Cougar habitat and prey selection on reclaimed coal mines in west-central Alberta

by

Meghan Beale

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ABSTRACT

Two reclaimed coal mines in west-central Alberta host a complex assemblage of large-bodied predator and prey populations, including cougars (*Puma concolor*), bighorn sheep (*Ovis canadensis*), elk (*Cervus elaphus*), and mule deer (*Odocoileus hemionus*). The presence of ungulate and predator populations are linked to landscape features that arose through mine reclamation. Although reclamation has been successful at attracting a diverse set of large mammals, reclamation also might be facilitating increased predation on bighorn sheep. Thus, our main objectives were to model habitat selection for bighorn sheep, elk, mule deer, and cougars on reclaimed mines to determine how ungulates responded to reclamation features, and to determine how cougars exploited landscape configuration while hunting. To evaluate ungulate habitat selection, we used direct ground counts on a fixed survey route between 2004-2017. We created a grid of 200 x 200-m non-overlapping sampling units for our study area and assigned each ungulate group to a sampling unit. We also assigned landscape features to each sampling unit to represent changes due to mining and reclamation. We modelled habitat selection pooled over four seasons by fitting exponential resource selection functions (RSFs) in a used vs. available design. Our results demonstrate that bighorn sheep, elk, and mule deer selected landscape features to increase access to quality forage and decrease predation risk. Bighorn sheep strongly selected high walls while elk and mule deer selected reclaimed grasslands. To model cougar habitat selection, we outfitted seven cougars with GPS collars between March 2017 – January 2018 and collected a GPS location every 1.5 hours. We visited clusters of GPS points to determine successful cougar predation events,

and collected species, age, and sex of prey. We estimated RSFs and step selection functions, using landscape features as covariates. At a fine scale, cougars selected rocky outcrops, forests, forest edges, and high relative availability of bighorn sheep when on the reclaimed mines. Further, cougar predation events on bighorn sheep were closer to forest edges than randomly expected, which supported that cougars exploited landscape configuration when hunting. Cougars specialized on bighorn sheep when on reclaimed mines. Findings from our study become increasingly relevant as government approve end land-use strategies for reclaimed mines in our study area. We recommend that ecologists consider wildlife to be a target for evaluating the success of ecological reclamation. Also, we suggest that managers consider configuring landscapes to reduce predation on bighorn sheep.

PREFACE

This thesis is original work by Meghan M. Beale. Field data for ungulate studies (Ch 2) were collected by Beth MacCallum and employees of Bighorn Wildlife Technologies Ltd. between January 2004 and December 2017, in accordance with annual wildlife surveys conducted for Teck Coal Limited, Cardinal Operations, and Westmoreland Coal Company. Field data for cougar studies (Ch 3) were collected by Meghan M. Beale, Samantha L. Widmeyer, and Mark S. Boyce between March 2017 and October 2018. These cougar data were collected in accordance with the Canadian Council on Animal Care (CCAC) guidelines and approved by the University of Alberta Animal Care and Use Committee (AUP00002113), and in accordance with Alberta Environment and Parks Research and Collection Permit (2017: #17-264; 2018: #18-011).

To date, no manuscripts have been submitted for publication. A version of Chapter 2 will be submitted to Restoration Ecology and includes M. Boyce as co-author. For this manuscript, B. MacCallum collected the data, M. Beale analyzed the data, and M. Beale wrote the manuscript. M. Boyce provided valuable feedback during analysis and writing. B. MacCallum provided valuable feedback during creation of GIS data and writing. A version of Chapter 3 will be submitted to the Journal of Wildlife Management, and includes S. Widmeyer and M. Boyce as co-authors. For this manuscript, M. Beale and S. Widmeyer collected the data, M. Beale analyzed the data, and M. Beale wrote the manuscript. M. Boyce provided valuable feedback during design, analysis, and writing.

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CHAPTER 1 – INTRODUCTION: ARE RECLAIMED MINES AN ECOLOGICAL TRAP OR

SANCTUARY?

Coal mine reclamation near Cadomin, Alberta has provided a mosaic of habitat features

for ungulates and their predators. In particular, the Gregg River and Luscar mines, located

approximately 6 km west of Cadomin, Alberta, were once part of the metallurgical coal

extraction industry in western Canada. Gregg River and Luscar opened in 1982 and 1969,

respectively, and reclamation began in 1982 and 1971, respectively. As of 2019, Gregg

River is almost entirely reclaimed and Luscar is partially reclaimed. The ecological

objective of reclamation is to mitigate negative effects of mining on the environment and

wildlife by revitalizing ecosystem functioning and returning the landscape to useable

habitats (Jones and Davidson 2016). Traditionally, the success of ecological reclamation is

determined by assessing vegetation communities, and in some cases, assessing

recolonization of wildlife (Jones and Davidson 2016). Reclamation at Gregg River and

Luscar (henceforth called 'the reclaimed mines') employed a combination of approaches,

including restoring pre-mine habitats, replacing lost habitats, and creating new habitats

to attract a variety of wildlife species (MacCallum and Geist 1992). Furthermore, the

reclaimed mines are located in the subalpine region of the Nikannassin Mountain Range.

