Nest-site Characteristics, Nest Distribution, and Home Range of Urban Merlins (*Falco columbarius*) in Winnipeg, Manitoba

By

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ABSTRACT

The development of urban forestry has created environments that have become ideal habitats for wildlife that exhibit generalist behaviours. Understanding the habitat features that urban wildlife rely upon is essential to protect species and to encourage successful human and wildlife cohabitation. Although not previously researched, Merlins (*Falco columbarius*) are commonly observed in the urban area of Winnipeg, Manitoba, Canada. Nests were documented throughout the city between 2017 and 2019. This research was conducted to understand how Merlins utilize this urban environment by developing baseline data on the nest site characteristics, nest density, nest site distribution, and breeding home range. To examine this, micro and macro-scale habitat features were measured using a subset of the total recorded nests (n=81). This subset included 30 occupied and 30 unoccupied nest sites that were surveyed and analyzed using generalized linear model (GLM) methods to determine nest site and habitat selection. Adult individuals (n=4) were tracked during the nesting period using high-precision GPS tags, and spatial movement patterns were analyzed using continuous movement methods (ctmm) to determine home range sizes. I found that all Merlin nests documented through the study were in unused American Crow (*Corvus brachyrhynchos*) nests located in mature spruce trees. The habitat selection analysis revealed nest trees were taller and fewer coniferous trees were found within 25 m of occupied nests as compared to unoccupied sites. Nest density of Merlins within Winnipeg was 6.73 nests/100 km^2 and the spatial distribution of nests was non-random, with significant clustering patterns. Mean nearest neighbour distances between nests was 1.67 km. The mean home range size of four breeding male Merlins was 1.79 km^2. During the 2018 field season, an apparent West Nile virus (WNV) outbreak occurred, and the impacts of this outbreak were documented within this study. Juvenile mortalities occurred at 71% of nests. A sample of 32 Merlins from the Winnipeg area presumably deceased due to WNV were necropsied at the Canadian Cooperative Wildlife Health Centre in Saskatoon, Saskatchewan, resulting in WNV positive results in 75% of the Winnipeg sample. Based on nest site selection results in this study, management for Merlins within Winnipeg should include the maintenance of mature spruce trees by both city planners and private property owners. Continued monitoring through community citizen science could provide important data to offer further insight into limiting factors and the ecology of Merlins in the urban environment of Winnipeg.

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I dedicate this thesis to my Solly, Poppa, Grandma and the birds. You all make me want to be my best self, thank you for all that you do.

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CHAPTER ONE

GENERAL INTRODUCTION

Urban environments are heterogeneous landscapes that contain a diversity of anthropogenic structures, linear features, fragmented green space and vegetation, and are made up of both native and non-native species (Lepczyk et al. 2017). Wildlife uses urban spaces in a variety of ways; some species avoid urban environments, some adapt, and others exploit urban areas (Blair 1996). Avian species have responded to urbanization in this manner, which has resulted in a shift of population dynamics, resulting in decreased species diversity and increased abundance for those that remain within urban environments (Isaksson 2018). The abundance of avian species provides resources that can aid in the adaptation and exploitation of environments by predator species like raptors. Although raptor response to urban environments is species specific, there are a number of species that have colonized and even thrived in urban habitats (Boal and Dykstra 2018). For those that have utilized urban environments successfully, there are documented trends of urban raptors maintaining higher population densities, greater reproductive success, decreased home ranges and changes in migratory patterns resulting in reduced energy expenditure (Warkentin and Oliphant 1990; Boal and Dykstra 2018).

Concentrated research attention on urban raptors has taken place over the past 35 years (Boal and Dykstra 2018). Yet, many information gaps remain, particularly using comparative studies of rural and urban inhabitants. However, the data that do exist indicate that characteristics contributing to the success and likelihood of a raptor inhabiting an urban environment includes species adaptability, abundance, and generalist behaviours (Boal and Dykstra 2018). While urban habitats offer many available resources, there are also limiting factors associated with the habitat. Urban environments are made up of densely distributed anthropogenic structures increasing the likelihood of building and window collisions, collisions with vehicles (especially when young have not honed their flight skills yet), predation by domestic animals like cats (Hager 2009), increased disease transmission, and nest disturbance or mortality caused by humans (Poppleton 2016; Kettel et al. 2018). Although some raptor species can adapt within urban environments such as Peregrine Falcons (*Falco peregrinus*), urbanization can negatively impact the success and status of other raptor species that do not exhibit generalist behaviours and rely on specific prey or habitat that may be reduced by urbanization (Boal and Dykstra 2018).

The Merlin (*Falco columbarius*) is a small member of the family Falconidae and is one of the raptor species that have thrived in urban environments. Since their colonization of urban environments in the 1960s (De Smet 2003) they have been a study species of interest. They are adaptable and abundant – they are widespread across North America, tolerant to human presence, and are opportunistic hunters. Merlins adapt to their surroundings and primarily select a diversity of avian prey based on seasonal abundance (Sodhi and Oliphant 1993; J. Josephson-Laidlaw pers. obs) but have been known to hunt insects, small mammals (Konrad 2004) and bats (Haak et al. in press).

True to most raptor species, Merlins are sexually dimorphic. Females are up to 30% larger than males (Warkentin et al. 2020) and distinct adult plumage differences are observed. There are nine recognized Merlin subspecies in the world. However, there are only three subspecies found in North America - the Richardson's Merlin (*Falco columbarius richardsonii*), Taiga Merlin (*Falco columbarius columbarius*), and Black Merlin (*Falco columbarius suckleyii*). Merlins often lack the prominent moustache found on American Kestrel (*Falco sparverius*) and have dark tails with bands and terminal bands that vary from light grey to white (Warkentin et al. 2020). Tail bands vary per subspecies but generally there is large variation between individuals. Subspecies hybridization of Taiga Merlins with both Black and Richardson's occurs due to overlapping breeding ranges (Warkentin et al. 2020; L. Oliphant personal communication; Haak et al. in press). However, Merlin plumage is often so variable that subspecies distinctions on plumage alone can be unreliable (Sealy 2004). The majority of Merlin plumage observed across North America is described to more frequently display a diversity of physical traits rather than traits that are observed in one single subspecies (Haak et al. in press).

Richardson's Merlins are the largest of the three and have historically been found in the central prairie regions of North America. Their larger wingspan and lighter external features complement open landscape hunting over prairie habitat (Hodson 1970; Trimble 1975). They are believed to primarily overwinter around Wyoming and Colorado (Warkentin et al. 1990) but have been reported as far south as Mexico (Haak et al. in press). Black Merlins are the smallest of the three and historically have been found in coastal habitats of western North America (Trimble 1975; Warkentin et al. 2020). They are typically a non-migratory species, but recent evidence has documented Black Merlins in several inland states, including New Mexico (Haak et

al. in press). Taiga Merlins inhabit the greatest diversity in their geographical range, commonly overlapping breeding ranges of both Richardson's and Black Merlins and are documented to exhibit the largest migrational span of the three – overwintering as far south as Peru (Trimble 1975). The documented habitat range of all three subspecies has demonstrated trends of expansion beyond the historical range maps (Haak et al. in press; Cava et al. 2014). The broad geographical range of all three subspecies is displayed in Figure 1.

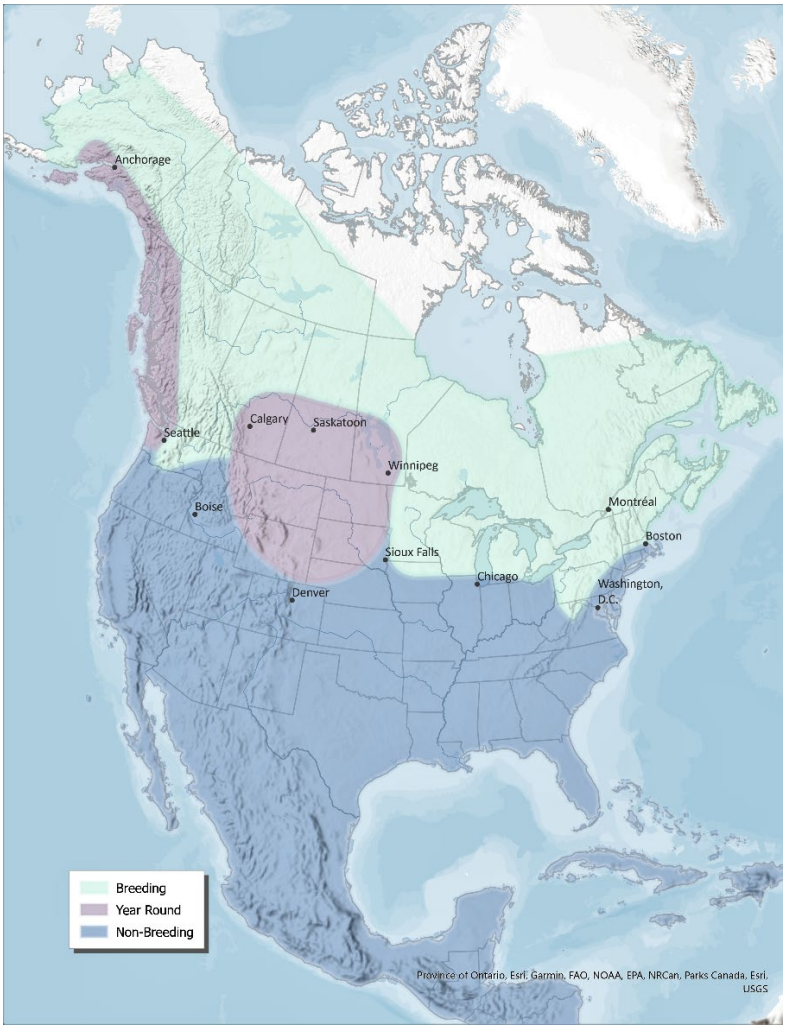


Figure 1. Range map of North American Merlins including breeding, year-round and non-breeding locations (Map produced by Kim Mauch 2020).

Merlins are currently listed as 'Not at Risk' by The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) (Government of Canada 2021) and 'Least Concern' by the International Union for Conservation of Nature (IUCN) (IUCN 2018). Populations across North America appear to be stable or growing (IUCN 2018; National Audubon Society 2020) and within Manitoba, populations have increased by 5.5% between 2005 and 2015, based on the reports by the Breeding Bird Survey (Sauer et al. 2017). However, Merlins were subjected to the impacts of pesticide application that began in 1940. Dichlorodiphenyltrichloroethane (DDT), and dieldrin are persistent chemicals that were used to reduce fungi growth and insect presence in agricultural crops (Cade 1988). The use of these chemicals and DDT's breakdown product, dichlorodiphenyldichloroethylene (DDE), created biomagnification processes in the food chain affecting predators such as Merlins due to their reliance on smaller avian and invertebrate species (Fyfe et al. 1976). The impacts of these chemicals were a reduction of shell-thickness, resulting in limited or failed nest success for many raptor species. Targeted studies on Merlins found higher concentrations of DDE to be correlated to eggshell thinning, hatch failure, decreased carrying capacity in prey base, and reduced to absent nest site defense (Hodson 1970; Fox and Donald 1980; Cade 1988). Despite these impacts, research by Fyfe et al. (1976) and Cade (1988), found Merlin productivity to be less susceptible to the impacts of DDE relative to other raptors, and an apparent population recovery was followed by the targeted approach to ban DDT and dieldrin in the 1970s (Stephens and Anderson 2002). It is worth noting that limited baseline data existed for the species prior to the DDT era creating unknowns regarding the accuracy of Merlin population recovery (Haak et al. in press).

After DDT and dieldrin were banned in Canada, research and sighting records continued to document the reduced to extirpated Merlin presence in Saskatchewan rural regions (Hodson 1970). These observations were a result of land cultivation creating habitat loss and decreasing prey availability (Hodson 1970). These habitat changes resulted in fewer nesting raptors and corvids and therefore fewer available nest sites, reducing breeding habitat for Merlins (Hodson 1970). As urban environments grew and natural habitat was lost, American Crows (*Corvus brachyrhynchos*) and Black-billed Magpies (*Pica hudsonia*) became less frequent in rural habitats with forested features and became common inhabitants in landscapes with lawns, parks, roads and residential neighbourhoods (Marzluff and Angell 2005). The distributional shift of crows and magpies into urban environments is perhaps the most significant factor contributing to

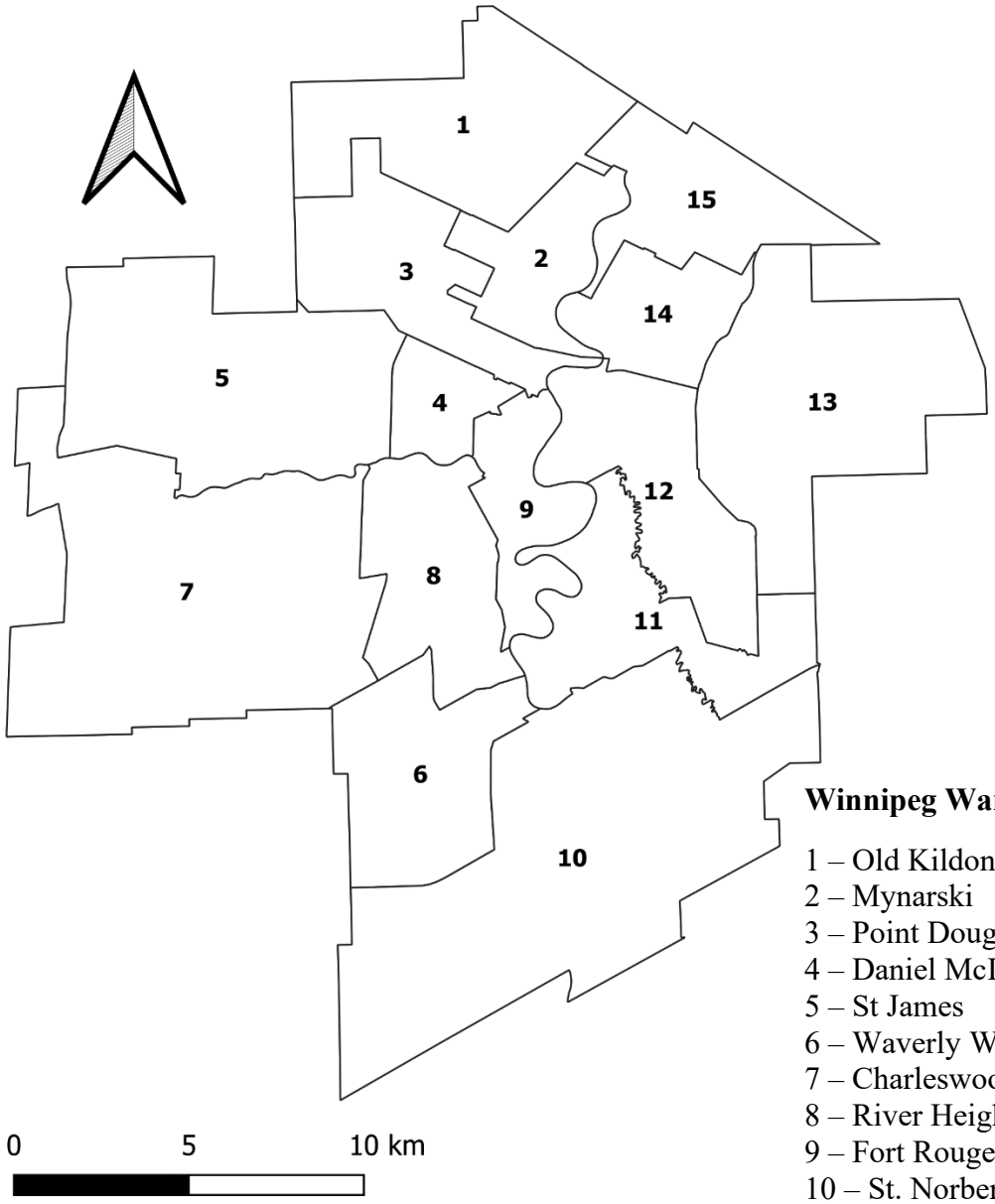
Merlins becoming urbanites, as crows and magpies offer an essential resource to Merlins - nests. Merlins, like most falcon species, do not build their own nests and rely on the availability of nests previously built by corvids or other raptors (Warkentin et al. 2020). A surge of urban forestry created a diversity of non-native tree species that offered corvids and therefore Merlins nesting habitat, year-round foraging opportunities, and habitat with reduced predation (Hodson 1970; James 1988; Hager 2009; Haak at al. in press). While the study of Merlin nesting habitat aims to examine the selection of Merlins, the importance of habitat selection by crows cannot be disregarded, and I believe it is crucial to acknowledge that the habitat examined in this thesis is a subset sample of initial crow habitat selection.

