SEASONAL ABUNDANCE AND HABITAT ASSOCIATIONS OF WANDERING CATS AND BIRDS IN A TEMPERATE ZONE BIODIVERSITY HOTSPOT

by

Olivia Wilson

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ABSTRACT

The interaction between birds and wandering domestic cats is an ongoing challenge for both wildlife conservation and cat welfare, particularly in regions where high avian diversity overlaps with dense human development and wandering cats. I examined the abundance, richness and community structure of birds and the abundance of wandering domestic cats (*Felis catus*), in the temperate biodiversity hotspot of the south Okanagan Valley, British Columbia, Canada, between Okanagan Falls and Osoyoos, across an entire annual period. I did this by pairing point counts and photos from trail cameras from 123 locations across five seasonal periods between March 2022 and March 2023, assessing the habitat associations of birds and cats across a variety of land use types, including urban, peri-urban, agricultural, and natural. I conducted a total of 2380 point counts and used hierarchical modelling and unconstrained ordination to examine bird abundance and species richness, and community composition, respectively. My results revealed distinct seasonal patterns of bird abundance and richness with these metrics being the highest during spring migration and the breeding season. Urbanization and human development impacted the distribution of birds year-round, especially in the non-breeding seasons when a large diversity of species used urban areas. Using the same locations as the point counts, but shifting cameras every 28 days, I examined local abundance of wandering cats. I showed that wandering cats were found in high abundances in urban habitats year-round but overall had the highest abundances during the early winter, spring, and summer. Wandering cats were detected at 100% of peri-urban sites, 97% of urban sites, 65% of agricultural sites and 42% of natural sites. I estimated an annual average of 6,557 wandering cats within the study area with up to 82% of them being unowned cats, equating to one cat for every two to three people. Overall, I demonstrate the importance of identifying where birds are and what habitats they are using across the entire year and not only during distinct periods within the annual cycle (e.g. breeding). The high numbers of wandering cats, combined with the diversity of birds and other wildlife, suggests that cats likely have significant impacts on birds and other wildlife year-round in the study region and likely elsewhere. In the south Okanagan Valley, management actions such as outreach initiatives should take a seasonal and habitat-based approach. Outreach should focus on encouraging urban residents to keep their cats indoors during the winter because many species move into urban areas at this time. Resident in peri-urban and agricultural habitats should be encouraged or incentivised to spay and neuter cats on their property and keep cats inside during spring, summer, and fall when high cat numbers overlap with high bird abundance and richness. Given the high abundance and richness of birds along with the highest currently reported abundances of wandering cats per capita, these results stress the urgent need for collaborative efforts among municipalities, stakeholders, and residents to mitigate the ecological impact of wandering cats and help preserve the biodiversity of this unique region.

PREFACE

Funding for this Masters was provided by Mitacs Accelerate Grant in partnership with the Stewardship Centre for British Columbia. Additional funding was supplied by Environment and Climate Change Canada (Wildlife Research Division and Canadian Wildlife Services), the Liber Ero Foundation, and a Graduate Entrance Research Scholarship from UNBC.

Permits for access to conduct research on private and provincial land were granted by Nature Trust of British Columbia, BC Parks, the BC Ministry of Forests, and the Osoyoos Desert Centre. Access to Vineyards, Wineries, and private residential property was granted by the occupying resident. I went door-to-door asking residents for access to their property and provided all of the information upfront, verbally and in a letter, working closely with residents to ensure willing participation. Residents were always informed when I would be accessing their property and were given the option to discontinue their involvement at any time.

Although I use first-person, singular, throughout this thesis in describing the work conducted, I would like to acknowledge that these studies represent, and will be published, as collaborative works, as I could not have conducted this research otherwise. While I was the lead researcher in the design, execution, analysis, and writing of the work presented here, several others contributed significantly to be included in authorship. As such, the anticipated authorship of the two main data chapters of the thesis (Chapters 2 and 3) will be, for Chapter 2, Tanya Luszcz and Elizabeth A. Gow, and for Chapter 3, Tanya Luszcz, DG Blair, Anna Skurikhina, and Elizabeth A. Gow.

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I am incredibly grateful to everyone with the Stewardship Centre for British Columbia for playing a pivotal role in launching this project and providing funding. Thank you, DG Blair, for your vision and for making this research possible! A special thanks to Anna Skurikhina for your behind-the-scenes work, your passion in outreach, and for being a friend when I was new to the area. Additionally, I would like to give a huge thank you to Tanya Luszcz with the Canadian Wildlife Services for funding my fifth season, which was crucial in completing a year-round study. Your guidance and support throughout the project have been invaluable. I also extend my appreciation to everyone who helped with fieldwork, with a special shout-out to David Bell who did an awesome job carrying on the project during the winter. Finally, I would also like to thank the Mitacs Accelerate Program for their financial support of this project.

I am also deeply thankful for the organizations and government agencies that facilitated the permits for conducting research in protected areas. Thank you to The Nature Trust of BC, especially the Okanagan Field Crew, The Osoyoos Desert Centre, BC Parks, and the BC Ministry of Forests. My sincere thanks also go to vineyards that hosted trail cameras: Arterra Wines Canada, Montakarn Estate Winery, Hidden Chapel Winery, River Stone Estate Winery, Red Horses Vineyard, and Burrowing Owl Estate Winery. Additionally, I am immensely grateful to the many residents who allowed us to place trail cameras on their properties – your generosity made this research possible, and it was a pleasure getting to know so many of you!

Finally, I would like to express my deepest thanks to my friends and family. To my lab mates, thank you for your constant support and encouragement. To my parents and sister, I appreciate you guys so much and I am so grateful to have you as my family. Your love, support, and encouragement have been a source of strength. To my partner, Andrew, thank you for your patience, understanding, and for helping me survive the stressful times. I couldn't have done this without all of you.

Chapter 1: General Introduction

1.1 Seasonality

Most organisms, including animals, closely follow the annual cycle of environmental fluctuations or changes, known as seasonality (Lisovski et al., 2021; Williams et al., 2017). These environmental changes may include day length, precipitation, and temperature, which ultimately influences resource availability (Lisovski et al., 2021). For animals, these seasonal changes promote changes in hormones that can help individuals prepare for, cope with, and adapt to seasonal changes (Bronson, 2009; Chen et al., 2020). Examples of some adaptations driven by hormone changes are mating or breeding, migration, and moulting (Chen et al., 2020). Additionally, as resource availability changes, animals will move to where resources are abundant, which can drive their seasonal behaviour and movements (Huston & Wolverton, 2009). These seasonal movements mean that the abundance and composition of species or individuals within a given area change throughout the year (Briese & Smith, 1974; Cameron et al., 2018; Scully et al., 2018). Seasonal changes in abundance and habitat selection have been studied in a range of animals, such as small mammals (e.g., shrews, Briese & Smith, 1974), larger mammals (e.g., seals, Cameron et al., 2018), domestic cats (Cove et al., 2023; Horn et al., 2011), and birds (Blair, 1996; Ortega-Álvarez & MacGregor-Fors, 2009; Tu et al., 2020). Every year, over 2.6 billion birds fly from the Southern USA, and Central and South America, to Canada during the spring and summer migration to follow seasonal changes in resource abundance (Dokter et al., 2018; Huston & Wolverton, 2009). Birds that can synchronize their movements with environmental patterns in their surroundings, such as those related to food and weather, tend to experience greater success in terms of survival and reproduction (Both et al., 2006). Both long- and short-distance migrants aim to arrive when resources are at their peak (Huston & Wolverton, 2009), or they could face a mismatch in resource abundance versus resource needs (Jones & Cresswell, 2010). Birds face many dangers during migration such as bad weather, collisions with glass or vehicles, pathogen transmission, and higher risk of predation (Jourdain et al., 2007; Lao et al., 2023; Lind & Cresswell, 2006; Van Den Broeke & Gunkel, 2021). Mortality during migration may be as much as fifteen times higher than during stationary periods for many species, such as the black-throated blue warbler (*Dendrioica caerulescens*, Lind & Cresswell, 2006).

One predator that may influence mortality risk of birds differently throughout the year are domestic cats. Globally, predation on birds by cats is well documented (Baker et al., 2005, 2008; Beckerman et al., 2007). Predation rates are typically highest in the spring and summer months when there are a high number of nestlings and fledglings that have less experience evading predators (Baker et al., 2005). But predation risk may be proportionately higher to individual birds in the late fall or winter if cat abundances do not change, but bird abundances are lower (Blancher, 2013). Cat presence can also indirectly affect nest productivity through a decrease in nest provisioning rates, in addition to predation rates by predators (Bonnington et al., 2013).

1.2 Domestic cats

Domestic cats (*Felis catus*) descended from a wild ancestor, the African wildcat (*Felis silvestris* lybica; Driscoll et al., 2007), a well-adapted and widespread predator, and have retrained many of the physiological and behavioural traits that make them effective hunters (Bradshaw, 2006). Cats have a short history of domestication and were domesticated to serve as pest controllers only ~10,000 years ago and have more recently been termed “wild companions” due to their dual roles in society as companion animals and pest controllers

(Crowley et al., 2020). Despite domestication, cats have retained their ancestral hunting behaviours, enabling them to adapt to a variety of environments into which they have been introduced (Bradshaw, 2006; Canadian Federation of Humane Societies [CFHS], 2017). These innate hunting abilities allow cats to capture a variety of wild animals, posing serious global threats to small vertebrate populations (Loss et al., 2013; Trouwborst et al., 2020).

Domestic cats are a common companion animal in Canada and the USA (CFHS, 2017; The Humane Society of the United States [HSUS], 2019), but unlike other companion animals, their life-styles vary depending on their access to the outdoors. The classification of domestic cats varies across studies, but in this thesis I will follow the definitions outlined by (Crowley et al., 2020). Owned cats that are fully confined indoors are considered to have no impact on birds or other wildlife, whereas those with indoor-outdoor access can wander unsupervised and have minimal ecological impact. In contrast, unowned cats pose a greater threat to wildlife, as they may rely on hunting for survival (van Heezik et al., 2010). Feral cats fall within the category of unowned cats and are often self-sustaining, surviving independently of human involvement. For this study, the term “wandering cats” refers to any cat that is outdoors and unsupervised or unconstrained, encompassing both owned indoor-outdoor and unowned cats. This term was found to be the most accepted by multiple groups of people and stakeholders based on focus groups run by Nature Canada’s Cats and Birds Campaign (Gow et al., 2024). Outdoor access provides cats with mental stimulation, physical activity, and opportunities to engage in natural behaviours, but it also exposes them to welfare risks (Rochlitz, 2005), highlighting the need for effective management strategies. When cats are allowed to wander unconstrained outside, they may experience increased risks of injury and illness through predation by other animals, vehicle collisions, and disease transmission of illnesses such as toxoplasmosis (reviewed in Tan et al., 2020). For these reasons, it is often recommended that cats are kept indoors, but many owners still allow their cats outside.

Various groups (e.g., veterinarians, shelters, humane societies, conservation organizations, policy makers, and members of the public) are concerned about issues surrounding wandering cats (detailed in Saunders et al., 2021; Wald & Peterson, 2020) and find common ground in the desire to curtail the number of wandering cats. Unowned cats may be fed by people, but their movements are not constrained. Unlike many other companion or domesticated animals, cats can persist, survive, and reproduce without any human intervention, and thus feral and unmanaged outdoor populations of cats are common (Legge et al., 2017; Tan et al., 2020). Quantifying the number of wandering domestic cats allows us to understand anthropogenic threats facing birds and other species of concern (Ferreira et al., 2011; Flockhart et al., 2016). It has also been a metric that is frequently requested by municipal government officials, and from non-profit organizations to identify where to direct limited resources to programs such as educational campaigns. Using this information, I can also create a baseline for wildlife and cat populations that helps to inform future management decisions and to assess the effectiveness of implemented management strategies. This thesis arose by a need and desire from key interested parties (i.e., the Stewardship Centre for British Columbia and Environment and Climate Change Canada) for a detailed study of the abundance of birds and cats in the southern Okanagan Valley (see below) to help direct educational campaigns and future funding and initiatives for other cat management actions in the region.

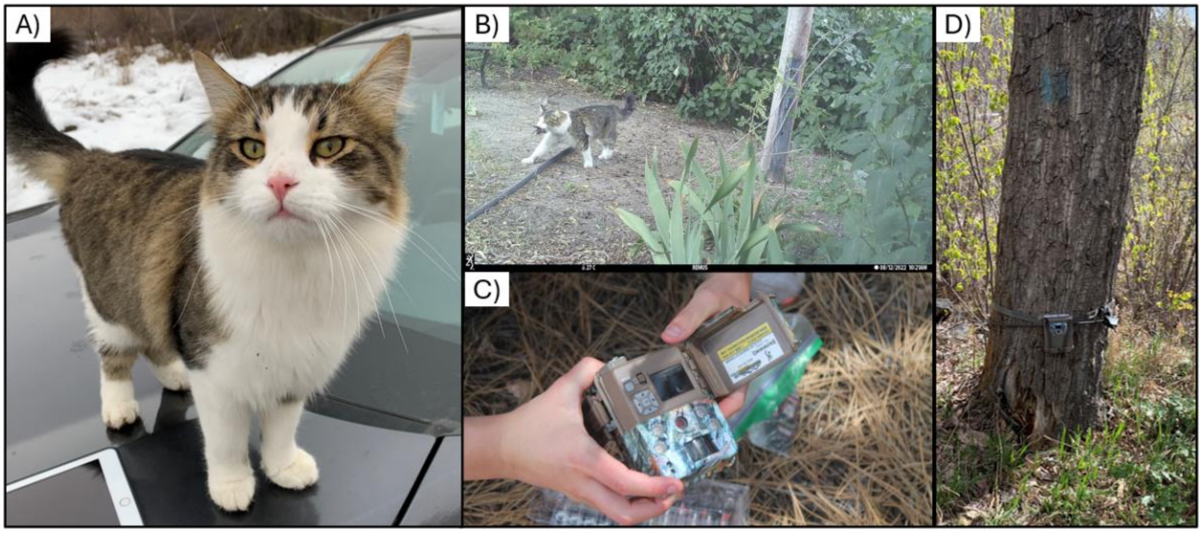

Figure 1.1. A) Wandering cat in the south Okanagan Valley. B) The same wandering cat seen on trail camera image with a bird in its mouth. C) Trail cameras used in this study to capture images of wandering cats. D) Trial camera strapped to a tree and secured with a lock.

Some methods for estimating local cat abundance are walking line transects and resident surveys, but they are sensitive to bias because they can limit the number of cats detected (Flockhart et al., 2016; Hand, 2019). In contrast, the use of trail cameras (Elizondo & Loss, 2016) can reduce bias seen in other methods and are increasingly being used as a common tool to assess population sizes and abundance of various animals around the world (Jhala et al., 2011; Sollmann et al., 2011; Tanwar et al., 2021), including wandering cats (Clyde et al., 2022; Coe et al., 2021; Cove et al., 2023; Gow et al., 2024). Trail cameras offer several advantages over other methods as they can detect both owned and unowned cats on private property (e.g., in people's yards) and during their peak activity periods (i.e., at night), and are particularly effective in rural areas. Recently, trail cameras were used to: assess the effectiveness of a trap-neuter-release management program for feral cats (Coe et al., 2021); estimate local abundance and habitat associations of wandering cats (Clyde et al., 2022; Gow et al., 2024); and, estimate occupancy and density of wandering cats in Washington, D.C

(Cove et al., 2023). For these reasons, I used trail cameras in this study to estimate the abundance of wandering cats in the south Okanagan Valley study (Figure 1.1).

1.3 Study Region

Located in the south-central region of British Columbia, the south Okanagan Valley (between Okanagan Falls and Osoyoos) is one of the most endangered ecosystems in Canada and is part of the bunchgrass biogeoclimatic zone that is characterized by grassland, shrub-steppe, wetlands, and dry open forests consisting of ponderosa pines, cottonwoods, and Douglas firs (Alldritt-McDowell et al., 1998). Its unique topography features a valley flanked by large mountains, making the south Okanagan Valley part of an important avian migratory route. These distinct habitats and topography also make it a significant area, supporting some of the highest diversity of birds in Canada, with over 330 recorded bird species, including 24 provincially listed species at risk, 22 federally listed species at risk under the Species at Risk Act (eBird, 2022; Okanagan Similkameen Stewardship [OSS], 2023). Many birds funnel through the valley during migration, including long-distance, short-distance, and altitudinal migrants. The temperate climate also ensures an abundance of food and other resources year-round, supporting many resident bird species that inhabit the south Okanagan Valley throughout the entire year. Given its ecological significance and vulnerability, there is an ecological need for research in this area to understand the challenges faced by bird populations and where birds may be at greatest risk of domestic cats to help target management actions.

I assessed four land class types during this study: urban, peri-urban, agricultural, and natural (Figure 1.2). While overlap exists between these four land classes, the following definitions outline the main attributes associated with each and classify them based on the amount of anthropogenic disturbance. Urban habitats have the highest level of anthropogenic disturbance and include a matrix of landscapes, such as pavement, buildings (residential and commercial), green spaces, and habitat patches (e.g., transportation corridors, riverbanks, and parks; Swanwick et al., 2003). Peri-urban habitats consisted of clusters of houses surrounded by agricultural (e.g., orchards, vineyards, and crops) and/or natural (e.g., desert, forest, riparian) habitat with houses often spaced apart (approximately 25-50 meters). Agricultural land was used for agriculture (such as vineyards and orchards) or other farming practices, targeting smaller vineyards/orchards that were <50 hectares. Natural habitats had the lowest level of anthropogenic disturbance and included any protected habitats conserved provincially, federally, or privately.

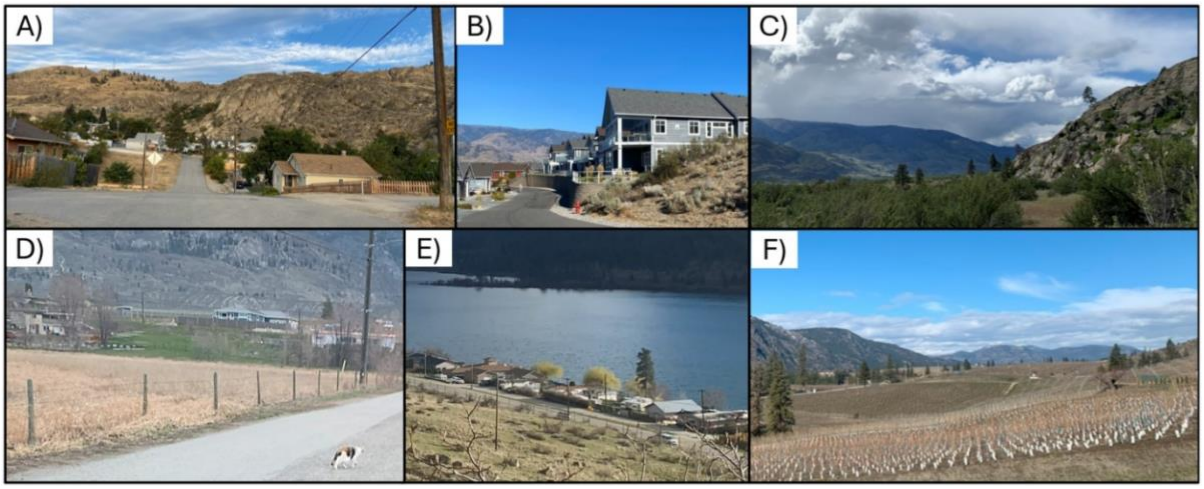

Figure 1.2. Representation of habitat types evaluated during this study, including urban (A and B; e.g., residential areas of Osoyoos and Oliver), natural (C; e.g., Haynes' Lease Ecological Reserve, Osoyoos), peri-urban (D and E; e.g., low-density residential areas around Okanagan Falls and Oliver), and agricultural (F; e.g., Montakarn Estate Winery).

1.4 Birds and Habitat

The abundance and diversity of birds in the south Okanagan Valley are shaped by their ability to adapt to different levels of urbanization and resource availability across habitats. Urbanization plays a large role in the abundance and richness of bird species, as urban areas provide more human-provided resources such as food, water, perches, and nesting places (Emlen, 1974; Mills et al., 1989). Some bird species take advantage of these resources provided in urban environments and can be labeled as “urban exploiters” (Blair, 1996). Across bird species, few are urban exploiters, which results in an overall reduction in the diversity of species in urban areas (Blair, 1996; Isaksson, 2018). Those who can exploit the urban environment often thrive, leading to high abundances of these species (Blair, 1996). Other species are “suburban adaptable” and can take advantage of human-provided resources that come with urbanization, while exploiting the additional natural resources provided in peri-urban areas. Lastly, many species are sensitive to urbanization and these “urban avoiders” are at their highest densities in natural areas.

Areas with lower levels of urbanization, specifically peri-urban and natural areas, tend to increase diversity and abundance of resources available to birds, and therefore support a wider range of niches (Blair, 1996). Urban areas lack resource diversity, thereby supporting few urban exploiters (Blair, 1996; Emlen, 1974). Other areas with lower levels of urbanization include peri-urban, agricultural, and natural areas, each being able to support a range of bird species. Agricultural areas in the south Okanagan Valley typically provide limited resources to birds due to simplification of vegetation (e.g., monoculture) and reduced insect populations due to pesticide use (Bain et al., 2020). This limits the number of birds that can use this habitat type throughout the year, but there are many species that can take advantage of the abundance of fresh fruit during the spring, summer, and fall. Natural and peri-urban areas have higher resource availability and diversification, with the highest abundance of resources available during the spring and summer (Blair, 1996; Jokimäki, 1999). Because of this, species richness and diversity are often highest in natural and peri-urban areas (because they often border natural areas), and lowest in urban habitats (Beissinger & Osborne, 1982; Emlen, 1974; Lancaster & Rees, 1979).

Abundance and diversity of birds in different habitats will vary between spring and winter due to changes in resource availability. During the spring and summer, resources are highest in peri-urban and natural areas leading to higher densities of species. While urban areas still provide abundant food resources, the quality of this food, compared to that which is available to birds naturally, enhances only a few feeding niches (Lancaster & Rees, 1979). When winter arrives, the increasing snow will reduce vegetation and prey availability, which will impact species distribution as snow levels change in the mountains and in the valley (Johnson & Sherry, 2001; Leech & Crick, 2007). In the winter, most species in the Okanagan Valley are residents, along with a few altitudinal migratory species. Additionally, during the winter, urban and peri-urban environments with higher human population densities provide food for birds, increasing abundance throughout the winter (Emlen, 1974; Lancaster & Rees, 1979). Many species will take advantage of the food and shelter provided by humans during the winter when options are limited.

1.5 Thesis Overview

Globally, predation on birds by cats is well documented (Baker et al., 2005, 2008; Blancher, 2013; Krauze-Gryz et al., 2019; Li et al., 2021; Sedano-Cruz, 2022; van Heezik et al., 2010; Woinarski et al., 2017). By quantifying the local abundance and numbers of wandering domestic cats, we can assess the potential risk that birds face from cats and identify where and when these threats may be highest. This study, developed in partnership with the Stewardship Centre for British Columbia and the Canadian Wildlife Service (Environment and Climate Change Canada), provides science-based information to guide targeted education campaigns, other potential cat management options, and bird-focused conservation and management, that overall aim to benefit people, birds, and cats. By examining both birds (Chapter 2) and wandering cats (Chapter 3) in this thesis, I am able to identify where there is overlap in local abundances of wandering cats, and high bird abundance and richness, and how these patterns vary across the habitats and seasons. While more study and analysis beyond the scope of this thesis would be needed to understand the true risks and impacts that birds face from cats in this region, I hope this thesis creates an initial, and valuable, starting point to identify potential predation pressure and risk to birds within the study region that can be applied more broadly. For example, if peri-urban areas have a high cat abundance and high bird abundance and species richness, this may indicate that focusing cat management or education in these habitats may provide the greatest benefit compared to urban areas, which I expect to have high relative cat abundance, but low relative bird abundance and species richness. Thus, through this type of approach that highlights areas and seasons of overlap between birds and cats, this thesis will contribute to helping to better inform conservation and management efforts that can then be used by a wide diversity of interested parties and identify where to direct limited resources.

I examined the year-round trends in abundance, species richness and community structure of birds and the abundance of wandering domestic cats in the south Okanagan valley, British Columbia, Canada, to understand where wandering domestic cats pose the greatest risks to birds, which has never been looked at in this type of landscape. My research was conducted between Okanagan Falls and Osoyoos, across a variety of habitats, including urban, peri-urban, agricultural, and natural areas across five seasonal periods relevant to birds (spring migration, breeding, fall migration, early non-breeding and late non-breeding). This study was developed in partnership with the Stewardship Centre for British Columbia and

Environmental and Climate Change Canada (Wildlife Research Division and Canadian Wildlife Services) to test hypotheses that will provide science-based information for targeted educational campaigns. Chapter 2 will focus on birds, with the overall objective of identifying how bird species richness and abundance varies across seasons and determine patterns of habitat associations among birds throughout the entire year. Chapter 3 will focus on wandering domestic cats, with the overall objective of identifying how cat abundance and population size varies seasonally and across habitats. Chapter 4 discusses the connections between birds (Chapter 2) and cats (Chapter 3).

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Chapter 2: Year-round Differences in Bird Abundance, Richness, and Community Composition Across an Urban to Rural Gradient

2.1 Introduction

Environmental change, including anthropogenic disturbance, creates altered environments that introduce new ecosystems and niches, thereby altering the spatial distribution of species (Adhikari & Hansen, 2018; Gaston et al., 2003; Semenchuk et al., 2022; Stein et al., 2014). Understanding how species respond to these alterations is crucial and can be achieved by examining species abundance, richness, and community composition across various land uses and seasons. Birds are suitable for studying the effects of land use and seasonality on species distribution patterns and composition because of their mobility, detectability, and diverse resource needs. Urbanization is a major form of land conversion that significantly alters ecological processes and resource availability for birds, within and around urban areas (Grimm et al., 2008; Isaksson, 2018; Rigal et al., 2023). Urban areas are designed to accommodate human needs, leading to similar habitats among cities and within cities, while still supporting diverse habitats within them through the diverse components and structure of the urban matrix (Swanwick et al., 2003). These urban environments can provide abundant resources such as food, water, perches, and nesting places to many species, favouring birds with flexible and broad niche requirements (Emlen, 1974; Isaksson, 2018; Mills et al., 1989). This uniformity in urban landscapes plays a significant role in influencing the patterns of bird communities and structure both within urban areas and in areas that have urban-like qualities (e.g. in peri-urban areas or urban boundaries; Callaghan et al., 2023; Emlen, 1974; Isaksson, 2018; Mills et al., 1989; Stagoll et al., 2010; Tu et al., 2020).

Birds can be classified based on their ability to inhabit a range of habitats, which can be categorized by varying degrees of urbanization. Certain bird species, known as “urban exploiters”, thrive in urbanized environments due to their ability to capitalize on the available resources (Blair, 1996). In contrast, “suburban adaptable” species, which can use both human-provided and natural resources, are commonly found in peri-urban areas, while “urban avoiders”, which are often native species, prefer natural habitats where the influences of humans are minimal (Blair, 1996; Mckinney, 2006; Mills et al., 1989). Areas with less homogenization and anthropogenic disturbance, such as peri-urban and natural habitats, offer greater resource diversity, supporting a wider range of ecological niches and higher species richness (Blair, 1996; Ortega-Álvarez & MacGregor-Fors, 2009; Stagoll et al., 2010). Conversely, homogenization among urban habitats is expected to result in low species richness but a high abundance of adaptable species that exploit available resources and niches, resulting in environments dominated by species that prefer to live near humans (i.e., synanthropic species; Urban Homogenization Hypothesis; Blair, 1996; Emlen, 1974; Mills et al., 1989; Morelli et al., 2016; Ortega-Álvarez & MacGregor-Fors, 2009; Tu et al., 2020).

