres, etc.) may represent regions with a higher density of observations as a result of higher sampling effort (Phillips et al. 2009; Ruiz-Gutierrez and Zipkin 2011). However, residential neighbourhoods also contain a high density of bird feeders and supplemental feeding is common at many high-traffic birding and outdoor leisure areas in the province (Jones and Reynolds 2008). Today, wild turkeys are heavily reliant on supplemental food sources, such as bird feeders and agricultural operations, during the winter in their northern range (Vander Haegen et al. 1989, Roberts et al. 1995, Paisley et al. 1996, Nguyen et al. 2003, Kane et al. 2007, Restani et al. 2009). As such, in this case, any underlying spatial bias in the dataset resulting from supplemental food sources and uneven sampling may assist in identifying regions that represent high-quality wild turkey habitat.

Turkeys are considered deciduous forest habitat generalists (Healy 1992, Porter 1992). However, forest cover contributed minimally to our models, with coefficients ranging from 0.3 to 0.89 depending on the modeling scenario. It has been reported that the ideal habitat composition for wild turkeys is an equal ratio of forest (primarily deciduous) to open land (Kurzejeski and Lewis 1985) but thriving turkey populations are found in landscapes with much less forest. For instance, some of the highest harvest densities of turkeys in Ontario are found in Wildlife Management Units (WMU's) with < 25% forest cover (OMNR 2007).

We evaluated model performance using the area under the receiver operator curve (AUC). The AUC provides a single measure of overall model accuracy and is not dependent on a predetermined habitat suitability threshold like other model performance evaluation techniques (Deleo 1993, Fielding and Bell 1997). This statistic is represented as a value between 0.5 and 1. A value of 0.8 indicates that for 80% of the time, the model will correctly predict wild turkey presence and that for 20% of the time, the model will predict wild turkey presence where there were no observations. Researchers suggest that an AUC value > 0.75 is necessary for SDM's to accurately model species distribution (Elith 2000, Nazeri et al. 2012). Our AUC values did not quite meet this threshold but were close, with mean test AUC's ranging from 0.723 to 0.73 depending on the climate variables used and whether observations were represented by thinned or un-thinned data. A known limitation of ROCs is that certain algorithms result in a high potential for commission errors: errors that result in false positives (Peterson et al. 2008). As such, one's interpretation of the AUC is dependent upon a subjective understanding of the cost of these false positive errors versus the cost of false negative errors. One of this study's goals was to identify a potential contemporary range map for this species and another was to estimate the current size of Ontario's wild turkey population. As such, AUC values < 0.75 may have resulted in an overestimation of the geographic range of this species and the overall size of the population as low AUC values can be associated with higher rates of commission (Peterson et al. 2008). However, wild turkeys are a highly adaptable and generalist species, known to thrive in a wide variety of habitats (Healy 1992). Therefore, it is reasonable to assume that geographic regions predicted by the modelling process to contain wild turkeys that do not, may represent regions that could be exploited by wild turkeys in the future as the population continues to expand their range over time.

Erikson et al. (2014) provide a useful overview of the perceived state of wild turkeys across the United States. However admittedly, researchers state that substantial variation exists among states in the methods used to collect population data and estimate ranges. Methods used include brood surveys, harvest information, winter flock counts, and the use of forest cover as a proxy for turkey density, which may be suitable in some areas of their range but would not be a suitable way to estimate turkey density in Ontario, as demonstrated by our findings that forest cover contributed minimally to all models.

There are few published empirical estimates of wild turkey densities in North America. Of the density estimates reported in the literature, there was only moderate agreement with Erikson et al. (2014). For instance, in Ohio, Erikson et al. (2014) estimated a density of 2 turkeys per km^2, whereas Donohoe et al. (1983) estimated 0.7 - 2.0 turkeys per km^2 using various survey techniques across several counties. Erikson et al. (2014) estimated 2.76 Wild Turkeys per km^2 in

Wisconsin, whereas Kubisiak et al. (1997) estimated 0.09 - 1.1 wild turkeys per km^2 using a helicopter survey. Erikson et al. (2014) estimate 2.01 in Mississippi, whereas Lint et al. (1995) estimated 0.34 - 0.87 adult males per km^2 using mark-recapture methods. The survey methods used to detect wild turkeys differ between studies making comparisons difficult. Furthermore, estimates were derived from data collected at different times of year and in different geographic areas. Wild turkey behaviour changes with the seasons (Kurzejeski et al. 1987, Badyaev et al. 1996, Humberg et al. 2009, Niedzielski and Bowman 2014, Gonnerman 2021), meaning that detection ability may differ, as well. We were unable to find an empirical field study that reports a wild turkey density as high as 4 wild turkeys per km^2. Thus, although it is using this mean wild turkey density that we derived total population size estimates similar to those reported in this species most recent management plan (OMNRF 2007), it is unlikely that this density of wild turkeys exists in Ontario given than this region represents their northern range limit and species densities are typically lowest at range limits (Caughley et al. 1988, Hampe and Petit 2005).

This is not the first time that community-collected data has been used to inform the distribution of wild turkeys. For instance, Donohoe et al. (1983:189) incorporated “reliable reports from interested citizens” into estimating the occupied range of wild turkeys in Ohio. Ontario’s provincial wildlife branch relied on personal interviews with area residents to determine the success of the trap-and-transfer efforts in the years directly following reintroductions (Ontario Government 1985, *unpublished manuscript*). The reporting platform eBird (2019) also provides relative abundance and geographic range estimates for many species, including wild turkeys, using a generalized additive modelling approach. In this case, relative abundance is defined as the count of individuals of a given species detected by an expert eBirder on a 1 hour, 1 kilometre traveling checklist at the optimal time of day (Fink et al. 2021). This approach estimates a mean relative wild turkey abundance of 0.04 for the province of Ontario 2021 (Fink, et al. 2022). Among other parameters, models tend to include a subset of variables selected from 60 environmental descriptors and 12 hourly weather variables (Fink et al. 2021). This estimate of relative abundance is useful in understanding broad-scale geographic patterns of distribution but may be less informative of regional predictors of density and abundance. For instance, eBird provides year-round relative abundance estimates, however, wild turkey behaviour differs dramatically with the seasons, and as such, the species detectability differs, as well. Our research investigates wild turkey distribution and population size during the winter months only, as this is when they are easiest to detect in Ontario. Our models also include fewer predictor variables and include only those with evidence from the literature to suggest their usefulness in addressing our research question. For instance, among the 60 environmental descriptors included in eBird's models, there may be numerous variables that are useful for predicting robust estimates of relative abundance across their range but are not necessarily useful for predicting regional Wild Turkey population size.

An informative addition to our community science project could be to request information regarding the presence of supplemental food sources near the turkey flock observations. Anthropogenic food sources (e.g. agricultural operations and residential bird feeders) are difficult to map as they are often located on private property and ephemeral in nature. Collecting information about the presence of these food sources at the time of the flock observation would allow researchers to investigate this important relationship directly, rather than relying on other anthropogenic variables as proxies. Although we did not explicitly request information about nearby supplemental food sources, 177 flock observations (3%) included notes about nearby residential bird feeders. Based on what we know about the frequency of bird feeding in residential areas (Jones and Reynolds 2008), it is likely that the true proportion of turkey sightings associated with bird feeders is much higher.

Our results demonstrate that community science can be a useful tool in estimating the distribution of reintroduced species across a large geographic scale. For generalist species with high adaptability to anthropogenic landscapes, filtering observations based on sampling effort may not be necessary to develop informative SDMs. Because wild turkeys exploit a wide variety of habitat types and food resources, especially anthropogenic food resources like agricultural operations and bird feeders, their detectability among community scientists was high. Furthermore, community scientists most commonly sample in and around anthropogenic landscapes (Tulloch and Szabo, 2012), overlapping with this species niche. For other species, like those that have neutral or avoidant relationships with anthropogenic landscapes, thinning the observations based on sampling effort may be a more important step in elucidating the true relationships between species presence and habitat variables.

Future research could involve applying our approach to species with similar distinctive morphologies, high visibility, and seasonal flocking behaviour, although there are a variety of factors to consider in identifying appropriate species. For instance, wild turkeys do have powered flight, but are primarily observed foraging or walking on the ground, which likely has a positive effect on their detectability. Additionally, virtual species reporting platforms are commonly used in North America, making community science data like wild turkey observations readily available and easily accessible. Most of our observations were derived through eBird (97%) which is significantly underutilized in regions like northern Africa and Russia, for instance (Fink et al. 2021).

We chose not to include topographic parameters, such as topographic index, as there is little evidence from the literature to suggest that they are important in predicting eastern wild turkey habitat suitability. There is also little variation in topography in most of the wild turkey's range in Ontario, suggesting that topographic variables are unlikely to inform wild turkey ranges within our study area. However, topography can be a limiting factor for wild turkey subspecies in habitats with higher topographic variability (Bakner et al. 2022).

Management Implications

This study provides empirical estimates of the size and distribution of Ontario's reintroduced wild turkey population. Prior to this, the most recent estimate of Ontario's wild turkey population size, published in 2007, was 70 000 individuals (OMNR 2007). However, this estimate was based on limited information and is not informative of how the population may continue to grow and change over time.

We present our findings in a provincial context. However, regional densities could also be estimated using this methodology. For instance, in Ontario, regions are designated as Wildlife Management Units (WMU's) and harvest rates are measured and managed within each unit. Estimating turkey densities inside each WMU based on SDM-informed predicted probabilities of presence would allow for finer-scale management (Francis et al. 2009).

In Ontario, spring turkey hunting is assumed to remove approximately 30% of the male population (OMNR 2007). Since 2008, there has generally been ~ 10,000 to 20,000 adult males harvested per year. Researchers in Texas found that the brood sex ratio (BSR) of wild turkey populations at two different study sites both skewed male with approximately 56% of sampled eggs being biologically male (Collier et al. 2007). If we assume a similar BSR in Ontario, and that approximately 10,000 to 20 000 individuals represent 30% of the male population, we can infer that Ontario should have a population of ~ 65,000 to 130 000 individuals. None of our modelling scenarios produced population size estimates this large even when estimating population size using an inflated mean density of 4.02 wild turkeys per km^2 (Table 3).

Survival rates are likely to differ between the sexes which may cause the sex ratio in the larger population to differ from the BSR. There is little published information regarding wild turkey survival rates at their northern range limit. One study estimates the mean annual survival rate of hens in Sudbury, Ontario (46° 10′ 0.0012" N, 80° 25′ 00.0012″ W)) to be 0.28 (Nguyen et al. 2003), while another conducted in the municipality of Northern Bruce Peninsula (45° 0′ 0″ N, 81° 19′ 0.0012″ W) estimated the mean annual survival rate of hens to be 0.37 (Niedzielski and Bowman 2014). However, neither study reported the survival rates of males in the same population. This is also true of survival rates estimated in other regions of the wild turkey's range. Researchers tend to estimate survival of one sex or the other, making comparisons across the sexes challenging (e.g. Keegan and Crawford 1999, Holdstock et al. 2006, Restani et al. 2009). One study conducted on both sexes in northern Indiana did find that there were substantial differences in survival between the two. Researchers estimated mean male and female annual survival rates to be 0.257 and 0.777, respectively (Humberg et al. 2009).

When using a mean density value representative of the average density across the wild turkey's USA range, we estimated an average of less than 30 000 individuals in Ontario (Table 3). Only when our mean was inflated to match the highest recorded density across their range did we estimate population sizes greater than 40 000 (Table 3). These results indicate that some regions in Ontario may exhibit a higher density of wild turkeys than expected based on densities reported for other geographic regions or previously for Ontario. Turkey populations can remain stable when up to 35% of the male population is harvested during the spring season (Vangilder 1997). However, our findings indicate that more than 30% of the male population may be being harvested during the spring season, particularly during the last several harvest years. This is a notable finding given the historical context of wild turkeys in Ontario. We recommend that the distribution, abundance, and harvest pressure of wild turkeys in Ontario continue to be monitored to gain a better understanding of their anthropogenic-centred habitat selection patterns and long-term population trends.

Table 3. Mean population size estimates and range size estimates under 4 different habitat suitability threshold scenarios and with the mean density inflated to match the density of wild turkeys (Meleagris gallopavo) reported in Alabama: 4.02 turkeys/km², the highest density reported for this species in the United States (Erikson et al. 2014). Threshold values represent divisions between regions estimated to be suitable or unsuitable, as predicted by model 2: the model built using un-thinned data. Range size estimates were calculated by multiplying the number of cells remaining after removing values below the specific threshold by the size of each cell (2.27 km²). Values in parentheses represent 1 standard deviation. Note: PH1 = post-hoc 1 and PH2 = post-hoc 2.

Threshold	Mean population size estimate	Range size estimate (km^2)
Mean Minimum Training Presence: 0.02 +/- 0.001	60 399.71 (33 077.04)	83 518.67
PH1: 0.12	43 557.93 (23 759.63)	56 093.04
PH2: 0.32	49 254.36 (23 491.05)	45 016.99
Mean 10th Percentile Training Presence: 0.49 +/- 0.006	54 075.14 (20 396.75)	40 083.97
Mean	51 821.78 (25 181.12)	57 283.73

Ongoing research efforts should continue to work towards a comprehensive understanding of wild turkey population density and distribution across the province while considering any habitat changes that are likely to occur because of climate change. Special attention should be paid to regions experiencing any changes in harvest pressure. wild turkey populations are most sensitive to hen survival and recruitment (Roberts and Porter 1995). However, established populations can fluctuate in size annually by as much as 50% of the long-term mean (Mosby 1967). So, monitoring changes over time is important as population densities are unlikely to remain constant.

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Chapter 3: How morphology and anthropogenic food sources inform wild turkey (*Meleagris gallopavo*) utilization distributions

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Contributions: Baici and Bowman conceived and designed the study; Baici analyzed the data and wrote the manuscript. Bowman critically reviewed the manuscript.

Abstract

Many animals exploit anthropogenic food sources, like agricultural crops and bird feeders, and it is likely that the distribution of these resources influences the size and composition of species' home ranges. Wild turkeys (*Meleagris gallopavo*) use anthropogenic foods across their range, particularly during the winter. As such, access to anthropogenic food sources may shape their seasonal movement patterns. Furthermore, the dominance rank of wild turkeys is an often cited but understudied aspect of turkey behavioural ecology that may be closely tied to the use of food sources. We estimated home range size and core use areas for 65 wild turkeys in south-central Ontario from 2017 to 2019. We used satellite and aerial imagery to evaluate the composition of anthropogenic food sources within turkey utilization distributions (UDs) and found that, for females during the summer, crop was positively associated with home range size, while livestock exhibited a strong negative relationship. For males, the opposite relationship was detected. Pasture was also strongly negatively associated with male home range size. We attempted to identify suitable proxies for dominance rank but found no relationship between social status and any morphological measurements, including mass and length, although we did identify a robust social structure for our study population. For males during the summer, measures of morphology were important predictors of utilization distribution, but the pattern differed depending on space use. Understanding the

anthropogenic factors that affect wild turkey home range size and composition is important for making informed management decisions for the species.

Keywords
Supplemental food, hierarchy, utilization distribution, home range, wild turkey

INTRODUCTION

Synanthropic species are those that can cohabitate with humans and thus exhibit commensalism or mutualism that is typically mediated by humans (Tomiałojć 1970, Johnston 2001). One way that synanthropic species benefit from anthropogenic activity is through access to supplemental food sources. For instance, many synanthropes (Johnston 2001) experience increased abundance as a result of exurban development (Hansen et al. 2005) and the increased availability of predictable human food sources (Kristan and Boarman 2003, Withey and Marzluff 2008, Webb et al. 2011, Newsome et al. 2014). For instance, ravens (*Corvus corax*) have been observed near human settlements, such as anthropogenic water-bodies and landfills, in the Mojave desert of California USA (Kristan and Boarman 2003). The abundance of crows (*Corvus brachyrhynchos*) is also strongly correlated with land cover providing anthropogenic resources such as residential neighbourhoods, parks, and landscaped yards (Withey and Marzluff 2008).

Wild turkeys (*Meleagris gallopavo*) are a synanthropic species that have experienced historical range contractions associated with regional extirpations. The combination of habitat loss and unregulated hunting resulted in the disappearance of turkeys from numerous regions of North America by the beginning of the 1900s (Blakey 1941, Davis 1949). The successful reintroduction of the wild turkey across its range can be largely attributed to harvest regulations, increased habitat availability following the abandonment of farmland in the 1920s and 30s, and well-coordinated trap and transfer initiatives. Starting in the 1950s and extending into the 1980s and 90s, wild turkeys were captured and relocated across North America (Mosby 1959, 1973, 1975, Bailey 1980, Kennamer 1986, Bellamy 2001). Present-day turkey populations are distributed across all U.S. states, six out of thirteen Canadian provinces or territories, and northern Mexico (Kennamer and Kennamer 1996, Tapley et al 2001, 2007, 2011). Recent observations indicate that wild turkeys can be found as far north as Neepawa, Manitoba (Koes 2019).

Winter is the season that primarily constrains the abundance of wild turkeys in the northern part of their range. (Niedzielski and Bowman 2015, 2016) and their northern range edge can fluctuate with winter severity (Leopold 1931, Schorger 1942, Mosby 1959). Both cold (MacDonald et al. 2016, Oberlag et al. 1990) and snow depth (Nguyen et al. 2003, Kane et al. 2007, Gonnerman et al. 2022, Baici and Bowman 2023) are important predictors of wild turkey movement, presence, and survival, particularly at northern latitudes. Turkeys do have several adaptations that allow them to withstand extreme temperatures (Buchholz 1996, Haroldson et al. 1998, Coup and Pekins 1999). The snood aids in heat dissipation, while dark and downy plumage allows turkeys to maintain their internal body temperature during cold temperatures (Buchholz 1996). However, during periods of heavy snow, wild turkeys experience restricted mobility, resulting in reduced foraging activity and other energy-conserving behaviours (Porter 1977, Porter et al. 1980, Roberts et al. 1995, Wright et al. 1996, Nguyen et al. 2003, Kane et al. 2007).

Like many synanthropic species, wild turkeys exploit anthropogenic foods, and can even become heavily reliant on them, particularly during the winter months when their natural foods are less available (Vander Haegen et al. 1989, Roberts et al. 1995, Paisley et al. 1996, Nguyen et al. 2003, Kane et al. 2007, Restani et al. 2009). For instance, the availability of supplemental corn increased winter survival (Porter et al. 1980, Restani et al. 2009) and spring body condition (Porter et al. 1980) for female turkeys during long periods of deep snow in Minnesota. Transplanted hens in central Minnesota also had higher winter survival in study areas with supplemental food sources compared to those without (Kane et al. 2007), and there have been several instances of wild turkeys succumbing to starvation when only a few kilometers from a supplemental food source (Kane 2003). As such, food is demonstrably an important limiting resource during harsh winter conditions. Some researchers argue that the persistence and stability of northern wild turkey populations may even depend on the availability of anthropogenic foods (Pekins 2007). Wild turkeys experience reduced movement during the winter due to deep snow (Austin and DeGraff 1975, Wunz and Hayden 1975, Porter 1977, Porter 1980, Roberts et al. 1995, Wright et al. 1996, Nguyen et al. 2003, Kane et al. 2007, Gonnerman et al. 2022), and as a result may constrict their seasonal ranges around important winter resources, such as anthropogenic foods.

Wild turkeys gather and move through the landscape in large groups during the winter months (Healy 1992). Group-living offers several potential benefits for wild turkeys, including increased foraging efficiency (Pitcher et al. 1982, Brown 1988, Krause and Ruxton 2002) and reduced predation risk (Bill and Hernkind 1976, Stacey 1986, Elgar 1989, Pays et al. 2013). Hierarchical animal societies have been described in insects (Choe 1994), fishes (Polačik and

Reichard, 2009), reptiles (Bush et al. 2016), birds (Devost et al. 2016) and mammals (Majolo et al. 2012), including humans (von Rueden et al. 2008). Interspecific dominance hierarchies among bird species have also been described (Miller et al. 2017). For social foragers like the wild turkey, dominance hierarchies within groups can promote the efficient use of resources by suppressing costly intraspecific aggression (Rowell 1974). Hierarchies also reduce the energy and time needed for social interactions, which can be spent performing other important activities such as foraging or predator avoidance (Dewsbury 1982, Drews 1993).