Early documentation of Merlins nesting in Manitoba was reported prior to 1950 when two nests were documented by Dr. Bob Nero (Oliphant 1985). Eight nests were recorded between 1954 and 1977 (Oliphant 1985); and the first urban nest was documented in the city of Brandon in 1977 (Sealy 2004), followed by the first nest in Winnipeg recorded in 1987 (Bancroft 1989). After 1987, considerable local population growth was documented over time (Bancroft 1989; De Smet 2003). The monitoring of Merlin presence within Manitoba has been limited after these observations; until summer surveys were conducted by the Wildlife Branch of Manitoba Sustainable Development between 1990 and 1994, followed by a collection of unpublished data from Dr. Merlin Shoesmith, Tanya Bowen, and voluntary reports from Winnipeg residents between 2011 and 2016. These data and other North American studies were the foundation upon which I developed my study.

STUDY AREA

The study area encompassed the 464.01 km^2 within the perimeter highway (PTH 100) of the city of Winnipeg (49.8951° N, 97.1384° W). Winnipeg is the capital city of Manitoba and is located within the Red River Valley. The city is made up of 15 wards (Figure 2) containing 236 neighbourhoods (City of Winnipeg 2021); the study area included all wards. The Red River Floodway runs along the east side of the city. The floodway is an artificial channel designated to control flood water and offer protection to the city of Winnipeg. Four rivers run through the city: Seine, La Salle, Assiniboine and Red River. The city is 234 m above sea level. The population of Winnipeg was 708,823 in 2016 (Statistics Canada 2019). Mean temperatures range from -23.9°C

with mean precipitation values of 22 mm in January; up to 25.7°C in July with mean precipitation values of 75 mm (Climate Data 2020). Winnipeg may experience extreme weather conditions, flooding, and long winters. Spring offers variable temperatures and conditions; one of the worst blizzards on record was documented on 5 April 1997. Winnipeg has been quantified as having one of the highest proportions of vegetation coverage in the world for a city (Dobbs et al. 2014) and at this time, 6% of Winnipeg is made up of park habitat (The Canadian City Parks Report 2020). The tree population in Winnipeg is primarily made up of 11 different genera: Ash (*Fraxinus spp*), Elm (*Ulmus spp*), Maple (*Acer spp)*, Oak (*Quercus spp*), Aspen (*Populus spp*), Birch (*Betula spp*), Willow (*Salix spp*), Basswood (*Tilia spp*), Spruce (*Picea spp*), Pine (*Pinus spp*), and Cedar (*Cedrus spp*) (Diduck 2012).



Winnipeg Wards

1 – Old Kildonan
2 – Mynarski
3 – Point Douglas
4 – Daniel McIntyre
5 – St James
6 – Waverly West
7 – Charleswood/Tuxedo/Westwood
8 – River Heights/Fort Garry
9 – Fort Rouge
10 – St. Norbert
11 – St. Vital
12 – St. Boniface
13 – Transcona
14 – Elmwood/East Kildonan
15 – North Kildonan

Figure 2. City of Winnipeg with 15 wards surveyed within this study.

STUDY OBJECTIVES

Research on urban raptors can influence management strategies for urban planning and inform residents of effective ways to cohabit with wildlife. Merlins are an abundant and accessible study species within the Winnipeg area. The objectives of this study were to determine nest site characteristics of Merlins in an urban (Winnipeg, Manitoba) environment by conducting vegetative surveys around the nest site, to estimate nest density, to document spatial distribution using observational techniques and nest surveys and to record the home range of breeding males utilizing radio transmitters. The research findings are outlined in a sandwich style thesis that is made up of five Chapters that follows the formatting style of the journal Ornithology.

Chapter 1 provides a general introduction of Merlins in North America; their physical description, distribution, past limiting factors and their colonization of urban areas that was documented in the 1960s. The sample area for this study and the early history of Merlins within Manitoba are described.

Chapter 2 describes the observed behaviour of Merlins during the breeding season, depicts habitat features on the micro and macroscale level of the nest site using unoccupied nest sites as a reference, and examines nest density and nearest neighbour distances for each field season. These objectives were aimed to address the nesting requirements of Merlins within this study area and understand how their external habitat may influence their spatial distribution. Within this chapter, previous studies are examined to offer insight into how resources and population dynamics may have influenced Merlin distribution across the urban landscape.

Chapter 3 examines the spatial movement patterns of four breeding male Merlins using new tracking technology, to provide their estimated home range. These objectives were aimed to address a significant gap in our understanding of habitat utilization of this species in Winnipeg. These data are contrasted with other studies that have examined the urban and rural breeding home ranges of Merlins within North America.

Chapter 4 investigates a potential West Nile virus outbreak in Winnipeg during the 2018 Merlin breeding season. Merlins with injury or illness were admitted to wildlife rehabilitation

centres for care; Merlins that survived were banded and released from the wildlife rehabilitation centres. The banding records from this study are included within this chapter.

Chapter 5 provides a general summary of the findings of the above chapters; outlines limiting factors; and recommends methods to build upon these data to support the continued monitoring of Merlin populations in Winnipeg.

CHAPTER TWO – NEST SITE CHARACTERISTICS AND DISTRIBUTION

ABSTRACT

Merlin (*Falco columbarius*) use of urban environments became commonly observed across North America in the 1960s. Urban habitats are dynamic and subject to change over relatively short periods. For this reason, it is important to understand the landscape features that wildlife rely upon within these habitats. This study took place within the city of Winnipeg, Manitoba, Canada, between 2017 and 2019. Over this period of time, 81 Merlin nests were recorded and monitored. All nests were found exclusively in nests previously built by American Crows (*Corvus brachyrhynchos*) and were located in spruce trees. The micro and macroscale habitat was analyzed to evaluate the vegetative and landscape characteristics that may influence nest selection and distribution of Merlins. The results show that Merlins selected nests that were located in taller spruce trees and sites that contained fewer coniferous species surrounding the nest tree. Merlin nests were recorded throughout the city, but patterns of nest clustering suggest required nest and/or prey resources may also be clustered. This study exemplifies the importance of maintaining existing spruce trees on private property and recommends the investment and planning of urban forestry that promotes tree longevity.

INTRODUCTION

Over the past 60 years, cities across North America have become ideal habitat for opportunistic species like Merlins (*Falco columbarius*) (Oliphant and Haug 1985). Urban forestry developed a population of native and non-native tree species including fruit bearing ornamental and coniferous trees. Ornamental trees can offer suitable environments and year-round food for songbirds and coniferous trees provide ideal nesting habitat for American Crows (*Corvus brachyrhynchos*) and Black-billed Magpies (*Pica hudsonia*) (Boal and Dykstra 2018). This abundance of songbirds and existing nests sites creates a foundation of resources for species like Merlins. The habitat selection of urban Merlins has been monitored in other areas of North America (Warkentin and James 1988; Haak et al. in press.) but there is a paucity of data for the species within Winnipeg, Manitoba. Targeted Merlin research in Saskatoon and the similarities

in climate, urban landscape and geographical location between Winnipeg and Saskatoon prompted interest in developing a base of data that would evaluate if Merlins displayed differences in their habitat selection and behaviours. The two study areas offered an interesting reference point but the difference in timeline makes it difficult to offer direct comparisons. Though, there is value in researching these dynamics as city environments are subject to habitat conversion and it is important to understand the habitat use of urban wildlife and document key features that are required to protect and sustain the environment that they rely upon for their survival (Boal and Dykstra 2018).

Previous research has described a diversity of nest site and habitat characteristics that have been utilized by Merlins but the most crucial piece to each geographical region and habitat that has been surveyed is the nest, as Merlins do not build their own nests. Other than the rare occurrence of building scrapes, which are shallow depressions. For this reason, the nest tree and the immediate surrounding habitat become valuable features to understand and offers insight into how urban populations adapt to their surroundings. Nest substrates utilized by Merlins vary from deciduous spp. in rural Saskatchewan (SK) (Hodson 1970), large Douglas Fir (*Pseudotsuga menziesii*) in Seattle (WA) (Haak et al. in press), cavities and scrapes in Europe (Bent 1961), ground nests in Churchill, MB (De Smet 2003) and the exclusive use of spruce trees within Canadian cities (Warkentin and James 1988; Stephens and Anderson 2002).

Merlins can display site fidelity to the same territory (Hodson 1970). However, the same nest is rarely used in consecutive years; factors such as the structural stability post-breeding season (Haak et al. in press), previous breeding failure at a site (Warkentin et al. 1991), and crow nest selection influences this behaviour. Nest site selection by crows within Winnipeg is documented to vary each year. Consecutive nest re-use is not common, and they may switch between deciduous and coniferous trees from one year to the next (Nature Manitoba 2011). Merlin preference to specific tree characteristics is observed beyond the breeding season in both migratory and non-migratory populations. Haney (1997) suggested that overwintering Merlins in Utah may select roost sites that represent natal imprinting and found adults to remain close to both past and future nesting territories over the winter season (Warkentin et al. 1991). The average life expectancy of Merlins ranges from 3.15 ± 1.91 years in females to 2.67 ± 1.51 years in males (Warkentin et al. 2020). However, breeding site fidelity can be observed in the species and is more commonly observed in males than females (Hodson 1970; James et al. 1988). In

Saskatoon 25% of males switched territories compared to 70% of females; this behaviour likely reduces in-breeding within populations (James et al. 1988). Selecting known territories on breeding grounds likely aids in maintaining territories, sustaining better quality sites and ease of efficiently navigating through well-known areas (Warkentin et al. 1991). When young disperse from their natal territories, documented distances to newly established sites vary from means of 3.7 ± .7 km in males, 3.8 ± .8 km in females to extremes up to 250 km away in the Saskatoon population (Warkentin et al. 2013).

Merlins rarely hunt at the nest site; they rely on open or sparsely vegetated habitats for foraging excursions (Hodson 1970; Sodhi et al. 1990; Konrad 2004; Haak et al. in press) and have historically utilized riparian areas within city areas that offer open habitat and greater diversity (Hodson 1970; Warkentin and James 1988; Sodhi et al. 1991; Konrad 2004). Parks, greenspace and golf courses make up only 7% of the landscape in Winnipeg and the city lacks a Master Plan for Greenspace; a policy plan that many other Canadian cities have adopted to protect and enhance green space. Without a management plan this leaves Winnipeg vulnerable to infrastructure development potentially resulting in loss of green space (CCPA 2020).

Merlin selection of spruce trees for nesting within urban areas has been documented within Canadian cities (Warkentin and James 1988) and previous nesting records within Winnipeg had exclusively recorded Merlin nests within spruce trees (M. Shoesmith unpub.). In addition to the importance of nest sites, perching substrates such as mature deciduous trees aid in food transfers, monitoring the nest, copulation, and communication between the male and female throughout the field season (Haak et al. in press). Colorado Blue Spruce (*Picea pungens*) and White Spruce (*Picea glauca*) are large coniferous trees that collectively make up 97% of the spruce tree population on public property in Winnipeg (City of Winnipeg 2020a). The high abundance of these two species makes them the most available spruce species available for nesting. However, spruce trees make up only 6% of the total tree population on public property in Winnipeg (City of Winnipeg 2020a). Both species are sensitive to their growing environment, requiring well drained areas and specific soil characteristics (Allen 2017). Although the two species are subject to the diseases ‘Sirococcus tip blight’ and ‘Cytospora banker’, their longevity and substantial growth demonstrates their resilience and ability to be sustained within Winnipeg. Two deciduous tree species that make up much of the urban forest in Winnipeg are American

Elm (*Ulmus americana*) and Green Ash (*Fraxinus pennsylvanica*) – both individually making up 40% of the tree population on public property for a total of 80% of Winnipeg's public urban forest (Diduck 2012; Trees Winnipeg 2020). Ash and elm trees are species that are characteristically sturdy and grow to be large in their maturity. In 1975 Dutch Elm disease (DED) was confirmed in Winnipeg (City of Winnipeg 2020b), DED is an invasive fungus that is responsible for killing off healthy elm trees (Trees Winnipeg 2020). Ash trees were the selected boulevard trees for replacement after the appearance of DED, creating a monoculture of ash trees after 1980 (City of Winnipeg 2020b). In 2017, Emerald Ash Borer (*Agrilus planipennis*) (EAB), an invasive beetle – that kills off healthy ash trees was confirmed within the city (Trees Winnipeg 2020). The management program within the city of Winnipeg involves removal of elm trees infected by DED (City of Winnipeg 2020b) and a combination of ash tree removal and pesticide injection to slow the spread of EAB (City of Winnipeg 2020c). Preventative measures are underway by the city of Winnipeg to plant tree populations that contain greater diversity that will offer the ecological benefits to humans and wildlife while reducing the high risk of disease and pest transmission that monoculture communities are susceptible to (City of Winnipeg 2020d). In other Canadian cities EAB has been documented to eradicate 90% of ash species and the loss of mature trees in Winnipeg may impact the distribution and behaviour of species within the urban environment (City of Winnipeg 2020d).

Merlins are reliant on the availability of mature trees that can support them throughout the breeding season and within the city of Winnipeg, deciduous and coniferous trees are subject to removal or damage. As spruce trees mature, they can become too large for the lot size and block natural light on personal property which may result in their removal, as observed in the thousands of spruce trees have been removed from older neighbourhoods in Winnipeg (G. Engel personal communication). During street construction and maintenance, physical space and soil volume required for supporting the longevity and health of planted trees are limited and therefore these factors limit where trees of substantial size can be planted. The exposure to these risks may only offer tree life expectancies of 10 years (City of Winnipeg 2020e). Newer neighbourhoods within Winnipeg do not commonly incorporate large trees into their landscape due to limited lot space, the type of grade that is used for drainage purposes and soil that is not conducive to trees with legacy (G. Engel personal communication). A 20-year urban forestry plan is being developed for the city of Winnipeg and at this time the only conifer that is considered within this

plan are cedar species (G. Engel personal communication). In Winnipeg, the continued loss from disease, the proclivity away from planting mature spruce and deciduous trees and the possible damage of existing trees has the potential to change to the vegetative landscape and can influence the behaviour and distribution of wildlife that utilizes these resources.

To better understand Merlin habitat selection within this study area, Merlin nests were monitored to evaluate the distribution of nests, the surrounding habitat characteristics, and the behaviour of Merlins during the breeding season in Winnipeg. Vegetative surveys surrounding the nest site were utilized to examine if there are specific habitat features that are selected by this population and nest survey methods was used to evaluate if there were areas of the city that demonstrated higher nest densities than other areas that may have different landscape features. These data were aimed to understand and identify features required during the breeding season and potentially understand the vulnerabilities that Merlins may be faced with in Winnipeg.

METHODS

NEST SURVEYS

Nest site surveys took place between the end of March to the beginning of July 2017 to 2019. The 15 wards within city limits were surveyed by bike, foot and car. This database was built using previously confirmed nest sites monitored by Dr. Merlin Shoesmith and Tanya Bowen, in-take records provided by Prairie Wildlife Rehabilitation Centre and Wildlife Haven Rehabilitation Centre, eBird records within Winnipeg during the breeding season, community science reports, data requests on Manitoba birding forums and territory monitoring from previous field seasons. All community citizen science and social media reports received were ground-truthed before sites were considered confirmed.

Nest sites were identified using aural and visual behaviour. Behaviours indicating potential nesting activity included: predictable/sedentary behaviour in a concentrated area, food transfers between pairs, vocal communication between pairs, and nestling presence. The nest site was not considered occupied without the observation of the male, female or both individuals at the nest. The surveying schedule varied greatly throughout the season based on seasonal changes in daylight. Sunrise began at 07:45 (CST) mid-March and began around 05:20 mid-May to mid-

June. However, I tried to be on site by 04:00, as nest site activity (vocalization, hunting, food transfers) were commonly observed by 04:20 between May and July. Peak activity periods were generally over by 09:00 and activity would often resume at approximately 17:00 lasting until 21:30. Nest surveys did not take place on rainy days as pairs were less active and vocal during those periods. Broadcast calls were played at 100 db from a cellular telephone during the courtship period between late March and early May but no later than this window to avoid disturbance during the incubation period which begins in May. Broadcast calls can elicit a vocal response from Merlin pairs and make their location easier to identify.