Urbanization and land conversion significantly influence bird communities, but our understanding of how these patterns change seasonally is limited due to a lack of full-annual cycle studies. Seasonal or migratory movements are critical for birds to align their resource needs with fluctuations in environmental productivity throughout the year (Fudickar et al., 2021; Huston & Wolverton, 2009). However, most studies focus on spring migration and breeding season, when abundance and richness are expected to be highest as birds move into habitats with peak plant and insect productivity (Bauer & Hoye, 2014; Huston & Wolverton, 2009; Mac Arthur, 1959; Somveille et al., 2015). This emphasis on the breeding season limits our understanding of bird habitat use during other seasonal periods. Studies examining seasonal bird movements have relied on citizen science (e.g., eBird) and complex modelling to predict migratory movements and factors that drive these movements (Fuentes et al., 2023;

Somveille et al., 2021) or have focused on individual species (Deshwal et al., 2021; Rueda-Uribe et al., 2024), rather than assessing community- or habitat-level differences. Therefore, year-round studies assessing birds abundance, richness, and community composition across the entire annual cycle and across habitats with varying degrees of urbanization are important to understand how anthropogenic disturbance impacts birds year-round (Marra et al., 2015).

Anthropogenic effects (such as land conversion) may affect resources, vegetative growth, and microclimates in human dominated landscapes (Clement et al., 2019; Rigal et al., 2023), yet their seasonal impacts on bird communities remains poorly understood. In North America, at least 25% of avian species are considered to be synanthropic (Johnston, 2001), yet, during the migration or non-breeding season, birds not typically considered synanthropic may rely on urban areas for extended periods (Morales et al., 2022; Poirier et al., 2024). During the non-breeding months (winter in the Northern Hemisphere), lower temperatures and increased snowfall reduce natural vegetation and prey availability, which may influence bird community composition and distribution, and patterns of abundance and species richness (Johnson & Sherry, 2001; Leech & Crick, 2007). However, urban environments can offer milder temperatures that may reduce metabolic costs for resident species or facultative migrants in the northern temperate zone winter/non-breeding season (Watson et al., 2024). As a result, urban areas may produce favourable conditions during certain periods of the year that may cause some species to avoid migration altogether (Bonnet-Lebrun et al., 2020), or short or altitudinal migrants may stay in habitats for longer periods to take advantage of the available resources (Atwell et al., 2011). Habitats with higher anthropogenic disturbance, such as urban or peri-urban habitats, may provide more resources through human provisioning, while areas that have less anthropogenic disturbance, such as agricultural and natural areas, may be less productive (Emlen, 1974; Lancaster & Rees, 1979).

Here, I assess how bird abundance, species richness, and community composition vary across habitats varying in degrees of anthropogenic disturbance and an entire annual cycle in the south Okanagan Valley, British Columbia, which is one of the most biodiverse regions in Canada. Using point counts conducted across five seasonal periods relevant for birds and habitats ranging from urban, peri-urban, agricultural, and natural habitats, I had three objectives. I aimed to: 1) identify how bird species richness and abundance varies across seasons; 2) test the Winter Urban Association, and Urban Homogenization Hypotheses (Table 2.1); and 3) determine patterns of habitat associations among birds throughout the entire year.

Table 2.1. Hypotheses and associated predictor variables used to explain local bird relative abundance and species richness in the south Okanagan Valley, B.C. Source(s) indicate a study or studies that provide evidence to support the hypothesis.

Hypothesis	**Hypothesized mechanism(s)**	**Predictor variable**	**Source(s)**
Urban Homogenization Hypothesis	Bird communities in urban areas are very similar because urban areas are all built the same to accommodate human needs. Therefore, the species that can exploit these areas thrive in high abundances, but this leads to low species richness, as these habitats are often dominated by urban exploiter species.	Landclass*season	(Blair, 1996; Emlen, 1974; Isaksson, 2018; Mckinney, 2006; Morelli et al., 2016; Ortega-Álvarez & MacGregor-Fors, 2009; Tu et al., 2020)
Winter Urban Association Hypothesis	During the non-breeding seasons, a lack of resources in natural and agricultural areas will draw birds into urban areas, as humans provide resources for birds. This will increase abundance and species richness in urban and peri-urban areas during the non-breeding seasons.	Landclass*season	Emlen, 1974; Jokimäki et al., 1996; Lancaster & Rees, 1979; Tilghman, 1987; Yaukey, 1996

2.2 Methods

2.2.1 Study Area

I studied seasonal patterns of bird abundance, richness, and habitat associations in the south Okanagan Valley (within and surrounding the communities of Okanagan Falls, Oliver, and Osoyoos) of British Columbia, Canada, which is the traditional, unceded, and ancestral territory of the Syilx Okanagan Nation (Figure 2.1). Located in the south-central region of British Columbia, the south Okanagan Valley is part of the northern extension of the Great Basin Bird Conservation Region and is one of the most endangered ecosystems in Canada. It is part of the bunchgrass biogeoclimatic zone, characterized by grassland, shrub-steppe, wetlands, and dry open forests consisting of ponderosa pines, cottonwoods, and Douglas firs (Alldritt-McDowell et al., 1998). The south Okanagan valley supports some of the highest diversity of birds in Canada, with over 330 recorded bird species, including 24 provincially listed species at risk and 22 federally listed species at risk under the Species at Risk Act (eBird, 2022; Okanagan Similkameen Stewardship [OSS], 2023). Many birds funnel through the valley during migration, including long-distance, short-distance, and altitudinal migrants. This area consists of a mosaic of natural habitats (grasslands, deserts, riparian areas, and forests), agriculture (e.g. vineyards and orchards), small urban centers, and peri-urban and exurban areas. The diverse bird species, seasonal variation, and habitat variation of the region emphasize the value of this region as a study area to test hypotheses about variation in species abundance, richness, and community composition across the entire year.

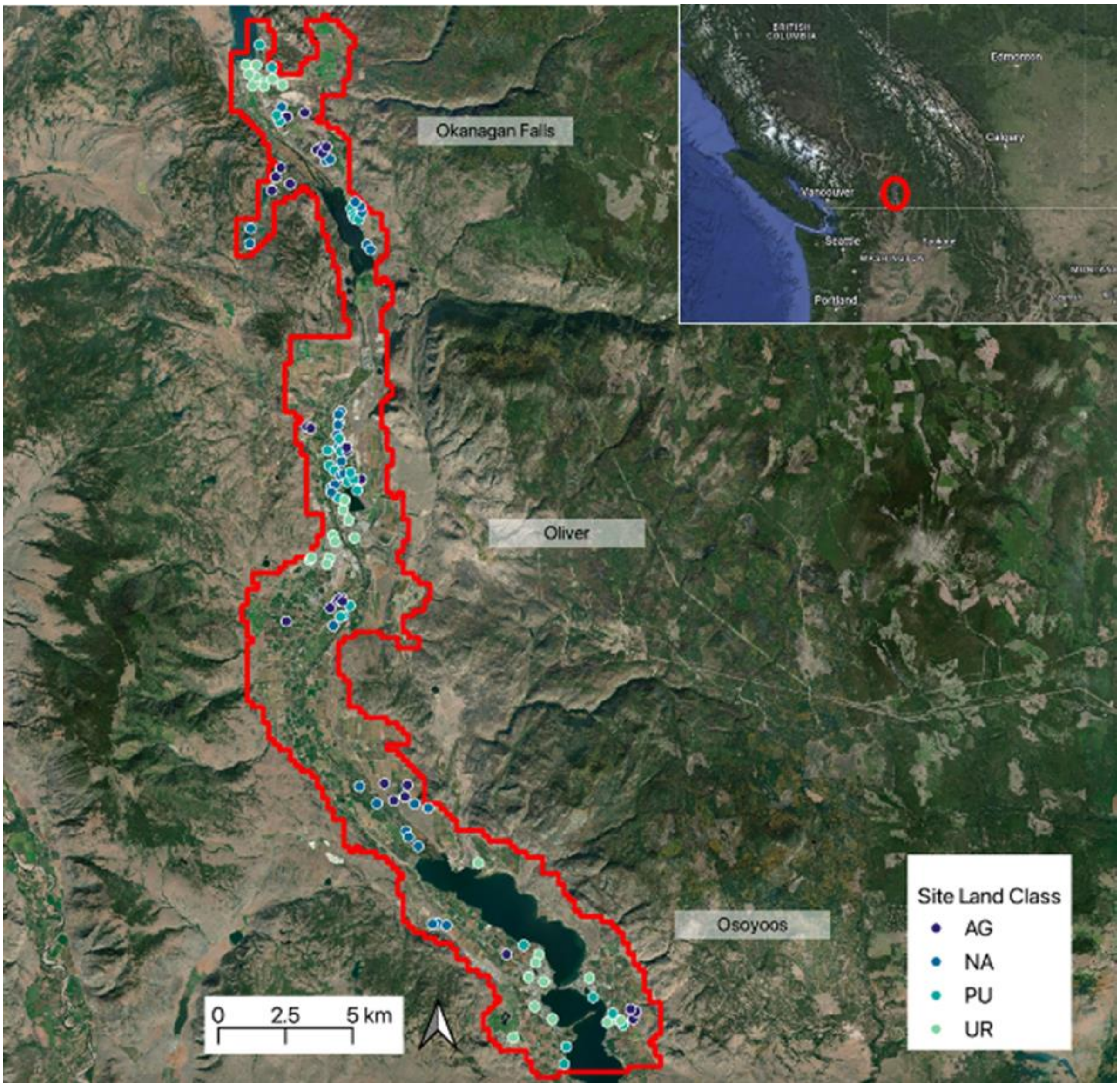

Figure 2.1. Map of the study area in the south Okanagan Valley showing trail camera locations in Okanagan Falls, Oliver, and Osoyoos. Location colours are classified by the following land classes: Agriculture (AG), Natural (NA), Peri-urban (PU), and Urban (UR). The red boarder represents the study area that was used to study wandering domestic cats and the study area boundaries (Chapter 3). The inset map in the top right corner shows a red circle around the south Okanagan Valley, located in the south-central area of British Columbia, Canada. Map is in EPSG:3005 coordinate system (NAD83, BC Albers projection).

2.2.2 Study Design

I assessed seasonal patterns of bird abundance, species richness, and habitat associations using a repeated-measures design stratified by season, conducting point count surveys from March 2022 to March 2023 across five distinct seasonal periods relevant to bird ecology and life history in this region. The five seasonal periods were spring migration (March 13 – May 18, 2022), breeding season (May 29 – July 29, 2022), fall migration (August 10 – October 12, 2022), early non-breeding season (October 20 – December 15, 2022), and late non-breeding season (January 13 – March 23, 2023). The non-breeding season was divided into early and late seasons because the resources available to birds are different during each of these two time periods. For logistical reasons, some sites were added, discarded, or moved throughout the year, leading to a total of 123 point count locations (see Appendix 1 for details). The number of locations with completed point counts were as follows: 120 locations for spring migration, breeding season, and fall migration; 119 locations for early non-breeding season; and 114 locations for late non-breeding season.

I conducted point counts to examine bird communities and species abundance and richness across different land use types. This research was part of a larger study examining the abundance of domestic cats (Chapter 3) across seasons and land use types and therefore, the point count locations were aligned with trail camera sites. This involved using a random stratified design with point count locations at least 100m apart stratified by four land use categories: urban, peri-urban, agricultural, and natural habitats (~30 point counts per land use type; Figure 2.1). While the trail cameras involved in the broader study were on private property (e.g. yards) or in natural habitats, to minimize disturbance to landowners, all point counts occurred either along roadways or on paths within natural areas or agricultural fields (see Appendix 2 for details of land permissions and site establishments).

I used satellite maps of the south Okanagan to categorize areas as urban, peri-urban, agricultural, and natural before deciding which permits I would need and which neighbourhoods to target for door-to-door recruitment. Definitions for each land use type often vary (Sahana et al., 2023; Wolff et al., 2021) and are context dependent and so I combined established definitions from the literature, with ground truthing, to ensure these definitions, and thus categories, were appropriate within the context of my study area. Therefore, I categorized urban areas as having extensive anthropogenic development that included a matrix of landscapes such as pavement, buildings (residential and commercial), and habitat patches (e.g., transportation corridors, riverbanks, and parks; Swanwick et al., 2003). Peri-urban areas consisted of clusters of houses surrounded by agricultural (e.g., orchards, vineyards, and crops) and/or natural (e.g., desert, forest, riparian) land with approximately 25-50 m between houses. Agricultural land was used for agriculture (such as vineyards and orchards) or other farming practices, targeting smaller vineyards/orchards that were <50 hectares. This included targeting houses residing next to agricultural land or buildings/barns within agricultural areas. Natural habitats included any protected areas conserved provincially, federally, or privately. Since these descriptors are continuous, I later classified sites using QGIS to create discrete variables and refined categories based on surrounding habitat variables for statistical analysis (see 2.2.3 GIS Analysis for more details). When choosing sites, true randomization was impossible because locations were mainly on private property and therefore dependent on the willingness of landowners and land managers (except natural sites). This resulted in some sites (two sets of sites; n = 4) being closer than 100 m apart (70 m and 40 m).

I incorporated a repeated measures component into each point-count session by conducting two back-to-back 5-minute point counts at each location. For clarity throughout, I will refer to a point count as each 5-minute survey, while a point-count session consists of two consecutive point counts at a single site. During each seasonal period, I visited each site twice, resulting in a total of 20 point counts per location (2 point counts per session x 2 visits x 5 seasons = 20 point counts per location), unless if a site was added or discarded, resulting in less than 5 seasons completed for a location (see Appendix 1 for more information). During point counts, observers watched, listened, and recorded all birds that were seen or heard, noting their distance from the surveyor. Distances were categorized into bins of 0–50 m and 50–100 m to align with the requirements of generalized distance sampling models (see below) we used to estimate bird abundance and richness. In some cases, the 100 m radius overlapped between point count sites, which meant that some point counts were not statistically independent, and it was possible, yet unlikely, that some birds were counted more than once. These methods have been used in other habitat-based point counts (Huff et al., 2000) and the Ontario Breeding Bird Atlas (Ontario Breeding Bird Atlas, 2021). While double counting can be a concern when estimating population sizes, for examining patterns of abundance across habitats and seasons, statistical independence is not required (Martijn et al., 2023). Thus, we did not remove point count locations that might have overlapped (i.e., <200m apart). At the start of each point-count session, the observer recorded the date, time, cloud cover, wind, and precipitation (defined in Table 2.2). Point count routes, consisting of 10–15 point count locations, were completed by highly skilled birders (3 different birders total). These routes began with morning civil twilight and were completed within 5 hours to maximize detection rates during peak bird activity periods. The order of point count routes was changed each time to account for temporal detection bias and activity variance between species.

2.2.3 GIS Analysis

I used QGIS (Version 3.32 Lima; QGIS.org, 2024) to obtain habitat variables. I assessed habitat variables that represent buildings, roads, water, vineyards, orchards, deserts, fields, forest, and altitude. I accessed detailed OpenStreetMap layers for roads, lakes, rivers, and buildings and imported them into QGIS. To obtain missing habitat information, I created polygons around forests, lakes, rivers, deserts, fields, orchards, and vineyards, and updated the buildings using the point creation function in QGIS. I used a 2023 Esri World Imagery Wayback satellite image as a base layer to trace the polygons (Esri, 2023). I created 200m buffers around each site and extracted the habitat variable information using the intersect tool. Within each buffer, I categorized habitat variables into roads, buildings, desert, field, forest, orchard, vineyard, and water. I then quantified the total length of roads, the total number of buildings, and the proportion of desert, field, forest, orchard, vineyard, and water within each 200 m buffer (defined in Table 2.2). I conducted these analyses in QGIS using the EPSG:3005 coordinate system (NAD83, BC Albers projection). The habitat variables extracted from the 200 m buffers were used to classify each point count location as either urban, peri-urban, natural, or agricultural (defined in Table 2.2).

Table 2.2. Description of detection and site covariates used to estimate relative abundance and species richness of birds in the south Okanagan Valley, B.C. Detection covariates influenced the detection of species during point counts and site covariates describe the site where point counts occurred.

Variables	Description
Detection covariates	
date	Julian date when the point count was conducted.
time	Time at the start of the point count observational period.
wind	Strength of wind during point count using the Beaufort scale (Scale 0-4)
cloud	Cloud cover % overhead during the point count.
precip	Binary observation during point count, where 0 = no precipitation and 1 = precipitation.
Site covariates	
desert	Percentage of desert within a 200 m buffer around the point count location.
field	Percentage of field within a 200 m buffer around the point count location. Fields include public spaces such as parks and school yards, land that is used as pasture or farming, and empty, overgrown lots.
orchard	Percentage of orchard within a 200 m buffer around the point count location.
vineyard	Percentage of vineyard within a 200 m buffer around the point count location.
forest	Percentage of forest within a 200 m buffer around the point count location.
water	Percentage of water within a 200 m buffer around the point count location.
roads	Privateroads and publicroads combined, to equal the total length of roads (m) within a 200 m buffer around the point count location.
privateroad	Total length of private roads, in meters, within a 200 m buffer around the point count location. Private roads include driveways (>40 m in length), vineyard/orchard roads, and other impervious surfaces that are not accessible to the public.
publicroad	Total length of public roads, in meters, within a 200 m buffer around the point count location. Public roads are maintained by the city and are easily accessible to the public.
buildings	All buildings within a 200 m buffer around the point count locations. This includes high-density housing and all other buildings.
landclass	The land class type of the point count locations: urban, peri-urban, natural, or agricultural
urban	Site buffer (200 m) containing >55 buildings
peri-urban	Site buffer (200 m) with 17 to 50 buildings and <49% vineyard and/or orchard.
agricultural	Site buffer (200 m) characterized by >28% agriculture and <20 buildings.
natural	Site buffer (200 m) characterized by >25% natural habitat, <16 buildings, and <29% agriculture.
altitude	Altitude (m) at the point count location.

2.2.4 Unmarked Analysis Design

I used distance sampling techniques to analyze data for unmarked animals (Fiske & Chandler, 2011). Distance sampling involves measuring the distance from a point to an observed organism and can be used to estimate abundance (Buckland et al., 2001). Distance can be recorded by measuring the exact distance to the observed organisms from the observer, however, since birds are often detected by their vocalizations, it can be difficult to detect their exact location. As a result, distances are frequently recorded by grouping observations into discrete distance intervals (Royle et al., 2004). I estimated bird abundance using generalized distance sampling models using 'gdistsamp' within the unmarked package (Fiske & Chandler, 2011) in Program R, V2.1.1. (R Core Team, 2021). The 'unmarked' models incorporate distance sampling and detection probability to estimate abundance, while accounting for species that may go undetected during sampling (Fiske & Chandler, 2011). Detection probability is the relationship between distance and the likelihood of detecting an organism (Buckland et al., 2001). Within the unmarked package, gdistsamp organizes data into three components: a distance matrix with observed birds per distance bin, the site covariates detailing environmental and site influences, and the detection covariates, which accounts for factors that would influence detection. Repeated measures are incorporated into these matrices. This model also accounted for the mobility of birds by incorporating temporary emigration, where individuals move in and out of the survey area throughout the sampling period (Chandler et al., 2011). This represents a superpopulation meaning the abundance estimates represent the number of individuals that could potentially use the point count area (Chandler et al., 2011). See Appendix 3 for distance sampling equation and assumptions.

I performed repeated counts (see above for how repeated measures are incorporated) and integrated environmental covariates into my models to reduce measurement errors (e.g., false detection), improve detection probability, account for variation, and provide more accurate estimates of abundance (Kellner et al., 2023). Site covariates, such as elevation and habitat variables, were used to account for site-specific differences. I also included several detection co-variates such as wind, time, and date. Prior to analysis, I standardized all detection and site covariates using a z-transformation (mean of 0 and SD of 1).

2.2.5 Abundance Estimates

I followed the unmarked analysis workflow outlined in Kellner et al. (2023) to estimate abundance. After completing the study design and data collection steps, I completed the data input and organization step by creating an unmarkedFrame to store an observation matrix, site covariates, and detection covariates. Given that most research on avian abundance has focused on a single season, current statistical approaches are unable to handle multiple seasons of point count data for sites, as well as multiple visits within each season. Thus, I evaluated each season separately. I identified the key function of ‘halfnormal’ to create the most parsimonious null model based on it having the lowest Akaike’s Information Criterion corrected for a small sample size (AIC_c) score (out of models with key functions of: normal, hazard, exponential, and uniform; (Bozdogan, 1987), and used it in subsequent gdistsamp models to estimate abundance.

I then fit the models using the gdistsamp function where each model includes 3 formulas estimating: abundance (λ), availability (ϕ), and detection (ρ). Local abundance is the superpopulation of birds that could potentially occupy the plot, availability is the portion of the superpopulation that could be detected. The observation matrix influences abundance estimates, while the site and detection covariates influence availability and detection, respectively. I then used a correlation matrix and constructed models with non-correlated combinations of covariates to assess potential covariation. In the model selection step, I compared all gdistsamp models using AIC_c scores to determine the most parsimonious model for each season. Finally, I used the most parsimonious gdistsamp model to estimate the abundance of birds at each site using the GetP function, with the top model for each season outlined in Table 2.3.

Table 2.3. Gdistsamp models used to estimate bird abundance for each season.

Season	**Abundance (λ)**	**Availability (φ)**	**Detection (ρ)**
Spring Migration	altitude + desert + field + agriculture + forest + water + privateroad + landclass	date + time	wind + cloud + date + time
Breeding Season	altitude + desert + field + agriculture + forest + water + buildings + landclass	date + time	wind + cloud + date + time
Fall Migration	altitude + desert + field + agriculture + forest + water + roads + landclass	date + time	wind + cloud + date + time
Early Non-breeding Season	altitude + desert + field + agriculture + forest + water + roads + landclass	date + time	wind + cloud + date + time
Late Non-breeding Season	altitude + desert + field + vineyard + forest + water + privateroad + buildings + landclass	date + time	wind + cloud + date + time

2.2.6 Species Richness

I calculated the average species richness at each site using the vegan package in R V2.1.1. (Oksanen et al., 2024; R Core Team, 2021). I analyzed each season separately by calculating the average species richness per site per season, including only birds that were observed within 100 meters from the observer. To determine average species richness, I summed the number of species observed during each point count and divided this sum by the number of repeated point counts at the site for each season. These average richness outputs were then used in subsequent analyses to understand which factors influence changes in species richness across seasons and land classes.

2.2.7 Seasonal Analyses

I used Linear Mixed Effects Models (LMEs) to examine seasonal variation in species richness and abundance. I used the average abundance and average species richness for each site during each season. The abundance and richness were normally distributed, and therefore, I used the 'lme4' and 'lmerTest' packages in program R to fit LMEs using the 'lmer' function (Bates et al., 2015; Kuznetsova et al., 2017) to test the Winter Urban Association and Urban Homogenization Hypotheses. Predictor variables included land class, season, and their interaction. The response variable was either abundance or richness, with site as a random effect to account for repeated measures between seasons. I created a pairwise comparison model, using lsmeans, to assess the patterns and interaction across season and land class, with a Tukey adjustment to control for multiple comparisons.

2.2.8 NMDS Analysis

I tested the Urban Homogenization Hypothesis (Table 2.1) using non-metric multidimensional scaling (NMDS), an unconstrained ordination, in R using the vegan package to assess how species composition varies seasonally (Oksanen et al., 2024). I analyzed each season separately and created an encounter history of the birds that were observed during each point count. I excluded rare species, and thus only included species that were observed at more than 10 point counts during a season (Spring, *N* = 35 species; Breeding, *N* = 48 species; Fall, *N* = 37 species; Early non-breeding, *N* = 31 species; Late non-breeding, *N* = 27 species). This approach simplified the model, lowered the stress scores, and focused on species that were important for understanding community composition. I used the Bray-Curtis distance metric because the ordination plots most accurately represented the bird community composition and structure (see Appendix 4 for further rationale). I chose three dimensions (k = 3) for each season to visualize these data because it resulted in the lowest acceptable stress score (0.2; see Appendix 4 for more details). For the fall migration and non-breeding seasons, I added a constant (0.05) to each observation because there were many sites where no birds were seen leading to too many zero values in the encounter histories and by adding the constant, I was able to improve robustness of the analysis (McCune & Grace, 2002). All stress values were below 0.2 (Appendix 5).

2.3 Results

I conducted 2380 point counts between March 2022 to March 2023, totalling 11,900 minutes (~200 hours) of observation. Throughout this year-long study, I identified 146 species (*N* = 51,682 individual birds; Table A6.1). The most common bird species were European starlings (*Sturnus vulgaris*; *N* = 10,432 observations), House sparrows (*Passer domesticus*; *N* = 9,463 observations), and California Quails (*Callipepla californica*; *N* = 5000 observations). The mean species richness per site was 5.74 ± 0.064 se (median = 6, range: 0–17) and the mean bird abundance per site was 82.8 ± 1.55 se (median = 80.62, range: 4.99–231.72). Overall, the number of species observed varied between seasons and land uses (totals for species observed during point counts are listed in Table A6.2), with natural habitats having the highest total number of observed species across each season, while urban areas had the lowest number of total species for every season except the late non-breeding season.

2.3.1 Mean Abundance and Richness Across Seasons

In my assessment of seasonal patterns using Linear Mixed Effects Models, I found that both abundance and richness were significantly higher during breeding season (120.1 ± 2.64 SE, 95% CI: 114.8–125.3), followed by spring migration (104.8 ± 2.64 SE, 95% CI: 99.6–110.0; spring migration vs breeding: $t = -6.363$, $P < 0.0001$; Figure 2.2). Additionally, mean bird abundance was not significantly different between early and late non-breeding season (LME, $P = 0.7346$; Figure 2.2), indicating that the abundance of birds did not change between October and March. For richness, all seasons were significantly different (LME, $P < 0.0001$; Figure 2.2). Richness was highest during the breeding season (7.99 ± 0.21 SE, 95% CI: 7.58–8.39), followed by spring migration (6.77 ± 0.21 SE, 95% CI: 6.36–7.17; spring migration vs breeding: $t = -4.974$, $P < 0.0001$).

2.3.2 Seasonal Abundance and Richness Across Land Classes

I showed support for the Winter Urban Association Hypotheses (Table 2.1) because abundance and species richness were elevated in urban and peri-urban areas during the early and late non-breeding seasons (Figure 3.4; Abundance: Table A6.3; Richness: Table A6.4). During the early non-breeding season, the mean abundance was highest in urban, peri-urban, and agricultural habitats, while mean species richness was highest in peri-urban, urban, and natural habitats. During the late non-breeding season, the mean abundance and mean species richness was highest in peri-urban and urban habitats, indicating that birds are using urbanized habitats (urban and peri-urban) more than other habitats during the non-breeding seasons.

There was support for the Urban Homogenization Hypothesis (Table 2.1) by comparing abundance and richness in different land classes across the most productive seasons (spring migration, breeding season, and fall migration). Abundance and richness displayed different seasonal results during these three productive seasons (Figure 2.4; Abundance: Table A6.3; Richness: Table A6.4). Mean abundance was highest in urban and peri-urban areas across all three seasons. During the breeding season, abundance in urban areas was very similar to abundance in natural habitats. Richness showed a different pattern where urban areas have low species richness across all three seasons. During spring migration, there is no significant difference in species richness between any of the land classes. During both the breeding season and fall migration, richness was lowest in urban areas, while abundance was high.