It is possible that an individual's position within a dominance hierarchy may affect their foraging opportunities. For instance, Whiteman and Côté (2004) found that, for sponge dwelling cleaning gobies (*Elacatinus prochilos*), larger, more competitively dominant individuals were able to monopolize areas with the highest food density and achieve the highest foraging rates. However, researchers found that for California condors (*Gymnogyps californianus*), rank did not predict the amount of time that a bird spent feeding at a carcass or the frequency that a bird was interrupted while feeding (Sheppard et al. 2013). So, foraging opportunities may not always be linked to hierarchical position. If dominance rank informs access to supplemental food resources by wild turkeys, then dominant birds might have access to the highest quality food resources and need to move around the landscape less to meet their daily nutritional demands (Kurzejeski and Lewis 1990). Relatively large home ranges are often attributed to poor quality foraging habitat (*e.g.* McNab 1963, Swanson et al. 1994, Godwin et al. 1996, Stewart et al. 1998, Lovari, et al. 2013, Skinner et al. 2022) and it has been suggested that food availability can determine wild turkey home-range size (Kurzejeski and Lewis 1990). Thogmartin (2001) found that, for nesting hens in Arkansas, heavier females occupied smaller home ranges than lighter females. Increased territory size in response to reduced availability of high-quality foraging habitat has also been reported for capercaillies (*Tetrao urogallus*), another large-bodied galliform (Wegge and Rolstad, 1986). We were interested in further evaluating this idea.

Inferring dominance hierarchies can be challenging, as they are often studied in animal groups by examining dyadic, agonistic interactions where there is a clear winner (dominant) and a loser (subordinate) (Drews 1993). Numerous statistical analyses may be employed to establish social hierarchies among animal groups (Sánchez-Tójar et al. 2017). However, all methods require recording agonistic dyadic interactions among individuals as input data. For wild turkeys, hierarchies been previously assigned qualitatively based on age; young individuals are subordinate to older individuals, and sex; females are subordinate to males (Healy 1992, Watts 1969, Watts and Stokes 1971, Schorger 1966). Based on qualitative observations, Watts (1969) found that the dominant-subordinate relationship between any two individuals was initially determined by a fight and later maintained by lesser forms of aggression. These relationships appear stable through all phases of flock activity, including feeding, roosting, and breeding. Dominance status changed with physical maturity and age but not with location (Watts 1969).

Numerous studies have attempted to quantitatively investigate the dominance hierarchy of wild turkeys (Watts 1969, Watts and Stokes 1971, Badyaev 1994, 1996, Badyaev et al. 1998, 1997, Krakauer 2005). However, those studies that have investigated this relationship have done so only using male-male interactions. Individuals within each age class can differ in body mass at breeding (Porter et al. 1983), allowing body mass to be employed as a proxy for social status in several studies (Badyaev 1994, 1996, Badyaev et al. 1998). Buchholz (1997) found that male turkeys involved in dyadic interactions differed by several kilograms in some cases but when measured alone, mass surprisingly did not have a significant effect on the outcome of the interaction (Buchholz 1997). They found that relaxed snood length was the most important predictor of dominance, with dominant males having longer relaxed snoods than their subordinate partners (Buchholz 1997) and that spur length was somewhat correlated with dominance, in that males with similar spur lengths fought one another sooner after meeting than dissimilar males (Buchholz 1997). An experimental field study in Sweden demonstrated that variation in the length of spurs is not a good predictor of dominance status in male ring-necked pheasants (*Phasianus colchicus*), a closely related species (von Schantz et al. 1989). However, studies of captive ring-necked pheasants found opposite results (Mateos and Carranza 1996).

Our objective was to test the hypotheses that wild turkeys, a synanthropic species, rely on supplemental foods during the winter months and that the use of supplemental food, along with dominance rank, can affect the size and composition of space use, as revealed by individual utilization distributions. Utilization distributions (UDs) represent the probability distribution defining the animal's use of space (Van Winkle 1995).

Animals generally use space disproportionately within the boundaries of their home range (Powell et al. 1997). Within home ranges, space-use is typically heterogeneous. Within core use areas, space-use is concentrated and can be informative for understanding the ecological factors determining use (Neale and Sacks 2001, Prange and Gehrt 2007, Chamberlin et al. 2003, Thompson et al. 2007). As such, to determine if we could detect a relationship between space-use and the exploitation of supplemental foods, we generated both 95% kernels to represent home ranges, and 50% kernels to represent core use areas.

We explored the relationship between seasonal utilization distribution size and several variables representing anthropogenic supplemental foods as well as individual turkey morphology as a proxy for rank. If wild turkeys rely on supplemental food sources in our study area during the winter months, we expect variables representing supplemental food to be important in predicting utilization distribution size. If wild turkeys are reliant on supplemental food sources during the winter months, but experiencing restricted mobility (Austin and DeGraff 1975, Wunz and Hayden 1975, Porter 1977, Porter 1980, Roberts et al. 1995, Wright et al. 1996, Nguyen et al. 2003, Kane et al. 2007, Gonnerman et al. 2022), we might expect to see a significant negative relationship between the size of winter utilization distribution and the proportion of the UD containing supplemental foods.

To determine whether we could detect a relationship between individual morphology and seasonal range sizes, we included morphological variables in our models as predictors of UD size. We predicted that larger, presumably more dominant, individuals would have more access to supplemental food resources than smaller, presumably less dominant, individuals and that we would see evidence of this in the size and composition of their utilization distributions, with larger individuals having smaller distributions.

METHODS

Study Area

Our research occurred in Peterborough County, situated in south-central Ontario. Ontario is a vast (Figure 1), a total land area of 908 699 km^2 and containing Canada's most populous city, Toronto (Statistics Canada 2017). The study area is characterized by mixed forests and both plant, and animal agriculture. In the southern part of the county there is a mix of agriculture, urban areas, and lakefront properties. The primary economic sectors are agriculture and tourism, with important crops including soybean, corn, and winter wheat. Pasture and hay fields also play an important role in the landscape, supporting animal agriculture (Statistics Canada 2023). Wild turkey density in this region is approximately 2.14 individuals per km^2 (Baici and Bowman 2023).

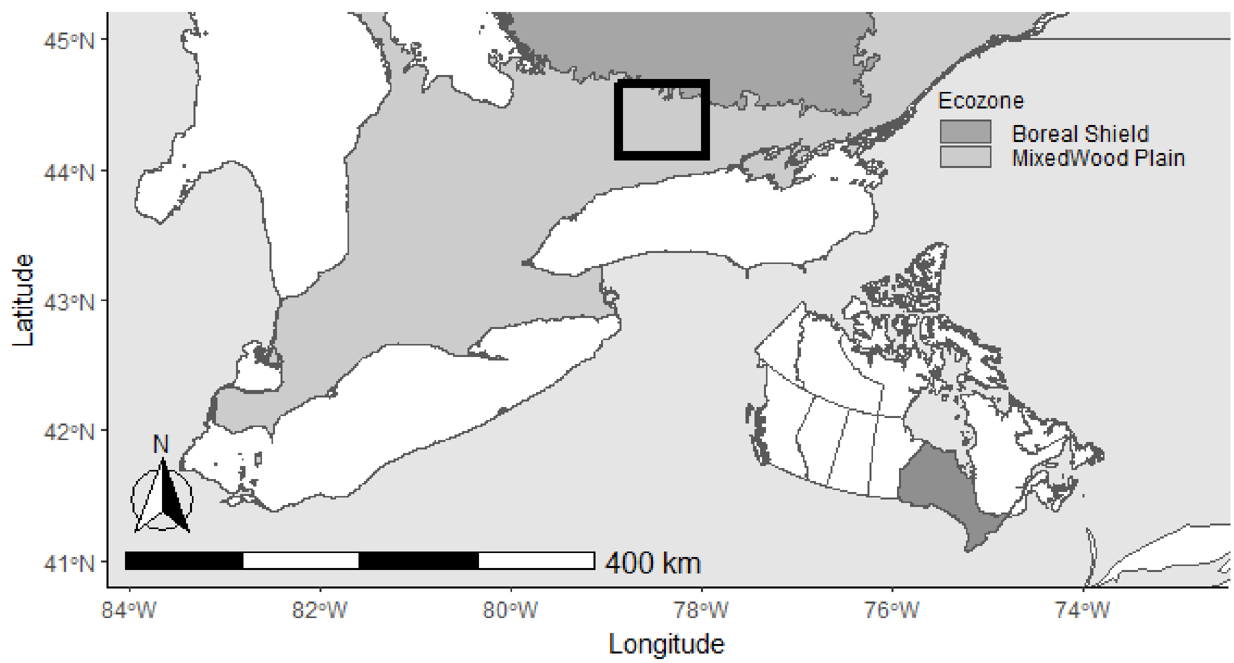

Figure 1. The area over which this study was conducted in Ontario, Canada. The bounding box represents Peterborough County (44° 30′ 0″ N, 78° 10′ 0″ W).

Capture and handling

During winter (December to March) 2017 through 2019, wild turkeys were trapped at locations baited with dried corn using a rocket net (Grubb 1988) following capture methods described by Niedzielski and Bowman (2014, 2016). Processing involved weighing, recording morphological measurements, sexing, collecting a blood sample for DNA extraction, and the attaching of GPS-transmitters (Model PP-VHF-3600L, Lotek, Newmarket, Canada). For each individual, we recorded tail length (mm), weight (kg), left tarsus length (mm), and left-wing chord (mm). For individuals with beards and spurs (males and bearded hens), we also recorded the length of both structures. Sex and age categories (juvenile male [“jake”], adult male [“tom”], female [“hen”]) were assigned based on wing chord, tail length, and tarsus length (Pyle 2008). For males, we also assigned age class based on beard and spur length. We considered hatch-year (HY) and second year (SY) males to be juveniles, and after second-year (ASY) males to be adults. Very few female individuals were classified as juveniles based on wing chord, tail length, or tarsus length (1 – 3 individuals depending on the measurement) and there was disagreement in age classification between measurements. As such, we included female individuals as one category within our dominance rank analysis, rather than dividing this category into juvenile and adult individuals.

Estimating utilization distributions

We estimated utilization distributions using GPS location estimates recorded by GPS-transmitters. The recording frequency of GPS locations varied, depending on the capture year and location, ranging from hourly to once every 4.25 hours, but remaining constant for each individual. Individuals were not included in the analysis if they were tracked for less than 33% of the season. To estimate the accuracy of location estimates derived from GPS-fixes, we placed 6 GPS-transmitters in known locations within our study area. For each GPS accuracy test, after 21 days, or approximately 218.83 (45 – 560) fixes per transmitter, we downloaded the location estimates and calculated the mean distance between the recorded GPS fixes and the known transmitter location. Again, we placed transmitters in a variety of different habitat types within our study area, with differing levels of canopy cover and understory vegetation.

We investigated net displacement over time to identify portions of the year when wild turkeys were exhibiting movement associated with static seasonal home ranges ("sedentarism") rather than seasonal dispersal or exploratory behaviours (Börger and Fryxell 2013, Bastille-Rousseau et al. 2016, Street et al. 2017). Distinct patterns in net-squared displacement (NSD) time-series are theoretically expected from specific movement strategies (Börger and Fryxell 2013, Bastille-Rousseau et al. 2016, Street et al. 2017). We conducted a visual assessment of time-series plots and identified two portions of the year when most individuals exhibited sedentarism: during the fall/winter from November 1 to March 15, and during the spring/summer from April 1 to July 31 ("winter" and "summer", respectively).

We trimmed the location data to retain fixes within each sedentary period and estimated home range and core use areas by calculating 95% and 50% Kernel Density Estimates, respectively. We generated isopleths and calculated the area of 95% and 50% isopleths to represent home range and core use area size in our models.

When generating isopleths, bandwidths were selected using an ad-hoc method in which we first generated isopleths using the reference bandwidth (Worton 1995, Seaman et al. 1999). 121.98. The use of the reference bandwidth results in over-smoothing of home ranges for most species (Worton 1995, Seaman et al. 1999). As such, we then with bandwidths ranging from 10-90% of the reference bandwidth, decreasing incrementally by 10%. The appropriate bandwidth and associated utilization distribution for each bird was selected by retaining the smallest bandwidth value that resulted in the same number of 95% isopleths as the reference bandwidth. For core utilization distributions, 50% isopleths, we retained the utilization distribution area estimates associated with the same bandwidths as was selected for the 95% isopleths in the previous step. We calculated all UDs in Program R (R version 3.2.5, www.r-project.org) and the adehabitatHR (Calenge 2006) and sp (Pebesma and Bivand 2005; Bivand et al. 2013) packages.

Principal Component Analyses on Morphology

To explore the relationships between morphological variables and to reduce them to proxies representing dominance rank in our utilization distribution models, we conducted Principal Components Analyses (PCAs), a method used to reduce the dimensionality of a dataset while preserving as much statistical information as possible (Jolliffe and Cadima 2016).

Wild turkeys are sexually dimorphic. As such, we conducted PCAs with morphological data from male and female wild turkeys independently. For females, as variables in the PCA, we included tail length (mm), wing length (mm), tarsus length (mm), and weight (kg). For males, we included two additional sex-specific characteristics: beard length (mm) and left spur length (mm). All variables were scaled in the PCA.

We investigated the proportion of variance explained by each principal component (PC) and extracted the PC scores for the first and second principal component. The scores were then included as predictor variables in our utilization distribution models as a proxy for each individual's rank.

Utilization Distribution Modelling

We modelled the relationship between wild turkey seasonal utilizations distributions (summer and winter home ranges and core use areas) and a combination of morphological and landscape features by fitting eight linear mixed-effects models (Bates et al. 2015) to our data. All models were fit under a Gaussian distribution using the "lme4" package for R Studio (Bates et al. 2015). Our response variables, seasonal wild turkey utilization distributions, were highly right skewed. As such, prior to modelling, we log-transformed each response variable so they more closely approximated a normal distribution.

In our models, in addition to PC 1 and 2 as proxies for rank, we also included as predictors of UD size, road density, and the proportion of crop, pasture, and livestock within each 50% and 95% utilization distribution.

Livestock locations were recorded and mapped using comprehensive road surveys and aerial imagery. We drove each road inside our study area (Figure 1) and recorded location information for each livestock operation visible from public access roads. We then used a combination of satellite imagery (LIO 2019) and qualitative observations recorded during ground-truthing to construct spatial polygons representing the approximate size and distribution of all livestock operations within the study area.

Using ArcMap (ESRI 2011), we created our crop and pasture variables by extracting these land cover classes from the OMNR Provincial Land Cover Dataset (2000). Cells (25 m x 25 m) representing cropland (row crops, hay, or open soil in areas of agricultural land use) were combined and exported as a new data layer to create our crop variable, and cells representing pasture and abandoned field (open grasslands with sparse shrubs in agricultural areas, including orchards) were combined and exported to create our pasture variable. Prior to modelling, we estimated the proportional overlap between each UD and our livestock, crop, and pasture polygon layers. Road data was derived from the Ontario Ministry of Natural Resources and Forestry (OMNRF) road network database (2010, 2013). We estimated road density by calculating the total road length, in kilometres, for each UD.

In temperate regions, wild turkey behaviour differs substantially with temperature (Haroldson et al. 1998, Coup and Pekins 1999) and environmental conditions associated with seasonality (Badyaev et al. 1998, Humberg et al. 2009, Kurzejeski et al. 1987). We accounted for this by conducting separate models for each season (winter and summer).

In addition to the sexual dimorphism displayed by this species, there are meaningful behavioural differences between the biological sexes. For instance, females are more vulnerable to starvation than males during inclement weather (Gray and Prince 1988) and may need to travel further to obtain necessary resources (Badyaev 1996). Food availability and distribution during the fall and winter seasons affect females more than males (Badyaev 1996). As such, we modelled the relationship between utilization distribution size and our predictor variables for each sex separately and predicted that females will have larger utilization distributions than males, particularly during the winter seasons.

Estimating Elo-Rank

To investigate social interactions among tagged turkeys, we set camera trap stations at three sites inside known turkey home ranges in the spring following our third year of trapping, Camera trap stations were set by placing a small pile of whole kernel corn in front of motion-triggered, camera traps (Model UltraFire XP9, Reconyx, Wisconsin USA). Camera trap stations were set in areas frequently used by tagged and colour-banded wild turkeys. Cameras were set to record one-minute video clips when motion-triggered. Bait piles were designed to mimic anthropogenic supplemental food sources that wild turkeys are known to exploit, such as bird feeders (Niedzielski and Bowman 2014) and cattle pastures (Vander Haegen et al. 1989).

We then reviewed each one-minute clip and recorded each instance of an agonistic, dyadic interaction. For each interaction, we assigned a winner, the dominant individual, and a loser, the subordinate individual (Shimmura et al. 2015). Examples of agonistic behaviours that we observed include pecking, darting, kicking, and flinching. In most cases, the instigating individual was assigned as the winner and the evading individual was assigned as the loser. For instance, an individual who pecks another individual who subsequently flees from the bait pile would be considered the winner, while the individual fleeing the site would be considered the loser of the interaction (Buchholz 1997). For every individual involved in an interaction, we recorded sex and age class. In cases where individuals were tagged and/or banded, we also recorded colour band pattern and tag type (GPS or VHF).

To estimate the dominance hierarchy based on the outcomes of dyadic interactions at bait piles, we calculated the randomized Elo-rank using the aniDom package for R Studio (Farine and Sánchez-Tójar 2019). The randomized Elo-rank is a variation of the original Elo-rating (Elo 1978): a method commonly applied to analyze sequential animal behaviours (Franz et al 2015; Snyder-Mackler et al. 2016; Strandburg-Peshkin et al. 2015). The randomized Elo-rank does not analyze interactions sequentially, but instead randomizes them, providing more robust estimates and a statistical measure of uncertainty (Sánchez-Tójar et al. 2017).

We estimated the relative Elo-rank for four sex and age categories: juvenile female, juvenile male, adult female, adult male. We also estimated the Elo-rank for 19 tagged individuals and investigated hierarchy shape for both groups. We estimated the robustness and uncertainty of our Elo-rank estimates using two different tests of repeatability (Farine and Sánchez-Tójar 2019). To gain insight about the accuracy of our morphological proxies of rank, we compared Elo-rank estimates to the PCs derived from our PCA.

To determine whether there is a relationship between the frequency of interactions occurring and the group's size or composition at the bait pile, we recorded the number of individuals in the frame at the 30-second timestamp for each clip and the sex and age structure of this sub-sample. We then filtered the dataset to include only video clips that contained a minimum of one individual, and a minimum of one interaction. We conducted a correlation test between the number of interactions that occurred within the one-minute clip and the number of individuals at the bait pile in the frame at the 30-second timestamp.

Finally, we evaluated the suitability of using morphological variables, such as mass, as a proxy for dominance rank in this species using a smaller subset of data derived from tagged and ranked individuals in our study population.

RESULTS

Utilization Distributions

We estimated the utilization distribution core use areas, represented as 50% and 95% kernel densities respectively, for all GPS-tagged individuals in our study population (Table 1). The sample size varied by year and season as individuals were depredated, lost, or captured. The mean home range size across all individuals in the summer season was 1.86km^2 (n = 42) and in the winter was 5.32km^2 (n = 65). The mean core use area size across all individuals in the summer was 0.12km^2 (n = 40) and in the winter was 0.28km^2 (n = 46). We also investigated differences in home range and core use area sizes between sexes (Table 1).

Table 1. Summary of home range and core use area sizes for each year, season, and sex. Value in parentheses represents one standard deviation.

		Winter km^2		Summer km^2	
Space Use	Year	Male	Female	Male	Female
Home Range (95% kernels)	2017	8.80km^2 (16.71) (n = 6)	1.56km^2 (0.95) (n = 3)	2.45km^2 (1.93) (n = 9)	2.26km^2 (1.20) (n = 3)
	2018	1.28km^2 (1.16) (n = 10)	2.41km^2 (1.74) (n = 10)	1.25km^2 (0.67) (n = 10)	1.60km^2 (0.95) (n = 8)
	2019		0.36km^2 (0.27) (n = 4)		3.80km^2 (1.82) (n = 3)
Core Use (50% kernels)	2017	0.03km^2 (0.03) (n = 6)	0.12km^2 (0.10) (n = 3)	0.10km^2 (0.15) (n = 9)	0.17km^2 (0.14) (n = 3)
	2018	0.08km^2 (0.10) (n = 9)	0.11(km^2) (0.15) (n = 9)	0.08km^2 (0.10) (n = 9)	0.15km^2 (0.23) (n = 7)
	2019		0.01km^2 (0.005) (n = 4)		0.32km^2 (0.43) (n = 3)

Principal Component Analyses on Morphology

Males were 1.5 times heavier ($t(24) = 8.62$, $p = 8.521e\text{-}09$) and had significantly longer tails ($t(5) = 2.6$, $p = 0.0441$) than females. Mean male and female weights were 6.7kg (+/- 1.32kg, n = 36) and 4.27kg (+/- 0.56kg, n = 37), respectively. Mean male and female tail lengths were 402.72mm (+/- 33.1mm, n = 36) and 360.94mm (+/- 29.7mm, n = 37), respectively. Weights ranged from 2.6 kg for the smallest female to 9.4 kg for the largest male and tail lengths ranged from 282 mm for the shortest female to 551 mm for the longest male.

The PCA on female morphological data indicated that 52.7% of the variance was explained by the first principal component (PC1) (± 1.45), followed by 27.4% explained by PC2 (± 1.04) (Figure 2A).