Once a pair had been identified the date, time and location, behaviour, interspecies presence within the nest vicinity, and perch substrate use/selection were recorded. After initial site confirmation, check-ups were made to determine the status of the site at least twice a month. A site was determined to be successful if the pair remained at the nest site until the fledging period and was determined abandoned or unsuccessful if the pair left prior to the fledging period. Visiting nest sites for extended time periods was avoided to reduce unnecessary disruption.

NEST SITE MEASUREMENTS – MICRO AND MACRO SCALE

Nest site parameters were measured at 30 Merlin occupied sites and 30 randomly selected sites containing a spruce tree with a crow nest unoccupied by a Merlin (Sieg and Becker 1990; Boal 1997; L. Oliphant personal communication). Random nests were identified using the Random Points feature in QGIS version 3.8. Using this feature, random coordinates were produced within the Winnipeg study area. Each location was surveyed and the closest spruce tree to the GPS coordinates containing an unoccupied nest within a 25 m radius was selected. If a spruce tree with a nest could not be identified, a Merlin occupied a nest within the 25 m radius, or permission was not given to be on the property, the location was deemed as unsuitable, and another random point or nest location would be selected. Nest site behaviour was monitored and recorded throughout the study. Obtaining temporal and spatial behaviour of adults and juveniles within the nest site radius was essential to effectively utilize observation periods at the nest and examine habitat use directly to hone in the study variables of interest. Microscale variables were selected based on observations made at the nest site and previous Merlin nest site studies (Warkentin and James 1988; Sieg and Becker 1990; Solensky 1997; Stephens and Anderson 2002; Drummond and Stillman 2019). Nest site and vegetation parameters were recorded late

August to September 2018 and 2019. This timeframe was selected as young have dispersed and human presence would not cause any disturbance to the nest.

Microscale measurements were taken at the nest tree and the 25 m radius around the nest tree. This radius was based on the nest site observations recorded through the season and previous studies (Drummond and Stillman 2019). The nest tree parameters measured included: GPS coordinates of the nest tree, nest tree height, nest height, circumference at breast height (CBH), nest tree species and nest orientation. All other parameters measured within the 25 m radius included: perch substrate (if within 25 m) and the height, CBH, species, and absolute distance from nest tree to all deciduous and coniferous trees within the radius. Deciduous and coniferous trees were analyzed separately as they appear to be utilized for different purposes by Merlins during the breeding season. All height measurements were recorded using a clinometer. Tree circumference was measured with a girthing tape and the distance from nest tree was measured with a range finder or open reel fiberglass tape.

The macroscale habitat was classified as anything beyond the 25 m radius of the nest. To examine the macroscale habitat in relation to the nest site, distances were measured from the nest tree to the closest surface water feature, habitat designated as park or greenspace, nearby utility pole (documented within and outside the 25 m radius, depending on site) and nearby building. These variables were selected based on nest site observations and previous studies examining Merlin habitat use (Warkentin and James 1988; Sieg and Becker 1990; Solensky 1997; Stephens and Anderson 2002, Drummond and Stillman 2019). Building details, parks and green space data were obtained from Winnipeg Open Data Portal (City of Winnipeg 2020a) and surface water features were obtained from Manitoba Land Initiative (Province of Manitoba 2020). The utility pole locations were obtained from Manitoba Hydro (M. Miller personal communication). Join attributes by nearest feature was run in QGIS version 3.10.9-A Coruña to measure the distances from the nest site to the macroscale features.

STATISTICAL ANALYSES

MICRO AND MACROSCALE HABITAT

A total of 21 variables (15 microscale and 6 macro macroscale) were selected based on the biological significance of Merlin behaviour and a literature review. These 21 variables were run through a correlation matrix test with package *corr.test* (Revelle 2020) in RStudio version

1.2.5042 (RStudio Team 2020) to examine multicollinearity across all variables. Variables that showed correlation of .70 and greater resulted in the removal of one of the two correlated variables (Table 1). The variables selected to remain were assessed based on biological relevance and accuracy of measurement. Biological relevance refers to the variable that holds more meaning to the surrounding environment/species; the intercorrelation between tree height and circumference resulted in removing circumference as the height offers a better indication of the tree features in this context. Accuracy of measurement refers to variables that had were less susceptible to human and equipment errors; measuring one tree would have decreased risk of error than measuring multiple trees within a nest radius.

A principal component analysis (PCA) was run to examine the variation across the remaining variables. The data were scaled to standardize the range of values and a PCA was run using *prcomp* (R Core Team 2019). The function *get.eigenvalue (*Kassambara and Mundt 2020) was used to extract eigenvalues from the PCA. Using the Kaiser criterion, variables with eigenvalues below one were removed (STHDA 2017; RPubs 2020; Statistics Solutions 2021) (Table 2). A biplot was produced using function *biplot* (R Core Team 2019) to visually assess the variable loadings (Figure 3). Variable loadings forming a small angle on this plot indicated autocorrelation and this resulted in the removal of one variable. The remaining variables from the PCA process were included in a generalized linear model (GLM) with package *glm* (R Core Team 2019) to examine if there were significant differences between occupied and unoccupied nest sites. These variables included: nest tree height, mean distance to deciduous, distance to utility pole, tallest deciduous tree, mean number of conifers, mean number of deciduous trees and the mean age of building (Table 3). The response variable assumed a binomial distribution and was classified as 0 = unoccupied and 1 = occupied.

NEST DENSITY AND DISTRIBUTION

To examine nest density, nests were analyzed using nests/100 km^2 for each Winnipeg ward and for the entire sample area for each year (Table 4). The nearest neighbour distances were analyzed using package *spatstat* (Baddeley and Turner 2005) in RStudio version 1.2.5042 (RStudio Team 2020). The function *nndist* produced the nearest neighbour distances based on the appropriate k-nearest neighbour (KNN) for each year. Site fidelity influenced the decision to analyze years separately for nest density and KNN. Separating years reduced potential bias

associated with sites selected within the same nesting territory. Analyzing years together may have resulted in nests appearing closer together. The k-value used for KNN was found by extracting the square root of n (nest count for the study year) and was rounded to the nearest whole number for each year (Prakash 2016). This produced the mean and range values of nest distances across the sample area (Table 5).

Spatial distribution of nest sites was analyzed with package *spatstat* (Baddeley and Turner 2005) with the Besag L-function *lest* (Baddeley and Turner 2005). The L-function is a transformation of Ripley's K that uses a square root transformation which stabilizes the variance and makes deviances in the data easier to observe (Baddeley et al. 2016). The purpose of this analysis was to examine the spatial distribution of nests by looking at the dispersion, clustering or random patterns of the datasets. The *lest* function requires raw Universal Transverse Mercator (UTM) coordinates to be converted into a point pattern object (*ppp*) and an applied edge correction to appropriately weight the border samples. The edge corrector "translate", was selected as it is suitable for smaller datasets, does not assume isotropy and applies to all window geometries (Baddeley et al. 2016). To visualize the nest distribution for each year and test for complete spatial randomness (CSR), I used a Monte Carlo simulation envelope with function *envelope* (Baddeley and Turner 2005) using .05 as the significance level. This method was selected as it is effective with small sample sizes (Baddeley et al. 2016) and can test the spatial distribution of nests using simulated distances of the dataset. The function rearranges the data 39 times within the sample area to produce what would be an expected distribution of the dataset and create the critical points for comparison against the empirical data (Figure 4) (Dixon 2002; Baddeley et al. 2016). A maximum range distance of 4250 m was selected as it was the lineal distance of the smallest ward and this offered a way to subset the sample area.

RESULTS

NEST SITE SURVEYS, MICRO AND MACRO SCALE HABITAT

The behavioural observations collected during nest site monitoring are described. Territory establishment and pair bonding is initiated as early as February and can continue through March into April. During territory establishment and courtship, the male often perches on a spruce tree within the nest site area and vocalizes to communicate with a mate or attract prospective mates. Interspecies aggression with crows is common during nest establishment and

during this study, conflict between crows and Merlins appeared to result in one pair of Merlins relocating from an established nest site as late as May. When pair establishment and bonding have occurred, mature trees surrounding the nest tree are utilized throughout the season as substrates for copulation, food transfers, roosting, nest defense, caching and a vantage point of the nest tree. The female's reliance on nearby deciduous trees changes throughout the season as she begins spending much more time at the nest itself during the incubation period which begins early to mid-May. During the incubation period, the male becomes the primary food provider for the pair and habitually utilizes specific perching substrates in the form of surrounding deciduous trees or utility poles to communicate with the female and monitor the nest and surrounding habitat. The female will remain very close to the nest between May and mid-June but will leave for short periods for feeding opportunities presented by the male that rarely take her out of sight of the nest tree. By mid to late-June when the female is not required to brood the young, she may spend less time at the nest, more time hunting and roosting away from the nest in nearby spruce or deciduous trees. In early July, the young begin to fledge and rarely return to the nest tree. They rely on mature trees – both deciduous and coniferous, utility poles and lines and roofs of buildings to roost and perch on while they hone their flight and hunting skills. By late July to early August the young have dispersed from the nest site. Merlins utilize the mature trees throughout many stages of the breeding and post-breeding cycles.

A total of 81 Merlin nests were located during the three-year study and 28 additional nests were reported by Dr. Merlin Shoesmith between 2011 and 2016 (M. Shoesmith personal communication) producing 107 nests in total. All nests used by Merlins were in spruce tree species containing unoccupied crow nests. The majority of both occupied and unoccupied sites were found on residential property. For occupied sites, 27 of 30 nest sites were on residential property and the remaining three were found on apartment, church and retail properties. On unoccupied sites, 9 were found on business, industrial and apartment building properties and 21 were on residential property.

Of the 21 variables analyzed, two variables were found to significantly contribute to defining the micro and macro scale habitat. Five variables were removed due to intercorrelation (nest diameter, nest height, mean height of surrounding conifers, mean distance to surrounding conifers and mean height of deciduous trees) and an additional nine were removed based on the

Kaiser criterion (Table 2). Biplot assessment resulted in the removal of mean deciduous diameter based on autocorrelation with nest tree height. This decision was made based on Merlin reliance on the nest tree and deciduous tree features described by three of seven variables that remained within the GLM model. Seven variables remained to be run through the GLM (Figure 3). The complete list of habitat variables for both occupied and unoccupied sites are included in Appendix A. The variables of significance were the mean nest tree height and number of conifers surrounding the nest site. The mean nest tree height ($Z = 2.601$, $P = .01$), indicated that the nest tree height was on average significantly taller at Merlin occupied sites than unoccupied sites and the mean number of conifers surrounding nest sites was trending towards significance ($Z = -1.810$, $P = .07$), indicating that there may be fewer conifers at Merlin occupied sites than Merlin unoccupied sites.

Though nest height was removed due to intercorrelation, nests were found located just over 2/3rds of the way up the nest tree at the 70th percentile, for both occupied (10.97 ± 2.23 m) and unoccupied sites (9.14 ± 3.63 m). All nests were near the trunk of the tree and very well concealed, nearly impossible to observe with the naked eye as reported in other studies. Although there were variables that were not found to be significant predictors overall and variables that did not contribute to the overall PCA and were therefore not included in the GLM, there are differences observed in the mean values between the variables recorded at unoccupied and occupied nest sites (Appendix A).

NEST DENSITY AND SPATIAL DISTRIBUTION

The mean number of nests between 2017 to 2019 was 27, the fewest nests reported was 17 in 2017 and the highest number reported was 35 in 2018. One ward, Old Kildonan did not report any nest sites over the study period. The neighbourhood with the greatest mean density based on ward area was Daniel McIntyre (area = 13.27 km, $\bar{x}$ = 15/100 km^2) (Table 4). Overall, the nest density for the city of Winnipeg was $\bar{x}$ = 6.73/100 km^2.

The mean nearest neighbour (KNN) distances across all study years was $\bar{x}$ = 1673.56 m, the mean minimum distance was $\bar{x}$ = 161.64 and the mean maximum distance was $\bar{x}$ =5836.46 m. The minimum distance between nests was 81.93 m in 2017 and the greatest distance between nests was 7185.80 m in 2018 (Table 5). Nest distribution using the L-function, displayed spatial patterns that suggested the documented nests are distributed in a non-random manner with

statistically significant patterns that indicated clustering at different levels of nesting distances (Figure 4).

DISCUSSION

MICRO AND MACROSCALE HABITAT

Broad habitat descriptions have been identified in previous studies across North America, but a general trend has been described within the nesting habitat examined for Merlins. This site description includes: a mature tree containing an unoccupied and well concealed corvid nest (Hodson 1970; Warkentin and James 1988; Sieg and Becker 1990; Haak et al. in press), surrounding habitat with sparse to low tree densities of greater height (Hodson 1970; Sieg and Becker 1990; Ayers and Anderson 1999; Haak et al. in press) and an open habitat for foraging within a close proximity to the nest site (Hodson 1970; Sieg and Becker 1987; Sodhi and Oliphant 1992; Konrad 2004). Sieg and Becker (1987) observed that grassland and agriculture habitats within their study area to be underutilized disproportionally relative to availability and they hypothesized that this may be associated with the lack of perching substrates in those habitats. Conversely, they found riparian habitats to be utilized at a higher proportion than anticipated and they suggested it was in relation to diversity and greater perching opportunity. Ayers and Anderson (1999) suggest that Merlins did not rely on a specific set of features within their study area and Hodson (1970) described trees with nests and an avian prey source to be the key features of nesting requirements for Merlins.

The results within this study determined the best predictors of Merlin nesting habitat to be height of the nest tree and the number of conifers surrounding the site; documenting that nest trees were significantly taller and there were fewer conifers within 25 m of the nest tree at occupied sites. Although not statistically significant, the general characteristics based on mean differences between occupied and unoccupied sites describe subtleties between the two sample groups (Appendix A).

Alternative tree species were not considered as potential nest sites within the analysis because spruce were the only tree species that were utilized for nesting within the study area between 2011 and 2019, based on the sample of 107 nests. A Winnipeg study conducted by Tom Reaume, found that 53% of crow's nests were found in coniferous trees. Of those nests, 50% were in spruce species, and 47% were in deciduous trees (Nature Manitoba 2011). This suggests

Merlins are preferentially selecting nests in spruce trees over deciduous nests as the availability of nests in both groups is approximately equal based on that study period. Spruce trees offer nest concealment and protection which can reduce the energy expenditure during extreme and variable weather (Haney 1997). The timeline for leaf emergence for deciduous trees in Winnipeg is variable but often does not occur until May as per observation within the study area. This period of the season is during or just prior to the incubation period for Merlins. Without foliage, deciduous trees would offer limited concealment or protection from the weather or predators until the breeding season was well underway. This protection may not be as essential to crows due to their larger size resulting in reduced risk of predation by species such as Great Horned Owls (*Bubo virginianus*) and variable weather (B. Haak. personal communication).

Based on the broad habitat features that were observed within this study and previous research, the adaptability of Merlins is evident and quite possibly the reason they have successfully adjusted to urban environments. Older neighbourhoods incorporate habitat features such as mature spruce trees for nests and perch substrates such as utility poles and mature deciduous trees – all features utilized by Merlins within this study area throughout the breeding season. The inclusion of coniferous landscaping within older neighbourhoods also provides suitable habitat for prey species such as House Sparrows (*Passer domesticus*) within Winnipeg, as flocks often utilize coniferous species as roost substrates (K. Fraser, pers. obs). In contrast, newer housing developments within the city encompass limited numbers of mature trees and reduced availability of utility poles as power lines are buried underground, likely for aesthetic purposes (B. Fougure personal communication). These factors may be limiting in nest and prey availability but do not appear to entirely deter Merlins from utilizing those sites and as those habitats develop and as mature trees are lost as a result of damage or disease in older neighbourhoods, they may become desirable locations.

NEST DENSITY AND SPATIAL DISTRIBUTION

The nest density of avian species is often associated with habitat quality and suitable nest availability (Newton 1991). As such, gathering data on habitat as an external factor that influences a species of interest provides the most accurate method of understanding how that species behaves and maintains ecological interactions both spatially and temporally.

