# Surrogacy and Umbrella Relationships of Grizzly Bears and Songbirds is Scale Dependent

by

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**Abstract**

The consideration of different spatial scales in ecological studies, including assessments of surrogacy, is often suggested, but less commonly implemented. Species utilize their environments at many spatial scales; therefore, the relationships between species including surrogate relationships, will vary depending on the scale of analysis. Here I study the potential for grizzly bears to act as a surrogate for songbird conservation in Alberta across three spatial scales: (1) the broad-scale umbrella effect that uses the scale of individual home ranges of bears, (2) an intermediate scale that uses seasonal patch-level resource selection function models (maps) to index local use (avoidance to selection) of habitats by grizzlies within a region, and (3) the localized scale that uses individual bear telemetry locations with known activity. The relationship between grizzly bears and songbird diversity changed over the three spatial scales examined – with the strongest results observed at the largest scale and little to no relationship at the intermediate and localized scales, respectively. This emphasizes the importance of spatial scale in surrogacy studies. As well as focusing on the surrogacy potential of grizzly bears more intensively, I also tested the idea that flagship species are equivocal to umbrella species by comparing the surrogate relationships of grizzly bears against two other well-known flagship species in Canada – the greater sage-grouse and woodland caribou – as surrogates for the same group of interest, songbirds. Grizzly bears were found to outperform the other candidate species at my largest spatial scale in Alberta (umbrella) – illustrating that a flagship can also be an umbrella species, but not in all cases. I also demonstrate that not all areas of grizzly bear range are equally effective in providing surrogate potential which emphasizes the consideration of geographic variation which can significantly affect the potential for a signal in surrogacy.

A primary motivation for using surrogate species is to simplify conservation action – using the management of one species' to guide the conservation of many. While this appears to be a theoretically simple idea, it has proved to be contentious in the literature and its efficacy circumstantially difficult to predict. This study contributes to knowledge on the applications of surrogacy – helping to emphasize the importance of spatial scale in evaluating a species' surrogacy potential.

## Preface

This thesis is an original work by Emily Cicon. No part of this thesis has been previously published.

## Acknowledgements

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## Chapter 1: General Introduction

### 1.1 Species surrogacy and umbrella species

Species surrogacy, as used in this paper, is the term used to describe the use of one species as a representative for another (Caro 2010). Within this over-arching definition, there are many sub-groups referring to specific uses (Simberloff 1998; Caro and O'Doherty 1999; Hunter Jr. et al. 2016). For example, surrogacy can be separated into categories such as flagship and umbrella species. The former of which is a charismatic species used to symbolize a wider reaching conservation plan and the latter a species whose conservation requires large tracts of land leading to the conservation of shared habitat – both theoretically encapsulating other species, and meeting the definition of surrogacy (Simberloff 1998; Caro and O'Doherty 1999). While these terms are not mutually exclusive, they are not equivalent, as flagship species are chosen for their charisma or ability to generate public interest, whereas umbrella species are chosen based on biological factors (e.g. large home range size) which impact species co-occurrence (Simberloff 1998; Caro and O'Doherty 1999; Leader-Williams and Dublin 2000). Regardless of specific sub-group, surrogacy is viewed as a management "shortcut" – proposed to increase the efficiency in managing multiple species, and to some degree, to be a necessity given the inavailability of data (Simberloff 1998; Favreau et al. 2006)

The use of species surrogacy for wildlife and land management is somewhat controversial, with both successes and failures. For example, it has been shown to be generally effective in numerous cases (Suter et al. 2002; Caro 2003; Branton and Richardon 2010; Morelli et al. 2015, 2017; Johnson et al. 2017). However, these positive findings can be equally contrasted with negative results (Andelman and Fagan 2000; Ozaki et al. 2006; Rubinoff 2001), 'caveats' (e.g. Favreau et al. 2006), and critiques by others on its impracticality and misuse (Andelman and

Fagan 2000; Murphy et al. 2011). As noted in a meta-analysis by Favreau et al. (2006), many surrogacy studies are concluded with caveats, including lack of transferability to all species and spatial-temporal contexts. For example, although Caro (2003) indicates that the reserves created in East Africa using the umbrella species principle were generally effective, it is noted that while some groups were beneficiaries from protection under the umbrella, others such as small mammals were not. While the study of Northern Goshawks conducted by Ozaki et al. (2006) didn't find goshawks to be an effective umbrella in northern Japan, they do note that a study of the same species at a different location (Italy) yielded the potential for effective surrogacy (Sergio et al. 2005). Given these conflicting findings and the number of components that contribute to the success or failure of a management strategy based on species surrogacy, the use of surrogate species selection criteria, such as large home range (Seddon and Leech 2008), and quantitative analyses, such as comparisons across spatial scales (Favreau et al. 2006), have been suggested as a means of improving the implementation of surrogacy. Therefore, within this study grizzly bears will be evaluated as a surrogate for songbirds across a number of spatial scales and geographic regions in Alberta.

## 1.2 Study species

### 1.2.1 Grizzly bears in Alberta

Grizzly (brown) bears (*Ursus arctos*) occupy over 40 countries of the northern hemisphere, occupying a diverse range of habitats, including deserts and forests (McLellan et al. 2017). In Canada, this generalist habitat trait is also evident with ranges occurring across 4 provinces and 3 territories – varying from arctic climates to temperate coastal rainforests (COSEWIC 2012). Habitat types used by individual grizzlies within specific regions are also highly variable, with factors such as season (and therefore food availability) (McLellan and Hovey 2001) or activity type – e.g. using forested areas for bedding vs. open areas for root digging (Munro et al. 2006) affecting selection.

Grizzly bears are omnivores with varying dietary components (and ratios of components) due to numerous factors such as season and occupied region (Munro et al. 2006; Mowat and Heard 2006; McLellan and Hovey 1995; Hamer and Herroro 1987). For example, in an interior region, such as west-central Alberta, grizzly diets could include a combination of plant matter (e.g. roots or berries), ungulates, arthropods, and other terrestrial vertebrates (Munro et al 2006). In contrast, coastal bear populations in British Columbia may incorporate other sources of protein, including salmon (Mowat and Heard 2006). Seasonal shifts in diet are expected, for example a shift from early summer ungulate hunting to frugivory in the fall (Munro et al. 2006), or the seasonal consumption of different plant species (Hamer and Herroro 1987).

Although globally *Stable*, grizzly bear is a species of *Special Concern* within Canada (McLellan et al. 2017; COSEWIC 2012). The current range of grizzly bears in Canada is estimated to be 76% of their historical range, with extirpations linked to human interference and habitat modifications (Banci et al. 1994), although there is no evidence of substantial further range constriction in the past three decades (COSEWIC 2012). In the province of Alberta, grizzlies are a *Threatened* species (AEP 2016) that occupies a 228,000 km$^2$ range in the western portion of the province (ASRD 2008). There are an estimated 691 mature bears in the province, with no official statement of population decline or growth (ASRD and ACA 2010a). The predominant causes of grizzly bear mortalities in Alberta are related to human action (ASRD 2008; Benn and Herrero 2002), with the two largest factors contributing to mortality being accidental deaths, such as road collisions and poaching (Govt. of Alberta 2018). Both of these factors are facilitated through access (e.g. industrial roads) (Nielsen et al. 2004b; McLellan

1990). The modification of grizzly bear habitat is also cited as being a potential threat, although this issue is further complicated by the potential benefits of modifications such as clear-cuts increasing food supply (ASRD 2008; Nielsen et al. 2004c).

### 1.2.2 Songbirds in Alberta

*Songbirds* include birds classified within the Order Passeriformes, with about 5000 species globally – over half the known bird species (Edwards and Harshman 2013). It is important to note that while there is a differentiation between the true grouping of songbirds, the Oscines, and the Suboscines (Edwards and Harshman 2013), many studies use the term songbird for species within both clades (e.g. Holmes 2007; Bonnot et al. 2018; Stralberg et al. 2018), including this thesis. Songbirds can be found on every continent (with the exception of Antarctica) (Edwards and Harshman 2013), and depending on their migratory patterns, can be found on multiple continents throughout their life cycles, as is shown in the movement of songbirds between North American breeding grounds and South American (or other Neotropic) wintering grounds (Stutchbury et al. 2009; Winger et al. 2014; DeLuca et al. 2015), referred to as Nearctic-Neotropical migration (Hayes 1995).

North American songbirds, with emphasis on the aforementioned Nearctic-Neotropical migrants, are in decline (Robbins 1989; Ballard et al. 2003). There are many possible causes of this decline including changes to habitat on breeding and wintering grounds, such as habitat loss (Robbins 1989; Taylor and Stutchbury 2016) and fragmentation (Robinson and Wilcove 1994; Donovan and Flather 2002; Taylor and Stutchbury 2016). Populations of birds in Canada have declined by over 10% in the past 40-50 years (NABCI 2012), with specific categories of birds, including aerial insectivores and grassland birds, exhibiting even stronger reductions (Peterjohn and Sauer 1999; Brennan and Kuvlesky 2005; Nebel et al. 2010). This conservation concern for birds is shown in the province of Alberta which contains three different ecological regions: the Western Boreal (spanning central to north Alberta), Prairies (south-east Alberta), and West Coast and Mountains (western edge of Alberta), each with a unique set of species and conservation issues, but all with habitat loss (or degradation) as a key issue (NABCI 2012).

Songbirds are a taxa of conservation concern on both a national and provincial level and are, mostly, protected under the Migratory Birds Convention Act (1994) (predicated upon migratory status), therefore necessitating management consideration. Provincially, of the 134 species (Alberta Bird Record Committee 2017) there are five species of songbird that are designated as *Special Concern* or are recommended to have a status of *Special Concern* (AEP 2016) and seven species which are designated as *Special Concern* or a more critical status by SARA (Government of Canada 2019). Given these threats and associated statuses of this taxa, in combination with the immense species diversity, it is a logical step to streamlining the management process through a method such as surrogacy.

### 1.3 Thesis objectives and organization

The broad objective of this study is to determine the ability for grizzly bears to act as a surrogate species for songbirds in Alberta, based on the assumption of mutually beneficial overlapping habitat. Specifically, my study aims to identify surrogacy relationships across three different scales: the umbrella species scale, the intermediate surrogacy scale for patches of habitat, and the localized surrogacy scale based on known grizzly bear locations with known behaviour. As a point of clarification, for the purposes of this thesis, the term *umbrella scale* will refer to the largest scale of assessment (e.g. evaluations using home range-sized areas), *localized scale* will be used to refer to evaluations of surrogacy at the smallest scale, where the term umbrella species does not apply, while species surrogacy will refer to the broad definition.

*Intermediate scale* will apply to site sizes which are too large to be deemed *localized*, but are not large enough to be considered representative of an umbrella scale effect.

At the larger umbrella scale, I assessed whether grizzly bears are effective surrogates when compared to other flagship species in Alberta, in terms of which species' range and associated conservation planning unit (management area) size would contain the highest songbird species richness or beta diversity. For the intermediate surrogacy scale, I examined the relationships between bear resource selection functions and bird richness. On the smaller localized surrogacy scale, I tested whether sites selected by grizzly bears for activities, such foraging locations, hold higher songbird species diversity or richness when compared to nearby random locations.

This thesis is organized into one data chapter and follows the formatting style of the Journal of Biodiversity and Conservation.

# Chapter 2: Grizzly Bears as a Surrogate Species for Songbirds at Multiple Spatial Scales

## 2.1 Introduction

In ecology, it has long been acknowledged that the consideration of spatial scale in both designing studies and interpreting results is crucial (Levin 1992; Wiens 1989). Spatial scale includes both the overall size of area considered, as well as the resolution of data within a given area (e.g. pixel size on a mapping software) (Palmer and White 1994; Pearson and Carroll 1999; McCarigal et al. 2016). Wiens (1989) argued that spatial scale must be varied based on the objective of the study, as well as the taxa or species of interest. This is further complicated by the idea that even within a species, habitat use and selection occurs at many scales (Wiens 1989), referred to by Addicott et al. (1987) as a species having many 'ecological neighborhoods'. These concepts have led to the inclusion or recommendation of multi-scale analyses within ecological studies, although upon review, it was found that the majority of habitat selection studies did not explore this aspect (McCarigal 2016).

Surrogacy studies are among those that are recommended to consider spatial scale on a number of levels (Favreau et al. 2006). Species surrogacy, when defined at its most basic form, is one species representing one or many others (Caro 2010) – predicating that there must be a form of co-occurrence between the species / their habitats. As noted by Wiens (1989) it must be considered that the scale at which one species uses an environment will differ from another: consider a vulture vs. a beetle – or in this case a grizzly bear vs. a songbird; therefore when determining the relationships of multiple species, scale is influential (Wiens 1989). It has been observed that the spatial scale – whether overall area or resolution – used for analysis can affect whether or not patterns of co-occurrence between species are observed and the strength of this relationship (Weaver 1995; Flather et al. 1997; Gaspar et al. 2010; Higa et al. 2016; Morelli et al.

2015). It cannot be assumed that the nature of this relationship between scale and surrogacy is consistent between studies and taxa. For example, the Gaspar et al. (2010) study of arthropod co-occurrence showed that surrogacy was generally higher when measured at their smaller scales – which they note is in contrast to the relationship of larger scale, more association proposed by Favreau et al. (2006).

Large carnivores, such as grizzly bears (*Ursus arctos*), are often considered as candidate large scale surrogates (referred to as umbrella species), given their need for land conservation over broader areas (Noss 1996; Andelman and Fagan 2000; Simberloff 1999). The grizzly bear, a Canadian species of *Special Concern,* occupies individual home ranges of hundreds of square kilometers (COSEWIC 2012); however, it is understood that individuals within species also have a number of ecological neighborhoods (Addicott et al. 1987). These neighborhoods include not just their home or geographic range, but also smaller scales such as specific sites, in the case of the grizzly bear based on localized presence of foods and habitat types (Nielsen et al. 2004a; Nielsen et al. 2004c; Stewart et al. 2013; Nielsen et al. 2017). This study therefore examines grizzly bear surrogacy on multiple scales to address the relationships not only at the umbrella scale, but also at finer scales where different 'ecological neighborhoods' may effect co-occurrence – ultimately to provide insight into a management "shortcut" (Simberloff 1998) for the conservation of a species rich group – songbirds (Edwards and Harshman n.d).