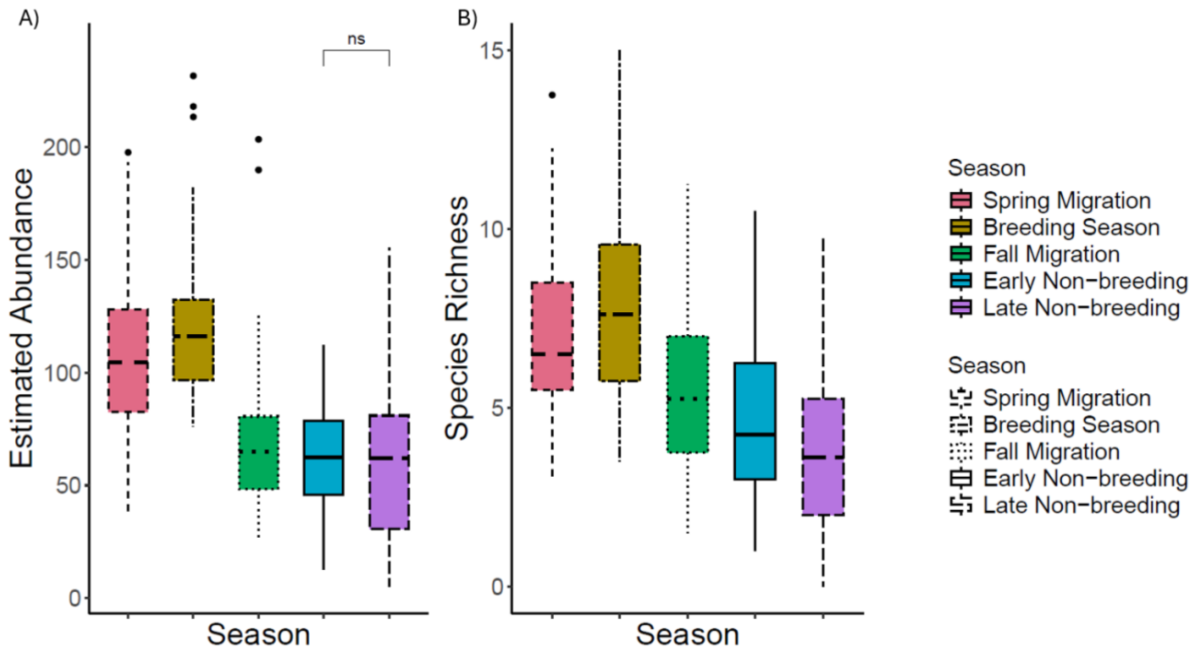

Figure 2.2. A) Mean seasonal abundance displaying non-significance between fall migration (green) and early and late non-breeding season (blue and purple). Mean abundance was highest during the breeding season (gold; 120 ± SE 29.4) and spring migration (pink; 104 ± SE 31.8), and lowest throughout the fall migration and early and late non-breeding seasons (fall migration 68.5 ± 28.4, early non-breeding 61.8 ± 21.4, late non-breeding 58.6 ± 31.3; mean bird abundance ± SE). B) Mean seasonal richness, showing non-significance between fall migration (green) and early non-breeding season (blue). Species richness was highest during the breeding season (gold; 7.98 ± SE 2.64) and spring migration (pink; 6.76 ± SE 1.99) and lowest in the late non-breeding season (purple). The horizontal black line within the box represents the median value, and the box represents 50% of the data, from the 25th to 75th percentile of each groups' distribution of values, which is the interquartile range (IQR). The whiskers extend to the highest and lowest points within 1.5 times the IQR and individual dots are outliers beyond the whiskers.

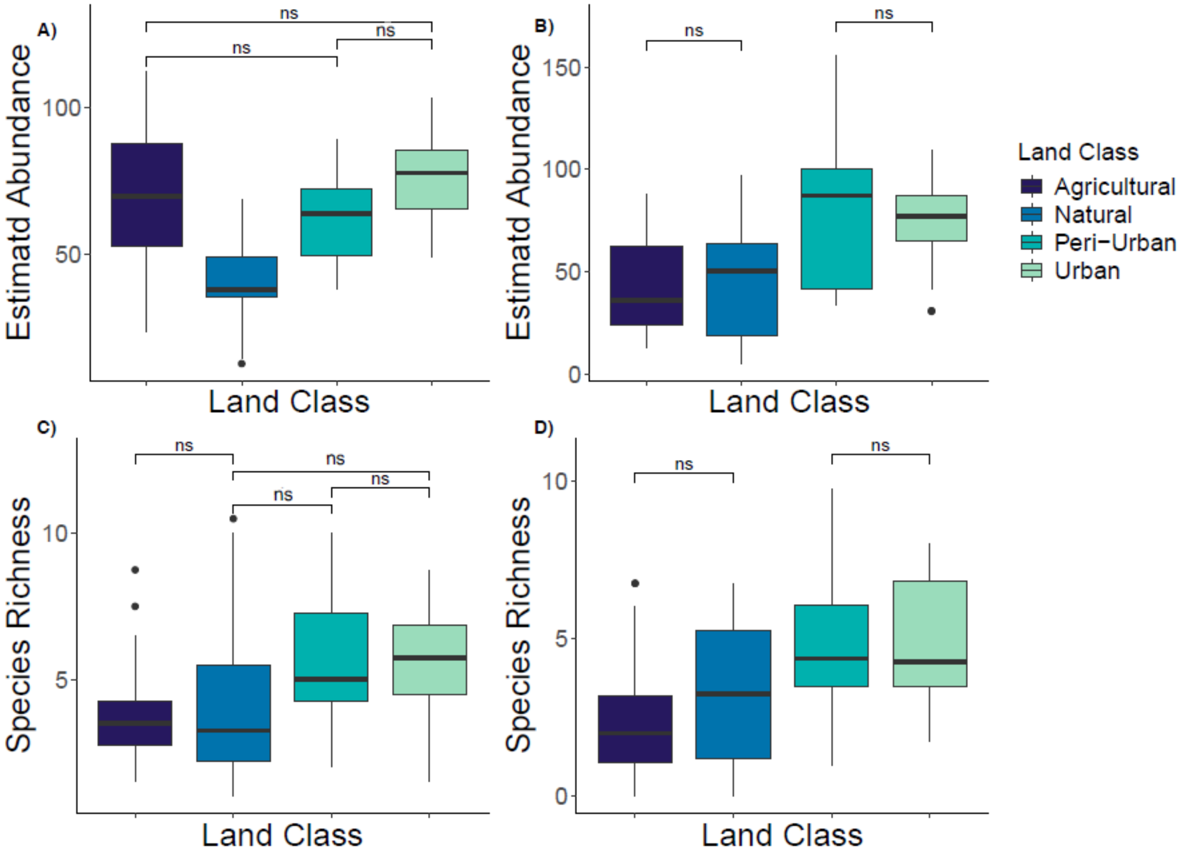

Figure 2.3. Box plots showing the estimated relative abundance during the early non-breeding seasons (A and C) and late non-breeding season (B and D). A) Estimated abundance was highest in urban areas (75.4 ± SE 3.96) though not significantly different from peri-urban (62.3 ± SE 4.37), and agricultural (70.1 ± SE 3.98) areas. B) Estimated abundance was highest in peri-urban (80.8 ± SE 4.42) and urban (74.3 ± SE 3.99) areas. C) Species richness was highest in urban (5.50 ± SE 0.378) and peri-urban (5.46 ± SE 0.420) areas, though not significantly different from natural habitats (4.25 ± SE 0.372). D) Species richness was highest in peri-urban (5.03 ± SE 0.427) and urban (4.79 ± 0.383) areas. See Table A6.3 and Table A6.4 for P-values. The horizontal black line within the box represents the median value, and the box represents 50% of the data, from the 25th to 75th percentile of each groups' distribution of values, which is the interquartile range (IQR). The whiskers extend to the highest and lowest points within 1.5 times the IQR and individual dots are outliers beyond the whiskers.

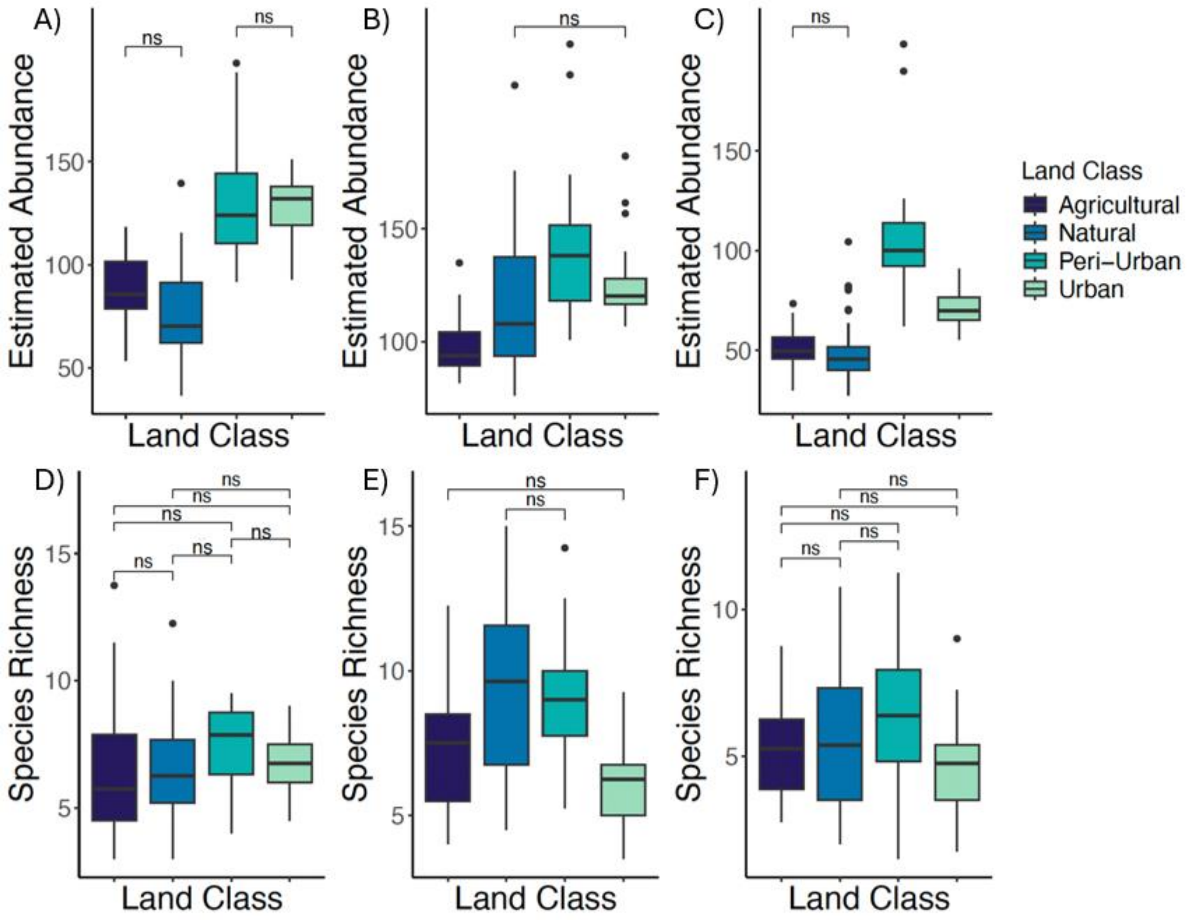

Figure 2.4. Box plots showing estimated abundance and species richness during the spring migration (A and D), breeding season (B and E) and fall migration (C and F). A) Estimated abundance was highest in urban (132.5 ± SE 4.30) and peri-urban (127.9 ± SE 3.95) areas during the spring migration. B) Estimated abundance was highest in peri-urban (142.2 ± SE 4.32) and urban (125.1 ± SE 3.96) areas during the breeding season, though similar to natural habitats (117.9 ± SE 3.90). C) During the fall migration, estimated abundance was highest in peri-urban (107.8 ± SE 4.32) and urban (71.9 ± SE 3.96) areas. D) Species richness was not significantly different between any land classes during the spring migration. E) Species richness was highest in natural (9.47 ± SE 0.372) and peri-urban (9.10 ± SE 0.413) areas during the breeding season. F) During the fall migration, species richness was lowest in urban areas (4.65 ± SE 0.378) and not significantly different between any other land classes. See Table A6.3 and Table A6.4 for P-values. The horizontal black line within the box represents the median value, and the box represents 50% of the data, from the 25th to 75th percentile of each groups' distribution of values, which is the interquartile range (IQR). The whiskers extend to the highest and lowest points within 1.5 times the IQR and individual dots are outliers beyond the whisker.

2.3.3 Species Communities and Habitat Types

I examined the dynamics of bird communities and their habitat associations (objective 2) by visualizing patterns of similarity or dissimilarity between seasonal bird communities. Synanthropic species, which were associated with buildings, included House Sparrows (*Passer domesticus*), Eurasian Collared-doves (*Streptopelia decaocto*), and California Quail (*Callipepla californica*). Species associated with vineyards included Vesper sparrows (*Pooecetes gramineus*), American Robins (*Turdus migratorius*), and European Starlings (*Sturnus vulgaris*). Species associated with deserts included Canyon Wrens (*Catherpes mexicanus*), Western Meadowlarks (*Sturnella neglecta*), Black-billed Magpie (*Pica hudsonia*), and Lazuli Bunting (*Passerina amoena*), and Western Bluebirds (*Sialia mexicana*). Forest habitat was strongly associated with the Western Wood-peewee (*Contopus sordidulus*). Natural habitat around water was associated with Song Sparrows (*Melospiza melodia*) and Red-winged Blackbirds (*Agelaius phoeniceus*). During the non-breeding seasons, water was associated with waterfowl, such as Mallards (*Anas platyrhynchos*), Buffleheads (*Bucephala albeola*), and Common Goldeneye (*Bucephala clangula*; figures and more details are in Appendix 7 and Figure A7.1). The late non-breeding season, the spring migration, and the breeding season show similar patterns, with urban and natural habitats have dissimilar species compositions (Figure 2.6; Figure A7.2).

In NMDS1 and NMDS2 there is a distinct separation between the natural and urban areas, which suggests that urban and natural habitats are composed of different species. Natural habitats also have greater variation in species (more spread out over NMDS1 and NMDS2) compared to urban sites where points are more clustered together, indicating that urban areas have similar community composition, which may be indicative of lower diversity. In contrast, in peri-urban and agricultural areas there is considerable overlap suggesting similarities in species composition. However, in agricultural areas there is more diversity of species across sites. The fall migration and early non-breeding seasons show different results from the other three seasons in terms of their species composition (Figure 2.6). During the fall migration, I observed an absence of discernible patterns across land classes, as there is a lot of overlap between the land use circles. This observation implies that there is a more uniform distribution of bird species across various sites. During the early non-breeding season, there is some division among land class circles, implying increased variation between the sites when compared to the fall migration.

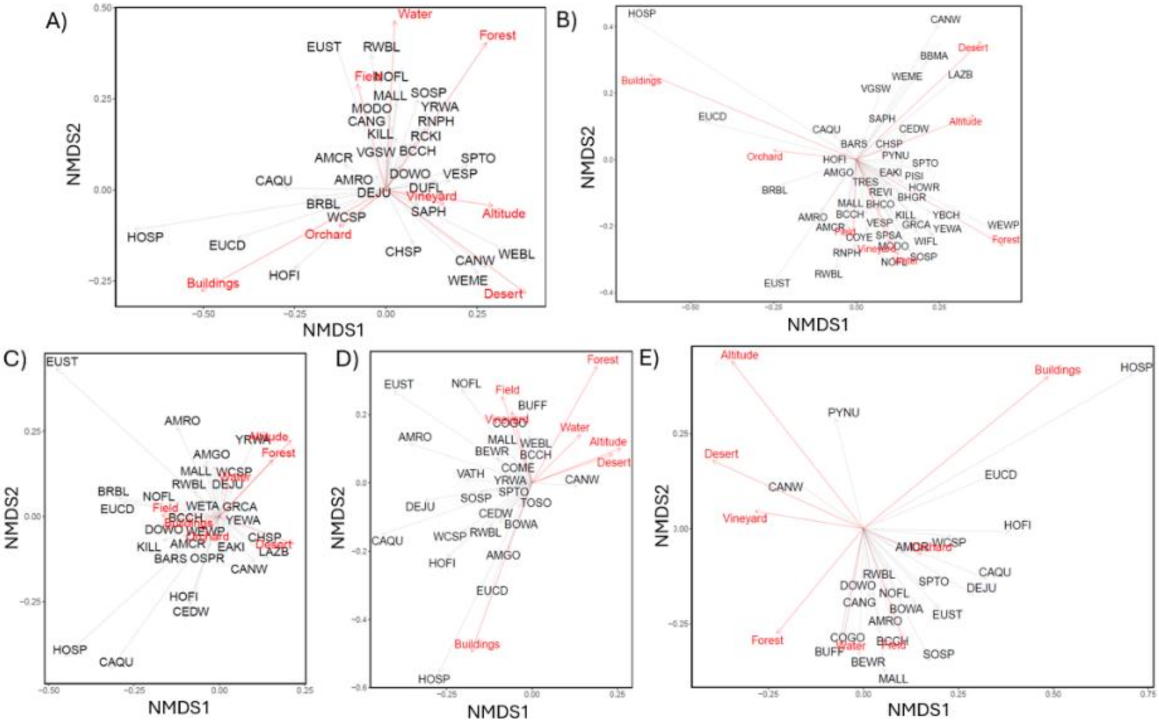

Figure 2.5. NMDS comparing species habitat associations across A) the spring migration B) breeding season, C) fall migration, D) early non-breeding season, and E) late non-breeding season. NMDS1 is on the X axis and NMDS2 is on the Y axis. Additional information can be found in Appendix 7 and remaining NMDS comparisons (1 vs. 3 and 2 vs. 3) can be found in Figure A7.1. Birds are represented by four-letter banding codes and the corresponding scientific and common names can be found in Table A6.1.

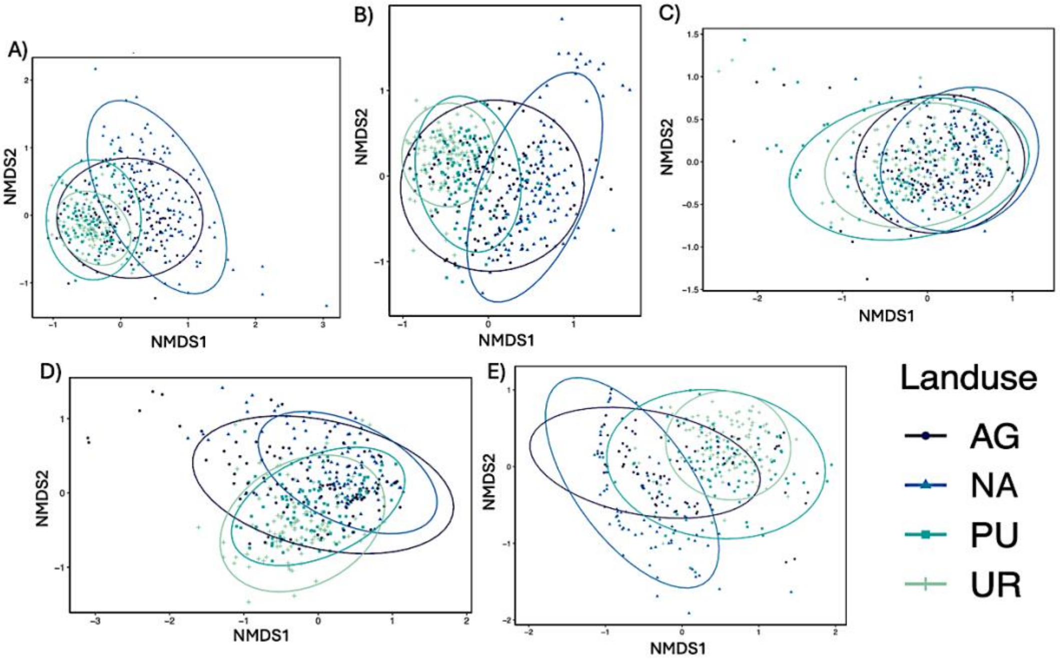

Figure 2.6. NMDS comparing species compositions across urban (UR), peri-urban (PU), agricultural (AG), and natural (NA) areas. I also compared four seasons: A) the spring migration B) breeding season, C) fall migration, D) early non-breeding season, and E) late non-breeding season. Each point in the figure represents a point count and its location on the NMDS is based on the birds that were observed during that point count. Points that are closer together are more similar in species composition and points that are further apart are dissimilar in their species composition. The circles around the points show a 95% confidence interval. I can see a distinct separation between the natural and urban areas, which suggests that urban and natural habitats are comprised of different species. Remaining NMDS comparisons (1 vs. 3 and 2 vs. 3) can be found in Figure A7.2.

2.4 Discussion

This study provides a comprehensive year-round analysis of bird abundance, richness, and community composition across a diversity of habitats and human modified landscapes. I demonstrate how current analyses focusing on a single season (e.g. the breeding season) can limit understanding of species habitat use and needs, and thus, ecological processes that may be influencing these patterns of use. For instance, my results showed that abundance and richness was highest during spring migration and breeding and that urbanization impacted the distribution and community composition of species year-round. In areas where there are mosaics of habitats, such as in my study region, the diversity of habitats may serve different ecological purposes and provide resources or other benefits for numerous diverse species communities, which can vary across the year. For example, urban, peri-urban, and agricultural habitats may provide very important habitat during non-breeding for numerous resident and migratory species, as these land class types had the highest abundances and species richness of birds during these seasons. The findings highlight the importance of year-round studies in assessing avian community composition and habitat selection, challenging long-held assumptions that these patterns remain consistent across regions.

Our findings do not align with past research on seasonal bird abundance in North America, which has predominantly focused on Eastern regions during the breeding and sometimes migratory periods, and this has led to a biased understanding, and often generalization, of seasonal patterns of avian abundance. My observation that local bird abundance was higher during spring migration and breeding compared to fall migration and non-breeding differs from results found in other regions of North America. Another study in western North America (southwestern Arizona) also observed higher bird abundance during spring migration compared to the fall migration (van Riper III et al., 2008). Yet, a common assumption in North America suggests that bird abundance peaks during the fall migration, which is opposite of what I observed in my study. These assumptions are based on numerous studies using mainly eastern North American sites, such as Dokter et al. (2018), which used weather radar data from across the USA and found that the total biomass of migrating birds was higher during the fall than the spring, and Horton et al. (2023), which used wood warbler banding data across North America to show that there were over three times as many wood warblers banded during the fall than the spring. One big reason for the differing patterns observed in western North America may be related to its diverse topography and abruptly changing habitats (Carlisle et al., 2009), such as numerous mountain ranges that create elevational gradients and additional obstacles for migrating birds (La Sorte, Fink, Hochachka, DeLong, et al., 2014; La Sorte, Fink, Hochachka, Farnsworth, et al., 2014). It is unclear if total abundances of birds is actually lower in western Canada, or whether birds take advantage of elevational gradients and higher elevations for food during the fall migratory period (Wilson & Martin, 2005). Based on my findings, it is evident that more research is needed on bird abundance and richness patterns throughout the year, especially in western North America. This will help with avoiding generalized assumptions on bird abundances and richness patterns based on studies from geographically different areas. Such generalization could have detrimental consequences if conservation or restoration initiatives apply standardized patterns of abundance across broad geographically different areas, emphasizing the need for more independent year-round assessments of bird abundance, richness, and associated community structures in western North America.

Urbanization may decrease the propensity of some birds to migrate and many species that remain year-round choose to stay within urban habitats (reviewed by Bonnet-Lebrun et al., 2020). While many birds migrate south during fall migration and the non-breeding season, I found that abundance remained stable throughout early and late non-breeding. This is unsurprising as many altitudinal and short-distance migrants remain in Canada during the winter (Bonnet-Lebrun et al., 2020; Boyle, 2017), with western North America having the highest number of altitudinal migrant species (Boyle, 2017). I also found that the birds that remained during non-breeding were primarily using urban and peri-urban habitats, as evidenced by the higher abundance and species richness compared to less urbanized agricultural and natural habitats. These patterns have also been seen in other North American studies (Clergeau et al., 1998; Murthy et al., 2016; Yaukey, 1996). In British Columbia, many species move out of high elevation habitats during the non-breeding season, selecting lower elevation ecosystems instead (Herbers et al., 2004), which may provide resources found in abundance in urban habitats. Supplemental food in the form of feeders, garbage, and cultivated plants can increase the abundance of birds in urban areas (McKinney, 2002), which can be beneficial for birds during the sparse non-breeding season (Grubb & Cimprich, 1990; Yaukey, 1996; Zuckerberg et al., 2011). Urban areas also act as heat islands, increasing the average temperature (McKinney, 2002), and thereby making urban habitats more hospitable for birds during the winter. These factors may all work together in the south Okanagan to drive birds into urban habitats, which is why urbanization is important in dictating the movement and community composition of birds during non-breeding (Bonnet-Lebrun et al., 2020; Leveau et al., 2021; Saunders et al., 2022; Zuckerberg et al., 2011), and why it is important to assess abundance, richness, and community composition of birds over a variety of habitats including urban areas during non-breeding seasons for proper management and conservation of overwintering species.

During early non-breeding, agricultural habitats had similar species composition and richness relative to natural habitats, and similarly high abundance compared to urban and peri-urban habitats. These findings differ from other studies and highlight the significant role of human altered landscapes in driving community structure during the non-breeding period. Recent studies have assessed the dramatic decline of farmland and grassland breeding birds in North America (Murphy, 2003; Stanton et al., 2018; Valiela & Martinetto, 2007; van Vliet et al., 2020), leading to the general assumption that agricultural habitat is bad for songbirds and allies and very few species can benefit from access to agricultural food sources (Murphy, 2003; Stanton et al., 2018; Valiela & Martinetto, 2007; van Vliet et al., 2020). However, all these studies were focused only on a single time period, the breeding season. While my results also showed that agricultural habitats had the lowest abundance and richness of birds during the breeding season, this was not the case year-round. Thus, my study shows that conducting research in only a single-season and making broad scale inferences of habitat quality or use for birds may be short-sighted, and these single season studies limit the full understanding of the benefits (or detriments) of agricultural land to a wide diversity of songbirds year-round. The high abundance of birds in agricultural habitat during early non-breeding may be linked to the availability of remaining grapes and seeds, with grapes being a major crop in the south Okanagan valley. Other studies using eastern North American eBird data have also shown that some birds transition to agricultural areas during fall migration (Zuckerberg et al., 2016), as changes in resource availability drive the movement and habitat selection of birds. The potential benefits of agricultural (and urban and peri-urban) areas for birds at various times of the year has typically been overlooked (or not even included) in past studies, but I have shown agricultural land plays a critically important role in support of bird communities outside of the breeding season.

Many species use urban habitats throughout the year, but species composition within each season is often similar across urban sites, typically dominated by urban exploiters such as Eurasian Collared-doves (*Streptopelia decaocto*) and House Sparrows (*Passer domesticu*s), resulting in homogenization characterized by low species richness and high abundance (Beissinger & Osborne, 1982; Butler, 2003; Fontana et al., 2011; Ortega-Álvarez & MacGregor-Fors, 2009). This pattern was evident during spring migration, breeding, and fall migration, where most species were urban exploiters, which were in high abundance. But during non-breeding, a wider diversity of species used urban habitats, including species not typically common in urban areas during other times of the year. This led to increased abundance and species richness, while maintaining homogenization as the same species were observed across all urban sites. Non-breeding homogenization appeared to result from broad, equal use of urbanized habitats by many species, including altitudinal migrants like Varied Thrushes (*Ixoreus naevius*) and Dark-eyed Juncos (*Junco hyemalis*). However, there may be elevated urban-associated risks (such as window collisions and predation by domestic cats) for some species that are not adapted to urban environments and may be drawn to urban areas for the abundant resources. For example, in Vancouver, British Columbia, Varied Thrushes (a common urban species during non-breeding in this study) are 76.9 times more likely to collide with glass compared to other species (De Groot et al., 2021). Homogenization can also have a negative, destabilizing effect on bird communities (Devictor et al., 2007), but targeted urban habitat management is important for proper bird conservation, not just during non-breeding, but also year-round. Community composition is unique in urban habitats and birds occupying this space face unique challenges that must be examined and considered within the unique and complex environments that urban areas create.

In summary, I demonstrate the importance of identifying where birds are and what habitats they are using across the entire year and not only during distinct periods within the annual cycle (e.g. breeding). In addition, I emphasize caution in applying broad generalization about patterns of bird abundance, richness, and communities at the continental scale without adequate geographic and context specific knowledge and understanding. Urban habitats provide important habitat for birds year-round yet are often only considered to support a limited number of species, but my study suggests that urban areas provide important resources for a large variety of bird species throughout the year and especially during migration and non-breeding seasons. Effective conservation of birds and habitat management requires the consideration of the entire annual cycle, the unique challenges that birds face in western North America, and their seasonal changes in habitat use. Thus, more research is needed that addresses and studies birds across the entire year and among a variety of ecozones and habitats.