Raptors are thought to space their nests in a regular pattern if there are sufficient resources such as nest sites and prey availability (Nilsson et al. 1982; Sodhi et al. 1991) but limited availability of these resources can result in nest clustering (Solonen 1993). The local density limitation model describes avian behaviour to maintain a hierarchal preference towards specific habitat features (Newton 1999). Within this model, habitats with ideal features will be utilized until they reach their highest capacity at which time, habitats of lesser quality will be occupied. If available crow nests in Winnipeg are limited and have varying degrees of ideal features or prey density is variable, hierarchal behaviour may be a factor that explains the trends of nest clustering observed in Winnipeg. Within the context of site availability, Merlins in Winnipeg are limited in where they can nest due their reliance on crow nests in mature spruce trees. Reaume's study of urban crows documented 47 crow's nests in Winnipeg between 2008 and 2009 and 50% of the nests were in spruce trees (Nature Manitoba 2011). The structural longevity of those crow's nests post-breeding season is unknown. Merlin monitoring in the city has documented nests previously utilized by Merlins falling out of trees after 10 years and one nest within the Transcona area of Winnipeg that remained for over 30 years (M. Shoesmith personal communication) but the structure of the nest itself post-breeding season is unknown and if lacking these limitations could be associated with clustering behaviour. It is difficult to ascertain how many potential nests of high quality exist within the city or a territory. In addition, preference to territory in the form of site fidelity can also influence clustering behaviour. As habitat fluctuations occur and sites become more desirable, site fidelity in raptors may result in behavioural adaptation in the form of territory reduction rather than movement away from neighbouring nests (Newton 1999).

The distribution trends documented within this study report a lower density relative to other Canadian urban studies; 20/100 km^2 in Edmonton, AB and 25.4/100 km^2 Saskatoon, SK (Sodhi et al. 1991). The nest density, 6.73/100 km^2 within this study was comparable to the rural study area of Montana, 3.8/100 km^2 (Sieg and Becker 1987) and the urban study area of Seattle, 8.79/100 km^2 (Haak et al. in press). These observations may be associated with the size of study area, all studies listed above are approximately $1/4^{th}$ of the study area that was selected for this research.

The lowest number of nesting records during a single year during this study was 17 nests as documented in 2017. The field surveys in 2018 and 2019 found population estimates of 35 and 29 sites, respectively. These recorded inconsistencies within the three study years were likely attributed to the number of residential contacts, research methods and database development rather than an accurate measure of actual Merlin nest abundance. The distribution of nest occupation within this study recorded varying degrees of nest presence between sample wards. Despite potential differences between neighbourhood features of both vegetative and structural forms within and across wards, there was only one ward (Old Kildonan) that did not contain a documented Merlin nest and maintained very few Merlin sightings throughout the three-year study (eBird 2021). These observations indicate that throughout Winnipeg there are areas within 14 of 15 wards that can support Merlin nesting habitat. One third of the habitat within the Old Kildonan ward contains non-residential habitat, spaces containing agriculture features and fewer trees than central areas of the city. These habitat features and the consideration of missed nests or areas that are inaccessible to survey may be associated with the lack of nest reports during the survey period. The Daniel McIntyre ward exhibited the highest density per size of ward was Daniel McIntyre ($\bar{x}$ = 15/100 km^2). This ward is in the core area of the city and documents one of the highest proportions of homes built before 1960 (67.9%) in the city (City of Winnipeg 2021). The abundance of older homes may be associated with site fidelity to nests that were utilized during initial colonization and an abundance of mature trees available for nesting.

These results demonstrate that there are many considerations to be addressed for evaluating the way in which Merlins utilize breeding habitat. Resource availability, historical habitat uses, crow abundance and nest site selection, Merlin preference to habitat features and survey accuracy may all play a role in understanding the distribution of breeding Merlins across the urban landscape of Winnipeg.

CONCLUSION

The exclusive selection of unused crow nests that are located in spruce trees by Merlins emphasizes the importance of the nest tree in Merlin breeding success. While this is an important feature to Merlins, spruce trees are not necessarily desirable tree species on residential properties. The size of spruce trees in their maturity, potential maintenance requirements and concern of disease may deter home owners from planting spruce trees on private property within

Winnipeg. The observation of clustered nests within this population indicates that there may be a lack of suitable nest sites available to Merlins. If spruce trees are removed within this study area, this may negatively influence or alter Merlin habitat use. As the majority of Merlin nest trees are located on private property, monitoring and education play important roles in ensuring we can maintain these important resources that currently exist for Merlins within the city of Winnipeg.

Table 1. Highly correlated variables, correlation (r) and the variables removed prior to analysis.

Correlated variables		r	Variable removed
NTheight	Nheight	0.83	Nheight
Ndiameter	NTheight	0.80	Ndiameter
Nheight	Ndiameter	0.70	Both were removed
DistTallestC	MeanDistC	0.79	MeanDistC
TallestC	MeanHeightC	0.72	MeanHeightC
MeanHeightD	TallestD	0.83	TallestD

NTheight = Nest tree height

Diameter = Nest tree diameter

Nheight = Nest height

DistTallestC = Distance to the tallest conifer within 25 m

TallestC = Tallest conifer within 25 m

MeanHeightC = Mean height of conifers within 25 m

MeanDistC = Mean distance to conifers within 25 m

MeanDiamC = Mean diameter of conifers within 25 m

DistTallestD = Distance to tallest diameter within 25 m

TallestD = Tallest deciduous tree within 25 m

MeanDistD = Mean distance to deciduous trees within 25 m

MeanDiamD = Mean diameter of deciduous trees within 25 m

MeanHeightD = Mean height of deciduous trees within 25 m

Coniferous = Number of coniferous trees within 25 m

Deciduous = Number of deciduous trees within 25 m

ParkSize = Park size of the park/green space closest to the nest tree

ParkDist = Distance to the nearest park/greenspace

YearBuilt = The mean age of home/building closest to the nest tree

BldgDist = Distance to the closest building from the nest tree

PoleDist = Distance to the closest utility pole from the nest tree

WaterDist = Distance to the closest water body from the nest tree

Table 2. Eigenvalues up to 10 dimensions and the contribution to the principal components. Eigenvalues below one were removed from analysis.

Dimension	Eigenvalue	Variance Percent	Cumulative Variance Percent
1	2.8001269	18.667513	18.66751
2	2.1767388	14.511592	33.17910
3	1.5804374	10.536249	43.71535
4	1.4262665	9.508443	53.22380
5	1.2890452	8.593635	61.81743
6	1.1290170	7.526780	69.34421
7	1.0442745	6.961830	76.30604
8	0.8630612	5.753742	82.05978
9	0.6768560	4.512374	86.57216
10	0.5227655	3.485103	90.05726

Table 3. Mean, standard deviation, ranges and p-values as produced from the GLM at occupied and unoccupied nests (n=30 for both samples groups). All measurements were taken or standardized to metres. Coniferous and deciduous variables are count data and the year built is a date.

Variables	Occupied	Unoccupied	*p*-value
NTheight	15.75 ± 2.6	12.69 ± 3.55	0.01
	(9.6 – 21.6)	(7.0 – 18.2)	
MeanDistD	15.76 ± 4.49	17.43 ± 5.31	0.19
	(5.09 – 24.98)	(7.88 – 24.95)	
PoleDist	126.3 ± 196.2	104.5 ± 129.4	0.19
	(7.02 - 800.38)	(11.23 - 476.54)	
TallestD	14.99 ± 3.87	13.60 ± 5.98	0.55
	(8.25 – 21.89)	(3.0 – 24.12)	
Coniferous	2.8 ± 1.97	6.0 ± 5.5	0.07
	(1-9)	(1-21)	
Deciduous	5.07 ± 3.04	6.9 ± 6.56	0.55
	(1-14)	(1- 32)	
YearBuilt	1957 ± 23.84	1968 ± 21.24	0.95
	(1888 – 1991)	(1915 – 2014)	

Table 4. Each ward and area in km^2. Nest counts recorded per season and mean nests per ward. Mean densities for each ward per 100 km^2.

Ward	**2017**	**2018**	**2019**	**x̄ Nests**	**Ward Area (km^2)**	**x̄ Density**
Old Kildonan	0.00	0.00	0.00	0.00	32.30	0.00
St. Norbert	0.00	1.00	2.00	1.00	52.34	2.00
St. Vital	0.00	1.00	2.00	1.00	44.87	2.00
Point Douglas	0.00	2.00	0.00	.67	32.30	2.00
Mynarksi	0.00	0.00	2.00	.67	23.35	3.00
Charleswood/Tuxedo/ Westwood	0.00	6.00	0.00	2.00	46.53	4.00
North Kildonan	0.00	2.00	2.00	1.33	27.12	5.00
St. Boniface	0.00	4.00	2.00	2.00	32.19	6.00
St. James	2.00	5.00	0.00	2.33	33.31	7.00
Waverly West	1.00	3.00	1.00	1.67	24.14	7.00
Fort Rouge	2.00	1.00	3.00	2.00	25.87	8.00
Elmwood/East Kildonan	2.00	3.00	1.00	2.00	17.77	11.00
Transcona	6.00	1.00	7.00	4.67	33.77	14.00
River Heights	1.00	4.00	6.00	3.67	24.58	15.00
Daniel McIntyre	3.00	2.00	1.00	2.00	13.27	15.00
	17	**35**	**29**	**27**	**Total area = 464.1 km^2**	**6.73/100 km^2**

Table 5. Study year, nest count totals, k-value, mean and range distance between nest sites based on the k-value in metres.

Year	Total nests	K-value	$\bar{x}$	min	max
2017	17	4	1765.17	81.93	6156.77
2018	35	6	1748.90	248.00	7185.80
2019	29	5	1506.60	155.00	4166.80
Total $\bar{x}$	**27**		**1673.56**	**161.64**	**5836.46**

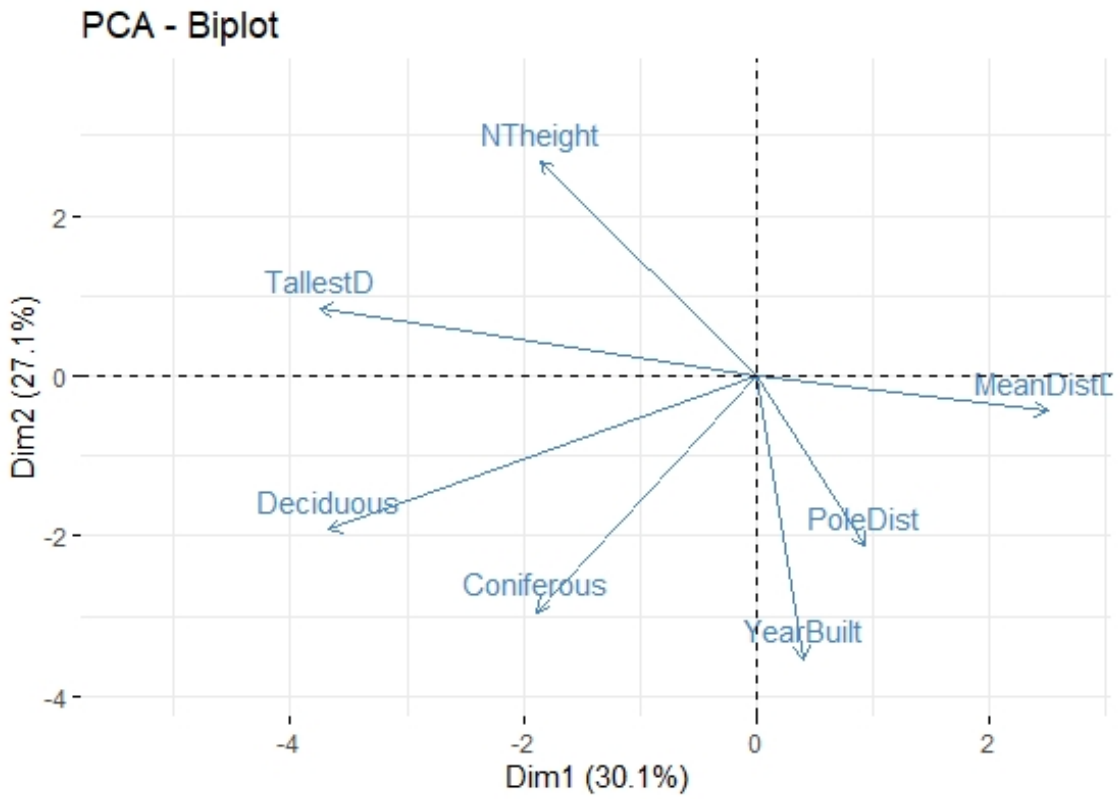


Figure 3. The seven variables selected for the GLM based on the greatest contribution to the PCA. Nest tree height (NTheight), tallest deciduous height (TallestD), mean number of deciduous (Deciduous) and coniferous (Coniferous) trees, mean distance to utility pole (PoleDist), mean distance to deciduous trees (MeanDistD) and mean age of home (YearBuilt).

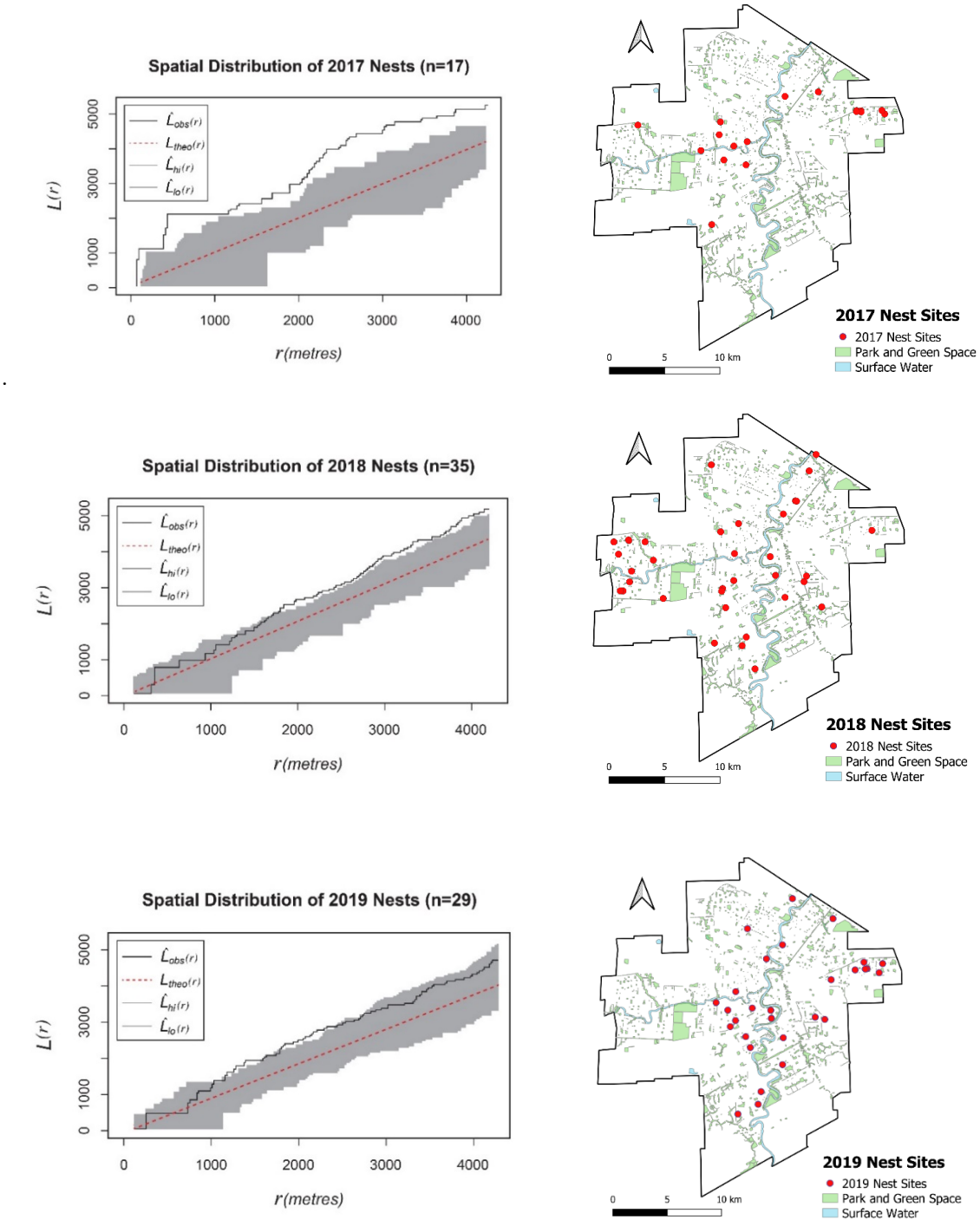


Figure 4. Monte Carlo simulation (left). Grey envelope designates upper range (L̂hi) and lower range (L̂lo) of the critical points based on 39 data simulations. Simulated range (Ltheo) is compared to the empirical data (L̂obs). Data outside of grey envelope are statistically significant. Nests per season (right).