With a focus on including spatial scale, my objective here is to determine whether grizzly bears are an adequate surrogate species for another well-studied taxa, songbirds. To address this objective, I will examine species co-occurrence on three separate spatial scales using field-derived data for the localized scale and modelled (mapped) data for both intermediate and large-scale studies. The broadest of the three studies will evaluate the grizzly bear's potential as an umbrella species through a comparison of bird diversity within the ranges of three flagship species in Alberta – while also addressing the effects of spatial variation in addition to scale. In contrast, the intermediate study will compare the relationship between grizzly bear habitat selection and bird richness. The localized study will focus on the relationship between sites selected by bears for denning, foraging, etc. and bird diversity observed in the vicinity of those sites.

## 2.2 Methods

### 2.2.1 Study areas

Analysis of surrogacy effects on all scales were undertaken in Alberta, Canada, although analyses varied in their scope and location within the province depending on the scale of interest (umbrella, intermediate, or localized) and species (e.g. grizzly bear, caribou, or sage-grouse).

*2.2.1.1 Umbrella scale*

The study area included (1) the 228,000 km$^2$ grizzly bear range on the western part of Alberta (ASRD 2008); (2) the 4,000 km$^2$ greater sage-grouse range in south-eastern Alberta (Aldridge 1998); and (3) the 134,833 km$^2$ woodland caribou range in west-central and northern Alberta (ASRD and ACA 2010) (Figure 2.1).

One scenario focused on core habitat within the grizzly bear and woodland caribou provincial ranges (Figure 2.2). Core habitat was defined as the most important areas for conservation (ASRD 2008). For the grizzly bears, core areas encompassed 37,274 km$^2$ (15.5% of total range) and was previously delineated based on factors such as low road density and high-quality habitat (Nielsen et al. 2009). No existing core ranges were publicly available for all woodland caribou ranges in Alberta and were therefore created here. Specifically, it was examined whether sub-watersheds had less than 65 percent disturbed habitat based on a 500 m buffer around mapped disturbances at a scale of 1:30,000 following caribou recovery rules from Environment Canada (2012). National Parks were also removed from caribou core areas to mimic core selection of grizzly bear habitat (Nielsen 2009, ASRD and ACA 2010). The resulting core area encompassed 37, 276 km$^2$ (24.8% of total range).

The last of the umbrella scale scenarios used Bear Management Areas (BMAs) (Figure 2.3). These sites divide the grizzly bear range into seven distinct areas and are composed of BMAs 1-7, referred in in this paper from north to south as Alberta North (BMA 1), Swan Hills (BMA 7), Grande Cache (BMA 2), Yellowhead (BMA 3), Clearwater (BMA 4), Livingstone (BMA 5), and Castle (BMA 6).

*2.2.1.2 Intermediate and localized scales*

Analysis of the relationship between grizzly bear habitat selection (resource selection function; RSF) and songbird species richness, as well as the analysis of localized surrogacy potential, was completed within the Yellowhead Bear Management Area (BMA 3), a 28,758 $km^2$ area in the central portion of the Alberta grizzly bear range (AEP 2016) (Figure 2.4), with an estimated population of 74 grizzlies (Stenhouse et al 2015). The study area for this localized surrogacy analysis did not encompass the entire BMA 3, but was focused instead on the north-western section of BMA 3 - specifically centered around 11˚ N  496671.98˚ W and 5892623.76˚ N (Figure 2.4).

**2.2.2 Planning units**

In this study, the term planning unit is used to describe individual sites of a defined size created in a hexagonal (umbrella) or square (intermediate) shape using the Repeating Shapes Extension (Jenness Enterprises 2012) in ArcMap (Esri 2016). These planning units act as areas of analysis for evaluations of pre-existing spatial data and can be thought of as management areas. For example, a planning unit could be a site whose size is defined by the home range of a species. These planning units are then varied in their location and sizes to create a range of planning unit scenarios. For example, a planning unit scenario testing the effect of planning unit size (area), would use a consistent planning unit location, but would compare many planning unit sizes across that location.

*2.2.2.1 Umbrella scale*

Four distinct planning unit scenarios were used address to assess the potential effectiveness flagship species as surrogates for songbirds. The first two initial scenarios were focused on the effects of planning unit size and planning unit location. Planning unit size was evaluated by comparing the songbird metrics among six different sized planning units: 25 $km^2$, 35 $km^2$ (greater sage-grouse), 100 $km^2$, 338 $km^2$ (grizzly bear), 382 $km^2$ (woodland caribou) and 1000 $km^2$, over the entire province – keeping the location of the planning units constant, while varying planning unit size.

Three of the planning unit sizes were based on the home range sizes of flagship species in Alberta defined by female range size for the two species of mammals and lek setback distance for sage-grouse. The area of planning units were 35 $km^2$ for greater sage-grouse (equivalent of a 3.2 km radius) (AESRD 2013); 382 $km^2$ for woodland caribou (Tracz et al. 2010), and 338 $km^2$ for grizzly bears (average of Alberta female ranges as listed by COSEWIC 2012). Three additional planning unit sizes (25 $km^2$, 100 $km^2$, and 1000 $km^2$) illustrate variation in the surrogacy effectiveness of other potential planning unit sizes, particularly those smaller and larger than were represented by the three flagship species used.

Planning unit location was evaluated with a constant planning unit size, set at 25 $km^2$, with the location of the units corresponding to the range of grizzly bear, woodland caribou, or greater sage-grouse. Using the respective planning unit sizes within each area, grizzly bear and woodland caribou core areas were compared. Lastly, within the grizzly bear range, BMAs were compared using the grizzly bear planning unit size (338 km$^2$), again analyzing for the effect of location, not planning unit size. Both of these scenarios (core and BMA) were separately evaluated for the 31 songbirds with a status of “Sensitive”, “At Risk”, or “May Be At Risk” (Government of Alberta n.d.), which are referred to in this study as Species at Risk.

#### *2.2.2.2 Intermediate scale*

Planning units for the intermediate study consisted of 27 327 square planning units, with 1 km$^2$ - square dimensions to match the pixel shape of the resource selection function models.

### 2.2.3 Study sites for field-based section

#### *2.2.3.1 Localized scale*

Grizzly bear use locations were determined from GPS telemetry data gathered from the Foothills Research Institute (fRI) in 2016 and 2017 using those sites with known activity based on visitation of telemetry locations by fRI staff. Bear activities include kill site, foraging, or bedding with attribute information on the individual including unique identity of bear and sex. Activity sites were selected and visited in the spring/early summer of 2017. Initially, sites were selected with an effort to balance sampling effort between individual bears and activity type, but given limitations in access, site selection was also based on ability to access sites within reasonable time parameters. Sites that were randomized to location were paired with grizzly bear activity sites for comparison and located further than 500 m, but within 3 km of the use micro-site (daily movement distance of female bears) (Berland et al. 2008) (see Appendix 1 for description of sites). Multiple random sites were first generated for each grizzly use site and the first site (randomly assigned) visited unless inaccessible in which case the next location was used, until an accessible site was reached. In the case where none of the randomized sites were accessible, a location was chosen in a random cardinal direction approximately 500 m from the use site.

### 2.2.4 Songbird data: abundance models and field collection

#### *2.2.4.1 Abundance models*

104 songbird species were included in the analysis of songbird diversity. These species were chosen based on the availability of relative abundance models from Alberta Biodiversity Monitoring Institute (ABMI) and vary in their provincial conservation status (ranging from Secure to At Risk) (Government of Alberta n.d.). The relative abundance models were derived from a combination of models relating to habitat association (based on point count, human footprint and vegetation data) and variation across space and climatic gradients (ABMI 2017). ABMI data for each species was converted from polygon to raster format and summarized for each planning unit (including RSF pixels) using zonal statistics in ArcMap (Esri 2016). Zonal statistics allowed for the calculation of average relative abundance for each species in each planning unit. Data were converted into presence-absence data, where an average relative abundance of 1 (ABMI's minimum value) was equivalent to absence, and any larger number (up to the maximum of 100) was recorded as presence.

#### *2.2.4.2 Field collection and analysis*

Autonomous recording units (ARUs), Wildlife Acoustics Song Meter SM2 and SM4 models, were placed at each site from early May until mid-July, 2017. Identical models (e.g. an SM2 paired with an SM2) were used within pairs to account for differences in detection ability between models (Yip et al. 2017). Specific ARU protocols followed ABMI guidelines for the placement of ARUs, such as distance above ground and orientation (Lankau 2015). Three minute recordings were taken at sunrise – which was constant within pairs, but varied between pairs up to a minute. ARUs were rotated through all sites over the sampling period to detect songbirds during migration and breeding phases.

In an effort to determine the radius surrounding the ARU in which a bird was detected, maximum detection distance values (MDD) from The Boreal Avian Monitoring Project for 37 of the 38 detected species were used based on values reported by Partners in Flight (The Boreal Avian Monitoring Project 2012). An average of all 37 species yielded an MDD of 159 m, while considering only species with 10 or greater observations (11 species) yielded an MDD of 152 m. We therefore chose a 150 m radius as representing the detection distance for all of our sites. This detection radius would be used to determine area surrounding ARU to be evaluated for vegetation, canopy cover, and bear resource selection factor (see Sections 2.2.5 and 2.2.6).

ARU recordings were analyzed for species presence and number of individuals using the software Audacity 2.1.3 (Audacity Team 2017), which produces a spectrogram as a visual aid in addition to the audio file (see Appendix 2 for example). Files were evaluated with a consistent volume level, with exception to files where the track was quiet, in which case volume was increased. Volume "levels" were determined using Bioacoustic Unit protocols (Lankau et al 2017) in order to create a standardized measure of volume for different computers and/or headphones. The first day when recordings for the grizzly use and paired random sites aligned (e.g. pairs of ARUs were not always able to be set up on the same day) was analyzed whenever possible. The first available date was not always used in the case when noise or weather impaired recordings or when issues occurred with ARU recording settings – in which case the next useable date was analyzed if the file was available. Sites with no alternative were removed from analysis, as well as ARUs whose clocks upon retrieval were inaccurate by two or more minutes, random sites with no bear activity check noted (checks were comprised of a site scan looking for clear signs of recent bear use), or other issues with protocol. This resulted in a total of sixty-five pairs of sites available for analysis. Individual birds which were not possible to be confidently identified to species were removed from analyses, but the site and any of its identifiable species were retained.

### 2.2.5 Resource selection function

Grizzly bear resource selection function (RSF) values were provided by the Foothills Research Institute and reflect the relative probability of occurrence of a grizzly within a 30 x 30 m pixel, thereby also allowing conclusions to be drawn about the habitat within the area (Nielsen et al. 2009). RSF values were derived from grizzly bear GPS collar data and measures of the bio-physical characteristics of the landscape (Nielsen et al. 2006, 2009). RSF values were provided for spring, summer, and fall, as well as a maximum overall RSF value for the year 2015. Although more recent RSF data was available, 2015 RSF values were chosen in order to better align with the ABMI songbird data (also dated as 2015) for the intermediate study. RSF values for the year 2016, which was the most recent RSF data available at time of analysis, were used in the localized study where RSF was evaluated in a 150 m radius buffer around each ARU point.

### 2.2.6 Classifying vegetation

Percent cover of vegetation types and average canopy cover was assessed at 150 m radii around each ARU point using 2017 land cover data (Nijland et al. 2015).

### 2.2.7 Measurements of diversity: species richness, beta diversity, and multi-variate analyses

All statistical evaluations were done in R version 3.5.1 (RStudio Team 2016).

*2.2.7.1 Umbrella scale*

Species richness was calculated using presence-absence of each species in each planning unit, in all four of the umbrella planning unit scenarios. Within each scenario, ANOVA or PermANOVA with pairwise Tukey tests were used to test the significance of differences in average species richness per planning unit. Beta diversity based on a Sorenson dissimilarity value was calculated for the core habitat and BMA scenarios using the package betapart 1.5.0 (Baselga et al. 2018).

*2.2.7.2 Intermediate scale*

Species richness was calculated using presence-absence of each species for each planning unit. RSF values (in each of the four categories: spring, summer, fall, and maximum) were then extracted for each site. These estimates of species richness were then evaluated with General Linear Mixed Models (GLMM) using the R Package glmmTMB (Brooks et al. 2017). The GLMMs used a random sample of 5000 data points, from the original 27,327 data points, using negative binomial distributions. Four models were created (five including the intercept model), using the four different RSF value types provided as fixed effects and songbird species richness as the response variable, with sub-watershed included in each model as a random effect to account for local correlation and scale of management unit being considered (Table 2.1).

*2.2.7.3 Localized scale*

Species richness was used as a metric of songbird diversity and was calculated using songbird species data from the ARU analyses. These species richness values were also evaluated with General Linear Mixed Models (GLMM) using the R Package glmmTMB (Brooks et al. 2017). Eight models (nine including the intercept model) were examined, all of which accounted for pairs of sites as a random effect, and used the status of either grizzly use or random location as a fixed effect (Table 2.2). Model 1 was structured to include the percent cover of vegetation type as a fixed effect at a 150 m radii and is referred to as Vegetation Type Cover 150 m. Model 2 simplified these vegetation measurements to average percent canopy cover within the 150 m radii, and is referred to as Canopy Cover 150 m. Model 3, referred to as Bear, used bear sex, individual ID, type of bear activity at the site, and number of days recording took place following May 1$^{st}$ as fixed effects. Model 4, referred to as Paired, contained only the use or random status as a fixed effect. Models 5 through 8 focused on grizzly RSF values for four different categories: maximum, spring (S1), summer (S2), and fall (S3) within a 150 m radius. A Pearson correlation matrix was used to identify correlated variables between continuous variables in Vegetation Type Cover 150 m (the only model with more than one continuous variable) and which covariates to be removed. All eight models were run as Poisson models, and the resulting Akaike information criterion (AIC) (Akaike 1973) values were then compared. Models were examined for over-dispersion and under-fitting of zeroes, and were found acceptable for the Poisson distribution. Sorenson dissimilarity values were estimated for grizzly use and random sites as measures of beta diversity using the package betapart 1.5.0 (Baselga et al. 2018).