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Chapter 3: Seasonal variation in domestic cat (*Felis catus*) abundance, population size, and habitat associations across a mosaic of habitats

3.1 Introduction

Domestic cats (*Felis catus*) have retained their ancestral hunting behaviours, enabling them to adapt to a variety of environments, both with and without human intervention (Bradshaw, 2006; Canadian Federation of Humane Societies [CFHS], 2017). These innate hunting abilities allow cats to capture a variety of wild animals, posing serious global threats to small vertebrate populations when they are allowed to wander unsupervised outdoors (i.e., wandering cats; Loss et al., 2013; Trouwborst et al., 2020). When cats are allowed to wander freely outside, they may experience increased risks of injury and illness through predation by other animals, vehicle collisions, and disease transmission of illnesses such as toxoplasmosis (reviewed in Tan et al., 2020). Consequently, various groups (e.g., veterinarians, shelters, humane societies, conservation organizations, policy makers, and members of the public) are concerned about issues surrounding wandering cats (detailed in Saunders et al., 2021; Wald & Peterson, 2020) and find common ground in the desire to curtail the number of wandering cats.

Data on local cat abundance and population sizes is needed for municipal governments to make informed policy changes and for organizations, including non-governmental organizations such as the Stewardship Centre for British Columbia, to develop educational campaigns and incentive programs promoting responsible cat ownership. The absence of such data on wandering cat populations poses a significant challenge to implementing effective management strategies, including municipal bylaws and policies. Furthermore, abundance and population estimates are valuable to establish a baseline for wandering cat populations, which is crucial for informing future management decisions and evaluating the effectiveness of implemented strategies. Yet, many studies on wandering cats have focused on only the spring and summer seasons (Bennett et al., 2021; Coleman & Temple, 1993; Flockhart et al., 2016; Hand, 2019; Hanmer et al., 2017; Kays & DeWan, 2004). Therefore, there is limited knowledge on the seasonal dynamics of wandering cat abundance and the factors that influence seasonal variation.

Wandering cats may inhabit and use a variety of habitats, which may be influenced by both human activities and the ecological needs of cats (reviewed by Tan et al., 2020), but only one study has directly explored seasonal variation in wandering cat abundance in North America (Clyde et al., 2022). They found that wandering cat local abundance estimates were higher in spring and summer compared to the fall and winter (Clyde et al., 2022), suggesting that human behaviour (e.g. owners not putting cats out as much in the winter) may influence cat abundances (Human Behaviour Hypothesis). Another study, conducted in Australia examined broad patterns of seasonal variation, showing that cat abundance decreased during dry periods and increased following periods of rainfall, suggesting that prey availability may drive these patterns (Prey Availability Hypothesis; Legge et al., 2017). Based on these two hypotheses, if cat abundances were highest during periods with most favourable temperature (i.e., not below 0°C nor above 25°C), then human behaviour might be the main factor influencing seasonal patterns. In contrast, if cat abundances are related to peak prey abundances, such as in the spring and fall, then prey availability may be influencing seasonal patterns.

The distribution and associated abundances of wandering cats are strongly influenced by human presence, cat ownership, and human and cat behaviour (Bennett et al., 2021; Clyde et al., 2022; Cove et al., 2023; Gow et al., 2024). In Canada, approximately 37% of households own at least one cat, with an average of 1.64 cats per household, and 28% of households allow their cat(s) unsupervised outdoor access (CFHS, 2017). When outdoors, cats typically have small home ranges, often remaining within 100 meters of their homes (Dunford et al., 2024; Kays et al., 2020), suggesting the highest densities of cats will be in areas with high human populations, such as urban areas where there is a high density of people and buildings (Human Presence and Cat Behaviour Hypothesis). Less urbanized areas, such as peri-urban areas, may also have high abundances of cats. Peri-urban areas tend to have fewer buildings than urban areas and are typically surrounded by agricultural lands or natural habitat. Cats living in peri-urban areas often have access to a wider variety of bordering habitats and have larger home ranges compared to those in urban habitats (Hall et al., 2016; Hanmer et al., 2017). Peri-urban households also frequently have more cats (Dozier et al., 2023) and are more likely to allow their cats outdoors (Clancy et al., 2003). Therefore, relative abundance estimates may be similar between urban and peri-urban habitats depending on human and cat behaviour (Hall et al., 2016; Hanmer et al., 2017; Kays et al., 2020).

Areas with lower levels of urbanization, such as agricultural and natural areas, are expected to have lower abundances of wandering cats. Agricultural habitats, such as orchards and vineyards, typically have few buildings, yet have similar numbers of cats per household to peri-urban areas (Coleman & Temple, 1993; Dozier et al., 2023). In addition to owned cats, unowned barn cats are common in agricultural areas (i.e., cats that may be fed but have no restrictions on movement). These cats are often unrestricted in their movement and used for pest control, leading to some cats displaying home ranges that are 1.6 times larger than urban wandering cats (Hall et al., 2016; Hanmer et al., 2017; Kays et al., 2020). Despite large home ranges and high densities on farm, the low human population densities and Human Behaviour Hypothesis lead me to expect that the abundance of wandering cats in agricultural habitats will be lower than urban and peri-urban habitats. Finally, natural habitats, which are characterized, in this study, by forests, deserts, wetlands, and few buildings, would have the lowest abundances of cats, if any. Cats within these areas likely have the largest home ranges (~ 70% larger than urban cats; Pirie et al., 2022) and may live in nearby households or be unowned and survive without (or with minimal) human intervention.

Here, I use trail cameras to quantify cat local abundance across the entire year and four land classes (urban, peri-urban, agricultural, and natural) in the south Okanagan Valley, British Columbia (from Okanagan Falls to Osoyoos). I have three objectives: 1) Identify how cat abundance and population size vary within the study region and across habitat types and seasons; 2) Examine the Prey Availability and Human Behaviour Hypotheses by assessing how local abundance of wandering cats varies seasonally; and 3) Test the Human Presence and Cat Behaviour Hypothesis by examining how local cat abundance varies across habitat types.

3.2 Methods

3.2.1 Study Area

I studied seasonal patterns of wandering domestic cat abundance and habitat associations in the south Okanagan Valley of British Columbia, Canada (within and surrounding the communities of Okanagan Falls, 49.34490N, -119.57328E; Oliver, 49.18276N, -119.54959E; and Osoyoos, 49.031778N, -119.465111E). My study area extends laterally to the hills that border the valley, covering a total area of 161.2 km^2 (Figure 3.1), with a total human population of ~22,217 (see Appendix 8 for more details of the study site boundaries and how population was determined). Located in the south-central region of British Columbia, the south Okanagan Valley is the traditional, unceded, and ancestral territory of the Syilx Okanagan Nation, and one of the most endangered ecosystems in Canada. The Okanagan Valley, including the study region, is heavily settled and has some of the highest numbers of species at risk in Canada (Coristine et al., 2018). Given this, there are numerous conservation plans, initiatives, and programs focused on species at risk recovery and habitat conservation in the region (e.g., Restoration of Okanagan Sockeye Salmon, Alexander et al., 2024 and Correia et al., 2024; Burrowing Owl recovery, Environment and Climate Change Canada, 2012; and Yellow-breasted Chat recovery, McKibbin & Bishop, 2008). It is part of the bunchgrass biogeoclimatic zone that is characterized by grassland, shrub-steppe, wetlands, and dry open forests consisting of ponderosa pines, cottonwoods, and Douglas firs (Alldritt-McDowell et al., 1998). The fertile lands, abundant lakes and rivers, long growing season, and warm climate also make this area highly attractive for agriculture (e.g. vineyards, and orchards), long-term residents, and seasonal recreation. This area consists of a mosaic of natural habitats (grasslands, deserts, riparian areas, and forests), agriculture (e.g. vineyards and orchards), small urban centers, and peri-urban areas (defined in Table 3.1).

Table 3.1. Description of detection and site covariates used to estimate relative abundance of cats in the south Okanagan Valley, B.C. Detection covariates influenced the detection of wandering cats and site covariates describe the area around the trail camera.

Variables	Description
Detection covariates	
date	Julian date when the point count was conducted.
time	Time at the start of the point count observational period.
wind	Strength of wind during point count using the Beaufort scale (Scale 0-4)
cloud	Cloud cover % overhead during the point count.
precip	Binary observation during point count, where 0 = no precipitation and 1 = precipitation.
Site covariates	
desert	Percentage of desert within a 200 m buffer around the point count location.
field	Percentage of field within a 200 m buffer around the point count location. Fields include public spaces such as parks and school yards, land that is used as pasture or farming, and empty, overgrown lots.
orchard	Percentage of orchard within a 200 m buffer around the point count location.
vineyard	Percentage of vineyard within a 200 m buffer around the point count location.
forest	Percentage of forest within a 200 m buffer around the point count location.
water	Percentage of water within a 200 m buffer around the point count location.
roads	Privateroads and publicroads combined, to equal the total length of roads (m) within a 200 m buffer around the point count location.
privateroad	Total length of private roads, in meters, within a 200 m buffer around the point count location. Private roads include driveways (>40 m in length), vineyard/orchard roads, and other impervious surfaces that are not accessible to the public.
publicroad	Total length of public roads, in meters, within a 200 m buffer around the point count location. Public roads are maintained by the city and are easily accessible to the public.
buildings	All buildings within a 200 m buffer around the point count locations. This includes high-density housing and all other buildings.
landclass	The land class type of the point count locations: urban, peri-urban, natural, or agricultural
urban	Site buffer (200 m) containing >55 buildings
peri-urban	Site buffer (200 m) with 17 to 50 buildings and <49% vineyard and/or orchard.
agricultural	Site buffer (200 m) characterized by >28% agriculture and <20 buildings.
natural	Site buffer (200 m) characterized by >25% natural habitat, <16 buildings, and <29% agriculture.
altitude	Altitude (m) at the point count location.

3.2.2 Study Design

I used 60 trail cameras to assess seasonal patterns of cat abundance, population size, and habitat associations using a repeated measures design stratified by season, collecting data across five distinct seasonal periods relevant to birds (a main prey sources of cats) from March 2022 to March 2023. The five seasonal periods assessed were spring (March 8 – May 27, 2022), summer (May 27 – July 30, 2022), fall (August 8 – October 13, 2022), early winter (October 13 – December 16, 2022), and late winter (January 13 – March 23, 2023). This research was part of a larger project also looking at bird abundance and habitat associations, and the resources available to birds are different throughout the winter; thus, winter was divided into early and late seasons. Throughout the study, some sites were added, discarded, or moved, leading to a total of 123 sites (see Appendix 1 for details). Total number of sites per season were as follows: 120 sites for spring migration, breeding season, and fall migration; 119 sites for early non-breeding season; and 114 sites for late non-breeding season. Trail cameras were deployed for at least 4 weeks at each site during each season, then moved to a second site within the same season for four weeks. I established these trail camera sites using a random stratified design, following a framework for completing ethical camera trapping fieldwork in urban environments (i.e., five P's from Herrera et al., 2021: *partnership, planning*, *placements*, *public participation*, and *processing*).

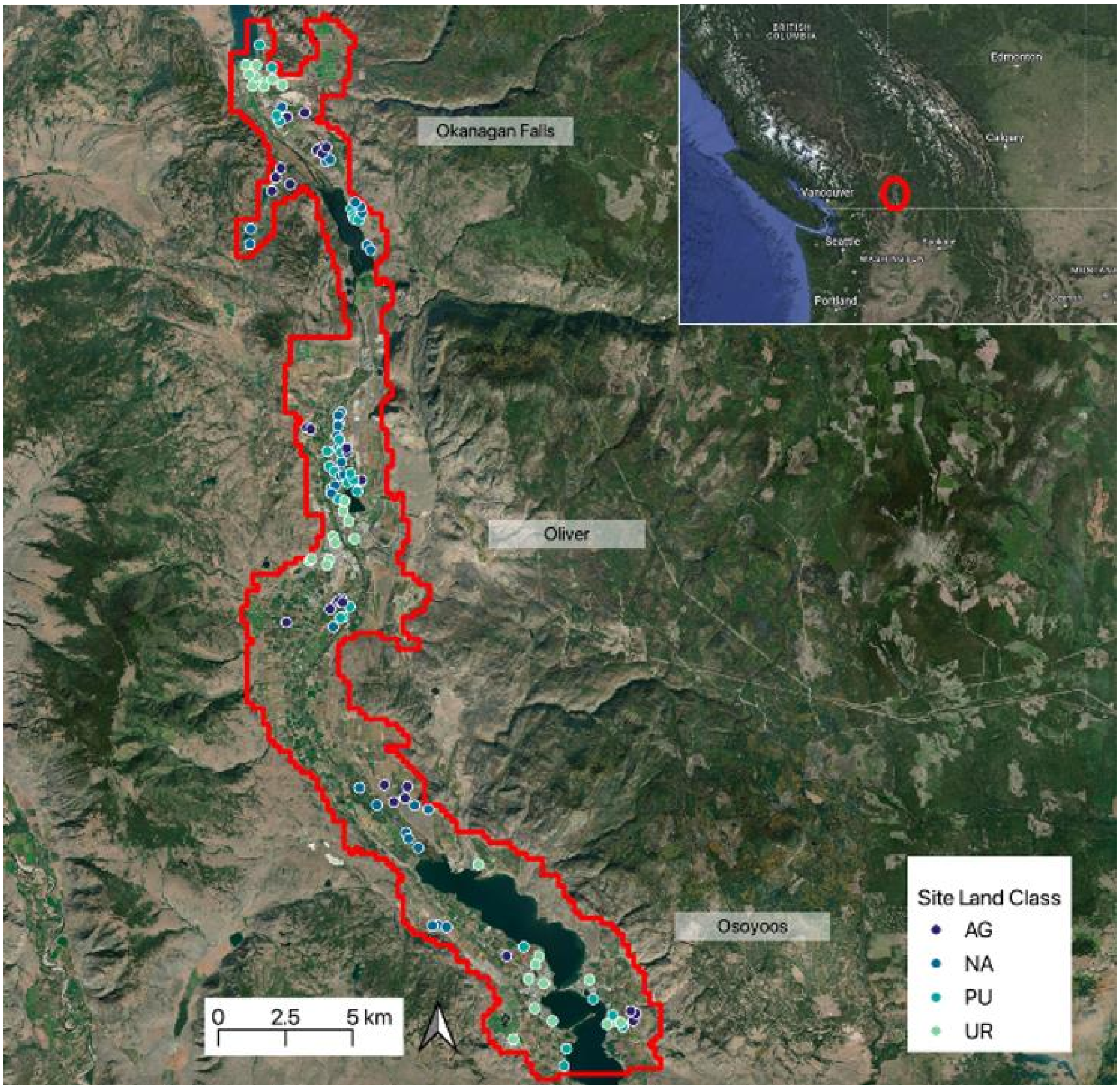

Figure 3.1. Map of the study area in the south Okanagan Valley showing trail camera locations in Okanagan Falls, Oliver, and Osoyoos. Location colours are classified by the following land classes: Agriculture (AG), Natural (NA), Peri-urban (PU), and Urban (UR). The red boarder represents the study area that was used to study wandering cats based on the perimeter of the grid. The inset map in the top right corner shows a red circle around the south Okanagan Valley, located in the south-central area of British Columbia, Canada. Map is in EPSG:3005 coordinate system (NAD83, BC Albers projection).

To determine wandering cat local abundance and habitat associations across an entire year, I used satellite maps of the south Okanagan Valley to identify habitats as urban, peri-urban, agricultural, and natural based on broad definitions of each land class type. At the stage of camera deployment, these broad categories helped me to identify which permits I would need and the neighbourhoods to target for door-to-door recruitment. For the purposes of camera deployment, I categorized urban areas as having extensive anthropogenic development that included a matrix of landscapes such as pavement, buildings (residential and commercial), and habitat patches (e.g., transportation corridors, riverbanks, and parks; Swanwick et al., 2003). Peri-urban areas consisted of clusters of houses surrounded by agricultural (e.g., orchards, vineyards, and crops) and/or natural (e.g., desert, forest, riparian) land with approximately 25-50 m between houses. Agricultural land was used for agriculture (such as vineyards and orchards) or other farming practices, targeting smaller vineyards/orchards that were <50 hectares or houses residing next to agricultural land. Natural habitats included any protected areas conserved provincially, federally, or privately. Since these descriptors are continuous, I later classified sites using QGIS to create discrete variables and refined categories based on surrounding habitat variables for statistical analysis (see 3.2.3 GIS Analysis for more details).

When choosing sites, true randomization and equal distribution of cameras across habitats during each 4-week deployment period was impossible because locations were mainly on private property and therefore dependent on the willingness of landowners and land managers (except natural sites). However, we aimed to have as equal distribution of cameras across habitats as possible during each deployment period. I balanced the distribution of cameras in each of the four land class types: urban (N = 32), peri-urban (N = 27), agricultural (N = 31), and natural (N = 33). This research was co-developed and designed with the Stewardship Centre for British Columbia and the Canadian Wildlife Service and Wildlife Research Division of Environment and Climate Change Canada (ECCC; key contributors and co-authors on this paper). I used several methods to recruit participants including word-of-mouth, networking, and media (see Appendix 9 for further details). Overall, I found that the most successful way of forming partnerships was by going door-to-door to speak with the residents directly and ask if they would be willing to participate in the study and providing them a letter to explain the study and provide contact information (Additional details in Appendix 10; Example letter in Appendix 11).

Once I established trail camera locations, I considered the placement of cameras to maximize cat detection, while ensuring the privacy of residents and the public. I used Browning Strike Force HD Pro infrared trail cameras (model BTC-5HDP) with a 1/3s picture trigger and a five-photo burst, which took enough pictures to identify individual cats and fast-moving animals (Clyde et al., 2022; see Appendix 12 for other technical considerations when using trail cameras). When possible, cameras were pointed towards animal corridors, such as fence lines and accessible paths to increase chances of detecting cats while avoiding capturing pictures of humans (see Appendix 13 for additional ethical considerations I took when placing cameras). Cameras were randomly placed as best as possible (given land use permissions, available posts for attaching cameras, and resident concerns or limitations) with a minimum of ~100 meters between cameras to maintain statistical independence, which is the maximum distance I expect cats to wander based on another study suggesting cats stay with ~100 m of their homes (Kays et al., 2020), Only two sets of sites ($n = 4$) were closer than the desired 100 m apart (70 m and 40 m; see section 3.2.8 Statistical analysis to see how I maintained statistical independence when one cat was identified at multiple sites).

Once pictures were collected, I processed ~7.1 million trail camera images to identify individual cats and create encounter histories of wandering cats over the 123 trail camera sites. Once the images were collected, images of human faces or other identifying features were removed and deleted. Images were processed by individually identifying cats based on their colour, coat pattern, and body size (Figure 3.2 highlights the diversity in coat pattern). Using these markers, along with other features such as the presence and colour of collars, I could identify 98% of cats to an individual level, which was consistent with other studies (Clyde et al., 2022; Elizondo & Loss, 2016). It was only difficult to identify numerous black cats at a trail camera site to an individual level, unless at night where pelage coats differ under infrared photography. In cases where cats appeared identical, I followed the same guidelines as Clyde et al. (2022) and Elizondo & Loss (2016), by assuming that there are as many cats as seen at the same time in one photo. I used the identity of individual cats to create encounter histories consisting of fourteen 24-hour sampling occasions (starting at midnight), counting the total number of cats/camera site during the 24-hour sampling occasion (see 3.2.8 Statistical analysis).

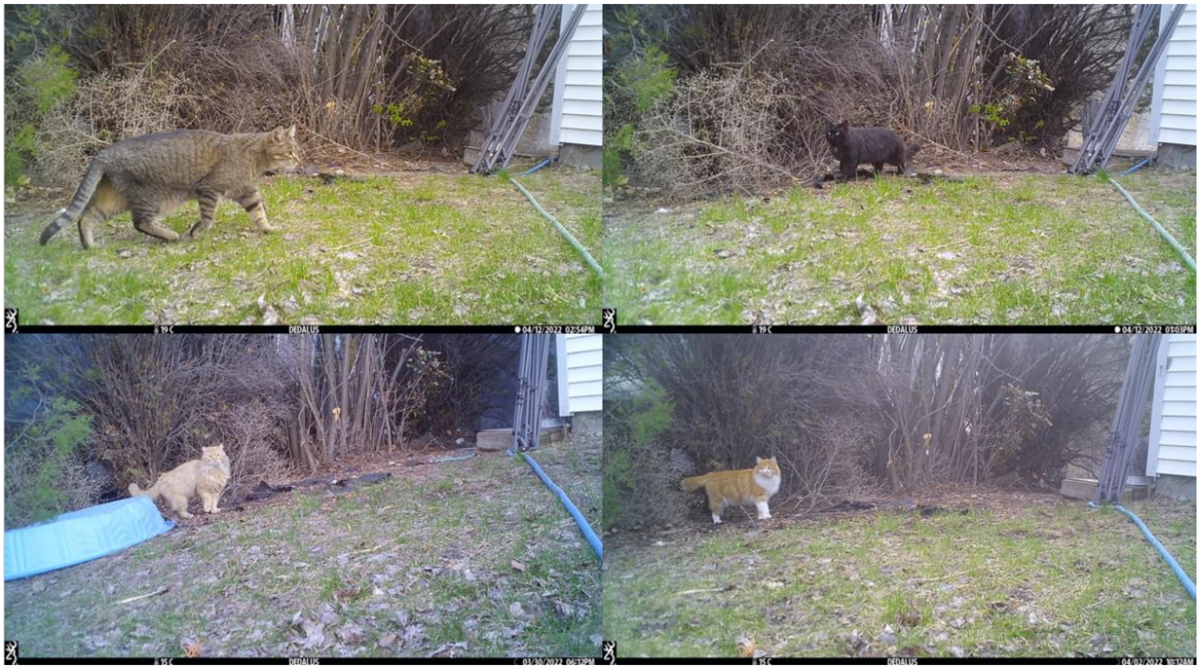

Figure 3.2. Domestic cat coat patterns, as seen on the trail cameras. Each cat has a different coat pattern, making them individually identifiable.

3.2.3 GIS Analysis

Camera locations were stratified based on four broad habitat categories, but to extract specific continuous habitat variables for statistical analyses, I used QGIS (Version 3.32 Lima; QGIS.org, 2024; EPSG: 3005 coordinate system; NAD83, BC Albers projection). I assessed habitat variables over the entire study area (Figure 3.1) that represented buildings, roads, water, vineyards, orchards, deserts, fields, forest, and altitude (defined in Table 3.1). I accessed detailed OpenStreetMap layers for roads, lakes, rivers, and buildings and imported them into QGIS. To obtain missing habitat variables, I created polygons around forests, rivers, deserts, fields, orchards, and vineyards, and updated the buildings using the point creation function in QGIS. I used a 2023 Esri World Imagery Wayback satellite image as a base layer to trace the polygons (Esri, 2023). I then created 200m buffers around each site to extract habitat variable information using the intersect tool (see figure 3.3). Within each 200 m buffer, I quantified the total length of roads, the total number of buildings, and the proportion of desert, field, forest, orchard, vineyard, and water (defined in Table 3.1). The habitat variables extracted from the 200 m buffers were used to re-classify each trail camera location as either urban, peri-urban, natural, or agricultural based on specific quantifiable definitions (defined in Table 3.1; N = 100 correctly classified at start).

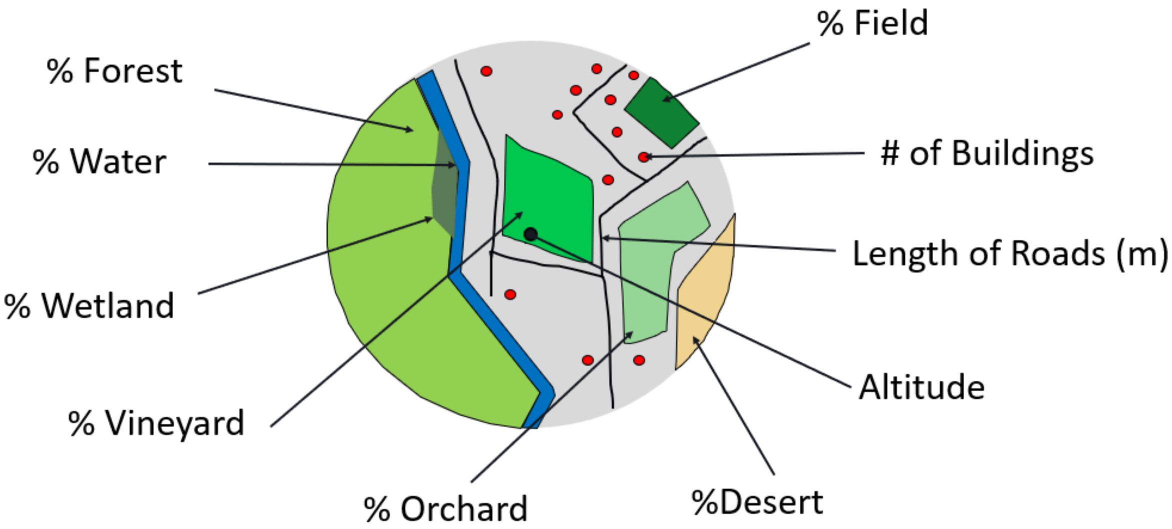

Figure 3.3. Example of 200-meter buffer divided into habitat variables. Variables within each 200 m buffer were used in the analyses.

3.2.4 Statistical analysis

To estimate local cat abundance, I used Hierarchical N-mixture models within the unmarked package (Fiske & Chandler, 2011) in Program R, V2.1.1. (R Core Team, 2021), specifically using the *pcount* model. I followed the methods in Clyde et al. (2022) and Gow et al. (2024), which, in short, involves using hierarchical N-mixture models that incorporate habitat and site covariates to estimate abundance and detection of domestic cats. Detection is the probability (*p*) of detecting an individual, given that there are *N* individuals available (Kéry & Royle, 2016). I used the number of individuals detected at a site during a 24-hour sampling occasion, while accounting for overdispersion and non-perfect detection (Kéry & Royle, 2016). This also keeps sampling periods consistent, reducing over- or under-estimation of local abundance estimates. The ‘unmarked’ model incorporates repeated measures and detection probability to estimate local abundance, while accounting for species that may go undetected during sampling (Fiske & Chandler, 2011). This hierarchical model is made up of two parts: a detection model and an abundance model (Kéry & Royle, 2016; equations in Appendix 14).

I modelled detection using three variables: average daily temperature (temp), Julian date (date), and total daily precipitation (precip; Table 3.1). Temperature can influence the likelihood of owners allowing their cats outdoors, as they may be less likely to allow them outdoors in extreme heat or cold, or cats may move around less during extreme temperatures (Forrest et al., 2023). Date accounts for any temporal patterns in either cat owner behaviour (e.g. people vacationing) or broader seasonal patterns of cats themselves. Cat owners may be less likely to put their cats outdoors when it is raining or snowing, or cats may be less active (and thus less likely to be detected) during periods of precipitation (Forrest et al., 2023; Goszczyński et al., 2009). Temperature and precipitation data was from ECCC from the Osoyoos CS weather station (49°01'41.850" N, 119°26'27.570" W; ECCC 2022). The average seasonal temperatures were 9.6 °C during the spring, 21.2 °C during the summer, 20.6 °C during the fall, 0.7 °C during the early winter, and 1.4 °C during the late winter. The total precipitation was 38.1 mm during the spring, 68.7 mm during the summer, 4.2 mm during the fall, 69.3 mm during the early winter, and 32.9 mm during the late winter.

Assumptions of this model include that the population remains closed during the sampling periods (i.e., each of the five seasons), all individuals have an equal probability of detection, and no animals are counted twice within a season. I assumed there was little to no death or birth within the population during each two-month sampling period and that cats could be equally detected across my study site by ensuring consistency in camera set-up and locations, meeting the first and second assumptions. To meet the third assumption, I took measures to avoid counting cats more than once by spacing cameras 100m apart as was shown to be effective for limiting capturing individual cats at more than one camera. However, there were 38 cats that were identified at more than one site in my study. When this happened within the same season, if possible, I shifted the encounter histories to have 14-day periods where the cat was only present at one site, and therefore only counted in one encounter history, meeting the third assumption. When this was impossible (N = 26) I flipped a coin to decide which site would be used in the encounter history.

