A Relative Landscape-Level Habitat Quality Model for the Burrowing Owls of the Canadian Prairies

by

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A thesis

presented to the University of Waterloo

in fulfilment of the

thesis requirement for the degree of

Master of Environmental Studies

in

Planning

Waterloo, Ontario, Canada, 2020

Author's declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners. I understand that my thesis may be made electronically available to the public.

Abstract

The range and population of the Burrowing Owls as flagship migratory species of the open prairie landscape are in decline across the northernmost portion of their global range in Canada. Multiple sources of degradation, including those induced by human footprint, are attributed to this declining trend. Yet the degradation caused by these factors is yet to be quantified and mapped at the landscape-level.

Using the InVEST habitat quality model, the habitat quality values for these endangered birds were quantified, mapped, and evaluated across both the historic and current ranges of these species in the Canadian Prairie ecosystem. In doing so, four different general categories of disturbance, namely the modified landscape, transportation network, urban areas, and energy infrastructure were considered. Also, variations of habitat quality values were modeled across the current range of these species upon the consideration of the different combinations of these sources of disturbance at this spatial extent.

The results of the study illustrate that despite the differences in the relative habitat quality values between the historic and current ranges of Burrowing Owls, these variations are not statistically significant between the two ranges when all sources of degradation are considered concurrently in the study area. Across the current range, also, the difference in habitat quality values is not statistically significant between the considered scenarios, even when specific habitat patches are assessed.

Nevertheless, the habitat quality was most affected by the transportation network data layer, followed by the energy and urban data layers. Consequently, the delineated spatial sources of disturbance can only be considered to have intensified the synergistic association between the other factors attributed to the decline of these species including the prolonged impact of grassland

conversion activities of the past, which has altered the configurational characteristics of the landscape, as well as to the other environmental factors affecting the population of these endangered species across the study area and beyond.

Considering the existing composition of land use/cover and share of specified sources of disturbance in habitat quality degradation across the current range of Burrowing Owls, conservation measures can be applied beyond the designated critical habitat boundaries for these species to preclude the potential future impacts of these sources of degradation. Further studies are still required to assess habitat quality values under the anthropogenic sources of degradation beyond the considered spatial extent and with regard to the other sources of disturbance across the global range of these species.

Acknowledgments

I would like to express my sincere gratitude to my Supervisor, Dr. Jeremy Pittman. His support was critical in the completion of this research. I thank him, first and foremost, for introducing me to the amazing world of conservation planning and landscape-level modeling. The initiation of a unique dialectic system between the two of us made me, in every sense of the word, a better scholar, and a more passionate researcher. I also would like to thank him for supporting me financially when I moved to the University of Waterloo to pursue my research dreams. Without his supports, this research work was impossible.

I would also like to express my deepest gratitude to Professor Keith Hipel, a lifelong mentor who introduced me to the field of multi-criteria decision analysis. The parameterization method utilized in this study was not possible without his encouragement to apply decision analysis techniques to different disciplines and research areas.

I also appreciate my committee member, Dr. Rob Feick, for his constructive feedback on this research, particularly on the MCDA section utilized for parameterization of the relative sensitivity scores. His feedback was critical in presenting the essential elements of the MCDA techniques for the readers of this thesis.

Last but not least, I would like to thank Dr. Richard Milgrom and Dr. David van Vliet of the University of Manitoba, and Colleen Sklar, my dear supervisor during my time at the Winnipeg Metropolitan Region. I thank them for allowing me to explore this unique nation and pursue my research dreams. Their support was fundamental in my understanding of the Canadian planning system, and will always be appreciated.

To Behrooz, Narges, Sahand, and Nayyer

for their support and never-ending love

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Chapter 1. Introduction

1.1 Context and Scope

The global biodiversity crisis is a major product and, interestingly, a concern of the late modern period, where due to the overexploitation of natural resources by the human race, terrestrial and marine ecosystems are losing their capacity to sustain different forms of life on Planet Earth (Johnson et al., 2017). Propelled by the need to increase the supply of provisioning ecosystem services, anthropogenic activities transformed the land and seascapes across the biosphere (Hoekstra et al., 2005). Technological advancements have led to unprecedented rates of natural resource exploitation, gradually destroying habitat for various taxa, the persistence of which benefits the very ecosystems and services we are using (MA, 2005).

Among different taxonomic groups, avifauna is the most influenced group with the highest range and population decline over the last two centuries due to the direct impact of human activities (Gaston and Blackburn, 1997). This thesis is a modeling study aimed to assess the impact of human footprint on the breeding habitat of one of the charismatic migratory bird species of the Americas "Burrowing Owls (Athene cunicularia)" across the Canadian Prairies, their breeding habitat in Canada (COSEWIC, 2017).

This breeding habitat is part of the Prairie Ecozone, which is the most altered terrestrial ecosystem in the country (Kerr and Deguise, 2004). The area is recognized by the flat to rolling prairie landscape, the significant proportion of which has been converted from grass to grain in the past couple of centuries (Riley et al., 2007). Notwithstanding the significant reductions in the rate of native habitat conversion across the landscape (Watmough and Schmoll, 2007), the population of wildlife species kept declining across this ecosystem over the past few decades. This trend is attributed to the more recent landscape change across the region, characterized by the combined impact of human-induced disturbances such as further configurational alterations (i.e., availability and distribution) of the native habitat, and the expansion of urban, transportation, and energy infrastructure in the area (Davis, 2004; Ludlow et al., 2015). Other environmental factors such as extreme climatic events across the region are also estimated to have accelerated the decline of biodiversity across this unique ecosystem (Jarzyna et al., 2016).

This thesis is a modeling and mapping study to quantify and portray the relative habitat quality values for Burrowing Owls under the anthropogenic sources of threat across the historic and current ranges of these migratory birds across the Canadian breeding grounds, where they are considered endangered. The results present, for the very first time, whether the considered sources of degradation can be a contributing factor behind the gradual decline of these species from the landscape.

What is important to note is that despite providing essential information on the potential causes of decline, the majority of the previous studies investigated the declining trend of these species considering only small samples of these birds across fragments of their breeding habitat. Consequently, these species-specific studies failed to portray a landscape-level image describing this trend at larger spatial extents, including their entire range in the Canadian Prairies. Also, there exists no precedent of quantified relative habitat quality values for these migratory birds among past studies.

Built upon the fundamental information collected from the existing literature on Burrowing Owls, this study is the first step to quantify the relative habitat quality values across the historic and current ranges of these species. In doing so, a coarse-filter landscape-level modeling framework was utilized to present habitat quality maps across both ranges for these avian species in the Prairie Ecozone. In this respect, key information was collected on the biology of Burrowing Owls, the ecological and environmental processes these species depend upon for survival, and their behavior in response to the human-induced configurational changes of the environment

1.2 Thesis Structure

The second chapter of this study presents a comprehensive literature review on what we know so far about the biology, demographic trends, habitat requirements, ecological processes and threats to Burrowing Owls both across their global range and in Canadian breeding grounds. Accordingly, the research gaps and questions are identified in the next step. Before that, however, a general review of the literature is presented on the history of human-induced landscape alterations of the global terrestrial ecosystems and the repercussions of these changes on living organisms. Also, the concept of habitat and its relationship with wildlife is briefly discussed from the lens of multiple scholars. In this respect, different habitat quality measurement techniques are compared for avian species.

The third chapter introduces the methods utilized in this study. It starts with a thorough description of the modeling framework adopted for the habitat quality measurement. In addition to the general framework, the rationale behind this modeling approach is presented with an in-depth explanation of the mathematical background to model development and the parameters required by this landscape-level modeling framework. Also, a simple multi-criteria decision rule is presented to parameterize one of the parameters required by this model with relevance to specific factors affecting the considered species.

Chapters four, five, and six are the results, discussion, and conclusion sections of this thesis. The results of this modeling study are presented in chapter four through appropriate statistical analyses over the modeled habitat quality values and the contributions made by the considered spatial sources of disturbance to habitat quality degradation. These results are then discussed in chapter five with reference to the available literature in ecology and conservation planning, and other scholarly works on Burrowing Owls. The implication for practice is briefly discussed afterward, and the section is finalized by the limitations associated with the modeling approach adopted in this research. The conclusion section summarizes the meaning of the results and proposes potential future research trajectories that could be adopted to further enlighten our understanding about the landscape-level habitat quality mechanisms affecting the rage and population of Burrowing Owls across the study area and beyond.

Chapter 2. Literature Review

2.1 Biodiversity Decline in the Age of Anthropocene

The first human-induced ecosystem structure change dates back to over two million years ago when the ecological niche of large species was destroyed by the genus of Homo in Africa (Johnson et al., 2017). Since then, human-driven destabilization of natural environments has caused vast configurational alternations to the natural settings. In the past millennials (i.e., the age of Anthropocene), the entirety of the global biodiversity has shrunk by Homo sapiens through destabilization of life and life-supporting processes across all ecosystems on Planet Earth (Wilson, 1989; Johnson et al., 2017). Led by the degradation of natural environments (Pereira et al., 2010; Rands et al., 2010), the human footprint on global ecosystems is now considered to cause a larger biome crisis (Hoeskstra et al., 2005), influencing all planetary biogeographic units (i.e., ecoregions), distinguished by unique climate, ecosystem, and biodiversity (Olson et al., 2001; Hoeskstra et al., 2005).

The relationship between biodiversity and ecosystem is a complex one (Mace et al., 2012). On the one hand, biodiversity is critical for ecosystem stability and the services it provides (MA, 2005). More specifically, biodiversity, from microorganisms to large vertebrates, can be seen as a regulator of ecosystem processes, a final ecosystem service, or a good (Mace et al., 2012). Accordingly, ecosystems are altered over time by the drivers of environmental change affecting wildlife and plant species (Hautier et al., 2015). On the other hand, the flow of organisms, materials, and energy is critical to maintaining biological diversity in different ecosystems through ecological processes (Crooks and Sanjayan, 2006). These processes connect species to one another. Consequently, the extirpation of only one ecological partner will influence others through the elimination of ecological functions required to sustain viability (Mace et al., 2012). Propelled by increasing human population and demand for a better life (Hanski, 2011), this mutual relationship has been disrupted through a complex web of interactions influencing living organisms and their environment (Nelson et al., 2006). Drivers of human-induced ecosystem change can take a variety of direct and indirect biological, physical, social, political, economic, demographic, and cultural forms (MA, 2005), disturbing ecological processes and different forms of life, in a combined fashion, over space and time (Nelson et al., 2006).

The land use/cover change, overexploitation of natural resources, environmental contamination, and species translocation are the most fundamental factors that have historically contributed to the declining trend of global biodiversity (Lande, 1998). Among these causes of decline, land use/cover change, particularly conversion of native land covers to agricultural land uses, is the oldest direct ecosystem change driver with the largest impact on global ecosystems and ecological processes defined in natural settings (Nelson et al., 2006). The anthropogenic land use/cover conversion to increase the net primary productivity has not only led to the decline of biodiversity through degradation and elimination of suitable wildlife habitat across the globe but also degraded the carrying capacity of different ecosystems in maintaining ecological balance required for the viability of organisms (Brooks et al., 2002; Gatson et al., 2003; Groom et al., 2006). Technological and scientific advancements in agriculture have accelerated the overexploitation of natural resources and the extent of land conversion activities (Nelson et al., 2006).

Global climate fluctuations along with modifications made to natural and semi-natural land use/cover, energy development, and urban and infrastructural developments are the more recent causes leading to ecosystem disruption and biodiversity decline across the globe (Kerr and Deguise, 2004; Luck et al., 2004; Bartlett et al., 2015; Maxwell et al., 2016; Scheffers et al., 2016). The underlying characteristics of all these factors are the elimination of suitable habitat and degradation of life-supporting processes for wildlife species across different ecosystems. More specifically, the synergistic association between these anthropogenic activities have accelerated habitat degradation by decreasing the natural rates of decay and succession across different habitat conditions for wildlife species (Sinclair et al., 1995; Sax and Gaines, 2003; Thomas et al., 2004; Ewers and Didham, 2006; Brook et al., 2008).

2.2 Wildlife Habitat and its Quality as Relative Concepts

Species habitat and its quality are integral to wildlife ecology and considered as the building blocks of biodiversity persistence in ecosystems (Hodgson et al., 2009). Accordingly, more and better habitat are sometimes deemed to be positively associated with occupation (i.e., inhabitancy), which in turn would increase species population and range in a given ecosystem (Sinclair et al., 1995; Hodgson et al., 2009; Hodgson, 2011). Given that habitat is a species-specific concept (Fischer and Lindenmayer, 2007), a variety of terms such as habitat occupancy, carrying capacity, habitat suitability, critical habitat, and habitat structure were used to describe habitat and its condition for different species (e.g., Boyd, 1986; Laymon and Barret, 1986; Alverson et al., 1988; McCoy and Bell, 1991; Block and Brennan, 1993; Anderson and Gutzwiller, 1994). For instance, Daubenmire (1968) considered habitat typology as the basis for comparing the quality of different land use/cover. Building upon this definition, some past studies (e.g., Gysel and Lyon, 1980; Peek, 1986; Laymon and Barret, 1986; Morrison et al., 1991; Samuel and Fuller, 1994) equated habitat with natural vegetation structure. This definition of habitat entails an only partial description of the habitat requirement for different groups of species, and thus caused confusion over the true nature of this concept in wildlife conservation.

As an effort to standardize habitat and habitat-related terminologies, Hall et al. (1997) defined habitat as “the resources and conditions present in an area that produce occupancy including survival and reproduction by a given organism." (p. 175). Unlike any single-criterion perception of habitat, this definition portrays the term as the sum of specific resources unique to each organism in an environmental setting (Hall et al., 1997). Habitat quality was accordingly defined as a continuous variable illustrating the environmental conditions required for survival, reproduction, and persistence of individuals and populations, and critical habitat for species was defined as areas with high habitat quality (Hall et al., 1997).

Despite presenting an intuitive definition of habitat quality, this definition masks specific requirements concerning the persistence of individuals and populations among different taxonomic groups (Johnson, 2007). For instance, access to high-quality habitat is prioritized for some individuals, even if it is limited across spatial scale, whereas for populations, the abundant average habitat would be of higher interest (Hobbs and Hanley, 1990; Pidgeon et al., 2006). More specifically, different groups of species require unique habitat conditions, which may vary based upon factors such as scale, spatiotemporal changes caused by the anthropogenic alteration of the landscape, and natural or induced habitat selection/avoidance behavior by species (Forman, 1995; George and Zack, 2001; Hanski, 2011; Haddad et al., 2015). These factors influence the wildlife-habitat relationship and the nature of the concepts such as habitat and habitat quality among different groups of species or even among individuals in the same taxonomic group (Johnson, 2007).

2.2.1 Considerations for Avian Species

During the past century, the population of the global avifauna has significantly reduced to just a quarter of the pre-agricultural era, making birds the most declined taxonomic group among different species (Fisher and Paterson, 1964; Wood, 1982; de Juanna, 1992; Gaston and Blackburn, 1997). The average global population of avifauna is estimated to be less than a hundred billion species, which shows dramatic declines across a range of tropical and temperate areas, including grassland/steppe, scrubland, temperate deciduous forests, savannah, and tropical woodland (Gaston et al., 2003). This trend has led to an increased vulnerability of approximately 40% of the global avian population (BirdLife International, 2018). Fragmentation and loss of native habitat, which in many real situations occur concurrently across the landscape (Forman, 1995), are the major causes of this declining trend. In addition, other anthropogenic drivers of change such as landscape modifications, energy development, reduced food availability, urban and infrastructural expansions, and climate change have been recognized as the other causes of global avian population decline (Wiens, 1995; BirdLife International, 2018).

Habitat for birds, can be defined as environmental factors and processes contributing to the evolutionary history and fitness (i.e., survival or reproductive success) of species (Block and Brennan, 1993). Habitat quality for birds, accordingly, can be measured either by assessing the attributes of a given habitat or through simultaneous assessment of the demographic, distributional, and performance (physical) indicators of bird species (Johnson, 2007). The former approach requires the consideration of critical factors such as availability of nests and food resources, as well as other indicators (e.g., predation, competition) limiting the use or accessibility to these essential resources (Johnson, 2007). More specifically, habitat quality should be examined considering not only the necessary resources for survival but also a set of factors affecting these essential resources (Morrison et al., 2006). Consequently, using only the crude vegetation index as a surrogate for avian habitat quality would yield inadequate results as it only provides a descriptive view of the species-habitat relationship (Morrison, 2001).

Incorporating the simultaneous impact of demographic, distributional, and performance indicators into the habitat quality assessment across a given landscape is not realizable, first and foremost, due to the financial and data limitations in avian research. Also, biased judgments on the importance of each indicator may overshadow the accuracy of habitat quality assessment (Johnson, 2007). Last but not least, these indicators cannot be individually utilized as they are unique to each species and may vary over space and time (Johnson, 2007).

For instance, demographic attributes such as abundance, survival, and reproduction are considered to be popular measures of habitat quality (Bock and Jones, 2004; Knutson et al., 2006). Yet, there might be diverging results between these factors across some ecosystems, particularly in those heavily influenced by human disturbances as conditions favoring one species might not necessarily be considered appropriate for the other (Franklin et al., 2000).

Similarly, factors such as time lags, or relaxation time as introduced by Diamond (1972), ecological traps, site fidelity, dispersed habitat patches, and intraspecific competitions may lead to different distributional patterns, which might result in the disproportionate use of habitat by avian species in diverse landscapes (Johnson, 2007), ultimately casting doubt on the accuracy of habitat quality results in studies constructed solely on species distribution across a given ecosystem (Jones, 2001; Manly et al., 2002; Morrison et al., 2006; Thomas and Taylor, 2006).

Likewise, despite being effective, particularly in migratory birds for which the demographic and distributional factors are hard to attain, the performance indicators (i.e., physical conditions) also partially describe the quality of habitat or reproductive success for avian species. This is because these measures can also vary over time and in response to environmental disparities between different ecosystems or behavioral differences among species. For instance, species with high levels of body fat might prefer habitats with scarce food resources and minimal danger of predation to habitats with abundant food resources and more serious danger of predation (Johnson, 2007).