CHAPTER THREE – SPATIAL MOVEMENT PATTERNS

ABSTRACT

Researching the home range of wildlife provides an in-depth understanding of the influence both internal and external factors can have on an animal's movement and behaviour within an environment. During the breeding season in 2019, four adult male Merlins (*Falco columbarius*) were fitted with radio transmitters to track their spatial movement patterns in Winnipeg, Manitoba, Canada. The mean home range and core use size for all four birds was 1.79 km^2 and .26 km^2, respectively. No home range overlap was recorded in the tracked individuals, but the home range of one Merlin encompassed the nest of an untracked Merlin within its home range. All but one tracked Merlin stayed within the city's boundaries during the tracking period and all home ranges included habitat designated as green space within their range.

INTRODUCTION

An important focus in the study of wildlife is how animals are structured within the habitat which they rely upon and how they successfully utilize and adapt to that environment for their survival (Rechetelo et al. 2016). Greater knowledge of movement patterns and habitat requirements increases the ability to make effective management plans and document behavioural changes in response to an animal's ecology. An animal's home range is generally defined as the space in which an animal carries out foraging, breeding, and caring for their young (Burt 1943; Ofstad et al. 2016). A home range can vary by individual (e.g., age and sex) or by season, year, population density, and available resources (Burt 1943; Sodhi 1993; Boal and Dykstra 2018). The benefits associated with the establishment of an individual's home range include the enhanced capability to protect and monitor their young and resources, source food efficiently, navigate their surroundings, and evade predators (Powell 2000). Home range can be responsive to nesting density and prey availability, increased prey availability near the nest site may reduce the need for extensive foraging excursions and as nest densities increase, the degree of home range overlap may increase (Mannan and Boal 2000).

Home range analyses examine the core use area and the home range boundary to evaluate an animal's use of their surrounding habitat and can be measured using the utilization

distribution – a method that examines the intensity of habitat use within a given area (Samuel et al. 1985; Clapp and Beck 2015). The core use area is a portion within the home range that is utilized at a greater proportion relative to other areas – it often contains the nest or den and is associated with an animal's territory, often providing information on the most crucial features of that animal's habitat for the sampling period (Samuel et al. 1985).

Monitoring migratory corridors and banding are methods that have been used to track raptors for many years but there are limitations in the data that can be retrieved from these methods due to the hands-off nature for observational monitoring, the reliance of finder reports for banding, and low re-encounter rates of only 3% (Goodrich et al. 2012). Radio telemetry equipment is another method of tracking avian species that has enhanced our understanding of their behaviour. Telemetry data were previously nonexistent for Merlins (*Falco columbarius*) in Manitoba, and although the data are limited across North America, previous studies have examined habitat use and spatial variation of breeding Merlins in both rural and urban areas. Home range sizes can indicate trends in population density and prey availability in an environment which is reflected in the previous research available that has reported smaller home ranges in urban areas where prey and nests tend to be more densely concentrated (Sieg and Becker 1987; Warkentin and Oliphant 1990; Sodhi 1991; Haak et al. in press).

This research aimed to better understand the habitat that Merlins utilize in Winnipeg during the breeding season by examining the spatial movement patterns and identifying the home range and core use area of four breeding male Merlins. These data were collected utilizing new telemetry methods for the species. This telemetry equipment are light weight models and were targeted to retrieve breeding data with limited disturbance to the nest site by means of periodic data retrieval without recapture, temperature reporting and equipment resistance to extreme weather.

METHODS

Four Pinpoint VHF-75 transmitters manufactured from Lotek Wireless Fish and Wildlife Monitoring were deployed. This is the first record of the use of these tags for Merlins. The tags weighed 3.5 g and were programmed in Standard FIX which provides an accuracy of ± 10 m. The battery life can last up to a year, dependent on user-set schedules. The transmitters were

mounted on the tail and attached at the base of the central tail feathers on the ventral side. The transmitter was secured with 4.5 kg test fishing line and epoxy quick dry adhesive. The GPS and VHF fixes were scheduled to record movements every half an hour between 04:30 and 09:30, and 17:30 and 21:30 (CST) to monitor peak activity times, as documented using observational methods and utilizing methods by Sodhi and Oliphant (1992) and as per observation throughout the study. This proposed schedule was anticipated to record a total of 18 positions per day. The data from each transmitter were downloaded remotely on a handheld PinPoint Commander. The remote download was scheduled one day a week, within a four-hour window for each transmitter. Individuals will be referred to using their transmitter identification throughout this chapter.

All four breeding males were trapped near their nest sites (Table 6) using dho-gazza nets with House Sparrows (*Passer domesticus*) to lure Merlins into the net. Trapping occurred during the breeding season at previously confirmed nest sites. The sites were monitored prior to trapping to ensure the nest was currently active and to better understand the pair's behaviour (hunting, roosting periods and preferred perch substrates). Having knowledge of this behaviour aided in trapping success and therefore reduced the time required at the nest site. Site assessment on the day-of trapping took place prior to sunrise to examine the behaviour and determine the Merlin's approximate location but trapping occurred between sunrise until 09:00 May to June. Although trapping occurred in 2018 and 2019, all transmitters were deployed between 25 and 29 May 2019 due to unsuccessful trapping and an apparent West Nile outbreak in the 2018 field season. While individuals were in hand, they were weighed using an abba (breathable cloth used to restrain raptors) and scale, sexed (based on weight or plumage), banded with a federal aluminum band and their wing chord and wing tracings were documented.

All necessary permits were obtained through the Animal Care Committee (ACC) at the University of Manitoba, Permit F18-013 and the Bird Banding Office (BBO), Permit 10876. All methods followed Animal Protocols outlined by and approved by the ACC and the Bander's Code of Ethics (USGS 2020) developed by the North American Banding Council. Care and consideration were taken for each animal involved in this study.

STATISTICAL ANALYSES

Autocorrelation kernel density estimator (AKDE) from the continuous movement model (ctmm) package *ctmm* (Fleming and Calabrese 2020) in RStudio version 1.2.5042 (RStudio Team 2020) was used to examine the home range of the four, radio tagged male Merlins. This is a relatively new method that selects models that are best suited to individual datasets and includes options that consider potential autocorrelation that can be associated with the natural movement patterns of tracked wildlife. The workflow that is outlined by Calabrese et al. (2016) was followed. The ctmm package relies on data to be transformed into telemetry objects and can then be plotted to examine the range of data and outliers. Low satellite signals resulted in data errors that were not included in the results. A total of 26 and 5 data points, respectively were removed due to these equipment errors. Empirical variograms are used to examine the time lags and autocorrelation of the data (Calabrese et al. 2020) (Figure 5). The shape and pattern of the data assists with interpreting the potential appropriateness of each model. After visually assessing the data with the *variogram* function, *variogram.fit* (Fleming and Calabrese 2020) was used to extract the parameters of the variogram into a maximum likelihood estimation (Calabrese et al. 2016). The appropriate models were ranked and selected using an Akaike Information Criterion (AICc). When the models were applied to the data a semi-variogram plot was created to assess asymptotes of the data and determine if home range analysis was suitable for each transmitter. Once the model was selected the smoothing bandwidth is automated within the model. The home range was calculated using the *akde* (Fleming and Calabrese 2020) function which results in a utilization distribution. With these results the estimated 50% core use area and 95% home range boundary was extracted. The home range boundaries were then exported into QGIS 3.10-A Coruña to visualize each individual home range in relation to habitat characteristics and nearby nest sites.

RESULTS

The Independent and Identically Distributed (IID) anisotropic model was selected from the AICc for three out of four tracked Merlins. This indicates that there was no autocorrelation but there was directional persistence in the range of three Merlins. The IID isotropic model was selected for transmitter 44656, which indicated there was no autocorrelation in the data and no patterns of directional persistence. The ctmm model maintains the ability to address

autocorrelation but if data are not autocorrelated, as was observed in these data, a Kernel Density Estimator (KDE) will be run (Fleming and Calabrese 2017). Using visual inspection of the plots produced by *variogram.fit* (Fleming and Calabrese 2020) (Figure 5), the asymptotes of each Merlin are reached at approximately day one of tracking, resulting in the approximate home range boundary at that time.

Due to equipment malfunctions, samples were not recorded every day. The sampling schedule was recorded at sporadic times with an average of two samples per day. Transmitter 44658 was damaged within 10 days of deployment, transmitter 44655 failed after 15 days of data collection and Merlins outfitted with transmitters 44656 (23 days tracked) and 44567 (45 days tracked) had failed nest sites, resulting in nest abandonment. The mean home range size of the four tracked Merlins was 1.79 km^2 (.67 – 6.95) and the mean core use area was .26 km^2 (.09 – 1.10). The mean number of data points collected was 39.75 (21 - 86) and the mean number of days tracked was 24.25 (10- 45) (Table 6). Movement from the nest appeared to become larger throughout the throughout the tracking period; transmitter 44657 recorded the largest range, approximately 4 km from the nest site. Transmitter 44657 was tracked 45 days, the longest tracking period within this study. Transmitters 44655 (Figure 6) and 44658 (Figure 9) were elongate shapes, transmitter 44656 was a circular home range (Figure 7) and 44657 (Figure 8) was an oval shape that contained multiple activity areas.

All home range boundaries included habitat that is designated as a park or greenspace, which could be habitat that was utilized for hunting; 44656 and 44657 included surface water features within their boundary, habitat that may have increased biodiversity and 44657 encompassed a neighbouring nest between the minimum confidence interval boundary for the home range and just outside the perimeter of the core use area. Three Merlins remained within the city perimeter and 44658 traveled approximately 1 km outside of the city more than one time (Figure 9). The core use area was centered around the nest site for all four birds.

DISCUSSION

The documented breeding home ranges of Merlins across North America show variability among the different study areas. Differences in prey abundance, inter and intraspecies nest density and vegetation/habitat types are factors described to influence home range behaviour (Sieg and Becker 1987; Sodhi 1991; Sodhi and Oliphant 1992; Sodhi 1993). The literature examined within this study contrast the home range sizes in both rural and urban habitats. Urban

home range sizes for raptors are theorized to be reduced relative to rural home range sizes (Rutz 2006; Boal and Dykstra 2018). Variables suggested to influence a reduction in urban home range size include increased prey abundance (Boal and Dykstra 2018; Sodhi 1993) resulting from increased growth periods of vegetation due to the higher temperatures in urban habitats and increased and predictable food availability from anthropogenic sources (Møller et al. 2012), and higher nest densities of conspecifics (Mannan and Boal 2000). It has also been suggested that noise pollution in urban environments may increase hunting success and therefore decrease the home range of predatory birds (Roth et al. 2008). The breeding home range of urban male Merlins in Saskatoon, SK within city limits reported mean ranges of 6.3 ± 1.3 km^2 (Sodhi 1991) and the breeding home range of rural Merlins in Montana reported mean ranges of 21.3 km^2. The mean breeding home range found within this study was 1.79 km^2, a far reduced range relative to the ranges reported in Saskatoon and Montana.

The mean home range and core use area across all sampled individuals was HR $\bar{x}$ = 1.79 km^2, CR $\bar{x}$ = .26 km^2; the core use area included the nest site four all four tracked birds. Individually, the home range boundary and core use area ranges of the transmitters were 44655 (HR $\bar{x}$ = 1.04, CR $\bar{x}$ = .14 km^2), 44656 (HR $\bar{x}$ = 1.42, CR $\bar{x}$ = .17 km^2) and 44658 (HR $\bar{x}$ = 1.15, CR $\bar{x}$ = .17 km^2) exhibited the most similarities. The range of tracking period was 10-23 days for the three individuals. Transmitter 44657 was tracked for 45 days total and exhibited the largest range from all four samples (HR $\bar{x}$ = 3.53, CR $\bar{x}$ = .55 km^2). In mid-June, transmitter 44657 became harder to locate, requiring download attempts up to 700 m away from the nest tree. This was unusual as prior to this time, both adults were within sight or range of download inside 10 minutes of being on site. By early July both adults had dispersed from the nest site and no fledglings were present, indicating that the nest had failed. The spatial patterns recorded later in the season appeared less habitual and more exploratory than what was documented at other sites or had previously been observed earlier in the season for this individual. This spatial behaviour may have been associated with the outcome of the nest rather than what may be a typically observed pattern throughout the breeding season. There appears to be individual variability in home range behaviour, but prey abundance is a factor that is associated with smaller home ranges. As prey abundance decreases later in the breeding season and females begin hunting again, home range sizes often increase (Sodhi 1993).

The home range shapes of individuals 44655 and 44658 were elongate and the home range shape of 44657 was elongate with multiple activity areas. All three followed a directional persistence which indicated that the three Merlins may have been utilizing specific foraging locations during the sampling period. The home range shape of transmitter 44656 was circular and did not indicate a directional persistence indicating that this Merlin may have been utilizing a number of different locations for foraging (Figs 6:9).

Spatial overlap of neighbouring Merlin's home ranges is not uncommon as observed in previous research on the home range of breeding male Merlins (Sieg and Becker 1987; Sodhi 1991; Sodhi and Oliphant 1992). One study reported Merlins exhibiting spatial overlap to have shared hunting sites on the same day but were not documented there at the same time (Sodhi 1991). Range overlap may increase Merlin hunting opportunities, success rates and the flight behaviour of their prey base makes prey defense difficult (Sodhi and Oliphant 1992). The degree of overlap can vary, within Saskatoon it was hypothesized that Merlins with nesting distances beyond 2 km did not have spatial overlap (Sodhi and Oliphant 1992) and overlap for Merlins nesting over 1 km away from one another did not overlap extensively (Sodhi 1993).

Within this study the mean distances between nests, as reported in Chapter 2 was 1.67 km (Table 5) which fits within the mean home range for the sampled individuals in this study ($\bar{x}$ = 1.79 km^2). This suggests that in some cases Merlins may be adjusting their home range size to neighbouring nests. However, home range overlap can be predicted in some cases within this study due to the short neighbouring distances between nest sites. The distribution of closely clustered nests between transmitters 44655 and 44658 (Figure 6) in the Transcona ward indicates the likelihood of home range overlap, as the distance between the two nests was only 155.93 m. Another example observed within this study was the home range documented for transmitter 44657 which encompassed one nest site within the home range boundary (Figure 8), the nest sites were located 739.40 m from one another. With consideration to both examples, the pooled nearest neighbour distances across all years reported the mean minimum distance between nest sites to be 161.64 m and the mean distance was 1.67 km (Table 5). These two distance parameters are within the mean home range ($\bar{x}$ = 1.79 km^2) of the four tracked Merlins within this study. Over the three years of this study, only three incidences of territorial displays were observed. Furthermore, the nests within these examples are both under 1 km away, which

suggests there may have been significant spatial overlap (Sodhi 1993) but without further tracking data, it is impossible to know how this habitat would have been shared between individuals of neighbouring nests.

CONCLUSION

The timeframe that was documented likely examined a period during the breeding season that relies on shorter and more frequent travels from the nest site to provide sufficient food for the female and nestlings. Behaviour observed during the breeding season is adaptive to the external environment and factors such as fluctuating prey abundance and the developmental stages of young can influence home range sizes. The small variation in spatial patterns and consistencies documented between the four tracked Merlins suggests that this research obtained an accurate snapshot of breeding male Merlins within the city of Winnipeg during the post incubation and pre-fledging period. The inclusion of green space within each track bird's home range demonstrates the importance of those landscape features within the urban environment of Winnipeg.

Table 6. Bird ID, total recorded locations (n), total days with successful data location, sampling interval in 2019 season, model selection from AKDE and the mean 95% home range boundary and mean 50% core use area with range of four tracked male Merlins.