For females, principal component 1 was primarily defined by weight (loading = 0.62) and tail length (0.61) but is also partially defined by tarsus length (0.39) and tail length (0.27). Principal component 2 was more strongly defined by tail length (0.78) and tarsus length (-0.62) and appeared to be defined very little by wing length (0.027) and weight (0.021) (Figure 2B).

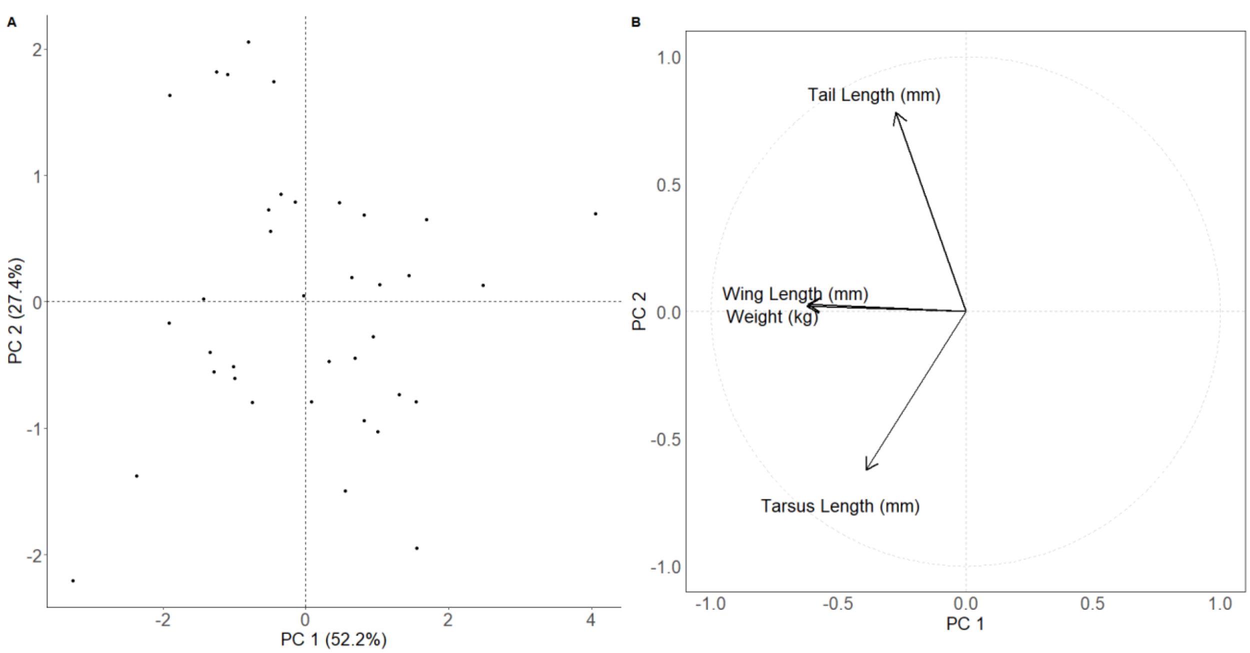

Figure 2. Results from the principal component analysis on female morphological data (n = 37). A. Biplot depicting PC1 and PC2 scaled scores, B. Scaled variable contributions to PC1 and PC2.

The results of the PCA on male morphological data indicated that 52.1% of the variance was explained by the first principal component (PC1) (± 1.76), and 14.5% of the variance was explained by PC2 (± 0.93) (Figure 3A).

For males, principal component 1 was primarily defined by weight (0.47), beard length (0.47), spur length (0.46), and wing length (0.44), but is also partially defined by tarsus length (0.31). Principal component 2 was almost entirely defined by tail length (0.96) and wing length (-0.21) (Figure 3B).

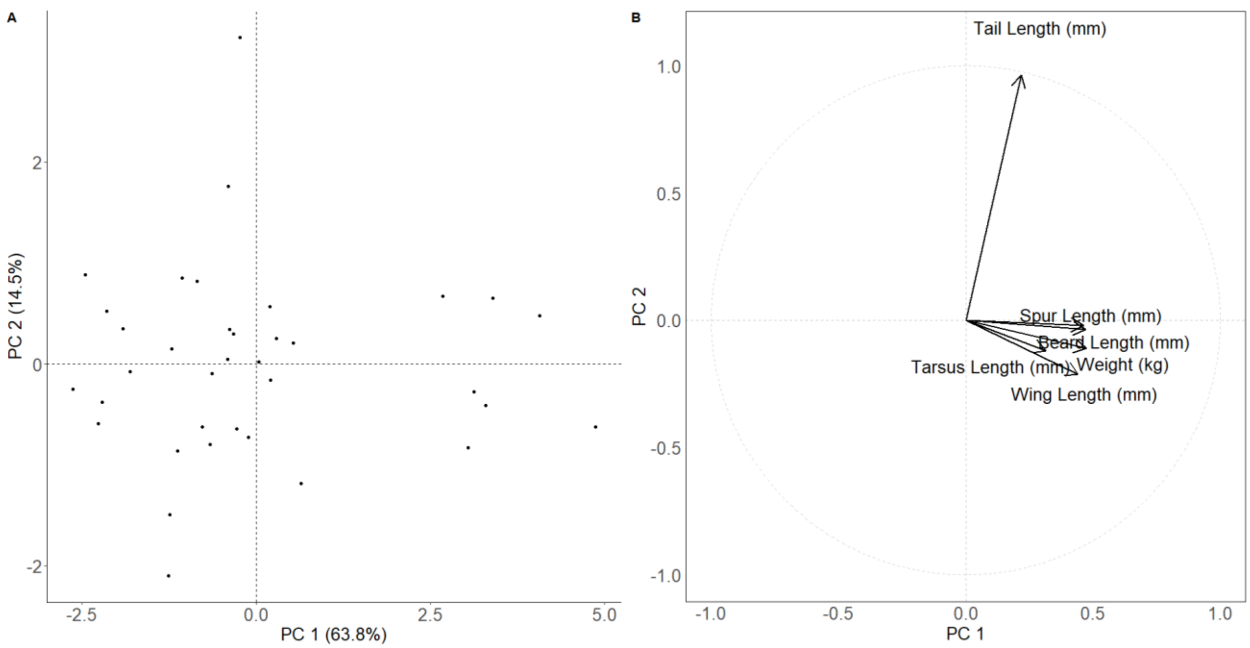

Figure 3. Results from the principal component analysis on male morphological data (n = 36). A. Biplot depicting PC1 and PC2 scaled scores, B. Scaled variable contributions to PC1 and PC2.

Modelling Utilization Distributions

When modelling summer home ranges, female UD size was negatively associated with livestock but positively associated with crop land. For males, the inverse was true. For males during the summer, home range size was also negatively associated with pasture, PC1 and PC2 (Table 2).

When modelling winter core use areas, crop land and pasture were significant predictors of UD size for females (Table 3). When modelling summer core use area size, PC2 was positively associated with UD size for males (Table 3). We did not identify any significant predictors of male or female winter home range size, male winter core use area size, or female summer core use area size.

Table 2. Coefficient estimates (± SE) for all generalized linear mixed models relating home range size to supplemental food source availability. Home range size was estimated as 95% kernel density estimates (km^2). Pasture, crop, and livestock were estimated as a proportion of the home range. PC1 and PC2 are principal components 1 and 2 derived from morphological data. All models were fitted under a Gaussian distribution, include year (2017 through 2020) and bird ID as a random effect (random effect is estimated as variance (± SD)). We did not detect any correlations between predictor variables (Appendix C: Supplemental Table 2). Significant variables are bolded.

	Winter		Summer	
Variable	Male (n = 17)	Female (n = 17)	Male (n = 18)	Female (n = 21)
(Intercept)	-1.33 (± 2.26) *p = 0.55*	1.52 (± 2.25) *p = 0.49*	**5.77 (± 0.95)** ***p < 0.0001***	-1.89 (± 1.38) *p = 0.17*
Pasture	-3.59 (± 4.75) *p = 0.44*	-4.81 (± 4.61) *p = 0.29*	**-12.59 (± 2.39)** ***p < 0.0001***	0.18 (± 2.2) *p = 0.93*
Crop	5.52 (± 4.61) *p = 0.23*	3.18 (± 3.09) *p = 0.3*	-3.87 (± 1.20) ***p < 0.0001***	**7.15 (± 2.16)** ***p < 0.0001***
Livestock	40.06 (± 53.03) *p = 0.44*	9.26 (± 11.82) *p = 0.43*	8.06 (± 9.53) ***p < 0.0001***	**-24.32 (± 5.19)** ***p < 0.0001***
PC1	-0.09 (± 0.21) *p = 0.67*	-0.37 (± 0.23) *p = 0.1*	-0.03 (± 0.07) ***p < 0.0001***	-0.03 (± 0.16) *p = 0.81*
PC2	0.97 (± 0.67) *p = 0.14*	0.25 (± 0.26) *p = 0.33*	-0.57 (± 0.23) ***p < 0.0001***	-0.3 (± 0.2) *p = 0.15*

Table 3. Coefficient estimates (± SE) for all generalized linear mixed models relating core use area size to supplemental food source availability. Core use area size was estimated as 50% kernel density estimates (km^2). Pasture, crop, and livestock was estimated as a proportion of the home range. PC1 and PC2 are principal components 1 and 2 derived from morphological data collected from the study population upon capture. All models were fitted under a Gaussian distribution, include year (2017 through 2020) and bird ID as a random effect (random effect is estimated as variance (± SD)). We did not detect any correlations between predictor variables (Appendix C: Supplemental Table 3). Significant variables are bolded.

	Winter		Summer	
Variable	Male (n = 13)	Female (n = 14)	Male (n = 17)	Female (n = 19)
(Intercept)	**-2.98 (± 1.0)** ***p = 0.002***	**-4.96 (± 0.7)** ***p < 0.0001***	-2.16 (± 1.62) *p = 0.18*	**-6.01 (± 2.28)** ***p = 0.008***
Pasture	-1.34 (± 2.24) *p = 0.54*	**3.77 (± 1.42)** ***p = 0.008***	1.29 (± 3.86) *p = 0.73*	4.33 (± 5.48) *p = 0.42*
Crop	4.49 (± 2.88) *p = 0.11*	**11.05 (± 2.41)** ***p < 0.0001***	-1.25 (± 1.69) *p = 0.45*	3.41 (± 2.8) *p = 0.22*
Livestock	115.28 (± 258.75) *p = 0.65*	-15.72 (± 16.85) *p = 0.35*	513.111 (± 458.24) *p = 0.26*	-22.19 (± 31.57) *p = 0.48*
PC1	-0.17 (± 0.37) *p = 0.64*	-0.27 (± 0.24) *p = 0.26*	-0.3 (± 0.19) *p = 0.12*	0.12 (± 0.58) *p = 0.83*
PC2	0.68 (± 0.86) *p = 0.43*	0.32 (± 0.32) *p = 0.3*	**1.67 (± 0.64)** ***p = 0.009***	-0.34 (± 0.92) *p = 0.7*

We recorded a total of 974 dyadic agonistic interactions at the bait pile (Appendix D: Supplemental Table 1). Most dyadic interactions occurred between hens (n = 581, 59.6% of the total interactions). Interactions between jakes and hens comprised 15.6% of the total interactions (n = 152). Of these, jakes exhibited dominance 63.8% of the time. Interactions between jakes composed 10% of the total interactions (n = 98), followed by bearded hens and hens (n = 48), toms and hens (n = 45), toms and jakes (n = 37), and bearded hens and jakes (n = 7). Toms interacting with hens, jakes, and bearded hens won 100% of the dyadic interactions that they were engaged in. When interacting with jakes, bearded hens won only 14.2% of the time, whereas they won 79.2% of interactions with un-bearded hens. There were only four instances of toms engaging in agonistic dyadic interactions with other toms at the bait pile.

Using intra-categorical interactions only (n = 291), we estimated the relative Elo-rank of the four sex and age categories. Each bird was assigned a rank from 1 (most dominant; high dominance ability) to approximately 4 (most subordinate; low dominance ability) (Sheppard 2013). The number of ranked classes is dependent on the number of individuals included in the analysis (Sheppard 2013). Among the sex and age categories, we found that adult males exhibited the highest position within the social hierarchy (Elo-rank = 1), followed by bearded hens (Elo-rank = 2.59), juvenile males (Elo-rank = 2.96), and hens (Elo-rank = 3.43).

We estimated the robustness of our Elo-rank estimates by randomising the order of interactions and calculating repeatability scores of the ranks (Farine and Sánchez-Tójar 2019). When estimating the Elo-rank among four sex and age classes, we calculated a repeatability score of 0.901 after 1000 iterations. A repeatability score of 0.8 or higher suggests a reasonably robust hierarchy (Sánchez-Tójar et al. 2017). We also estimated the uncertainty of our Elo-rank estimates by dividing the data into two exclusive halves and calculating the correlation of the Elo-ranks for individuals across the exclusive halves of the data (Farine and Sánchez-Tójar 2019). We calculated a Spearman's rank correlation coefficient of 0.6548 after 1000 iterations.

A correlation coefficient of 0.5 or higher suggests a dominance hierarchy with low uncertainty (Sánchez-Tójar et al. 2017).

We found that, during an agonistic dyadic interaction, there was a 65% probability that the higher ranked individual would win the interaction, regardless of the difference in rank between the two individuals. More specifically, we found that, for dyads with a difference in rank of 1 (such as hens and bearded hens), there was an 87% probability that the higher ranked individual (the bearded hen) will win the agonistic interaction. For dyads with a difference in rank of 2 (such as jakes and toms), there was a 65% probability that the higher ranked individual will win the interaction, and for dyads with a difference in rank of 3 (such as hens and toms) the higher ranked individual (the tom) will win 100% of the time (Appendix C, Supplemental Figure 1).

We conducted an additional Elo-rank analysis in which we only included tagged or otherwise identifiable individuals (n = 19) (Appendix C, Supplemental Figure 2). We calculated a repeatability score of 0.99 and a Spearman's rank correlation coefficient of 0.78 after 1000 iterations. In this analysis, each bird was assigned a rank from 1 (most dominant; high dominance ability) to approximately 24 (most subordinate; low dominance ability) (Sheppard 2013). On average, females exhibited a more subordinate position within the flock hierarchy at the bait piles than males. The average male Elo-rank was 7.4, (n = 7 individuals) while the average female Elo-rank was 14.57 (n = 12). There are no tagged bearded hens in this study, so we were unable to compare the Elo-rank of tagged hens and tagged bearded hens in this analysis. Our analysis does include an "anonymous bearded hen" category to account agonistic dyadic interactions between untagged bearded hens and tagged individuals. As expected, based on the overall dominance hierarchy of the flock, tagged adult males exhibited a higher position within the hierarchy than tagged juvenile males. The average Elo-rank among tagged adult males was 3.309 (n = 2), while the average Elo-rank among tagged juvenile males was 9.041 (n = 5).

Rank and Morphology

For tagged and ranked individuals, we found a significant negative correlation between rank and tail length ($r(22) = -0.584$, $p = 0.02$) and between rank and capture weight ($r(22) = -0.54$, $p = 0.04$). Because Elo rank is interpreted inversely, with individuals with lower rank values exhibiting fewer dominant behaviours and residing closer to the bottom of the flock hierarchy (Sánchez-Tójar et al. 2017), our significant negative correlation coefficients indicate that heavier individuals with longer tails are more likely to be successful upon entering into an antagonistic dyadic interaction over access to a supplemental food source. Across all individuals, we did not detect a significant correlation between rank and wing length, tarsus length, or beard length ($p > 0.05$).

When investigating the relationship between rank and each morphological variable for males and females independently, we identified a significant relationship between rank and tail length for female individuals ($r(8) = -0.86$, $p = 0.001$). For males, we identified a significant relationship between rank and wing length ($r(2) = 0.95$, $p = 0.04$). However, our sample size of male individuals was quite small (n = 4). There was no significant correlation between rank and any other morphological variables when we investigated these relationships for each sex independently ($p > 0.05$).

We found no significant correlation between rank and PC1 ($r(12) = -0.3$, $p = 0.28$) or PC2 ($r(12) = -0.39$, $p = 0.16$). When parsing the data by sex we also found no significant correlation between rank and PC1 or PC2 scores for either male or female wild turkeys ($p > 0.05$ for all tests).

When evaluating the effect of group size on the rate of interactions, we found a slightly positive but statistically significant relationship between the interaction rate and the number of individuals present at the bait pile ($r(277) = 0.55$, $p < 2.2e-16$; Figure 4).

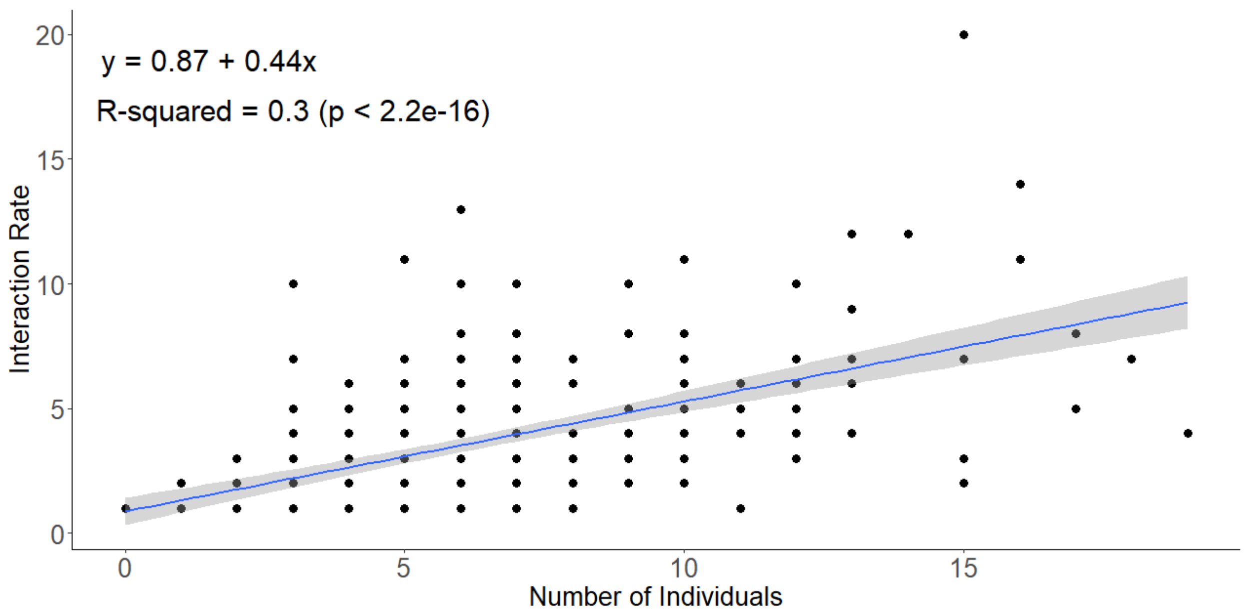

Figure 4. The relationship between the number of interactions in each one-minute video clip and the number of individuals present in the frame at the 30-second timestamp for each video clip (n = 279). Some points overlap on this figure. The fitted line represents the linear regression equation.

DISCUSSION

Most research to date has addressed population-level effects of human-derived resource subsidies on synanthropic species, while relatively few studies have investigated how food subsidies alter aspects of individual behavior, such as foraging patterns and space use

(Boutin 1990, Schoech and Bowman 2001, Scarpignato and George 2013, West et al. 2016). Furthermore, many studies report seasonal home ranges and movement patterns for wild turkeys (Ellis and Lewis 1967, Porter 1977, Everett et al. 1979, Bidwell et al. 1989, Smith et al. 1989, Kurzejeski and Lewis 1990, Gonnerman et al. 2022), but few attempt to identify factors that might explain individual variation in utilization distribution size while accounting for social factors like dominance hierarchy (Badyaev 1994, 1996).

We found support for our hypothesis that there is a relationship between morphology and UD size for male wild turkeys but not for females (Tables 2 and 3). For males, the coefficients for PC1 (primarily driven by weight, beard length, and spur length) and PC2 (primarily driven by tail length) were both negative when modelling home range size. The loadings for PC1 and PC2 were both positive, indicating that as overall male turkey mass increases and sexually selected features become more prominent, summer home range size decreases (Table 3). The opposite was true when modelling male core use areas: the coefficient for PC2 was positive, indicate that as overall body size increases, core use area size increases (Table 3).

Our findings suggest that there is a significant relationship between utilization distribution size and supplemental food source availability for this species. We found evidence to support our prediction that wild turkeys will shrink their utilization distributions around important supplemental resources. However, in contrast to our prediction, the variable coefficients for pasture, crop, and livestock were positive in some models and negative in others, indicating that wild turkeys are contracting their utilization distributions around supplemental food sources during some seasons and may be expanding their UDs to include more resources, both supplemental and natural, during others.