MetaMDS analyses were then run using the R vegan package 2.5-2 (Oksanen et al. 2018) and ecodist (Goslee and Urban 2007). MetaMDS evaluated relationships between the species composition at sites based on (1) activity type and (2) sex of bear. Bray-Curtis distance was used in both analyses, with k = 2 dimensions. For the purposes of increasing clarity, two outlying data points were removed.

## 2.3 Results

### 2.3.1 Umbrella scale

*2.3.1.1 Comparison of planning unit sizes*

Average bird species richness per planning unit ranged from 89 species (SE = 0.06) at the 25 $km^2$ size to 92 species (SE = 0.27) at the 1000 $km^2$ size. Average bird species richness per planning unit differed significantly between the six planning unit sizes- when location in Alberta was held constant ($F_{5,54826} = 58.534$, $p<0.001$) (Appendix 3; Figure 2.5). However, differences were not found for every Tukey pairwise comparison (Appendix 4). Generally, larger sized planning units had higher average species richness per planning unit than smaller sized planning units, with the exception of the three largest planning unit sizes (338 $km^2$ (grizzly bear), 382 $km^2$ (woodland caribou), and 1000 $km^2$), where the increase in area between these planning units did not result in a significant addition of species.

*2.3.1.2 Comparison of planning unit locations (flagship ranges)*

Average bird species richness per planning unit when planning unit size was held constant at 25 $km^2$ was also found to differ between the three surrogate species ranges: grizzly bear, woodland caribou, and greater sage-grouse provincial range ($F_{2,14161} = 254.4$, $p<0.001$) (Appendix 5; Figure 2.6). All three ranges were found to have different species richness per planning unit, with grizzly bear range having the highest ($\bar{x} = 88$, SE = 0.1) and greater sage-grouse having the lowest ($\bar{x} =72$, SE = 0.31) (Appendix 6).

*2.3.1.3 Comparison of planning unit locations (grizzly and caribou core habitat)*

Average bird species richness per planning unit was higher for grizzly bear core habitat ($\bar{x}$ = 92, SE = 0.89) than for caribou core habitat ($\bar{x}$ = 86, SE = 0.58), when accounting for all songbird species ($p<0.001$), as well as in the separate analysis for Species at Risk ($\bar{x}_{grizzly}$= 19, SE = 0.19; $\bar{x}_{caribou}$ =18, SE = 0.17) ($p<0.001$) (Appendix 7, Appendix 8, Figure 2.7).

Sorensen dissimilarity values for woodland caribou core habitat planning units were higher than those for the grizzly core areas for both all songbirds and Species at Risk (Table 2.3)

*2.3.1.4 Comparison of planning unit locations (Bear Management Areas)*

Average bird species richness per planning unit differs between the seven Bear Management Areas, for all songbird species ($p<0.001$) and Species at Risk ($p<0.001$) (Appendix 9, Appendix 11, Figure 2.8, Figure 2.9); however, in Tukey pairwise comparisons of Bear Management Areas, they do not all differ from each other and the results vary between the analysis of all songbird species and only Species at Risk (Appendix 10, Appendix 12).

Sorensen dissimilarity values have a clear divide between the more northern bear management units of AB North, Yellowhead, Grande Cache, and Clearwater with that of the more southern Livingstone, and Castle for all songbird species and Species at Risk (with the exception of the northern Swan Hills BMA which exhibited a value closer to the southern BMAs) (Table 2.4).

**2.3.2 Intermediate scale**

*2.3.2.1 RSF and songbird richness*

The most supported of the four RSF models was found to be the Max RSF model (AIC = 51678.2) (Table 2.5), which indicated a significant positive effect between RSF values and songbird richness ($\beta$ = 0.050, SE = 0.004, p<0.001) (Table 2.6), equating to a 5 % increase in bird species richness with a one unit increase in RSF.

### 2.3.3 Localized scale

#### *2.3.3.1 GLMM: species richness*

Results comparing continuous variables from vegetation cover types at a 150 m radius indicated that percent cover of anthropogenic and upland treed vegetation were highly correlated (value > 0.7) (Appendix 13). Upland treed vegetation was removed from subsequent models and calculations.

AIC comparisons of models indicated that canopy cover at a 150 m radius (AIC = 512.0) had the lowest AIC and thus greatest support (Table 2.6). Within the canopy cover at a 150 m radius model the only significant effects found were in canopy cover at 150 m ($\beta$ = -0.005, SE = 0.002, p<0.05), equating to a 0.5% decrease in songbird species richness with a one percent increase in canopy cover. It should be noted that the intercept model is within 2 AIC from the most supported model (AIC = 513.7).

#### *2.3.3.2 Beta diversity*

Sorenson dissimilarity values calculated for both grizzly and non-grizzly use sites show no difference in community composition of birds between used and random sites (Table 2.9).

#### *2.3.3.3 MetaMDS*

MetaMDS results for both songbird species composition relationships with activity type (e.g. foraging, carcass, bedding, and randomly selected) at the sampling site (Figure 2.10) and sex of bear associated with sampling site (Figure 2.11), show no clustering of species based on either factor.

## 2.4 Discussion

Our broadest scale of assessment, referred to as an umbrella effect, demonstrated a relationship between bird species richness and locations of grizzly bears at an individual home range scale throughout their provincial range and core areas, when compared to other flagship species in Alberta. This relationship between grizzlies and songbirds, however, was not as distinct when the planning unit scale was decreased and analyses were restricted to the Yellowhead Bear Management Area. At our intermediate scale of 1 $km^2$ planning units – where grizzly resource selection factor represented habitat selection, a significant, but only slightly positive relationship was found. This relationship was no longer apparent at our smallest, 150 m radius (localized) scale – where habitat selection was derived from bear collar data. Changes in strength and relationships with scale were anticipated given the widely known effect of scale on species co-occurrence (e.g. Wiens 1989) and as noted in a meta-analysis of surrogacy by Wolters et al. (2006), stronger relationships in cross-taxon surrogacy are more likely at larger scales (> 1 $km^2$) – a finding mirroring the pattern observed in this study and others (e.g. Burrascano et al. 2018). This has been attributed to the species-area relationship (Favreau et al. 2006); however, the more complex nature of spatial scale and species co-occurrences relates to responses of species to environmental factors applicable to different scales – e.g. broad climatic factors at large scales vs. micro-topography at smaller scales (Burrascano et al. 2018). Given my findings of discrepancy between scales and their relationships to songbird species richness, I will address each scale and the environmental factors that may be influencing the results. I will also discuss the analyses of space undertaken at the umbrella scale, which illustrate the variability of surrogacy across large areas.

The effects of spatial scale are apparent in the factors which could be impacting species richness at our umbrella scale (e.g. the large spatial scale is impacted by factors relevant to its size); however, within this scale, variation in physical location (space) is also important in explaining species richness variation. At the umbrella scale, where individual home ranges acted as the spatial units – across provincial ranges and core areas, the relationships between songbirds and planning unit size and location were examined independent of each other. First, isolating the size of planning unit from the effects of spatial variation, all planning unit sizes above 25 $km^2$ had significantly higher species richness increasing until an area of 100 $km^2$ to 338 $km^2$ (grizzly) – independent of location. This reflects the well-known principle of species-area relationships (Cain 1938). Assuming the goal is to have the highest songbird species richness within the least area, the grizzly bear sized planning unit outperformed the other flagship species, as the 338 $km^2$ grizzly planning unit holds an equivalent amount of songbird species richness as the larger 382 $km^2$ caribou planning unit. In our analysis of flagship ranges, independent of planning unit size, clear differences were observed between the three species examined, which is likely driven by environmental spatial heterogeneity associated with these regions/ranges. Spatial heterogeneity is a well-known factor in shaping patterns in bird diversity (Roth 1976; Bohning-Gaese 1997; Rahbek and Graves 2001; Hurlbert and Haskell 2003). For example, the range in elevation (per planning unit) for the woodland caribou and grizzly bear ranges are approximately seven times higher than that of greater sage-grouse range ($range_{GB}$ = 2461 m; $range_{WC}$ = 2482 m; $range_{GSG}$ = 340 m). Greater elevation ranges result in more climatic variation and thus vegetation variability which leads to higher bird diversity (Ruggiero and Hawkins 2008). Lower differences in elevation within the greater sage-grouse range suggest less spatial heterogeneity in the environment and thus helps explain its poor performance as an umbrella for songbird diversity.

Regardless, there are some important conservation values of habitat associated with greater sage-grouse including its shared use by other sagebrush species (Rowland et al. 2006; Carlisle et al. 2018). Although sage-grouse spatial heterogeneity is low, elevation ranges between grizzly bear and woodland caribou ranges were quite similar, so why was species richness higher for grizzly bears? Interestingly, a recent study by Coops et al. (2016) used LIDAR-based forest structure data (e.g. tree height) with climatic data to index bird species richness for much of the caribou and grizzly bear ranges and showed a clear pattern of higher bird richness in the western part of the province and areas of low richness in the east. This is similar to our results. The Foothills and Rocky Mountain natural regions, which account for ~25 and 20 % of grizzly range in Alberta, respectively, have the greatest topographical variation (Natural Regions Committee 2006). In contrast, the Boreal region, while representing ~ 50% of grizzly range, is the predominant natural region of the caribou range (~85%). The Boreal region, while exhibiting some topographical variation is more strongly associated with subtle variations in topography and less climatic variation (Natural Regions Committee 2006). Topographical variation is related to vegetation diversity at numerous scales (Riera et al. 1998; Opedal et al. 2015; Irl et al. 2015), which is in turn influences vertebrate diversity (Fraser 1998) - a relationship described by Guo et al. (2017). It is important to note that there were slightly higher beta Sorensen values detected for woodland caribou core habitat compared to grizzly core, however these values are not accompanied with significance values and therefore the species richness evaluations are treated as more informative for the conclusions drawn. Aside from variation between the flagship species ranges, within the grizzly bear provincial range one could anticipate overall variation between the BMAs in terms of both richness and beta diversity given the high degree of topographic and vegetative variation over an area as large as the grizzly bear provincial range – for context, this range contains 14 different natural subregions. For example, there is a general pattern of higher overall species richness in the southern BMAs and lower in the most northern BMA – as could be anticipated given the high plant diversity of the Rocky Mountains (Zhang et al. 2015) and their exclusion from the most northern BMA. Beta diversity, however, was generally higher in the northern BMAs than the southern – an unexpected result given that montane regions tend to exhibit higher beta diversity due to variations in altitude (Dobrovolski et al. 2011; Melo et al. 2009). Regardless of the contrary results, this illustrates that beyond spatial scale, variation in space (location) is also influential – a concept that is intuitive, but given the large area covered by this scale of analysis and the management implications of this data, is important to note.

At our localized and intermediate scales there are other factors, aside from general topographic variability, that relate to specific habitat selection which could be affecting the strength of the surrogate relationships observed. Habitat selection by grizzly bears is driven by numerous factors. For instance, local abundance of bears and habitat selection can be affected by local habitat quality related to the combined presence of ungulates and fruit (Nielsen et al. 2017), presence of edges (anthropogenic and natural) and clear-cuts – with variations based on sex, season, and time of day (Nielsen et al. 2004a, 2004b; Stewart et al. 2013). The type of grizzly bear activity also affects their habitat selection, with mixed/open forests being preferred for fruit foraging, forested areas preferred for bedding, and various types of forest and non-vegetated areas being associated with kill sites (Munro et al. 2006). Unlike grizzly bears, songbirds, particularly those in the shrub and ground nesting/foraging guilds, may be negatively affected by some factors known to benefit bears, such as the presence of deer, which can reduce vegetative cover that songbirds depend on (Teichman et al. 2013, Holt et al. 2011); however, other factors such as individual songbird species habitat preferences and therefore their responses to the creation of edges add further complications to the explanation of the surrogate relationship. Forest interior birds would decline with increases in anthropogenic edges, while early successional species of birds, or those that are synanthropic, can increase near edges (Farwell et al. 2016). Bird diversity can therefore be higher in the early successional stages of clear-cut forests due to higher numbers of edge species (Keller et al. 2003). Given these diverse relationships between songbirds and habitats and the breadth of factors which can contribute to grizzly bear habitat selection, the interaction between songbirds and grizzly bear habitat is complex and difficult to generalize precise causes of the positive relationship at the intermediate scale and lack of relationships at a localized scale.

### 2.5 Conclusion

Scale affected the relationship between grizzly bear habitat and songbird richness, therefore affecting the potential for grizzly bears to act as a surrogate species for songbirds. This adds to the growing body of literature emphasizing the integration of spatial scale into surrogacy studies (e.g. Favreau et al. 2006). While previous studies have suggested or even evaluated grizzly bears as a candidate surrogate species, and in particular umbrella species (Noss 1996; Carroll et al. 2001), these studies have not quantitatively integrated scale variation.