The process of habitat selection is another important consideration in determining habitat quality for avian species (Johnson, 1980). According to Hutto (1985), habitat selection is a "hierarchical process involving a series of innate and learned behavioral decisions made by an animal about what habitat it would use at different scales of the environment" (p. 458). Also, a four-step hierarchical process for habitat selection was introduced by Johnson (1980), which starts with the geographical range selection, and is followed by the selection of home range, selection of specific patches across the landscape, and selection of the available trophic options, respectively. The outcome of this process is the disproportionate use of some resources over others across the landscape, which was named "habitat preference" by Hall et al. (1997).

Using similar definitions, George and Zack (2001) identified regional, landscape, macro, and micro-habitat scales as spatial hierarchies, the dynamics of which influence species behavior across a given ecosystem. Consequently, the interdependence of habitat selection and the issue of scale is of the utmost importance in determining habitat quality for avian species (Johnson, 2007). That is, habitat selection by species is a scale-dependent process whereby factors or dynamics at the higher scales (e.g., at the landscape scale) influence the micro-scale processes (Allen et al., 1987).

The other dimension that needs to be simultaneously investigated with the process of multi-scalar habitat selection is the time factor and its relevance to habitat preference by species (George and Zack, 2001). Accordingly, both spatial and temporal scales of habitat selection depend on species dispersal across the environment and its perception of the dynamics within the ecosystem (Kotliar and Wiens, 1990). More specifically, small-scale habitat selection preference may be influenced by factors occurring at a fraction of time (e.g., availability of prey or presence of competitors), whereas large-scale habitat selection depends on the long-term evolutionary processes such as geological or geomorphological changes, which occur over decades or even centuries. In addition, large-scale habitat selection is considered to be a genetic factor (Hutto, 1985), whereas micro-scale habitat selection is determined by factors such as environmental learning and perception, which differs from time to time and from one species to another.

The Anthropogenic-driven changes in the landscape might accelerate the occurrence of some of the long-term processes or environmental perceptions of habitat preference among some species (Cody, 1985). Thus, the large-scale ranges of many species might be affected due to human-induced alterations of natural ecosystems (George and Zack, 2001). Consequently, the interplay between the time, scale, and species-specific preference factors need to be considered when the wildlife-habitat interaction is the subject of habitat quality studies, and the acquired results should not be extrapolated beyond the scales at which these factors might change (Hall et al., 1997).

2.3 Burrowing Owls as Endangered Flagship Species of the Canadian Prairies

The Burrowing Owl (Athene cunicularia) is a tiny ground-dwelling migratory owl species of the open grassland areas, which is closely associated with burrowing (fossorial) mammals (COSEWIC, 2017). It weighs no more than 238 g, stands 20 cm tall, and has the smallest facial disk among owl species (Scobie et al., 2013, COSEWIC, 2017). This charismatic grassland bird species has long legs, big rounded yellow eyes, a short tail, and a brownish body that has cream and beige spots in the chest and feather areas (Environment Canada, 2012). From southern Canada to southern South America, the open grassland systems of the Americas, are the breeding range of this flagship predatory species (Environment Canada, 2017). The western Burrowing Owls (hereafter Burrowing Owls) and the Florida Burrowing Owls are the two subspecies of this bird that breed across North America (Dechant et al., 2002).

These species were historically considered as common elements of the Canadian prairies and the small portions of the southern British Columbia. Nevertheless, their range and population have significantly declined across their breeding grounds in Canada. The majority of the remaining population (=254 mature individuals) currently breed from southcentral Alberta to southern Saskatchewan (COSEWIC, 2017). From the reintroduced birds, only 16 species are living in British Columbia. Despite a very short period of reoccupation in Manitoba, following a reintroduction program (Environment Canada, 2012), the most recent field observations illustrate a zero occupancy rate across this westernmost prairie province in the past few years (COSEWIC, 2017).

As migratory species, Burrowing Owls return to their Canadian breeding grounds in early April and remain in these areas until September (Haug, 1985; Haug and Oliphant, 1990). These species lay an average clutch of 9 eggs, of which three to five eventually fledge (Wellicome, 2000). Hatchlings gain sustained flight ability within forty days of birth, and dispersal of the fledglings begins only thirty days later (Wellicome, 1997). However, owls remain in the natal site until August and start migrating to their wintering habitat in the south during September (Haug and Oliphant, 1990).

Adult pairs cooperate over nest maintenance; yet during the breeding season, male birds involve in active vigilance and food provisioning activities outside of the Burrows, but female owls brood hatchlings. Both adults and juveniles might use several non-natal burrows - also known as satellite burrows - located in close proximity (≤ 30 m) to the nest burrows. This behavior is an anti-parasite and anti-predation strategy, which is adopted by adults to protect juveniles and increase the probability of hatchling survival (Desmond, 1991; Plumpton and Lutz, 1993; Desmond and Savidge, 1999). Site fidelity (i.e., returning to the natal sites) differs between adults and first returning generations. This number varies from 0 km (i.e., absolute fidelity) to 3500 km among different individuals across the breeding grounds (Wellicome et al., 1997; De Smet, 1997; Duxbury, 2004).

During the past forty years, the breeding range of Burrowing Owls in the Canadian Prairies has shrunk to only a third of their historic range in the early 1900s (Environment Canada, 2012). This range contraction occurred concurrently with dramatic back to back decennial declines of 90% and 64% from 1990 to 2000, and from 2005 to 2015, respectively (COSEWIC, 2017).

Burrowing Owls were first designated as "Endangered" species in 1995 by the Committee on the Status of Endangered Wildlife in Canada (Wellicome and Haug, 1995). In 2003, the status of these birds was also confirmed as "Endangered" by the Species at Risk Act (SARA) developed by the federal government to protect, manage, and recover endangered and threatened species (Environment Canada, 2012; COSEWIC, 2017). Despite this designation across the Canadian breeding grounds, these birds are still considered as "Apparently Secure" across their global range (NatureServe, 2019).

2.4 Habitat

2.4.1 Availability of Burrows

As their name suggests, Burrowing Owls have a very high association with their burrows. The burrows utilized by these species differ depending on the type of use; while some are used for nesting, some others are used for roosting purpose (Environment Canada, 2012). Since the availability of appropriate burrows is a critical factor for these semi-colonial species (Bent, 1961; Haug et al., 1993), they tend to locate their nests in areas with abundant burrows excavated by fossorial mammals including Coyote, Foxes, American Badger, Black-tailed Prairie Dogs, and Richardson's Ground Squirrel (Salt and Wilk, 1958; Bent, 1961; Stewart, 1975; Desmond 1991; Haug et al. 1993; Wellicome and Haug, 1995; Desmond and Savidge, 1996, 1999; Wellicome 1997; Leupin and Low, 2001; COSEWIC 2006; Thiele et al, 2013).

Among these species, the abundance of Black-tailed Prairie Dogs is estimated to be positively linked with the owls' population across its North American breeding range (Butts and Lewis, 1982). There is also strong evidence that across the Canadian range, burrows excavated by other mammals such as Richardson's Ground Squirrels and American Badgers are frequently utilized as nesting sites by these ground-dwelling bird species (COSEWIC, 2017). The results attained from field research also suggest that these species might also select artificial burrows - excavated as conservation measures - to maintain and expand their existing range (De Smet, 1997; Wellicome et al., 1997; Leupin and Low, 2001; Nadeau, 2015; Riding and Belthoff, 2015). For instance, the reintroduced population of Burrowing Owls in their westernmost Canadian range in BC tend to utilize the artificial burrows (Mitchell, 2008). However, the occurrence of the wild pairs in the Prairie Provinces of Canada, particularly Alberta and Saskatchewan, depends mainly on the abundance of burrows in the colonies of the Black-tailed Prairie Dogs, or the ones that are excavated naturally by other fossorial mammals (COSEWIC, 2006).

Studies on the Prairie Dog colonies (Bent, 1961; Butts and Lewis 1982; MacCracken et al., 1984; Desmond and Savidge, 1996; Toombs, 1997) showed considerable rates inhabitancy by Burrowing Owls in both active and inactive burrows. However, the owl species are inclined to occupy the active burrows as nest depredation rates are much lower in active colonies (Butts and Lewis 1982; Desmond and Savidge, 1999; Toombs, 1997). In the absence periodical maintenance by the Prairie Dogs, nest depredation rates are much higher, and burrows are usually prone to the encroachment of dense vegetation cover (Desmond, 1991; Desmond and Savidge, 1999).

The areas within the conspicuous high density colonies of the Black-tailed Prairie Dogs in the western block of the Grasslands National Park in Saskatchewan are designated as critical habitat for almost 15% of the Burrowing Owls population in Canada (Environment Canada, 2012). Outside this range, the low remaining burrows are distributed across a vast spatial scale. Other burrows excavated by fossorial mammals, including Richardson's Ground Squirrel and American Badger are rarely occupied by the Owls across their Canadian range (Environment Canada, 2012).

2.4.2 The Landscape-Level Habitat Requirements

Despite the contraction of their historic breeding range, Burrowing Owls have been recorded across a wide variety of land use/cover categories in Canada and the U.S. (Dechant et al., 2002; Environment Canada, 2012; COSEWIC, 2017). More specifically, if high dependency on nest burrows is not considered, Burrowing Owls can be regarded as both prey and habitat generalists, which is an indication of their relative resistance to environmental changes across the landscapes they tend to occupy throughout the year (Environment Canada, 2012).

The results of the available modeling and empirical studies to date suggest that Burrowing Owls could occur, reproduce, and survive across both natural (i.e., grasslands and wetlands) and semi-natural environments (i.e., croplands, introduced pasture). Consequently, conservation practices targeting specific land use/cover are deemed to be lacking the spatial extent needed to be considered for controlling the declining population of these migratory birds (Environment Canada, 2012).

According to some studies (Hjertaas and Lyon, 1987; Restani et al., 2001; Murphy et al., 2001; Klute et al., 2003; Warnock and Skeel, 2004; Holroyd and Trefry, 2011), the significant native habitat loss due to the pre-1980s landscape conversion activities is the primary reason for the contraction of the historic range of Burrowing Owls in the Canadian Prairies. Despite this contraction, however, many U.S. and Canadian studies since the 1940s have investigated the suitability of different land use/cover as nesting and foraging habitats for Burrowing Owls (Dechant et al., 2002). The results of these studies illustrated various habitat preferences among individual owl species (Dechant et al., 2002; Environment Canada, 2012; COSEWIC, 2017).

The choice of nesting location depends heavily on two factors: a) availability of short and sparse grass cover, which is regularly grazed by livestock or consumed by rodent mammals, and b) proximity to areas containing a high density of nesting and roosting burrows (James et al., 1991; Plumpton and Lutz, 1993; Faanes and Lingle, 1995; Wellicome and Haug, 1995; Clayton and Schmutz, 1999; Poulin et al., 2005; Thiele et al., 2013). At the landscape level, Burrowing Owls occupy treeless areas with flat plains (Haug et al., 1993; Thiele et al., 2013). Regardless of the type of the grass cover (native or non-native), these species typically tend to occupy burrows with immediate grass surroundings (Clayton and Schmutz, 1999; Poulin et al., 2005).

The results of an earlier study by Wedgwood (1976) in southcentral Saskatchewan illustrated strong nesting avoidance across the tilled grounds, woody vegetation, and even reintroduced grasslands. Yet, nest success was reported to be higher in tame pasture than in native grasslands in another study by Haug (1985). Thus, suitable nesting grounds consist of open grassland systems such as prairie landscapes (which may include non-grass vegetation such as sagebrush), grazed pasture, and sometimes, edges of agricultural fields (Poulin et al., 2005). Evidence from past studies (Salt and Wilk, 1958; Bent, 1961; Stewart, 1975; Wedgwood, 1976; Haug, 1985) also confirms the association between the nesting location and vegetative cover in immediate nest surroundings.

Despite being the dominant landscape across the breeding range of Burrowing Owls, croplands are rarely utilized as nesting grounds by these avian species (Haug, 1985; Poulin et al., 2005; Stuber et al., 2018). Furthermore, nest success is typically low in these areas as these lands are regularly cultivated (Dechant et al., 2002). In Alberta, only 41% of the identified nest sites were located within 500 meters of croplands. In 59% of times that Burrowing Owls were observed across croplands, no nesting locations were identified (Schmutz, 1997). This might indicate that farming landscapes are more often used as foraging or loafing destination for these avian species.

Habitat selection for breeding was examined at a larger extent in Alberta and Saskatchewan in a later study by Clayton and Schmutz (1999) across four different land use/cover categories including native pasture, tame pasture, cropland, and other. Among these, native pasture was recorded as the most suitable habitat for nesting and roosting over the two-year observation period in twenty-one study sites in Alberta. In Saskatchewan, however, both nesting and roosting sites were equally divided between the native and non-native pasture (Clayton and Schmutz, 1999). In rare instances, other land use/cover categories, including hayland, fallow fields, roadsides, and even urban areas were recorded as breeding sites. Nevertheless, these areas were considered as non-suitable breeding habitat for Burrowing Owls, and their use is only limited to occasional foraging endeavors by these species (Konrad and Gilmer, 1984; Haug, 1985; James et al., 1990; Haug et al., 1993).

The maximum recorded diurnal home range for these species is 250 m, which belongs to the adult male owls spending a significant portion of the day protecting the nest burrows (Haug and Oliphant, 1990; Scobie, 2015). The nocturnal home range, however, demonstrates an entirely different behavior among these species. While Burrowing Owls typically select immediate nest surrounding for foraging purposes, a large body of evidence (Haug and Oliphant, 1990; Plumpton, 1992, Sissons et al., 2001; Sissons, 2003; Marsh et al., 2014a; Marsh et al., 2014b; Scobie, 2015) suggest that these species might fly to further distances which may extend their home ranges beyond the preferred land use/cover for foraging purposes. This distance could potentially increase under extensive cultivation regimes (Haug, 1985; Wellicome and Haug, 1995), or be diverted to roads and roadsides, which, despite the high relative average prey availability (Sissons et al., 2001), are not considered as preferred destinations for these species (Marsh et al., 2014b).

Prey abundance could not be considered as the only factor affecting foraging habitat selection by Burrowing Owls as this behavior was observed over different compositions of land use/cover (Sissons et al., 2001; Marsh et al., 2014b). Yet, instances of successful foraging attempts illustrated a high association with sparse vegetation across the foraging grounds with different land use/cover (Marsh, 2014a). Among these, croplands, pasture, and fallow fields were investigated simultaneously in multiple studies (Butts and Lewis, 1982; Desmond, 1991; Haug et al., 1993; Wellicome, 1994; Sissons et al., 2001; Sissons, 2003; Marsh et al., 2014a; Marsh et al., 2014b).

In southern Saskatchewan, the results of the studies by Wellicome (1994) and Wellicome and Haug (1995) across different land use/cover categories illustrated that prey abundance was lower across the periodically-plowed landscapes such as croplands and fallow fields. Also, habitat with taller vegetation cover (30-60 cm) including some areas of native grasslands, roadside vegetation, and mature croplands had more prey abundance than areas covered with tame pasture (Wellicome, 1994). Yet, strong foraging avoidance was observed across mature crops with average height of at least 100 cm, even when prey sources were abundant. This behavior is attributed to the hindrance caused the owls' flying ability by the dense vegetation cover, which ultimately lead to extended foraging time and energy across these fields. Two more recent studies by Marsh et al. (2014a; 2014b) approved this negative relationship between the vegetation height and foraging time across different land use/cover categories in Alberta and Saskatchewan.

In an effort to narrow down the factors determining the critical breeding habitat for Burrowing Owls, Stevens et al. (2010) modeled the distribution of these species with reference to the impact of both biotic and abiotic environmental factors across a landscape consisting of 30% of their current breeding habitat in the Canadian Prairies. Although not providing absolute habitat suitability values, the results of the model suggested that abiotic factors, including elevation and slope, were much more important predictors than biotic factors for nesting site selection. Nevertheless, biotic factors, such as land use/cover, remain relatively important in the 4.8 square kilometers of the immediate nest surroundings (Stevens et al., 2010). Among these land use/cover categories, medium patches of grasslands, areas with positive native grass growth, and ephemeral wetlands were estimated to be more suitable habitat for these species. The results of the model also illustrated that no relationship could be inferred between the home range selection and nest success or fledgling production (Stevens et al., 2010).

Empirical studies of the last two decades in Alberta and Saskatchewan (Haug and Oliphant, 1990; Sissons et al., 2001; Sissions, 2003) revealed varying results with reference to foraging home range and reproductive success of adult Burrowing Owls across a combination of patches considered as habitat for these birds. While cultivated croplands were the dominant patches (more than 50%) in two of the local study sites located in Saskatchewan (Haug and Oilphant, 1990; Sissons et al., 2001), farming areas only comprised a negligible share (only 1%) of the habitat composition across the local study site in Alberta (Sissons, 2003). Despite a relatively small share (less than 5%) of the land use/cover composition in both sites, ephemeral wetlands were also recorded among the most popular areas for foraging by the adult Burrowing Owls (Haug and Oilphant, 1990; Sissons, 2003).

While the lowest nest success was observed across the croplands of Saskatchewan (Haug and Oliphant, 1990), the least number of fledglings were recorded in Alberta, where the landscape was dominated by pasture areas (Sissions, 2003). Home ranges also varied widely, even among the areas dominated by cultivated lands. Across the local study sites in Saskatchewan, for instance, home ranges vary, on average, between 0.34 and 2.41 square kilometers of the nest sites (Haug and Oilphant, 1990; Sissons et al., 2001). This range was around 3.28 square kilometers in a pasture-dominated study site in Alberta (Sissons, 2003).

The results of these studies suggest that home ranges for adult Burrowing Owls are not necessarily correlated with the abundance of croplands or pasture areas. Evidence from another study by Shyry (2005) on the nocturnal foraging behavior of the juvenile owls in Alberta confirms the results of empirical studies on adult Burrowing Owls in the area. In this study, no significant association was found between foraging tendency and the availability of natural grassland areas. Quite surprisingly, at a finer scale, owls preferred unconventional destinations for foraging, which have minimum vegetation cover such as truck trails and petroleum well pads (Scobie et al., 2013).