Male wild turkeys contracted their summer home ranges around crop land and pasture, while females expanded their home ranges to include more crop land during the summer. Males also expanded their summer home ranges to include more areas of the landscape containing livestock, while females contracted their summer home ranges around this resource.

Female wild turkeys increased their core use areas to include more pasture and crop during the winter. Core areas are considered to contain the most frequently visited part of a range containing the features most important to the individual (Ewer 1968). As such, it may be that female wild turkeys are increasing the size or their core use areas during the winter when conditions are harsher and natural resources are harder to access (reference). We also detected a positive association between male core use area size and PC2 during the summer. Positive coefficients in combination with positive PC loadings indicate that as overall body size increases, male summer core use area size also increases.

In most years, we did not detect a substantial difference in home range size between the two sexes (Table 1). One exception to this is in 2017, when mean male winter home range size was 5.5 times larger than female winter home range size. This inflated mean is the result of seemingly unusual behaviour from one individual who travelled 22.2 kilometers from their capture location during the winter of 2017, and returned to nearby their original location later that season. At the level of core use, we detected support for our hypothesis that females are likely to have larger UDs than males, as mean female core use area size was larger than that of males in both 2017 and 2018. Female wild turkeys are more vulnerable to starvation than males during inclement weather (Gray and Prince 1988) so they may be expanding their core use areas to include more supplemental foods during the winter to meet their daily nutritional demands. Females are also more strongly affected by the distribution of resources than males during other times of year (Badyaev 1996), which may be evident in their larger summer core use areas.

In general, both male and female wild turkeys typically have smaller utilization distributions during the winter than during the summer (Porter 1977; Miller et al. 1985, Gonnerman et al. 2022). However, in our study, the relationship between season and UD size differs depending on year, sex, and space-use. During the summer, wild turkeys are able to easily move across the landscape and access more abundant natural foods. In contrast, during periods of heavy snow, wild turkeys experience restricted mobility, resulting in reduced foraging activity and other energy-conserving behaviours (Porter 1977, Porter 1980, Roberts et al. 1995, Wright et al. 1996, Nguyen et al. 2004, Kane et al. 2007). When deep snow does not limit turkey movements, they have been noted to have larger winter home range sizes, particularly if they need to use more of the landscape to find food (Dickson 1992). It may be that, in some years, we did not experience enough snow accumulation to induce a measurable effect regarding wild turkey movement patterns at the individual level.

We did not find any evidence to support our hypothesis that hierarchical rank affects the size or composition of individual utilization distributions in this species. However, by investigating dyadic interactions associated with a supplemental food source, we were able to identify a very defined rank among sex and age classes with low uncertainty in our rank estimates. This indicates that social interactions may affect fine-scale access to resources, like anthropogenic food sources, but this dynamic does not necessarily translate to landscape level space-use patterns.

Overall, our findings indicate that there are several factors that play an important role in determining the size and composition of wild turkey utilization distributions, including sex, season, and supplemental food source type and availability. However, there may be additional factors affecting turkey space use that we did not investigate in our study. For instance, the resource dispersion hypothesis (RDH) predicts that increased dispersion of resources in fragmented habitats would increase home ranges of birds and mammals (Johnson et al. 2002, Macdonald 1983). Previous evidence of the RDH has been found in wild turkeys in that the movement distances and home ranges of translocated turkeys were greater at more fragmented sites than at less fragmented sites (Marable et al. 2012). We did not quantify fragmentation in our study. However, Peterborough County is located in a well-developed region of the province with high road density (Statistics Canada 2023). Thus, it is highly likely that habitat fragmentation associated with roads and human dwellings are also defining wild turkey space use in this region.

Dominance Rank

Dominance rank within turkey flocks has previously been assigned based on age; young individuals are subordinate to older individuals (Buchholz 1997), and sex; females are subordinate to males (Healy 1992, Buchholz 1997). However, this study is the first to investigate the hierarchical rank of a wild turkey flock across all discernible sex and age classes. This study is also the first to examine the social hierarchy of wild turkeys as it relates to supplemental food source acquisition.

In our study population, there appears to be a very defined rank among sex and age classes and measures of robustness indicate low uncertainty in rank estimates. We found that interactions between individuals within the same sex/age category were much more common than interactions between individuals of different sex/age categories, particularly between hens (Appendix C: Supplemental Table 1). We also found that individuals with large differences in rank between them were much less likely to interact than individuals with lower differences in rank between them. Minor, non-injurious threats, posturing, and display behaviours, all reinforce social hierarchies within the group as individuals learn to evaluate their chances of successfully challenging dominant conspecifics (Parker 1974, Whitehead 2008, Sheppard et al. 2013). As such, we might expect to see more interactions between similarly ranked individuals among the hierarchy, than those with greater differences in rank between them. This also indicates that there may be avoidance behaviours occurring. For instance, a hen never instigated an interaction with an adult male and an adult male never lost an interaction except to another adult male. This is consistent with qualitative observations made by Watts (1969). Furthermore, there were only four instances of adult male – adult male interactions of the total 974 recorded interactions. Avoidance behaviour in wild turkeys has been previously detected using male-decoy trials (Buchholz 1997). Males tended to avoid feeding on seeds near decoy males with longer snoods. Thus, it is possible that adult males in our study were simply avoiding either unnecessary or too-risky interactions at the bait pile.

Another explanation for this phenomenon is that kin selection among males is driving interactions associated with access to supplemental food resources. At the end of winter, prior to breeding season, male wild turkeys participate in aggressive fights involving the use of their spurs and engage in "wrestling" behaviours (Watts 1969, Watts and Stokes 1970, Healy 1992). As a result of these interactions, typically involving sibling brothers, a dominant male is determined. Later in the season, sibling males will often engage in cooperative courtship displays (Krakauer 2005) in which brothers will gobble, strut, and fan their tails together in an attempt to attract a female (Watts 1969, Watts and Stokes 1971, Healy 1992). Dominant and presumably high-quality males are much more likely to reproduce than subordinate males in the courtship coalition (Krakauer 2005). However, the subordinate males, if closely related to the successful male, receive indirect fitness benefits as calculated by Hamilton's rule (Hamilton 1964, Lucas et al. 1996). This shift in behaviour from agonistic to cooperative indicates that fights between individuals likely serve a very specific, adaptive function in this species (Parker 1974). For instance, it may be that kin selection among males is driving interactions associated with access to food resources in addition to interactions associated with breeding and courtship displays. Watts (1969) observed the interactions between male siblings at a small bait pile of sorghum and noticed that individuals often took turns pecking from the bait pile rather than engaging in agonistic interactions over access, providing further support for this hypothesis.

We recorded few interactions involving bearded hens. However, this is likely reflective of the proportion of bearded hens in the overall population rather than the affinity for these individuals to visit the bait pile or interact with other individuals. In wild populations, 1 – 29% of female wild turkeys may possess beards, a typically male sexually-selected trait (Lewis 1967, Williams and Austin 1988, Beasom 1970). However, there has been little empirical investigation into what factors may be driving regional variation of this frequency and the potential social dynamics involved. Of the hens in our overall study population, only ~3% possessed beards (n =

1). The only bearded and tagged female in our study population was captured in 2018 but did not appear in any of our one-minute video clips. As such, all the bearded hens included in the rank analysis are anonymous, and the true sample size is unknown.

We did not measure the snood length of captured individuals in this study. However, previous research indicates that a male's relaxed snood length, another sexually selected feature (Buchholz 1995), can be predictive of the outcome of male-male competition, with dominant males tending to have longer snoods (Buchholz 1997). Snood length has also been shown to be negatively correlated with coccidian parasite load and positively correlated with overall male body condition (Buchholz 1995). Furthermore, researchers have found that displaying males have significantly deeper caruncles than non-displaying males (Buchholz 1997). As such, future studies wishing to identify suitable proxies for hierarchical rank in wild turkeys should account for differences in head ornamentation, particularly for male individuals.

Although snood length can be associated with hierarchical dominance (Buchholz 1997), and mass can be significantly correlated with snood length (Buchholz 1997), we did not find any evidence for a direct relationship between mass and rank in this species, a positive association that has been previously assumed (Badyaev 1994, 1996, Badyaev et al. 1998).

Our study investigates agonistic dyadic interactions associated with access to an anthropogenic food source. However, there are likely agonistic interactions occurring in relation to access to other resources, as well. For instance, A. V. Badyaev reports personal observations of aggressive territorial interactions between dominant and subdominant female wild turkeys associated with roost sites (Badyaev et al. 1998) and Watts (1969) also notes preferential perch selection among certain individuals.

There were several limitations of this methodology that are worth noting. Although the coloured leg bands allowed us to identify some individuals using camera trap footage, in many instances the colour of the band was difficult to determine, requiring observers to review the one-minute clips numerous times to achieve a correct identification. Deep snow and ground vegetation often obscured the turkeys' legs and covered their coloured bands. When the leg bands were visible, low-light conditions and the reflection of the sun on the coloured metal bands both posed a challenge for discerning band colours. The significant time-cost associated with re-watching videos could be reduced by using an alternative tagging system. For instance, patagium markers (Knowlton et al. 1964, Watts 1969, Hoffman 1991).

Additionally, it is important to consider that the interactions recorded in this study represent only a proportion of the interactions occurring among the flock during each one-minute clip. For instance, there may have been individuals outside of the frame eliciting reactionary behaviours from individuals inside the frame while remaining unobserved.

Management Implications

Our results indicate that wild turkeys do utilize supplemental food sources throughout the year. However, we did not test for potential population-level effects of resource exploitation, nor should a positive effect on population size be assumed. For instance, previous observations suggest that individual avian foragers in rural environments may have access to comparatively fewer resources (Shochat et al. 2004, 2006, Faeth et al. 2005) despite urban habitats containing richer and more productive food sources (Mills et al. 1989, Atchison and Rodewald 2006, Leston and Rodewald 2006). This relationship can occur when resource overexploitation results in population densities that exceed those sustainable by the landscape

(Shochat et al. 2004). Wild turkeys are not considered an urban species but are regularly found within or in close proximity to peri-urban and rural residential neighbourhoods (Fuller et al. 2013, Baici and Bowman 2023, Gonnerman et al. 2022). Furthermore, stands of soybean or corn are not sufficient to support wild turkey populations unless natural foods are available (Nguyen et al. 2004). As such, the species-specific impacts of supplemental feeding should be considered prior to engaging in this practice as a population management strategy.

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Chapter 4: Wild turkeys forage and roost with kin during the non-breeding season

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Contributions: Baici and Bowman conceived and designed the study; Baici analyzed the data and wrote the manuscript. Bowman critically reviewed the manuscript. Adey contributed significantly to data collection and analysis, and Walker conducted all laboratory methods and microsatellite amplification interpretation.

Abstract
Kin-selection is an evolutionary mechanism employed by many species in diverse contexts. To understand the role of kin-selection for wild turkeys (*Meleagris gallopavo*), we investigated conspecific foraging and roosting associations during the non-breeding season. Wild turkeys were fitted with either GPS- or VHF- transmitters and were genotyped at thirteen microsatellite loci. Association indices indicate that, during the summer, wild turkeys roost and forage with unrelated conspecifics at a similar rate to with kin. However, during the winter we found that pairs that were more genetically related were more likely to roost and forage together. We also individuals that were captured together exhibited higher average pairwise relatedness than simulated random groups of the same size. It is well known that wild turkeys demonstrate kin-selection during the breeding season in the form of cooperative courtship. However, we present the first empirical evidence that wild turkeys may be exhibiting kin-selection during the non-breeding season, as well.

INTRODUCTION

Kin-selection is an evolutionary mechanism employed by many species in diverse contexts and describes a phenomenon in which individuals that are closely related may benefit indirectly from the fitness of their relatives through inclusive fitness (Hamilton 1976a, 1976b).

As such, kin-selection may evolve when relatives are in spatial proximity because they can then have a direct influence on each other's fitness (Ralls et al. 2001). Wild turkeys (*Meleagris gallopavo*) exhibit kin-selection during the breeding season in the form of cooperative courtship (Krakauer 2005). Nonbreeding males assist closely related males, typically brothers, in courtship by gobbling and displaying in conjunction with the breeding male, as well as guarding females from other nearby males (Watts and Stokes 1971). It has not yet been explored, however, whether turkeys exhibit kin-selection during the non-breeding season.

In temperate regions, eastern wild turkeys cluster to form large wintering flocks (Vander Haegen et al. 1989). These groups typically remain together until the spring when males begin to gobble and females begin to disperse in search of mates and nesting territories (Healy 1992). By using microsatellites to estimate allele frequencies, Rhodes et al. (1995) were the first to demonstrate that there was significant genetic variation between wintering flocks of Rio Grande turkeys (*Meleagris gallopavo intermedia*) in Kansas. Turkeys shared more alleles with individuals from the same wintering flock than they did with individuals from other wintering flocks (Rhodes et al. 1995). The following year, Boone and Rhodes (1996) demonstrated that the same was true for the eastern wild turkey (*Meleagris gallopavo silvestris*). The researchers sampled 36 male and 36 female eastern wild turkeys from four different wintering flocks and found that there were significant genetic differences in allele frequencies between flocks (Boone and Rhodes 1996).

It is not well-understood why closely related wild turkeys might be more likely to flock together during the winter. Boone and Rhodes (1996) theorize that distinct localized variation in the wild turkey is most likely a function of mating tactics and dispersal behaviour. However, this idea has not been empirically tested. Kin-selection during the non-breeding season has been explored in another cooperatively breeding bird, the long-tailed tit (*Aegithalos caudatus*). Napper and Hatchwell (2016) found kinship to be an important factor in establishing initial winter flocks, as nuclear family groups joined the same wintering flock at the beginning of the season (Napper and Hatchwell 2016). Napper and Hatchwell (2016) also found that individuals that switched flocks mid-season typically did so in conjunction with one or more closely related individuals. In addition to demonstrating the importance of kinship in establishing wintering flocks, researchers used playback experiments to demonstrate that a lack of aggression toward individuals belonging to other flocks was influenced by kinship, indicating that kin associations are likely resulting from a preference to flock with relatives rather than an aversion to flocking with non-relatives (Napper and Hatchwell 2016). Furthermore, as revealed by social network analyses, kin in close association during the nonbreeding season, were more likely to assist one another during the following breeding season (Napper and Hatchwell 2016).

Kinship may also play a role in influencing social interactions between individuals during specific daily activities, such as roosting and foraging. For instance, Siberian jays (*Perisoreus infaustus*) exhibit a high degree of genetic nepotism both during foraging and predator encounters (Griesser, et al. 2015). In the greater horseshoe bat (*Rhinolophus ferrumequinum*), the greatest positive spatial associations were observed between female kin, specifically females and their daughters (Rossiter, et al. 2002). Contrarily, in southern flying squirrels (*Glaucomys volans*) (Garroway, et al. 2013) and big brown bats (*Eptesicus fuscus*) (Methen et al. 2008), kinship does not correlate with nesting or roosting associations. In golden-crowned sparrows genetic relatedness did not influence social foraging behaviour.

Wild turkeys are highly social birds that roost communally year-round in trees, except when females are nesting (Watts and Stokes 1971, Healy 1992). Roost selection may be influenced by factors such as predator avoidance, disturbance, microclimate, and proximity to foraging areas (Lewis 1996, Béchet et al. 2010, Adey et al. 2023). Adey et al. (2023) tested the hypothesis that multiple benefits exist for roost tree selection, including thermoregulation, resource acquisition, and protection from predators, and found that thermoregulation was not the driving force behind wild turkey roost selection but instead, that predator avoidance appears to play the most important role, with some weaker evidence in support of proximity to resources (Adey, et al. 2023).

Roost lability is also an important aspect of eastern wild turkey behavioural ecology that has likely evolved to minimize predation risk in landscapes where available roosts are widespread, consistent with other species that distribute roost sites throughout their ranges (Lewis 1995). If predator avoidance is an important driver of roost site selection for this species, it is likely that wild turkey sociality plays an important role in this dynamic. For instance, individuals may benefit from the increased vigilance of sleeping in a group with conspecifics (Paclík and Weidinger 2007, Wiens et al. 2014, O'Brien et al. 2021).

In addition to during roosting, wild turkeys also exhibit sociality while foraging. Individuals form foraging groups of varying sizes depending on the season and move through the habitat as a group while foraging on natural foods, like tree mast, fruit, and seeds (Healy 1992), and supplemental foods, like waste grain associated with plant agriculture (Otieno and Frenette 2017), livestock feed associated with animal agriculture (Porter et al. 1980, Vander Haegen et al. 1989, Thompson et al. 2009) and bird feeders (Niedzielski and Bowman 2016, Baici and Bowman 2023). The foraging behaviour of wild turkey broods is well-understood (e.g., Chamberlain et al. 2020, Bakner et al. 2022, Nelson et al. 2022, Tebo et al. 2021), and the fitness benefits associated with brood interactions between hen and poults (parent and offspring) and among poults (siblings) are clear (Hamilton 1976a, 1976b, Ashbrook et al. 2017). However, we do not yet understand how kinship may be affecting roosting or foraging associations between adult individuals in this species.

We sought to quantify roosting and foraging associations in wild turkeys during both the breeding and nonbreeding seasons by determining genetic relatedness within and between flocks and quantifying social interactions associated with specific behavioural states. Specifically, we examined foraging and roosting interactions between individuals. We hypothesized that kin selection is occurring during both the breeding and nonbreeding seasons, and as a result expected individuals to be more closely related to other individuals within their foraging and roosting flocks than they were to individuals belonging to other flocks.

Kin-selection evolves in species when related conspecifics are in close proximity to one another as it is during these instances that they may have an influence on one another’s fitness (Hamilton 1976a, 1976b, Ralls et al. 2001). As such, to examine the evolutionary potential for kin-selection to occur in relation to specific behaviours, we also estimated indices of the mean number of foraging and roosting flocks within our study population, the size of these flocks, and the impact of seasonality on both of these metrics.

METHODS

Study Area

Our study was conducted in Peterborough County: a region in south-central Ontario, characterized by mixed forest, plant, and animal agricultural operations (Figure 1 in Chapter 3). The southern section of the county is a mix of agriculture, urban and lakefront properties. The dominant industries are agriculture and tourism and the main crops grown are soybean, corn, and winter wheat. Pasture and hay fields are also significant components of the landscape to support animal agriculture (Statistics Canada 2023). Wild turkeys in this region may occur in relatively high densities (Baici and Bowman 2023) and readily use supplemental food sources (Baici and Bowman 2023).

Capture and DNA Collection

Wild turkeys were captured at baited locations in Peterborough County during each winter, 2017 – 2019, using a rocket net (Grubb 1988) following methods outlined by Niedzielski and Bowman (2014, 2016). Wild turkeys are captured during the winter in Ontario, as their winter flocking behaviour allows for the easy capture of multiple individuals at once. Processing involved weighing, sexing, and attaching GPS- (Model PP-VHF-3600L, Lotek Wireless Inc., Newmarket, Canada) or VHF-transmitters (Model Series A15400, Advanced Telemetry Systems, Isanti, MN, USA). Transmitters were attached using a backpack-style harness made of shock cord (Norman et al. 1997, Niedzielski and Bowman 2014, 2016). Five individuals were not fitted with a transmitter due to feather loss at the site of potential attachment. Both GPS- and VHF-transmitters weighed less than the recommended maximum of 3% of adult female weight (Millspaugh and Marzluff 2001).

In the field, whole blood samples were collected using sterile needles by puncturing the brachial vein; samples were stored in vacutainer tubes. To prevent blood samples from freezing in the field, tubes were stored in an insulated container with hot water bottles for several hours prior to centrifuging. Blood samples were centrifuged at maximum speed (3250 RPM) to for 1 minute isolate the serum from the whole red blood cells. The serum was then removed from each tube and stored separately. Samples were stored at -20 degrees Celsius until DNA extraction and amplification.

Animal Locations

For turkeys fitted with GPS-transmitters, fix intervals varied depending on the capture year and location and ranged from once per hour to once every 4.25 hours. For turkeys fitted with VHF-transmitters, we derived location estimates by first triangulating individuals in the field and then estimating locations using Location Of A Signal (LOAS) (version 4.0.3.8, Ecological Software Solutions LLC, Florida, USA), software developed to calculate locations for radio-telemetry studies. Individuals were triangulated 2 - 4 times a week for the duration of the study. Whenever possible, a minimum of 3 bearings were taken to estimate an animal's location. However, in some situations where 3 bearings were not possible, we estimated locations using 2 bearings.

We completed most triangulations from roads, recording all bearings in <30 minutes. We used a hand-held three- or five-element directional Yagi antenna and a VHF receiver (Model R-1000 Telemetry Receiver, Communications Specialists Inc.) to determine the direction of each turkey from the observer's location (Niedzielski and Bowman, 2014, 2016). Locations of direct observations of tagged individuals were also recorded and included in our analyses.