Table 2.1 Songbird species richness and grizzly bear resource selection functions (RSF) are compared using general linear mixed models (GLMM) at the intermediate scale. Models were all evaluated using a negative binomial distribution with sub watershed used as a random effect.

| Model | Model name | Fixed Effects (continuous) | Random Effects |
|---|---|---|---|
| 1 | Max RSF | Max RSF | Sub-watershed |
| 2 | S1 RSF | Spring RSF | Sub-watershed |
| 3 | S2 RSF | Summer RSF | Sub-watershed |
| 4 | S3 RSF | Fall RSF | Sub-watershed |
| 5 | Intercept | | Sub-watershed |

Table 2.2. Songbird richness is compared to grizzly bear habitat selection factors using general linear mixed models (GLMM) at the localized scale of analysis. Models were all evaluated using a Poisson distribution with pairs of sites as a random effect and the status as a confirmed grizzly use or random site as a fixed effect.

| Model | Model name | Fixed Effects (continuous) | Fixed Effects (categorical) | Random Effects |
|---|---|---|---|---|
| 1 | Vegetation Type Cover (150 m) | Upland Treed<br>Upland Herb<br>Shrub<br>Barren<br>Anthropogenic<br>Wetland Herb<br>Wetland Treed | Grizzly Use / Random | Pairs |
| 2 | Canopy Cover (150 m) | Canopy Cover | Grizzly Use / Random | Pairs |
| 3 | Bear | Days Since May 1st | Grizzly Use / Random<br>Bear Activity Type<br>Bear ID<br>Sex of Bear | Pairs |
| 4 | Paired | | Grizzly Use / Random | Pairs |
| 5 | Max RSF | Max RSF | Grizzly Use / Random | Pairs |
| 6 | S1 RSF | Spring RSF | Grizzly Use / Random | Pairs |
| 7 | S2 RSF | Summer RSF | Grizzly Use / Random | Pairs |
| 8 | S3 RSF | Fall RSF | Grizzly Use / Random | Pairs |
| 9 | Intercept | | | Pairs |

Table 2.3. Sorensen (Beta Sor) values for the planning units within grizzly bear and woodland caribou core habitat in Alberta, Canada.

| **Planning Unit Location and Species Group** | **Beta Sim (turnover)** | **Beta SNE (nestedness)** | **Beta Sor** |
|---|---|---|---|
| All Songbird Species | | | |
| Grizzly Bear Core | 0.037 | 0.231 | 0.267 |
| Woodland Caribou Core | 0.122 | 0.174 | 0.296 |
| | | | |
| Songbird Species at Risk | | | |
| Grizzly Bear Core | 0.067 | 0.241 | 0.308 |
| Woodland Caribou Core | 0.139 | 0.202 | 0.341 |

Table 2.4. Sorensen (Beta Sor) values for the planning units within Bear Management Areas in western Alberta, Canada.

| Planning Unit Location and Species Group | Beta Sim (turnover) | Beta SNE (nestedness) | Beta Sor |
|---|---|---|---|
| All Songbird Species | | | |
| AB North | 0.286 | 0.497 | 0.782 |
| Grande Cache | 0.226 | 0.296 | 0.522 |
| Swan Hills | 0.020 | 0.172 | 0.192 |
| Yellowhead | 0.063 | 0.547 | 0.611 |
| Clearwater | 0.011 | 0.495 | 0.506 |
| Livingstone | 0.007 | 0.128 | 0.135 |
| Castle | 0.023 | 0.035 | 0.058 |
| | | | |
| Songbird Species at Risk | | | |
| AB North | 0.340 | 0.479 | 0.820 |
| Grande Cache | 0.297 | 0.345 | 0.641 |
| Swan Hills | 0.049 | 0.331 | 0.380 |
| Yellowhead | 0.175 | 0.508 | 0.682 |
| Clearwater | 0.004 | 0.569 | 0.573 |
| Livingstone | 0.015 | 0.263 | 0.278 |
| Castle | 0.044 | 0.089 | 0.133 |

Table 2.5. AIC values for five tested general linear mixed count (negative binomial) models. Models examine the relationships relationship between bear resource selection function (RSF) values for three seasons - spring (S1), summer (S2), fall (S3) - and the maximum across all seasons (MAX) with that of songbird species richness in the Yellowhead Bear Management Area of Alberta, Canada. Detailed descriptions of models available in Table 2.1.

| **Model** | **AIC** | **logLikelihood** |
|---|---|---|
| MAX RSF | 51678.2 | -25849.1 |
| RSF S3 | 51704.4 | -25848.2 |
| RSF S1 | 51706.2 | -25849.1 |
| RSF S2 | 51706.9 | -25849.4 |
| Intercept | 51815.6 | -25904.8 |

Table 2.6. Results of a negative binomial general linear mixed models (GLMM) that examine the relationship between bear resource selection function (RSF) values for three seasons - spring (S1), summer (S2), fall (S3) - and the maximum across all seasons (MAX) with that of songbird species richness in the Yellowhead Bear Management Area of Alberta, Canada, as well as the intercept model.

| Model | Variable | Beta | Standard Error | p-Value |
|---|---|---|---|---|
| Max RSF | Intercept | 4.035 | 0.097 | <0.001 |
| | MAX RSF | 0.050 | 0.004 | <0.001 |
| RSF S1 | Intercept | 4.066 | 0.096 | <0.001 |
| | RSF S1 | 0.046 | 0.004 | <0.001 |
| RSF S2 | Intercept | 4.071 | 0.097 | <0.001 |
| | RSF S2 | 0.044 | 0.004 | <0.001 |
| RSF S3 | Intercept | 4.067 | 0.097 | <0.001 |
| | RSF S3 | 0.045 | 0.004 | <0.001 |
| Intercept | Intercept | 4.279 | 0.084 | <0.001 |

Table 2.7. AIC values for nine tested general linear mixed count (Poisson) models. Models examine the relationships between factors associated with bear use and random sites with that of songbird species richness (count). Detailed descriptions of models available in Table 2.2.

| **Model** | **AIC** | **logLikelihood** |
|---|---|---|
| Canopy cover at 150 m radius | 512.0 | -252.0 |
| Intercept | 513.7 | -254.9 |
| Pair | 515.4 | -254.7 |
| Max RSF at 150 m radius | 515.9 | -253.9 |
| S2 RSF at 150 m radius | 515.9 | -253.9 |
| Max RSF at 150 m radius | 515.9 | -253.9 |
| S3 RSF at 150 m radius | 516.0 | -254.0 |
| S1 RSF at 150 m radius | 516.7 | -254.3 |
| Bear | 520.4 | -252.2 |
| Percent vegetation type at 150 m radius | 522.6 | -252.3 |

Table 2.8. Results of general linear mixed model analysis of nine Poisson models. Models examine the relationships between factors associated with bear use and random sites with that of songbird species richness (counts). Detailed descriptions of models available in Table 2.2.

| Model | | Variable | Beta | Standard Error | p-Value |
|---|---|---|---|---|---|
| Model 1 | Vegetation Type Cover (150 m) | Intercept | 1.183 | 0.107 | <0.001 |
| | | Paired (unused) | -0.060 | 0.095 | 0.528 |
| | | Upland Herb | 0.002 | 0.008 | 0.793 |
| | | Shrub | 0.004 | 0.005 | 0.483 |
| | | Barren | -0.008 | 0.013 | 0.532 |
| | | Wetland Herb | 0.038 | 0.026 | 0.137 |
| | | Wetland Trees | -0.016 | 0.018 | 0.373 |
| | | Anthropogenic | 0.002 | 0.001 | 0.189 |
| Model 2 | Canopy Cover (150 m) | Intercept | 1.461 | 0.106 | <0.001 |
| | | Paired (unused) | -0.073 | 0.094 | 0.440 |
| | | Canopy Cover | -0.005 | 0.002 | <0.05 |
| Model 3 | Bear | Intercept | 2.103 | 0.847 | <0.05 |
| | | Paired (unused) | -0.052 | 0.094 | 0.579 |
| | | Carcass | 0.010 | 0.157 | 0.949 |
| | | Foraging | 0.054 | 0.133 | 0.683 |
| | | Bear ID | -0.004 | 0.005 | 0.426 |
| | | Gender (Female) | 0.111 | 0.110 | 0.311 |
| | | Days Since May 1 | -0.005 | 0.003 | 0.066 |
| Model 4 | Pair | Intercept | 1.264 | 0.070 | <0.001 |
| | | Paired (unused) | -0.053 | 0.094 | 0.573 |
| Model 5 | MAX RSF (150 m) | Intercept | 0.978 | 0.244 | <0.001 |
| | | Paired (unused) | -0.067 | 0.095 | 0.480 |
| | | Max RSF | 0.038 | 0.031 | 0.217 |
| Model 6 | S1 RSF (150 m) | Intercept | 1.071 | 0.241 | <0.001 |
| | | Paired (unused) | -0.057 | 0.094 | 0.543 |
| | | S1 RSF | 0.027 | 0.031 | 0.399 |
| Model 7 | S2 RSF (150 m) | Intercept | 1.001 | 0.228 | <0.001 |
| | | Paired (unused) | -0.069 | 0.095 | 0.466 |
| | | S2 RSF | 0.037 | 0.030 | 0.221 |
| Model 8 | S3 RSF (150 m) | Intercept | 1.013 | 0.228 | <0.001 |
| | | Paired (unused) | -0.067 | 0.095 | 0.478 |
| | | S3 RSF | 0.035 | 0.030 | 0.243 |
| Model 9 | Intercept | Intercept | 1.238 | 0.053 | <0.001 |

Table 2.9. Sorensen (Beta Sor) values for grizzly use and random location sites based on songbird species richness (counts).

| Variable | Beta Sim (turnover) | Beta SNE (nestedness) | Beta Sor |
|---|---|---|---|
| Grizzly | 0.943 | 0.021 | 0.964 |
| Random | 0.940 | 0.023 | 0.964 |

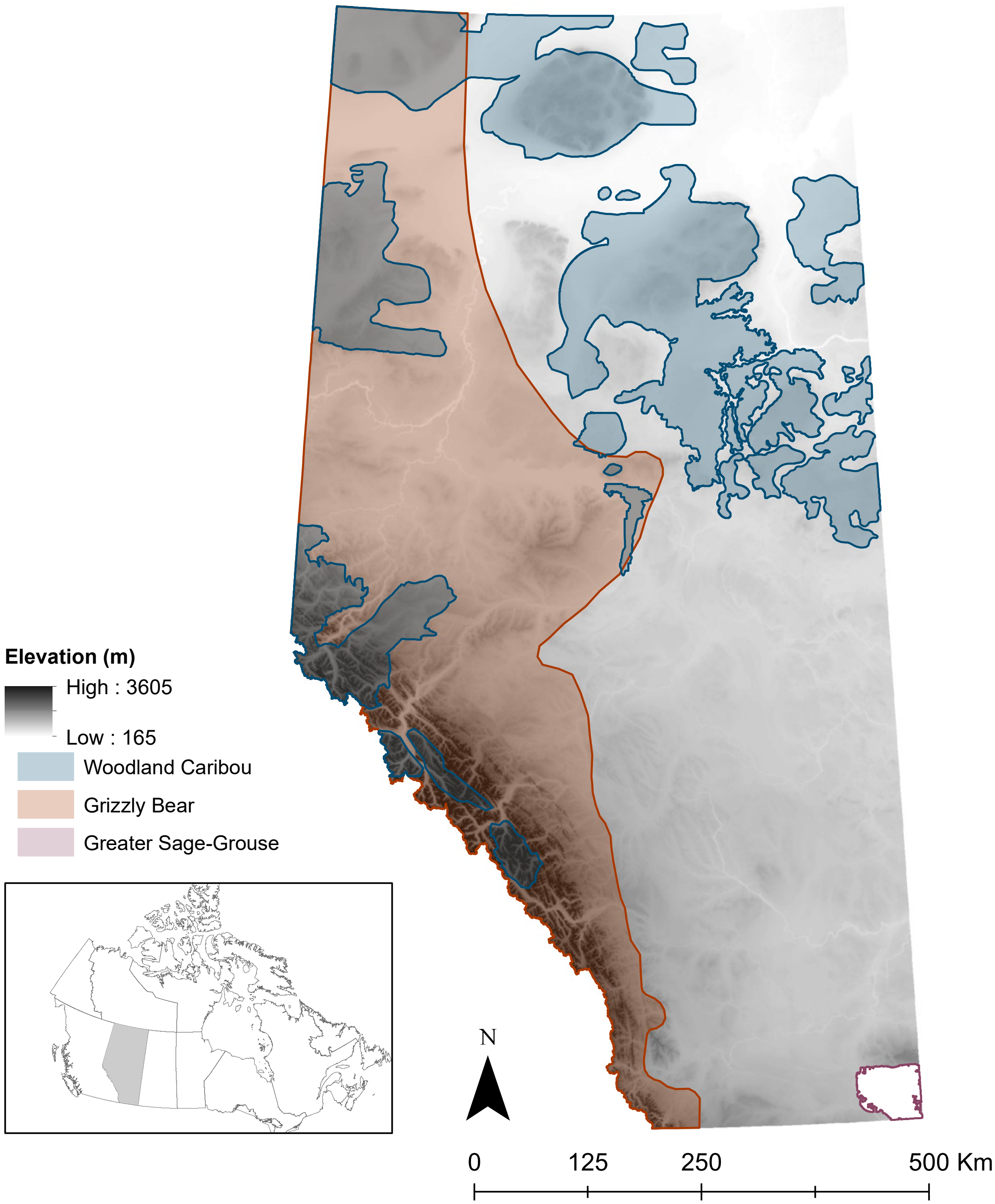


Figure 2.1. Provincial species ranges of three flagship species in Alberta, Canada: woodland caribou, greater sage-grouse, and grizzly bear.

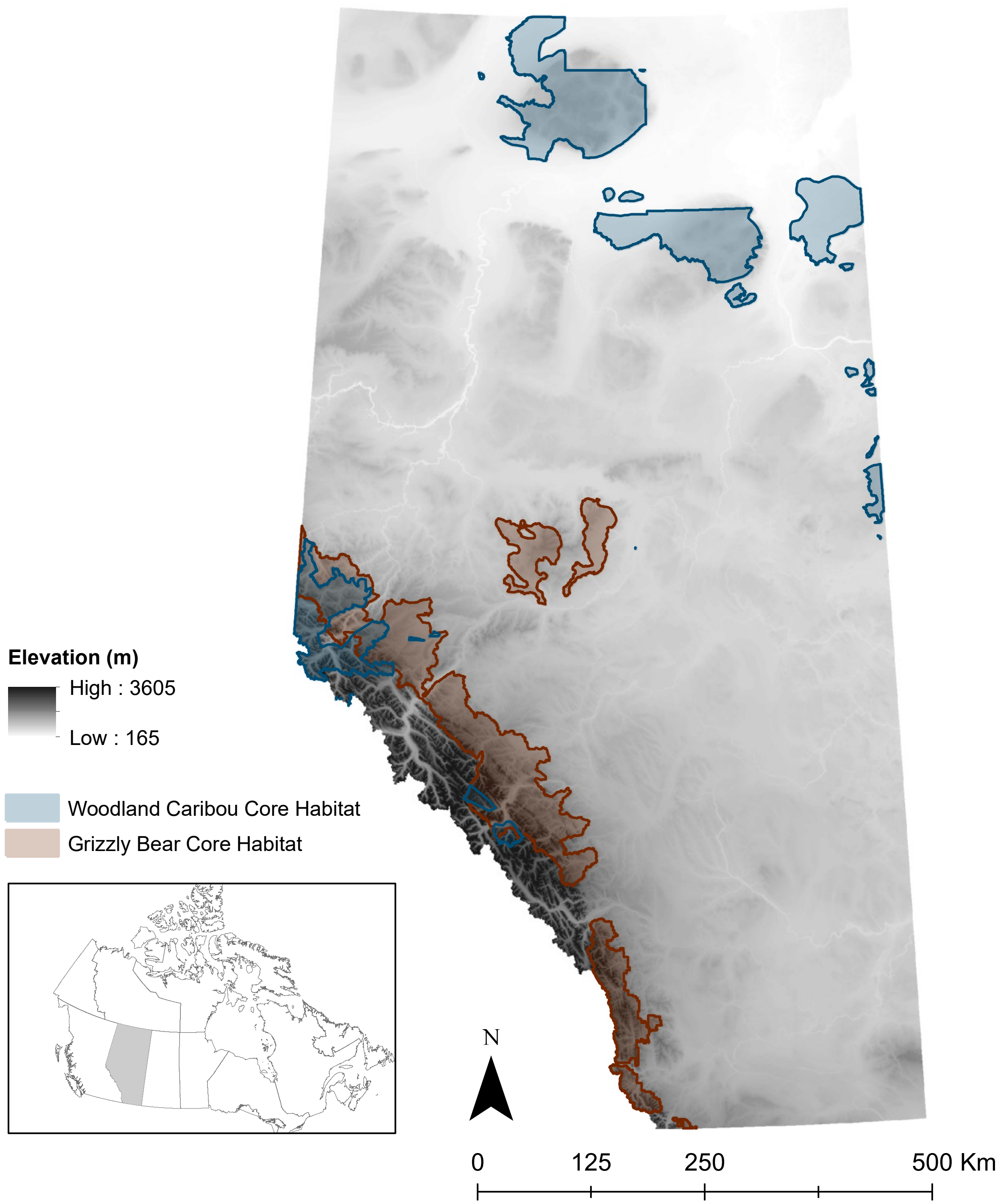


Figure 2.2. Core habitat for grizzly bear and woodland caribou. Grizzly bear core habitat represents areas of low road density and high-quality grizzly bear habitat. Caribou core areas were derived from federal government habitat disturbance thresholds.