2.5 Sources of Habitat Degradation

The contracted range and decreasing population of Burrowing Owls might raise some questions concerning some intrinsic vulnerabilities among these species (Lantz et al., 2004). Except for the small remaining population across the edges of the current range, however, the rest of the owls in the Great Plains do not illustrate associations with intrinsic vulnerability characteristics such as endemism, extreme habitat specificity, low dispersal, genetic isolation, and hybridization (Clayton and Schmutz, 1999; Poulin et al., 2005; Todd, 2001a; Todd, 2001b; Thiele et al., 2013). The results of a study by Korfanta (2001) illustrated that genetic isolation and low dispersal of these species across their breeding range are wrong hypotheses.

Highly specific habitat requirements are also rejected as these birds have historically illustrated adaptations to moderate changes in the land use/cover, leading scholars to categorize them as habitat and prey generalists (Haug et al. 1993; Warnock and James 1997; Clayton and Schmutz 1999; Orth and Kennedy 2001). Nevertheless, productivity, survival, and high associations with nest burrows are unique attributes of the Burrowing Owls' ecology that might be interpreted as increasing intrinsic vulnerability characteristics (Lantz et al., 2004) or limiting factors (COSEWIC, 2017), disrupting ecological processes across the breeding range of these endangered birds, particularly in the presence of anthropogenic sources of disturbance.

The declining population of Burrowing Owls across their breeding range in Canada cannot only be attributed to a single external factor. Rather, this decline is a result of the cumulative impact of multiple extrinsic threats and critical limiting factors influencing these birds in a complex way (Environment Canada, 2012). These factors include categories such as loss of burrows, loss of trophic options, increased predation, extreme weather conditions, landscape modification through the application of environmental contaminants, energy development, and collisions with vehicular traffic (Environment Canada, 2012; COSEWIC, 2017).

These degradation categories are directly and, sometimes, indirectly correlated in a complex way, may encompass or lead to other causes of degradation, portray unique historical trends, and result in the cumulative impact of very high for these endangered species (COSWEIC, 2017). Undoubtedly, loss of native habitat through extensive crop production, petroleum explorations, and urban development is the most critical factor driving the landscape alteration in the Canadian Prairies during the past decades (COSWEIC, 2017). Nevertheless, the real impact of these activities on the declining population of Burrowing Owls in Canada remains poorly quantified (Environment Canada, 2012), and is only limited to speculations in a number of studies (Hjertaas et al., 1995; Sheffield, 1997; McDonald et al., 2004).

2.5.1 Loss of Native Habitat

Loss of native habitat could be considered as one of the degradation sources adversely influencing the historic and current ranges of Burrowing Owls. Although occurred with different reported intensities across the Great Plains, the historical conversion of native grasslands to croplands in the

Canadian Prairies throughout the past decades (WWFC, 1987; Samson and Knopf, 1994) has been considered as the primary reason leading to habitat loss for Burrowing Owls (Wellicome and Haug, 1995; Clayton and Schmutz, 1999; Poulin et al., 2011). Despite its significant influence on the past population trends, the conversion of grasslands to cultivated land uses, including agriculture and introduced pasture, is currently considered a low-impact threat across the existing range of these species (COSEWIC, 2017). However, in southwestern Saskatchewan, where approximately 40% of the native habitat had been converted to agriculture in the past few decades, native habitat loss is still considered a medium-impact threat for these birds (Parks Canada Agency, 2016).

The remaining native habitat is estimated to undergo low conversion rates in the near future, primarily due to inappropriate soil structure and moisture level required for seeding annual crops (Olimb and Robinson, 2019). Nevertheless, this consideration might be affected by unpredicted market forces (e.g., changes in crop values) or improved crop seeds with high resistance to different geomorphological conditions, including those found across the remaining patches of the grasslands in the prairies (Gjetvaj and Bentham, 2014).

Interestingly though, the population of Burrowing Owls has been declining with faster rates than the loss of native habitat in southern Saskatchewan (Holroyd and Trefry, 2011). Evidence presented by Skeel et al. (2001) illustrated an annual loss of six percent of the native plant species from the late 1980s to the early 1990s in southern Saskatchewan. In this period, however, the population of Burrowing Owls declined with rates much higher (approximately four times higher) than the loss of grasslands in the area (Skeel et al., 2001). At a larger scale in the Canadian Parries, the rate of population decline over the past three decades outpaced the rate of native habitat conversion (COSEWIC, 2006). Thus, habitat loss due to land conversion activities cannot be presumed as the sole major factor propelling this declining trend (Holroyd and Trefry, 2011).

2.5.2 Ecosystem Modification, Severe Weather Condition, and Declined Trophic Options

The combined impact of ecosystem modifications and diminishing trophic options is a fundamental threat for the endangered population of Burrowing Owls across their existing range in Canada (Environment Canada, 2012; COSEWIC, 2017). The systematic use of rodenticides and insecticides negatively affects the prey availability for Burrowing Owls through reductions of offspring production and survival rates among these species (James et al., 1990).

Abundant prey is a fundamental factor influencing offspring production during the breeding season (Environment Canada, 2012). According to Poulin et al. (2011), availability of prey sources such as voles and grasshoppers is a critical factor for the persistence of these species across the landscape; abundant prey might lead to increased populations through improving the survival rates among the adults and juveniles (Poulin, 2003). This association became apparent in 1997, when an unprecedented eruption in the number of voles across the prairies led to increased rates of nest success and post-fledgling survival across the breeding grounds of Burrowing Owls in Alberta and Saskatchewan (Wellicome et al., 1997; Wellicome, 2000; Todd et al., 2003).

The use of chemicals as pest control measures on agricultural fields and pasture areas might also adversely influence Burrowing Owls. That is, the ingestion of the carcasses of the poisoned insects or other prey species would indirectly poison these small raptors (Environment Canada, 2012). Due to a negative association that exists between the application of pesticides and nest and reproductive success of Burrowing Owls (James and Fox, 1987; James et al., 1990; Mineau and Whiteside, 2013; Mineau and Palmer, 2013), a 0.5 km buffer is considered, as a preventive measure, across the prairie farmlands of Alberta upon the application of chemicals, particularly carbofuran and Carbaryl (Environment Canada, 2009).

Nevertheless, the use of these and other chemical substances across the wintering grounds of Burrowing Owls in the U.S. and Mexico could also be considered as a severe threat for these migratory birds, resulting in lower rates of return to the Canadian breeding grounds per annum (McDonald et al., 2004). The results of the few available studies on the application of insecticides to the livestock population for parasite treatment (Floate et al., 2008; Suarez et al., 2008) illustrated a negative correlation between the implementation level of chemicals and the availability of the insects in the pasture areas. The application of pesticides also adversely influences the population of burrowing mammals through the direct or indirect poisoning incidents (Proulx, 2014). This would, in turn, increase the areas of ungrazed pasture and decrease the availability of suitable burrows and foraging grounds for Burrowing Owls (Hjertaas and Lyon, 1987; Marsh et al., 2014b).

Severe weather conditions and altered grazing patterns are considered to be the other factors influencing the existing population of Burrowing Owls by undermining the suitable habitat and decreasing the trophic options for these small predatory birds (Heisler et al., 2014; Marsh et al., 2014b). Accordingly, extreme weather condition - mostly associated with storm events, prolonged winters, or extended droughts - could adversely influence the population of these species (Wellicome et al., 2014), particularly through nest destruction or abandonment, reduced offspring production (Fisher and Bayne, 2014), restricted foraging range (Wellicome, 2000), or the decline of rodent mammals across the prairies. Altered grazing patterns due to changes in the management and ownership regimes might be another contributing factor, as well. In this respect, the transfer of ownership and management of grazed pasture might serve as a major reason behind the declining population trend. With the conversion of grazed pasture to croplands and the subsequent loss of grazing livestock, the nesting and foraging requirements for Burrowing Owls are seriously affected across their breeding grounds in the prairie landscape (COSEWIC, 2017).

2.5.3 Landscape Fragmentation, Sensory Disturbances, and Increased Predation

Despite being relatively unquantified, the fragmentation of native landscape due to the dramatic reduction of the prairie habitat is also considered as a contributing factor to the declining population of Burrowing Owls in several studies (e.g., Konrad and Gilmer, 1984; Ratcliff, 1986; Haug et al., 1993). A fragmented landscape may result in less pair-bonding among the small and localized groups (Klute et al., 2003). It might also result in extended home range, higher road mortalities across the semi-natural matrices including vast areas of croplands (Warnock and James, 1997; Clayton and Schmutz, 1999), and increased population of mammalian predators and large avian raptors preying on these tiny birds during the breeding season (Wellicome and Haug, 1995).

If not the most critical degradation source, increased predation, according to a number of studies (Wellicome and Haug 1995; Leupin and Low, 2001; Todd et al., 2003; Shyry, 2005), is one of the major reasons behind the increasing fatality of Burrowing Owls across their breeding range. In the absence of larger mammalian predators due to extensive alterations made to the landscape structure for crop production, the population of the smaller predators such as striped Skunk, Raccoon, and Coyote has increased in the prairies. This increase is understood as a major cause of nest depredation across the existing breeding grounds (Environment Canada, 2012).

The increased population of larger avian raptors is another predation threat for Burrowing Owls. Utility poles, trees, and availability of perch features of this kind is the main reason for increased predation across the Burrowing Owls' range (Houston and Bechard, 1983). This trend is primarily attributed to the combined impact of declined prey availability and landscape fragmentation (Todd 2001a,b).

Numerous artificial perch sites across the prairies also attract adult male Burrowing Owls during the breeding season (Scobie, 2015). Vigilant attentiveness and communicating potential predation threats with burrows is one of the major nest protection strategies adopted by the male Burrowing Owls from the nesting to post-fledgling periods (Chipman et al., 2008). The abundance of artificial perch sites (e.g., fence posts) across the landscape, however, has altered this behavior significantly by attracting more owl species to the elevated features developed concurrently with the land use/cover transition across the changing prairie landscape (Martin, 1973). The co-occurrence of the abundant anthropogenic features such as the elevated perch sites and road networks may influence the quality of audible waves transmitted to and from the nest locations (Parris and Schneider, 2009).

The results of a study by Scobie (2015) on the combined impacts of the anthropogenic perch features (e.g., fence posts) and traffic sound on the diurnal space usage of the adult male Burrowing Owls in southeastern Alberta and Saskatchewan illustrated strong perching avoidance from features located in the proximity of roads with average passing traffic speed of 80 km/h. Also, roads in many areas of the prairies are situated above the surrounding land uses, and, consequently, might cause visual disturbance to protective vigilance activities by these species.

In response to this disturbance, owls might utilize road surface areas (Scobie, 2015). This behavior, would increase the probability of predation by larger avian predators flying along these corridors during the daytime (Meunier et al., 2000). Additionally, using roads as perch locations could also increase the probability of collision with passing vehicular traffic, which is deemed to be a major factor leading to increased fatality rates among these avian species (Wellicome, 1997; Clayton and Schmutz, 1999; Todd, 2001b; Shyry, 2005).

The impacts of acoustic sensory disturbances emanating from some of the anthropogenic features were investigated on the nocturnal foraging habitat of Burrowing Owls in the same study by Scobie

(2015). Accordingly, when the atmospheric attenuation is not considered, Burrowing Owls are capable of detecting sounds up to an average maximum of 0.2 km, 0.2 km, and 1 km from oil wells, small vehicles, and larger vehicles with passing traffic speed of 97 km/h or higher, respectively. Despite relying on both sensory and visual leads while foraging, the space usage of Burrowing Owls is more affected by the footprint of the physical elements across the landscape (Scobie, 2015).

2.5.4 Other Sources of Degradation

Urbanization, mining, and energy development are the other factors estimated to be influencing the existing population of Burrowing Owls by further dividing the suitable habitat for these endangered species across the prairie landscape (Environment Canada, 2012; COSEWIC, 2017). However, the real impact of these factors is poorly studied across the breeding range of these birds in Canada. Among these factors, the impact of urban areas was investigated on the non-migratory subspecies of Burrowing Owls in a number of studies in the urban-rural gradient context in the southern United States (Millsap and Bear, 2000; Chipman et al., 2008) and South America (Cavalli et al., 2018; Franco and Marcal-Junior, 2018). Accordingly, despite relative adaptation to the built environment (Franco and Marcal-Junior, 2018; Cavalli et al., 2018) and availability of prey sources in these areas (Chipman et al., 2008), the population of these subspecies remains highly sensitive to increasing urban density and the associated anthropogenic sources of disturbance (e.g., road kills, pets, etc.), in the built environments (Millsap and Bear, 2000).

There is no published study on the impact of the mining industry on the suitable habitat for these birds, nor there exist any relevant research investigating the potential impacts of the growing renewable energy projects, particularly wind farms, on the current range of these birds in Canada. Nevertheless, compared to non-renewable sources of energy, which are currently experiencing an economic downturn, the potential expansion of the wind energy projects over the next few decades is estimated to further undermine the habitat quality for these endangered species (COSEWIC, 2017).

The impact of wind turbines on Burrowing Owls, however, was directly assessed in a number of studies across a wind resource area in California, USA (Smallwood et al., 2007; Smallwood et al., 2009; Smallwood et al., 2013). Accordingly, nearly 600 owl species perish per annum due to collisions with wind turbines (Smallwood et al., 2009; Smallwood et al., 2013). Most collisions occurred in the vicinity of wind farms located in a range of 15 m from the Ground Squirrel Burrows, 20 m of intensive livestock grazing areas, and 90 m from the nest sites. Burrowing Owls fly more frequently within 50 m of the turbines in the area. They also tend to perch in areas with lower concentration of wind turbines (Smallwood et al., 2007).

Despite causing sensory disturbances to Burrowing Owls, petroleum infrastructure, operation sites, and the associated linear features are regarded as features which influence the space usage of Burrowing Owls (Scobie, 2015). The area occupied by these features directly influences the habitat for these species by creating edge effects and changing the optimum height and density of vegetation structure. More specifically, these alterations have serious consequences on critical factors, including prey abundance, perch availability, and predation risk, all with a direct impact on the persistence of these endangered birds across the landscape (Scobie, 2015).

There are federal regulations to mitigate the impact of upstream petroleum infrastructure on Burrowing Owls and their nest locations (Environment Canada, 2009). Although not legalized, these regulations set timing restrictions and setback guidelines to buffer nest sites both in time and space at federal as well as provincial scales (Scobie et al., 2013). According to these regulations, the maximum recommended setback buffer is 0.2 km from the nest sites and satellite burrows throughout the breeding period for low-impact developments (Environment Canada, 2009).

In 2011, provincial restrictions were enacted for new drilling activities in Alberta for upstream oil and gas infrastructure on public lands (Government of Alberta, 2011). In Saskatchewan as well, there are provincial guidelines, but these guidelines bear no legal restriction and are only limited to recommendations (Government of Saskatchewan, 2017). Since provincially-owned lands are the target of these guidelines, none of these documents apply to privately owned lands across this landscape. Furthermore, these guidelines target new developments only and do not address the disturbance caused by the existing operating petroleum infrastructure.

2.6 The Existing Gaps and Research Questions

As discussed, habitat and its quality are fundamental, but relative concepts influencing wildlife across a variety of terrestrial ecosystems. Built upon the wildlife-habitat interaction, this relativeness has historically served as a contributing factor to the development of species-specific habitat measurement methods used by different experts. Nevertheless, indicators used in these methods are hard to measure simultaneously, due to the time, financial, and data limitations, and can only portray a partial picture of habitat quality when considered individually.

Further confusion in habitat quality measurement is caused due to lack of knowledge about the impact of the multi-scalar processes on the behavioral attributes of species in different ecosystems, the time-dependency of factors associated with the evolutionary characteristics of species, and the scope of natural and human-induced alterations across different terrestrial ecosystems.

Habitat quality measurement for the Burrowing Owls in the Canadian Prairies is no exception to these partial and relative measurement approaches. More specifically, the majority of the past studies on habitat preference of Burrowing Owls (e.g., Haug, 1985; Haug and Oliphant, 1990; Schmutz, 1997; Sissons et al., 2001; Scobie, 2015) were based upon the demographic, distributional, or performance indicators of small samples of these birds and across fragments of their current range in the Canadian Prairies. Despite providing critical information regarding the contracted range and the species-habitat interactions in small population samples, these studies overlooked the spatiotemporal factors affecting the quality of habitat across still a wide prairie landscape deemed to be relatively suitable within the current range of these species.

Given that there is still a gap in the existing body of knowledge concerning the cumulative influence of multiple anthropogenic sources of degradation on the landscape-level habitat quality values for Burrowing Owls in Canada, more comprehensive relative habitat quality models are needed to improve our understanding with reference to the influence of the existing spatial sources of disturbance and their distribution on habitat quality values for these endangered species.

The purpose of this thesis, consequently, is to study the relative habitat quality values for Burrowing Owls under the existing spatial composition of anthropogenic sources of degradation across the historic and current ranges of these endangered species in the Canadian Prairies. More specifically, this study aims to answer the following sets of questions:

1. What are the relative habitat quality values across the historic and current ranges of Burrowing Owls in Canada considering the existing spatial distribution of the transportation, energy, urban, and the modified land uses? And whether the resultant habitat quality values differ dramatically across both spatial extents?

2. How does the habitat quality value change for Burrowing Owls when the different combinations of spatial sources of degradation are considered across the current range of

these species in the Canadian Prairies? And whether variations between the resultant habitat quality values are significant?

Chapter 3. Methods and Materials

3.1 Study Area

The study area in this research encompasses both the recent (2004) and historic (pre-1970s) ranges of Burrowing Owls in the Canadian Prairies (Figure 3.1). This area is considered the northernmost range of these endangered bird species in North America. Both of these ranges are located within the boundaries of a landscape known today as the Prairie Ecozone of Canada, which has the total area of 465,094 square km, an area equal to approximately 4.7% of the land surface area in the country.