We estimated the accuracy of location estimates derived from triangulation by conducting field trials in which we placed VHF-transmitters in known locations within our study area. We then triangulated the test transmitter using the same methods used to triangulate VHF-tagged turkeys. To account for variability in frequency detection and triangulation conditions, we conducted field trials using three VHF transmitters, each with different signal frequencies, under varying levels of canopy cover, ranging from open field to dense white cedar (*Thuja occidentalis*) stands.

For individuals fitted with GPS-transmitters, we derived location estimates directly from the fix locations collected and stored by the transmitters. We used a PinPoint Commander (Lotek Wireless Inc.) and a hand-held three- or five-element directional Yagi antenna to download data directly from each GPS-transmitted in the field.

To estimate the accuracy of location estimates derived from GPS-fixes, we placed 6 GPS-transmitters in known locations within our study area. For each GPS accuracy test, after 21 days, or approximately 218 (45 – 560) fixes per transmitter, we downloaded the location estimates and calculated the mean distance between the recorded GPS fixes and the known transmitter location. Again, we placed transmitters in a variety of different habitat types within our study area, with varying levels of canopy cover and understory vegetation. The mean accuracy of location estimates derived from triangulation was 198 meters (+/- 76.93), whereas the mean accuracy of location estimates derived from GPS-transmitters was 19.8 meters (+/- 10.7).

Roosting and Foraging Flocks

We applied findings from Byrne et al. (2014) to identify periods of the day that wild turkeys were most likely to be foraging to estimate the approximate location of foraging flocks.

Using first passage time analysis, Byrne et al. (2014) determined that wild turkeys tended to move towards bird feeders during midday and were more likely to remain in a 110 radius during this time. From this, researchers inferred that wild turkeys were likely foraging during this time compared to later in the afternoon when birds were more likely to move quickly through localized areas (Byrne et al. 2014). They defined midday by first dividing the day into three segments, beginning one-half hour before sunset and ending one-half hour after sunset. We define midday as 12:00 pm to 13:45 pm local time to represent locations at which wild turkeys were likely foraging.

Nesting hens are more likely to forage alone during the incubation period and roost alone while the poults are young (Healy 1992, Chamberlain et al. 2020). As such, we removed both VHF and GPS locations associated with known nesting events within our study population when estimating foraging and roosting association indices. Individuals who were tracked for less than 13 days for birds fitted with VHF transmitters, or 15% of the season for birds fitted with GPS transmitters, were also excluded from the analysis (Niedzielski and Bowman 2016).

We used the leaderCluster R package (Arnold 2014) for R Studio (Pace et al. 2017) to group wild turkey locations based on proximity and derive foraging or roosting clusters of individuals. The leaderCluster (Arnold 2014) clustering algorithm allows users to set the approximate radius of clusters rather than the desired number of clusters (Arnold 2014). This is useful when attempting to assign observations to clusters based on geographic distance. We set the foraging clustering radius based on Byrne et al. (2014; 110 m) and the average GPS error estimated by our field trials (19.9 m +/- 26.2 m). We added these values together to set the maximum distance between two individuals within the same cluster (156.1 m) and divided this value by two as the clustering algorithm calls for a radius rather than a diameter. This resulted in a foraging clustering value of 78.05 representing and a clustering area of 0.019km^2.

To calculate roosting clusters, we chose a much smaller clustering radius as wild turkeys are sleeping while roosting and are thus stationary. We considered individuals roosting in the same tree to be associating with one another. Wild turkeys roost on branches in the tree canopy (Healy 1992) and in our study area, there were no tree canopies larger in diameter than the average GPS error estimated by our field trials (19.9 m +/- 26.2 meters). As such, we chose this value (19.9 m) for our roosting clustering radius, resulting in a circular roosting area of 0.0012km^2.

To account for the differences in accuracy between our two transmitter types, we chose to exclude the location estimates for VHF-tagged birds from the clustering analysis and instead generated spatial clusters using GPS-fixes only. We then compared the afternoon location estimates (between 12:00 and 13:59 EST) for VHF-tagged birds with the location of foraging clusters and assigned each VHF-tagged individual to the geographically closest foraging cluster, within 198 meters (the mean VHF-transmitter triangulation error). VHF fixes farther than 198 meters from the nearest foraging cluster were considered to represent a new foraging cluster. Individuals assigned to the same foraging flock were considered to be foraging together.

Roosting flocks were identified by clustering GPS fixes recorded over night between 21:00 and 05:59 EST. Again, we assigned VHF-tagged birds to the roosting flock in closest proximity to their evening locations within 198 m (+/- 76.93). Individuals assigned to the same roosting flock were then considered to be roosting together.

Flock Size and Number

We derived a seasonal index of flock size for both behaviours (foraging and roosting) by calculating the number of tracked individuals in each daily foraging and nightly roosting cluster. We were interested in the effect of seasonality on flock size, so data were pooled across years but parsed by season (winter and summer). We then calculated mean flock size for each season and behaviour. We also derived a seasonal index of the number of foraging and roosting flocks by calculating the number of daily foraging and nightly roosting clusters and then calculating the mean number of clusters for each season and behaviour.

Association Indices

We used the ANTs R package (Sosa et al. 2020) to estimate wild turkey association indices and metrics of index precision. The half-weight index (α) was calculated as:

$$\alpha = x/(x + yAB + 1/2(yA + yB))$$

where *x* was the number of days individual *A* and *B* roosted or foraged together, *yA* was the number of days that individual *A* was foraging or roosting, *yB* was the number of days that individual *B* was foraging or roosting, and *yAB* was the number of days where both *A* and *B* were observed but not associated. We used the half-weight index (α) (Whitehead 1997; 2008) as it is the most commonly used index of association and accounts for bias associated with unidentified individuals (Whitehead 1997; 2008). Wild turkeys are highly gregarious (Healy 1992) and our tagged study population represented only a small proportion of our study area's total wild turkey population. As such, the half-weight index was the most appropriate index of association for our study. To account for the effect that seasonality may have on kin-selective behaviours, we calculated foraging and roosting pairwise sharing association indices annually for each dyad and also for each calendar season (fall, winter, spring, summer).

Genotyping and Relatedness

Whole blood (50 uL) obtained from 77 wild turkeys was used for genomic DNA extraction using the E.Z.N.A.® tissue DNA Extraction system (Omega Bio-tek) following the manufacturer's guidelines. We initially analyzed 15 microsatellite loci: WT54, WT75, WT83, WT10, WT90-2 and WT38-2 (Latch et al. 2022), MNT11, MNT13, MNT17 and MNT20 (Reed et al. 2002), TUM-1, TUM32, TUM50, TUM6, and TUM23 (Table 1; Huang et al. 1999, Mock et al. 2002). Amplification was done in two multiplex reactions with fluorescently labelled forward primers: Multiplex WT-A included TUM1, TUM50, TUM32, WT54, MNT17, WT90-2, WT75 and MNT11; Multiplex WT-B included WT38-2, TUM23, WT83, MNT13, MNT9, WT10 and MNT20. We dropped TUM1 and WT90-2 from Multiplex WT-A, due to irregular binning patterns that created inconsistent scores across runs. A 10x2µM primer mix of 1mL was made for each multiplex reaction and frozen at -20^{o}C. Total volume for each reaction was 12µL with 2µL of DNA (standardized to 2.5ng/ µL), 1x primer mix, 1x Qiagen Multiplex PCR Master Mix (Qiagen, Toronto ON) and 2.8µL of DNAase-free molecular grade water. Cycling conditions were as follows: initial denaturation and polymerase activation at 95 ^{o}C for 15 mins, followed by 30 cycles of 94 ^{o}C for 30 s, 57 ^{o}C (for both multiplexes WT-A and WT-B) for 90s, and 72 ^{o}C for 60 s, followed by a 60 min final extension at 60 ^{o}C. We combined 6µL of GenScan 500 LIZ size standard (Applied Biosystems) with 1 mL of HiDi Formamide and then visualized amplified DNA on an ABI3730 (Applied Biosystems) by combining 0.75µL of PCR product with 10µL of HiDi Formamide-500 LIZ mixture. Samples were genotyped with standardized bins in Genemarker v7.1 (SoftGenetics). For each locus, we calculated the number of alleles in our study population and estimated observed and expected heterozygosity of each gene using the "adegenet" package (Jombart 2008, Jombart and Ahmed 2011) for R Studio (Pace et al. 2017).

Table 1. Summary of molecular data used to amplify genotypes for wild turkeys including the number of alleles per locus (N_A), the observed heterozygosity (H_o), and the expected heterozygosity (H_e).

Locus	N_A	H_o	H_e	Source of the primers
WT54	12	0.81	0.8	Latch et al. 2002
WT75	8	0.65	0.71	Latch et al. 2002
TUM50	12	0.87	0.82	Huang et al. 1999, Mock et al. 2002
MNT17	15	0.82	0.79	Reed et al. 2002
MNT11	5	0.34	0.56	Reed et al. 2002
TUM32	10	0.86	0.85	Huang et al. 1999
TUM6	11	0.81	0.82	Huang et al. 1999
MNT20	5	0.55	0.63	Reed et al. 2002
WT38-2	9	0.92	0.86	Latch et al. 2002
TUM23	7	0.71	0.75	Huang et al. 1999, Mock et al. 2002
WT83	3	0.38	0.57	Latch et al. 2002
WT10	5	0.31	0.31	Latch et al. 2002
MNT13	12	0.79	0.8	Reed et al. 2002

Several methods exist for estimating pairwise relatedness using microsatellite genotype data (e.g. Lynch and Ritland 1999, Queller and Goodnight 1989, Wang 2002). Using the "related" (Pew et al. 2015) package for R Studio (Pace et al. 2017), we evaluated the suitability of the Li et al. (1993), Lynch and Ritland (1999), Queller and Goodnight (1989), and Wang (2002) estimators by generating simulated genotypes of known relatedness based on our population's observed allele frequencies and calculating relatedness between simulated genotypes using each of the four estimators. We generated the same number of simulated genotypes as were observed (n = 77). We then estimated Pearson's correlation coefficients between observed and expected relatedness values for each estimator (Table 2).

Each of our seasonal association indices were right-skewed and zero-inflated. As such, for both behaviours (foraging and roosting) we first classified dyads as individuals that had either never flocked together (0) or as individuals that had flocked together, even just once during the study period (1). To test for a relationship between roosting and foraging social behaviour, 0 or 1, and genetic relatedness of each pair (Queller and Goodnight 1989), we then conducted a Kruskal-Wallis test (Kruskal and Wallis 1952): a non-parametric test commonly employed to compare means between groups. To account for the variation in environmental factors associated with season in temperate regions, we conducted separate Kruskal-Wallis tests for each season (fall, winter, spring, summer).

We tested for a relationship between the estimated proportion of time a pair roosted and foraged together (the half-weight index; Whitehead 1997, 2008) and pairwise relatedness (with Mantel tests). Again, to account for seasonal variation, we conducted separate Mantel tests for each season (fall, winter, spring, summer). Using the “ade4” (Dray and Dufour 2007) package for R Studio (Pace et al. 2017), we conducted permutation tests (9999) to estimate summary statistics.

Capture Associations

To explore the potential effect of kinship on associations at supplemental food sources, we investigated the relationship between genetic relatedness and capture associations. We considered wild turkeys to belong to the same capture group when they were captured on the same date and at the same location. In our study, no more than one capture event occurred per day.

We estimated pairwise relatedness for each dyad within each capture group using the Queller and Goodnight (1989) estimator and calculated mean pairwise relatedness for each capture group. We then created 16 simulated capture groups of the same size as our observed capture groups but comprised of individuals randomly selected from the population. We estimated mean pairwise relatedness for each simulated capture group and conducted a t-test to determine if there was a significant difference in mean relatedness between capture groups and random groups of the same size.

RESULTS

Genotyping

We successfully amplified genotypes for 77 individuals at 13 microsatellites for analyses. The number of alleles per locus ranged from three to 15 and we did not detect any substantial differences between observed and expected heterozygosity for any loci (Table 1).

Foraging and Roosting Associations

We collected 76,536 GPS fixes representing location estimates for 51 wild turkeys. From those, we generated 4016 foraging clusters over 924 study days and 34 062 roosting clusters over 1097 study nights. For 21 individuals fitted with VHF transmitters, we determined 1743 locations. Of those, <5% were derived using only 2 bearings. When assigning VHF tagged individuals to foraging clusters, no VHF fixes were further than 198 meters from the nearest foraging or roosting cluster. As such, all VHF-fixes and associated individuals were assigned to GPS-derived clusters.

The mean number of daily flocks also varied by season, with more flocks detected during the summer than the winter for both behaviours (Figure 1). The mean number of daily foraging flocks during the winter was 2.84 (SD = 2.1, n = 297) while the mean number of daily roosting flocks during the winter was 2.04 (SD = 1, n = 545). The mean number of foraging and roosting flocks during the summer was 2.23 (SD = 1.19, n = 627) and 1.46 (SD = 0.36, n = 552), respectively.

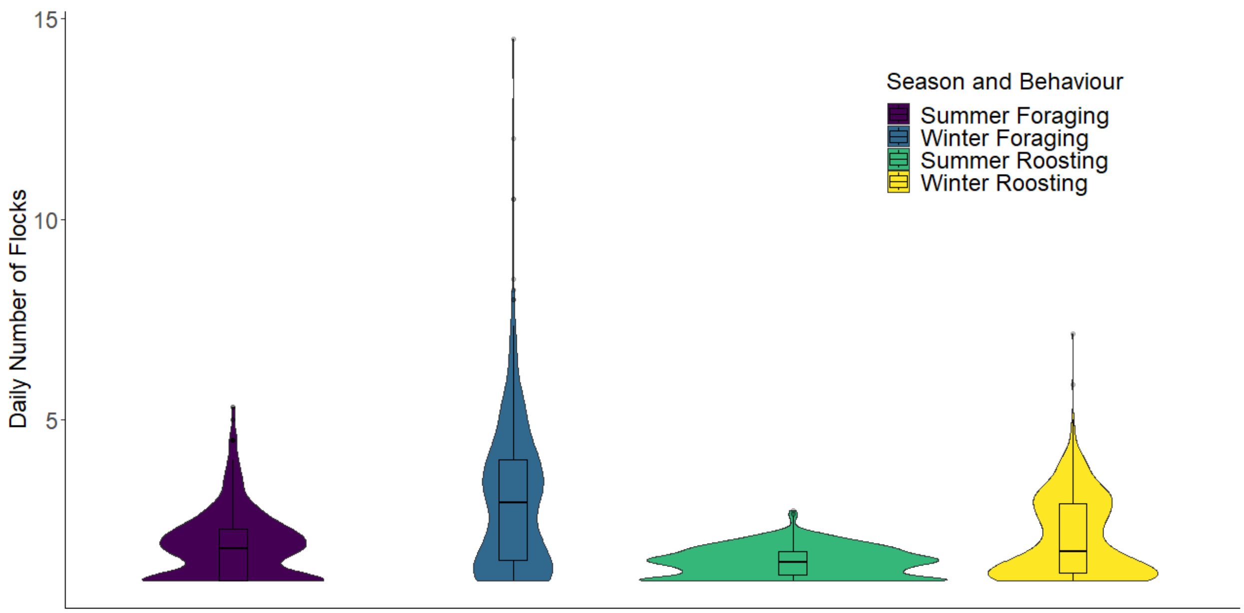

Figure 1. A violin and box plot of the number of daily foraging and roosting flocks of wild turkeys (Meleagris gallopavo) site in Peterborough County, Ontario, Canada during January 2017 to December 2019. Data is divided by season and behaviour but pooled across years.

Effects of Relatedness and Seasonality on Sociality

Of the four estimators that we compared, we found that the Queller and Goodnight (1989) estimator was the most appropriate for our dataset as the correlation coefficient between observed and expected values was highest for this estimator (Table 2). The Queller and Goodnight (1989) method of estimating relatedness is similar to both the Pamilo (1984) and Crozier et al. (1984) methods and to a measure based on the F-statistic estimates of Weir and

Cockerham (1984). Their estimate represents a modification to the multidimensional regression coefficient in that it corrects the bias due to small number of groups sampled. It has also been argued that it provides a more natural and consistent method of combining information from different loci by giving more weight to more informative loci (Queller and Goodnight 1989).

Table 2. Correlation coefficients between observed and expected values for four commonly used pairwise relatedness estimators.

Estimator	Correlation coefficient between observed and expected values
Wang 2002	0.820
Li et al. 1993	0.812
Lynch and Ritland 1999	0.757
Queller and Goodnight 1989	0.827

There was no relationship between genetic relatedness and sociality when coding association as a binary variable: in association with one another (1) or not (0). During the summer, we found no significant difference in genetic relatedness between individuals that roosted together and those that did not (Chi square = 0.01, p = 0.89, df = 1). We detected the same pattern in roosting sociality during the winter (Chi square = 0.62, p = 0.42, df = 1) and in foraging sociality during both the summer (Chi square = 0.07, p = 0.79, df = 1) and winter (Chi square = 1.51, p = 0.21, df = 1).

When considering association to be a continuous variable (i.e., the half-weight index), the Mantel tests indicated that relatedness between pairs was not a good predictor of the estimated proportion of time those individuals foraged ($n_{\text{individuals}} = 27$, $r = -0.02$, $p = 0.48$) or roosted ($n_{\text{individuals}} = 27$, $r = 0.19$, $p = 0.14$) together during the summer. However, during the winter we found that genetic relatedness between pairs was a significant predictor of how often those same individuals both foraged ($n_{individuals}$ = 42, r = 0.21, p = 0.0088) and roosted ($n_{individuals}$ = 42, r = 0.36, p = 0.001) together.

Capture Associations

There were 16 capture events during our study. Capture groups ranged from 3 to 8 individuals with a mean group size of 4.8 individuals. We found a significant difference in average pairwise relatedness between capture groups and random groups of the same sample size, with capture groups having significantly higher r-values than random groups ($t(24.18)$ = 3.45, 0.002; Figure 2). Three capture groups displayed below average pairwise relatedness (Figure 2).

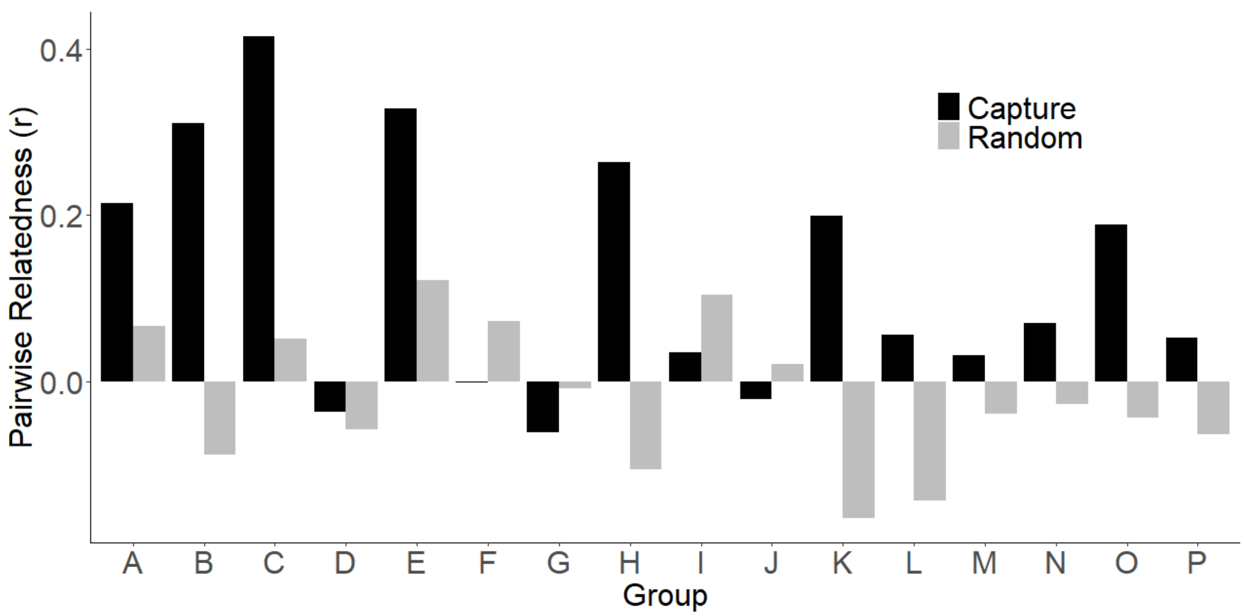

Figure 2. Estimated pairwise relatedness values for 16 capture groups within our study area in Peterborough, Ontario and mean simulated pairwise relatedness values for 16 groups of the same size but with random group membership. Negative pairwise relatedness values indicate that groups are less related than average, while positive pairwise relatedness values indicate that groups are more related than average.