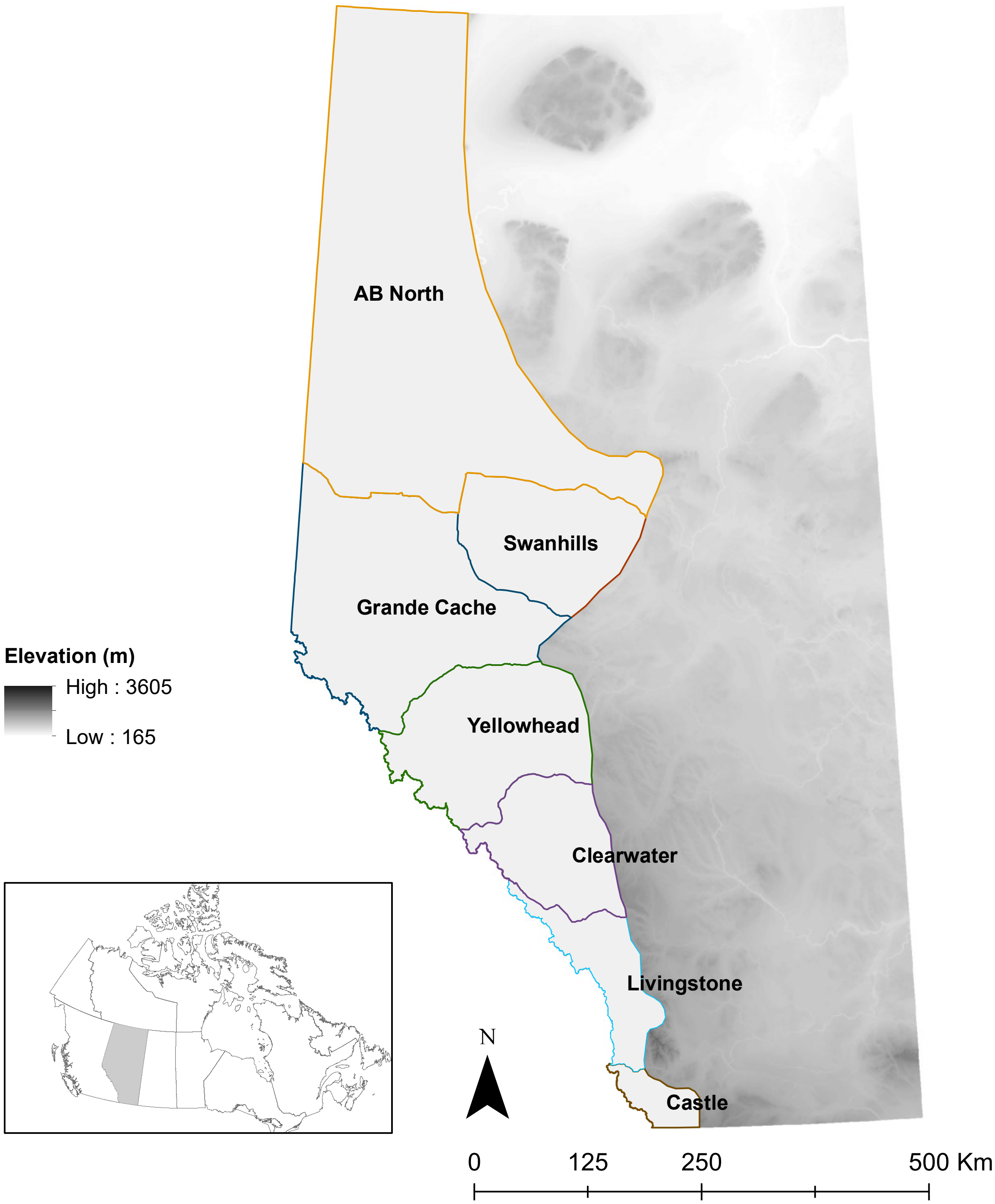


Figure 2.3. Bear Management Areas for Alberta, Canada grizzly bear range.

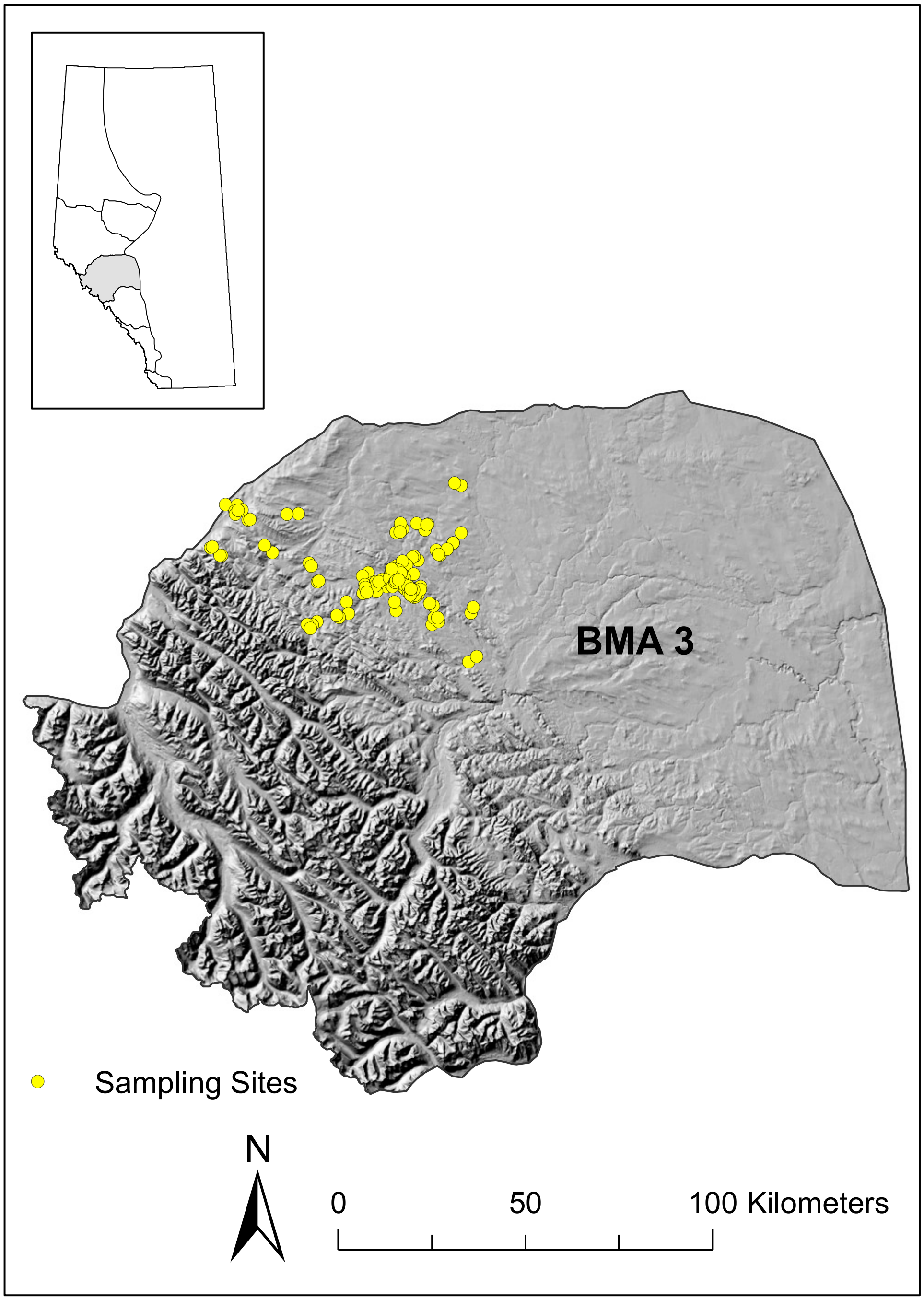


Figure 2.4. Study locations within the Bear Management Area (BMA) 3 / Yellowhead population unit in Alberta, Canada. Yellow circular symbols represent the 130 field sample sites used in this study.

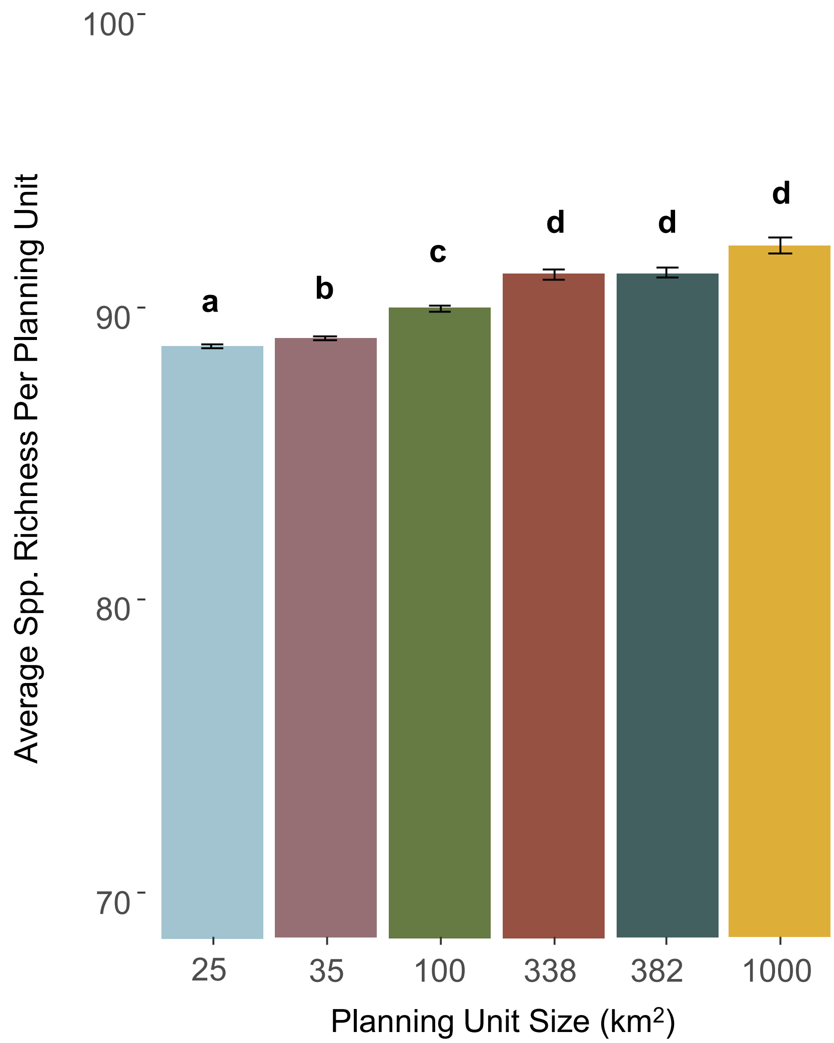


Figure 2.5. Comparisons of the average species richness per planning unit for six different sized (in km$^2$) planning units across Alberta, Canada. Planning unit sizes for greater sage-grouse, grizzly bear, and woodland caribou individual home ranges are represented by 35 km$^2$ (n = 18 300), 338 km$^2$ (n = 1841), and 382 km$^2$ (n = 1630), respectively, while the 1000 km$^2$ (n = 597), 100 km$^2$ (n = 6409), and 25 km$^2$ (n = 26055) sizes are meant for additional sampling of scales.

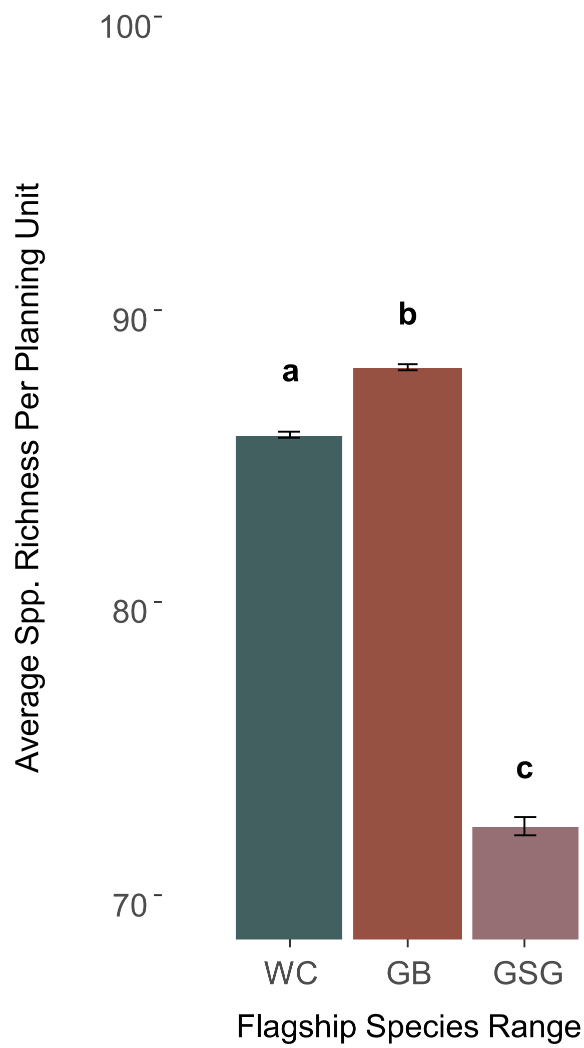


Figure 2.6. Comparisons of the species richness per planning unit for 25 km$^2$ sized planning units placed in three different flagship species ranges. Planning units were within with greater sage-grouse (GSG) (n = 104), grizzly bear (GB) (n = 9199), and woodland caribou (WC) range (n = 4861), respectively.