The historic breeding range of Burrowing Owls covers approximately 450,000 km^2 of this landscape. It stretches as far east as the Red River Valley in Manitoba and west to the foothills of the Rocky Mountains in Alberta (Environment Canada, 2012). From the south, the landscape has borders with the US states of Montana, North Dakota, and Minnesota, and from the north, it is boarded by Boreal Plains Ecozone (ESTR Secretariat, 2014). The historic range of Burrowing owls had shrunk by 27% in the 1970s (Wedgwood, 1978) and by 53% by the early1990s (Wellicome and Haug, 1995). With only 160,000 km^2 of breeding area, the current range of the Burrowing Owls covers only 36% of its historic range in the Prairie Ecozone, mainly including the prairie landscape of southcentral and southeastern Alberta and southwestern Saskatchewan (COSEWIC, 2017).

As one of the fifteen national terrestrial ecological zones (ESWG, 1995), the Prairie Ecozone is an ecosystem characterized by flat and rural lands rich in agriculture and energy production. The landscape has historically been considered as suitable habitat for a wide variety of plant and animal species (ESTR Secretariat, 2014). This ecosystem is also distinguished by its variable climate marked

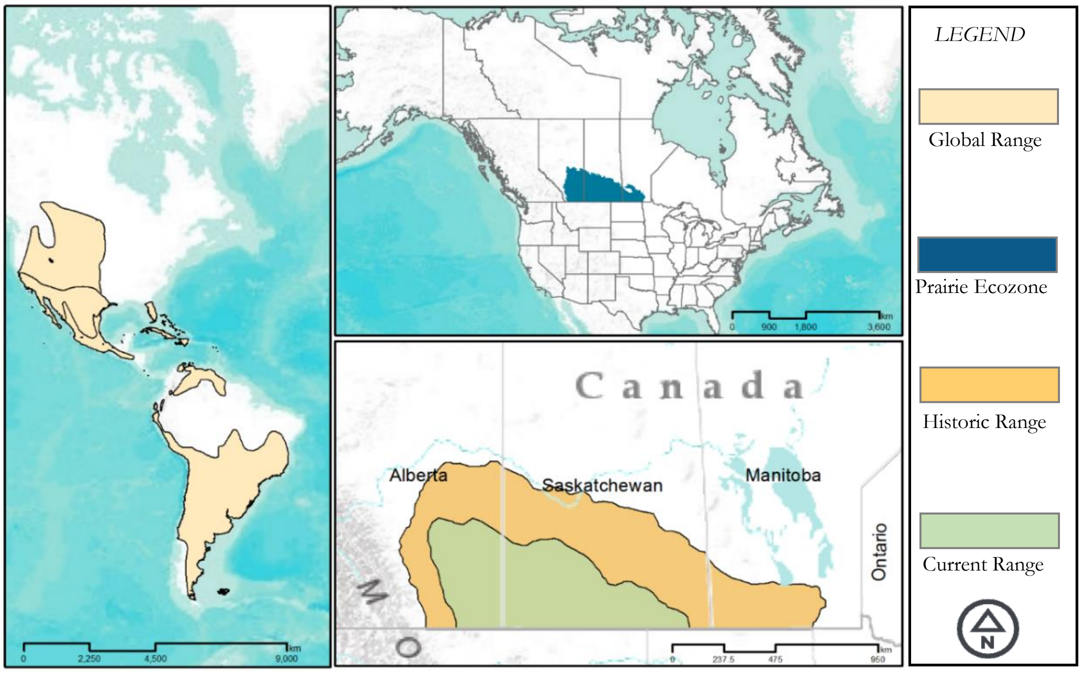

Figure 3.1 The Global and Canadian Ranges of Burrowing Owls

with freezing winters (-6 °C ≤ average temperature range ≤ -17 °C), warm moist summers (15 °C ≤ average temperature range ≤ 19°C), and variable average annual precipitation ranging from 288 mm/year to 540 mm/year (ESTR Secretariat, 2014). The varying degrees of moisture and high evaporation created a suitable condition for the growth of native temperate grass vegetation in the region (Samson et al., 2004).

The landscape was historically covered by large areas of mixed-grass, moist mixed-grass, as well as very small patches of fescue grass and tallgrass in the westernmost and southeastern part of the ecosystem (Askins et al., 2007; Doherty et al., 2017). Historically, natural fire regimes caused by lightning, periodic drought, and grazing by large mammals (e.g., Elk, Bison, Pronghorn) and small rodents (e.g., Black-tailed Prairie Dogs and Richardson's Ground Squirrel) were the major ecological processes involved in the formation and maintenance of the native vegetation across the landscape (Steinauer and Collins, 1996; Gauthier et al., 2003).

With much smaller shares of the natural land cover across the ecozone, wetlands and woodland areas are the other essential components of the prairie landscape. The Prairie Ecozone has millions of shallow ephemeral wetlands or potholes, which were formed during the last ice age due to retreating masses of subterranean ice (Askins et al., 2007). Accordingly, vast areas of the flat to rolling landscape of the Prairie Ecozone are also known as the "prairie pothole" region (Doherty et al., 2017). Despite covering only 3% of the area and the climate-dependency of these ephemeral potholes, the ecological role of these wetlands has been central to the regional biodiversity in this ecosystem (ESTR Secretariat, 2014).

Significant loss of wetlands, however, occurred since European settlers arrived in this area and continued ever since as more wetlands were drained for cultivation purposes (ESTR Secretariat, 2014). Woodlands are the other native land category across the prairie landscape and cover a very small proportion (≤ 5%) of this area. Despite the overall declines across the entire ecosystem, woodlands have shown small increase across some areas as a result of the altered fire regime and extirpation of large mammalian grazers (ESTR Secretariat, 2014).

Due to the natural fertility of the soils in the Prairie Ecozone, almost the entire landscape with suitable soil and slope condition has been converted from grass to grain or tame pasture (Askins et al., 2007). Approximately 70% of the native vegetation in the area was converted by the late 20th century (Riley et al., 2007), but the conversion rate has slowed down afterward. Despite the past conversion rates, land protection measures failed to protect much of the remaining native land cover in this vulnerable ecosystem. As such, only 4.5% of the remaining native land cover is situated within the boundaries of the protected areas in this ecosystem (ESTR Secretariat, 2014).

Among the three Prairie Provinces, Manitoba has the least grassland cover, and almost all the native grass species have been replaced with cultivated lands. Approximately 57% and 80% of the native grasslands had already been converted in Alberta and Saskatchewan, respectively (Samson and Knopf, 1994). The result of this conversion is a highly fragmented landscape, with the majority of the remaining native habitat patches having surface areas of less than 10 ha (ESTR Secretariat, 2014). The predominantly agricultural landscape of the Prairie Ecozone is dedicated to annual crop production (i.e., cereal grain and oilseed) with the remaining areas of native and tame grass utilized for livestock grazing and cattle production (ESTR Secretariat, 2014).

Historically, the natural periodic grazing of large herds of ungulates (e.g., bison, pronghorn) sustained the variability of grass species in the area. However, the overall vigor of the native grasslands had declined since the natural grazing regime was replaced with intensive annual livestock grazing, which, unlike natural grazing patterns across the landscape, is spatially confined to certain patches (Knopf, 1994). The native grassland patches with homogenous density and height are less likely to attract some species of grassland birds (Knopf, 1994; Robins and Dale, 1999) and sustain their population during periodic droughts (George et al., 1992).

Apart from the extensively cultivated landscape, roads, urban areas, and energy infrastructure are the other human land uses fragmenting the Prairie Ecozone. Rapid urban expansion has altered, what has historically been known as a rural landscape, to an urbanized ecosystem with multiple population centers, growing in size and spatial impact (ESTR Secretariat, 2014). Extensive road and railroad networks have been built to support the transport of people and freight between these growing urban centers (Thorpe & Godwin, 1999; ABMI, 2018).

The Prairie Ecozone is also a landscape rich in petroleum reserves. As such, extensive oil drilling projects have been central to anthropogenic disturbance to wildlife across the prairie landscape (e.g., Walker et al., 2007). Given the promising potential of this area for harnessing renewable sources of energy, the number of renewable energy projects such as wind and solar power has increased across this landscape. Further areas are also expected to be allocated for the development of renewable energy projects of this kind in the years and decades to come (Copeland et al., 2011).

3.2 The Modeling Framework

3.2.1 InVEST: A Spatially Explicit Decision Support System

The modeling framework utilized in this study to measure the landscape-level habitat quality for Burrowing Owls is the "Habitat Quality" module of the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) models. (Sharp et al., 2019). The InVEST modeling toolset is the product of the Natural Capital Project, a joint partnership between the World Wildlife Fund, the Nature Conservancy, Stanford University, and the University of Minnesota.

These models equate ecosystem services with environmental services and facilitate decision-making on natural resource management through mapping and quantifying the past, current, and potential future status of the natural environments and ecosystems. Using ecological production functions, this modeling framework considers both spatial and tabular data of land use/cover status, change, and management in conjunction with other environmental or, if applicable, economic information required to analyze single or multiple environmental services and processes, the synergistic association between these services, and potential tradeoffs among them (Tallis and Polasky, 2011; Sharp et al., 2019).

These models are a set of Decision Support Systems (DSS) which produce spatially explicit outcomes that could be interpreted through social, economic, environmental, or conservation perspectives for different points in time. (Tallis and Polasky, 2011). As such, experts, practitioners, managers, and other stakeholders could be integrated into the scenario development, parameterization, and decision-making stages of this modeling framework (Tallis and Polasky, 2011). Figure 3.2 illustrates the framework of the InVEST modeling toolset.

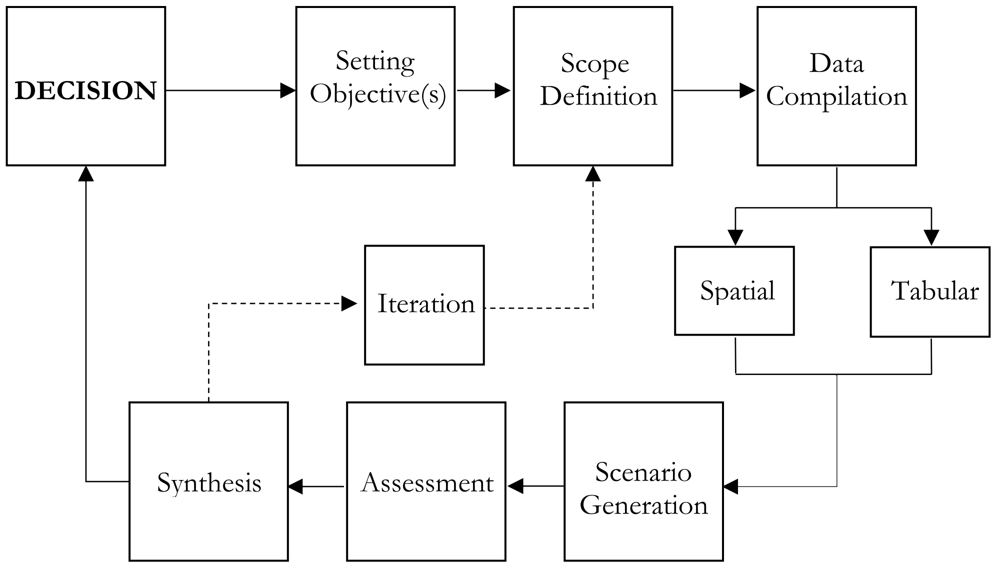

Figure 3.2 The Framework of the InVEST DSS

This toolset, according to Sharp et al. (2019), is categorized into four groups: I) The final ecosystem services assessment tools - which consider those provisioning and regulating environmental services with direct benefit to human beings, II) the supporting ecosystem service assessment tools (e.g., habitat risk assessment and habitat quality), III) tools to facilitate ecosystem service assessment, and IV) supporting tools (e.g. scenario generator tool).

3.2.2 The InVEST Habitat Quality Module: A Landscape-level Habitat Quality Model

As a supporting ecosystem assessment tool, the InVEST habitat quality module was developed based upon the habitat and habitat quality definitions by Hall et al. (1997). This model utilizes the information on land use/cover and spatial sources of disturbance (threats) to biodiversity to produce relative habitat quality maps under the environmental and anthropogenic sources of disturbance. Biodiversity in this context is not considered as an ecosystem service with embedded economic value. Rather, it is perceived as an independent variable of natural environments that has its own intrinsic values, including the conservation of ranges of genes, species, populations, or habitats (Sharp et al., 2019).

Depending on the scale of consideration, this assessment provides key insights into the relative extent and degradation of habitat in any terrestrial ecosystem at any given time for a target conservation objective, which can be a single species, group of species, or biodiversity in general (Polasky et al., 2011). More specifically, habitat quality in this context is considered as a proxy for more detailed measures of biodiversity. As a result, ecosystems with higher habitat quality will be representative of better living environments for the target conservation species (Sharp et al., 2019). This modeling framework, therefore, is particularly useful for making initial conservation assessments (Terrado et al., 2016).

The InVEST habitat quality framework represents a coarse-filter approach considering the landscape-level habitat assessment that uses the vegetation index as a basis to identify habitat suitability of different land use/cover for the target conservation objective (Sharp et al., 2019). While this approach looks over detailed species occurrence data, it represents unique simplifications and advancements concerning conservation and conservation-dependent decision-making practices across different landscapes (Sharp et al., 2019).

First, the model considers the sensitivity of species or their habitat to different spatial sources of disturbance, which is a variable factor when different land uses and taxonomic groups are considered. In addition, the model takes into account the relative impact of the spatial sources of disturbance on selected land use/cover categories or a given species. This, in turn, enables analysts to assess habitat quality results based upon the varying degrees of habitat degradation caused by different spatial sources of threat across the landscape. Another fundamental consideration in this modeling framework is the impact of the distance factor and the degree to which an ecosystem is degraded due to the proximity to spatial sources of disturbance. In this respect, proximity is considered as a critical determining factor for habitat degradation. Last but not least, the habitat quality model accounts for the degree to which the land is legally, socially, or physically protected and assumes that protection is effective in maintaining habitat quality, and thus persistence of the conservation targets in a given environment (Nelson et al., 2011; Sharp et al., 2019).

The model utilizes raster data of land use/cover where each cell is assigned to a unique class, varying based on the details of classification determined by the analyst. Following this classification, each land use/cover class is assigned a suitability score, which could take either a binary value (i.e., 0 = not suitable or 1 = suitable) or a range of suitability scores between 0 and 1, depending upon the modeling perspective (i.e., species-specific consideration or biodiversity in general) and the habitat preference by the subject of study (generalist species vs. specialist species). More specifically, if the data is limited or species of interest illustrate heavy reliance on specific land use/cover, then a binary approach would be an ideal way to illustrate habitat quality scores.

In contrast, generalist species such as Burrowing Owls might be able to use multiple patches of natural and semi-natural areas across a given ecosystem (Franklin and Lindenmayer, 2009). Thus, habitat quality modeling based on a range of values would be more appropriate for these species. When a continuum of suitability scores is considered, more areas could be subject to land management and conservation/restoration practices, and habitat quality scores will be illustrative of the combined impact of land protection practices and species' habitat preferences. In contrast, a binary consideration of habitat suitability dismisses the interrelationship of the habitat extent, importance, and suitability with its quality (Sharp et al., 2019).

The model also requires raster data of spatial sources of degradation, where each grid cell on the threat raster(s) is assigned a value of 1 or 0, indicating the presence or absence of the spatial sources of degradation, respectively. To produce habitat quality scores, the model first considers the cumulative impact of the spatial sources of degradation to determine habitat degradation level in each grid cell. This pixel-based degradation value is calculated by the following equation:

$$D_{xj} = \sum_{r=1}^{R} \sum_{y=1}^{Y_r} (W_r \,/\, \sum_{r=1}^{R} W_r)\, r_y i_{rxy} \beta_x S_{jr} \quad \text{(eq. 1)}$$

where D_{xj} represents the total threat level in grid cell x with land use/cover type j; r indicates each source of threat; y illustrates all grid cells on r's raster map, and Y_r denotes the set of grid cells on threat r's raster map.

Each threat's weight is denoted by W_r, which can take any value between 0 and 1; the higher the value of this parameter is the higher degree of degradation it causes. β_x represents the level of accessibility to the habitat as an indicator of its protection status. This parameter can take any value from 0 to 1, with values close or equal to 1 showing higher accessibility to the habitat and values close or equal to zero, illustrating lower accessibility to the habitat being studied. S_{jr} denotes the relative sensitivity of habitat or species to the spatial sources of disturbance.

This parameter, also, can take a range of values between 0 and 1. In this respect, the higher values illustrate higher sensitivity of the land use/cover or species of interest to the spatial sources of degradation. i_{rxy} illustrates the impact of threat source r, which originates in grid cell y, r_y, on habitat in cell x. This threat level is mediated by the distance between the source and the habitat cell and is calculated using either one of the following distance-decay equations:

$$i_{rxy} = 1 - (d_{xy}/d_{r\,max}) \quad \text{(eq. 2)}$$

$$i_{rxy} = \exp(-(2.99/d_{r\,max}))d_{xy} \quad \text{(eq.3)}$$

where d_{xy} is a linear distance between grid cells x and y, and the $d_{r\,max}$ is the maximum effective distance beyond which the impact of the spatial source of disturbance fades across space. The underlying assumption in equation 2 highlights a linear relationship between the impact emanating from a given source of threat and the maximum effective distance beyond which the impact of that threat dissipates across space. In contrast, equation 3 considers an exponential distance-decay function, where the impact of a given source of threat across space changes as exponents of its maximum effective distance.