DISCUSSION

Kin-selection refers to the phenomenon whereby individuals can benefit indirectly by helping a relative (Hamilton 1976a, 1976b, Lucas et al. 1996). Kin-selection can explain the evolution of cooperative courtship in wild turkeys (Krakauer 2005). In cooperative courtship coalitions, subordinate males do not themselves reproduce, but their indirect fitness as calculated by Hamilton's rule more than offsets the cost of helping (Krakauer 2005). Here, we present the first empirical evidence that wild turkeys forage and roost during the non-breeding season with kin.

During the summer, both related and unrelated individuals roosted and foraged together at a rate not appreciably different than if associations were random (Queller and Goodnight 1989). However, our results indicate that, during the winter, related individuals preferentially roosted and foraged with kin. We detected this relationship only when coding association as a continuous variable representing the proportion of days individuals either roosted or foraged together. When considering association to be a binary variable (birds did not interact with one another (0), or they have interacted a minimum of once (1)), we did not detect such a relationship between sociality and genetic relatedness. This indicates that, although wild turkeys are sometimes roosting and foraging with unrelated conspecifics, they are preferentially associating with kin while foraging and roosting during the winter.

In our study, trapping occurred exclusively during the winter. We found that individuals that were captured together were more closely related to one another than expected if the groups were composed of random members, indicating that individuals may be preferentially foraging with kin at both natural and supplemental foraging sites. We did not investigate daily changes in group membership for foraging or roosting flocks nor did we compare capture groups with foraging or roosting flocks. However, it may be informative to quantify changes in group membership over time to better understand the seemingly dynamic relationship between social associations and genetic relatedness in this species.

Theoretical modelling using evolutionary game theory indicates that group formation in animals is likely an evolutionarily adaptive behaviour (Javarone and Marinazzo 2017). Empirical studies of vertebrates and invertebrates have demonstrated that animals typically form groups for one or more of five functional reasons: (i) predator avoidance; (ii) resource acquisition; (iii) mate acquisition; (iv) offspring care; and (v) homeostasis (Bourke 2011, Hofmann et al. 2014). In the context of foraging, wild turkeys are likely to experience benefits associated with both increased foraging efficiency (Waite and Grubb 1988, Templeton and Giraldeau 1995, 1996, Smith et al. 1999) and greater predator vigilance (Elgar 1989, Magrath et al. 2009, 2015, Fallow and Magrath 2010, Carrasco and Blumstein 2012) when joining foraging flocks. These benefits may be even more pronounced when flocking with kin, as indirect fitness implications may act as incentive to cooperate and share information related to finding food and detecting predators, particularly during the winter in temperate regions when environmental conditions are generally harsher and natural food sources are more difficult to access (Vander Haegen et al. 1989, Roberts et al. 1995, Paisley et al. 1996, Nguyen et al. 2003, Kane et al. 2007, Restani et al. 2009).

In the context of roosting, wild turkeys are also likely to experience benefits associated with predator detection and avoidance when roosting communally rather than independently. Recent findings from Adey et al. (2023) suggest that thermoregulation is not an important predictor of roost selection, but that predator avoidance appears to play an important role. We also found some weaker evidence in support of roost selection in response to proximity to resources (Adey et al. 2023), perhaps lending support for the idea that communal roosts for wild turkeys act as information sharing centers for food source locations (Ward and Zahavi 1973). We did not investigate the distribution of roost and forage flock locations within the study area, nor did we examine membership overlap between roosting and foraging flocks occurring close together in time. However, it may be that the same factor is driving association in foraging and roosting flocks during the winter for this species: resource acquisition.

In our study, we expected the mean number of daily foraging and roosting flocks to differ between seasons, as wild turkey sociality is strongly seasonal in temperate regions (Healy 1992). The number of foraging and roosting flocks differed seasonally, with more flocks occurring in the winter for both behaviours. Wild turkeys typically form large groups towards the end of fall and remain in these groups until the spring when both male and female dispersal behaviour is driven by reproduction (Healy 1992). As such, we expected to detect fewer, larger flocks during the winter, indicating that more individuals were flocking together. However, our findings were contrary to this expectation. In our study, mortality was highest during the winter and spring. As such, some individuals present in our analysis during the winter were not present in our analysis during the summer, potentially causing a negative bias in the mean number of daily foraging and roosting flocks. Furthermore, it is important to note that we estimated the mean number of daily foraging and roosting flocks based on tagged individuals only, and that there were visibly many more wild turkey foraging and roosting flocks in the region during the same period. The mean number of tagged individuals comprising daily winter foraging and roosting flocks was 2.98 (SD = 3.8, n = 1020) and 2.4 (SD = 3.16, n = 13 675), respectively. The mean number of tagged individuals comprising daily summer foraging and roosting flocks was 2.29 (SD = 2.01, n = 2996) and 1.65 (SD = 1.46, n = 20 387), respectively. However, the proportion of tagged individuals in each foraging and roosting flock varied between flocks and days. As such, we are unable to interpret the number of tagged individuals comprising roosting and foraging flocks as a proxy for mean daily flock size. However, it may be informative for future research to investigate how mean daily flock size changes with the seasons in the context of foraging and roosting sociality in this species.

Interactions are the basis of Hinde's (1976) framework for the study of social structure. The definition of interaction is clear: "the behaviour of one animal is affected by the presence or behaviour of another". Whitehead (2007) describes associations as state measures that are usually more easily measured than interactions. Association is usually defined based on spatial proximity plus a behavioural state measure (Whitehead 2007). In our study, we defined social associations of foraging and roosting based on proximity and time of day as a proxy for a behavioural state measure. Associations can also be reasonably interpreted as occurring "within range of communication" because communication involves the active or passive transmission of information that may change the behaviour of the recipient (Bradbury and Vehrencamp 1998). There is little published information regarding wild turkey conspecific communication and the distance limitations of this communication. Autonomous recording units (ARUs) were used to record gobbling activity during two spring breeding seasons in Georgia and determined that the units were able to record a gobble from a bird on the roost up to 207 m away (Colbert et al.

2015). It is important to note that this is indicative of how far the sound can travel but is not necessarily of the distance at which a wild turkey might be able to detect that sound.

Dyads are typically considered to be in “association” if they are in a situation in which interactions usually take place (Whitehead and Dufault 1999). Associations are state measures and are thus more easily measured than direct interactions (Whitehead 2007). Associations can also be reasonably assigned when individuals are “within range of communication” because communication involves the active or passive transmission of information that may change the behaviour of the recipient (Bradbury and Vehrencamp 1998).

The maximum distance between two GPS-tagged individuals within the same foraging or roosting flock was 156.1 m and it is likely that wild turkeys intentionally socializing with others are using both auditory and visual communication signals to share information. As such, individuals in the same foraging and roosting flocks were likely within range of communication with one another. Furthermore, in most cases, errors in choosing a suitable distance threshold at which to define association will not profoundly affect the subsequent analysis (Whitehead 2007). A too small value will omit some interactions, and a too large one will include noninteracting dyads, but if a large data set is collected and there is no systematic bias (such that might be caused by pairs of individuals who generally interact at ranges just greater than the chosen distance), an informative social model should emerge (Whitehead 2007). Nonetheless, it may be beneficial for future studies aiming to explore wild turkey sociality and associations to investigate communication distance thresholds for this species.

There are several limitations to our methodology that may have impacted our results. For instance, we recognize that the mean VHF-transmitter triangulation error was fairly large (198 m +/- 76.93). This large triangulation error may have introduced both false positives and false negatives into our analysis. For instance, birds may have been classified as associating with another individual when they were in fact not, or vice versa. In our study, 29.16% of individuals (21/72) were fitted with VHF-transmitters and 0.79% of the location estimates (606/76 536) were derived from triangulating these individuals. As such, it is unlikely that our presumably high triangulation error had a substantial impact on our final results or inferences.

Some of the individuals have GPS-transmitters on a one-hour fix schedule, while others have their transmitters on longer fix schedules (4.25 hours). This means that when producing clusters, some individuals with one-hour fix schedules may be contributing more than one GPS-point to the cluster depending on the times at which the fixes occurred, while the birds on longer fix schedules (4.25 hours) may only be contributing one point. Furthermore, it may be that some individuals with shorter fix-schedules contributed points to more than one cluster, and as such, exhibited sociality with multiple groups of individuals during foraging and roosting. Because we investigated the total proportion of days that individuals were together, rather than the number of fixes in close proximity, it is unlikely that the difference in fix schedule had any impact on our results or inferences.

Our findings indicate that wild turkeys are preferentially associating with kin while foraging and roosting during the winter. We did not test for a direct effect on individual fitness. However, it may be that wild turkeys are preferentially associating with kin while foraging and roosting in response to selective pressures to congregate with conspecifics and maximize inclusive fitness. To identify kin-selection, future research could focus on testing hypotheses related to associations between kin and metrics of fitness, such as clutch size (Godfray et al.

1991) and poult survival rate. Future research surrounding kin-selection in wild turkeys could also focus on investigating kinship associated with other behavioural states. For instance, broods spend up to 89% of their time foraging (Chamberlain et al. 2020) and broods from different hens readily combine to form multi-brood flocks (Healy 1992, personal observations by Baici and Adey). Healy (1992) reports that the hen's responses to one another determine whether broods combine, which may be related to genetic similarity between hens. Regional habitat differences may alter wild turkey foraging or roosting behaviour such that kin-selection becomes more or less important in fitness and survival. As such, the measurable relationship between genetic relatedness and social associations may reflect interesting evolutionary differences in behavioural ecology between subspecies.

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Chapter 5: Synthesis

The general objective of my thesis was to further our understanding of the ecology, behaviour, and social structure of Ontario's reintroduced wild turkey population. To do so, in Chapter 2, we employed community-collected data and MaxEnt modelling to estimate the size and distribution of Ontario's contemporary wild turkey population and explore the relationship between population density and habitat features like forest cover and agricultural operations. Building density, the most important environmental feature, was positively correlated with turkey density. Winter severity, represented by snow depth and temperature, was also positively correlated with turkey density. We found that our results and interpretation were similar when attempting to account for spatial bias in the community collected datasets, indicating that this may not always be an essential practice when managing data like these.

In Chapter 3, I narrowed the scope of study from a provincial to a regional context and estimated the size and composition of individual wild turkey home ranges and core use areas in Peterborough County. We found that body size was an important predictors of space use, but the pattern differed depending on sex and season. We also attempted to identify suitable proxies for rank in this species but found no relationship between social status and any morphological measurements, including mass and length. By reviewing extensive camera trap footage of the flocks and interpreting wild turkey behaviour, we were able to identify a robust social structure for our study population.

In Chapter 4, to gain further insight into the social structure of local wild turkey populations, I quantified social interactions between individuals on the basis of proximity and compared this to pairwise genetic relatedness. We learned that this species not only exhibits kin-selection during the breeding season, but also during the non-breeding season while foraging and roosting. In our study, wild turkeys did not exclusively socialize with related conspecifics. However, kin-selection is clearly an important factor shaping wild turkey behaviour and social structure.

The Wild Turkey in Ontario

This work highlights several features of this species in Ontario. As in other parts of their range, wild turkeys are highly gregarious in the province and seem to be significantly limited by winter conditions in northern regions. They are highly abundant in the Mixedwood Plains Ecozone (Chapter 2) and are becoming increasingly common in residential and urban areas. Despite Ontario representing the northern portion of their range, there is some evidence that there may be a relatively high density of wild turkeys in the province when compared to densities reported in other parts of their range (Erikson et al. 2014).

There are several factors commonly associated with successful species reintroductions that may have influenced the seemingly disproportionate success of wild turkeys in Ontario. Wildlife reintroductions are most successful when several factors are accounted for and managed. Predation is often a major factor influencing the reintroduction success of birds, mammals, and reptiles (Fischer and Lindenmayer 2000, Moseby et al. 2011). For instance, White et al. (2012) found that high predation significantly reduced the overall success rates of parrot (psittacine) reintroductions by reducing both post-release survival and the probability of breeding by released birds. Predation was also determined to be the primary cause of the failure of reintroduction attempts for the thick-billed parrot (*Rhynchopsitta pachyrhyncha*; Snyder et al. 1994) and the ultramarine lorikeet (*Vini ultramarine*; Ziembicki et al. 2003).

Furthermore, the high success rate of reintroductions of the kākāpo (*Strigops habroptilus*), kākā (*Nestor meridionalis*), and kākāriki (*Cyanoramphus novaezelandiae),* have been primarily attributed to the release of these species on predator-free islands or in regions with relatively low numbers of introduced mammalian predators (e.g., Elliot et al. 2001, Powlesland et al. 2006, Ortiz-Catedral et al. 2010).

In Ontario, the most frequent cause of wild turkey mortality is mammalian predation, primarily by coyotes (*Canis latrans*). This is true regardless of the turkey's age, proximity to supplemental food, and habitat use (Niedzielski and Bowman 2015). Furthermore, Adey et al. (2023) found that reducing predation risk is likely the most probable explanation for the selection of tall, large diameter trees by this species. It may be that, for Ontario's wild turkey population, the pressure from mammalian predation is driving roost selection behaviours but is not limiting overall population size and distribution.

Human factors also have important implications for the success of wildlife reintroductions (Auster et al. 2021, McCann et al. 2021, Martins et al. 2022). For instance, potential human-wildlife conflict, whether real of perceived (Messmer 2000; Torres, Oliveira, and Alves 2018), can hinder the success of projects and the way wildlife managers interact with individuals affected by conflicts can either prevent or contribute to the escalation of such conflicts (Treves, Wallace, and White 2009; Decker et al. 2014, 2015, 2016). Wild turkeys are commonly involved in both perceived and real human-wildlife conflict. For instance, wild turkeys are often blamed for damage to standing crops like corn and soybeans (Otieno and Frenette 2017), whereas empirical evidence suggests that they subsist primarily on plant parts (e.g. acorns, berries, leaves, ferns), arthropods, and occasionally waste grain (Hurst 1992, Vander Haegen et al. 1989, Vangilder and Kurzejeski 1995, Yarrow 2009, Otieno and Frenette 2017). One study has investigated landowner feelings around reintroduced wild turkeys in northwestern Minnesota USA and found that the majority of survey respondents (89%) reported positive feelings towards turkeys, while 9% were indifferent, and only 2% reported negative feelings for turkeys (Parent et al. 2012). However, they note the importance of understanding landowners' acceptance capacity for this species as the population size is still expected to increase (Parent et al. 2012).

In Ontario, perceptions of wild turkeys likely differ depending on the region and turkey demography. Wild turkeys, often incorrectly, perceived as a crop pest by many farmers, which is common across their range (Groepper et al. 2013). However, increased hunting and wildlife viewing benefits associated with this species likely have a substantial positive impact on the way they are perceived by many Ontarians (Parent et al. 2012). Positive perceptions associated with wild turkey reintroduction projects in Ontario may have played a role in the success of this species re-establishment, persistence, and high contemporary densities. Our investigation into the contemporary size and distribution of Ontario's wild turkey population is the first conducted post-reintroduction. As such, we do not yet know how the population is expected to change over time or how the acceptance capacity may change with it.

Kinship and Dominance Rank

In Chapter 4, we found evidence of association between kin during both the breeding and non-breeding season in that wild turkeys are roosting and foraging with kin more often than with unrelated conspecifics. We were also able to derive a well-defined dominance rank for a small subset of individuals in Peterborough County (Chapter 3). We investigated these phenomena separately. However, associations between kin can contribute to an individual's rank relative to other individuals in the group (Holekamp et al. 2011, Seil et al. 2016, Chakrabarti et al. 2020). For instance, kinship plays an important role in predicting insubordination among dyads of female rhesus macaques (*Macaca mulatta*; Seil et al. 2016), and in maintaining large cooperative coalitions among male Asiatic lions (*Panthera leo persica*; Chakrabarti et al. 2020). This is not always the case, however. Researchers investigating the social structure of a reintroduced population of California condors (*Gymnogyps californianus*) found that a well-defined dominance hierarchy regulated competitive access to food sources, but also identified an absence of kin-based social groups and the role of kinship in defining the hierarchy (Sheppard et al. 2013). Elaborating upon our methods for defining dominance hierarchies for wild turkeys, future research could examine genetic relatedness within the hierarchy to determine the role of kinship, if any, in establishing the pecking order.

Climate Change, Anthropogenic Foods, and Northward Expansion

We are in a period of rapid environmental change, which is causing a shift in species distributions (Walther et al. 2002). Under future climate change scenarios, researchers predict shallower snow depths and warmer temperatures for temperate regions (Pörtner et al. 2022), both climatic factors that may limit wild turkey range expansion (MacDonald et al. 2016, Nguyen et al. 2003, Kane et al. 2007, Gonnerman et al. 2022, Baici and Bowman 2023). Thus, we may expect to see northward range expansions of this species in the future. However, Gonnerman et al. 2022 found that the potential for wild turkey range expansion will be determined in part by the availability of habitat types that allow this species to withstand periods of inclement weather, like conifer forests. In addition, evidence suggests that anthropogenic food sources are likely to continue to play a substantial role in shaping the size and distribution of wild turkeys (Nguyen et al. 2003, Kane et al. 2007, Restani et al. 2009, Baici and Bowman 2023). As such, furthering our understanding of the anthropogenic factors that drive individual wild turkey utilization distribution size and composition will allow us to make better predictions about where Ontario's reintroduced turkey population may expand under climate change and what resources, both natural and anthropogenic, are likely to be predictive of their survival in novel, northern environments. Given that climate change modelling predicts milder temperatures overall but an increase in the occurrence of extreme weather (IPCC 2022), range expansion will also partly be determined by the availability of habitat that allows wild turkeys to withstand sporadic periods of inclement weather (Gonnerman et al. 2022).

Anthropogenic food sources, such as animal agriculture operations and residential bird feeders, can be difficult to quantify as they are often dynamic and can be challenging to locate, even using a combination of aerial imagery and ground-truthing (Jones 2017). We used a combination of ground-truthing and proxies to identify and represent anthropogenic food sources in our studies. As anthropogenic food sources become increasingly important for northern populations, future research may benefit from more robust methods to identify anthropogenic food sources in the landscape and account for their dynamic nature.

Management Implications and Recommendations

In Chapter 2, we present population size estimates that indicate some regions in Ontario may exhibit a higher density of wild turkeys than expected based on densities reported for other geographic regions or previously for the province. Given the variation in size estimates for

Ontario's population over time (Ontario Ministry of Natural Resources 2007, Baici and Bowman 2023), and the propensity for reintroduced (Nichols and Armstrong 2012) and peripheral populations (Thomas et al. 1994, Curnutt et al. 1996) to experience boom-and-bust dynamics, we recommend that the species be monitored using empirical methods such as systematic surveys. Fuller et al. (2020) argue that one fundamental change needed in decision making for complex problems that exist at the intersection of ecological, social, and economic spheres, such as wildlife management, is widespread recognition that an explicit framework is useful to arrive at defensible and robust decisions. As such, in addition to ongoing monitoring of Ontario's wild turkey population, we also suggest that an explicit framework be developed to guide future wild turkey reintroductions and harvest management, especially given their history of regional overharvest and extirpation. Historical reintroductions, introductions, and anticipated range shifts in combination with the continued harvest of wild turkeys highlights the need for future empirical studies investigating habitat suitability, resource use, and persistence in northern environments.

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APPENDIX A

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APPENDIX B

Chapter 2: Supplemental Materials

1. Systematic Survey Methods

Peterborough County: Aerial Survey

From a helicopter (Eurocopter EC130), we flew pre-determined line transects over a 380km^2 survey area within Peterborough County. The survey was conducted over three days in March 2018. To allow for refueling during the survey, the survey area was divided into three survey blocks. One survey block was 8 x 15 km and the other two were 10 x 13 km in size, each covering 120 – 130 km^2 of survey area and ~ 200 km of survey flight lines.

Survey lines consisted of consecutive, parallel, NW-SE oriented 8-10 kilometer long transects, which in most cases followed or ran parallel to local concession roads. Line transects were spaced 700 meters apart requiring a wild turkey detection range of 350 meters. This detection range was tested prior to surveying using a small flock of decoy turkeys placed at measured distances from known locations. During the survey we maintained a cruising altitude of ~75 meters AGL and cruising flight speed of 80 km/h (43 knots). Our survey consisted of three observers: a front-right observer and navigator, a back-right observer, and a back-left observer. We observed a total of 33 flocks and 522 individuals using this method (Table A2.2).

Peterborough County: Community Science Survey

Wild turkey observations within Peterborough County were requested from the public from 1 December 2017 to 31 March 2018. We requested that observations be submitted through two commonly used community science reporting platforms; eBird and iNaturalist. In iNaturalist a project titled “Peterborough Wild Turkey Count” was created to both advertise this initiative and curate observations. Additionally, we advertised this survey through various social media platforms, email listservs, relevant local organizations, such as the Ontario Federation of Anglers and Hunters and the Peterborough Field Naturalists Club, the local newspaper, and using posters located in local college and university campuses and public libraries. A total of 42 flocks and 469 individuals were observed and reported by citizen scientists (Table A2.2). Of these, the majority were reported to eBird (38/42 = 90.4%). Two were reported to iNaturalist, and although not requested, one was reported to us directly via email correspondence.