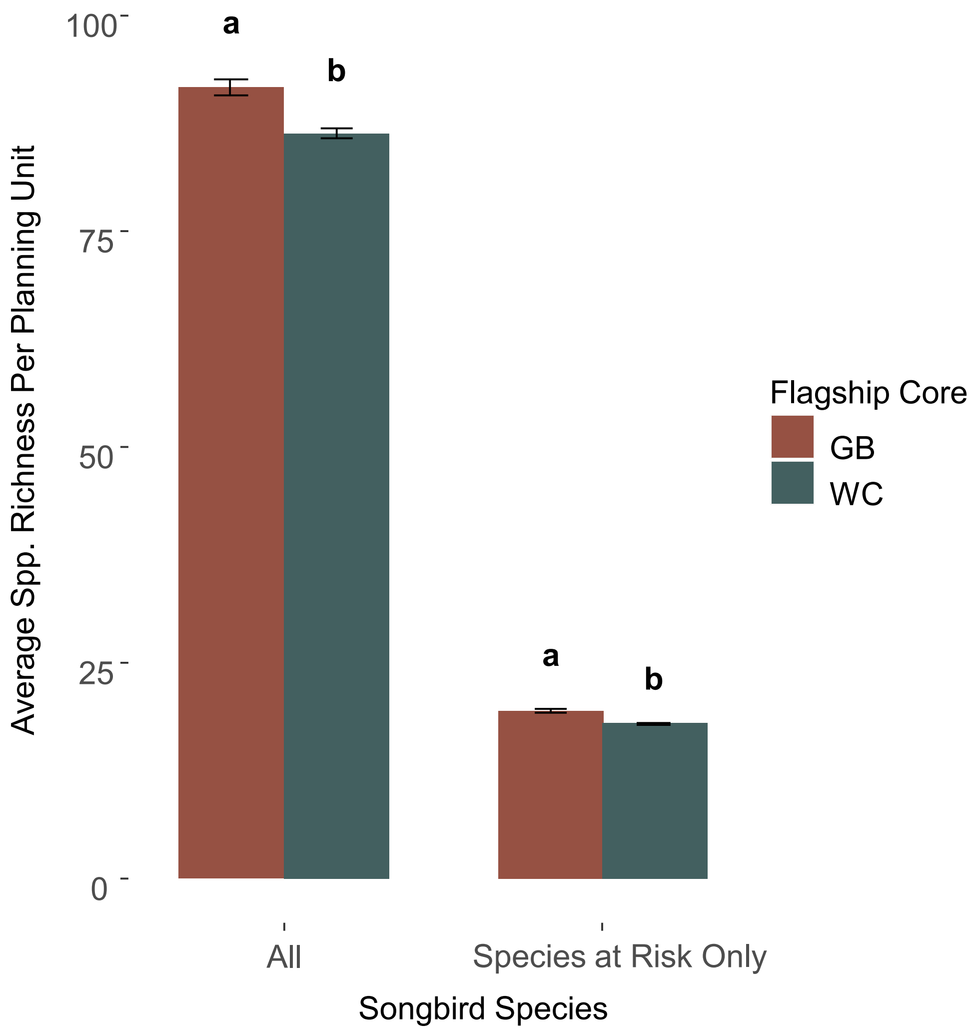


Figure 2.7. Comparisons of the species richness per planning unit for planning units placed in grizzly bear (GB) (n = 32) and woodland caribou (WC) core habitat (n = 28).

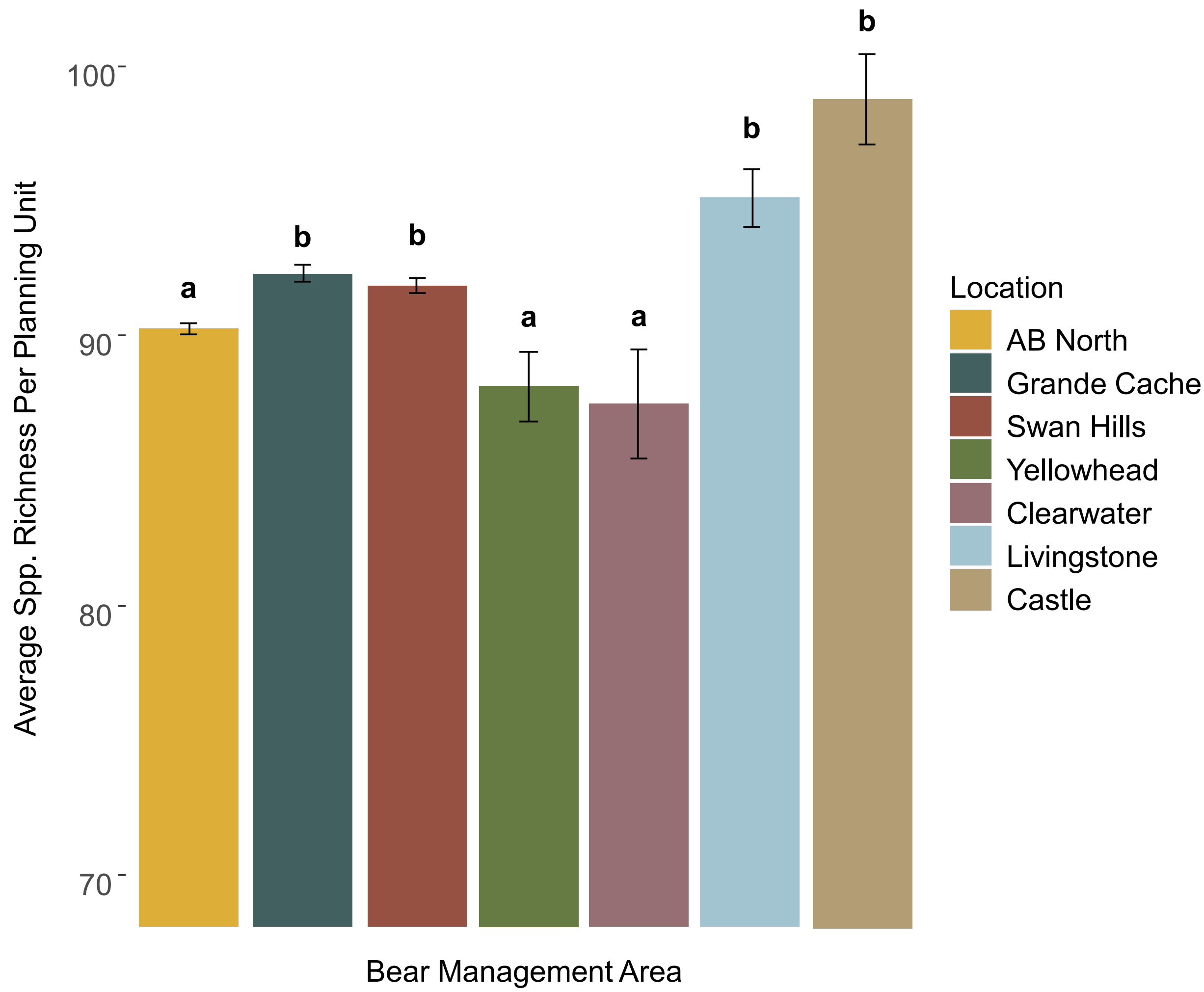


Figure 2.8. Comparisons of the songbird species richness per 338 $km^2$ planning unit (grizzly bear individual home range size) for Alberta, Canada Bear Management Areas.

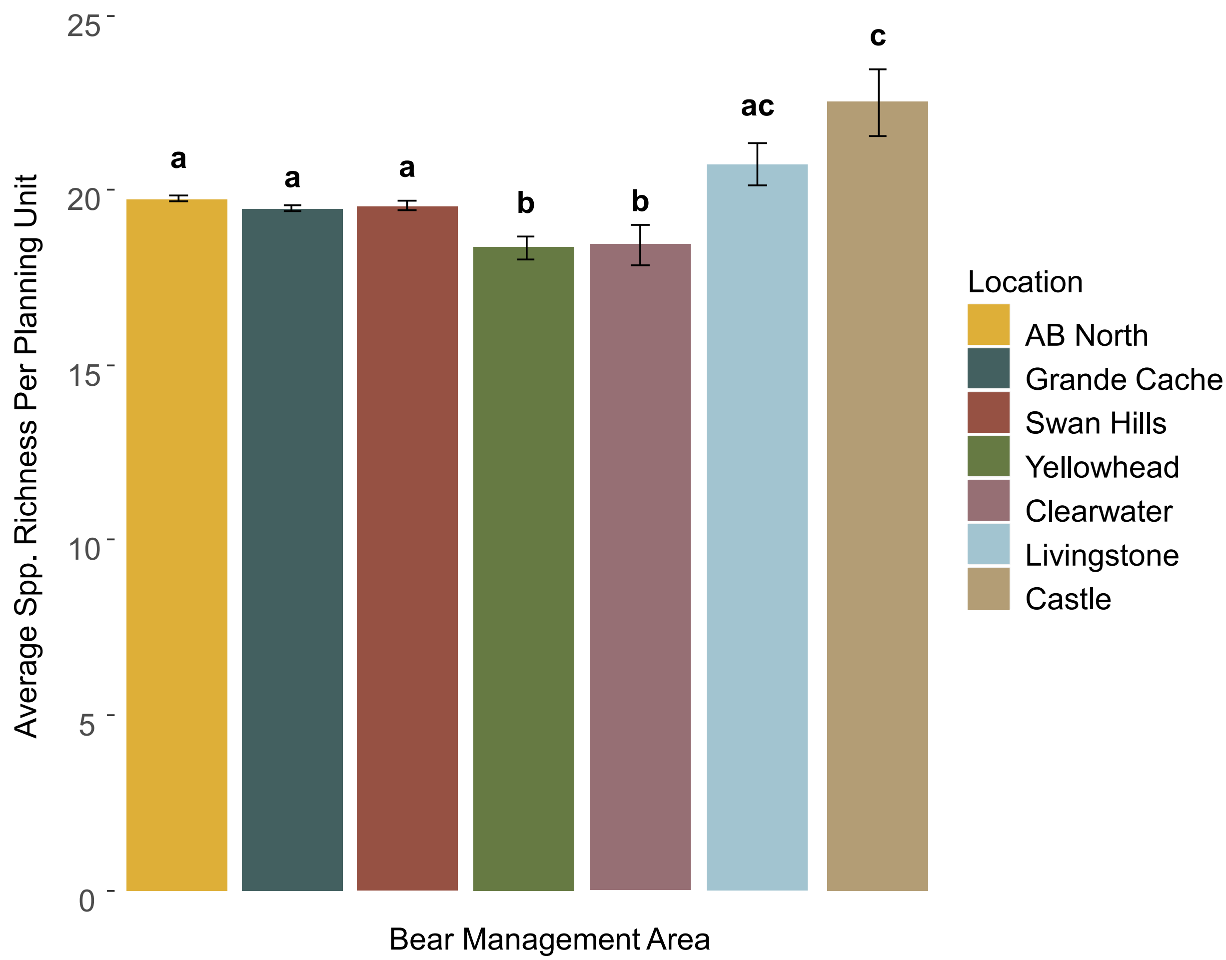


Figure 2.9. Comparisons of the songbird species richness per 338 $km^2$ planning unit (grizzly bear individual home range size) for Alberta, Canada Bear Management Areas, using only Species at Risk.

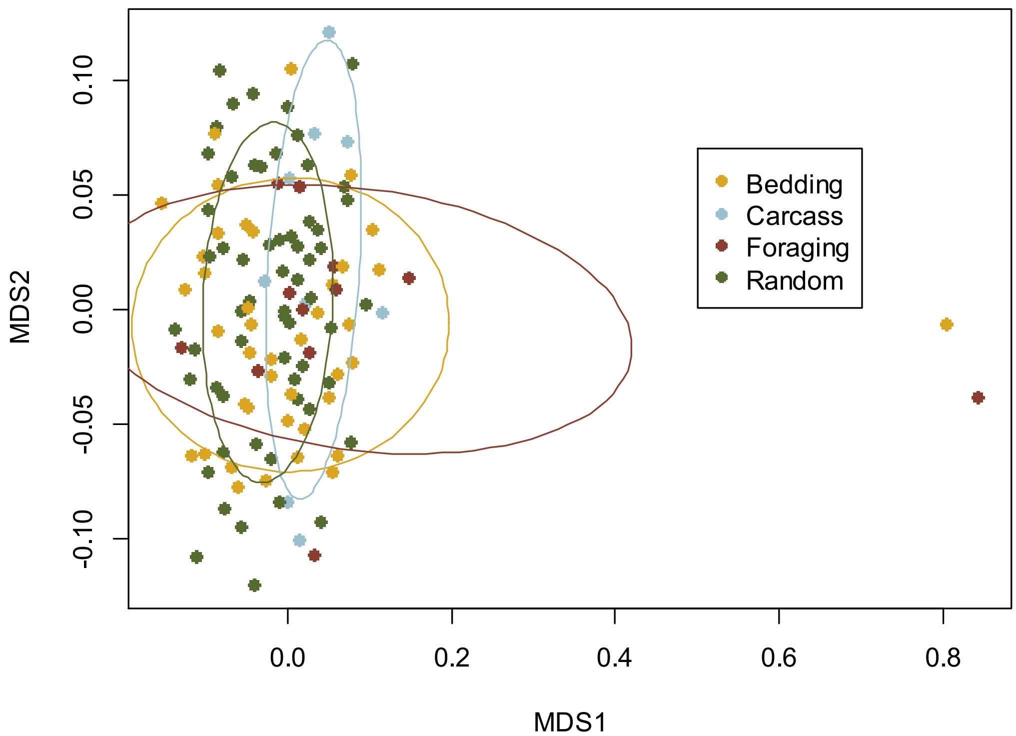


Figure 2.10. MetaMDS of songbird species composition at field sampling sites, with associated bear activity as a grouping mechanism. “Random” designation is attributed to randomly selected sites.