When mining for coal in mountainous regions, extraction is constrained by locations of

coal seams. Consequently, mining at Gregg River and Luscar resulted in a mosaic of

undisturbed forest patches, interspersed with reclaimed and active areas, and this

landscape mosaic provided habitats for bighorn sheep (*Ovis canadensis*), elk (*Cervus*

elaphus), deer (*Odocoileus* spp.), grizzly bears (*Ursus arctos*), cougars (*Puma concolor*), and wolves (*Canis lupus*), despite continuing mining activities.

The bighorn sheep population that inhabits the reclaimed mines and surrounding areas (coined 'the Cadomin bighorn population') have been deemed an important bighorn sheep population by hunting organizations in North America. The population has been monitored by Beth MacCallum of Bighorn Wildlife Technologies Ltd. (BWT) since 1985, and has also produced multiple trophy rams (BWT, pers. comm.). The reclaimed mines and surrounding areas are located in a Chinook zone (Natural Regions Committee 2006) that brings warm westerly winds, which ameliorate harsh winter conditions and may allow rams to thrive. This milder microclimate may have allowed the Cadomin rams to grow larger than in other regions of Alberta. Additionally, the Cadomin bighorn population provided over 400 individuals for various translocations projects across North America, between 1989 and 2015 (Beth MacCallum, BWT, unpublished data). Albertan sheep hunters wish to ensure that trophy rams continue to be produced by the Cadomin bighorn population because trophy hunting bighorn sheep is legal on wildlife management units adjacent to the reclaimed mines. Ultimately, managers wish to conserve habitat, maintain biodiversity, and sustain viable bighorn populations on the reclaimed mines (Government of Alberta 2013).

Anecdotal evidence suggests that cougars might be targeting bighorn rams on the reclaimed mines. In the last decade, over thirty bighorn sheep have been killed by cougars on the reclaimed mines, which some Albertan hunters believe may be problematic. Cougars select vulnerable targets, such as old (Husseman et al. 2003), shy

(Reale 2003), and young (Knopff et al. 2010a, Smith et al. 2014) individuals. Thus, cougars might select rams in late winter, when rams are in their poorest body condition. Cougar predation on bighorn sheep is a complex and long-standing phenomenon in ecology (Rominger 2018). For example, multiple studies across North America (Alberta, Canada ((Ross et al. 1997, Festa-Bianchet et al. 2006); British Columbia, Canada (Harrison and Hebert 1988); California, USA (Wehausen 1996, Hayes et al. 2000, Johnson et al. 2013); New Mexico, USA (Rominger et al. 2004); and South Dakota, USA (Blake and Gese 2016b)) have addressed the predator-prey relationship between cougars and bighorn sheep. Although cougars are generalists (Elbroch and Wittmer 2013), individuals can specialize (Cooley et al. 2008, Knopff et al. 2010a, Elbroch and Wittmer 2013, Lowrey et al. 2016), and many cases of cougar specialization on bighorn sheep have been documented (Harrison and Hebert 1988, Ross et al. 1997, Blake and Gese 2016b). Further, multiple studies have found that cougar specialization on bighorn sheep can result in population-level effects for sheep (Harrison and Hebert 1988, Wehausen 1996, Hayes et al. 2000, Rominger et al. 2004, Festa-Bianchet et al. 2006), which supports that cougar predation might reduce bighorn sheep populations and reduce the availability of rams for hunters.

In a recent review, Rominger (2018) identified multiple ecological patterns that exacerbate cougar predation on bighorn sheep, including apparent competition (Johnson et al. 2013), specialist predators (Cooley et al. 2008, Knopff et al. 2010a, Elbroch and Wittmer 2013, Lowrey et al. 2016), vulnerable prey populations (i.e. small (Bourbeau-Lemieux et al. 2011) or translocated (Rominger et al. 2004)), and subsidized predator

diets (Rominger et al. 2004). However, Rominger (2018) failed to include how landscape configuration might facilitate cougar predation on bighorn sheep. Cougars are stalk and ambush predators (Beier et al. 1995) that use forest edges (Beier et al. 1995, Laundre and Hernandez 2003, Laundre and Loxterman 2006, Holmes and Laundre 2006) to approach prey while remaining undetected. Therefore, if landscapes are configured in such a way that cougars can gain increased access to sheep, it is possible that spatial configuration of landscape also might exacerbate cougar predation on bighorn sheep.

We evaluated whether landscape configuration on the reclaimed mines exacerbates cougar predation on bighorn sheep. Local biologists debate whether the reclaimed mines are an ecological trap or sanctuary for bighorn sheep (Kneteman 2016). Some believe that the numbers of sheep observed on the reclaimed mines represent population recruitment due to nutritious forage and protection from hunters (MacCallum and Geist 1992). In contrast, others believe that these numbers represent shifts in ranges by existing bighorn sheep populations (Kneteman 2016). Some believe that bighorn sheep are enticed to the reclaimed mines by non-native forage, engineered escape terrain, and protection from hunting, but experience higher rates of predation due to residual landscape configuration (Kneteman 2016). Further, some believe that high walls retained from mining do not provide adequate escape terrain for bighorn sheep and may, instead, provide a predator pantry for cougars (Chris Watson, Hinton Conservation Officer, pers. comm.). Despite conflicting hypotheses regarding the ecological significance of the reclaimed mines, no research has evaluated whether the reclaimed mines are an ecological trap or sanctuary. We evaluated one aspect of the ecological trap hypothesis.