It is important to note, however, that if the exponential function is utilized to describe the impact of degradation caused by sources of disturbance, then the model will disregard the i_{rxy} values that are too small or close to zero to expedite the modeling process. If the i_{rxy} values are greater than 0, then the habitat grid cell x is considered to be in threat r's disturbance zone (Sharp et al., 2019). The D_{xj} values are then converted into the habitat quality scores using the following equation:

$$Q_{xj} = H_j(1 - (D_{xj}^z/(D_{xj}^z + K^z))) \quad \text{(eq.4)}$$

where Q_{xj} is the habitat quality score in cell x that is situated in land use/cover j; H_j is an indicator of habitat suitability of the land use/cover at pixel j. Habitat quality scores have values between 0 and 1, where a higher habitat quality score in each cell indicates the lower degradation of habitat in that grid cell. If habitat quality scores are equal to zero, then either the land use/cover is not a suitable habitat or the habitat being assessed is totally degraded by the cumulative impact of the spatial sources of degradation. The k and z =2.5 are the scaling parameters (constants); the value for the parameter k is determined by the user but is typically set equal to half of the highest degradation score in habitat degradation map (Sharp et al., 2019). To perform this model calibration, the model needs to be run by the default value of k=0.5, and then a new k value will be determined based on the highest degradation score. The rank of grid cells in habitat quality maps is invariant of these changes as the value of k is only related to the spread and central tendency of habitat quality values.

It is important to note that if habitat quality models target specific species, all model parameters need to be considered for the target species only (Sharp et al., 2019).

3.3 Models and Scenarios

The habitat quality models in this study (Figure 3.3) were specifically developed to answer the research questions outlined in section 2.4. Accordingly, the cumulative impacts were modeled and mapped for all threats as well as three distinct combinations of threats (scenarios) across the historic and current ranges of Burrowing Owls using the habitat quality module of the InVEST 3.7.0 toolset.

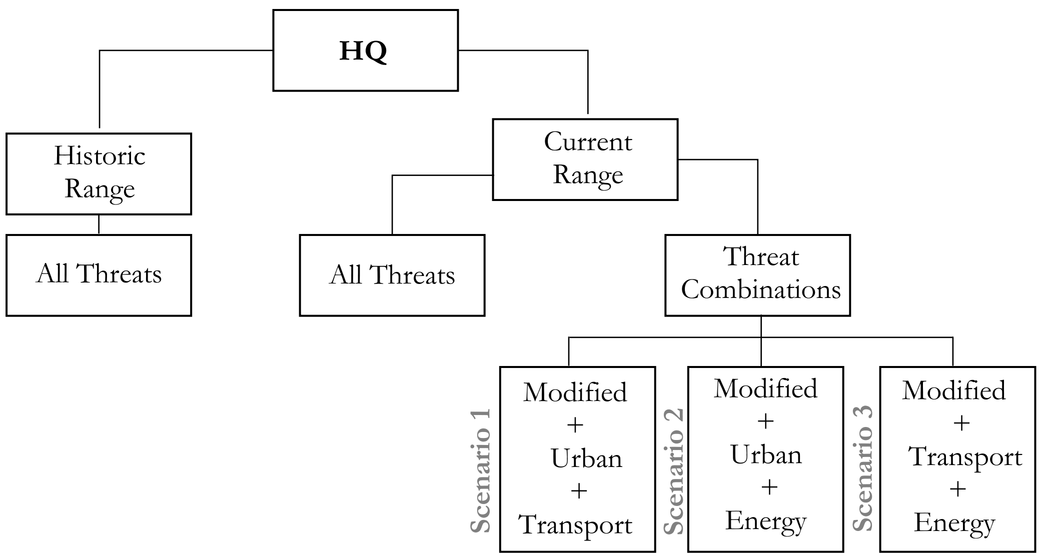

Figure 3.3 The Habitat Quality Models for the Burrowing Owls

In this respect, the habitat quality scores attained from the scenarios applied to the current range of Burrowing owls would be illustrative of key conservation requirements and priorities, as well as the impact of the different combinations of spatial threats on the overall habitat quality. The habitat quality scores acquired from the deterministic model run in the historic range would be illustrative of degradations made to the habitat when all sources of disturbance are considered concurrently across this spatial extent. This would be critically essential to determining the potential role of the considered sources of degradation in the contracted range of Burrowing Owls in the study area.

The spatial sources of disturbance in this model are those anthropogenic features or land use/cover categories causing direct disturbance to these species or to the ecological processes supporting these birds across both ranges. Therefore, the spatial data layers for agricultural fields, tame pasture, major roads, secondary roads, railroads, urban centers, active oil facilities, inactive or abandoned oil facilities, and wind turbines were utilized to perform different model runs across both ranges. These spatial data layers were classified into four general threat groups including

a) The Modified Landscape (Agriculture + Tame Pasture)
b) The Transportation Network (Major Roads + Secondary Roads + Railroads)
c) The Urban Centers (Urban Areas)
d) The Energy Infrastructure (Active Oil Facilities + Inactive Oil Facilities + Wind Turbines)

To compare habitat quality values under different scenarios across the current range of Burrowing Owls, the data layers in the modified landscape category was considered to be integral to different combinations of threat in the scenarios.

3.3.1 Spatial Data Layers Preparation

The land use/cover data map is the basic spatial data layer required by the InVEST habitat quality model. Accordingly, a reclassified version of the 2018 Annual Crop Inventory (ACI) data layer was clipped to the boundaries of the study area and utilized as the input land use/cover layer to the model. The ACI is the product of the science and technology branch of the Agriculture and Agri-food Canada (AAFC) and has the overall target accuracy of at least 85% at a final spatial resolution of 30 m (AAFC, 2019).

Given that both the historic and the current ranges of Burrowing Owls are located within the boundaries of the Prairie Ecozone, and to prevent inflated habitat quality scores at the edges of these ranges, the ACI layer was clipped by the Prairie Ecozone boundary using ArcGIS 10.7. Since there are no precedents of habitat quality mapping for Burrowing Owls at this spatial scale, the 2018 data layer was considered to be the baseline landscape-level modeling year for this species. As such, the reclassified version of the clipped raster data layer was the only layer utilized in different model runs for the considered scenarios. In this respect, the clipped ACI land use/cover data layer was reclassified into five distinct classes, namely grassland, pasture, cropland/fallow, wetland, and other based upon the habitat preferences of Burrowing Owls (Figure 3.4). The description of the reclassified land use/cover, as well as the assigned code to each class, are outlined in Table 3.1.

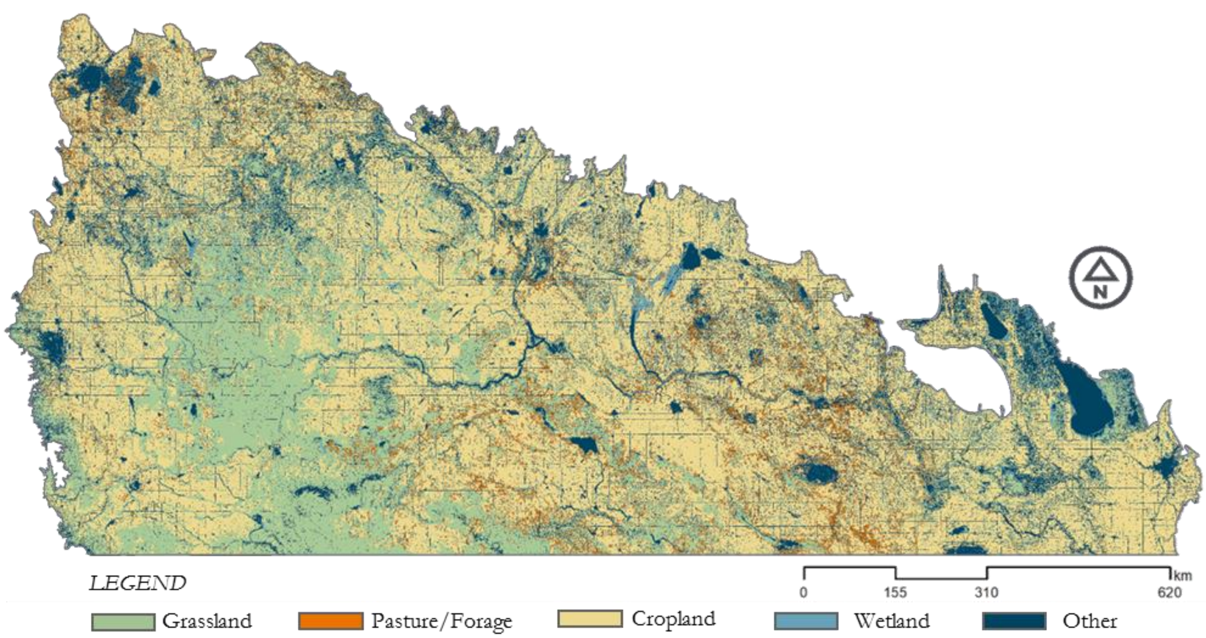

Figure 3.4 The Reclassified Land Use/Cover Input Data Layer

The model also requires a spatial data layer for the protected areas in the study area. Accordingly, a shapefile of the national protected areas was obtained from the Canadian Protected and Conserved Areas Database (CPCAD), and then clipped to the boundaries of the study area. This spatial data layer was compiled and managed by Environment and Climate Change Canada (ECCC) and contains information on terrestrial protected areas and other effective area-based conservation measures (ECCC, 2019).

Using the guidelines of the International Union for Conservation of Nature (IUCN) for categorizing protected areas (Dudley, 2008), the accessibility scores required by the model were assigned to each of the protected areas in the CPCAD data layer. The map of protected areas, as well as a table, including the name, description, and assigned accessibility values of protected areas, are included within the Appendix section of this thesis.

Table 3.1 The List and Description of the Reclassified Land Use/Cover Classes

Land use/cover class	Code	Description
Grassland	1	Includes predominantly native grasses and other herbaceous vegetation and may include some shrubland cover. The ACI code for this land cover is 110 in the 2018 data layer.
Pasture/Forage	2	Periodically cultivated land uses including tame grasses and other perennial crops such as alfalfa and clover grown alone or as mixtures for hay, pasture or seed. The ACI code for this land cover is 122 in the 2018 data layer.
Cropland/Fallow	3	Annually cultivated land uses (i.e., cereals, wheat, oilseeds), as well as vegetables and fresh products except those produced in greenhouses. In addition, other categories including too wet to be seeded and fallow areas are all considered as seeded landscape for agricultural production. The ACI code for this group of land uses is 130-197 in the 2018 data layer.
Wetland	4	Includes lands with a water table near/at/above soil surface for enough time to promote wetland or aquatic processes. The ACI code for this land cover is 80 in the 2018 data layer.
Other	5	Includes all land use/cover layers which are not generally considered habitat in the considered range. These are water, barren fields, urban and developed areas, greenhouse, shrubland, coniferous trees, broadleaf trees, and mixed wood areas. The ACI code of these layers are 20, 30, 34, 35, 50, 210, 220, and 230 in the 2018 ACI data layer.

Data layers for the spatial sources of degradation are the second group of data inputs required by the model. In this respect, the agriculture and tame pasture layers were extracted from the raster land use/cover data layer and considered as distinct spatial threat layers input to the model. The road network shapefile was acquired from the road network file of Statistics Canada for year 2018 (Statistics Canada, 2018), and then clipped by the boundaries of the Prairie Ecozone. This spatial data layer consists of five distinct road categories, which were divided into the major roads and secondary roads data layers, according to the street rank description within the national road network reference guide (Statistics Canada, 2018).

In this respect, the first three categories (i.e., Trans-Canada highway, national highway system, and major highways) were considered as the major road network layer, and the other two categories (i.e., secondary streets and all other streets) were classified as the secondary road layer input data to the model. The latest railroad network data was obtained from the Atlas of Canada National Scale Data source (Natural Resource Canada [NRCAN], 2014) and clipped to the boundaries of the Prairie Ecozone. All the transportation group layers were then converted to raster data layers input in ArcGIS, to be integrated into the modeling framework.

The urban areas data layer was obtained from the 2016 census boundary files (Statistics Canada, 2019), which encompasses the latest boundaries of population centers and rural areas classification in Canada. This shapefile was then clipped to the boundaries of the study area and converted to the raster data file required by the model runs. The location of wind farms in Canada was extracted from the Renewable Energy Power Plants, 1MW or more – North American Cooperation on Energy Information (NRCAN, 2018), and the exact locations of the wind turbines were digitized accordingly.

In this regard, the maximum area occupied by each turbine pad was considered to be a circular buffer equal to 581 m^2, an area of a wind turbine pad with a 27 m diameter (Gipe, 1993). The data layers for petroleum energy facilities were obtained from the provincial resources (Alberta Energy Regulator [AER], 2019; Government of Saskatchewan, 2019; Government of Manitoba, 2019), and then reclassified into the active oil facilities and inactive oil facilities (suspended and abandoned) data layers in the study area. The surface area of the considered petroleum facilities was set to the average lifetime area of oil well drilling pads, which is equal to 10500 m^2 (Buto et al., 2010). Accordingly, a circular buffer area of 58 m was considered for both oil facility classes in the study area. All the spatial sources of disturbance are illustrated in Figure 3.5.

3.3.2 Model Parameterization

The habitat suitability of the considered land use/cover classes in this study was equated to the definition of habitat preference by Hall et al. (1997). To determine the suitability scores, 17 post-1980s studies were selected and reviewed based on the habitat preference of Burrowing Owls across the delineated land use/cover classes. Among these studies, 14 assessed habitat use of Burrowing Owls within the Canadian breeding grounds in Alberta, Saskatchewan, or both of these Prairie Provinces.

These studies are based on the recorded occurrence incidents of Burrowing Owls or the likelihood of their occurrence across the delineated land use/cover classes. Habitat suitability scores were determined according to the reported occurrence incidents or likelihoods of occurrence in different classes. These reports were classified into two groups: a) habitat use for nesting and roosting purposes and b) habitat use for foraging and loafing purposes.

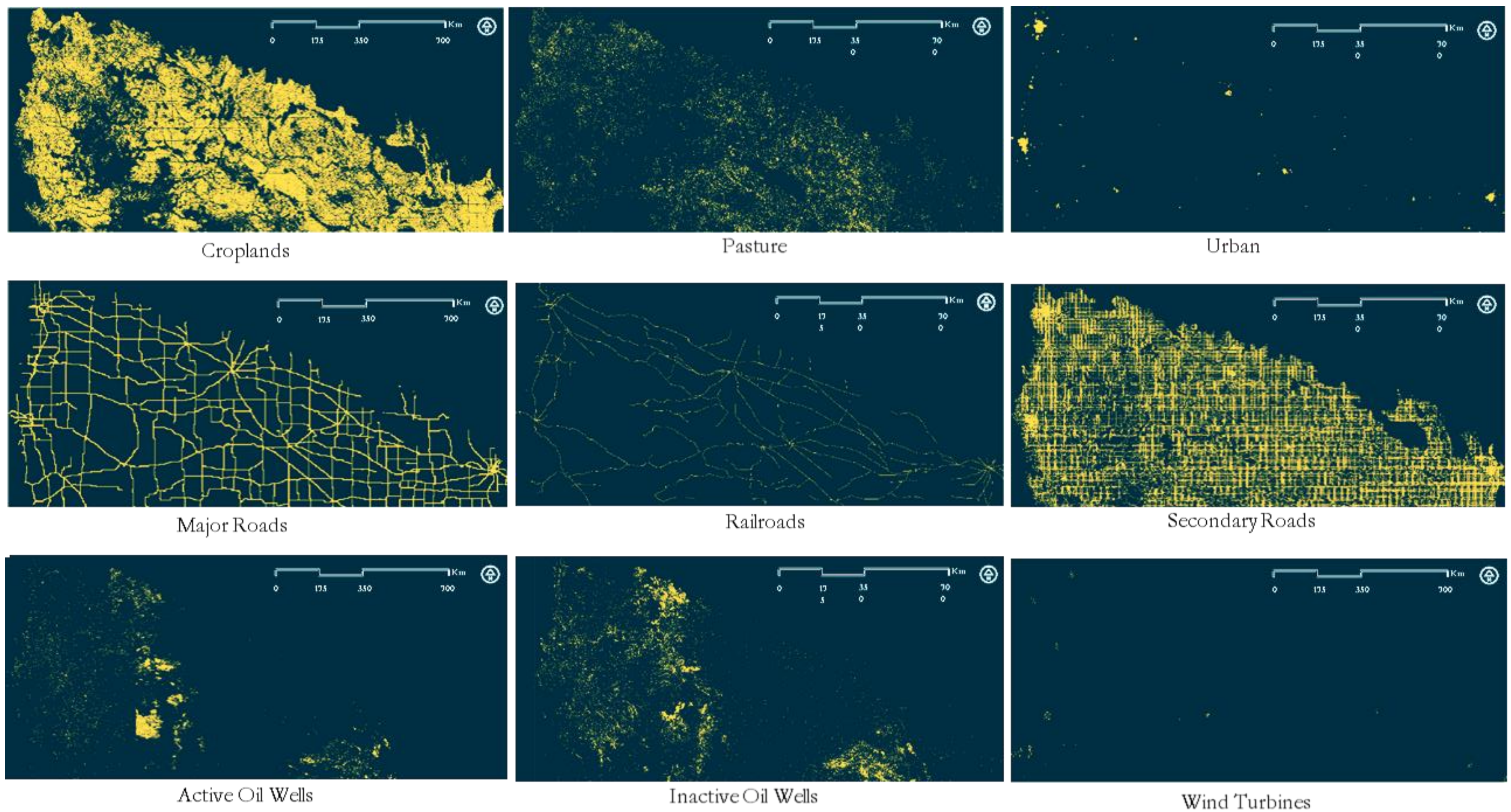

Figure 3.5 The Spatial Sources of Disturbance in the Prairie Ecozone

Accordingly, the percentage of studies reporting the occurrence of these species across individual land use/cover categories was calculated for each group separately, and then averaged among the two habitat groups to determine the habitat suitability of each category for these bird species. For instance, grassland areas were utilized in 92% of the studies assessing the nesting and roosting behavior of Burrowing Owls and in 100% of the studies reporting the occurrence of these species for foraging and loafing activities. Habitat suitability of the grasslands, consequently, was calculated (averaged) as 96%, which is equal to 0.96 within the [0 1] score continuum. In this approach, classes that are underutilized in relation to their abundance were not taken into consideration. The calculated habitat suitability scores along with the considered studies are listed in Table 3.2.

The maximum effective distances were determined for the delineated spatial sources of degradation from the literature on Burrowing Owls. In this regard, the regulations or recommendations on appropriate buffer distance was considered as an initial factor for setting the distance parameter in the model. However, if buffer distances did not apply to the identified sources of degradation, the maximum effective distance would be determined based on the distance beyond which Burrowing Owl's perception of sensory disturbance fades away across space.