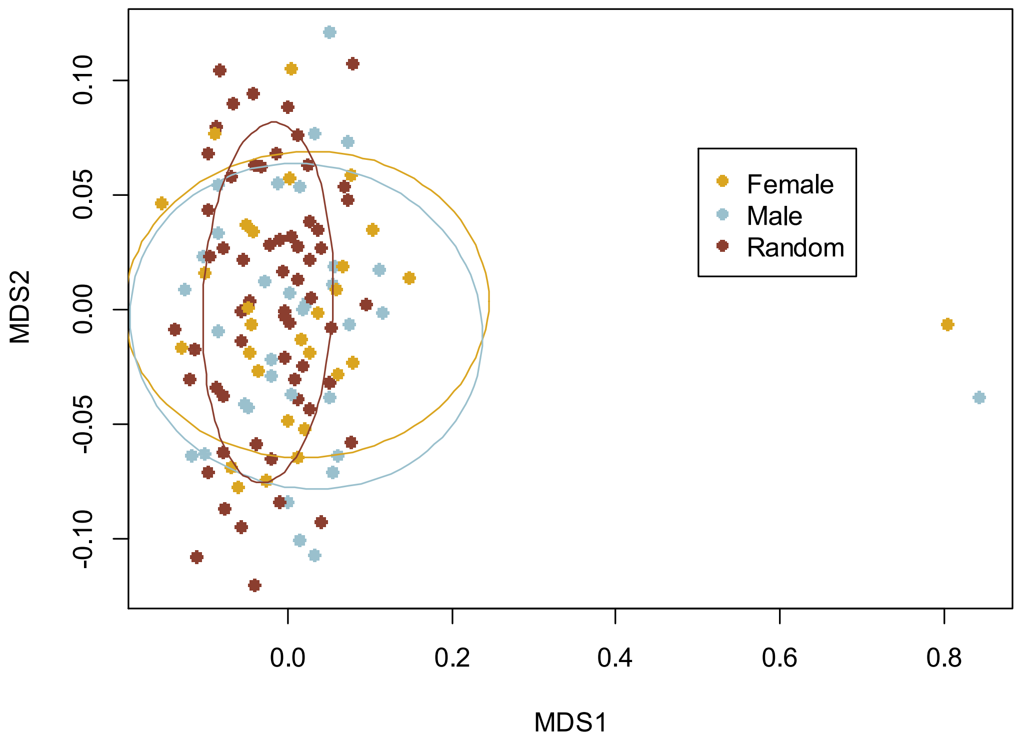


Figure 2.11. MetaMDS of songbird species composition at field sampling sites, with sex of bear at sampling sites as the grouping mechanism. "Random" designation is attributed to randomly selected sites.

## Chapter 3: General Conclusions

### 3.1 Summary

The broad objective of this study was to determine whether grizzly bears are an adequate surrogate species for songbirds, while accounting for and comparing the effects of spatial scales on its efficacy. Overall, I found that grizzly bears have the potential to serve as a surrogate species within their Alberta range – with conditions. Variation in spatial scale was highly influential on the strength of the co-occurrence of grizzlies and songbirds, resulting in grizzly bears being an effective umbrella species (large scales), but lacking a clear relationship at intermediate to smaller scales.

### 3.2 Management implications

Results from this study provide guidance to land managers and environmental organizations using flagship species for conservation initiatives. In both industry and government, land management and project planning require consideration of many environmental factors. For industry, this includes not only biophysical components, but also numerous other protected wildlife species utilizing the landscape. This complexity leads to a natural question of "Can this process be simplified?" The results of this study suggest that perhaps grizzly bear can be used as a surrogate species to assist in the management of songbirds within Alberta at large scales; however, aside from whether or not there is overlap between the species in question (here grizzly bears and songbirds), there are other considerations that need to be addressed when determining whether a species could act as an adequate umbrella. This includes the criteria proposed by Seddon and Leech (2008). For example, they indicate, logically, that the management of the umbrella and their habitat must also benefit the species under said "umbrella"; therefore, it would have to be considered whether the management actions undertaken to benefit the grizzly bear, such as limiting open access roads (publicly accessible) in grizzly habitat, with stricter guidelines in key areas (e.g. core habitat), and banning grizzly bear hunting (ASRD 2008) would have a positive, if any effect on the bird diversity of the shared habitat. This consideration would be tied to whether the land managing entity is using this information in the effort to conserve songbirds incidentally – i.e. focusing conservation on the grizzly flagship, in which case the above criteria would be of high importance – or, creating conservation areas for songbirds within the flagship habitat. Additionally, a key consideration in the comparison of the grizzly bear and caribou as umbrella species may lie in the proposed criteria of population persistence (Seddon and Leech 2008). The grizzly bear, a Canadian species of *Special Concern* (COSEWIC 2012), has shown signs of recent population increases (Stenhouse et al. 2015), whereas the *Threatened* (Boreal) (COSEWIC 2014a) / *Endangered* (Central Mountain) (COSEWIC 2014b) woodland caribou's persistence is questionable given industrial developments and current predation rates (Festa-Bianchet et al. 2011; Hervieux et al. 2013). Therefore, although the grizzly bear meets some suggested umbrella species guidelines, there are other aspects that would require further investigation prior to its adoption as an umbrella species for conservation and management of biodiversity in Alberta. It is important to note that in addition to selecting a surrogate species considering multiple factors (including those listed above), it is also necessary to monitor for efficacy following the implementation of a surrogate-based management action (Favreau 2006; Wiens 2008).

### 3.3 Future research

This study addresses a basic question regarding grizzly bears as songbird surrogates, providing a general understanding of the relationship between grizzly bear habitat and songbird richness;

however, there is the potential to build upon this idea and provide more detailed information and recommendations. I acknowledge that while this study addresses species diversity largely via species richness, other measures such as species abundance could be integrated to expand on my findings. Additionally, it is assumed in this study that high diversity is the goal, this may not always be the case because some species poor areas may overlap with rare species (concept of complementarity); therefore, future studies could seek to identify individual songbird species relationships with grizzly habitat – particularly those of species at risk.

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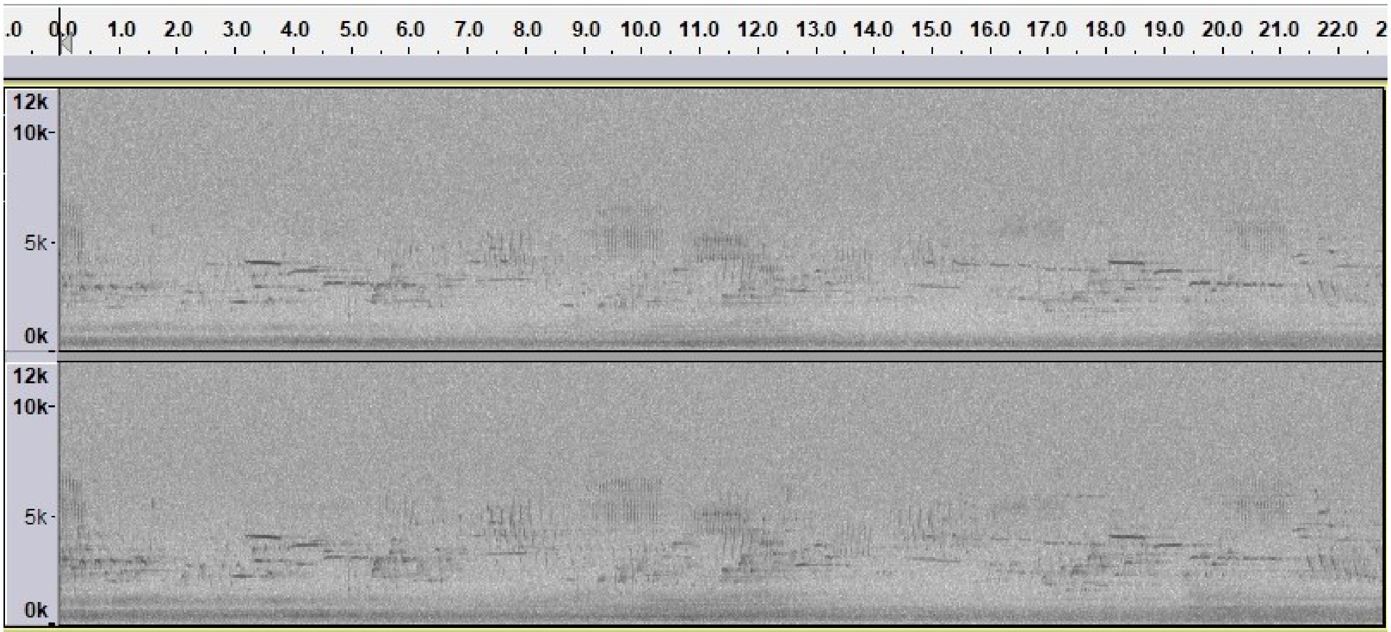
**Appendix**

Appendix 1. Description of 130 field sites (65 pairs) near Hinton, Alberta, Canada used in the analysis of the grizzly bear and songbird richness localized scale surrogate relationship. Bear Use represents sites with known bear activity determined via telemetry data by fRI. The Random designation represents sites randomly generated between 500 and 3000 m from the paired Bear Use site. Songbird species richness was assessed at each site using 3 minute dawn chorus recordings, captured using autonomous recording units (ARUs).

| | Bear Use | | | | | Random | | |
|---|---|---|---|---|---|---|---|---|
| **Site Pair Number** | **Sex of Bear** | **Bear Activity** | **Easting** | **Northing** | **Songbird Species Richness** | **Easting** | **Northing** | **Songbird Species Richness** |
| 1 | Male | Bedding | 454088 | 5897361 | 4 | 454593 | 5897306 | 4 |
| 2 | Male | Bedding | 452120 | 5899576 | 3 | 451658 | 5899380 | 5 |
| 3 | Female | Foraging | 499773 | 5892464 | 3 | 500797 | 5893242 | 4 |
| 4 | Female | Bedding | 500702 | 5884910 | 1 | 501130 | 5882525 | 6 |
| 5 | Male | Foraging | 492189 | 5891800 | 1 | 493650 | 5892676 | 1 |
| 6 | Male | Foraging | 504222 | 5889034 | 2 | 503213 | 5888654 | 2 |
| 7 | Male | Carcass | 522723 | 5870319 | 2 | 520524 | 5868934 | 2 |
| 8 | Male | Carcass | 516403 | 5900669 | 5 | 518522 | 5903412 | 4 |
| 9 | Male | Carcass | 503045 | 5904353 | 3 | 501101 | 5903464 | 2 |
| 10 | Male | Foraging | 516403 | 5900669 | 4 | 506962 | 5895967 | 6 |
| 11 | Male | Carcass | 516835 | 5916655 | 4 | 518560 | 5916016 | 2 |
| 12 | Male | Bedding | 509633 | 5905581 | 5 | 508830 | 5904087 | 4 |
| 13 | Male | Foraging | 502225 | 5903619 | 6 | 502390 | 5905897 | 2 |
| 14 | Male | Bedding | 505536 | 5896917 | 1 | 506023 | 5897128 | 6 |
| 15 | Male | Carcass | 500634 | 5891075 | 2 | 501942 | 5890017 | 4 |
| 16 | Male | Bedding | 507056 | 5886788 | 5 | 505933 | 5886182 | 2 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| 17 | Female | Bedding | 502143 | 5892843 | 2 | 503955 | 5894706 | 3 |
| 18 | Female | Bedding | 500141 | 5892215 | 2 | 500999 | 5893244 | 2 |
| 19 | Female | Bedding | 503646 | 5888349 | 4 | 502673 | 5889234 | 1 |
| 20 | Female | Foraging | 501707 | 5892513 | 4 | 501259 | 5894470 | 4 |
| 21 | Female | Foraging | 503523 | 5891186 | 4 | 505889 | 5892380 | 2 |
| 22 | Female | Bedding | 501696 | 5893179 | 5 | 502829 | 5895682 | 5 |
| 23 | Female | Foraging | 501409 | 5890614 | 4 | 499738 | 5891913 | 2 |
| 24 | Female | Bedding | 494550 | 5889853 | 4 | 492316 | 5887254 | 1 |
| 25 | Male | Bedding | 488501 | 5881758 | 6 | 488001 | 5884920 | 3 |
| 26 | Male | Bedding | 485305 | 5881303 | 5 | 485951 | 5880792 | 3 |
| 27 | Male | Carcass | 495671 | 5890175 | 5 | 493189 | 5888514 | 4 |
| 28 | Male | Carcass | 495693 | 5890185 | 4 | 492794 | 5889450 | 4 |
| 29 | Male | Foraging | 497735 | 5890220 | 5 | 498641 | 5888913 | 2 |
| 30 | Female | Bedding | 501275 | 5890408 | 3 | 500900 | 5891510 | 3 |
| 31 | Male | Foraging | 502678 | 5889854 | 5 | 501945 | 5889973 | 2 |
| 32 | Male | Bedding | 512881 | 5897524 | 4 | 511879 | 5898719 | 5 |
| 33 | Male | Carcass | 521805 | 5883422 | 3 | 521134 | 5881927 | 4 |
| 34 | Female | Bedding | 501892 | 5892642 | 3 | 501637 | 5893164 | 3 |
| 35 | Female | Bedding | 495764 | 5889186 | 1 | 495981 | 5887667 | 1 |
| 36 | Female | Bedding | 506383 | 5886302 | 6 | 504650 | 5887387 | 3 |
| 37 | Male | Bedding | 480605 | 5890543 | 0 | 480114 | 5890237 | 1 |
| 38 | Female | Bedding | 499861 | 5892399 | 4 | 500994 | 5893581 | 4 |
| 39 | Female | Bedding | 501492 | 5889936 | 2 | 503034 | 5892117 | 3 |
| 40 | Female | Bedding | 500457 | 5891120 | 4 | 501869 | 5893341 | 2 |
| 41 | Male | Bedding | 512469 | 5897613 | 7 | 514823 | 5898926 | 6 |
| 42 | Male | Bedding | 509319 | 5905459 | 9 | 506579 | 5905904 | 7 |
| 43 | Male | Foraging | 512488 | 5879694 | 5 | 510751 | 5878904 | 6 |
| 44 | Female | Bedding | 499932 | 5891043 | 2 | 497428 | 5891027 | 5 |
| 45 | Female | Bedding | 499377 | 5891353 | 1 | 496598 | 5890265 | 5 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| 46 | Female | Bedding | 493320 | 5887460 | 5 | 492974 | 5888736 | 5 |
| 47 | Female | Bedding | 499933 | 5893769 | 5 | 502437 | 5892583 | 8 |
| 48 | Male | Bedding | 478533 | 5894501 | 1 | 477919 | 5895276 | 0 |
| 49 | Male | Bedding | 462171 | 5906830 | 3 | 461662 | 5906787 | 3 |
| 50 | Male | Bedding | 466034 | 5900038 | 2 | 468185 | 5898097 | 3 |
| 51 | Male | Bedding | 460014 | 5909562 | 2 | 458793 | 5910771 | 3 |
| 52 | Male | Bedding | 458151 | 5909140 | 6 | 455615 | 5910917 | 3 |
| 53 | Male | Bedding | 478774 | 5878766 | 2 | 480056 | 5879664 | 2 |
| 54 | Male | Bedding | 478239 | 5877750 | 5 | 477477 | 5878893 | 2 |
| 55 | Female | Bedding | 500849 | 5890088 | 2 | 500212 | 5888921 | 4 |
| 56 | Female | Bedding | 506202 | 5886697 | 3 | 507706 | 5888979 | 2 |
| 57 | Female | Carcass | 512286 | 5880660 | 4 | 511259 | 5880728 | 5 |
| 58 | Female | Bedding | 510135 | 5884632 | 2 | 511165 | 5883861 | 7 |
| 59 | Female | Bedding | 505400 | 5887659 | 3 | 503346 | 5889409 | 3 |
| 60 | Female | Bedding | 505834 | 5888031 | 6 | 507579 | 5888207 | 5 |
| 61 | Female | Foraging | 505047 | 5887670 | 6 | 504397 | 5889056 | 6 |
| 62 | Female | Bedding | 501914 | 5890890 | 5 | 501169 | 5890506 | 5 |
| 63 | Female | Bedding | 505064 | 5888377 | 3 | 504959 | 5886881 | 1 |
| 64 | Male | Bedding | 458974 | 5909446 | 2 | 458241 | 5908264 | 1 |
| 65 | Male | Bedding | 471988 | 5908390 | 2 | 474981 | 5908508 | 1 |