In particular, we evaluated how multiple prey populations integrated mining and reclamation features into their habitats. We also evaluated whether landscape configuration on the reclaimed mines facilitated cougar predation on bighorn sheep. Ultimately, the reclaimed mines will soon return to Crown land and the Province of Alberta will be responsible for management of land. Thus, understanding how spatial landscape patterns influence cougar predation on bighorn sheep is integral to ensure the persistence of thriving bighorn populations.

CHAPTER 2 – UNGULATE HABITAT USE IN RESPONSE TO COAL MINE RECLAMATION

ABSTRACT

Extracting coal through mining alters natural habitats and can result in substantial changes to vegetative and topographic structure. Reclamation aims to mitigate the negative effects of mining on the environment and wildlife by revitalizing ecosystem functioning and returning the landscape to useable habitats. We evaluated interspecific differences in response to mining and reclamation for sympatric bighorn sheep (*Ovis canadanesis*), elk (*Cervus elaphus*), and mule deer (*Odocoileus hemionus*) populations, on reclaimed mines near Cadomin, Alberta. We used direct ground counts on a fixed survey route to obtain count data for bighorn sheep, elk, and mule deer between 2004-2017. We created a grid of 200 x 200-m non-overlapping sampling units for our study area and assigned each ungulate group to a sampling unit. We also assigned landscape features to each sampling unit to represent changes due to mining and reclamation. Landscape features included high walls, which are steep walls created by strip mining that were retained to function as escape terrain. We modelled annual habitat selection by fitting exponential resource selection functions in a used vs. available design. Bighorn sheep, elk, and mule deer selected landscape features to increase access to quality forage and decrease predation risk with bighorn sheep strongly selecting high walls while elk and mule deer selected reclaimed grasslands. Ungulates also responded to mining and reclamation features in ways that we did not anticipate. Bighorn sheep and elk selected for haul roads, which we attributed to seeking human refuge from predators. Bighorn sheep also selected main roads, which we hypothesized was due to mineral salts

available on roadways. Further, mule deer avoided haul roads and selected disturbed areas, which was opposite to elk. Understanding interspecific differences in response to external environment is crucial for industrially and anthropogenically modified landscapes, where humans determine landscape design. If creating habitats for wildlife is the primary goal of reclamation, then future landscape design should focus on developing high walls and flat ridges seeded with forbs and grasses. Findings from our study become increasingly relevant as government approves management strategies for reclaimed mines in our west-central Alberta study area.

INTRODUCTION

Industrial expansion is increasing rapidly on a global scale and includes the development of extraction industries such as oil, natural gas, coal, and forestry. Although Alberta, Canada is known for its expansive oil and gas industry, mining for metallurgical coal is also a major industry in the Province's Rocky Mountain foothills. Surface and open pit mining drastically alter natural habitats because they remove the top layers of soil to extract the underlying coal seams, stripping the land of its natural topography (Wickham et al. 2013), introducing soil compaction (Larkin et al. 2008), and altering vegetation communities (MacCallum and Geist 1992, Holl and Cairns 1994, Wickham et al. 2006, 2013). During this process, forests are logged, and topsoil is salvaged and stockpiled. Blasting and mechanized shoveling are used to remove overburden and expose coal seams for extraction. Haul roads connect areas of active blasting with processing facilities to allow the transport of coal using heavy equipment. These modifications of land can result in changes to species distribution (Weir et al. 2009) and decreases in species diversity

(Larkin et al. 2008, Wickham et al. 2013, Ardente et al. 2016). Surface mining, in particular, is associated with substantial changes to vegetative structure (Holl and Cairns 1994), a loss of forested habitats (Schueler et al. 2011, Javed and Khan 2012, Wickham et al. 2013), and the conversion of interior forest to edge habitats (Wickham et al. 2006), which may not provide the same services for wildlife (Harper et al. 2005).

Reclamation aims to mitigate effects of mining on the environment and wildlife over the long-term by revitalizing ecosystem functioning and returning the landscape to useable habitat (Jones and Davidson 2016). During reclamation, topographic manipulation reintroduces slopes of various aspects, then topsoil is replaced (Hingtgen and Clark 1984, MacCallum and Geist 1992). The land is seeded with a grass-legume-forb mixture to facilitate efficient succession of grasslands while promoting soil development and reducing soil erosion (MacCallum and Geist 1992, Swab et al. 2017). Native tree and shrub species are planted, and undisturbed forest patches are retained (MacCallum and Geist 1992). Throughout the process of reclamation, specific features may be preserved or rebuilt for wildlife habitat, including talus slopes, rock piles, stream beds, logging debris, tree poles, bodies of water, and high walls (MacCallum and Geist 1992).

Reclaimed landscapes are a mosaic of features that serve as valuable habitat for wildlife (MacCallum and Geist 1992). Abandoned structures can provide hibernacula (Whitaker and Rissler 1992, Sherwin et al. 2000) and breeding sites (Johnson et al. 1978, Heath et al. 1986) for bats and birds. Grasslands reclaimed from surface mines can provide grazing opportunities for Dall's sheep (*Ovis dalli)* (Elliott and McKendrick 1984) and bighorn sheep (*O. canadensis*) (MacCallum and Geist 1992, Poole et al. 2016).