Bird ID	**Capture location**	**Data points (n)**	**Days tracked**	**Sampling period**	**Model**	**Home range (km^2)**	**Core use (km^2)**
44655	49.89868, -97.01606	26	19	05/26 – 06/13	IID anisotropic	1.04 (.67 - 1.48)	.14 (.09 - .19)
44656	49.88312, -97.16939	26	23	05/28 – 06/19	IID isotropic	1.42 (.92 - 2.02)	.17 (.11 - .24)
44657	49.86037, -97.12477	86	45	05/26 – 07/10	IID anisotropic	3.53 (1.25 - 6.95)	.55 (.20 – 1.1)
44658	49.90329, -96.98056	21	10	05/28 – 06/07	IID anisotropic	1.15 (.69 – 1.72)	.17 (.11 - .26)
Total		39.75	24.25			1.79	.26

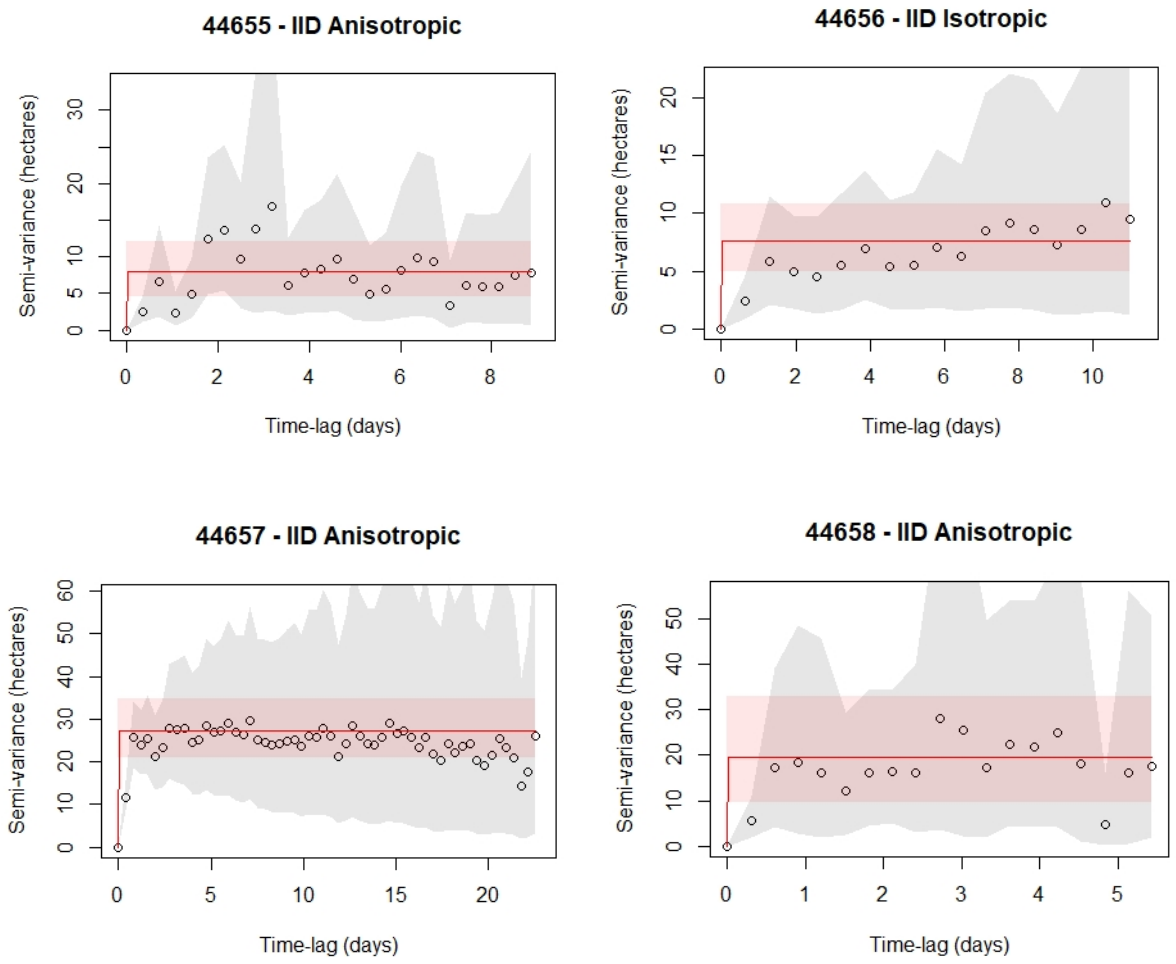


Figure 5. Semivariogram plots showing the asymptotes at approximately one day of tracking establishing the approximate home range boundary within that period.

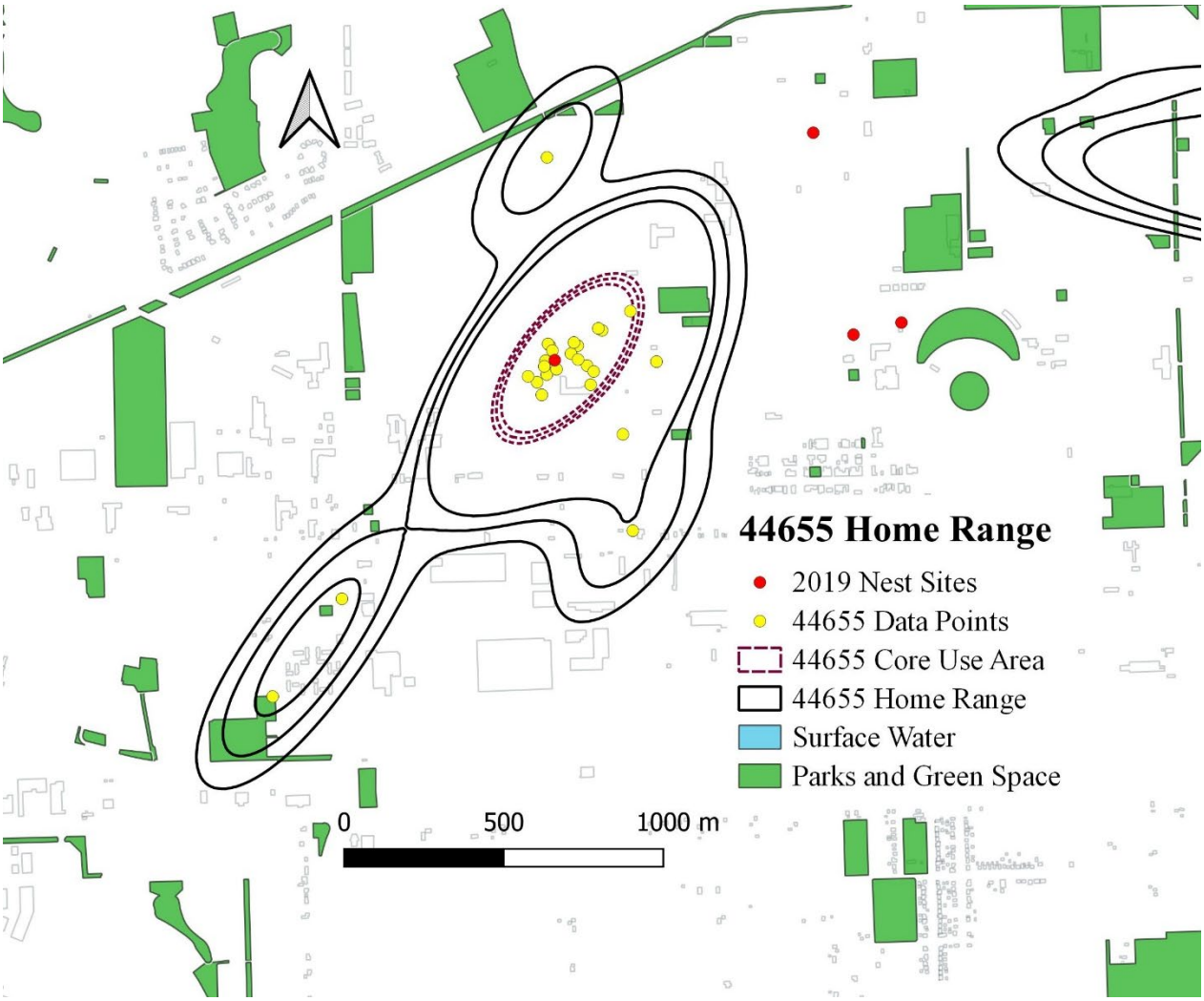


Figure 6. Mean home range (1.04 km^2) and core use area (.14 km^2) of transmitter 44655. Black lines demonstrate the 95% confidence interval of the home range and purple dashed lines demonstrate the 50% confidence interval, as designated as the core use area. Merlin was tracked for 19 days with a total of 26 locations recorded between 26 May and 13 June.

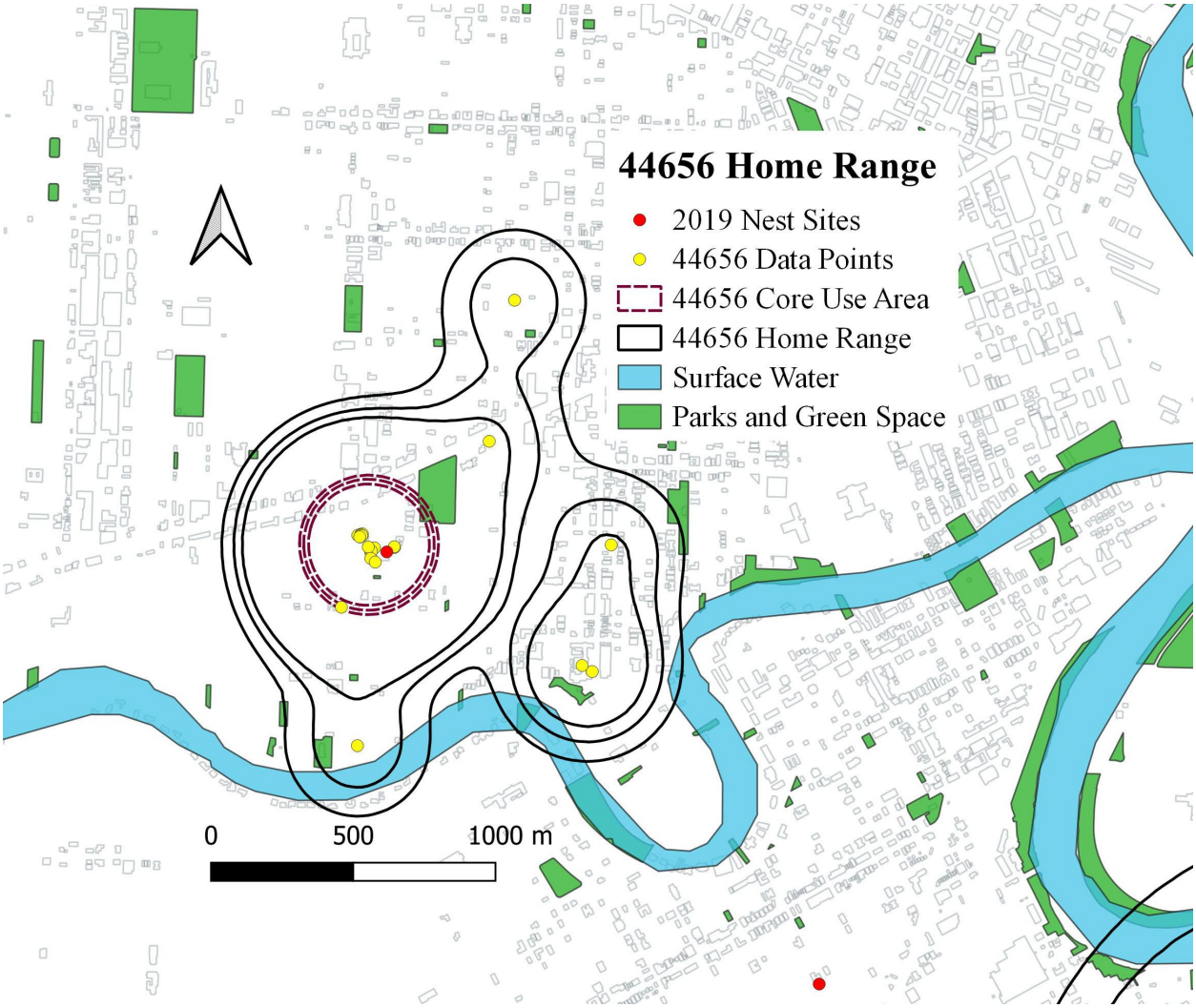


Figure 7. Mean home range (1.42 km^2) and core use area (.17 km^2) of transmitter 44656. Black lines demonstrate the 95% confidence interval of the home range and purple dashed lines demonstrate the 50% confidence interval, as designated as the core use area. Merlin was tracked for 23 days with a total of 26 locations recorded between 28 May and 19 June.

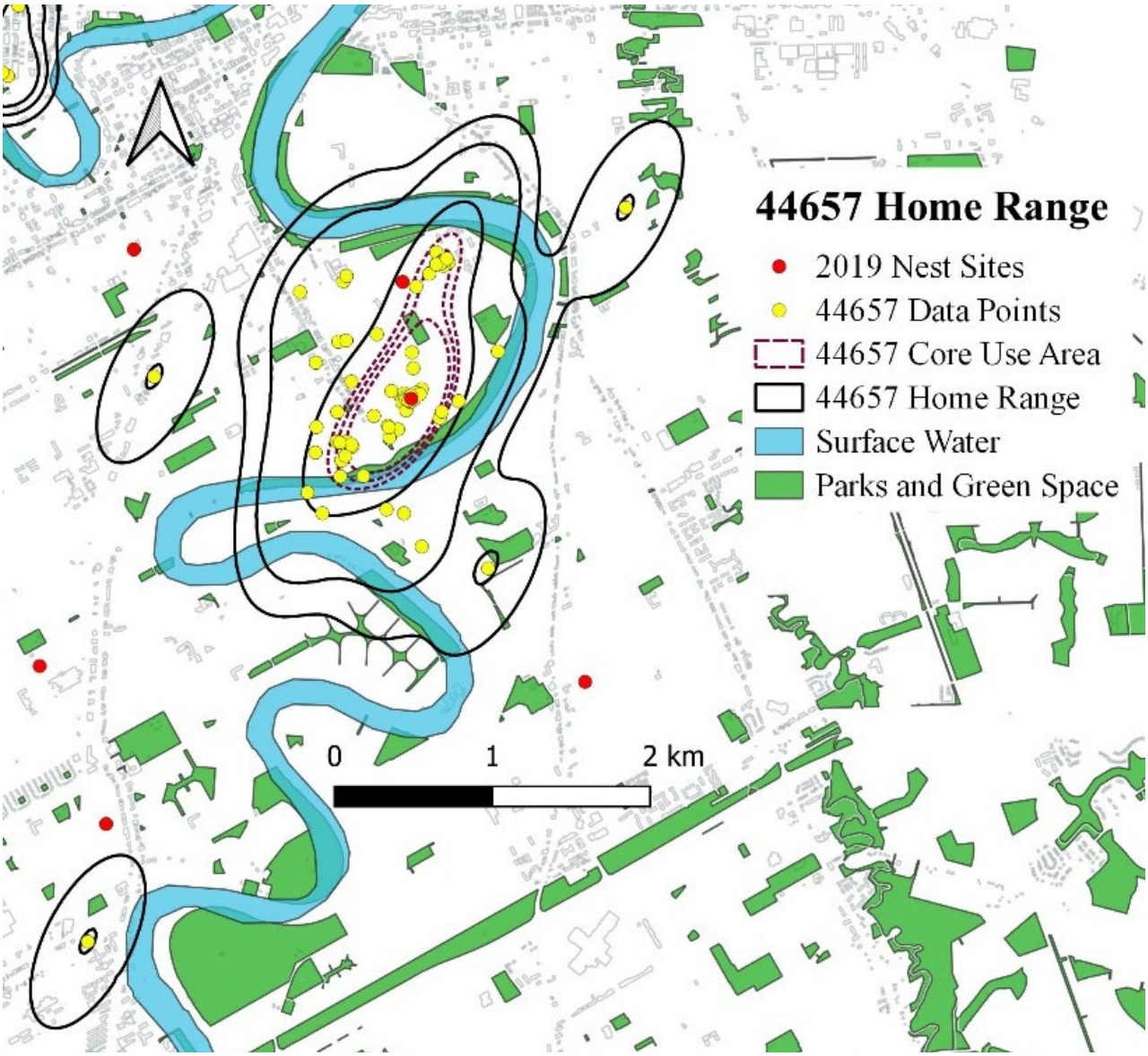


Figure 8. Mean home range (3.53 km^2) and core use area (.55 km^2) of transmitter 44657. Black lines demonstrate the 95% confidence interval of the home range and purple dashed lines demonstrate the 50% confidence interval, as designated as the core use area. Merlin was tracked for 45 days with a total of 86 locations recorded between 26 May and 10 July.