3.2.5 Model Selection and Seasonal Abundance Estimates

I followed the unmarked analysis workflow outlined in Kellner et al., (2023) to estimate the abundance of wandering cats. Prior to analysis, I standardized all detection and site covariates using a z-transformation (mean of 0 and standard deviation of 1). Currently, the unmarked package is unable to handle multiple seasons across multiple sites, as well as multiple visits within each season, and therefore I examined each season separately. I used the Variance Inflation Model (VIF) to assess multicollinearity amongst covariates in the global model (included all detection and abundance covariates) and used the chi-squared goodness-of-fit tests for *N*-mixture models, using the *Nmix.gof.test* function from the AICcmodavg package (Mazerolle, 2023) in Program R, V2.1.1. (R Core Team, 2021). All variables had a VIF value between 1 and 4.4, suggesting little to no multicollinearity among the covariates (Murray & Sandercock, 2020).

I created separate models for each season and used the Quasi Akaike Information Criterion (QAIC; Burnham et al., 2011; Burnham & Anderson, 2002) to help account for some overdispersions and the dredge function to determine the top detection and abundance models (ΔQAIC = 0) for each season separately. I first determined the top detection model by creating a detection-only model with all three detection covariates (date, precipitation, and temperature), holding abundance (state) at a constant of 1. For determining the top abundance model, I created a full model that included habitat variables (water, field, vineyard, orchard, natural, roads, and buildings) and carried forward the top detection covariate(s) from the detection-only model. Finally, I used the seasonal local abundance model to estimate the local abundance of wandering cats within each buffer using the *predict* function.

3.2.6 Seasonal Analyses

I used a Kruskal-Wallace test followed by a post-hoc Dunn's test to examine the seasonal variation of wandering cat abundance between seasons and land class types. The abundance data were not normally distributed, favouring non-parametric methods to examine the significance of median differences in seasonal abundance. I used a Dunn's post-hoc test to compare abundance across seasons (total number of wandering cats estimated within the study area during each season), and local abundance across land class types (the number of wandering cats within a specific land class during each season). I generated maps of cat local abundance by first obtaining the habitat variables in 200m x 200m grid cells over the entire study area, then estimated the local abundance of wandering cats within each grid cell using the top seasonal abundance model(s) (without the detection covariate) and set the remaining habitat variables in the model to their mean (additional mapping methods in Appendix 15).

3.2.7 Predicting Total Cat Abundance

The local abundance estimates for each 200 m x 200 m grid cell were combined to estimate the total abundance of wandering cats in the south Okanagan Valley, B.C. I assumed that each grid cell accurately estimated the local abundance of urban wandering cats because cats typically have a home range of 100 m from their homes (Kays et al., 2020). Since water isn't usable habitat for cats, I removed any grid cell that had a proportion of water greater than 50%, which avoided overestimating the population size. Assessing each season separately, I combined the grid cell local abundances for each map created (as described above) and calculated the average total abundance across all habitat covariates being assessed for that season. Therefore, the population of wandering cats is described as the seasonal estimate of wandering cats within the boundaries of the south Okanagan Valley. While the total study area was 161.2km^2, after removing grid cells with more than 50% water, the remaining area was 141.2km^2, which I used to calculate density of wandering cats.

I verified and compared my estimated population size to the estimated number of owned wandering cats based on surveys conducted in Canada and the total number of dwellings in the south Okanagan Valley. When combining the number of dwellings from Okanagan Falls, Oliver and Osoyoos, I get a minimum number of 7,002 and maximum number of 11,587 dwellings within the towns and surrounding areas that surpass my study area (Government of Canada, 2023). Across Canada, an average of 36.9% of households have at least one domestic cat, with an average of 1.64 cats per household and 28% of these residents allow their cats to roam freely outdoors (CFHS, 2017). Surveys conducted in British Columbia found that 36% of households had a domestic cat with an average of 1.69 cats per household, and 45% of cat owners allowed their cat unsupervised access to the outdoors (Stewardship Centre for British Columbia, 2019). I subtracted the estimated number of owned cats, that was calculated based on survey data, from the estimated population of wandering cats in the south Okanagan Valley (based on my models), to approximately estimate the number of unowned cats.

3.3 Results

I identified a total of 374 different cats across the year, with a range of 0 – 12 uniquely identified cats per site and an average of 3.5 cats per camera site, or 4.8 cats per camera site where I observed at least one cat. Wandering cats were detected by trail cameras at a total of 92 of 123 sites (74.8%). I detected the highest number of unique wandering cats during spring (N = 220), followed by fall (N = 186), summer (N = 172), early winter (N = 156), and late winter (N=153; Table 3.2). The percentage of sites where cats were detected was highest during the spring (61.7%), followed by the early winter (57.6%), fall (57.5%), summer (54.2%), and late winter (50%; Table 3.2). Wandering cats were detected at 100% of peri-urban sites, 97% of urban sites, 65% of agricultural sites and 42% of natural sites (Table 3.2). Based on local abundance estimates, wandering cats were most prevalent in urban and peri-urban habitats during each season, followed by agricultural and natural habitats (Table 3.2). Of the 374 wandering cats detected, 38 were observed to have traveled between two or more camera locations; 22 wandering cats were detected at two camera locations, 7 cats were detected at three sites, and 9 cats were observed at four sites. This was most prevalent at agricultural (N = 12) and peri-urban (N = 9) sites, compared to urban and natural (N = 3 sites each). Many of these cats travelled far distances, with 23 cats travelling more than 200 m between sites and 8 cats that travelled more than 400m, including two of which travelled 567 meters between two peri-urban sites.

Table 3.2. Seasonal summary statistics, including the number of sites where wandering cats were detected, organized by land class, and the total number of individually identified wandering cats identified using the trail camera during each season.

Land Class	**Spring**	**Summer**	**Fall**	**Early Winter**	**Late Winter**	**Full Year**
Urban	27/31 = 87.1%	27/31 = 87.1%	28/31 = 90.3%	26/31 = 83.9%	24/29 = 82.8%	31/32 = 97%
Peri-urban	19/26 = 73 %	19/26 = 73%	16/26 = 61.5%	21/25 = 84%	18/24 = 75%	27/27 = 100%
Agricultural	19/31 = 61.3%	13/31 = 41.9%	17/31 = 54.8%	17/30 = 56.6%	11/29 = 37.9%	20/31 = 65%
Natural	9/32 = 28.1%	6/32 = 18.75%	8/32 = 25%	4/32 = 12.5%	4/32 = 12.5%	14/33 = 42%
TOTAL	74/120 = 61.7%	65/120 = 54.2%	69/120 = 57.5%	68/118 = 57.6%	57/114 = 50%	92/123 = 74.8%
Number of Individually Identified Cats	220	172	186	156	153	374

Table 3.3. Top models for detection probability (*p*) when assessing the number of domestic cats photographed at trail camera sites during all five seasons in south Okanagan Valley, B.C. Global model included the co-variates: temp, date and precip (precipitation). Abundance estimates (λ) are held constant (λ (.)).

	Model	df	logLik	QAIC$_c$	ΔQAIC$_c$	weight
SPRING	*p* (date) λ (.)	4	-937.002	1882.4	0	0.194
	p (.) λ (.)	3	-938.191	1882.6	0.22	0.173
	p (temp+ date) λ (.)	5	-936.253	1883.1	0.70	0.136
	p (precip + date) λ (.)	5	-936.255	1883.1	0.70	0.136
	p (precip) λ (.)	4	-937.358	1883.1	0.71	0.136
	p (temp + date + precip) λ (.)	6	-935.516	1883.9	1.47	0.093
	p (temp) λ (.)	4	-937.958	1884.3	1.91	0.074
	p (temp + precip) λ (.)	5	-937.117	1884.8	2.43	0.057
SUMMER	*p* (.) λ (.)	3	-771.359	1549.0	0	0.345
	p (precip) λ (.)	4	-770.765	1550.0	0.99	0.210
	p (temp) λ (.)	4	-771.323	1551.1	2.11	0.120
	p (date) λ (.)	4	-771.327	1551.1	2.12	0.120
	p (date + precip) λ (.)	5	-770.740	1552.2	3.17	0.071
	p (temp + precip) λ (.)	5	-770.751	1552.2	3.20	0.070
	p (temp+ date) λ (.)	5	-771.301	1553.3	4.29	0.040
	p (temp + date + precip) λ (.)	6	-770.719	1554.4	5.41	0.023
FALL	*p* (temp + precip) λ (.)	5	-898.048	1806.7	0	0.356
	p (temp + date + precip) λ (.)	6	-897.698	1808.3	1.56	0.163
	p (date + precip) λ (.)	5	-898.846	1808.3	1.60	0.160
	p (temp) λ (.)	4	-900.150	1808.7	1.99	0.132
	p (date) λ (.)	4	-900.687	1809.8	3.07	0.077
	p (temp+ date) λ (.)	5	-899.892	1810.4	3.69	0.056
	p (precip) λ (.)	4	-901.697	1811.8	5.09	0.028
	p (.) λ (.)	3	-902.822	1811.9	5.17	0.027
EARLY WINTER	*p* (temp + date + precip) λ (.)	6	-737.016	1486.9	0	0.339
	p (date + precip) λ (.)	5	-738.546	1487.7	0.82	0.225
	p (temp + precip) λ (.)	5	-738.728	1488.0	1.18	0.188
	p (temp+ date) λ (.)	5	-739.065	1488.7	1.86	0.134
	p (temp) λ (.)	4	-740.906	1490.2	3.34	0.064
	p (date) λ (.)	4	-741.155	1490.7	3.84	0.050
	p (precip) λ (.)	4	-745.000	1498.4	11.53	0.001
	p (.) λ (.)	3	-748.977	1504.2	17.33	0.000
LATE WINTER	*p* (date + precip) λ (.)	5	-707.597	1425.8	0	0.394
	p (temp + date + precip) λ (.)	6	-706.591	1426.0	0.22	0.353
	p (date) λ (.)	4	-709.703	1427.8	2.02	0.144
	p (temp+ date) λ (.)	5	-708.993	1428.6	2.79	0.098
	p (temp + precip) λ (.)	5	-711.461	1433.5	7.73	0.008
	p (temp) λ (.)	4	-714.058	1436.5	10.73	0.002
	p (precip) λ (.)	4	-714.449	1437.3	11.51	0.001
	p (.) λ (.)	3	-716.518	1439.3	13.50	0.000

3.3.1 Detection probability

The global model for detection probability included average daily temperature (temp), Julian date (date), and total daily precipitation (precip; Table 3.1). The Variance Inflation Model (VIF) showed that none of the variables were collinear. During the spring, the top model contained one covariate, date (Table 3.3), with detection probabilities positively related to date. The mean detection probability during the spring was 0.2261 ± 0.0323 (lower CI: 0.1366, upper CI: 0.3594). During the summer, the top model contained no covariates (Table 3.3), and detection held constant at 0.1822 ± 0.0293 SE (lower CI: 0.1315, upper CI: 2468). During the fall, the top model contained temperature and precipitation, where detection was positively related to precipitation and negatively influenced by temperature. The mean detection probability was 0.237 ± 0.0312 (lower CI: 0.117, upper CI: 0.527). During the early winter, the top model contained all three co-variates with detection positively related to temperature but negatively influenced by date and precipitation. The mean detection probability during the early winter was 0.1045 ± 0.028 (lower CI: 0.0114, upper CI: 0.3611). During the late winter, date and precipitation were the top variables for estimating abundance, with date positively and precipitation negatively influencing detection, and a mean detection probability was 0.1753 ± 0.0316 (lower CI: 0.0433, upper CI: 0.3794).

3.3.2 Local Cat Abundance

The global model for local cat abundance during each season included the proportion of vineyards, orchards, water, fields, and natural habitat (desert, forest, and wetlands combined), as well as the number of buildings and the length of roads. None of the covariates showed multicollinearity. Natural habitat, such as wetlands, forests, and deserts, were included in the top models for each season indicating cats were the least abundant in natural areas compared to other habitats (Table 3.4). Vineyards were also included in four top models; cats were rarely found in the middle of vineyards, away from humans and buildings.

The habitat variables influencing local wandering cat abundance differed between seasons, with a more consistent set of predictors during spring, summer, and early winter (natural habitats and vineyards), and additional habitat variables becoming important in fall and late winter (fall: natural habitats, vineyards, water and roads; late winter: natural habitats, fields, and buildings). The mean estimated abundance of wandering cats per site during the spring was 2.21 ± 0.498 (lower CI: 0.006, upper CI: 8.21), increasing during the summer to 2.33 ± 0.606 (lower CI: 0.004, upper CI: 8.114), and increasing again during the early winter to 3.25 ± 1.084 (lower CI: 0.003, upper CI: 17.159). The mean estimated number of wandering cats per site during the fall was 1.87 ± 0.478 (lower CI: 0.022, upper CI: 10.992) and during the late winter it was similar 1.86 ± 0.476 (lower CI: 0.019, upper CI: 10.34; Figure 3.4b).

3.3.3 Seasonal and land class analyses

The median seasonal estimated abundances were very similar, with no significant differences between seasons except the early and late winter (Figure 3.4). Urban sites, characterized by a higher number of buildings within their buffer areas, consistently exhibited the highest estimated local abundance of wandering cats across all seasons (For all five seasons: Kruskall-Wallace, $P < 0.0001$; post hoc Dunn Test, urban versus all other land classes, $P < 0.01$; Figure 3.5), supporting the Human Presence and Cat Behaviour Hypothesis. Over the 12-month sampling period, urban habitats had the highest median estimated abundance of 4.1 wandering cats per site (Figure 3.4).

Table 3.4. Top models for abundance estimates (λ) when assessing the number of domestic cats photographed at trail camera sites during all five seasons in south Okanagan Valley, B.C. Global model included the co-variates: natural, vineyard, orchard, field, water, roads, and buildings. Detection probability (*p*) is held constant (*p*(.)).

	Model	df	logLik	QAIC$_c$	ΔQAIC$_c$	weight
SPRING	*p* (.) λ (natural + vineyard)	5	-912.063	1834.7	0	0.154
	p (.) λ (natural + vineyard + roads)	6	-911.685	1836.3	1.49	0.073
	p (.) λ (natural + vineyard + field)	6	-911.732	1836.4	1.58	0.070
	p (.) λ (natural + vineyard + orchard)	6	-911.738	1836.3	1.59	0.069
	p (.) λ (natural + vineyard + water)	6	-911.889	1836.6	1.89	0.060
SUMMER	*p* (.) λ (natural + vineyard)	5	-751.523	1513.7	0	0.105
	p (.) λ (natural + vineyard + buildings)	6	-750.768	1514.5	0.77	0.072
	p (.) λ (natural + vineyard + roads)	6	-750.826	1514.6	0.89	0.067
	p (.) λ (natural + vineyard + field)	6	-750.850	1514.6	0.94	0.066
	p (.) λ (natural + vineyard + orchard)	6	-751.189	1515.3	1.63	0.047
FALL	*p* (.) λ (natural + vineyard + roads + water)	7	-881.102	1777.4	0	0.124
	p (.) λ (natural + vineyard)	5	-883.700	1778.0	0.63	0.091
	p (.) λ (natural + vineyard + water)	6	-882.859	1778.6	1.20	0.068
	p (.) λ (natural + vineyard + roads)	6	-882.990	1778.9	1.47	0.060
	p (.) λ (natural + vineyard + orchard + roads + water)	8	-880.721	1779.0	1.59	0.056
EARLY WINTER	*p* (.) λ (natural + vineyard)	5	-726.030	1462.6	0	0.128
	p (.) λ (natural + vineyard + buildings)	6	-725.051	1462.9	0.28	0.111
	p (.) λ (natural + vineyard + field)	6	-725.296	1463.4	0.77	0.087
	p (.) λ (natural + vineyard + orchard)	6	-725.656	1464.1	1.49	0.061
	p (.) λ (natural + vineyard + orchard + field)	7	-724.666	1464.4	1.80	0.052
LATE WINTER	*p* (.) λ (natural + buildings + field)	6	-695.826	1404.4	0	0.129
	p (.) λ (natural + roads + buildings + field)	7	-695.348	1405.8	1.32	0.067
	p (.) λ (natural + vineyard + roads)	6	-696.709	1406.2	1.77	0.053
	p (.) λ (natural + buildings + water + field)	7	-695.632	1406.3	1.89	0.050
	p (.) λ (natural + orchard + buildings + field)	7	-695.698	1406.5	2.02	0.047

The median estimated local abundance of wandering cats across the year was significantly higher at peri-urban sites compared to agricultural sites, while agricultural sites had significantly higher median local abundance than natural sites (Figure 3.4). Peri-urban habitats generally supported more wandering cats than agricultural habitats, while natural habitats had lowest median estimated local abundance except during fall, where the local abundance was not significantly different from agricultural habitats (For all seasons except fall: Kruskall-Wallace, $P < 0.0001$; post hoc Dunn Test, natural versus all other land classes, $P < 0.01$; Figure 3.5). Across seasons there were no significant differences in median estimated local abundance between peri-urban and agricultural sites (Kruskall-Wallace, $P < 0.0001$; post hoc Dunn Test, peri-urban versus agricultural, $P > 0.05$; Figure 3.5).

3.3.4 Estimating wandering domestic cat population size

I estimated the population size of wandering cats for the spring, summer, and early winter, abundance using predicted values from top models that contained the natural and vineyard covariates. For fall, I used the natural, vineyard, water, and roads covariates, and for late winter I used natural, buildings, and field covariates (All covariates mapped in Figure 3.6). The estimated average population size of wandering cats was highest during the early winter (8,964; average density of 63.47 wandering cats/km^2; Table 3.5), followed by the spring (7,048; average density of 46.29 wandering cats/km^2; Table 3.5) and summer (7,031; average density of 49.78 wandering cats/km^2; Table 3.5). The population size was lowest during the fall (5,245; average density of 37.14 wandering cats/km^2; Table 3.5) and late winter (4,496; average density of 31.84 wandering cats/km^2; Table 3.5). On average, the population across the year and study area was 6,557 (range: 4,496 – 8,964) wandering cats, with an average density of 46.44 (range: 37.14 – 63.47) wandering cats/km^2. Seasonal predicted cat abundance based on natural habitat showed low cat abundance in natural areas (Figure 3.7), which was similar to the seasonal maps based on vineyards (Figure 3.8), where cats were not associated with vineyards. All other habitat covariate maps for the fall and late winter can be found in Appendix 16 (Figure A16.1 for the fall and Figure A16.2 for the late winter).

I used provincial and national cat owner surveys to estimate the number of owned and unowned wandering cats in the south Okanagan Valley. Based on the provincial surveys and the minimum number of dwellings, I estimated 1,917 owned wandering cats, with the number increasing to 3,172 wandering cats with the maximum number of dwellings. Using national owner surveys, the number of owned wandering cats was estimated to be 1,186 using the minimum number of dwellings, and 1,963 wandering cats with the maximum number of dwellings. Using the average population size of 6,557 wandering cats estimated in this study and subtracting the estimated owned cats from owner surveys, I estimated the number of unowned cats to range from 4,485 to 4,640 based on provincial surveys, and from 4,594 to 5,371 based on national surveys, indicating the 68-83% of wandering cats were unowned.

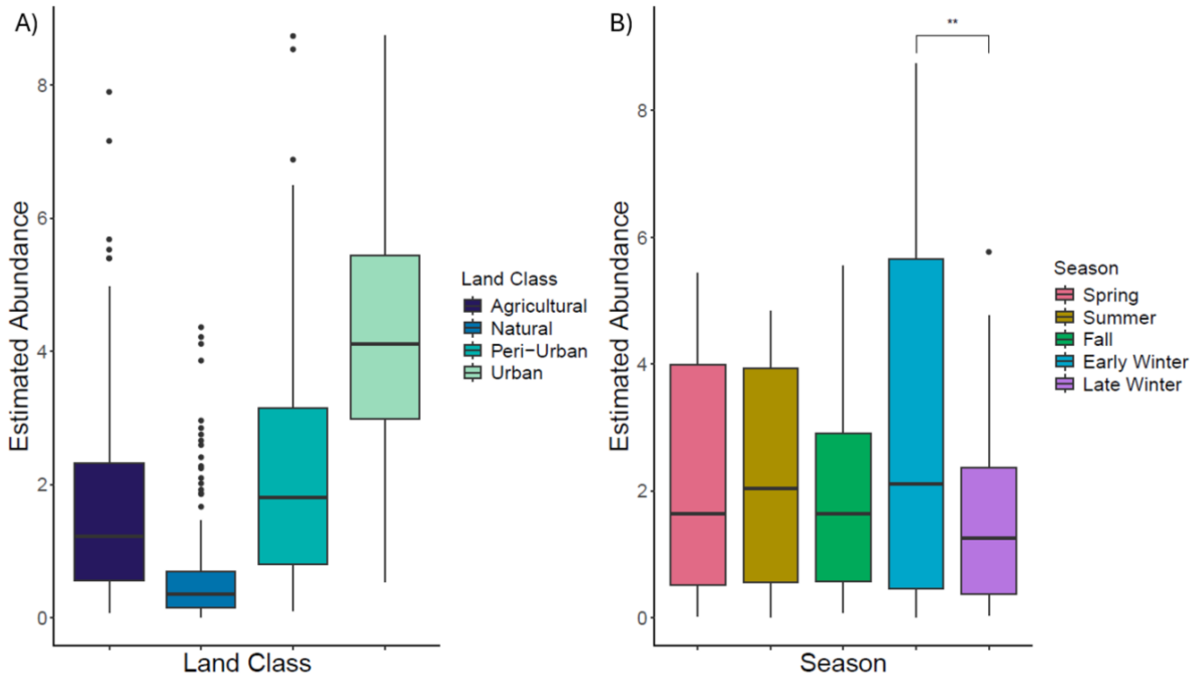

Figure 3.4. Box plots showing the estimated abundance A) in different land classes throughout the year and B) during each season. All land classes were significantly different ($p < 0.05$ for all seasonal comparisons). Median abundance of cats per site was highest in urban habitats (4.1), followed by peri-urban habitats (1.8), agricultural habitats (1.2), and natural habitats (0.4). For seasonal comparisons, all land classes were not significantly different ($p > 0.05$), except between early and late winter, which were significantly different ($p < 0.05$). The median abundance of domestic cats per site was highest in the early winter (2.1) and lowest in the late winter (1.3). The horizontal black line within the box represents the median value, and the box represents 50% of the data, from the 25th to 75th percentile of each groups' distribution of values, which is the interquartile range (IQR). The whiskers extend to the highest and lowest points within 1.5 times the IQR and individual dots are outliers beyond the whiskers.

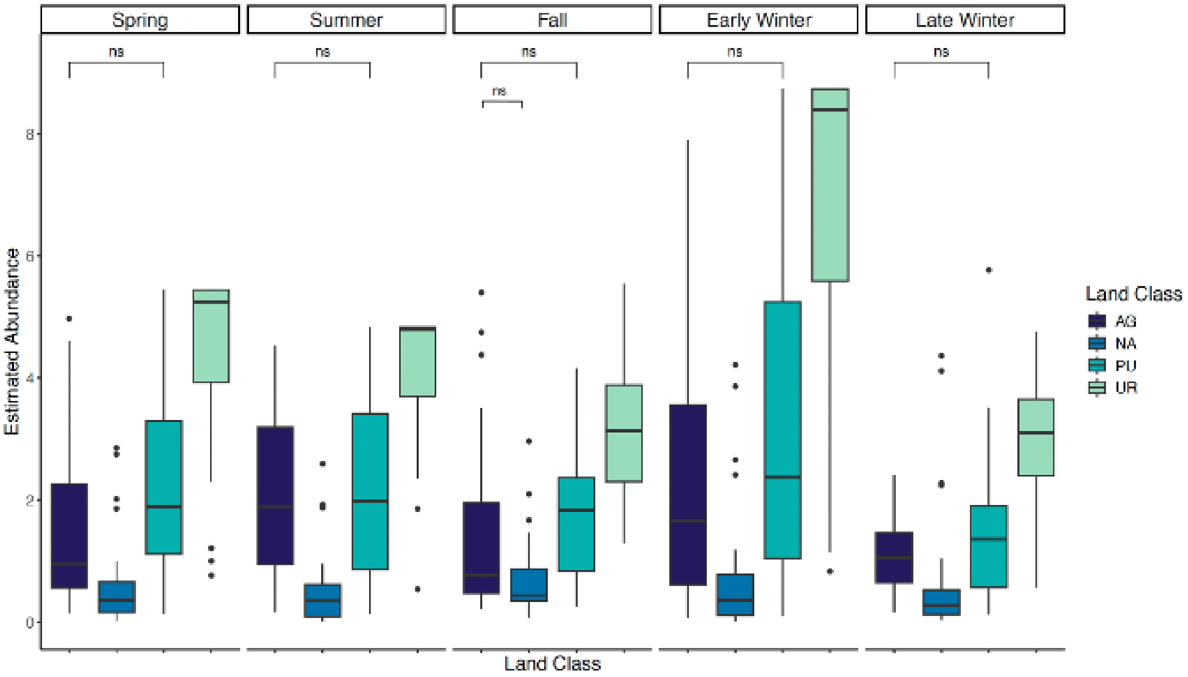

Figure 3.5. Box plots showing the estimated abundance for each land class separated by season. Mean abundance was significantly highest in urban habitats (UR) consistently across each season ($p < 0.05$). Mean abundance in peri-urban (PU) and agricultural (AG) sites are not significantly different in any season ($p > 0.05$). Mean abundance is significantly lowest in natural (NA) habitats during each season ($p < 0.05$) except during the fall, where agricultural and natural sites do not significantly differ ($p > 0.05$). The horizontal black line within the box represents the median value, and the box represents 50% of the data, from the 25th to 75th percentile of each groups' distribution of values, which is the interquartile range (IQR). The whiskers extend to the highest and lowest points within 1.5 times the IQR and individual dots are outliers beyond the whiskers.

Table 3.5. Variables for each season used for predicting cat abundance. Natural is wetlands, desert, and forests combined. Density was calculated using the study area of 141.2km^2.