Further, reclaimed sites often have increased landscape heterogeneity, which provides a complex habitat structure that can support a greater variety of wildlife than the original habitat (Müller et al. 2017). Specifically, in Denmark, red deer (*Cervus elaphus*) reintegrated reclaimed mines into their ranges and Müller et al. (2017) attributed this reintegration to the patchy and heterogeneous landscape newly available, post-reclamation. Moreover, smaller mammals, such as gray fox (*Urocyon cinereogenteus*) and red foxes (*Vulpes fulva*), are also attracted to the increased landscape heterogeneity available on reclaimed landscapes (Yearsley and Samuel 1980).

Although reclamation can restore productivity to industrially modified ecosystems, we still must be concerned with understanding how ungulates respond to mining and reclamation. Ungulates are susceptible to large-scale changes caused by mining because they depend on vegetation for forage (Merkle et al. 2016), security cover (Sheehy and Vavra 1996, Anderson and Long 2012, Webb et al. 2013), and thermal cover (Sawyer et al. 2007, Anderson and Long 2012, Webb et al. 2013). Ungulates are integral to food web functioning, as they serve as prey for large-bodied predators, including grizzly bears (*Ursus arctos*), wolves (*Canis lupus*), and cougars (*Puma concolor*). Further, ungulates are economically important to humans as ungulates provide a source of food, trophy hunting opportunities, and aesthetic importance for wildlife enthusiasts. Thus, we focus our research on three ungulate species found in our Alberta, Canada study area: bighorn sheep, elk, and mule deer.

Mines near Cadomin, Alberta, our study area, provide a landscape for investigating ungulate response to coal mining and reclamation. Our study area includes

three mines: two reclaimed mines that were once operational open pit mines for metallurgical coal and one active coal mine that is currently operating. Bighorn sheep, elk, and mule deer occur sympatrically on all three mines and all species' populations appear to be thriving since systematic monitoring began in 1985. The pre-identified end land use of the mines is wildlife habitat and watersheds. Specifically, Alberta's provincial Environmental Protection and Enhancement Act (EPEA) (Province of Alberta 2000) establishes that mining corporations must include reclamation as an integral part of the entire coal extraction process. Mining corporations must identify an end land use before the Province issues a permit to mine the land. This approach ensures that mined landscapes are returned to an end land use that encompasses the environmental, biological, and social needs of the particular region.

We anticipate that target ungulate species will respond differently to disturbed, attractive, and risky landscape features. Disturbed landscape features include active and inactive mining pits, waste rock dumps, stored equipment, buildings, and parking lots (these areas henceforth called 'disturbed areas'). At fine scales, bighorn sheep avoid disturbed areas and select undisturbed habitat patches (Jansen et al. 2006) but at coarser scales, use active mines more than inactive mines (Bleich et al. 2009, Jansen et al. 2009). Bighorn sheep select similar habitat features on mines as they would off mines (Oehler et al. 2005, Jansen et al. 2006) and the reclaimed areas of active mines can provide sheep with rugged topography as escape terrain and seeded grasslands for forage (MacCallum and Geist 1992, Bleich et al. 2009, Poole et al. 2016). Bighorn sheep inhabit mines in our study area, and we expect them to exhibit fine-spatial-scale avoidance of disturbed areas

and haul roads. Moreover, elk avoid roads (Johnson et al. 2000, Rowland et al. 2000, Stewart et al. 2002, Ager et al. 2003, Stewart et al. 2010), resource extraction developments (Buchanan et al. 2014), and noises from mining disturbances (Kuck et al. 1985). Mule deer also avoid roads (Stewart et al. 2002), resource extraction developments (Sawyer et al. 2006, 2009), and highly disturbed patches when migrating through active surface mines (Blum et al. 2015). However, mule deer use roads to spatially separate and avoid exploitative competition with elk (Johnson et al. 2000, Ager et al. 2003, Stewart et al. 2010, Lendrum et al. 2012), so we expect mule deer to be more likely than elk to select haul roads and disturbed areas in our study area.

High walls and seeded grasslands are likely to attract ungulates in our study area. In our study area, high walls function as escape terrain, which is the single most critical habitat component for bighorn sheep (Smith et al. 1991, Andrew et al. 1999, Singer et al. 2000, Bleich et al. 2009). High walls are comprised of three different types of escape terrain: bench walls, foot walls, and free-dumped talus. In general, these walls are steep rocky walls that were left-over from strip mining for coal seams below the ground. Bench walls are multi-level walls that were engineered to have benches of seeded forage layered in with vertical rocky walls (BWT, pers. comm.). Foot walls are smooth rocky faces, directly left-over from coal extraction and have not been engineered (BWT, pers. comm.). Free-dumped talus is free-dumped rocks and boulders (BWT, pers. comm.). As such, we henceforth refer to bench walls, foot walls, and free-dumped talus collectively as 'high walls'. High walls are especially important for pregnant and post-parturient sheep, as they depend heavily on escape terrain to avoid predators (Berger 1991, Bleich et al.