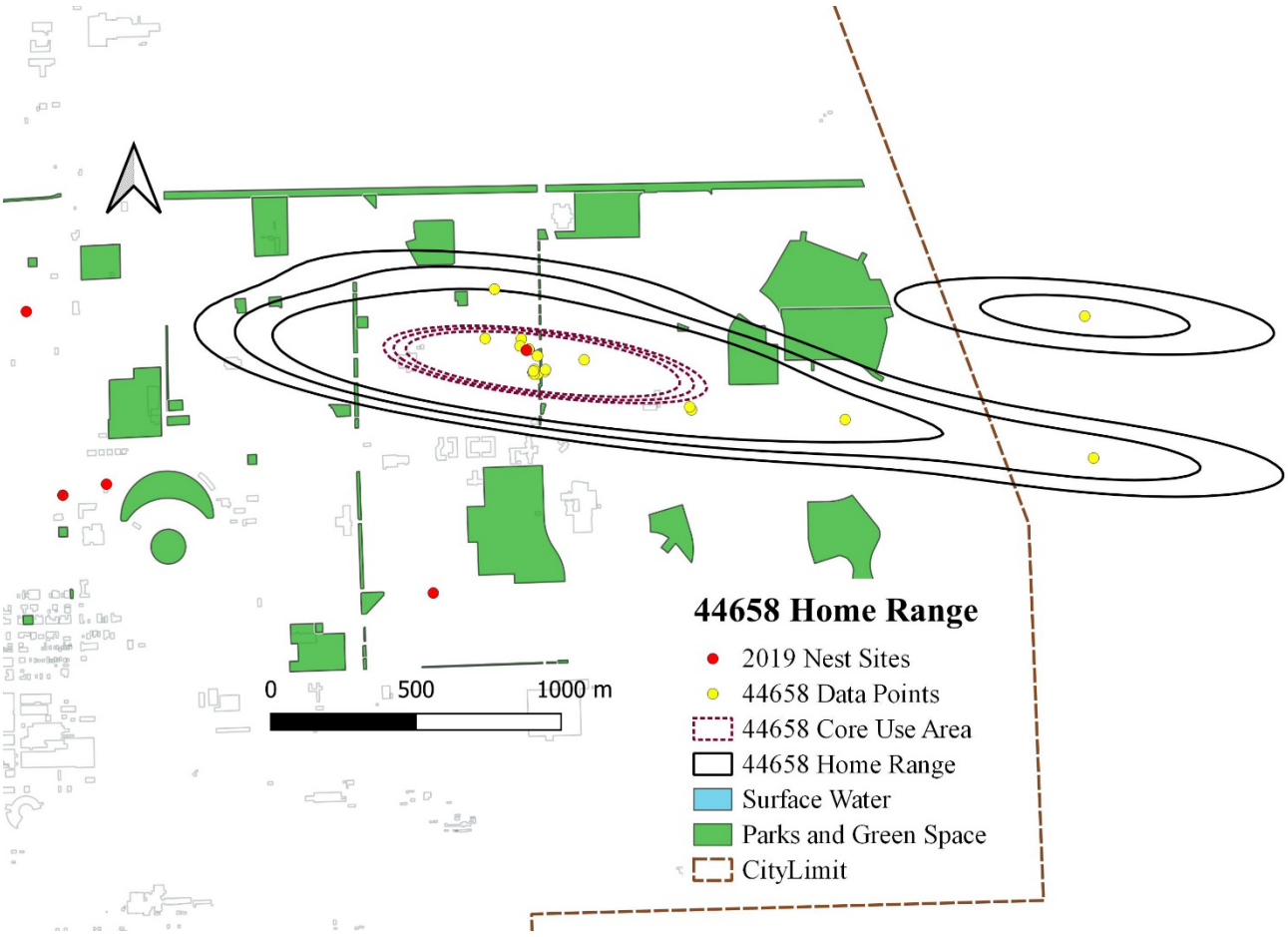


Figure 9. Mean home range (1.15 km^2) and core use area (.17 km^2) of transmitter 44658. Black lines demonstrate the 95% confidence interval of the home range and purple dashed lines demonstrate the 50% confidence interval, as designated as the core use area. Merlin was tracked for 10 days with a total of 21 locations recorded between 28 May and 7 June.

CHAPTER FOUR – POPULATION MONITORING AND WEST NILE VIRUS

ABSTRACT

The impact West Nile virus (WNV) has on wildlife can vary based on weather conditions, species interactions, habitat and individual response, among other factors. During the 2018 breeding season, an apparent WNV outbreak appeared to be associated with the mortality of juvenile Merlins (*Falco columbarius*) in Winnipeg, Manitoba, Canada. Merlin specimens found within this area were necropsied at the Canadian Wildlife Cooperative Health in Saskatoon, Saskatchewan, Canada. The results found that 78% of the Winnipeg sample (n=32) tested positive for WNV and 66% of that total were juveniles. Of the recorded and monitored nests within the study area during 2018, only 9% of nests appeared to have no juvenile mortalities prior to the fledging period. Juveniles that were admitted to Prairie Wildlife Rehabilitation Centre and Wildlife Haven Rehabilitation Centre in Manitoba with apparent WNV symptoms and were deemed 'releasable' were banded. This research outlines the details of those band records and all other Merlins banded during this study.

INTRODUCTION

Wildlife that inhabits urban environments can be at higher risk of transmission and the impacts of disease as compared to those inhabiting rural environments (Bradley and Altizer 2007). Factors within the urban environment that contribute to this susceptibility include higher temperatures creating ideal breeding conditions for parasites and vectors, varying habitat quality (Gil and Brumm 2013), reduced habitat resulting in increased contact with conspecifics (Bradley et al. 2008), and decreased biodiversity (Bradley and Altizer 2007). Trichomoniasis (Bradley et al. 2008) and West Nile virus (WNV) (Bradley and Altizer 2007) are two examples of diseases that can impact avian species inhabiting urban environments at greater rates than those in rural environments. During the 2018 field season of this study, an apparent WNV outbreak occurred within Manitoba.

WNV is a vector born virus first documented in North America in 1999 (Kilpatrick et al. 2006). WNV requires a mosquito vector and an amplifying host such as an avian species.

Mammalian species act as a 'dead end' host in the cycle as mammals do not carry a sufficient level of the virus within their blood to continue the cycle (Taieb et al. 2020). High mortality rates are observed in avian species that have contracted WNV but fewer WNV mortalities are observed in raptors relative to other avian groups (Nemeth et al. 2008). Avian species within the population that survive infection can develop anti-bodies to the virus that can be sustained for different lengths of time depending on the species. This can vary based on life history traits and have been reported to last up to four years in raptors (Nemeth at al. 2009). As more individuals in populations develop WNV antibodies, herd immunity and passive immunity to young from mothers can reduce the impacts of WNV outbreaks on a population (Nemeth et al. 2008).

WNV prevalence is found to be greater in urban environments as stagnant water, dense vegetation, mature trees and sewer systems associated with older housing are all features that provide suitable habitat for mosquitoes (Bradley et al. 2008)**.** The impacts of climate change can have an impact on WNV outbreaks. Rising temperatures and greater precipitation can increase mosquito populations and can reduce the larvae incubation length. Conversely, drought can make outbreaks more severe as larvae lay dormant until ideal conditions exist, and fewer water sources can attract greater diversities of mosquito and avian species to the same water resources (Paz 2015).

This research investigates the cause of a large mortality of juvenile Merlins (*Falco columbarius*) during an apparent WNV outbreak in the city of Winnipeg. Specimens that were found deceased or died in care at wildlife rehabilitation centres tested for WNV at the Canadian Cooperative Wildlife Health Centre in Saskatoon. Merlins that were admitted to local wildlife rehabilitation centres with suspected WNV symptoms and were deemed 'releasable' were banded. These data were used to examine potential limiting factors for Merlins within this study area and better understand the vulnerability of Merlins to WNV and other factors.

METHODS

WEST NILE VIRUS

Between July and August 2018, nestling and fledgling Merlins were found to be suffering from neurological issues, starvation, pinched off feathers, immobility and morality near nest sites within Winnipeg. These are symptoms that are commonly observed in birds that have contracted WNV (Vidaña et al. 2020). Many reports of Merlins suffering from these symptoms and juvenile

mortalities were made by residents within Winnipeg due to the residential inhabitance of Merlin nests. Local wildlife rehabilitation centres were contacted by residents and contacts within this study area alerted me to their observations. Live Merlins were transferred to local wildlife rehabilitation centres where palliative care by the employees and volunteers at the centres was provided.

Each nest was monitored to determine if adults, juveniles or nestlings appeared to be displaying these symptoms or mortality. All Merlins and any other wildlife that had died at the nest site were bagged, labeled and frozen. In some cases, specimens had already been thrown out by the property owner either prior to making the report or prior to the arrival of pick-up. Those specimens were not included in the final mortality record as the species identification was confirmed first hand. Nests were categorized as 'survived', 'mortality' or 'abandoned'. Nests were designated as 'survived' if all young fledged from the nest, 'mortality' if some/all young died at the nest site or in wildlife rehabilitation centres and 'abandoned' if the adults dispersed from the nest site prior to the fledging period and there were no young observed at the nest site.

A Wildlife Export Permit from Manitoba Sustainable Development was obtained for transfer of the specimens to Saskatoon for necropsy by Dr. Trent Bollinger, wildlife veterinary pathologist at the University of Saskatoon and Director of the Western and Northern Region of the Canadian Cooperative Wildlife Health Centre (CCWH). In the lab polymerase chain reaction (PCR) tests were run on liver, spleen, heart, brain and kidneys. Histological examinations were conducted by CCWH lab employees to examine for lesions of WNV.

MARK AND RECAPTURE DATA

Healthy Merlins were trapped at their nest sites and those that were admitted into wildlife rehabilitation care and categorized as 'releasable' into the wild were banded with a Federal aluminum bird band. Lock-on size 4 bands were used for females and butt-end size 3A for males. However, discretion was used if sex was unknown, or the individual had larger tarsus than what was commonly observed in males. Sexing juvenile Merlins using weight was more difficult for rehabilitated birds, likely as a result of decreased muscle mass and increased food availability. Banding took place between May and September 2018 and 2019. A dho-gazza net was used to trap adults and fledged juveniles. Nestlings were banded at the nest tree within the first to second week of July at the nest site; the nest tree was climbed, and the nestlings were transferred down

in bird bags. When banding juveniles and nestlings at the nest site, nest sites were observed to ensure that feedings were not taking place. The first feedings of the day generally occurred around sunrise - banding would occur just after sunrise when the adults were less active at the nest site but early enough that the temperatures had not peaked. When young were in hand, they were weighed using an abba, banded and their general condition was assessed before bringing them back to the nest. When Merlins were admitted to local wildlife rehabilitation centres due to injury, illness or unknown reasons and categorized as "releasable", they would be banded and brought back to their nest site or appropriate habitat for release. If the Merlin was a juvenile or nestling and released back to the nest, the nest site would be monitored to ensure no further issues occurred and nestlings had not prematurely fledged from the nest.

RESULTS

WEST NILE VIRUS

A total of 35 deceased Merlin specimens were collected within the city of Winnipeg and 13 live Merlins were admitted to wildlife rehabilitation centres for care and released late in the season. Reports to myself and the wildlife rehabilitation centres recorded 21 more specimens were found deceased at the nest sites but had been thrown away prior to arrival. The sample submitted for necropsy from Winnipeg included 17 females, 9 males and 6 of unknown sex; two specimens were too desiccated to conduct necropsies on (both of unknown sex); for a total of 32 necropsied Merlins. The PCR test reported WNV positive results for 25 specimens, included in these results are one specimen that was reported as was 'presumptive positive', and seven negative WNV results were reported (Table 7). The cause of death for the WNV negative specimen included aspergillosis (4), necrosis (1), crop inflammation resulting in necrosis (1) and unknown for the remaining specimens. Two House Sparrows (*Passer domesticus*) were necropsied that were found beside the body of a deceased juvenile Merlin at a nest site in the city of Winnipeg; both House Sparrows tested positive for WNV.

A total of 25 nests were categorized as 'mortalities' – 19 appeared to have all young deceased and 6 appeared to have cases of both deceased young and fledglings, based on observation. A total of seven nests were categorized as 'abandoned' prior to the fledging period and three nests were categorized as 'survived' and appeared to have all young fledge from the nest (based on no moralities on site and the number of fledglings 4+) (Table 9).

MARK AND RECAPTURE

A total of 44 Merlins were banded between 2018 and 2019: 22 females and 22 males. A total of 21 Merlins were banded prior to release from rehabilitation centres, this total included 14 females (12 juvenile; 2 adults) and 7 males (5 juveniles; 2 adults). A total of 23 Merlins were banded at the nest site, this total included 15 male Merlins (5 juveniles; 10 adults) and 8 females (4 juveniles; 4 adults). During the WNV outbreak 13 Merlins that were admitted to the rehabilitation centres with WNV symptoms were banded and released at their nest sites or suitable habitat (Table 8).

Six band reports have been made as of October 2020. Three Merlins were reported as deceased during the WNV outbreak. A juvenile Merlin that had been admitted to a wildlife rehabilitation centre with apparent WNV symptoms in the summer of 2018 and was found deceased in Iowa, USA on 5 January 2019. A Merlin was reported deceased due to an apparent window collision in the summer of 2019; the Merlin had been banded at the nest on 4 July 2018 and found approximately 14 km from the nest site in Transcona August 2019. An adult male that was banded at the nest site in June 2019 was reported as a mortality in a residential back yard near a bird feeder approximately 550 m from the nest site in October 2020. Two banded Merlins have been observed within previously utilized breeding territories during the breeding season of 2020, but their unique identification number was not decipherable.

DISCUSSION

WEST NILE VIRUS

There are records of the impact of WNV on Merlins, but I am not aware of any other studies that have reported such a mass mortality within one area. Previous studies have documented Merlins to be one of the most common avian species to become seropositive – meaning they have detectable antibodies within their blood stream (Taieb et al. 2020). Avian populations in urban areas are 2.5 times more likely to become seropositive than populations in rural areas (Bradley et al. 2008). This is suggested to be related the exposure associated with the abundance of *Culex* mosquitoes in urban environments (Government of Manitoba 2020). Decreased species diversity in urban habitats may result in greater severity on populations of higher densities but a reduced mortality rate overall on populations as the impact may be more

localized due to the reduced foraging and movement exertion requirements for urban inhabitants due to accessible food sources, like bird feeders (Bradley et al. 2008).

Within this study it is possible that sample bias influenced the high rates of mortality reporting. The intensive documentation of Merlin nests within the city and increased resident awareness of Merlin presence due to research communication may have heightened awareness of the outbreak. The impact of the outbreak appeared to disproportionally impact juvenile Merlins. Necropsy results documented 66% of WNV positive specimens to be juveniles. Nestlings can be more susceptible to WNV as they are sedentary in one location and have immune systems that are more vulnerable that adults (Soltész et al. 2017). A multitude of factors may have influenced the severity of this outbreak – weather conditions, the clustered distribution of nest sites, and features in the urban environment like stagnant water. Record setting temperatures and dry conditions were reported during the summer of 2018 (Global News 2018), both are conditions that can amplify outbreaks and shorten incubation periods (Fenech and Chiotti 2012). Interestingly, 2018 was a relatively low count year for mosquitoes within the city of Winnipeg as per the count traps (City of Winnipeg 2020a) but higher than average WNV positive mosquitoes were recorded within city traps; 39 in 2016, 41 in 2017 and 168 in 2018 (Government of Manitoba 2020).

Raptor species that rely on avian prey are associated with higher rates of infection and mortality to WNV due to oral transmission of WNV infected prey items (Vidaña et al. 2020). The WNV positive House Sparrows necropsied within this study found at a Merlin nest supports speculations that oral transmission in addition to mosquito-borne transmission could have affected the Merlin population at a higher proportion level due to their predatory behaviours and reliance on prey items like House Sparrows. House Sparrows are a common host to WNV due to their abundance and the nature of their life history traits reduce the likelihood of local population herd immunity; due to shorter lifespans and higher population turnover (Nemeth et al. 2009). However, the species is documented to have a lower mortality rate relative to their abundance (Kilpatrick and Wheeler 2019) which may increase the likelihood of passing WNV on to predator species like Merlins.

Monitoring the prevalence of WNV on avian species can be used as a predictor into the potential impact of WNV on humans. Mosquitoes often select avian species early to mid-summer, in relation to their abundance and sedentary nestling behaviour. As summer progresses,

mosquitoes will often shift their preference to mammalian species, like humans, as birds disperse across their breeding territories and begin migration in late summer to early autumn (Kilpatrick et al. 2006; Thiemann et al. 2011; Taieb et al. 2020). There are limited studies on the long-term impacts of WNV on avian populations but based on the data that are available the impacts appear to be low to moderate (Kilpatrick and Wheeler 2019). The impacts of juvenile mortality observed at 71% of nest sites over the breeding season in 2018 within this study are still unknown at this time, to date significant effects on the breeding population has not be observed as per my field surveys in 2019 and data collection by Dr. Shoesmith in 2020 (M. Shoesmith personal communication). Over time the mortality of breeding adults may result in a decrease in future breeding populations. However, at this time it appears that the population was resilient to these mortalities.