However, the underlying assumption in this case was that this distance must reach beyond the average diurnal home range radius of 0.25 km for these species. If none of these initial conditions applied, then the maximum effective distance would set equal to the average diurnal home range radius of Burrowing Owls. It was also assumed that the threats emanating from all the identified sources of degradation follow a linear distance-decay function.

Table 3.2 The List of Habitat Suitability Scores and the Studies Reviewed to Determine the Suitability Scores

Behavior									Nesting and Roosting									
Study	Study1	Study2	Study3	Study4	Study5	Study6	Study7	Study8	Study9	Study10	Study11	Study 12	Study 13	Study 14	Study 15	Study 16	Study 17	Suitability
Location	CO	NE	AB-SK	SK	SK	SD	SK	AB	SK	SK	SK	SK	AB-SK	AB-SK	AB-SK	AB	SD	
Land use/cover																		
Grassland	*	*	*	*	*	*	*	*	*		*	NA	NA	NA	*	*	*	0.92
Pasture			*		*		*		*	*	*	NA	NA	NA	*	*	*	0.64
Cropland/Fallow								*				NA	NA	NA		*	*	0.21
Wetland												NA	NA	NA				0
Other												NA	NA	NA				0
Behavior									Foraging and Loafing									
Land use/cover																		
Grassland	NA	NA	*	NA	*	NA	*	*	*	NA	*	*	*	*	NA	*	NA	1
Pasture/Forage	NA	NA	*	NA		NA	*			NA	*	*	*	*	NA	*	NA	0.7
Cropland/Fallow	NA	NA		NA		NA		*		NA	*		*	*	NA	*	NA	0.5
Wetland	NA	NA		NA		NA				NA	*		*	*	NA	*	NA	0.4
Other	NA	NA		NA		NA				NA					NA		NA	0

Land use/cover	Grassland	Pasture	Cropland/Fallow	Wetland	Other
Final Suitability Scores	0.96	0.67	0.35	0.2	0

List of Studies: Study 1 (Plumpton and Lutz, 1993), Study 2 (Fannes and Lingle, 1995), Study 3 (Clayton and Schmutz, 1999), Study 4 (Poulin et al., 2005), Study 5 (Haug, 1985), Study 6 (MacCracken et al., 1984), Study 7 (Haug et al., 1993), Study 8 (Schmutz, 1997), Study 9 (Wedgwood, 1976), Study 10 (James et al., 1990), Study 11 (Haug and Oliphant, 1990), Study 12 (Sissons et al., 2001), Study 13 (Marsh et al., 2014a), Study 14 (Marsh et al., 2014b), Study 15 (Clayton, 1997), Study 16 (Stevens et al., 2010), Study 17 (Thiele et al., 2013)

The US states and Canadian Provinces listed in Table 2: CO: Colorado, NE: Nebraska, SD: South Dakota, AB: Alberta, SK; Saskatchewan

The impacts of the spatial sources of disturbance were directly derived from the threat assessment worksheet enclosed within the latest status report for these endangered species (COSEWIC, 2017). Within this framework, the threat impact for each spatial source of degradation was calculated considering the scope (i.e., the proportion of species or ecosystems that could be affected by the spatial sources of disturbance within a decade) and severity (i.e., the level of damage to the species or ecosystems within the three-generation period) of different sources of degradation (IUCN, 2017; COSEWIC, 2017).

Since land modifications (i.e., the application of pesticides and rodenticides) occur across the cultivated landscape, the threat scores from this source of threat were taken into account for the agricultural fields, as well as the pasture areas. All of the spatial sources of degradation are listed in Table 3.3, along with the corresponding maximum effective distance and threat impact scores.

Table 3.3 The Maximum Effective Distance (km) and Threat Impact Scores for the Considered Sources of Degradation

Threat Group	**Modified Landscape**		**Transportation**			**Urban**	**Energy**		
Spatial Source of Degradation	Agriculture	Pasture	Major Roads	Secondary Roads	Railroad	Urban	Active Oil	Inactive Oil	Wind Turbines
Maximum Effective Distance	0.5	0.5	1	0.25	0.25	0.25	0.25	0.25	0.25
Impact Level	High	High	Medium	Medium	Medium	Negligible	Negligible	Negligible	Medium
Assigned Impact Score	0.8	0.8	0.6	0.6	0.6	0.2	0.2	0.2	0.6

3.3.3 The Elimination Method: A Simple Multi-Criteria Decision Analysis Rule

Complex environments typically represent challenges that need to be addressed through more than one criteria at the same time. That is, the order of preferences for available alternatives should be defined considering a variety of criteria defined by analyst(s) (Radford, 1989). Multiple Criteria Decision Analysis (MCDA) techniques are a suite of decision making approaches developed based upon the coexistence of multiple criteria and different alternatives from which decision makers need to choose the most preferred one(s) by reference to an explicit set of pre-defined objectives, determined on a common set of values.

In the Multiple Criteria Decision Analysis (MCDA), the combination of three main factors needs to be considered by analysts before a suitable technique is employed to address the problem being studied. These factors are a) availability of numeric measures of progress for assessing the considered criteria, b) whether or not the same measurement unit is available for the assessment, and c) whether the relative prioritization of criteria could be presented using a numerical format or ordinal preferences should also be considered (Radford, 1989). Central to all MCDA techniques is the judgment of analyst(s) in determining the objectives, criteria, relative weighting system, and the degree to which the alternatives meet the defined criteria. Although this might raise questions on the subjectivity of the MCDA approaches, these techniques are open to adjustments by the discretion of analyst(s) or decision makers(s), and upon the availability of more and better information.

The common element of all MCDA techniques is the performance matrix (Dodgson et al., 2009). This matrix consists of rows and columns including the criteria and options considered in a decision situation. The entries of the matrix cells are the performance levels for each option against the defined criteria. These performance levels are more often assessed through numerical (cardinal) factors, but may take non-quantitative representations such as color coding or binary categorization. The majority of the MCDA techniques (e.g., linear additive models, analytical hierarchy process) convert these matrices into consistent numerical values using approaches referred to as compensatory methods or the combination of scoring and weighting analyses applied to the options and criteria in the performance matrix, respectively (Dodgson et al., 2009). Thus, most MCDA techniques vary from one another based upon the numerical operations applied to the elements of the matrix.

In many decision situations, however, the progress of the available alternatives towards considered criteria cannot entirely be measured by quantitative factors. Sometimes, this issue could be addressed by assuming proxy measures for some factors that are not easily quantifiable. But there are instances where, due to the deficit of suitable numeric weighting framework, priorities amongst the considered set of criteria could be presented only at ordinal scale (MaCrimmon, 1973). In these decision situations a suite of qualitative techniques (Nijkamp and Van Delft, 1977), such as qualitative outranking approaches can be utilized to find best alternatives or to rank alternatives in order of importance.

Unlike compensatory MCDA techniques, which require numerical operations on performance matrices, these matrices could also be considered independently as a basis for judgment of alternatives. In other words, the dominance of options could be determined using non-compensatory approaches, where the ranking procedure is done considering no external numeric factor and by simply taking into account new threshold levels for different criteria or through the perceived importance of criteria and elimination of options one at the time in an iterative process until the most suitable alternative is determined. The former is achieved using a non-compensatory method known as conjunctive and disjunctive selection procedure and the latter is performed using another method of this kind, known as lexicographic ordering (Dodgson et al., 2009).

The MCDA method utilized in this study is a variant of the Elimination by Aspects, which is a combined non-compensatory method constructed based upon the selection procedure and lexicographic ordering (Dodgson et al., 2009). The Elimination method is a simple MCDA rule that could be utilized when ordinal scales are also integral to the criteria assessment procedure (MaCrimmon, 1973; Radford, 1989). It is based upon ranking a set of alternatives considering their contribution to individual criterion developed from the available factors in a multi-criteria decision environment. When ordinal-scale assessments are required, each of the considered alternatives is assessed in terms of a letter grade (e.g., A, B, C etc.) or its extension (e.g., A- or B+ etc.), representing the degree to which that alternative meets a criterion defined for the assessment (Please see Table 2 in the Appendix).

By placing different alternatives in the order of preference against the list of criteria through sequential evaluation, analysts would be able to rank the alternatives. This is particularly useful in practical decision situations, where identifying a dominant alternative is not possible due to the complexities and the interrelationship between the assessment criteria. For instance, in real-life situations, alternatives that highly satisfy a single criterion might score poorly in some other criteria or very high in a criterion which contradicts with the initial factor being considered (Radford, 1989).

To evaluate the alternatives against the considered criteria, a minimum or maximum threshold, also known as the performance level, can be utilized through the principal objectives upon which the factors are defined. Starting from the top factor, alternatives that do not meet the performance levels at each row are eliminated using a cross (×) sign in the corresponding cell (please see Table 3 in the Appendix). This way, all the alternatives could be ranked sequentially. Accordingly, the alternatives which are eliminated when measured against the higher performance levels are the least preferred ones, and those eliminated upon assessment by a lower priority performance level are of the higher preference.

The rank of those alternatives that fail in the same initial performance levels is determined based on the total number of eliminations. If this procedure fails to determine the dominance of some alternatives over others, the performance levels can be changed by analyst(s) - to create a new threshold level- and ranking procedure should be repeated (Radford, 1989). Some decision situations might involve tradeoffs or synergies between different factors. That is, the lower performance level in one factor could be compensated by a higher level performance in a less preferred factor.

Alternatively, the influence of a criterion may be complemented by another criterion or a set of criteria, or it might be dependent on or independent from the other assessment criteria.

To define new threshold levels which accounts for these synergistic (or tradeoff) associations, considered criteria can be connected by conjunctive, disjunctive, and conditional linkages, represented by "AND", "OR", and "IF" statements, respectively (Please see Table 4 in the Appendix). In this respect, all linked criteria must fulfill the described performance level when the conjunctive (AND) statement is considered. The disjunctive statement (OR), however, only requires one of the linked criteria to be assumed credible, and the conditional statement (IF) can be assumed credible if the performance level meets the condition laid out by the analyst (Radford, 1989).

3.3.4 Applying the Elimination Method to Parameterize the Sensitivity Scores in the InVEST Habitat Quality Model for Burrowing Owls

The relative sensitivity of land use/cover or species to the spatial sources of disturbance is the final parameter required by the InVEST habitat quality model. Given that this concept can also be inferred from an opposite term, the "resistance" when species are the target conservation objective (Nelson et al., 2011), the relative resistance of species to the spatial sources of disturbance can also be considered to ascertain the relative sensitivity scores required by the model.

Accordingly, the direct and indirect impacts of the identified spatial sources of disturbance on Burrowing Owls and on the ecological processes these birds rely on to survive in the environment were considered to determine the relative sensitivity scores specific to these endangered species. This was done using the elimination method where the spatial sources of disturbance were ranked against the set of criteria developed based on the ecological processes and other environmental factors influencing these bird species.

The spatial sources of disturbance were thereupon set to be the alternatives the priority of which was determined through a range of assessment criteria determining the direct and indirect sensitivity levels in the environment being studied. These criteria were classified into five overarching groups considering the literature on Burrowing Owls' use of habitat, dependency on other species, and persistence in the face of the anthropogenic sources of disturbance. These groups are loss of burrows, landscape fragmentation, collisions, landscape modification, and sensory disturbance. To assess the factors within each group, an ordinal-scale assessment measure was utilized whereby accordance and discrepancy of the listed alternatives and defined criteria were determined using letters "Y" and "N", respectively (Table 3.4).

Table 3.4 The Considered Criteria and the Performance of the Spatial Sources of Disturbance

Sensitivity Criteria/Spatial Sources of Degradation	**Cultivated Landscape**		**Transportation**			**Urban**	**Energy**		
	Agriculture	Tame Pasture	Major Roads	Secondary Roads	Railroad	Urban	Active Oil Wells	Abandoned oil Wells	Wind Farms
Loss of Burrows									
Direct Destruction of Burrows	Y	N	N	N	N	N	N	N	N
Declined Population of the Prairie Dogs	N	Y	N	N	N	N	N	N	N
Declined Population of at least one other associated burrowing mammal (Squirrel & Badger)	Y	N	Y	N	N	N	N	N	N
Fragmentation									
Native Habitat Removal	Y	Y	Y	Y	Y	Y	Y	Y	Y
Increased Predation of Owls	N	N	Y	Y	N	N	N	N	N
Extended Home Range	N	N	N	N	N	N	N	N	N
Collisions									
Fatal Collisions of Owls	N	N	Y	Y	N	N	N	N	Y
Landscape Modification									
Declined Prey Population	Y	Y	N	N	N	N	N	N	N
Increased Indirect Poisoning	Y	Y	N	N	N	N	N	N	N
Declined Nest Success	Y	N	N	N	N	N	N	N	N
Declined Reproductive Success	Y	Y	N	N	N	N	N	N	N
Sensory Disturbances									
Source of Audio Disturbance	N	N	Y	Y	N	N	Y	N	N
Source of Visual Disturbance	N	N	Y	Y	N	N	N	N	Y

In cases where the association between the listed threats and criteria was missing from the literature, no accordance was considered. In this assessment scheme, supplementary information was also utilized to support the determined accordance and disparity between the identified threats and the defined criteria (Watson, 2005; Tuckwell and Everest, 2009; COSEWIC, 2011; COSEWIC, 2012; Roch and Jaeger, 2013; Cabrera-Cruz and Villegas-Patraca, 2016; Watson et al., 2018), and to ascertain if there is an association between the listed sources of degradation in terms of their potential impact on Burrowing Owls and the ecological processes these endangered bird species depend upon to persist in the study area (Barrientos et al., 2019).

Criteria within each group were then linked with each other or with the criteria from other groups using conjunctive, disjunctive, and conditional linking statements to define new threshold levels against which the threat alternatives were assessed. These performance levels were used to eliminate the spatial sources of disturbance sequentially (Table 3.5).

Table 3.5 The Linked Criteria for the Elimination of Spatial Sources of Disturbance and the Relative Sensitivity Scores

Threat Groups	Modified Landscape		Transportation			Urban	Energy		
Spatial Source of Degradation	Agriculture	Tame Pasture	Major Roads	Secondary Roads	Railroad	Urban	Active Oil	Inactive Oil	Wind Turbines
Linked Assessment Criteria									
Lead to the direct destruction of burrows AND affect prey availability		×	×	×	×	×	×	×	×
Lead to the decreased population of the prairie dogs OR at least one other associated burrowing mammal				×	×	×	×	×	×
Cause increased predation OR fatal collisions of owls	×	×			×	×	×	×	
Cause increased indirect Poisoning			×	×	×	×	×	×	×
Lead to the declined reproductive success. IF not, it should lead to the declined nest success			×	×	×	×	×	×	×
Lead to extended home range OR native habitat removal									
Serve as a source of Audio OR Visual Disturbance	×	×			×	×		×	

Spatial Source of Degradation	Agriculture	Tame Pasture	Major Roads	Secondary Roads	Railroad	Urban	Active Oil	Inactive Oil	Wind Turbines
Ranking	1	3	2	4	6	6	5	6	4
Relative Sensitivity Scores	1	0.66	0.86	0.5	0.16	0.16	0.33	0.16	0.5

The spatial sources of degradation were then ranked contrary to their elimination stage. However, it is critical to note that within this ranking process, higher ranks were considered to be representative of higher sensitivity of Burrowing Owls to the sources of degradation. Accordingly, six ranks were assigned to the threats as some of them were eliminated at the same stage of assessment. Finally, considering the number of ranked positions, the relative sensitivity scores were assigned to each source of degradation using its relative position in the final ranking. For instance, railroads, urban areas, and inactive oil facilities were determined as the sources of perturbation to which Burrowing Owls illustrate the highest resistance or lowest sensitivity level. Consequently, these group of threats were assigned a ratio of 1/6, which is equal to the relative sensitivity score of 0.16 in the [0 1] scores continuum.

3.4 Data Analyses

As discussed in section 3.2.2, the outputs of this modeling process are habitat quality maps rendered across the study area with every single pixel describing the habitat quality values calculated based on eq. 4. The results of the study are investigated through appropriate statistical analyses, using the IBM SPSS Statistics, along with an index used as a benchmark to assess the relative degradation caused by different sources of disturbance to habitat quality values. More specifically, the former method of analysis is used to measure the statistical significance of differences in habitat quality values between the historic and current ranges of Burrowing Owls, and the values attained under the three defined scenarios across the current range of these endangered species. The relative destruction caused by the sources of disturbance can be inferred from the following formula:

$$HQ_{\text{change index}} = (\mu_{\text{HQ Scenario } i} - \mu_{\text{HQAll}}) / \mu_{\text{HQAll}} \quad \text{(eq. 5)}$$

where $HQ_{\text{change index}}$ illustrates the relative change of habitat quality values across the landscape when the values in each scenario are compared to habitat quality values when all sources of degradation are considered concurrently across the landscape.

This benchmark was developed based on the concept of Ecosystem Change Index, which is used to measure the temporal changes of a given ecosystem service at time X with reference to its state in a baseline time (I) across a given landscape (Matlock and Morgan, 2011). However, instead of a temporal scale, eq.5 takes into account the mean habitat quality results in each scenario ($\mu_{HQ\ Scenario\ i}$) and compares it to the mean habitat quality values when all sources of degradation are considered (μ_{HQAll}). Accordingly, the higher this index is for each scenario, the higher the relative habitat quality will be under that scenario. Since each scenario was constructed by excluding one of the spatial threat data layer groups, higher habitat quality change index values in a given scenario will be indicative of the higher degradation caused by the spatial threat data layer missing from that scenario.

Chapter 4. Results

4.1 Variations of Habitat Quality between the Historic and Current Ranges of Burrowing Owls

The quality of habitat for Burrowing Owls at every single pixel of the input land use/cover data layer was calculated considering the assigned values to model parameters and mapped into the habitat quality raster subsequently. The historic and current ranges of Burrowing Owls were then extracted from this output data layer to illustrate the habitat quality values across both ranges (Figure 4.1).