1997). Therefore, we expect that bighorn sheep will strongly select high walls in our study area. However, elk and mule deer do not escape to high walls as their primary anti-predator strategy. Rather, at fine temporal scales (i.e. on the order of minutes and hours), elk forage in productive grasslands (Collins and Urness 1983) but move into forests for security cover when threatened (Unsworth et al. 1998, Fortin et al. 2005, Creel et al. 2005, Buchanan et al. 2014). Mule deer select less productive grasslands than elk (Collins and Urness 1983), but selection for sub-optimal forage can be due to elk competitively excluding mule deer from productive areas (Collins and Urness 1983, Lendrum et al. 2012). Similar to elk, mule deer retreat to forests for security cover when threatened (Collins and Urness 1983, Lendrum et al. 2012) but mule deer also select forests (Carson and Peek 1987, Ager et al. 2003, Serrouya and D'Eon 2008) to feed on browse. Given the coarse temporal scale of our study (i.e. data were collected months apart), we expect mule deer will select forests, but do not expect that elk will select forests, which only occurs at fine temporal scales (Creel et al. 2005). Bighorn sheep, elk, mule deer select nutritious forbs, such as alfalfa (*Medicago sativa*), if available (MacCallum and Geist 1992, Torstenson et al. 2006, Webb et al. 2013, DeVore et al. 2016). Alfalfa is a staple forb in the agronomic mixture seeded onto reclaimed grasslands in our study area, which we predict will drive all study species to select reclaimed grasslands. Because there is such a vast extent of reclaimed grasslands available in our study area, we expect that we will not be able to detect the fine spatial avoidance between elk and mule deer that Stewart (2002) did.

Certain habitats, such as those proximate to forest edges, near riparian areas, and at low relative topographic positions render ungulates more vulnerable to predation by cougars or wolves. Cougars are effective predators on bighorn sheep (Wehausen 1996, Ross et al. 1997, Festa-Bianchet et al. 2006, Knopff et al. 2010a, Rominger 2018), feed extensively on deer (Knopff et al. 2009, 2010a, Bacon et al. 2011), and occasionally take elk (Knopff et al. 2010a, Bacon et al. 2011) . Cougars are stalk and ambush predators (Beier et al. 1995, Pelletier et al. 2006) and use forest edges (Beier et al. 1995, Laundre and Hernandez 2003, Laundre and Loxterman 2006, Holmes and Laundre 2006) to approach prey while remaining undetected. Wolves also select edges while hunting (McPhee et al. 2012) to target elk where they are vulnerable (Bergman et al. 2006). Telemetry data show that cougars inhabit mines in our study area (Ch 3), while surveys from Bighorn Wildlife Technologies Ltd. (BWT) suggest that wolf presence in our study area is high (BWT, unpublished data). Therefore, we expect that all study species will avoid forest edges when selecting habitat to reduce risk of predation by cougars and wolves. In addition to forest edges, riparian areas including ravines and streams often act as travel corridors for wolves (Kunkel and Pletscher 2000, 2001, Bergman et al. 2006, Kauffman et al. 2007) and cougars (Dickson and Beier 2002), and create physical obstacles that may make it difficult for elk to escape attack (Kauffman et al. 2007). Mule deer may also avoid riparian areas due to increased wolf attacks (Kunkel and Pletscher 2001), but may also select for riparian habitat to browse (Carson and Peek 1987) and use rugged terrain in riparian corridors to escape predation (Lingle 2002, Lingle et al. 2008). Thus, we expect that elk will avoid riparian habitat and also reason that mule deer may

either select or avoid riparian habitat, depending on the trade-off between predation risk and accessing forage. Riparian areas, valleys, and ravines exist at low relative topographic positions (RTPs), whereas high relative topographic positions are the tops of hills, mountains, and cliffs. High RTPs should provide advantageous locations for ungulates to forage while remaining vigilant and so we expect all study species will similarly select for high RTPs (Cassirer et al. 1992, Kunkel and Pletscher 2001).

Bighorn sheep, elk, and mule deer select and avoid landscape features with varying strengths, which we predict will influence their strength of response to mining and reclamation. For instance, bighorn sheep are particular in their response to external environment (Hudson et al. 1976), such as their high fidelity to suitable escape terrain (Smith et al. 1991, Andrew et al. 1999, Singer et al. 2000, Bleich et al. 2009) and selection for high quality forbs (Hobbs et al. 1983). In contrast, elk have wider niche breadths (Hudson et al. 1976, Hobbs et al. 1983, Collins and Urness 1983) and have weaker responses to environmental parameters than both bighorn sheep and mule deer (Hudson et al. 1976). Hence, we expect that bighorn sheep will respond most strongly to mining and reclamation through strong selection of high walls. Elk should have weaker responses than bighorn sheep to mining and reclamation, as their life histories are not as highly dependent on any particular feature. We expect that mule deer will respond to mining and reclamation with similar strength as elk.

Here we evaluate interspecific differences in response to mining and reclamation by evaluating habitat use and strength of response to mining and reclamation among bighorn sheep, elk, and mule deer. Firstly, we modelled habitat selection and intensity of

habitat use on the reclaimed mines to understand habitat requirements for each species.

For the ease of describing our methods, we refer to 'habitat use' as a collective term to

encompass both 'habitat selection' and 'intensity of habitat use'. Secondly, we compared

beta coefficients for landscape features appearing in top habitat use models to quantify

strength of each species' response to mining and reclamation. Ultimately, understanding

habitat use and responses to mining and reclamation among the ungulate guild on the

reclaimed mines is integral to ensuring sustainable ecological management of

economically important ungulates in our study area.