Peterborough County: Opportunistic Road Survey

We conducted an opportunistic road survey from 1 December 2017 to 31 March 2018. During this period, the location of any flocks observed within the study area was recorded along with the number of individuals in the flock. This survey was opportunistic in that all observations were made during regular travel between radiotelemetry and/or trapping sites. All roads within the study area were driven at least once however no effort was made to ensure equal sampling across the study area. A total of 35 flocks and 507 individuals were observed using this method (Table A2.2).

To ensure that density estimates were comparable across survey methods, all estimates were produced for a 422km^2 region within the larger study area in which all three survey methods were employed.

Peterborough County: Detecting Duplicate Observations

Because all three survey methods were employed during the same time-period and within the same survey area it is possible that some flocks were detected multiple times using the same method and that some flocks were detected by more than one method. To estimate the number of flocks detected across survey methods we first removed duplicate observations within each survey method.

In ArcMap (version 10.7.1, ESRI 2011), the reported location of each flock observation was plotted and a buffer equivalent to the average home range size of wild turkeys during this time-period was added around each point. Home range size was estimated by calculating 95% minimum convex polygons (MCP's) using GPS data derived from wild turkeys that were previously trapped and tagged in the area. Birds were captured at baited locations using a rocket net (Grubb 1988) following methods outlined by Niedzielski and Bowman (2014; 2016). MCP's were calculated in R Studio (R Core Team version 1.4), using the "sp" (Pebesma and Bivand 2005; Bivand et al. 2013) and "adehabitatHR" (Calenge 2006) packages. The average home range size of tagged individuals during this study period was 1.50km^2 (n=9). The buffer tool in ArcMap adds a circular buffer at a specified distance around each point. Thus, we added a buffer equal to the radius of a 1.50km^2 circle; 0.69km.

We considered an observation to be a duplicate if the distance between it and any other observation was less than that of the radius of the circular buffer. In other words, we considered an observation to be a duplicate if it was located inside the buffer of another observation. We retained the observation reported earliest in the study period and removed all duplicate observations of the same flock. In cases where the number of individuals reported differed between duplicate observations, we retained the observation that reported the highest number of individuals.

We then removed duplicate observations across survey methods using the same methodology (Tables A2.2 and A2.3). Of duplicate observations across survey methods, we retained the observation reporting the highest number of individuals regardless of survey method.

Bruce Peninsula: Census

We conducted ad hoc road surveys during winter 2012 (January through March inclusive). The surveys were conducted during wild turkey capture efforts. All roads on the Bruce Peninsula north of Ferndale were travelled multiple times throughout the winter to identify flocks for trapping and also to track transmittered birds. Researchers were able to census all flocks and birds in the study area.

2. Supplemental Tables

Table A2.1. Home ranges, represented by 95% kernels, estimated for 45 GPS-tagged individuals (24 M, 21 F) in Peterborough County and the Bruce Peninsula. Kernels were estimated from GPS fixes collected during the winter season (21 December to 20 March) in 2011 - 2012, 2017 - 2018, 2018 - 2019, and 2019 - 2020.

Bird ID	**Winter home range area (km²)**				**Sex**
	2011 - 2012	2017 - 2018	2018 - 2019	2019 - 2020	
46318		3.733346	0.321246		M
46319		9.80E-04	0.004378		M
46320		1.094468	0.941199		M
46321		0.091661	0.362633		M
46322		1.698507	2.897565		M
46323.2			0.82327		M
46323		2.472596			M
46324.2			4.485722		M
46324		0.67655			M
46325.2			3.853413		M
46325		1.40637			M
46326		2.091556	6.581876		M
46327		1.155332			M
46328		1.48681			M
46329.2			1.040931		M
46329		1.527266			M
46330		0.066322	0.062175		F
46331		1.427152	0.182978		F
46332.2			1.866992		M
46332.3			2.936232		M
46338		1.388389	0.585791		F
46371			1.468284	1.820743	F
46373			3.477688	1.018712	F
46374			3.043664	1.971639	F
46375			1.268942	0.138026	M
46376			1.068082	0.294379	M
46377			3.431996		F
46379			0.402614		F
46380			2.688241	0.076966	F
46381.2				1.328696	F
46381			3.464302		F
46382			1.153459	0.672541	M
46383			3.448104	0.984083	F
46384			3.556854	0.251519	F
46385			3.794423		F
46719				0.661847	F

46721		1.069605	F
MNR15	1.235541		F
MNR20	1.797744		F
MNR23	1.928839		F
MNR24	3.05874		M
MNR30	4.201094		M
MNR31	5.127398		M
MNR34	3.428839		F
MNR37	3.165882		M

Table A2.2 The number of flocks and individuals observed in Peterborough County (422km^2) using each survey method and in total before removing duplicate observations.

Survey Method	**Number of Flocks Observed**	**Number of Individuals Observed**
Aerial	33	522
Citizen Science	42	469
Road	35	507
Total	110	1498

Table A2.3. The number of flocks and individuals detected in Peterborough County ($422km^2$) using each survey method and in total after removing duplicate observations across all survey methods.

Survey Method	Total Number of Unique Flocks Detected	Flock density (number of unique flocks per km^2)	Proportion of total flocks detected (%)	Total Number of Unique Individuals Detected	Individual density (number of unique individuals per km^2)
Aerial	30	0.07	56.5%	493	1.16
Citizen Science	12	0.02	22.6%	228	0.54
Road	11	0.02	20.7%	184	0.43
Total	53	0.12		905	2.14

Table A2.4. The number of flocks and individuals detected using a road survey in the Bruce Peninsula ($1140km^2$).

Survey Method	Total Number of Unique Flocks Detected	Flock density (number of unique flocks per km2)	Total Number of Unique Individuals Detected	Individual density (number of unique individuals detected per km2)
Road	10	0.008	228	0.2

Table A2.5. A summary of wild turkey density estimates for each state. Derived from Erikson, et al. 2014.

State	Density Estimate (number of turkeys per km²)
Minnesota	0.530189621
Illinois	0.880355375
Indiana	1.297353399
Louisiana	1.311045559
Michigan	1.328771219
Maine	1.394523327
Delaware	1.447876448
West Virginia	1.537567909
Vermont	1.645548791
Rhode Island	1.65471594
Arkansas	1.775358031
Pennsylvania	1.791795929
Missouri	1.795057646
Virginia	1.883948757
New Hampshire	1.930501931
Ohio	2.003886325
Ohio (Donohoe et al. 1983)	0.7 - 2.0
Mississippi	2.013725553
Mississippi (Lint, et al. 1995)	0.34 – 0.87 (adult males only)
Iowa	2.08263912
South Carolina	2.106002106
Kentucky	2.151210545
North Carolina	2.225512938
Massachusetts	2.400576138
Tennessee	2.401042471
New York	2.574002574
Wisconsin	2.757881632
Wisconsin (Kubisiac et al. 1997)	0.09 - 1.1
Maryland	2.831402831
New Jersey	3.301277451
Georgia	3.391477773
Connecticut	3.715498938
Alabama	4.207504208
Mean	2.078941683
Standard deviation	0.817567703
Low density estimate (mean - 1 SD)	1.26137398
High density estimate (mean + 1 SD)	2.896509385

Table A2.6. Maxent and post-hoc presence thresholds identified from Figure A3.1, the associated area proportion estimates for each threshold and population size estimates. Model 2: Building density + snow depth + agriculture + road density + forest cover.

Presence threshold	Predicted area	Mean density estimate
Minimum Training Presence: 0.0364	0.991	19487.3
Post-hoc 1: 0.1	0.7029	16146.38
Post-hoc 2: 0.325	0.5454	18638.77
Tenth Percentile Training Presence: 0.3976	0.5099	19845.75

Table A2.7. Summary statistics from Welch's Two Sample t-test of the effect of data thinning on habitat suitability inside wild turkey winter home ranges (95% kernels), winter core use areas (50% kernels), available habitat (buffered 100% MCP's), and winter available habitat (buffered 100% MCP's).

Region	Year	Spatial Scale	t	P	Degrees of freedom	95% CI
Peterborough	Year 1 (2016 -2017)	Annual available habitat	0.61829	0.5368	352.77	-0.01, 0.02
Peterborough	Year 2 (2017 -2018)		-0.40922	0.6824	1815.9	-0.01, 0.01
Peterborough	Year 3 (2018 – 2019)		-1.015	0.3112	230.06	-0.03, 0.01
Bruce Peninsula	Year 5 2012		-1.3738	0.1707	246	-0.09, 0.01
Peterborough	Winter 1 (2016 -2017)	Winter available habitat	0.51253	0.6092	124.78	-0.02, 0.03
Peterborough	Winter 2 (2017 -2018)		0.085533	0.9319	185.66	-0.02, 0.02
Peterborough	Winter 3 (2018 – 2019)		-0.40543	0.6858	127.89	-0.03, 0.02
Bruce Peninsula	Winter 5 2012		-1.5016	0.1347	207.99	-0.1, 0.01
Peterborough	2017	Winter home range (95% kernels)	0.52	0.6	20.61	-0.02, 0.03
Peterborough	2018		0.96	0.34	46.64	-0.02, 0.05
Peterborough	2019		0.65	0.52	20.45	-0.03, 0.06
Bruce Peninsula	2012		-1.02	0.32	11.79	-0.04, 0.01
Peterborough	2017	Winter core use (50% kernels)	1.61	0.11	21.025	-0.004, 0.03
Peterborough	2018		1.6	0.12	43.25	-0.005, 0.04
Peterborough	2019		1.27	0.22	19.44	-0.01, 0.05
Bruce Peninsula	2012		-0.68	0.51	7.79	-0.04, 0.02

Table A2.8. Mean winter and annual wild turkey home range sizes estimated using 95% kernels. Polygon buffers represent the diameter of a circle equivalent in area to the mean winter and annual home range sizes for each year. Buffers were added to winter available habitat polygons and annual available habitat polygons.

Year	Mean winter home range size (km^2)	Winter available habitat MCP buffer (km)	Mean annual home range size (km^2)	Annual available habitat MCP buffer (km)
Year 1 (2016 – 2017)	1.32 (SD = 0.68, n = 10)	1.3	11.13 (SD = 20.06, n = 19)	3.76
Year 2 (2017 – 2018)	1.99 (SD = 1.8, n = 24)	1.59	2.68 (SD = 2.74, n = 21)	1.85
Year 3 (2018 – 2019)	1.12 (SD = 1.1, n = 13)	1.19	4.68 (SD = 10.02, n = 10)	2.44
Year 5 (2011 – 2012)	3 (SD = 1.29, n = 8)	1.95	10.21 (SD = 7, n = 5)	3.6

3. Supplemental Figures

Figure A3.1 A poster developed to create awareness about our community science campaign. This poster was shared virtually with field naturalist groups and nature clubs across the province. It was also shared on various social media platforms including Facebook, Instagram, and Twitter.

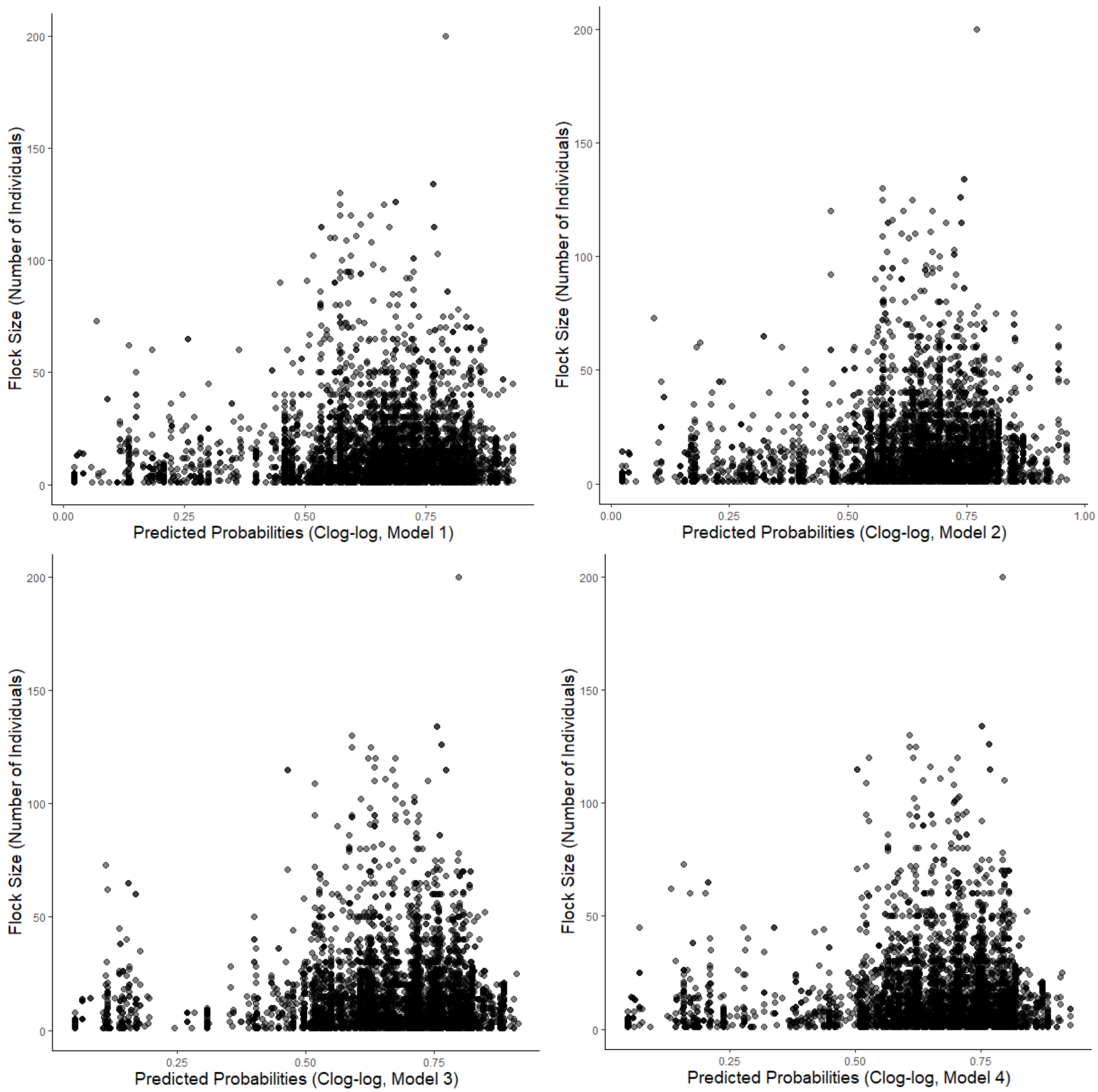
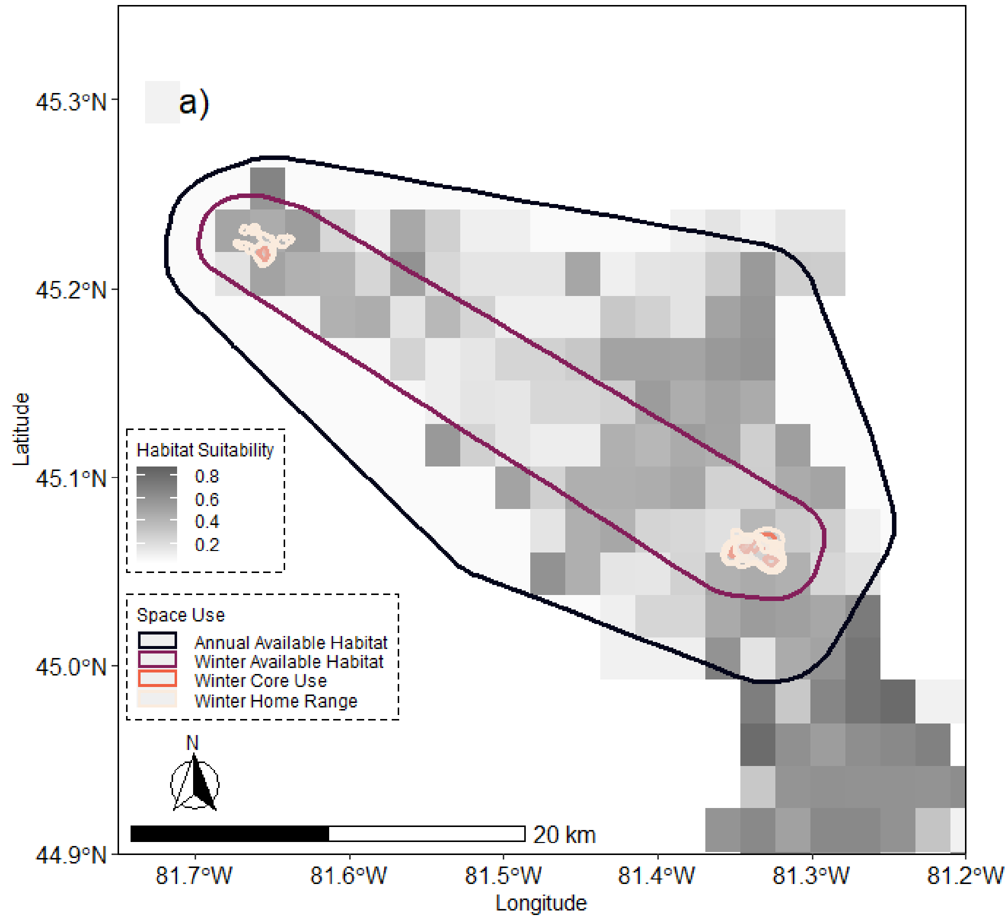
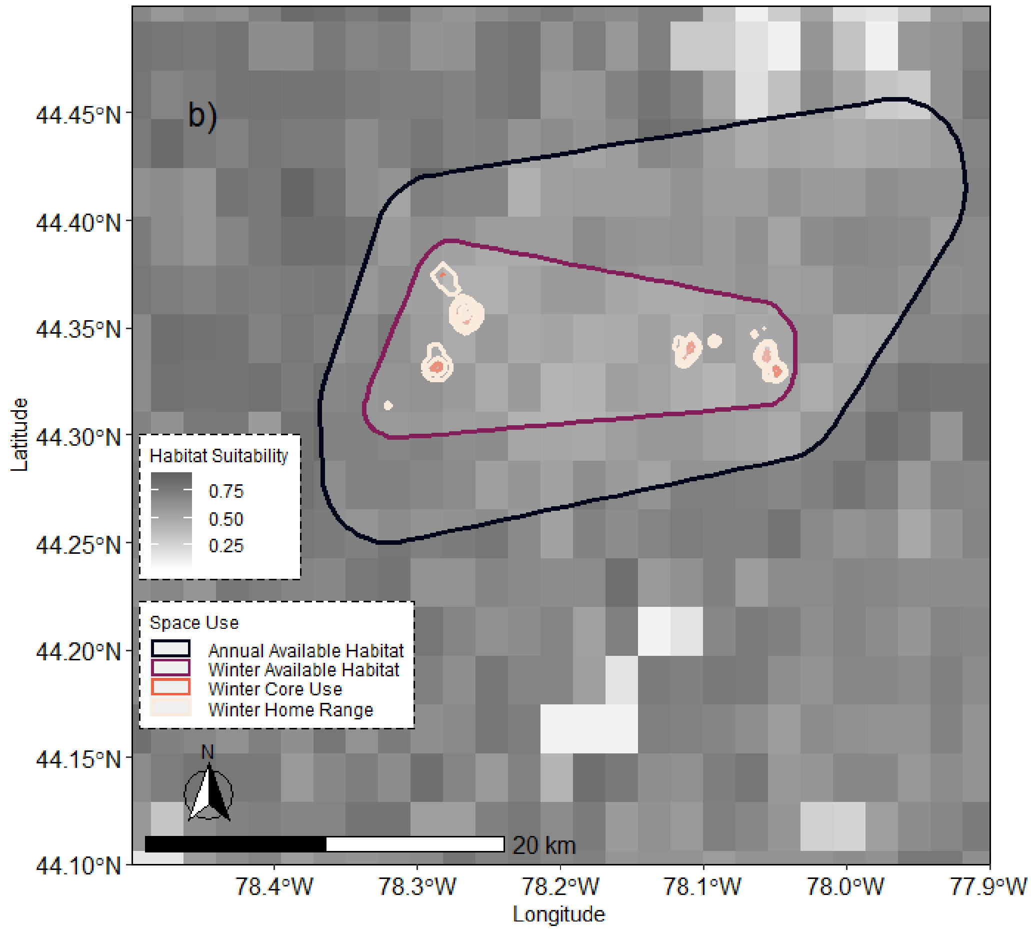