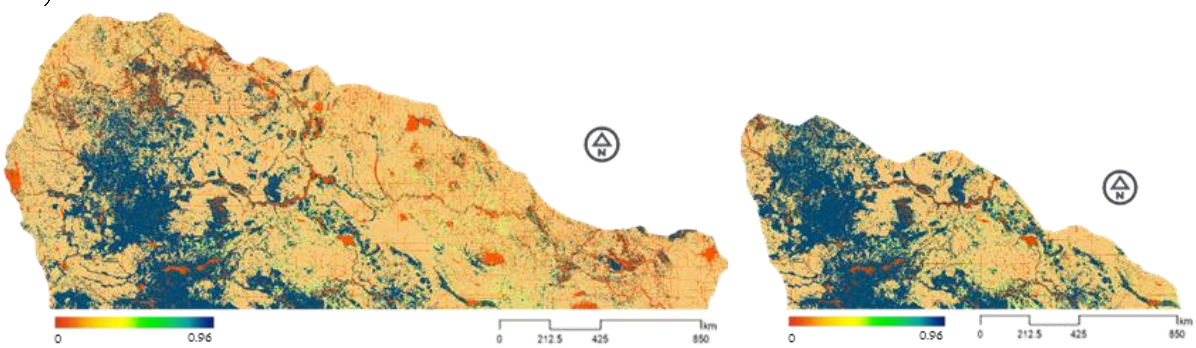

Figure 4.1 The Habitat Quality Maps across the Historic and Current Ranges of Burrowing Owls

Figure 4.2 illustrates the relative distribution of habitat quality values across the historic and current ranges of Burrowing Owls. The two distributions are relatively similar, illustrating multiple peak points (i.e., modes) accumulating around certain ranges. These ranges belong to the delineated land use/cover categories in the area, defined as a factor of their suitability level considered in the model.

As such, when the most unsuitable areas (i.e., $H_j=0$ *and* $HQ=0$) are considered, the proportion of the landscape which has an absolutely zero habitat quality value is much higher within the historic range of Burrowing Owls than the proportion of non-habitat areas across the current range of these species. At the other end of the habitat quality spectrum, however, the proportion of highly suitable habitat areas are much higher in the current spatial extent, compared to historic range of these species. Unlike the distribution of habitat quality values at the two ends of the continuum, the proportion of habitat with low to moderate quality is similar across both ranges and only illustrates slight differences (≤4%) between the two spatial extents.

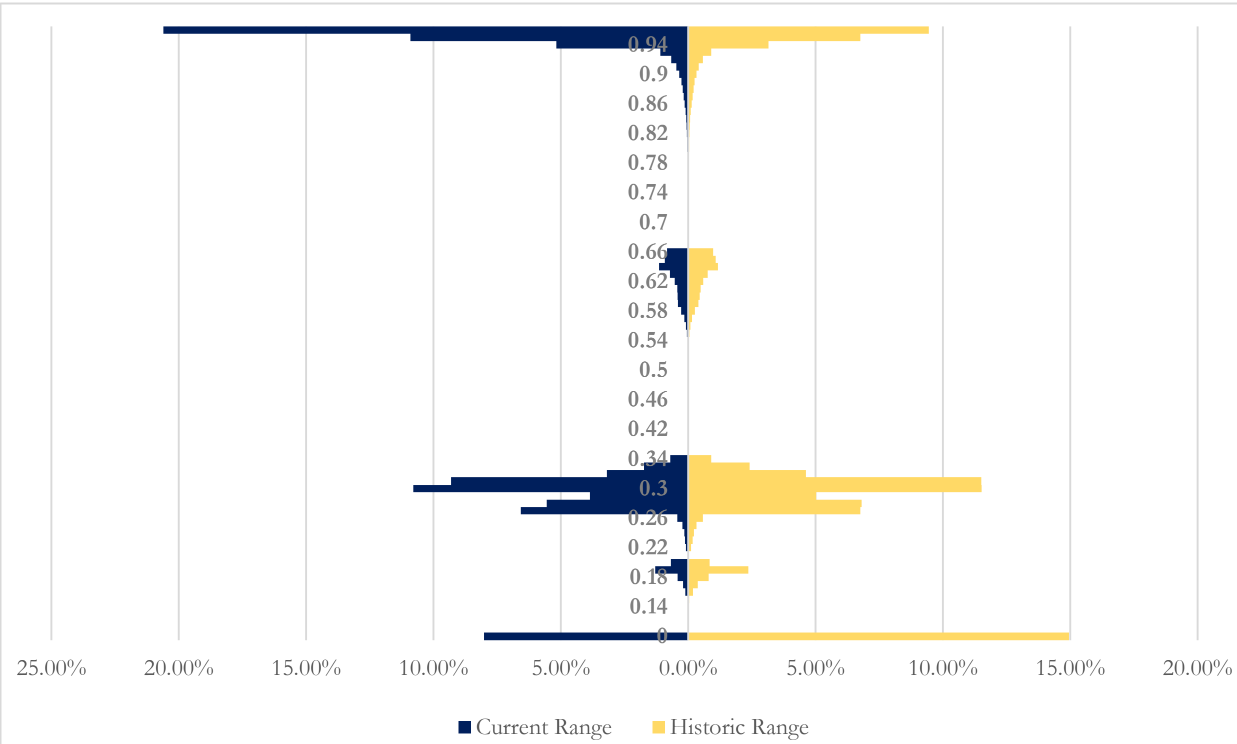

Figure 4.2.The Relative Frequency of the Habitat Quality Values

To determine if there is statistically significant difference in the habitat quality values, as a continuous dependent variable between the two ranges, the ranked-based non-parametric Mann-Whitney U test was performed over random samples (*n=100*) of the pixel groups at each of the ranges considered in the study (Table 4.1).

The result of this test illustrated that despite the differences in the habitat quality values between the historic (*Mdn=0.56*) and current (*Mdn=0.57*) ranges of Burrowing Owls, these differences cannot be considered statistically significant (*P=0.424*), thus resulting in the rejection of an alternative hypothesis.

Table 4.1 The Result of the Mann-Whitney U Test

Null Hypothesis	Test	Sig.	Decision
The distribution of HAQ is the same across categories of scenarios	Independent-Samples Mann-Whitney U Test	0.424	Retain the Null Hypothesis

The significance level is 0.05

4.2 Variations of Habitat Quality under Different Scenarios in the Current Range

4.2.1 Assessment of Habitat Quality across the Landscape

A One-Way ANOVA test (Table 4.2) was performed to determine if habitat quality values differ significantly between the three scenarios (groups) in the current range of Burrowing Owls. Accordingly, random samples of size 20 ($n=20$), representing different groups of pixels from each scenario were selected to perform the analysis.

Table 4.2 The Result of the One-Way ANOVA Test between Different Scenarios

HQ	Test Statistic (F-Test)	Sig.
Between Groups	2.707	0.075

There were no outliers among these samples and the data was not normally distributed for each scenario as assessed by the Shapiro-Wilk test ($p<0.05$). In addition, there was homogeneity of variances as assessed by Levene's test of homogeneity of variances ($p=0.309$). Despite the violation of normality, the results of One-Way ANOVA was considered as this test is fairly robust to deviations from normality, particularly when sample sizes are smaller than 50 (Lix et al., 1996), which is the case in this study. The results of this statistical test illustrated that the difference in habitat quality values between the three considered scenarios was not statistically significant ($p>0.05$). The habitat quality maps for these scenarios are illustrated in Figure 4.3.

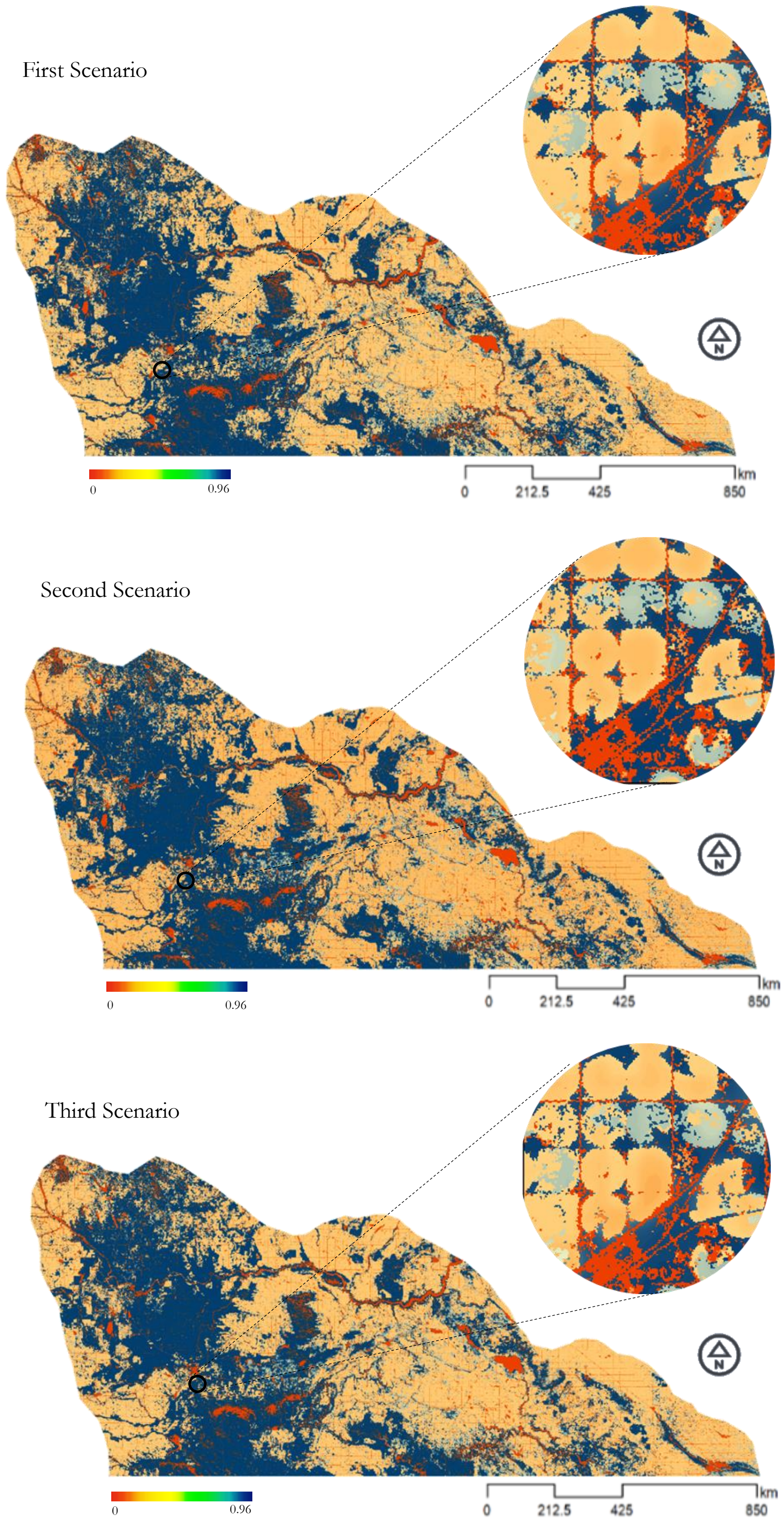

Figure 4.3 The Habitat Quality Maps under the Three Scenarios

Despite the rejection of the alternative hypothesis on the existence of statistically significant differences between the mean habitat quality values among the selected samples, using the habitat quality change index in eq. 5, the mean values in each scenario was compared to the mean habitat quality of the current range when all sources of degradation are considered in this spatial extent.

The change index was calculated as 0.05, 0.09, and 0.04 for the first, second, and third scenarios, respectively. These numbers suggest that the transportation data layer group, followed by the energy and urban data layers, result in the highest degrees of degradation to the current range of Burrowing Owls, respectively. The results of the habitat quality change index are plotted as a radar chart in Figure 4.4.

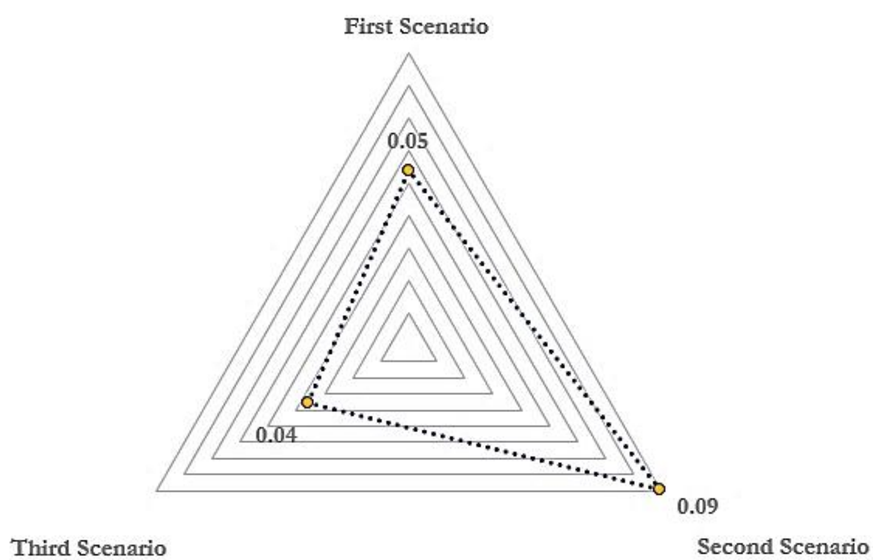

Figure 4.4 The Habitat Quality Change Index for Different Scenarios

4.2.2 Assessment of Habitat Quality Results in different Land Use/Cover Categories

In addition to testing the statistical significance of the differences in the mean habitat quality values under the three considered scenarios, a more-in-depth analysis was performed on the variant habitat quality results across the specific land use/cover categories in the current range of Burrowing Owls.

Accordingly, four different One-Way ANOVA tests were performed on samples of size 20 (*n=20*), which unlike the samples in the ANOVA test in the preceding section, were randomly selected from the specific land use/cover categories in the study area. The results of the performed tests for all categories were not statistically significant at $\alpha=0.05$. For the Grassland and Pasture categories, the results of the Welch ANOVA tests were reported as the homogeneity of variances was violated in the samples selected from these categories, according to Levene's test for equality of variances ($p<0.05$). The results for these statistical tests are listed in Table 4.3.

Table 4.3 The Results of the ANOVA Tests for the Specific Land Use/Cover Categories

HQ _Grassland	Test Statistic	Sig.
Welch	0.115	0.892
HQ_Cropland		
ANOVA-Between Groups	0.511	0.603
HQ_Pasture		
Welch	0.425	0.657
HQ_Wetland		
ANOVA-Between Groups	0.839	0.438

To further investigate the impact of different sources of degradation on each category, the Box and Whisker Plots of the habitat quality values were considered for different land use/cover categories (Figure 4.5). As such, the impact of energy, transportation, and urban data layers on habitat quality across each of the classes was determined from the analyses of the first, second, and third scenarios, respectively.

The habitat quality values under the second scenario were the highest among the three considered scenarios across each of the land use/cover categories in the study area. The values under this scenario illustrated the highest means and minimum variations (SD) across all categories (i.e., 0.87 ± 0.06 in grasslands, 0.3 ± 0.02 in croplands, 0.61 ± 0.04 in pasture areas, and 0.18 ± 0.02 in wetlands). Consequently, the spatial sources of degradation in the transportation data layer have the highest degradation impact on the patches of habitat at different categories.

Among the other two scenarios, the first scenario represented slightly higher mean habitat quality values and lower variations than the third scenario, meaning that the considered energy sources of disturbance degrade habitat more than the urban centers across the different land use/cover categories in the current range of Burrowing Owls.

Figure 4.5 The Box and Whisker Plots of the HQ Values across the Specific Land Use/Cover Categories

Chapter 5. Discussion

5.1 The Results in the Context of Literature

The analyses of the results attained from the habitat quality models in this study confirm that despite the existence of differences in the habitat quality values between the two ranges, these variations are not statistically significant. Furthermore, variations in relative habitat quality values between the different scenarios are also not statistically significant across all land use/cover categories in the current range of these species, and when the individual land use/cover categories are the subject of investigation. Nevertheless, the mean habitat quality values in the current breeding grounds are more affected by the existing transportation network, followed by the energy and urban data layers, respectively. This trend is also observed across the individual land use/cover categories specified in the study area.

The disparity of habitat quality values between the historic and current ranges of burrowing owls illustrates that the delineated spatial sources of degradation cannot cause considerable degradation to the habitat of these avian species, when the existing configuration of land use/cover categories is considered across the study area. That is, despite the fact that the current range of burrowing owls is situated within the boundaries of the historic range, habitat degradation caused by the cumulative impact of these sources of degradation cannot merely explain the contracted range and the declined population of these species. Thus, an alternative explanation must exist to shed light on the extirpation of these migratory birds from their historic range.

Given that the majority of the landscape across the historic range of Burrowing Owls was converted from grass to grain, a more plausible explanation for the contracted range of these species would be the relationship that exists between the spatial configurations of habitat patches at this spatial extent.

More specifically, where significant proportions of the landscape had been transformed, area (patch size), and isolation are the factors influencing the population of species in terrestrial ecosystems (Andren, 1994).

That is, the spatial configuration of habitat patches becomes the dominant determining factor as the proportion of converted landscape increases. Yet, the impact made by the considered sources of degradation is not sufficient to attribute the contracted range of these species to the cumulative degradation caused solely by these data layers across the historic range. Rather, this might be attributed to the isolation and patch size effects caused by the past landscape transformation processes (Samson and Knopf, 1994).

According to Andren (1994), a minimum extinction threshold (i.e., the native habitat proportion) of 30% must exist for the persistence of avian species across the terrestrial ecosystems. This means that regardless of the rate of colonization, at a certain point in a continuum of disturbed habitat configuration characterized by fragmentation at one end and complete habitat loss at the other, the metapopulation of species loses viability (Hanski, 2011). The results of this modeling study illustrate that the proportion of habitat in the native vegetation patches (i.e., grasslands) across the historic range of Burrowing Owls is 21%, which falls well below the 30% native habitat extinction threshold defined by Andren (1994).