Season	**Model Co-variate**	**Estimated Abundance**	**Density (cats per km^2)**
Spring	Natural	6537 (range: 4264 to 10,098)	46.29 (range 30.19 to 71.50)
	Vineyard	7559 (range: 4754 to 12,297)	53.52 (range 33.66 to 87.07)
Summer	Natural	6231 (range: 3901 to 10,069)	44.11 (range: 27.62 to 71.29)
	Vineyard	7832 (range: 4473 to 14,419)	55.45 (range: 31.67 to 102.09)
Fall	Natural	5331 (range: 3633 to 7941)	37.74 (range: 25.72 to 56.22)
	Vineyard	6423 (range: 4197 to 10,096)	45.49 (range: 29.72 to 71.50)
	Water	7328 (range: 4912 to 10,959)	51.91 (range: 34.79 to 77.64)
	Roads	1894 (range: 451 to 11,013)	13.41 (range: 3.19 to 77.97)
Early Winter	Natural	7914 (range: 4208 to 14,964)	56.03 (range: 29.79 to 105.95)
	Vineyard	10,015 (range: 4538 to 22,693)	70.91 (range: 32.13 to 160.67)
Late Winter	Natural	4744 (range: 2898 to 7854)	33.59 (range: 20.52 to 55.61)
	Buildings	2401 (range: 1467 to 3934)	17.00 (range: 10.38 to 27.85)
	Field	6345 (range: 3123 to 21,129)	44.93 (range: 22.11 to 149.60)

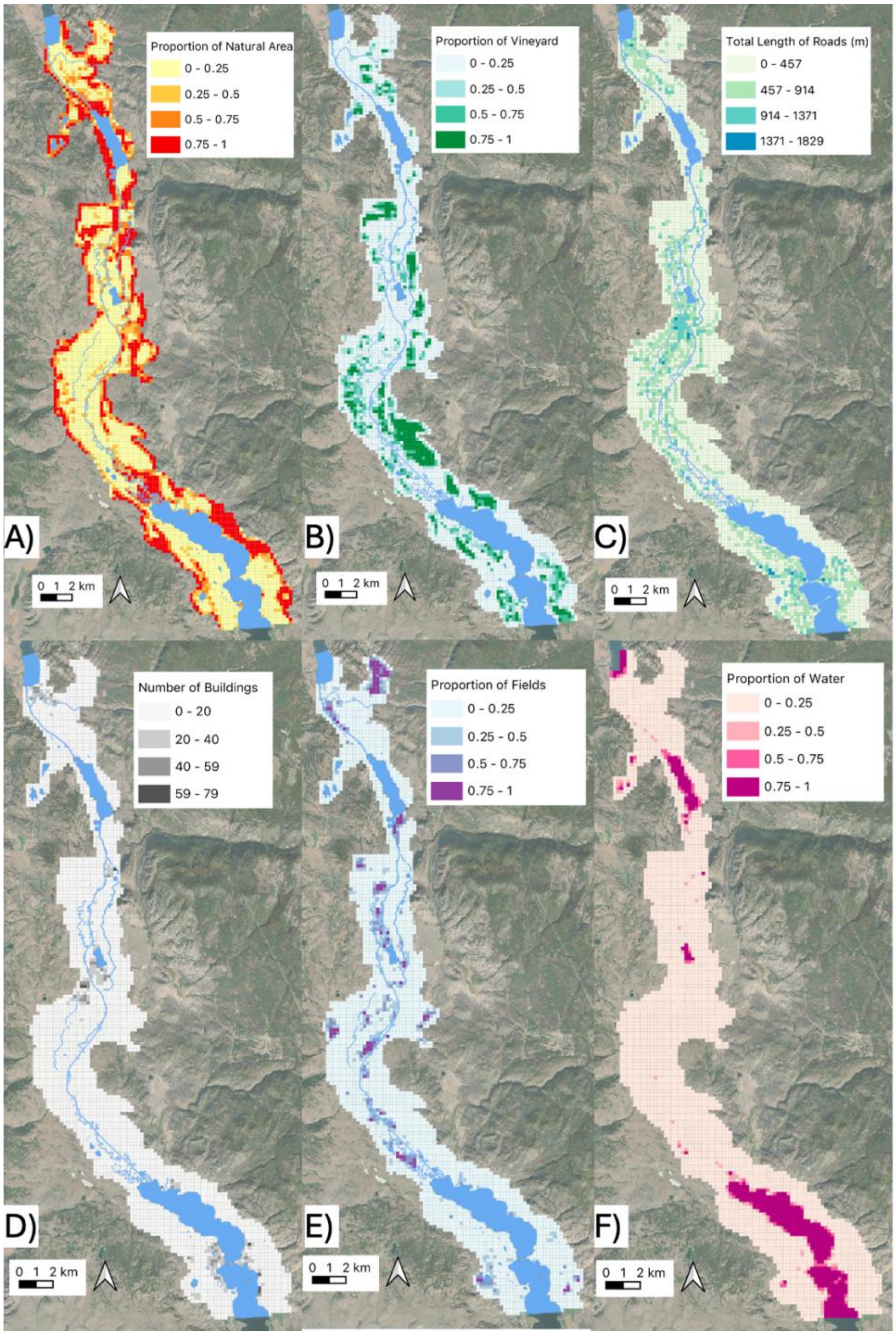

Figure 3.6. Habitat variables used in top models to estimate wandering cat populations in the south Okanagan Valley, B.C. Maps show each grid square and the proportion of natural habitats (A), vineyards (B), roads (C), buildings (D), fields (E) and water (F). Darker colours represent higher values, while lighter colours represent lower values.

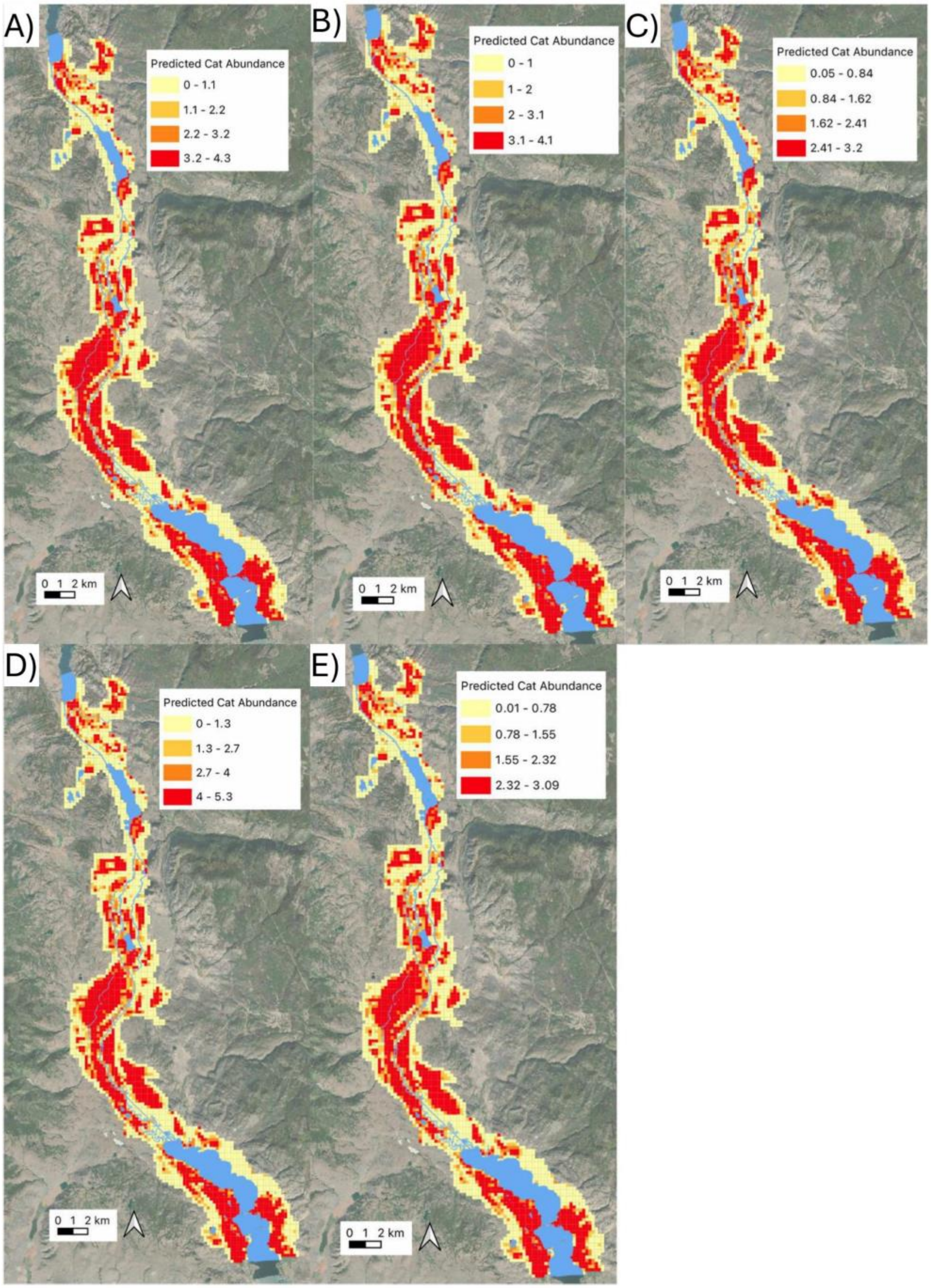

Figure 3.7. Predicted abundance of wandering cats based on the proportion of natural habitat in each cell during each season: A) spring, B) summer, C) Fall, D) Early winter, and E) Late winter. Darker areas have higher densities of wandering cats. While maps look the same, the local abundance per grid cell differs across seasons.

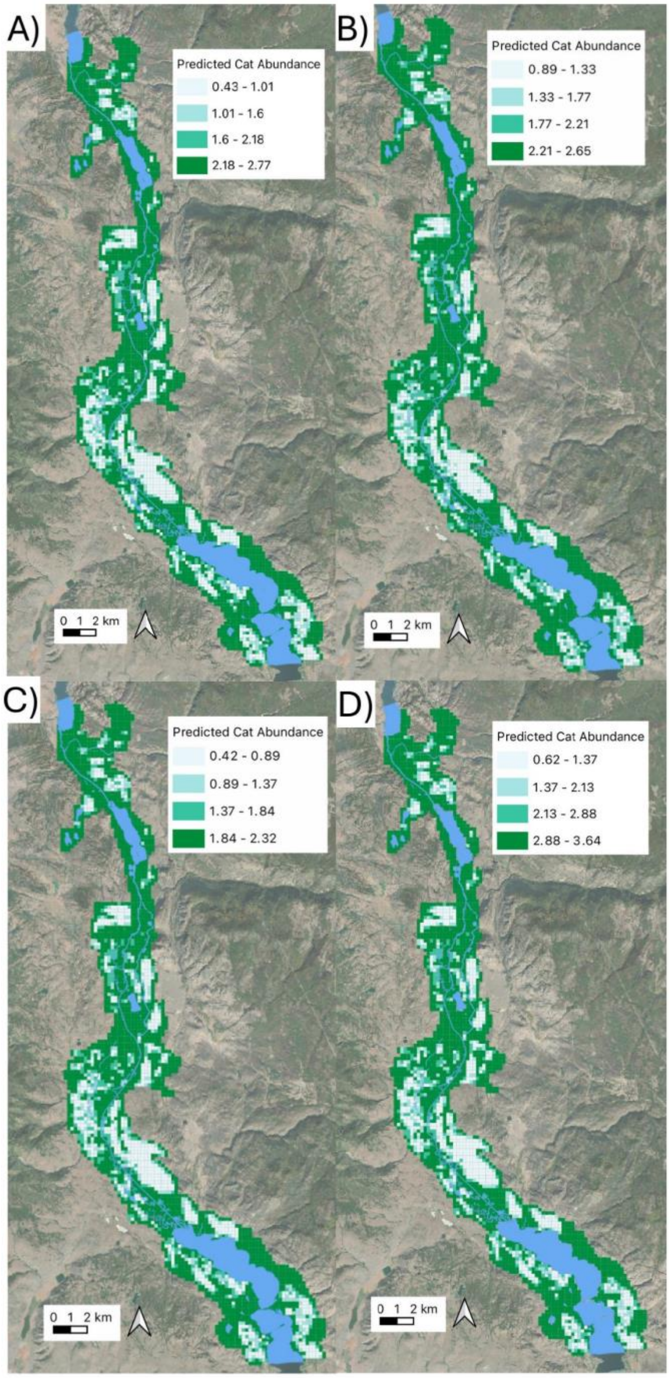

Figure 3.8. Predicted abundance of wandering cats based on the proportion of vineyards in each cell during four seasons: A) spring, B) summer, C) Fall, and D) Early winter. Darker areas have higher densities of wandering cats.

3.4 Discussion

In this multi-season and -habitat focused study, I demonstrated that wandering cats are most abundant in urban habitats but also occur widely, and often in high local abundances, across land classes. This suggests that regions like the south Okanagan, which feature a mosaic of urban, agricultural, peri-urban and natural areas, harbour a proportionately large number of cats in relation to the area and human population. Based on my estimated cat population size, the south Okanagan Valley had at least 1 cat for every 3 people, which represents the highest per capita density of wandering cats recorded in a North American based study thus far (Cove et al., 2023; Flockhart et al., 2016; Gow et al., 2024). With an average predicted density of 46.44 cats/km^2 across all habitat types, the density in the south Okanagan Valley is higher than the only other multi-habitat assessment (i.e., Australia, 0.13-0.73 cats/km^2; Legge et al., 2017), and at least one urban area in North America (41.45 cats/km^2, 1 cat for every 93 people, Washington, DC, USA, Cove et al., 2023). However, the density is lower than two Canadian cities but higher based on wandering cats/person; Guelph, ON (88.36 cats/km^2, one cat for every 16 people, Flockhart et al., 2016) and Gatineau, PQ (62.24 cats/km^2, one cat for every 14 people, Gow et al., 2024). Cat population estimates were highest during the early winter, spring, and summer, with fewer cats in fall and late winter. Population estimates of wandering cats using trail cameras were nearly double those extrapolated from owner surveys, meaning that the vast majority of wandering cats are likely unowned or households were more likely to put their cats outdoors than national or provisional averages.

While my estimates provide valuable insights into the seasonal abundance of wandering cats, there is inherent uncertainty in my results. My abundance models account for factors influencing both cat abundance and detection, but other unmeasured variables may contribute to fluctuations that I cannot fully capture. These limitations may lead to under- or overestimation of wandering cat abundance across seasons. These uncertainties apply to both owned and unowned wandering cat estimates; however, my results remain valuable in identifying the seasons and habitats where wandering cat abundance may be highest and have the greatest impact on wildlife. Despite limitations, my findings establish an important foundation for understanding the seasonal patterns in wandering cat abundance and highlight the need for further research to refine estimates and improve confidence in population assessments. Wandering cats were most abundant in urban areas, while natural habitats supported the lowest, but not zero, abundance, reflecting the strong influence of human presence and habitat type on their local abundance, and supporting the Human Presence and Cat Behaviour Hypothesis. This association between cats and urban areas aligns with other studies in Ontario (Clyde et al., 2022) and Washington, DC (Cove et al., 2023), further highlighting the influence of human presence in shaping wandering cat abundances across different landscapes. While other studies have focused on binary classifications of habitat types, such as rural vs urban (Kauhala et al., 2015), this approach often overlooks the nuanced land-use patterns in wandering cat abundance. In peri-urban and agricultural habitats, human influence also shaped the distribution of wandering cats, with lower wandering cat abundances likely reflecting the reduced number of buildings and residents in these areas. As predicted, natural habitats supported the lowest abundance of wandering cats, reflecting the low levels of urbanization, greater distance from human settlements, and limited human influence. However, the high overall abundance of wandering cats in the south Okanagan Valley may be attributed to the unique integration of peri-urban and urban areas, where urbanization intensity differs from that of a larger metropolitan center. The distinct landscape, combined with the region's temperate climate and abundant resources, may create more favourable conditions for wandering cats. However, research on wandering cats in small urban centers is limited, making it difficult to determine the mechanisms driving these patterns.

While there were fewer wandering cats in natural habitats, they were still present and can pose a threat to local wildlife, particularly birds, by exerting predation pressure, especially if unowned or partially owned (e.g., barn cats). A study tracking cat movement across four countries (USA, Australia, UK, and New Zealand) found that 11% of tracked cats spent most of their outdoor time in natural habitats (Kays et al., 2020), while another study in Guelph, ON, Canada, showed that 4% of GPS locations occurred in natural habitats or greenspaces (Pyott et al., 2024). While I did not use tracking collars, these findings may explain the use of natural habitats by wandering cats observed in my study. Cats that wander into natural areas from nearby houses raise concerns for local wildlife, as studies in Great Britain found that cats living adjacent to natural habitats killed more prey than inner suburban cats, with an average of 7.91 prey/cat/year, 25% of which were birds (Pirie et al., 2022). While predation pressure on birds and other animals may be highest in urban areas due to greater cat densities (Legge et al., 2020), the impact of a single cat in natural habitats is concerning, particularly due to the higher numbers of species at risk, including numerous bird species found in the south Okanagan Valley (Chapter 2). Further research that tracks cat movement and estimates predation rates of owned and unowned cats across different habitat types in this region would provide more precise insights into the magnitude of their impact on bird (and other animal) populations in this region.

Understanding how seasonal weather patterns influence the activity level and detection of wandering cats is critical for accurately assessing changes in local abundances and population sizes throughout the year. During fall and winter, the detection of wandering cats was largely influenced by changes in weather, such as temperature and rain. During fall, I saw a decrease in the abundance and detection of wandering cats, which may have been influenced by the dry, hot weather. Similar dry hot weather has been associated with reduced wandering cat density in Australia due to lower prey availability, with this pattern driven predominantly across the dry regions of the country (Legge et al., 2017). Seasonal patterns of feral cat abundance in Australia may be driven by changes in prey availability (i.e., more prey during wet periods), thus it is possible that the seasonal variability that I observed in local wandering cat abundance and population estimates may similarly be explained by prey availability (i.e., Prey Availability Hypothesis), especially if the high estimated proportion of unowned cats in my study area relies heavily on hunting animals to survive (see Chapter 2 for details about birds in the region). In addition, weather conditions can influence human behaviour, such as reducing the likelihood of residents allowing their cats outdoors during hot or cold temperatures (or rain or snow), thus lowering the abundance of wandering cats (Forrest et al., 2023; Goszczyński et al., 2009). My study region is a popular tourist destination, and the departure of vacationers in fall and the potential decrease in activity of unowned cats due to hot, dry conditions could have further contributed to the observed decline in wandering cat abundance estimates during the late winter, showing some support for the Human Behaviour Hypothesis. While I was unable to examine the mechanism explaining the seasonal abundance differences directly, it is possible that, given the large estimated number and local abundance of unowned cats, ecological factors (e.g., prey availability) and human factors (e.g., supplemental feeding, lack of spay-neuter programs, pet dumping, etc.) may be more likely to drive wandering cat populations in this region, in contrast to potential dynamics in large urban centres that may be influenced by a dynamic between shelters, owned cats, and unowned cats (Flockhart et al., 2024; Flockhart & Coe, 2018).

Seasonal weather patterns, particularly snowfall, may influence the activity levels of wandering cats, leading to the highest local population and local abundance estimates during the early winter and decreasing during the late winter. Early winter local abundance and population estimates were higher than those of all other seasons, possibly due to increased cat activity levels as they searched for food before snow accumulation, and residents potentially granting their owned cats more outdoor access as temperatures cooled and precipitation increased following the fall. Additionally, during early winter, many birds moved into urban areas where food is more readily available (Chapter 2). As cats are primarily found in urban areas, there may be greater potential for hunting birds during this period of overlap between high wandering cat abundance and high bird abundance, thereby increasing cat activity levels and detection. Increased activity of wandering cats during the winter also aligns with findings from GPS-tracked cats in Belfast, Ireland, which showed larger home ranges in winter compared to summer (Dunford et al., 2024). While cats may have been less active when snow accumulated, many were still outdoors, as evidenced by numerous observations of cats walking through snow. Overall, early winter may provide the most realistic estimate of the wandering cat population, as owned cats are likely outdoors, and unowned cats are more detectable as they may be actively searching for food. During late winter, I observed a decrease in the wandering cat abundance, potentially due to cats not surviving the winter and higher rainfall, which has been shown to decrease cat activity level (Goszczyński et al., 2009). During this period, I also saw a decrease in the number of prey species in the south Okanagan Valley (Chapter 2), which may have contributed to decreased activity levels, such as reduced exploratory and hunting behaviours. My findings suggest that wandering cats have the potential to pose a significant year-round threat to birds and other small animals, relative to seasonal prey availability (e.g. bats, insects, reptiles, amphibians, mammals etc.).

3.4.1 Management implications

Unfortunately, the high number of unowned wandering cats in the study region creates wandering cat population management challenges and thus subsequent economic investments for municipalities and other governments if they chose to take action. Without significant multi-stakeholder actions and funding, welfare concerns (e.g., cats being hit by cars, predation of cats, disease transmission, etc.), risks cats pose to wildlife, and subsequent calls for initiatives to manage cat populations are likely to escalate and magnify. For example, media reports indicate that the Okanagan Human Society has seen a substantial increase in low-income spay neuter requests, which are estimated to cost up to $50,000 per month (Black Press Media, 2024). This places significant strain on shelters and puts an economic burden on local communities tasked with managing these unowned cats. While this recognizes the growing public interest in controlling wandering cat populations, the associated costs often prohibit effective action. Additionally, the economic burden of cat-dependent diseases may extend beyond the local level, with human healthcare costs in Australia potentially reaching billions of dollars due to medical costs and lost productivity (Legge et al., 2020). Effectively managing the high number of unowned wandering cats and their ecological, welfare, and economic impacts requires a unified approach with considerable collaboration and education across multiple sectors, including local municipalities, animal welfare organizations, and the public.

In conclusion, my study highlights the widespread presence of wandering cats across diverse land classes, raising significant concerns for both cat welfare and wildlife conservation locally and more broadly. Here, I showed that cats were not confined to urban areas, and their activity across the landscape would require landscape-wide management strategies to mitigate their potential negative impacts on wildlife. By quantifying seasonal variation in abundance and habitat use, I identified periods of increased cat activity and potential greater risk to wildlife, such as birds that are also heavily driven by seasonal patterns. By also identifying seasonal drivers of wandering cat detections, such as temperature and precipitation, I was able to highlight periods of increased wandering cat abundance, which could help in the development of strategies that account for these seasonal dynamics. Effective solutions require a multifaceted approach that balances the welfare of cats with the protection of vulnerable wildlife populations. Such approaches could involve a combination of educational campaigns to reduce outdoor access for owned cats and the implementation of new regulations by local governments, such as licensing or requiring spay-neutering (Cecchetti et al., 2021). Collaborative efforts among municipalities, stakeholders, and residents will be critical to mitigate the ecological impact of wandering cats and help preserve the biodiversity of this unique region. Given the ecological sensitivity of the area, public outreach and strong enforcement measures are needed, which ultimately requires municipal government to legislate and enforce by-laws that curtail wandering cats and address the environmental and economic impacts.

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Chapter 4: Conclusions, Future Directions, and Management and Conservation Implications

4.1 Discussion and Future Directions

The high numbers of wandering cats in the south Okanagan Valley throughout the year, combined with the diversity of birds and other wildlife, suggests that cats could have significant impacts on birds and other wildlife year-round. Wandering cats pose a significant threat to bird populations globally, contributing to the extinction of 22 species on islands (Medina et al., 2011), and killing billions (if not trillions) of birds worldwide each year (Blancher, 2013; Krauze-Gryz et al., 2019; Loss et al., 2013; Sedano-Cruz, 2022; Woinarski et al., 2017; Woods et al., 2003). Cats may also affect mainland bird populations (Loss et al., 2013). The impacts of wandering cats on wildlife are not evenly distributed, creating distinct local ecological risks. The south Okanagan Valley acts as a funnel for migratory birds in central B.C., which was evident from my findings (Chapter 2), where I identified 146 bird species during point counts, though over 330 bird species inhabit this area across the year, including 24 provincially listed species at risk and 22 federally listed species at risk under the Species at Risk Act (eBird, 2022; Okanagan Similkameen Stewardship [OSS], 2023). Many of these species at risk are found in natural habitats or border peri-urban or agricultural habitats (e.g., Yellow-breasted Chat [*Icteria virens auricollis*], Lewis's Woodpecker [*Melanerpes lewis*], and Bobolink [*Dolichonyx oryzivorus*]), but are still at risk of predation by cats that wander into natural areas. There are other habitats and seasons where the abundance patterns of cats and birds overlap considerably, such as during the non-breeding seasons when there is a high abundance and species richness of birds in urban areas along with extremely high local abundances of wandering cats. I demonstrated the importance of natural areas, especially during breeding for species of birds that are uncommon or were not identified in other habitats, including numerous species at risk that may face predation by domestic cats, such as Yellow-breasted Chat (*Icteria virens auricollis*), Bank Swallow (*Riparia riparia*), and Bobolink (*Dolichonyx oryzivorus*). The fact that cats were observed in these natural areas is concerning, especially given the potential impact, either directly or indirectly, of a single cat on birds.

Although about half of owned cats kill wild animals (Loyd et al., 2013), up to 82% percent of wandering cats in my study area are potentially unowned, with potentially up to 91% preying on wildlife (McGregor et al., 2015), collectively intensifying predation pressure on migratory and non-migratory birds in the study region, as well as other animals including native mammals, amphibians, reptiles and insects. This is exacerbated by the disproportionately high abundance of wandering cats in urban and peri-urban areas in relation to people, their presence in natural areas, and the presumably larger home ranges of rural cats (Hanmer et al., 2017; Pirie et al., 2022). Although the local abundance estimates in natural habitats were low, which may have also been attributed to the presence of predators, cats were still detected at 42% of natural sites during the year. At six natural sites, cats appeared more than 100 m from the nearest building, including one observation in a forest that was 217 m from the closest structure. With this extensive movement into natural areas, birds are at greater risk of encountering a wandering cat, and therefore risk of predation. Blancher (2013) estimated the number of birds killed by cats every year in Canada using an average predation rate of 2.8 birds killed per owned cat per year and 24 birds per unowned cat per year. Although the exact predation rates in the south Okanagan Valley are currently unknown, extrapolating these rates suggests that upwards of 100,000 birds are killed by wandering cats in this region, and this would likely include many species at risk. The potential impact on native small mammal species (including many endangered bat species) is also exceptionally high. If using predation rate estimates based on those used in Loss et al. (2013), ranging from 177.3 to 299.5 small mammals killed per cat annually, I estimate at least 1 million small mammals are killed every year by domestic cats within the study region. While these estimates are based on average and not median predation rates from studies in other regions of the world, researching predation rates in the south Okanagan Valley would provide a much-needed next step to better estimate the true impact that wandering cats have on bird populations and other small animals in this region. Such research, when combined with the cat population and abundance estimates (Chapter 3) along with bird abundance estimates (Chapter 2), would provide important context to understand what the true impacts of wandering cats are on birds and other animals that could be applied to other ecosystems.

4.2 Management and Conservation Implications

Despite the high per capita density of wandering cats in my study, many residents and the general public overall are unaware of their prevalence due to the crepuscular and elusive behaviour of cats, which makes them rarely seen (Merčnik et al., 2023). Numerous residents were surprised to learn, through the trail camera images, that cats regularly frequented their yards. This lack of awareness may be attributed to differing perspective on wandering cats, as non-cat owners tend to express greater concern regarding their presence than cat owners (Booth & Otter, 2024). The public's unawareness and division regarding cat regulations complicates efforts to build support for effective management strategies, such as enforceable no-roam bylaws (Trouwborst et al., 2020). While their presence often goes unnoticed, wandering cats have ecological impacts comparable to or exceeding those of wild carnivores, with their high densities amplifying predation pressure on local wildlife (Kays et al., 2020). Allowing cats to wander also imposes several welfare risks to the cats themselves, including predation from other animals (e.g. coyotes, cougars, or bears), vehicle collisions, and disease transmission (Tan et al., 2020). Cats are well known to acquire or transmit numerous zoonoses through interactions with one cat to another (e.g., feline leukemia virus and rabies; Fehlner-Gardiner et al., 2024; Powers et al., 2018), from wildlife (e.g., avian influenza and toxoplasmosis; Burrough et al., 2024; Rahimi et al., 2015), and even transmit diseases to other wildlife and humans (e.g., toxoplasmosis; Hollings et al., 2013; Rahimi et al., 2015). While stakeholder groups such as Trap-Neuter-Release (TNR) groups, rescues, shelters, and veterinarians (Canadian Veterinary Medical Association [CVMA], 2020) may be aware of the challenges posed by wandering cats, municipalities – who have the power to implement change – often underestimate these challenges or chose to ignore them (Canadian Federation of Humane Societies [CFHS], 2017). This was evidenced by a 2017 survey in which the majority of municipalities stated that cat overpopulation was not a concern (CFHS, 2017).

Removing all or most wandering cats from the landscape in the study region and elsewhere is likely impossible, but management strategies may help mitigate their environmental impact, benefiting wildlife, people, and the cats themselves. One proposed approach for curtailing the number of owned cats in natural habitats involves buffer zones around natural areas. This approach focuses on protecting birds and keeping cats out of natural areas, which we've identified as important habitat for birds during spring migration, breeding, and fall migration. Residents within the buffer would be prohibited from allowing their cats outdoors and restricted from providing resources to unowned cats (e.g. can't feed or house barn cats that can leave the barn; Dunford et al., 2024; Pyott et al., 2024; R. L. Thomas et al., 2014), and cats could be actively trapped in these buffers. In my study, only two cats travelled more than 500 m between trail camera sites, suggesting that buffers of 750 meters (Dunford et al., 2024) or 840 m (Pyott et al., 2024) may effectively protect wildlife in natural habitats by restricting cat access and thereby reducing the risk to wildlife. This would be critical for sensitive habitat in the south Okanagan Valley such as areas neighbouring Important Bird Areas (IBA) and Migratory Bird Sanctuaries. For example, Vaseux Lake in Okanagan Falls is a Migratory Bird Sanctuary, and the surrounding area is part of an IBA, yet 11 of 19 sites in this area detected at least one cat with a total of 14 wandering cats.