MARK AND RECAPTURE

The sample of 44 banded Merlins that bred or hatched within the city of Winnipeg builds a foundation of potential data resources that extend beyond the time frame of this study. WNV results from this study demonstrate the potential vulnerability of juvenile Merlins to variables such as disease transmission in the urban environment. Additionally, previous studies have documented juvenile vulnerability with mortality rates of 70% versus adult mortality rates of 30% as documented in Saskatoon (Warkentin et al. 2020) and survivorship of Merlins over one year of age to be estimated at 62% by Stephens and Anderson (2002) .

These data may offer insight into the habitat selection and spatial distribution post-breeding season and the limiting factors that Merlins may be subjected to within Winnipeg and other utilized habitats. Unfortunately, these may be far-reaching expectations to obtain from banding data based on low band recovery rates (Goodrich et al. 2012) and the small band sizes that are required for Merlins. Without unique identifiers such as colour bands or leg markers, identifying individuals from a distance is nearly impossible. However, the urban inhabitance of Merlins increases the likelihood of human contact with injured or deceased wildlife and this factor may increase the likelihood of potential band reports. At this time, six bands have been reported, representing 14% of the total sample of banded Merlins. Continued investment into these methods are a relatively cost-effective way to monitor urban populations with the objective of obtaining long-term understanding of Merlins within Winnipeg.

CONCLUSION

The vulnerability of juvenile Merlins appeared to have contributed to the mortality that was observed over the breeding season. At this time, the loss does not appear to have affected the population but increasing temperatures related to climate change may increase the chances of WNV outbreaks occurring. Continued monitoring of the population can offer the opportunity to better understand what these outbreaks would mean long-term for populations like this one. WNV was one factor that documented the mortality of Merlins within Winnipeg. However, banding records have demonstrated that there are a number of different factors that can put Merlins at risk such as collisions with windows and other anthropogenic landscape features. Understanding those risks to Merlins may have broader implications to other avian species that inhabit urban environments.

Table 7. WNV results from necropsy on Merlin samples collected within Winnipeg. A total of 25 (78%) Merlins tested positive and 7 tested negative (22%). A larger proportion of females (53%) and juveniles (78%) were represented in this sample. ASY = known to have hatched earlier than the calendar year, hatch year unknown and Unk = Unknown.

	Positive	Negative	Total
Female Juvenile	11	3	14
Female ASY	0	1	1
Female Unk. Age	2	0	2
Male Juvenile	7	1	8
Male ASY	0	0	0
Male Unk. Age	0	1	1
Unk. Sex Juvenile	3	0	3
Unk. Sex and Age	2	1	3
Total	24	7	32

Table 8. Overview of all banded Merlins: separated by general banding location, approximate age (HY = known to have hatched in the calendar year, ASY = known to have hatched earlier than the calendar year, hatch year unknown), total released from rehabilitation centres after apparent WNV infection and total from sample reported as mortality.

	Female	Male	Total	Proportion
Banded at the nest	8	15	23	52%
Banded at rehab centre	14	7	21	48%
Hatch-year (HY)	16	10	26	59%
After second year (ASY)	6	12	18	41%
Release from rehab centre post-WNV	8	5	13	30%
Mortality (WNV and other)	3	3	6	14%

Table 9. Total number of nests with suspected survival of all young to fledgling period (9%), mortalities of some or all young (71%) and sites abandoned prior to fledging period (20%).

	Survived	Mortalities	Abandoned
Total nests	3	25	7
Proportion of nests	9%	71%	20%

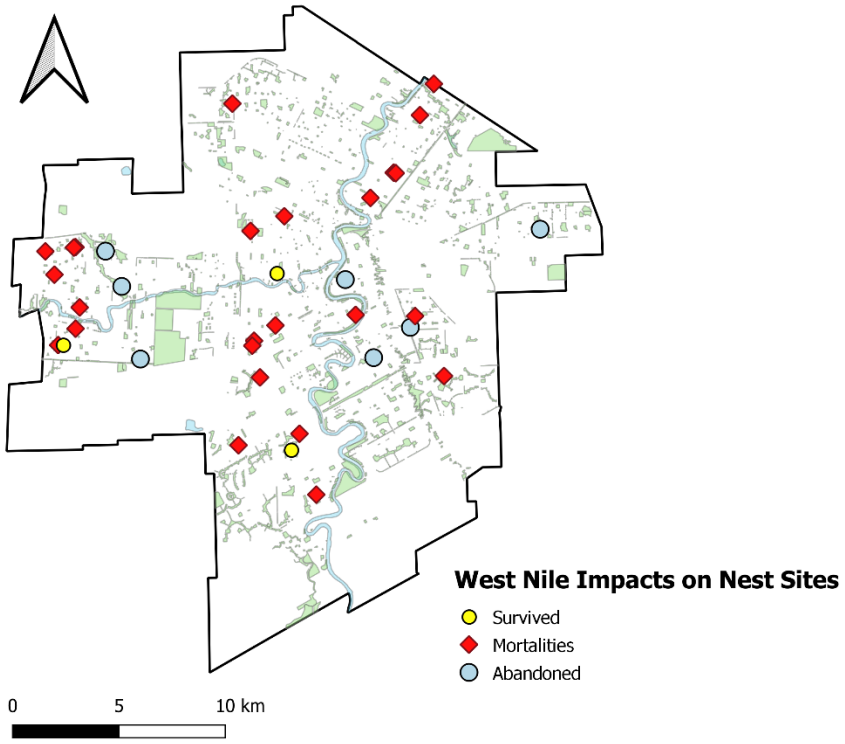


Figure 10. The outcome of 2018 nest sites impacted by WNV. Site designated as survived (n = 3); all young fledged from the nest, mortalities (n = 25); all or some young died and abandoned (n = 7); adults dispersed prior the fledging period with no young observed.

CHAPTER FIVE

SUMMARY AND FUTURE DIRECTIONS

This study has documented the key habitat features of nesting Merlins to be mature spruce trees with unused crow nests and the area within 25 m of the nest tree to contain fewer trees - specifically fewer coniferous species - at Merlin nest sites. These features likely aid in nest accessibility and an enhanced vantage point of the surrounding environment for this visual predator. The dispersal of nest sites across the study area in all wards except one, indicates that there is suitable habitat and available nest sites throughout Winnipeg. Patterns of nest clustering and nearest-neighbour distances suggest that nests may be limited, or that areas of the city may contain higher quality habitat than others.

The home range of breeding males documented within this study demonstrates the importance of the habitat and resource availability within the approximate 1.79 km^2 boundary surrounding the nest site. In addition, the high degree of habitat utilization within the core use area which ranges approximately 260 m surrounding the nest site emphasizes the significance of that space during the breeding period. While the distribution and range of males during the breeding season suggests home ranges overlap, few intraspecies conflicts were observed at monitored nests during this study suggesting there is conspecific tolerance to individuals at neighbouring nests.

The outcome of a WNV outbreak during this study and band reports outline the potential areas of vulnerability to include juvenile exposure to viruses such as WNV, collision with anthropogenic sources such as windows and unknown variables that can take place in overwintering habitats, as documented by a deceased Merlin in Iowa.

The data produced from this study developed baseline information to enhance the understanding of Merlins within the city of Winnipeg. However, this was a brief interpretation of a complex and large study area and there are many aspects to consider for future research. These areas of interest include resource allocation of prey availability and nest sites, nest selection and behaviour of crows, and education on habitat and Merlin life history timelines for urban residents and city planners that will assist in the development of urban forestry management and residential landscaping.

Research on songbird abundance within Winnipeg and prey preferences for Merlins would provide a less species-specific approach and a broader perspective on the environment.

Pairing these questions with productivity may gather understanding of habitat quality and utilization in relation to foraging and the influence prey has on nest density and distribution.

Interspecific interactions between Merlins and crows would be an invaluable component to better understand the nest habitat and availability for both species. Continued development on the data of the specific nesting habitat selection of crows would create data that would allow researchers to cross examine habitat selection between the two species. Spatial and temporal nest occupancy, and interspecies tolerance would develop an in-depth and long-term understanding of future habitat use.

Data sharing and educating the public, arborists and city planners are crucial components to effectively manage wildlife in an urban environment. Residents who have Merlins nesting in their backyard are exposed to daily contact with a breeding pair. Public perception of Merlins varies widely, as they are a vocal species that is active early in the morning and prey on songbirds. Their loud nature and predatory behaviour can result in residents removing a nest tree from their yard. If this decision is made, it is important for tree removal to occur post-breeding as the destruction to a falcon nest is illegal under the Manitoba Wildlife Act Section 49 (Province of Manitoba 2021). Tree removal during the season is disruptive to the nest and may result in nest failure or mortality. Experience obtained through this study provided insight in the value of education and communication with residents to explain the value of these mature trees and the ecological significance Merlins contribute to the environment. Often, once this information was provided, resident perception of the species appeared to become more favourable. The accessibility of nesting raptors on residential property offers many opportunities for data collection and reporting. The continued investment into community citizen science and education efforts put forward by Dr. Merlin Shoesmith and through contacts made during this study builds a foundation for this future research. Finally, the loss of mature trees within the city of Winnipeg as a result of disease or damage and landscape preferences away from large trees that can sustain biodiversity in new neighbourhoods are likely to be the greatest limiting factors within the city of Winnipeg. Management and protection of existing mature trees, in particular spruce trees, and urban planning that encompasses the value in planting diverse tree communities that are resilient to disease and are likely the most effective management strategies for this species.

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APPENDIX A

Means, standard deviations and ranges of all 21 study variables and p-values produced from the ANOVA at occupied and unoccupied nests (n=30 for both sample groups). All measurements were taken or standardized to metres and coniferous and deciduous variables are count data.

Variable	Occupied	Unoccupied	*p*-value
NTheight	15.75 ± 2.6 (9.6 – 21.6)	12.69 ± 3.55 (7.0 – 18.2)	0.000162
Ndiameter	1.37 ± 0.23 (.94 – 1.92)	1.05 ± 0.25 (.48 – 1.51)	0.001888
Nheight	10.97 ± 2.23 (7.0 – 17.4)	9.146 ± 3.63 (3.1 – 13.98)	0.097161
DistTallestC	4.27 ± 6.90 (1.63 - 24.66)	12.02 ± 8.0 (1.15 – 24.8)	0.517627
TallestC	13.01 ± 4.68 (5.0 – 22.2)	13.58 ± 5.08 (4.9 – 27.88)	0.697758
MeanHeightC	10.67 ± 3.76 (5.0 – 18.60)	9.97 ± 3.74 (4.61 – 21.39)	0.234077
MeanDistC	14.23 ± 5.25 (2.87 – 23.54)	12.33 ± 6.38 (2.07 – 24.72)	0.746370
MeanDiamC	.85 ± .39 (.43 – 1.64)	.96 ± .79 (.35 – 1.23)	0.082473
DistTallestD	7.82 ± 6.38 (3.51 – 24.99)	18.29 ± 6.14 (5.63 – 24.95)	0.782583
TallestD	14.99 ± 3.87 (8.25 – 21.89)	13.60 ± 5.98 (3.0 – 24.12)	0.680084
MeanDistD	15.76 ± 4.49 (5.09 – 24.98)	17.43 ± 5.31 (7.88 – 24.95)	0.234737
MeanDiamD	1.29 ± .59 (.42 – 3.34)	1.01 ± .63 (.21 – 1.68)	0.098323
MeanHeightD	11.57 ± 2.74 (6.9 – 17.75)	10.52 ± 4.0 (3 – 17.35)	0.736609
Coniferous	2.8 ± 1.97 (1-9)	6.0 ± 5.5 (1-21)	0.266333
Deciduous	5.07 ± 3.04 (1-14)	6.9 ± 6.56 (1- 32)	0.293214

ParkSize	25267 ± 28587 (227.1 – 110404)	264563 ± 631115 (594 – 2921975)	0.189618
ParkDist	140.1 ± 109.2 (12.18-435)	211.70 ± 148.40 (72.68 – 687.25)	0.281873
YearBuilt	1957 ± 23.84 (1888 – 1991)	1968 ± 21.24 (1915 – 2014)	0.644388
BldgDist	130.8 ± 86.14 (5.08 – 287.5)	104.5 ± 104.0 (1.4 – 403.6)	0.616205
PoleDist	126.3 ± 196.2 (7.02 - 800.38)	104.5 ± 129.4 (11.23 - 476.54)	0.153909
WaterDist	409.80 ± 438.5 (72.92 – 685)	392.90 m ± 317.6 (10.91 – 1119)	0.692580

NTheight = Nest tree height

Diameter = Nest tree diameter

Nheight = Nest height

DistTallestC = Distance to the tallest conifer within 25 m

TallestC = Tallest conifer within 25 m

MeanHeightC = Mean height of conifers within 25 m

MeanDistC = Mean distance to conifers within 25 m

MeanDiamC = Mean diameter of conifers within 25 m

DistTallestD = Distance to tallest diameter within 25 m

TallestD = Tallest deciduous tree within 25 m

MeanDistD = Mean distance to deciduous trees within 25 m

MeanDiamD = Mean diameter of deciduous trees within 25 m

MeanHeightD = Mean height of deciduous trees within 25 m

Coniferous = Number of coniferous trees within 25 m

Deciduous = Number of deciduous trees within 25 m

ParkSize = Park size of the park/green space closest to the nest tree

ParkDist = Distance to the nearest park/greenspace

YearBuilt = The mean age of home/building closest to the nest tree

BldgDist = Distance to the closest building from the nest tree

PoleDist = Distance to the closest utility pole from the nest tree

WaterDist = Distance to the closest water body from the nest tree

APPENDIX B

Banded Merlins between 2018 and 2019. Females are designated with prefix 1004 and males are designated with prefix 1583. HY = known to have hatched in the calendar year ASY = known to have hatched earlier than the calendar year, hatch year unknown. WNV

Band #	Age	Weight (g)	Location	Additional Comments
100416342	HY	223	Nest	
100416343	HY	186	Nest	
100416344	HY	198	Nest	Deceased; WNV; 07/2018
100416345	ASY	194	Rehab	Admitted for starvation; released
100416346	HY	194	Nest	Died from WNV at nest site
100416347	HY	201	Rehab	Admitted for WNV; released
100416349	HY	200	Rehab	Admitted for WNV; released
100416350	HY	207	Rehab	Admitted for WNV; released
100416351	HY	201	Rehab	Deceased; ruptured air sac; 07/2018
100416352	HY	200	Rehab	Admitted for WNV; released
100416353	HY	178	Rehab	Admitted for WNV; released
100416354	HY	234	Rehab	Admitted for WNV; released
100416355	HY	231	Rehab	Treated and released
100416356	HY	153	Rehab	Treated and released
100416358	ASY	200	Rehab	Treated and released
100416359	ASY		Nest	
100416360	ASY	289	Nest	
100416361	HY	219	Rehab	Treated and released
100416362	HY	204	Rehab	Treated and released
100416363	ASY	257	Nest	
100416364	ASY	235	Nest	
100416365	HY	190	Rehab	Treated and released
158382801	HY	144	Nest	
158382802|	HY	171	Nest	
158382803	HY	160	Nest	Died from WNV at nest site

158382804	HY	164	Nest	Died from WNV at nest site
158382805	HY	132	Rehab	Admitted for WNV; released
158382806	ASY	176	Nest	Transmitter 44657
158382807	HY	137	Rehab	Admitted for WNV; released
158382808	HY	134	Rehab	Admitted for WNV; released
158382809	ASY	164	Nest	Transmitter 44655
158382810	ASY	166	Nest	Transmitter 44658
158382811	ASY	166	Nest	Transmitter 44656
158382812	ASY	164	Nest	
158382813	ASY		Nest	Deceased; bird feeder mortality; 10/2020
158382814	ASY	207	Nest	
158382815	HY	167	Nest	
158382816	ASY	152	Nest	
158382817	HY	235	Rehab	Released back to nest site
158382818	HY	151	Rehab	Released back to nest site
158382819	ASY		Nest	
158382820	ASY		Rehab	
158382821	ASY	150	Rehab	
158382825	ASY	169	Nest	