While this threshold might seem to be unrealistic to habitat generalists like Burrowing owls (Clayton and Schmutz 1999; Orth and Kennedy 2001), the extent of dispersal, variant site fidelity (Wellicome et al., 1997; De Smet, 1997; Duxbury, 2004), the limiting factors particular to these species (Lantz et al., 2004), and their close association with fossorial mammals (Sidle et al., 1998; Wellicome 1997; Leupin and Low, 2001) are the ecological attributes that might have contributed to the declined resistance level among these species across their historic breeding grounds.

Consequently, despite being relatively effective in reducing the habitat quality values across the historic range of Burrowing owls, the modifications made to the landscape, as well as the edge and isolation effects caused by the considered spatial sources of degradation, can only be presumed to have a complementary impact on the changes occurred to the landscape configuration through historic land conversion activities.

Across the current range of Burrowing Owls, where despite the disproportionate rates of native grass loss and population declines over the past decades (Skeel, 2001; Holroyd and Trefry, 2011) these species are still extant across the prairie landscape of Saskatchewan and Alberta, the relative frequency of high-quality habitat is 38%, which is higher than the 30% conversion threshold considered for the persistence of avian species across a given ecosystem.

Nevertheless, the relative habitat quality values under the specified spatial sources of disturbance at this spatial extent yield no significant differences with the habitat quality values across the historic range of these species, nor they differ from one another when different scenarios are considered. These results corroborate the findings of Scobie (2015) in the mixed-grassland areas of southern Alberta and Saskatchewan, where notwithstanding the owls' avoidance from certain anthropogenic features such as roads with high traffic speed, human footprint is yet to affect the breeding habitat for Burrowing Owls, and these birds might even prefer nest sites surrounded by cultivated landscapes or even surface of roads perceived to be less menacing.

Given that at the landscape level, the wildlife response is not instantaneous and represents nonlinearities to native habitat loss across terrestrial ecosystems (Hanski, 2011), the observed declining trends in the population of Burrowing Owls across their current range could be attributed to the relaxation time (Diamond, 1972) associated with the past conversion activities, and, to a much less extent, to the edge and isolation effects caused by the spatial data layers considered in this modeling study.

However, since the extinction threshold is not yet violated across the current range of these species, other potential factors such as severe weather condition (Heisler, 2014) and loss of trophic options (Poulin, 2003), whereby the ecological processes are being disrupted at the landscape (Hutto, 1985) and micro-habitat (Allen et al., 1987) levels, might better justify the observed and continuing demographic trends of these endangered birds. In addition to these potential causes of habitat degradation, the declining population of Burrowing Owls as migratory birds might also be related to factors affecting the returning wild pairs outside of the considered spatial extent in this study (e.g., landscape modifications), particularly across the wintering grounds in the southern portions of their global range (McDonald et al., 2004).

5.2 Implications for Practice

This study presents the first relative habitat quality model at the landscape level with reference to the most cited spatial anthropogenic sources of degradation affecting the habitat for the Burrowing Owls across their breeding grounds in the Canadian Prairies. Based on the existing theories about the critical habitat (Hall et al., 1997) and landscape-level extinction threshold for species (Andren, 1994), the current spatial extent for these endangered avian species can potentially be considered as a spatial extent where the action plans should be implemented.

That is, the proposed fire and grazing management strategies, habitat restoration, population monitoring and management, and reintroduction practices (Parks Canada Agency, 2016) can be implemented at similar or close rates to what is being currently implemented across the conspicuous colonies of the Black-tailed Prairies Dogs, designated as the existing critical habitat boundaries for Burrowing Owls (Environment Canada, 2012). More specifically, the existing home ranges should not entirely be equated to the currently occupied habitat patches as high-quality habitat patches across the landscape might be selected for occupation at a later time (Hall et al., 1997), provided that new burrows be excavated or allocated for reintroduction purposes.

Since a significant proportion of the high-quality habitat areas is located within the grassland patches of the current range, the results of this study can be used in conjunction with concurrent probabilistic models of grassland conversion (e.g., Gage et al., 2016; Olimb and Robinson, 2019), as well as the spatially explicit habitat suitability and resource selection model for Burrowing Owls (Stevens et al., 2010) and other grassland birds (Fedy et al., 2018) in the remaining grassland areas of the prairies to target the high quality habitat patches with different conversion risks and probability of occupation. Since Burrowing Owls are considered as flagship species (Environment Canada, 2017), adoption of conservation strategies at this spatial extent would also benefit the associated burrowing mammals, and other avian species with high association with the grassland patches and similar sensitivity levels to the identified sources of disturbance across the study area (Askins et al., 2007).

Despite illustrating no statistical significance, the differences in the mean habitat quality values under different scenarios in the current range of these species could be used to mitigate the relative habitat degradation caused by the considered spatial data layers across this spatial extent. As there is no precedent for the quantification of habitat quality values under the anthropogenic sources of degradation across the current range of Burrowing Owls, the habitat quality change index calculated for transportation, energy, and urban data layers can be considered as a basis for prioritizing the recovery actions to mitigate the impact of these anthropogenic features across the landscape and different land use/cover categories.

Considering the existing spatial composition of these spatial data layers, conservation, recovery, and reintroduction measures should be implemented by prioritizing the impacts of the transportation network, energy infrastructure, and population centers, respectively. This is of the utmost importance as the further expansion or intensification (e.g., use) of these sources of disturbance across the current range of Burrowing Owls might lead to statistically significant differences, which may ultimately cause extinction cascades (Fischer and Lindenmayer, 2007) when combined with further native habitat loss across this landscape.

5.3 The Limitations and Modeling Advancements

Like any modeling study, the results of this research should be interpreted by considering some limitations and simplifications in the habitat quality simulation framework. The first salient point in this process is the proxy-based nature of the habitat quality model. Despite the inclusion of ecosystem structure, the proxy-based measures in ecology can only capture a small section of a vast web of cause and effect mechanisms characterized by nonlinearities associated with the ecological causal pathways in a given ecosystem (Stephens et al., 2015).

As such, the results of this study should not be seen in lieu of the species-specific studies (e.g., Marsh et al., 2014 a,b; Scobie, 2015) about our endangered birds. Rather, these results should be interpreted through the prevalence and heterogeneity of different habitat patches and the degree to which the suitable landscape is disconnected by the human footprint. This modeling approach, therefore, provides further insights to the studies assessing the demographic, distributional, and physical characteristics of these birds. Furthermore, degradation to habitat patches is calculated considering the additive impact of the delineated spatial sources of degradation on the landscape. However, the cumulative impact caused by these sources of disturbance could be much greater than the sum of its parts (Sharp et al., 2019).

Besides, not all the anthropogenic sources of degradation are considered in this modeling study. For instance, due to the dearth of knowledge on the impact of mining and industrial areas on the habitat quality values for Burrowing Owls, these areas are excluded from the threat data and only considered as land use/cover categories with zero suitability level. Similarly, the impact of petroleum infrastructure was considered to be focused in the proximity of the drilling well pads, and as such, the potential influence of pipelines or new technological advancements (Brittingham et al., 2014) that enable hydraulic fracturing at more depths and reoperation of abandoned wells were not integral to this modeling study.

Moreover, the results of this study should only be considered for a fixed spatial extent and point in time and are not generalizable over different spatiotemporal scales. That is, the temporal changes across the landscape are not part of the habitat quality assessment in this study, mainly because this research serves as a first habitat quality model at the landscape level, and should be used as a baseline model for conservation actions at the current range of these species with reference to the considered sources of degradation. Consequently, extrapolating the results of this study to a larger spatial extent beyond the considered range of these species would be inappropriate due to differences in ecosystem structure and composition of spatial sources of degradation across other spatial extents.

Nevertheless, the incorporation of the elimination method into the parameterization of the relative sensitivity scores could be considered as a stepping stone in this modeling framework, whereby Burrowing Owls' sensitivity to different sources of disturbance is determined using a set of criteria defined based upon the direct and indirect factors affecting the resistance of these species to environmental changes across the study area. More specifically, factors such as edge effect, accessibility to trophic options, or extended home ranges that might not be directly considered through the spatial configuration of the habitat patches and sources of degradation, are incorporated into the ranking system which defines the perceived level of threat, and thus species' relative sensitivity to the sources of degradation.

In addition, this approach is aligned with the participatory nature of the InVEST decision-making toolset and provides a more robust and flexible parameterization approach by incorporating a variety of factors, which could be directly acquired from the literature on the species or expert surveys on different criteria and their order of importance, rather than relying on general principles of ecology (e.g., Forman, 1995) or inferences from cognate studies that might lead to ill-structured modeling assumptions, when species-specific habitat quality measurement is the purpose of assessment.

Chapter 6. Conclusion

This research was conducted to answer the essential knowledge gaps in the literature on the impact of the anthropogenic sources of disturbance on habitat quality for Burrowing Owls across their breeding grounds in the Canadian Prairies. Based on the adopted landscape-level habitat quality model, the existing spatial distribution of the spatial sources of degradation did not result in statistically significant variations in the habitat quality values between the historic and current ranges of these endangered migratory birds. Similarly, different combinations of sources of disturbance illustrated no dramatic differences in the mean habitat quality values across the current range of these species.

Accordingly, the contracted range and the declining population of these ground-dwelling birds of the open prairie landscape could be attributed to other factors, such as the relaxation time associated with past landscape transformation activities, inclement weather conditions, stochastic environmental changes, and access to trophic options across the study area. Yet, since the considered spatial extent serves as the northern edge of the global range for these small migratory birds and that these endangered avian species only occupy the study area during the breeding season, their decline from the landscape might also be the result of the synergistic association between various factors leading to habitat quality degradation across their wintering grounds in the southern parts of their global range.

Despite showing no statistical significance, the different habitat quality values under the considered scenarios across the current range of Burrowing Owls were integral to the calculation of the habitat quality change index as a measure of habitat degradation caused by the different groups of data layers considered in this modeling process. Accordingly, the transportation, energy, and urban data layers were determined to cause, in descending order of importance, the highest levels of degradation across the current range of Burrowing Owls.

As such, conservation measures (e.g., buffer zones) targeting these spatial sources of degradation can be prioritized according to the calculated habitat quality change index to preclude statistical significance that might be caused through the expansion or intensification of the sources of disturbance, which may, in conjunction with further habitat transformation, lead to extinction cascades among these charismatic migratory birds across the Canadian Prairies. However, since the extinction threshold is not yet violated across the current range of these species, conservation measures at the existing critical habitat boundaries could be expanded to match the spatial extent of the current range in order to maintain, and if possible, improve the relative habitat quality for these endangered species at a larger spatial scale.

This study is an initial step, from a landscape modeling preservative, to delve deeper into the influence of the human footprint on habitat quality for Burrowing Owls at the northernmost portion of their global range. However, since habitat and its quality are both relative concepts, the results of this study should only be judged at a fixed spatiotemporal scale, and thus need to be considered as a baseline habitat quality model in the absence of preexisting landscape-level models across the prairies. As such, the results should not be extrapolated beyond this spatial extent and point in time. Consequently, future studies should evaluate the potential impacts of the other anthropogenic or environmental sources of degradation at various spatial scales and points in time.

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Appendix

Table 1. The list and definitions of IUCN categories and the assigned accessibility values

IUCN Categories	Assigned Accessibility Value	Description
Ia: Strict Nature Reserves	0	Protected areas that are strictly set aside to protect biodiversity and also possibly geological/geomorphological features, where human visitation, use and impacts are strictly controlled
Ib: Wilderness Area	0.28	Protected areas that are usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, which are protected and managed so as to preserve their natural condition.
II: National Parks	0.42	Large natural or near natural areas set aside to protect large-scale ecological processes, along with the complement of species and ecosystems characteristic of the area, which also provide a foundation for environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.
III: Natural Monument or Feature	0.57	Protected areas set aside to protect a specific natural monument, which can be a landform, sea mount, submarine cavern, geological feature, or even a living feature such as an ancient grove. They are generally quite small protected areas and often have high visitor value.
IV: Habitat/Species Management Area	0.71	Protected areas aiming to protect particular species or habitats and management reflects this priority. Many category IV protected areas will need regular, active interventions to address the requirements of particular species or to maintain habitats, but this is not a requirement of the category.
V: Protected Landscape	0.86	Includes areas where the interaction of people and nature over time has produced an area of distinct character with significant ecological, biological, cultural and scenic value: and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values.
VI: Sustainable use of natural areas	1	Protected areas that conserve ecosystems and habitats, together with associated cultural values and traditional natural resource management systems.

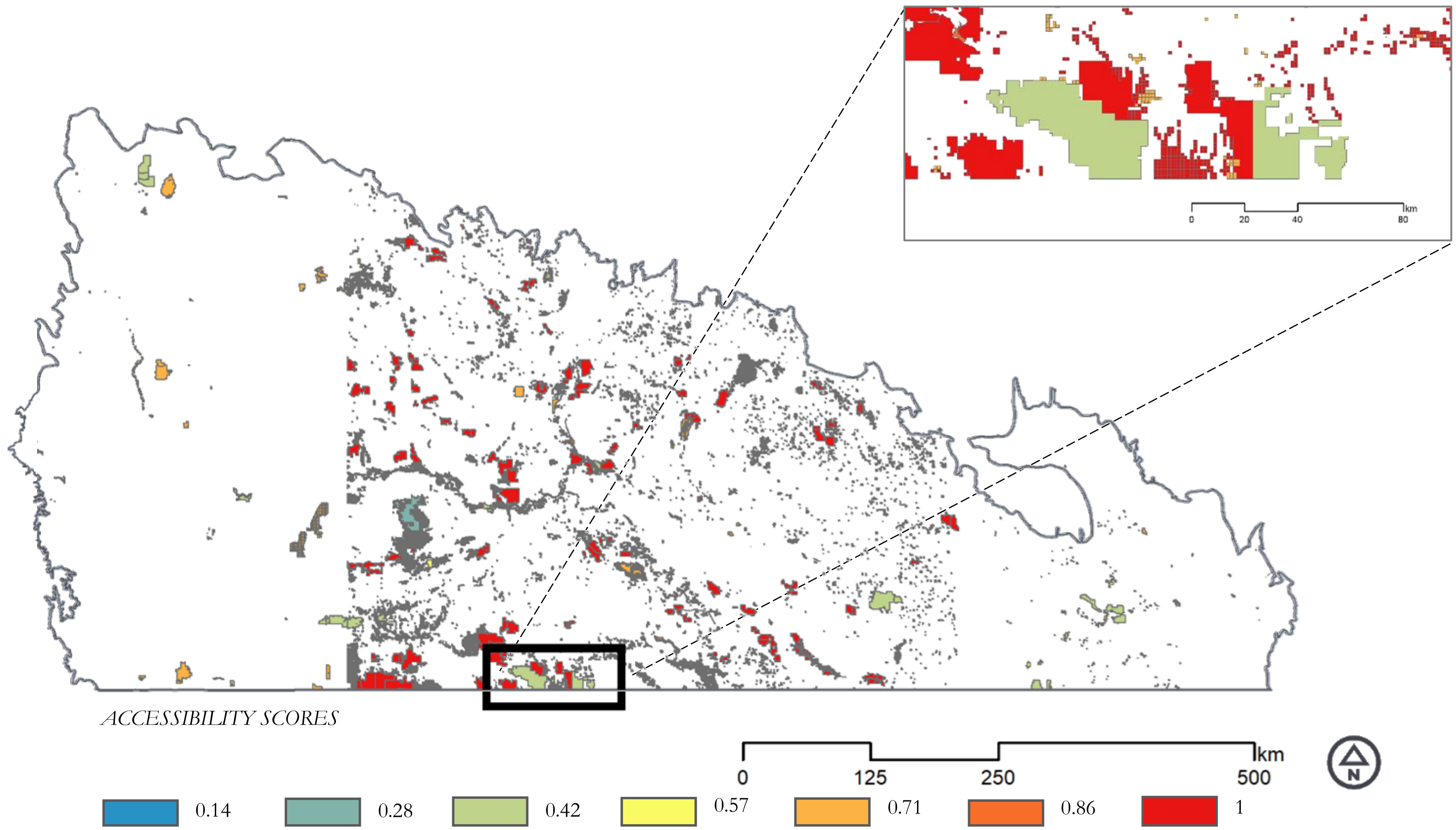

Figure 1. The Protected Areas across the Prairie Ecozone

Table 2. Sample Factors and Alternatives in a MCDA Situation

	Alternatives				
Factors	Alternative 1	Alternative 2	Alternative 3	Alternative 4	Alternative 5
Factor 1	11.9	15.6	10.5	13.1	12.4
Factor 2	B-	A	C-	C+	B+
Factor 3	B-	B-	B	C+	C+
Factor 4	B	C-	A	B	C
Factor 5	C+	C+	B+	C-	C+
Factor 6	8.7	15.6	6.9	17.8	11.4
Factor 7	21	84	14	67	37

Table 3. Sequential Elimination Process in a Sample MCDA Situation

	Alternatives				
Criteria without linking statements (In decreasing order of importance)	Alternative 1	Alternative 2	Alternative 3	Alternative 4	Alternative 5
Factor 1 must be greater than 11			×		
Factor 2 must be C+ or better			×		
Factor 3 must be C+ or better					
Factor 4 must be B or better		×			×
Factor 5 must be B or better	×	×		×	×
Factor 6 must be 10 or less		×		×	×
Factor 11 must be less than 50		×		×	

Alternatives	Alternative 1	Alternative 2	Alternative 3	Alternative 4	Alternative 5
Initial ranking	1	4	5	2	3

Table 4. Sequential Elimination and Ranking based on Linked Criteria

	Alternatives				
Criteria using linking statements (In decreasing order of importance)	Alternative 1	Alternative 2	Alternative 3	Alternative 4	Alternative 5
Factor 1 must be greater than 10 AND Factor 5 must be C+ or better				×	
Factor 2 must be better than B OR Factor 4 must be at least B+	×				
IF Factor 3 is less than B, then Factor 6 must be less than 10		×			×
IF factor 2 is B or more, then Factor 7 must be less than 40		×			

Alternatives	Alternative 1	Alternative 2	Alternative 3	Alternative 4	Alternative 5
New ranking	4	3	1	5	2