Appendix 2. Spectrogram produced using Audacity 2.1.3 (Audacity Team 2017). Spectrograms were evaluated along with associated audio data to determine species composition at grizzly use and random sites.

Appendix 3. ANOVA table comparing songbird species richness per planning unit for 6 different sized planning units placed across Alberta, Canada (25 $km^2$, 35 $km^2$, 100 $km^2$, 338 $km^2$, 382 $km^2$, and 1000 $km^2$).

| **Source** | **DF** | **Sum Sq** | **Mean Sq** | **F Value** | **p-Value** |
|---|---|---|---|---|---|
| Planning Unit Size | 5 | 26280 | 5256 | 58.534 | <0.001 |
| Residuals | 54826 | 4923060 | 89.8 | | |

Appendix 4. Tukey pairwise comparisons of the species richness per planning unit for 6 different sized planning units placed across Alberta, Canada where numbers indicate the area of the planning units in $km^2$ (e.g. Twenty-five – 25 $km^2$) and the planning units associated with the flagship species are listed as greater sage-grouse (35 $km^2$), grizzly bear (338 $km^2$), and woodland caribou (382 $km^2$).

| **Planning Unit Size** | **Difference** | **Lower** | **Upper** | **P Adj** | |
|---|---|---|---|---|---|
| Greater Sage-Grouse - Grizzly Bear | -2.043 | -2.703 | -1.383 | <0.001 | * |
| Hundred - Grizzly Bear | -1.173 | -1.887 | -0.459 | <0.001 | * |
| Thousand - Grizzly Bear | 0.990 | -0.282 | 2.262 | 0.229 | |
| Twenty-five - Grizzly Bear | -2.333 | -2.985 | -1.682 | <0.001 | * |
| Woodland Caribou - Grizzly Bear | 0.019 | -0.899 | 0.938 | 1.000 | |
| Hundred - Greater Sage-Grouse | 0.871 | 0.479 | 1.263 | <0.001 | * |
| Thousand - Greater Sage-Grouse | 3.033 | 1.910 | 4.156 | <0.001 | * |
| Twenty-five - Greater Sage-Grouse | -0.290 | -0.551 | -0.030 | <0.001 | * |
| Woodland Caribou - Greater Sage-Grouse | 2.063 | 1.365 | 2.761 | <0.001 | * |
| Thousand - Hundred | 2.162 | 1.007 | 3.318 | <0.001 | * |
| Twenty-five - Hundred | -1.161 | -1.537 | -0.784 | <0.001 | * |
| Woodland Caribou - Hundred | 1.192 | 0.443 | 1.941 | <0.001 | * |
| Twenty-five -Thousand | -3.323 | -4.441 | -2.205 | <0.001 | * |
| Woodland Caribou - Thousand | -0.971 | -2.262 | 0.321 | 0.266 | |
| Woodland Caribou - Twenty-five | 2.353 | 1.663 | 3.042 | <0.001 | * |

Appendix 5. ANOVA table comparing songbird species richness per planning unit for 25 $km^2$ sized planning units placed within three different flagship species ranges in Alberta, Canada (greater sage-grouse, grizzly bear, and woodland caribou).

| Source | DF | Sum Sq | Mean Sq | F Value | p-Value |
|---|---|---|---|---|---|
| Flagship Range | 2 | 39913 | 19956.5 | 245.4 | <0.001 |
| Residuals | 14161 | 1151604 | 81.3 | | |

Appendix 6. Tukey pairwise comparisons of the species richness per planning unit for 25 $km^2$ sized planning units placed in three different flagship species ranges in Alberta, Canada (greater sage-grouse, grizzly bear, and woodland caribou).

| **Flagship Range** | **Difference** | **Lower** | **Upper** | **p Adj** | |
|---|---|---|---|---|---|
| Greater Sage-Grouse - Grizzly Bear | -15.6612 | -17.7456 | -13.5768 | <0.001 | * |
| Woodland Caribou – Grizzly Bear | -2.32008 | -2.69489 | -1.94527 | <0.001 | * |
| Woodland Caribou - Greater Sage-Grouse | 13.3411 | 11.24635 | 15.43585 | <0.001 | * |

Appendix 7. PermANOVA/ Tukey comparison output for comparison of the songbird species richness per planning unit between woodland caribou core habitat and grizzly bear core habitat in Alberta, Canada.

| **Core Comparison** | **Difference** | **Lower** | **Upper** | **p Adj** | |
|---|---|---|---|---|---|
| Woodland Caribou – Grizzly Bear | -5.295 | -7.490 | -3.100 | <0.001 | * |

Appendix 8. PermANOVA / Tukey comparison of the Species at Risk songbird species richness per planning unit between woodland caribou core habitat and grizzly bear core habitat in Alberta, Canada.

| **Core Comparison** | **Difference** | **Lower** | **Upper** | **p Adj** | |
|---|---|---|---|---|---|
| Woodland Caribou – Grizzly Bear | -1.442 | -1.954 | -0.930 | <0.001 | * |

Appendix 9. PermANOVA output for comparison of the songbird species richness per planning unit between Alberta, Canada Bear Management Areas.

| Source | DF | R Sum Sq | R Mean Sq | Iter | p-Value |
|---|---|---|---|---|---|
| Bear Management Area | 6 | 1719.3 | 286.547 | 5000 | < 2.2e-16 |
| Residuals | 528 | 15383 | 29.135 | | |

Appendix 10. PermANOVA / Tukey comparison of the songbird species richness per planning unit between Alberta, Canada Bear Management Areas.

| Bear Management Area | Difference | Lower | Upper | p Adj | |
|---|---|---|---|---|---|
| Clearwater – AB North | -2.819 | -5.811 | 0.172 | 0.080 | |
| Yellowhead – AB North | -2.132 | -4.418 | 0.155 | 0.086 | |
| Grande Cache – AB North | 2.123 | 0.325 | 3.921 | <0.05 | * |
| Livingstone – AB North | 4.904 | 0.522 | 9.286 | <0.05 | * |
| Swan Hills – AB North | 1.598 | -0.867 | 4.064 | 0.469 | |
| Castle – AB North | 8.618 | 0.570 | 16.667 | 0.027 | * |
| Yellowhead - Clearwater | 0.688 | -2.810 | 4.185 | 0.997 | |
| Grande Cache - Clearwater | 4.942 | 1.743 | 8.141 | <0.001 | * |
| Livingstone - Clearwater | 7.723 | 2.604 | 12.842 | <0.001 | * |
| Swan Hills - Clearwater | 4.418 | 0.801 | 8.034 | 0.006 | * |
| Castle - Clearwater | 11.438 | 2.965 | 19.910 | <0.05 | * |
| Grande Cache - Yellowhead | 4.254 | 1.703 | 6.806 | <0.001 | * |
| Livingstone - Yellowhead | 7.036 | 2.294 | 11.777 | <0.001 | * |
| Swan Hills - Yellowhead | 3.730 | 0.671 | 6.789 | 0.006 | * |
| Castle - Yellowhead | 10.750 | 2.500 | 19.000 | <0.05 | * |
| Livingstone – Grande Cache | 2.781 | -1.745 | 7.308 | 0.536 | |
| Swan Hills – Grande Cache | -0.524 | -3.238 | 2.189 | 0.998 | |
| Castle – Grande Cache | 6.496 | -1.632 | 14.624 | 0.216 | |
| Swan Hills - Livingstone | -3.306 | -8.136 | 1.525 | 0.400 | |
| Castle - Livingstone | 3.714 | -5.343 | 12.772 | 0.889 | |
| Castle – Swan Hills | 7.020 | -1.281 | 15.321 | 0.160 | |

Appendix 11. PermANOVA output for comparison of the Species at Risk songbird species richness per planning unit between Alberta, Canada Bear Management Areas.

| Source | DF | R Sum Sq | R Mean Sq | Iter | p-Value |
|---|---|---|---|---|---|
| Bear Management Area | 6 | 190.76 | 31.794 | 5000 | < 2.2e-16 |
| Residuals | 528 | 1282.97 | 2.43 | | |

Appendix 12. PermANOVA / Tukey comparison of the Species at Risk songbird species richness per planning unit between Alberta, Canada Bear Management Areas.

| Bear Management Area | Difference | Lower | Upper | p Adj | |
|---|---|---|---|---|---|
| Clearwater – AB North | -1.361 | -2.225 | -0.497 | <0.001 | * |
| Yellowhead – AB North | -1.401 | -2.061 | -0.740 | <0.001 | * |
| Grande Cache – AB North | -0.272 | -0.791 | 0.248 | 0.715 | |
| Livingstone – AB North | 0.947 | -0.318 | 2.213 | 0.289 | |
| Swan Hills – AB North | -0.227 | -0.939 | 0.485 | 0.965 | |
| Castle – AB North | 2.733 | 0.408 | 5.057 | <0.05 | * |
| Yellowhead - Clearwater | -0.040 | -1.050 | 0.970 | 1.000 | |
| Grande Cache - Clearwater | 1.089 | 0.165 | 2.013 | 0.009 | * |
| Livingstone - Clearwater | 2.308 | 0.830 | 3.786 | <0.001 | * |
| Swan Hills - Clearwater | 1.134 | 0.089 | 2.178 | <0.05 | * |
| Castle - Clearwater | 4.094 | 1.647 | 6.541 | <0.001 | * |
| Grande Cache – Yellowhead | 1.129 | 0.392 | 1.866 | <0.001 | * |
| Livingstone - Yellowhead | 2.348 | 0.978 | 3.717 | <0.001 | * |
| Swan Hills - Yellowhead | 1.173 | 0.290 | 2.057 | <0.05 | * |
| Castle - Yellowhead | 4.133 | 1.751 | 6.516 | <0.001 | * |
| Livingstone – Grande Cache | 1.219 | -0.089 | 2.526 | 0.086 | |
| Swan Hills – Grande Cache | 0.044 | -0.739 | 0.828 | 1.000 | |
| Castle – Grande Cache | 3.004 | 0.657 | 5.352 | <0.05 | * |
| Swan Hills - Livingstone | -1.174 | -2.569 | 0.221 | 0.164 | |
| Castle - Livingstone | 1.786 | -0.830 | 4.401 | 0.403 | |
| Castle – Swan Hills | 2.960 | 0.563 | 5.357 | <0.05 | * |

Appendix 13. Pearson correlation matrix for continuous environmental variables representing amount of vegetation cover types within a 150 m radius from study sites.

| | Upland Treed | Upland Herb | Shrub | Barren | Anthro | Wetland Herb | Wetland Trees |
|---|---|---|---|---|---|---|---|
| **Upland Treed** | 1 | -0.01 | -0.09 | -0.1 | -0.91* | -0.07 | 0.02 |
| **Upland Herb** | -0.01 | 1 | -0.05 | -0.1 | -0.18 | -0.04 | -0.04 |
| **Shrub** | -0.09 | -0.05 | 1 | 0.07 | -0.19 | 0 | -0.03 |
| **Barren** | -0.10 | -0.1 | 0.07 | 1 | -0.02 | -0.05 | -0.07 |
| **Anthro** | -0.91* | -0.18 | -0.19 | -0.02 | 1 | -0.01 | -0.12 |
| **Wetland Herb** | -0.07 | -0.04 | 0 | -0.05 | -0.01 | 1 | 0.36 |
| **Wetland Trees** | 0.02 | -0.04 | -0.03 | -0.07 | -0.12 | 0.36 | 1 |