METHODS

Study area

The study area, totaling 483 km^2, is located in west-central Alberta, Canada

(approximately 53°04'N 117°26'W) and includes three coal mineral surface leases (MSLs)

(Figure 2.1). MSLs describe the boundaries in which mining companies can legally extract

minerals. The MSLs are situated in the Rocky Mountain Foothills, in the subalpine region

east of the Nikanassin Mountain Range. Two of the MSLs include Luscar and Gregg River,

two neighbouring reclaimed mines. The reclaimed mines are located approximately 50

km south of Hinton, Alberta and 6 km west of Cadomin, Alberta. Mining began at Luscar

and Gregg River in 1969 and 1982, respectively, and reclamation began in 1971 and 1982,

respectively. Luscar, owned by Teck Resources Ltd. (Teck; Vancouver, Canada), totals 53

km^2 and is partially reclaimed (60%), as of 2017. Gregg River, owned by Westmoreland

Coal Company (Westmoreland; Colorado Springs, USA), totals 37 km^2 and is almost

entirely reclaimed (99%), as of 2017. No active mining remains on Gregg River, but some

areas of Luscar are active, including a haul road, which connects to a third MSL. The third

MSL includes Cheviot, an active coal mine located approximately 8 km south of Luscar

and Gregg River that is owned by Teck. Coal extracted from Cheviot is transported via the

haul road to a processing facility on the Luscar mine site. The Cheviot MSL is 72 km^2 in

total but the eastern half of the lease is undisturbed and has yet to be mined, as of 2017.

Thus, we included only the western half of the Cheviot MSL (totaling 44 km^2) in our study

area. MSLs include undisturbed, actively mined, and reclaimed areas.

The study area is characterized by a sub-alpine Cordilleran climate (cold winters

and cool summers) with a short growing season (Strong and Leggat 1992) and a Chinook

that descends onto the MSLs and brings warmer temperatures (Natural Regions

Committee 2006). Meteorological data collected between 2007 and 2016 indicates that

the study area has an average daily minimum temperature of -4°C, average daily

maximum temperature of 8°C, average daily precipitation of 1.7 mm, and average daily

accumulated precipitation of 2870 mm (Alberta Agriculture and Forestry 2017). High

winds above 80 kph are common (Alberta Agriculture and Forestry 2017).

The current land cover in the study area is approximately 50% forest, 22% barren,

14% shrub, 13% grassland, and 1% water, although land cover is ever-evolving as active

mining and subsequent reclamation continue. Grasslands on the reclaimed mines are

dominated by non-native forage species including Kentucky bluegrass (*Poa pratensis*),

Canadian bluegrass (*Poa compressa*), red fescue (*Festuca rubra*), orchard grass (*Dactylis

glomerata*), alfalfa, cicer milkvetch (*Astragalus cicer*), and clover (*Trifolium hybridum*)

(Strong 2002). Woody species such as Engelmann spruce (*Picea engelmannii*), lodgepole pine (*Pinus contorta*), balsam poplar (*Populus balsamifera*), willow (*Salix* spp.), and alder (*Alnus* spp.) are planted during reclamation to create shrubland and, eventually, mature forests (MacCallum and Geist 1992). Natural forests are conifer at higher elevations and are mainly lodgepole pine, white spruce (*Picea glauca*), and Engelmann spruce (Strong 2002). At lower elevations, balsam poplar and trembling aspen (*Populus tremuloides*) are present in mixed conifer-deciduous forests.

Habitats between Luscar and Gregg River are contiguous; that is, there are no man-made boundaries between the two mines that limit animal movement. To the east of the MSLs, human activities include motorized and non-motorized recreational activity, forest harvesting, and drilling by oil and gas industries. Highway 40 receives heavy vehicular traffic and is the main road that services the mines, the town of Cadomin, and oil and gas roads. Protected land (Whitehorse Wildland Provincial Park and Jasper National Park) dominates the west side of the MSLs. Non-motorized public trails exist on the reclaimed mines and allow passage to the wildlife management units to the west, mainly for hunting purposes.

Ungulate surveys

We conducted direct ground counts of ungulates between 2004 and 2017, using a fixed survey route, a minimum of four times per annum ($\bar{x}$ = 4.1 surveys per year). We completed the same survey route, in its entirety, during each survey. Due to the length of the survey route (159 km) and available manpower, the entire survey route could not be

completed in one day. Thus, groups of 2-3 trained observers completed the entire survey route within 2-5 days. Observations were made by vehicle or by foot using a spotting scope or binoculars to record the total number and composition of individuals in each species group (MacCallum 1989). Ungulate groups were defined based on spatial aggregation, as well as life history and behavioural traits of each species (e.g., sheep spatially aggregate into ram and nursery groups (Geist and Petocz 1977)). The centroid of each ungulate group was estimated visually and plotted by hand on a large-scale paper map. If ungulate groups moved during observation, the initial ungulate group centroid (henceforth called ungulate centroid) was recorded. Trained observers communicated with handheld radios regularly during each field day to confirm that groups of ungulates were not counted twice. We delineated spatial 'blocks' within the study area based on natural boundaries that limit animal movement, such as rivers and mountain ranges. Each block was completed at the end of each survey day to reduce the likelihood of counting a group of ungulates on a subsequent day. Ungulate centroids were transferred visually to spatial software and a Universal Transverse Mercator (UTM) northing and easting was calculated for each ungulate centroid. If the entire survey route could not be completed (i.e. due to poor visibility or weather conditions), results from the uncompleted survey were not included in our analyses. We defined the extent of our study area by measuring the perpendicular distance from each ungulate centroid to the nearest point on the survey route, then applying a buffer twice the distance of our furthest ungulate centroid to the survey route (Buckland et al. 2001). We determined that ungulate centroids were

located between 0 – 2,269 m from the survey route and, therefore, we applied a 4,538-m buffer to the survey route to determine study area extent.