Other options to curtail the number of wandering cats includes by-law enforcement, public outreach, and targeted funding, all of which would effectively reduce predation pressure on birds throughout the south Okanagan Valley (also see Cecchetti et al., 2021). For example, by-laws prohibiting owners from allowing their cats to wander outdoors would effectively reduce the number of owned wandering cats. This has been introduced in Osoyoos, but is not enforced, as one third (30.7%; N = 115) of the wandering cats I identified were wandering in Osoyoos. Enforcing such by-laws requires resources, which could be supported through licensing fees and penalties for wandering cats. Public education is also crucial to raise awareness of the risks posed to wandering cats and threats cats pose to wildlife (CFHS, 2017; Proulx, 1988). Outreach is often the first step in informing the public and municipal governments about the environmental impacts and economic implication of wandering cats, an important step in the south Okanagan Valley. Funding initiatives, such as those spearheaded by the Stewardship Centre for British Columbia, are key to addressing the high number of wandering cats in the south Okanagan Valley. In the south Okanagan Valley, outreach initiatives should focus on peri-urban and agricultural habitats, where high cat numbers overlap with high bird abundance and richness during the spring, summer, and fall. During winter, outreach should target urban habitats to encourage residents to keep their cats indoors, as many species move into urban areas at this time. Through collaboration, outreach, and targeted funding, these efforts aim to reduce the impact of wandering cats on this ecologically diverse and sensitive region.

4.3 Conclusion

In conclusion, the high population of owned and unowned wandering cats in the south Okanagan Valley places significant predation pressure on birds and other wildlife, yet controlling their numbers remains a controversial topic that requires input and collaboration from various stakeholders. Targeted management strategies, such as establishing buffer zones around natural habitats, strengthening by-law enforcement, enhancing public outreach, and fostering community-based initiatives, are essential to mitigate the risks that wandering cats pose to wildlife, themselves, and people. Using these data, I have established baselines that can be used if repeated in future years to compare the success of cat management strategies. I have also provided key seasons where birds and other wildlife may be most vulnerable to predation. This provides a foundation for outreach efforts in the south Okanagan Valley, with the goal of curtailing the number of wandering domestic cats, and in the process, reducing the risk to birds and other wildlife.

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Appendix 1

Addition, movement, or loss of trail camera sites.

Trail camera sites were moved, discarded, and added throughout the year. Trail camera sites were added when homeowners reached out after the completion of the first season (N = 1). Trail camera sites were lost due to the following reasons: homeowners no longer agreed to participate in the study after one season (N = 1), the homeowners moved and I therefore lost access to the site (N = 2), the homeowners didn't reply when attempting to expand the project for a 5th season (N = 2), or the trail camera was accidentally destroyed during the last season, along with the fence it was attached to (N = 1). No theft occurred during this study and any other loss of data was a result of technical errors. Trail camera sites were moved when the vegetation grew too tall but couldn't be removed and the camera couldn't be repositioned in the same place (N = 1), therefore I found the next closest spot to place the camera.

Appendix 2

Additional Study Design information regarding site selection.

Sites in natural habitats were obtained by collaborating with the Ministry of Forests, B.C. Parks, and Nature Trust of British Columbia to acquire permits. In agricultural areas, I obtained permission by emailing vineyards and speaking with vineyard owners in-person. For urban and peri-urban sites, I collaborated with local organizations to help advertise for volunteers and went door-to-door to seek landowner participation in areas with low numbers of volunteers. Local organizations (e.g., Castanet and Penticton Naturalist Club) helped advertise the project, reach out to homeowners, and share contact information with those interested in participating.

Appendix 3

Distance Sampling equation and assumptions.

Distance sampling broadly belongs to the category of multinomial outcomes and estimates abundance using the following equation:

$$y_{it} \sim Multinomial\ (N_{it}\ , \pi_{it})$$

Where y_{it} represents counts per plot i on occasion t, N_{it} denotes the total number of individuals available for detection in plot i on occasion t, and π_{it} represents multinomial probabilities derived from detection probabilities, p (Chandler et al., 2011).

There are two other main assumptions for the gdistsamp model; first, that objects at the point are detected with certainty and second, measurements of these detections are exact (Thomas et al., 2010). The model addresses the first assumption by assigning higher detection probabilities to birds closer to the observer, where they naturally have a greater likelihood of being detected. To manage distance-dependent detection probability, my surveys categorized observations into distance bins: 0-50 meters and 50-100 meters. This categorization allowed me to estimate observation distances more accurately, thereby addressing the second assumption of exact measurements.

Appendix 4

Bray-Curtis distance metric rationale.

I compared multiple distance metrics (Bray-Curtis, Canberra, Euclidean, and Gower) and multiple axes (k = 1 to k = 4) for each season. The model with the lowest acceptable stress (below 0.2) was chosen for each season and a stress plot was generated to assess the adequacy of the model. Despite the Euclidean distance metric yielding the lowest stress scores, the ordination plots produced did not accurately represent these data because Euclidean distance does not take species identity into account (Legendre & Legendre, 2012).

Appendix 5

NMDS Stress Values.

Spring migration = 0.18, breeding season = 0.18, fall migration = 0.18, early non-breeding season = 0.17, and late non-breeding season = 0.16.

Appendix 6

Table A6.1. All species observed during point counts during the spring migration (SM; N = 480 point counts), breeding season (BS; N = 480 point counts), fall migration (FM; N = 480 point counts), early non-breeding season (EN; N = 476 point counts), and late non-breeding season (LN; N = 464 point counts). Birds that are at higher risk of predation by a domestic cat are denoted with an asterisk. The total number of individuals for each species and season are calculated, as well as the total number of species seen during each season. Common name, scientific name, and four-letter banding codes are provided. Only birds seen within 100 meters from the observer are provided.

Common Name	Scientific Name	4-letter banding code	SM	BS	FM	EN	LN	Total
Alder flycatcher	*Empidonax alnorum*	ALFL	0	2	0	0	0	2
American coot	*Fulica americana*	AMCO	15	6	44	6	0	71
American crow	*Corvus brachyrhynchos*	AMCR*	118	114	70	36	36	374
American dipper	*Cinclus mexicanus*	AMDI	0	0	0	4	0	4
American goldfinch	*Spinus tristis*	AMGO*	155	215	276	124	120	890
American kestrel	*Falco sparverius*	AMKE	2	3	4	1	0	10
American pipit	*Anthus rubescens*	AMPI	0	0	4	1	0	5
American robin	*Turdus migratorius*	AMRO*	848	511	335	1146	373	3213
American wigeon	*Mareca americana*	AMWI	6	0	0	0	2	8
American tree sparrow	*Spizelloides arborea*	ATSP*	2	0	0	3	1	6
Bald eagle	*Haliaeetus leucocephalus*	BAEA	3	0	2	5	12	22
Barrow’s goldeneye	*Bucephala islandica*	BAGO	0	0	0	9	0	9
Bank swallow	*Riparia riparia*	BANS	0	33	0	0	0	33
Barn swallow	*Hirundo rustica*	BARS*	5	45	66	0	0	116
Black-billed magpie	*Pica hudsonia*	BBMA	19	38	10	27	16	110
Black-capped chickadee	*Poecile atricapillus*	BCCH*	105	118	199	130	170	722
Black-chinned hummingbird	*Archilochus alexandri*	BCHU	0	1	0	0	0	1
Belted kingfisher	*Megaceryle alcyon*	BEKI	0	1	21	9	0	31
Bewick’s wren	*Thryomanes bewickii*	BEWR*	8	4	12	17	15	56
Brown-headed cowbird	*Molothrus ater*	BHCO*	20	58	3	0	0	81
Black-headed grosbeak	*Pheucticus melanocephalus*	BHGR	0	15	6	0	0	21
Bobolink	*Dolichonyx oryzivorus*	BOBO*	0	1	0	0	0	1
Bohemian waxwing	*Bombycilla garrulus*	BOWA	12	0	0	281	309	602
Brewer’s blackbird	*Euphagus cyanocephalus*	BRBL*	224	322	425	0	0	971
Bufflehead	*Bucephala albeola*	BUFF	31	0	0	39	116	186
Bullock’s oriole	*Icterus bullockii*	BUOR	5	30	0	0	0	35
Cassin’s finch	*Haemorhous cassinii*	CAFI	3	7	5	0	1	16
Calliope hummingbird	*Selasphorus calliope*	CAHU	0	6	0	0	0	6
Canada goose	*Branta canadensis*	CANG	19	118	88	81	126	432
Canyon wren	*Catherpes mexicanus*	CANW*	18	30	33	14	25	120
California quail	*Callipepla californica*	CAQU*	1358	914	1004	803	921	5000
California scrub jay	*Aphelocoma californica*	CASJ	0	0	0	1	0	1

Cedar waxwing	*Bombycilla cedrorum*	CEDW*	1	586	525	216	48	1376
Chipping sparrow	*Spizella passerina*	CHSP	42	84	31	0	0	157
Chukar	*Alectoris chukar*	CHUK*	0	0	0	7	0	7
Clark's nutcracker	*Nucifraga columbiana*	CLNU	0	7	6	0	0	13
Cliff swallow	*Petrochelidon pyrrhonota*	CLSW	0	5	0	0	0	5
Common goldeneye	*Bucephala clangula*	COGO	2	2	0	115	58	177
Cooper's hawk	*Accipiter cooperii*	COHA	1	0	1	4	5	11
Common merganser	*Mergus merganser*	COME	10	0	0	32	6	48
Common raven	*Corvus corax*	CORA	3	0	13	30	8	54
Common yellowthroat	*Geothlypis trichas*	COYE	3	41	2	0	0	46
Dark-eyed junco	*Junco hyemalis*	DEJU*	135	0	15	1028	726	1904
Downy woodpecker	*Picoides pubescens*	DOWO*	18	18	28	8	16	88
Dusky flycatcher	*Empidonax oberholseri*	DUFL	14	0	0	0	0	14
Eastern kingbird	*Tyrannus tyrannus*	EAKI	0	30	70	0	0	100
Eurasian collared-dove	*Streptopelia decaocto*	EUCD*	425	408	377	202	273	1685
European starling	*Sturnus vulgaris*	EUST*	1034	679	2492	5450	777	10432
Evening grosbeak	*Coccothraustes vespertinus*	EVGR*	2	2	16	0	0	20
Fox sparrow	*Passerella iliaca*	FOSP	0	0	0	2	0	2
Gadwall	*Mareca strepera*	GADW	3	0	0	0	0	3
Great blue heron	*Ardea herodias*	GBHE	5	3	2	2	1	13
Golden-crowned kinglet	*Regulus satrapa*	GCKI	0	0	0	1	0	1
Great-horned owl	*Bubo virginianus*	GHOW	9	0	2	0	0	11
Gray catbird	*Dumetella carolinensis*	GRCA*	0	46	51	2	0	99
Gray flycatcher	*Empidonax wrightii*	GRFL	0	1	0	0	0	1
Greater scaup	*Aythya marila*	GRSC	0	2	0	0	0	2
Greater white-fronted goose	*Anser albifrons*	GWFG	0	0	0	2	0	2
Green-winged teal	*Anas carolinensis*	GWTE	0	0	0	1	4	5
Harry woodpecker	*Leuconotopicus villosus*	HAWO*	0	0	1	2	0	3
Herring gull	*Larus argentatus*	HERG	0	0	0	3	0	3
Hermit thrush	*Catharus guttatus*	HETH	0	0	1	1	0	2
House finch	*Haemorhous mexicanus*	HOFI*	493	272	480	901	528	2674
Hooded merganser	*Lophodytes cucullatus*	HOME	10	1	0	10	0	21
House sparrow	*Passer domesticus*	HOSP*	2528	1787	1336	1556	2256	9463
House wren	*Troglodytes aedon*	HOWR*	2	25	7	0	0	34
Killdeer	*Charadrius vociferus*	KILL*	102	81	48	0	4	235
Lesser goldfinch	*Spinus psaltria*	LEGO	0	0	0	1	0	1
Lark sparrow	*Chondestes grammacus*	LASP*	0	13	9	0	0	22
Lazuli bunting	*Passerina amoena*	LAZB*	0	87	28	0	0	115
Least flycatcher	*Empidonax minimus*	LEFL	0	1	0	0	0	1
Lesser scaup	*Aythya affinis*	LESC	7	0	0	12	28	47
Lincoln's sparrow	*Melospiza lincolnii*	LISP	1	0	9	2	0	12
Mallard	*Anas platyrhynchos*	MALL	93	37	63	629	317	1139
Marsh wren	*Cistothorus palustris*	MAWR	3	5	0	4	0	12
Merlin	*Falco columbarius*	MERL	0	1	0	2	2	5
MacGillivray's warbler	*Geothlypis tolmiei*	MGWA*	0	0	1	0	0	1
Mountain bluebird	*Sialia currucoides*	MOBL*	3	0	0	1	0	4

Mountain chickadee	*Poecile gambeli*	MOCH*	0	6	1	0	8	15
Mourning dove	*Zenaida macroura*	MODO*	29	55	53	4	0	141
Nashville warbler	*Leiothlypis ruficapilla*	NAWA	11	2	0	0	0	13
Northern flicker	*Colaptes auratus*	NOFL*	340	200	224	284	159	1207
Northern goshawk	*Accipiter gentilis*	NOGO	0	0	0	3	0	3
Northern harrier	*Circus cyaneus*	NOHA	4	1	0	2	0	7
Northern waterthrush	*Parkesia noveboracensis*	NOWA	0	0	5	0	0	5
Northern shrike	*Lanius borealis*	NSHR	0	0	0	4	7	11
Northern rough-winged swallow	*Stelgidopteryx serripennis*	NRWS	39	58	2	0	0	99
Orange-crowned warbler	*Vermivora celata*	OCWA	5	0	0	0	0	5
Olive-sided flycatcher	*Contopus cooperi*	OSFL	0	0	6	0	0	6
Osprey	*Pandion haliaetus*	OSPR	10	6	45	0	0	61
Pacific wren	*Troglodytes pacificus*	PAWR	0	0	0	2	0	2
Pied-billed grebe	*Podilymbus podiceps*	PBGR	7	4	9	0	0	20
Peregrine falcon	*Falco peregrinus*	PEFA	4	9	2	0	3	18
Pine siskin	*Spinus pinus*	PISI*	8	39	52	0	0	99
Pileated woodpecker	*Dryocopus pileatus*	PIWO	0	0	1	0	0	1
Pygmy nuthatch	*Sitta pygmaea*	PYNU	12	32	33	30	123	230
Ring-billed gull	*Larus delawarensis*	RBGU	0	9	3	0	0	12
Red-breasted merganser	*Mergus serrator*	RBME	0	0	0	2	0	2
Red-breasted nuthatch	*Sitta canadensis*	RBNU*	20	19	5	4	7	55
Ruby-crowned kinglet	*Regulus calendula*	RCKI	61	0	1	3	0	65
Red crossbill	*Loxia curvirostra*	RECR	19	58	18	12	17	124
Red-eyed vireo	*Vireo olivaceus*	REVI	0	11	4	0	0	15
Ring-necked duck	*Aythya collaris*	RNDU	0	0	0	0	14	14
Ring-necked pheasant	*Phasianus colchicus*	RNPH*	48	41	4	1	3	97
Rock pigeon	*Columba livia*	ROPI*	43	12	65	328	120	568
Rock wren	*Salpinctes obsoletus*	ROWR*	2	0	0	0	2	4
Red-tailed hawk	*Buteo jamaicensis*	RTHA	6	1	3	12	4	26
Ruddy duck	*Oxyura jamaicensis*	RUDU	12	9	0	6	0	27
Ruffed grouse	*Bonasa umbellus*	RUGR	1	0	0	0	0	1
Rufous hummingbird	*Selasphorus rufus*	RUHU	1	0	0	0	0	1
Red-winged blackbird	*Agelaius phoeniceus*	RWBL*	464	432	81	171	69	1217
Say’s phoebe	*Sayornis saya*	SAPH	109	38	23	2	7	179
Savannah sparrow	*Passerculus sandwichensis*	SAVS*	0	0	8	0	0	8
Snow bunting	*Plectrophenax nivalis*	SNBU	0	0	0	4	0	4
Sora	*Porzana carolina*	SORA	0	8	0	2	1	11
Song sparrow	*Melospiza melodia*	SOSP*	114	101	29	154	137	535
Spotted sandpiper	*Actitis macularius*	SPSA	5	21	5	0	0	31
Spotted towhee	*Pipilo maculatus*	SPTO*	60	44	22	37	27	190
Sharp-shinned hawk	*Accipiter striatus*	SSHA	0	0	5	6	1	12
Steller’s jay	*Cyanocitta stelleri*	STJA*	0	0	7	3	0	10
Townsend’s solitaire	*Myadestes townsendi*	TOSO	7	5	0	21	23	56
Tree swallow	*Tachycineta bicolor*	TRES	21	37	6	0	0	64
Trumpeter swan	*Cygnus buccinator*	TRUS	0	0	0	3	8	11
Turkey vulture	*Cathartes aura*	TUVU	38	0	0	0	0	38
Varied thrush	*Ixoreus naevius*	VATH*	0	0	0	35	9	44

Veery	*Catharus fuscescens*	VEER	0	8	0	0	0	8
Vesper sparrow	*Pooecetes gramineus*	VESP*	38	52	31	0	0	121
Violet-green swallow	*Tachycineta thalassina*	VGSW	213	142	6	0	0	361
Warbling vireo	*Vireo gilvus*	WAVI	2	1	1	0	0	4
White-breasted nuthatch	*Sitta carolinensis*	WBNU*	1	1	0	2	4	8
White-crowned sparrow	*Zonotrichia leucophrys*	WCSP*	952	0	38	111	22	1123
Western bluebird	*Sialia mexicana*	WEBL	57	9	24	159	38	287
Western kingbird	*Tyrannus verticalis*	WEKI	2	6	0	0	0	8
Western meadowlark	*Sturnella neglecta*	WEME*	39	48	10	0	1	98
Western tanager	*Piranga ludoviciana*	WETA	3	0	19	0	0	22
Western wood-peewee	*Contopus sordidulus*	WEWP	2	201	113	0	0	316
Willow flycatcher	*Empidonax traillii*	WIFL*	0	95	0	0	0	95
Wilson's warbler	*Cardellina pusilla*	WIWA	1	5	1	0	0	7
Wood duck	*Aix sponsa*	WODU	13	6	22	39	14	94
White-throated sparrow	*Zonotrichia albicollis*	WTSP*	0	0	0	1	0	1
White-throated swift	*Aeronautes saxatalis*	WTSW	2	30	0	0	0	32
Yellow-breasted chat	*Icteria virens*	YBCH*	6	38	2	0	0	46
Yellow rail	*Coturnicops noveboracensis*	YERA	0	0	3	0	0	3
Yellow warbler	*Setophaga petechia*	YEWA	7	164	67	2	0	240
Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	YHBL	6	8	0	0	0	14
Yellow-rumped warbler	*Setophaga coronata*	YRWA	107	0	78	19	1	205
Total individuals			10876	8888	9358	14431	8129	51682
Total Species			93	92	88	82	59	146

Table A6.2. Total number of observed species in each land class during each season.

	Urban	**Peri-urban**	**Agricultural**	**Natural**
Spring migration	41	50	56	74
Breeding Season	45	53	59	79
Fall Migration	38	56	56	66
Early Non-breeding Season	37	51	40	56
Late Non-Breeding Season	30	35	27	52

Table A6.3. LMER pairwise abundance comparison across land classes for each season. I assessed the four land classes, Agriculture (AG), Natural (NA), Urban (UR), and Peri-urban (PU), across 5 seasons.

Spring Migration	**Estimate**	**SE**	**DF**	**P-value**
AG – NA	13.540	5.56	270	0.0729
AG – PU	-43.765	5.86	271	<.0001
AG - UR	-39.184	5.60	270	<.0001
NA - PU	-57.305	5.80	274	<.0001
NA - UR	-52.724	5.54	273	<.0001
PU - UR	4.581	5.84	274	0.8615
Breeding Season				
AG – NA	-19.765	5.57	270	0.0025
AG – PU	-44.113	5.87	271	<.0001
AG - UR	-27.009	5.61	270	<.0001
NA - PU	-24.349	5.81	274	0.0002
NA - UR	-7.245	5.55	273	0.5607
PU - UR	17.104	5.86	274	0.0197
Fall Migration				
AG – NA	1.315	5.57	270	0.9953
AG – PU	-55.807	5.87	271	<.0001
AG - UR	-19.964	5.61	270	0.0025
NA - PU	-57.123	5.81	274	<.0001
NA - UR	-21.279	5.55	273	0.0009
PU - UR	35.843	5.86	274	<.0001
Early Non-breeding Season				
AG – NA	29.193	5.57	270	<.0001
AG – PU	7.832	5.91	276	0.5470
AG - UR	-5.250	5.61	270	0.7855
NA - PU	-21.362	5.85	279	0.0018
NA - UR	-34.443	5.55	273	<.0001
PU - UR	-13.081	5.89	274	0.1204
Late Non-breeding Season				
AG – NA	0.472	5.59	273	0.9998
AG – PU	-37.576	5.97	284	<.0001
AG - UR	-31.096	5.66	277	<.0001
NA - PU	-38.003	5.89	284	<.0001
NA - UR	-31.523	5.58	277	<.0001
PU - UR	6.480	5.96	287	0.6971

Table A6.4. LMER pairwise species richness comparison across land classes for each season. I assessed the four land classes, Agriculture (AG), Natural (NA), Urban (UR), and Peri-urban (PU), across 5 seasons.

Spring Migration	**Estimate**	**SE**	**DF**	**P-value**
AG – NA	-0.1621	0.531	418	0.9901
AG – PU	-1.1387	0.560	419	0.1774
AG - UR	-0.3598	0.535	418	0.9075
NA - PU	-0.9766	0.555	423	0.2948
NA - UR	-0.1977	0.530	422	0.9823
PU - UR	0.7789	0.559	423	0.5045
Breeding Season				
AG – NA	-2.1248	0.531	416	0.0004
AG – PU	-1.7552	0.561	417	0.0100
AG - UR	1.1972	0.536	416	0.1155
NA - PU	0.3696	0.556	420	0.9104
NA - UR	3.3221	0.531	419	<.0001
PU - UR	2.9525	0.560	420	<.0001
Fall Migration				
AG – NA	-0.3234	0.531	416	0.9293
AG – PU	-1.0235	0.561	417	0.2627
AG - UR	0.5924	0.536	416	0.6860
NA - PU	-0.7001	0.556	420	0.5898
NA - UR	0.9158	0.531	419	0.3121
PU - UR	1.6159	0.560	420	0.0214
Early Non-breeding Season				
AG – NA	-0.4386	0.531	416	0.8424
AG – PU	-1.6541	0.566	423	0.0190
AG - UR	-1.6899	0.536	416	0.0093
NA - PU	-1.2155	0.561	425	0.1346
NA - UR	-1.2513	0.531	419	0.0873
PU - UR	0.0358	0.565	425	0.9999
Late Non-breeding Season				
AG – NA	-0.7745	0.535	421	0.4704
AG – PU	-2.5829	0.574	433	0.0001
AG - UR	-2.3457	0.543	425	0.0001
NA - PU	-1.8084	0.567	432	0.0082
NA - UR	-1.5712	0.535	424	0.0182
PU - UR	0.2372	0.574	436	0.9762

Appendix 7

NMDS Seasonal Bird Communities

During spring migration, positive NMDS1 and negative NMDS2 were linked with desert habitat and species such as Canyon wrens (*Catherpes mexicanus*), Western Meadowlarks (*Sturnella neglecta*), and Western Bluebirds (*Sialia mexicana*). Conversely, negative NMDS1 values were associated with buildings and species like House Sparrows (*Passer domesticus*), Eurasian Collared-doves (*Streptopelia decaocto*), and California Quail (*Callipepla californica*). NMDS2 displayed positive associations with natural habitat variables including water, forests, and fields, which corresponded to species such as European Starlings (*Sturnus vulgaris*), Red-winged Blackbirds (*Agelaius phoeniceus*), Northern Flickers (*Colaptes auratus*), and Song Sparrows (*Melospiza melodia*). For NMDS3, positive values were associated with White-crowned Sparrows (*Zonotrichia leucophrys*), while negative values associated with vineyards, altitude, and species such as American Robin (*Turdus migratorius*), Say's Phoebe (*Sayornis saya*), Dark-eyed Junco (*Junco hyemalis*), and Vesper Sparrow (*Pooecetes gramineus*; Figure 2.5a and Figure A7.1).

During the breeding season, negative NDMS1 values corresponded to buildings and orchards, as well as species like House Sparrow, Eurasian Collared-dove, and Brewer's Blackbirds (*Euphagus cyanocephalus*). The forest habitat variable was strongly associated with both positive NMDS1 and negative NMDS2, which highlighted the strong association between forests and species such as the Western Wood-peewee (*Contopus sordidulus*). Other natural habitat variables, fields and water, were associated with both negative NMDS2 and NMDS3 values, and with species such as European Starling, Song Sparrow, Red-winged Blackbird, and Common Yellowthroat (*Geothlypis trichas*). Some species had a strong association with negative NMDS3, such as Willow Flycatcher (*Empidonax traillii*) and Cedar

Waxwing (*Bombycilla cedrorum*). NMDS2 and NMDS1 revealed positive correlations with desert habitats and species such as Canyon Wren, Western Meadowlark, Black-billed Magpie (*Pica hudsonia*), and Lazuli Bunting (*Passerina amoena*). Positive NMDS3 values were associated with vineyards and altitude, and species including Vesper Sparrow, California Quail, American Robin, and Spotted Towhee (*Pipilo maculatus*; Figure 2.5b and Figure A7.1).

During the fall migration, NMDS1 values were positively associated with deserts and species such as Lazuli Bunting and Chipping Sparrows (*Spizella passerina*). Negative NMDS1 values related to fields, buildings, and species including European Starling, California Quail, Brewer's Blackbird, House Sparrow, and Eurasian Collared-dove. NMDS2 showed positive associations with altitude, American Robin, and Yellow-rumped Warblers (*Setophaga coronata*). For NMDS3, positive values were tied to natural habitat variables such as water, forests, and fields, and with species including Cedar Waxwing, Northern Flicker, Western Wood-peewee, and Yellow Warbler (*Setophaga petechia*). Negative NMDS3 values were associated with orchards, buildings, and species like American Crows (*Corvus brachyrhynchos*) and Eurasian-collared Dove (Figure 2.5c and Figure A7.1).

During the early non-breeding season, NMDS1 values were positively associated with the natural habitat variables of desert, forest, water, altitude, and only the Canyon Wren. Negative NMDS1 values are associated with species like American robin, European starling, California quail, Dark-eyed junco, and House finch (*Haemorhous mexicanus*). Positive NMDS2 values exhibited associations with fields, forests and vineyards, and species such as Northern Flicker and European Starling, while negative NMDS2 values related to buildings, Eurasian Collared-dove, House Sparrow, and American Goldfinch (*Spinus tristis*). For NMDS3, positive values are associated with fields and water, as well as species like House

Finch, Song Sparrow, Mallards (*Anas platyrhynchos*), Buffleheads (*Bucephala albeola*), and Common Goldeneye (*Bucephala clangula*). Negative NMDS3 values correlated with altitude and species such as Black-billed Magpie, Dark-eyed Junco, American Robin, and Western Bluebird (Figure 2.5d and Figure A 7.1).

In the late non-breeding season, positive NMDS1 values were associated with buildings and species like House Sparrow, Eurasian Collared-dove, House Finch, and California Quail. Negative NMDS1 values are associated with habitat variables such as vineyards, altitude, and forests, with no specific bird species associations. NMDS2 showed positive values related to altitude, deserts, and buildings, and species such as Pygmy Nuthatch (*Sitta pygmaea*), House Sparrow, and Canyon Wren. Conversely, negative NMDS2 values are associated with natural habitat variables like forests, water, and fields, and species including Mallard, Song Sparrow, Bufflehead, Common Goldeneye, Black-capped Chickadee (*Poecile atricapillus*), and Bewick's Wren (*Thryomanes bewickii*). For NMDS3, positive values are linked to species such as American Robin, European Starling, and Black-billed Magpie, while negative values are associated with Dark-eyed Junco and Canada Goose (*Branta canadensis*; Figure 2.5e and Figure A7.1).