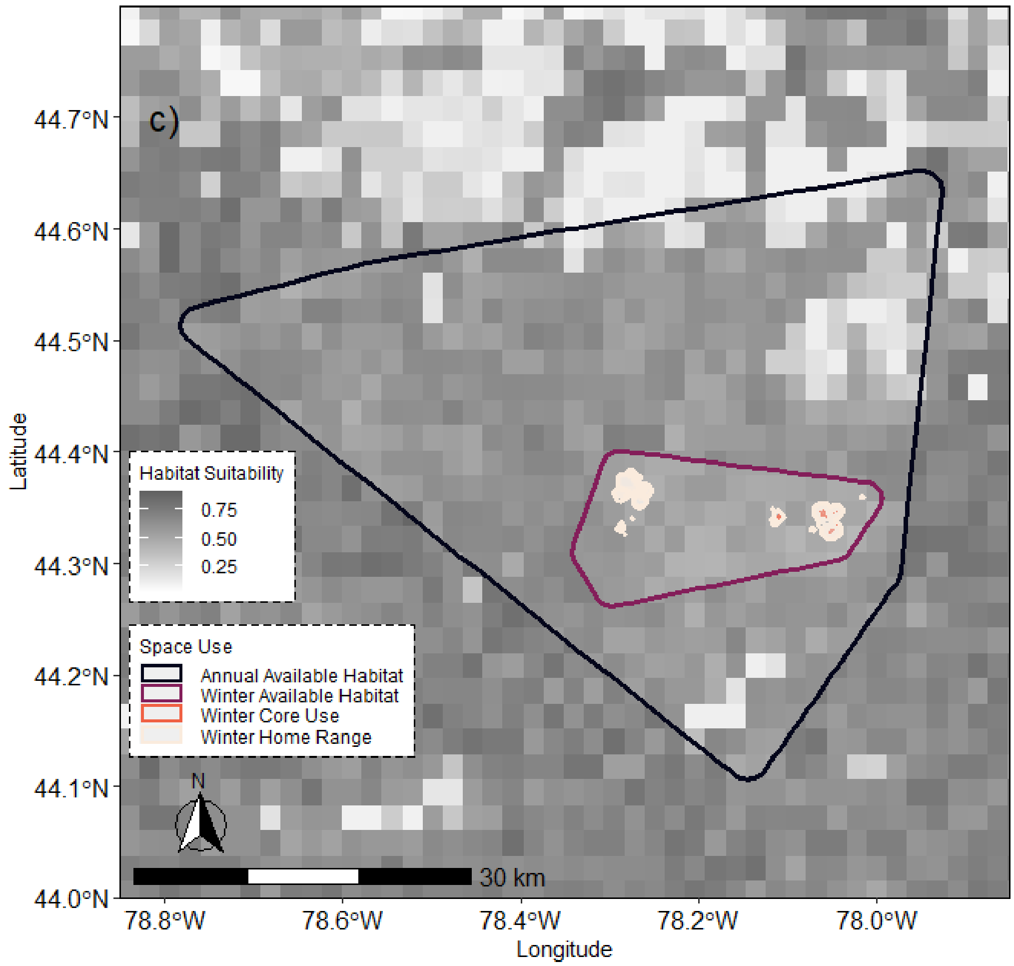
Figure A3.2 Scatterplots depicting the predicted probabilities in relation to flock size for Models 1-4 (Chapter 2). Figures include replicates in the dataset and were used only to identify potential habitat suitability thresholds using a visual assessment, post-hoc. Potential habitat suitability thresholds were identified at 0.1 (post-hoc 1) and 0.325 (post-hoc 2). Top left: Model 1, top right: Model 2, bottom left: Model 3, bottom right: Model 4.



a)
45.3°N
45.2°N
45.1°N
45.0°N
44.9°N
Latitude
81.7°W
81.6°W
81.5°W
81.4°W
81.3°W
81.2°W
Longitude
Habitat Suitability
0.8
0.6
0.4
0.2
Space Use
Annual Available Habitat
Winter Available Habitat
Winter Core Use
Winter Home Range
N
20 km



b)
Habitat Suitability
0.75
0.50
0.25
Space Use
Annual Available Habitat
Winter Available Habitat
Winter Core Use
Winter Home Range
N
20 km
44.45°N
44.40°N
44.35°N
44.30°N
44.25°N
44.20°N
44.15°N
44.10°N
Latitude
78.4°W
78.3°W
78.2°W
78.1°W
78.0°W
77.9°W
Longitude



c)
Habitat Suitability
0.75
0.50
0.25
Space Use
Annual Available Habitat
Winter Available Habitat
Winter Core Use
Winter Home Range
N
30 km
44.7°N
44.6°N
44.5°N
44.4°N
44.3°N
44.2°N
44.1°N
44.0°N
Latitude
78.8°W
78.6°W
78.4°W
78.2°W
78.0°W
Longitude

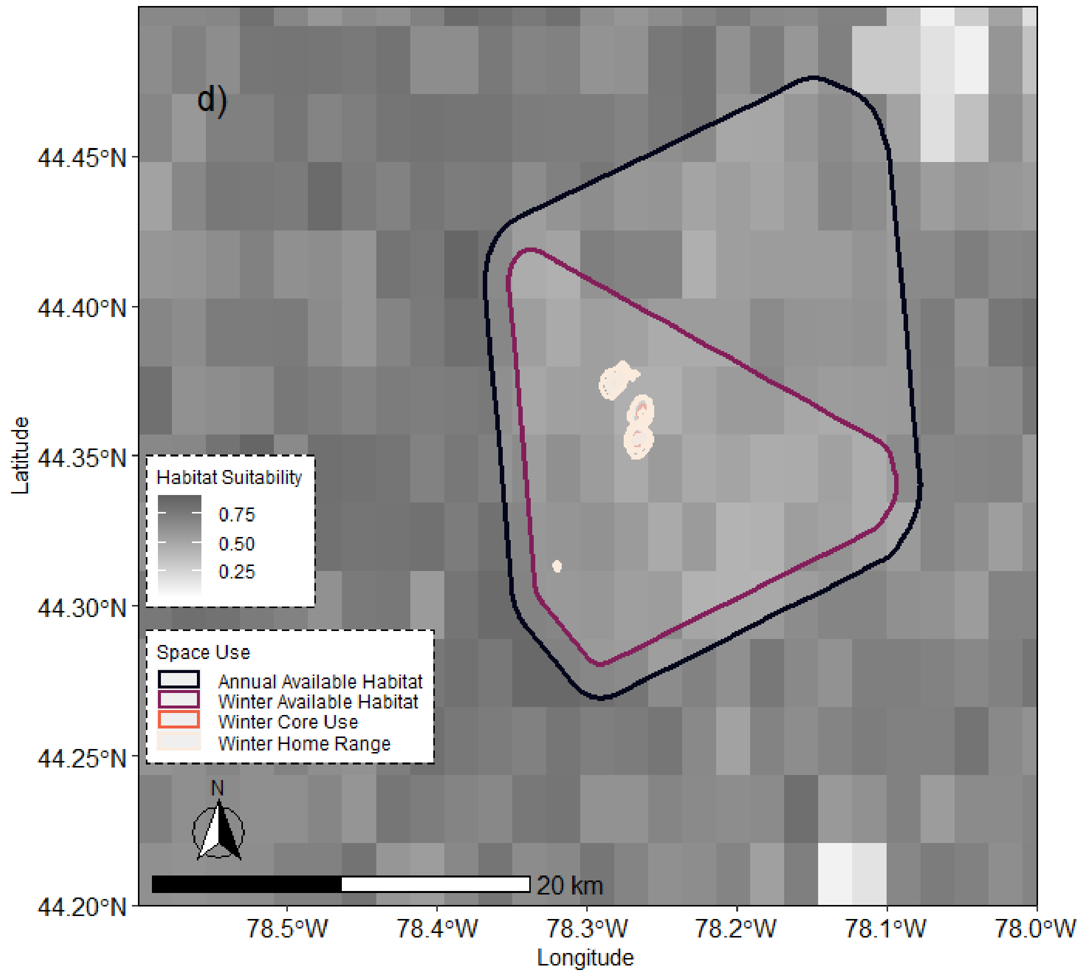


Figure A3.3. Polygons depicting annual available habitat (buffered 100% MCPs around pooled GPS locations), winter available habitat (buffered 100% MCPs around winter GPS locations), wild turkey home ranges (95% kernels), and wild turkey core use areas (50% kernels). a) Bruce Peninsula in 2012, b) Peterborough County in 2017, c) Peterborough County in 2018, d) Peterborough County in 2019. The habitat suitability raster depicted was generated using un-thinned wild turkey observations and pixel size is 2.27km^2.

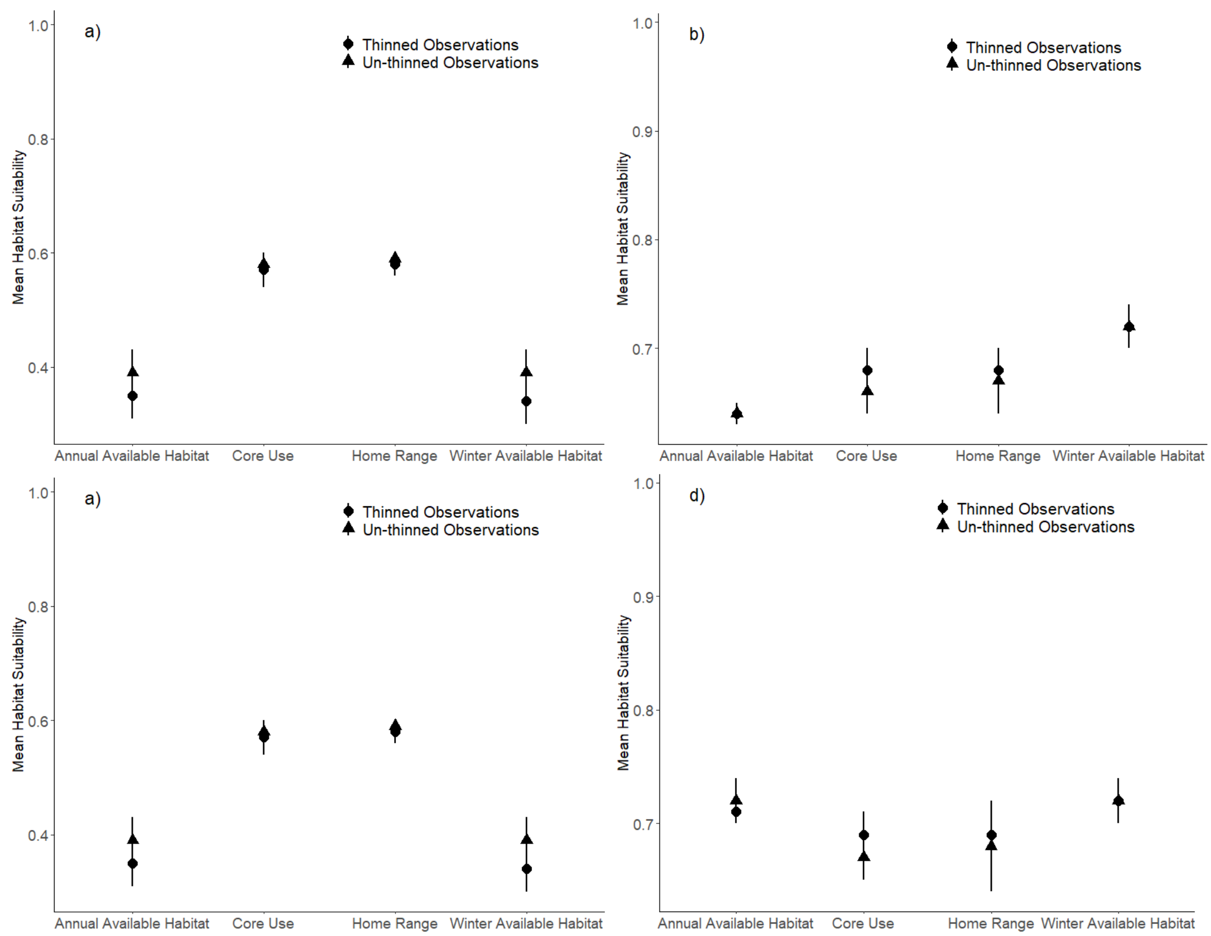


Figure A3.4. Mean habitat suitability inside polygons representing four levels of wild turkey space use: annual available habitat (buffered 100% MCP's), winter available habitat (buffered 100% MCPs), core use (50% kernels), and home range (95% kernels). Data is parsed by year and region: a) Bruce Peninsula 2012, b) Peterborough County 2017, c) Peterborough County 2018, d) Peterborough County 2019. Points represent mean habitat suitability values inside each polygon and lines represent 95% confidence intervals calculated from un-thinned data. 95% CIs did not differ between thinned and un-thinned data as error bars overlapped substantially. As a result of survival differences between individuals, sample sizes vary with space-use category, year, and region. It ranges from 8 individuals on the Bruce Peninsula in 2012 (year 5) to 28 individuals in Peterborough County in 2017-2018 (year 2).

APPENDIX C

Chapter 3: Supplemental Materials

1. Supplemental Tables

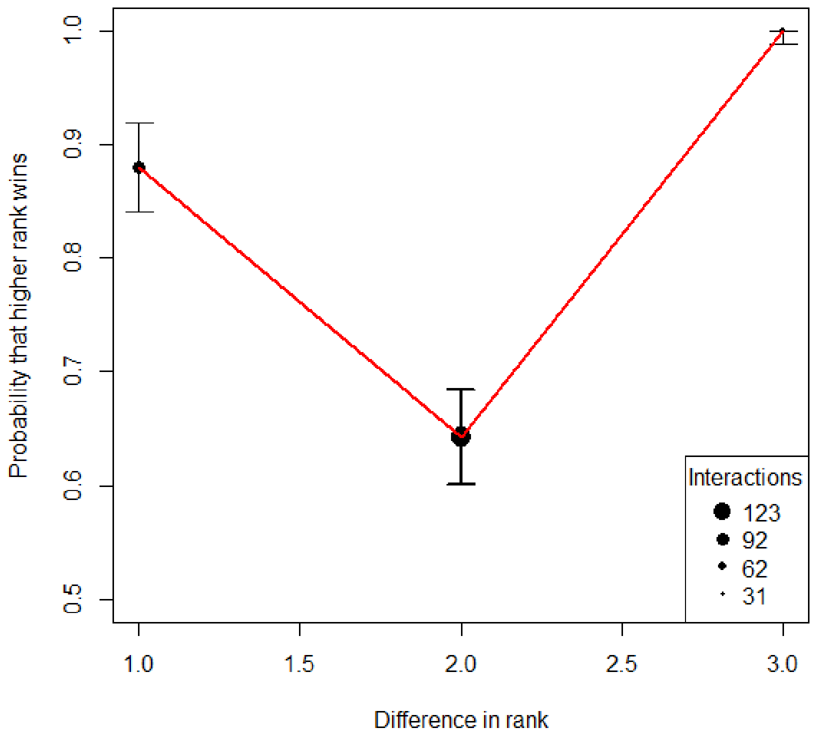
Supplemental Table 1. Correlation coefficients for each pair of numerical variables used to model wild turkey home range size (95% kernels).

	Utilization distribution size (m^2)	Utilization distribution size (km^2)	Crop Proportion	Pasture Proportion	Livestock Proportion	PC1	PC2
Utilization distribution size (m^2)	1.0	1.0	0.04	-0.03	0.10	-0.14	-3.05e-05
Utilization distribution size (km^2)	1.0	1.0	0.04	-0.03	0.10	-0.14	-3.05e-05
Crop Proportion	4.14e-02	4.14e-02	1.0	-0.38	0.43	-0.17	-1.88e-01
Pasture Proportion	-3.2e-01	-3.2e-01	-0.38	1.0	-0.03	0.21	3.09e-01
Livestock Proportion	1.0e-01	1.0e-01	0.43	-0.03	1.0	-0.11	-1.32e-01
PC1	-1.4e-01	-1.4e-01	-0.17	0.21	-0.11	1.0	1.73e-01
PC2	-3.05e-05	-3.05e-05	-0.18	0.30	-0.13	0.17	1.0

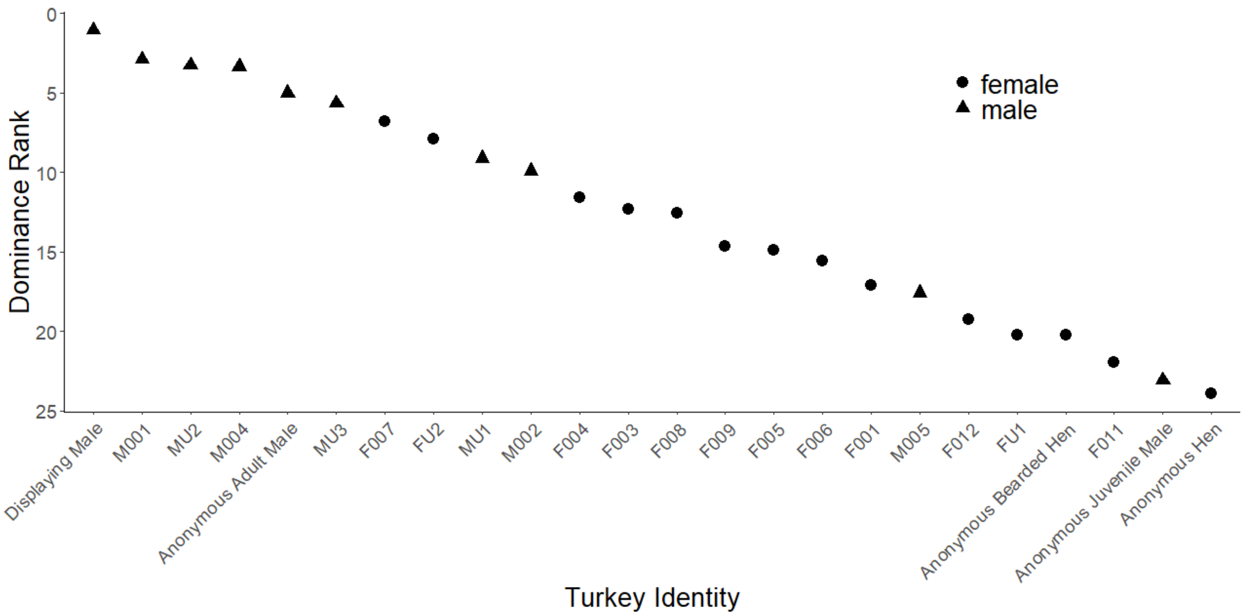
Supplemental Table 2. Correlation coefficients for each pair of numerical variables used to model wild turkey home range size (50% kernels).

	Utilization distribution size (m^2)	Utilization distribution size (km^2)	Crop Proportion	Pasture Proportion	Livestock Proportion	PC1	PC2
Utilization distribution size (m^2)	1.0	1.0	0.10	0.13	-0.06	-0.10	0.05
Utilization distribution size (km^2)	1.0	1.0	0.10	0.13	-0.06	-0.10	0.05
Crop Proportion	0.10	0.10	1.0	-0.23	-0.01	0.03	-0.21
Pasture Proportion	0.13	0.13	-0.23	1.0	-0.07	0.12	0.26
Livestock Proportion	-0.06	-0.06	-0.01	-0.07	1.0	0.008	-0.30
PC1	-0.10	-0.10	0.03	0.12	0.008	1.0	0.28
PC2	0.05	0.05	-0.21	0.26	-0.30	0.28	1.0

2. Supplemental Figures



Supplemental Figure 1. The shape of the dominance hierarchy among four wild turkey sex and age classes. The probability of the higher ranked individual winning as it relates to the difference in rank between the two individuals. The error bars represent the 95% confidence interval and the red line depicts the shape of the hierarchy.



Supplemental Figure 2. The dominance rank of tagged wild turkeys (n = 19 individuals, n = 396 interactions). Five additional categories are included; "displaying male" to represent untagged displaying males, and four anonymous categories to represent untagged individuals in each sex/age class that were identifiable by other means (i.e., broken tail feathers, walking with a minor limp). Anonymous individuals are only included when they were involved in an interaction with an identifiable individual.

APPENDIX D

Chapter 4: Supplemental Materials

1. Supplemental Tables

Supplemental Table 1. A summary of the agonistic interaction outcomes for each dyad category. The winner of each interacting pair is italicized.

Interacting pair (*winner* vs loser)	Number of interactions	Proportion of total interactions
Bearded hen **vs hen**	38	0.039
Bearded hen **vs jake**	1	0.001
Hen **vs bearded hen**	10	0.010
Hen **vs hen**	**581**	**0.596**
Hen **vs jake**	55	0.056
Jake **vs bearded hen**	6	0.006
Jake **vs hen**	97	0.099
Jake **vs jake**	**98**	**0.100**
Tom **vs bearded hen**	2	0.002
Tom **vs hen**	45	0.046
Tom **vs jake**	37	0.037
Tom **vs tom**	**4**	**0.004**
	n = 974	