We created a grid of 12,373 non-overlapping 200 x 200-m sampling units covering the entirety of our study area using a random point as the starting point. We selected this sampling unit size in an effort to encompass any positional sampling error in ungulate centroids. We estimated positional accuracy using the Viewshed tool in ArcGIS Spatial Analyst (Environmental Systems Research Institute 2017). The Viewshed tool uses a digital elevation model (DEM) and an observer survey route as input to determine areas that are visible and non-visible from the survey route, based on topography. We determined ungulate centroids that were located in non-visible areas (n = 22) and calculated the Euclidean distance to nearest visible area. The maximum distance was 50 m, which we assumed was a conservative estimate of positional accuracy. Further, we chose to use sampling units to account for the subjective nature of a human observer defining an ungulate group. We then assigned each ungulate centroid a sampling unit based on geographical location. We recorded species and number of individuals for each ungulate centroid, as well as the date of each observation. We defined four seasons based on timing of birth and rut for each species (Table 2.1) and assigned each observation one of four seasons, when we modeled intensity of habitat use.

Correcting for detectability

We examined ungulate detectability in two habitat types using program DISTANCE version 7.1, Release 1 (Thomas et al. 2010). First, we stratified our study area into two

habitat types due to presumed detection differences in open vs. forest habitats. We pooled annual data between 2004-2017 for each of the three target species, within each of the two habitat types, creating a total of six detectability analyses. We assigned either 'open' or 'forest' habitat type to each ungulate centroid in ArcGIS using land cover classification data (see below for the development of these land cover classification data). We considered grasslands, shrubs, non-vegetated (i.e. barren/rocky), and water to be open habitat, as these habitats do not have canopy cover. We assumed all open areas had similar ungulate detectability for a trained human observer. Using ArcGIS, we split the 159-km survey route into 89 segments, or transects, and calculated the lengths of each transect. For each habitat type and species, we calculated the perpendicular distance from each ungulate centroid to the nearest point on the survey route and identified to which transect the centroid belonged.

For each habitat type and species, we fit a preliminary model, a half-normal key function with a cosine series expansion, to non-truncated data, as recommended by Buckland et al. (2001). Then, we examined histograms to determine the right-truncation distance (henceforth called truncation distance) w where $g(w)$ = 0.15 (Table 2.2) for each habitat type and species. Here, the truncation distance represents the distance at which detection drop and observations may be unreliable (Buckland et al. 2001). Truncation distances may differ slightly between open vs. forest habitats, and among ungulate species (Table 2.2). As a solution for accounting for detectability issues, we opted to rely on the accuracy of our original group counts by truncating our modelling dataset to remove observations further than the minimum truncation distance for each species (i.e.

300 m for bighorn sheep; Table 2.2). We are confident that our defined truncation

distances are conservative enough that observers were able to count individuals

consistently and reliably within these distances. Although we explored the concept of

inflating the number of individuals per group based on detectability (Appendix 2.1), we

opted not to inflate counts before habitat modelling as this method was biologically

unreasonable and depended heavily on weakly supported assumptions.

To remove observations beyond truncation distances, we first assigned a distance

from each sampling unit to the survey route. To do this, we subdivided each 200 x 200-m

sampling unit into 100 20 x 20-m pixels. We calculated the perpendicular distance from

each pixel centroid to the nearest point on the survey route. Within each 200 x 200-m

sampling unit, we calculated the mean distance to survey route for n = 100 pixels and

assigned the mean distance to each sampling unit. We chose this method to ensure that

the tortuosity and configuration of the survey route was adequately accounted for when

calculating mean distances to assign per sampling unit (Appendix 2.2). Thus, we removed

sampling units with mean distances > w, based on w for each species, from further

habitat modelling.

Landscape features

We created a database of biologically relevant landscape features (Table 2.3) to be used

as covariates in habitat use models. We obtained spatial data from numerous sources

including Foothills Research Institute (FRI; Hinton, Alberta), Teck, Bighorn Wildlife

Technologies Ltd. (BWT; Hinton, Alberta), and GeoBase Series (GeoBase; Natural

Resources Canada). We obtained spatial data for years 2004-2010 from the Boyce Lab at the University of Alberta as these data were created for previous research in the study area (Cristescu et al. 2016). We updated all spatial data between 2011-2017 to reflect annual landscape changes due to mining by interpreting annual orthorectified aerial photography. We updated and verified 2011-2017 spatial data using the same methods as Cristescu et al. (2016) to ensure consistency between years.