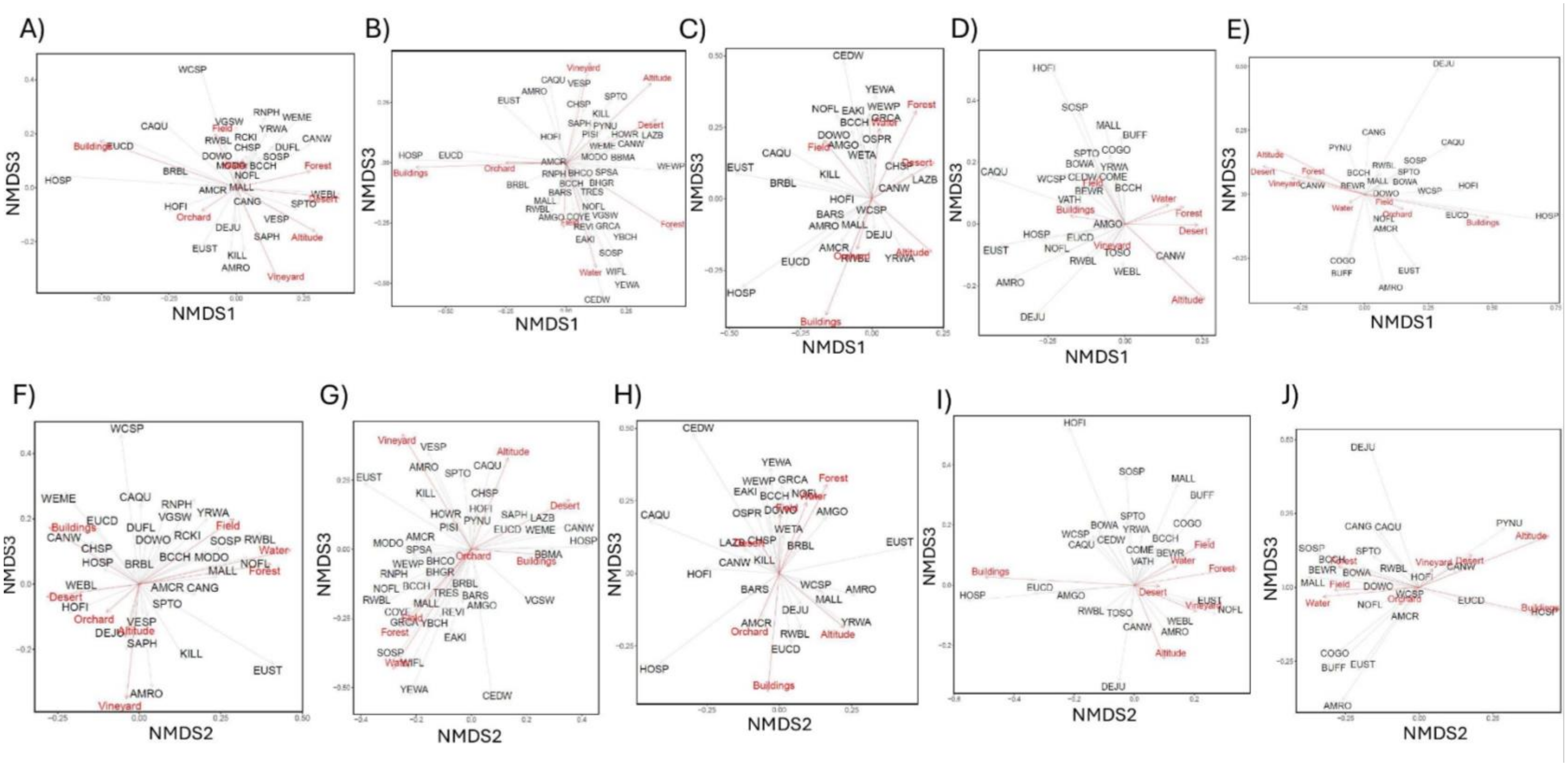


Figure A7.1. NMDS comparing species habitat associations across A and F) the spring migration B and G) breeding season, C and H) fall migration, D and I) early non-breeding season, and E and J) late non-breeding season. The top row is all NMDS1 compared to NMDS3, while the bottom row is NMDS2 compared to NMDS3. Birds are represented by four-letter banding codes (See Table A1 for 4-letter banding code and associated common and scientific names). Each row shows the three-dimensional comparisons for that season

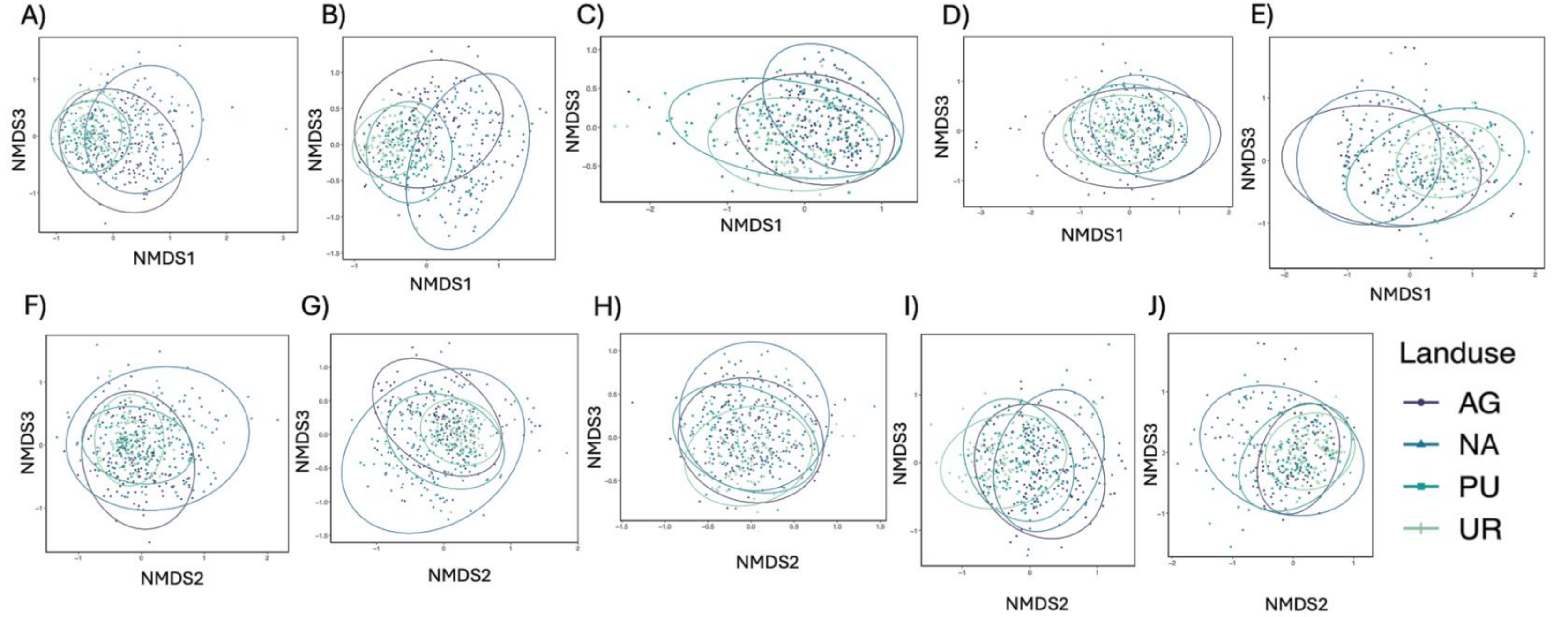


Figure A7.2. NMDS comparing species compositions across urban (UR), peri-urban (PU), agricultural (AG), and natural (NA) areas. I also compared four seasons: A and F) the spring migration B and G) breeding season, C and H) fall migration, D and I) early non-breeding season, and E and J) late non-breeding season. The top row shows NMDS1 vs NMDS3 and the bottom row shows NMDS2 vs NMDS3. Each point in the figure represents a point count and its location on the NMDS is based on the birds that were observed during that point count. Points that are closer together are more similar in species composition and points that are further apart are dissimilar in their species composition. The circles around the points show a 95% confidence interval.

Appendix 8

South Okanagan Valley, B.C., study area details and human population estimates.

The study area consists of three main sections, each encompassing a town and its surrounding area. Okanagan Falls is the smallest and most northern of the three towns, with a population of 2,266 (Government of Canada, 2023). Okanagan Falls is part of Electoral District D, which is 583.93 km^2 with a population of 4016 (Government of Canada 2023). Oliver, which is south of Okanagan Falls, has a population of 5,094 and is roughly 5.49 km^2 (Government of Canada, 2023). The study area surrounding Oliver is part of Electoral District C, which is 444.75 km^2 in size and has a population of 3,986 (Government of Canada, 2023). The most southern town is Osoyoos, which covers a total of 8.41 km^2 and has a population of 5,556 (Government of Canada, 2023). Most of the study area surrounding Osoyoos is part of Electoral District A, which has a population of 2,139 and is 258.04 km^2 in size (Government of Canada, 2023). The area northeast of Osoyoos is part of the census area of Osoyoos 1 Indian Reserve, belonging to the Osoyoos Indian Band, with a population of 1,426 and a size of 130.34 km^2 (Government of Canada, 2023). To estimate the total human population size, I combined all census data, which covers a total of 1,430.96 km^2 and has a total population of 22,217. This is an overestimation of the population size within the study area, but due to a lack of data, I can only say the population is between 12,916 (total of all three town populations) and 22,217 (total population estimates of towns and surrounding areas that surpass my study area).

Appendix 9

Additional methods regarding partnership in using trail cameras.

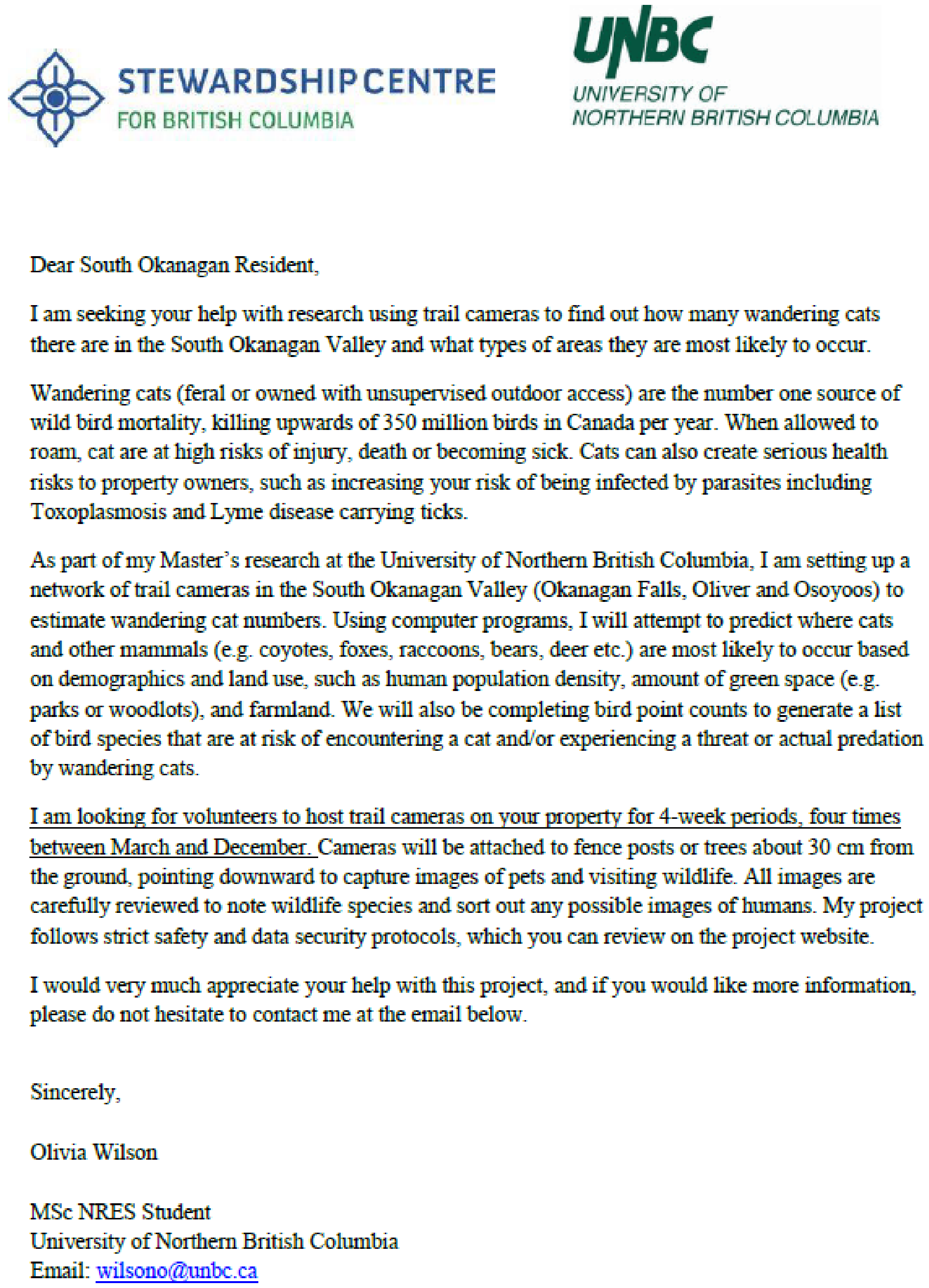
The Stewardship Centre for British Columbia's mission is to strengthen ecological stewardship through programs and resources for organizations, governments, private sectors, and the general public. This research aims to create the foundation of these resources and programs in the south Okanagan Valley to curtail the number of wandering cats, reducing welfare risks for both cats and wildlife. Partnership was also critical throughout the planning and deployment phases of this research, especially in finding camera sites for this study. I worked with local organizations and government groups to attain permits (e.g., The Nature Trust of British Columbia and BC Parks), and I connected with local groups (e.g., Penticton Naturalist Club) that would help me find interested participants to host trail cameras on private property. Permits were obtained from the Ministry of Forests, BC Parks, and The Nature Trust of British Columbia to put up cameras in natural habitats. There was also an article in the local news (Castanet) which discussed my research and search for household residents to let me put up cameras (Richardson, 2022). Additionally, the Stewardship Centre for British Columbia helped in the creation and distribution of blogs that updated the public on the research and any interesting findings throughout fieldwork (Wilson & Skurikhina, 2022a, 2022b).

Appendix 10

Additional information regarding the acquisition of trail camera sites.

Approximately 50% of the landowners I spoke with expressed interest in participating, emphasizing the importance of door-to-door engagement in strategically placing cameras within urban and peri-urban habitats. Speaking with residents face-to-face allowed me to explain the project in detail, showcase the cameras, and address their questions. Going door-to-door was also essential for finding agricultural sites; while many vineyards were initially contacted by email, responses were rare until follow-ups were conducted in person or over the phone. This initial meetings and subsequent interactions during camera switches ensured that property owners or workers became active participants in this research. I encouraged them to talk to me about what they observed in their yards or properties and I in turn shared highlights of interesting animals or other observations from the trail camera photos.

Appendix 11



STEWARDSHIP CENTRE
FOR BRITISH COLUMBIA

UNBC
UNIVERSITY OF
NORTHERN BRITISH COLUMBIA

Dear South Okanagan Resident,

I am seeking your help with research using trail cameras to find out how many wandering cats there are in the South Okanagan Valley and what types of areas they are most likely to occur.

Wandering cats (feral or owned with unsupervised outdoor access) are the number one source of wild bird mortality, killing upwards of 350 million birds in Canada per year. When allowed to roam, cat are at high risks of injury, death or becoming sick. Cats can also create serious health risks to property owners, such as increasing your risk of being infected by parasites including Toxoplasmosis and Lyme disease carrying ticks.

As part of my Master's research at the University of Northern British Columbia, I am setting up a network of trail cameras in the South Okanagan Valley (Okanagan Falls, Oliver and Osoyoos) to estimate wandering cat numbers. Using computer programs, I will attempt to predict where cats and other mammals (e.g. coyotes, foxes, raccoons, bears, deer etc.) are most likely to occur based on demographics and land use, such as human population density, amount of green space (e.g. parks or woodlots), and farmland. We will also be completing bird point counts to generate a list of bird species that are at risk of encountering a cat and/or experiencing a threat or actual predation by wandering cats.

I am looking for volunteers to host trail cameras on your property for 4-week periods, four times between March and December. Cameras will be attached to fence posts or trees about 30 cm from the ground, pointing downward to capture images of pets and visiting wildlife. All images are carefully reviewed to note wildlife species and sort out any possible images of humans. My project follows strict safety and data security protocols, which you can review on the project website.

I would very much appreciate your help with this project, and if you would like more information, please do not hesitate to contact me at the email below.

Sincerely,

Olivia Wilson

MSc NRES Student
University of Northern British Columbia
Email: wilsono@unbc.ca

Figure A11 Letter to landowners to assist with hosting trail cameras. Original letters only included four periods between March and December, because the late winter was added later in the project with the help of additional funding from Canadian Wildlife Services.

Appendix 12

Additional technical considerations when using trail cameras.

Multiple steps were taken to ensure the continuous capture and longevity of the trail cameras. I used lithium batteries to keep the Browning Strike Force HD Pro infrared trail cameras (model BTC-5HDP) functioning for as long as possible and checked the cameras bi-weekly to replace batteries and SD cards when necessary. Past experience with these cameras has indicated alkaline batteries last between 4–8 months, and lithium batteries ~8 months. Given the short deployment time of each camera, I did not expect cameras to run out of batteries. Despite this, there were some technical difficulties (e.g., some cameras stopped working when temperatures were too hot), which caused the camera to stop taking pictures for extended periods. The Okanagan had multiple heat waves during July and August, which held temperatures above 30°C, even at night, for weeklong periods, some days reaching over 40°C. In addition to malfunctioning, this seemed to have caused cameras to die quickly, making it necessary to replace batteries every 1-2 weeks during the spring and summer seasons. I mainly used SD cards that were 32 GB in capacity, with an additional 5 SD cards with 128 GB of storage for sites where high volumes of photos were common. Many SD cards filled up with pictures during windy periods or periods of vegetation growth, as the grass grew to the height of the camera and triggered the sensor.

Appendix 13

Additional ethical considerations when using trail cameras

Many of the cameras were placed along public walking paths in natural habitats, but because cameras were positions low to the ground to capture cats (30-60 cm above ground, i.e., cat height), I was often able to avoid capturing human faces, avoiding privacy concerns. In urban, peri-urban, and agricultural habitats, camera position was discussed with landowners to ensure compliance with any of their conditions, such as avoiding windows, high traffic areas, or spaces frequented by winery guests. This discussion with landowners was critical to ensure they were completely comfortable with the study, camera position, and how images would be stored or removed (i.e. images of human faces or identifying features were removed).

Appendix 14

Unmarked model abundance and detection equations.

Abundance is calculated using the following equation:

$$N_i \sim Negative\ Binomial\ (\lambda_i\ , \alpha)$$

Where N_i is the abundance, λ_i is the mean local abundance of cats at site i, and α is the dispersion parameter. The second part of the equation models detection using the following equation:

$$y_{ij} | N_i \sim Binomial\ (N_i\ , p_j\ )$$

Where y_{ij} is the detection or non-detection of an individual cat at site i during the jth occasion and p_j is the detection probability of an individual cat during the jth occasion (Kéry & Royle, 2016).

Appendix 15

Mapping Predicted Cat Local Abundance in the South Okanagan Valley

I created predictive maps of cat abundance across Okanagan Falls, Oliver, and Osoyoos using a predicted local cat abundance grid. I mapped a grid over the south Okanagan Valley in QGIS, with each grid being 200 x 200 metres. The grid was placed to focus on covering the towns of Okanagan Falls, Oliver, Osoyoos, and the space between and around the towns (Figure 2.1 outlines the gridded study area in red). The grid extended laterally to the mountains on either side, maintaining the focus in the valley, and longitudinally from the top of Okanagan Falls to the Canada/USA border. I extracted the habitat variables for each grid cell using the intersect tool, the same as used for extracting the buffer habitat variables (see GIS Analysis). After the habitat variables were obtained for each grid cell, I estimated the local abundance of wandering cats within each grid cell using the seasonal abundance model (without the detection covariate). I mapped each habitat variable within the seasonal abundance model separately and set the remaining habitat variables in the model to their mean. Maps showing predicted local cat abundance were created using QGIS by joining the predicted local abundance value with its corresponding grid cell and creating a graduated colour scheme, breaking the abundance estimates into 4 classes of equal intervals.

Appendix 16

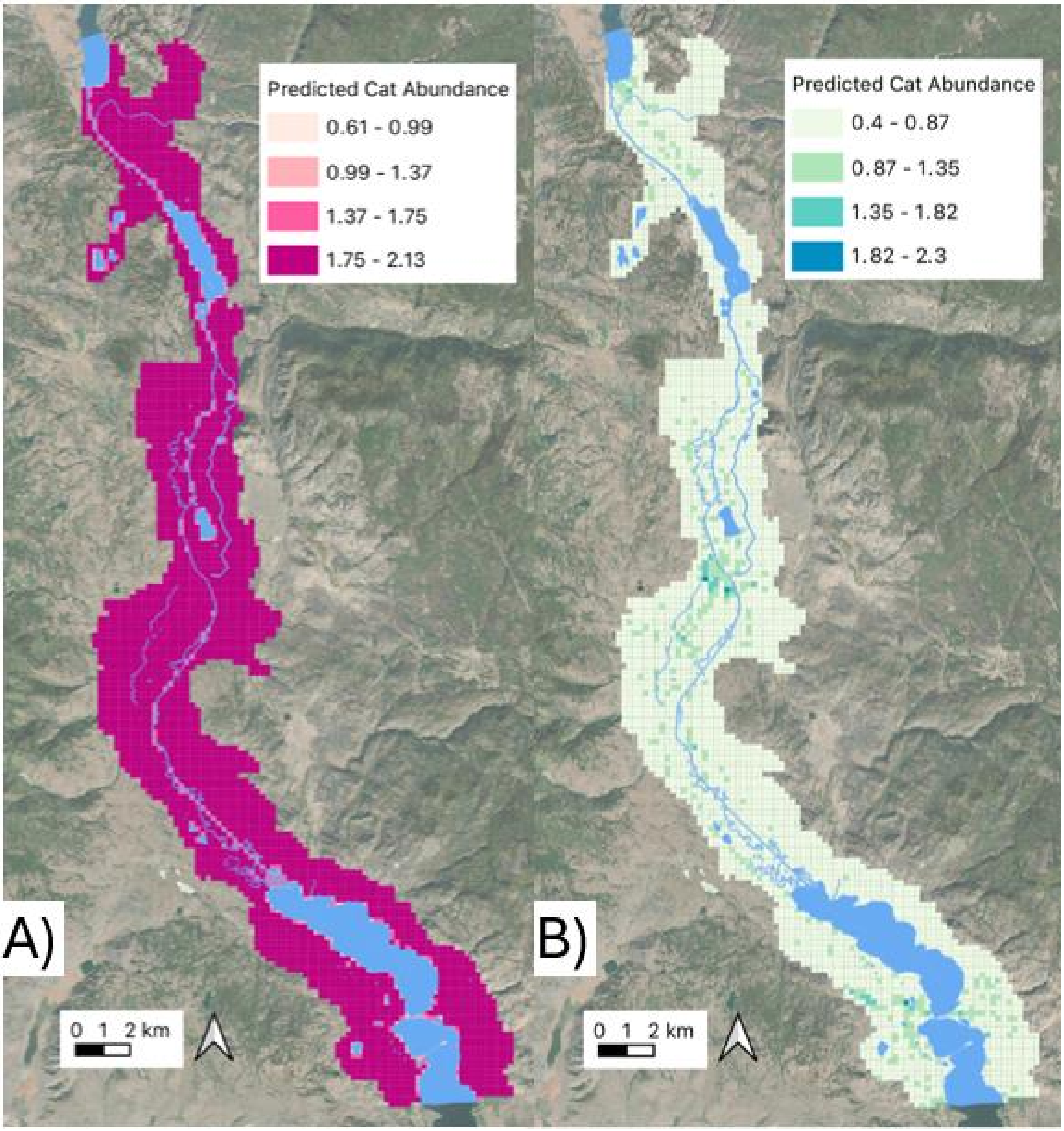


Figure A16.1 Predicted abundance of wandering cats during the fall based on A) the proportion of water in each cell and B) the length of roads in each cell.

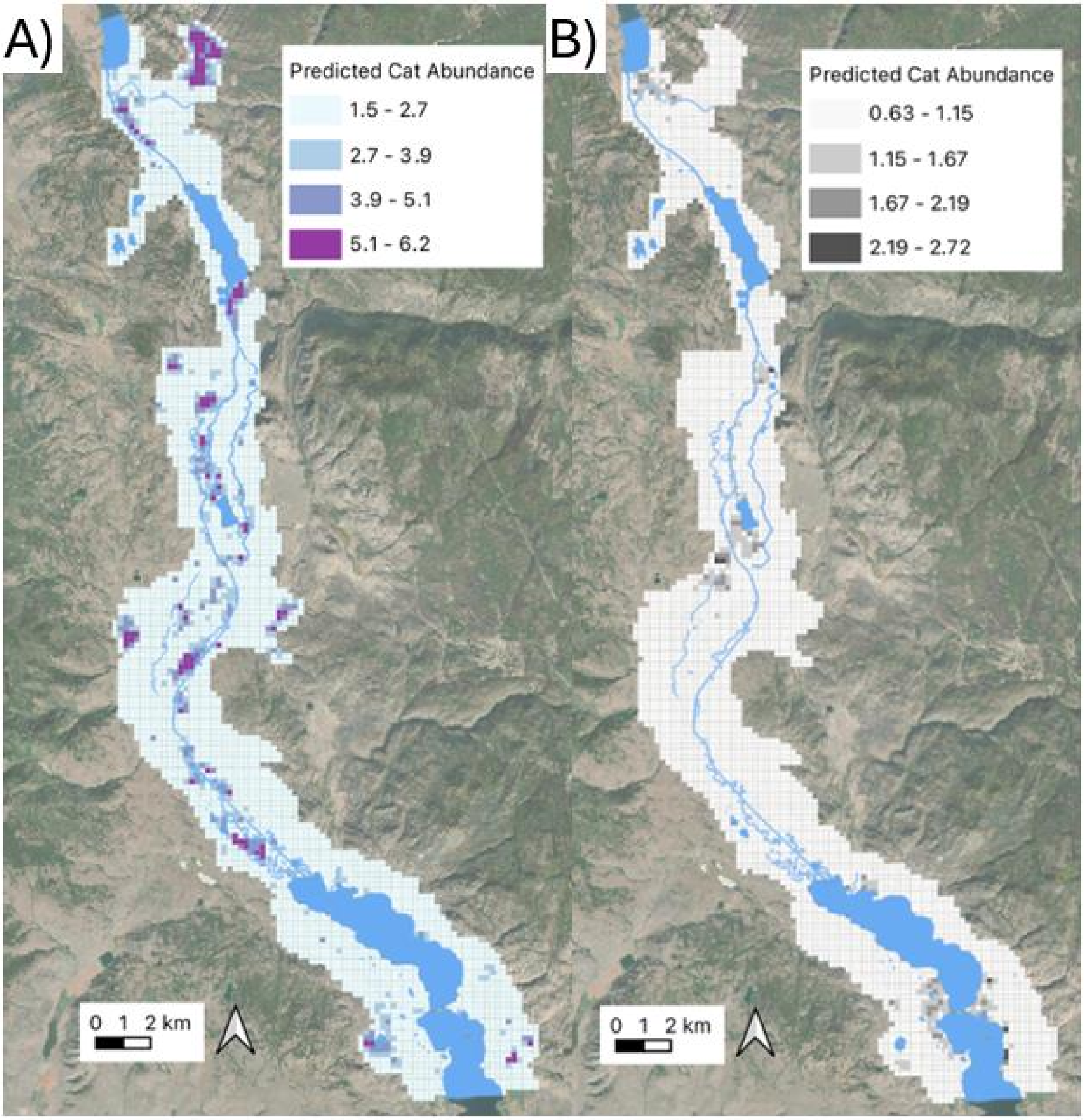


Figure A16.2 Predicted abundance of wandering cats during the late winter based on A) the proportion of fields in each cell and B) the number of buildings in each cell.

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