We obtained annual land cover classification data (grain: 30 x 30 m), which categorized our study area into five categories: forest, grassland, shrub, water, and barren/rock (McDermid 2005). We created a dummy variable for forest land cover, where we coded forest with '1' and all other land cover classes with '0'. We defined forest edge as the linear boundary between forest land cover and another non-forest land cover classification, to represent edge habitat. We did not differentiate between edge habitat caused by mining activities and edge habitat caused by alternative sources of fragmentation (i.e. highways, cutlines, rugged terrain). However, mining is a leading cause of forest fragmentation in our study area, which is why we identified edge as a mine specific feature. We applied a 20-m buffer on both the inside and outside of all forest edges. We selected 20 m because this distance represents suitable stalking cover and hunting habitat for cougars (Beier et al. 1995, Laundre and Hernandez 2003, Holmes and Laundre 2006). We also created a dummy variable for grassland land cover, but further split all grasslands coded with '1' into either reclaimed grasslands or other grasslands. We defined reclaimed grasslands as grasslands that arose due to post-mining reclamation through seeding of agronomic species, rather than alternative anthropogenic

causes (i.e. seeding verges after road construction). We titled all grassland that was not introduced through mining reclamation as 'other grassland'; other grasslands arose due to a combination of anthropogenic activities (other than mining) and natural environmental processes. We represented permanent river and streams as riparian areas. We defined haul roads as wide gravel roads (3 − 4 lanes wide) that receive regular daily traffic from heavy equipment (i.e. haul trucks) transporting coal on the mine sites. We defined main roads as roads with ≥ 2 lanes, servicing only areas off the mine sites (i.e. Highway 40). We defined disturbed areas as areas of active or inactive mining, excluding the haul road. We represented bench walls, foot walls, and free-dumped talus as 'high walls'. We included mining high walls because they represent sheep escape terrain (MacCallum and Geist 1992). We used DEMs to calculate a terrain ruggedness index (TRI; (Riley et al. 1999)) and a relative topographic position index (RTP; (Jenness 2006)). TRI is highly correlated with slope, but we chose to include TRI because it was most relevant for all study species. We chose not to include elevation as a covariate because the elevation range in our study area is narrow. We chose not to include aspect as a covariate to reduce extraneous covariates and because previous studies show that ungulates in west-central Alberta select south-facing aspects (Telfer 1978).

For each landscape feature, we created a 30 x 30-m raster layer to represent the Euclidean distance from the center of each raster cell the nearest point of each respective feature. Using this method, we calculated distance to nearest forest, reclaimed grassland, other grassland, riparian area, haul road, main road, disturbed area, and high wall for each raster cell. Because the scale of our 200 x 200-m sampling units is

coarser than the scale at which we created our raster layers, we reassigned raster values to each sampling unit. We reassigned distance to nearest forest, reclaimed grassland, other grassland, riparian area, haul road, main road, disturbed area, and high wall based on the mean value of raster cells within each sampling unit. We reassigned TRI and RTP based on the mean index value across each sampling unit. Because we buffered linear forest edges with a ± 20-m buffer to create the edge habitat feature, calculating a Euclidean distance to the nearest edge habitat would not be applicable. Thus, we calculated the proportion of edge habitat within each sampling unit.

Habitat modelling

Habitat selection – For each species, we employed a used vs. available design (Manly et al. 2002, Johnson et al. 2006) to create an exponential resource selection function (RSF), fitted with logistic regression:

$$RSF(x) = \exp(\beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \cdots + \beta_n x_n) \qquad \text{(Eq 2.1)}$$

where β_i represents the selection coefficient for covariate x_i, for n total covariates. Because ungulates were surveyed a mean of 4.1 times per year, the results from surveys represent instantaneous examples of habitat selection and we cannot be exactly certain how each species is selecting habitat during the remainder of each year. Thus, we interpreted sampling units with > 0 ungulate centroids as each species' 'used' habitat among a choice of 'available' habitat. We defined our 'available' habitat as a random sample of sampling units within the specific truncation distance, w, for each species (Table 2.2). We selected 'available' habitat at a ratio of 1 used sampling unit to 5 available

sampling units, based on the mean number of sampling units used annually. We replicated the same suite of 'available' sampling units each year for the 14-year study period, to represent the same domain of availability each year.

Intensity of habitat use – We used linear regression to model the number of individuals per group as a function of landscape covariates:

$$IHU(x) = \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \cdots + \beta_n x_n \qquad \text{(Eq 2.2)}$$

where IHU is intensity of habitat use, β_i represents the selection coefficient for covariate x_i, for n total covariates. We log-transformed our response variable before modelling to improve assumptions of normality. We used linear regression to model intensity of habitat use after count methods for estimating intensity of habitat use (i.e. Poisson or negative binomial distributions) failed (Appendix 2.3).

For habitat selection models, we designed each species' candidate set of models *a priori* to evaluate alternative hypotheses (Table 2.4). For intensity of habitat use models, we were less certain about which covariates would influence group size and, thus, approached model selection differently. We considered all combinations of covariates up to moderately complex models (i.e. models containing five covariates) in our candidate set of models. For both habitat selection and intensity of use models, we standardized ($\bar{x}$ = 0, SD = 1) all continuous covariates, then tested them for multicollinearity using Pearson's correlation coefficient. We included only biologically relevant and non-highly-correlated ($|r| < 0.67$, determined *a priori*) covariates in models. We did not consider interaction terms or non-linearities in covariates. Because distance to high wall and

distance to reclaimed grassland were highly correlated, we did not include both

covariates (Appendix 2.4). We chose to include distance to high wall in bighorn sheep

models because bighorn dependence on escape terrain is well documented (Smith et al.

1991, Andrew et al. 1999, Singer et al. 2000, Bleich et al. 2009). Moreover, we chose to

include distance to reclaimed grassland in elk and mule deer models because high walls

have little biological significance to either species. Further, we did not consider distance

to forest or riparian areas in bighorn sheep models because there is little support from

the scientific literature that bighorn sheep select or avoid either forests or riparian areas.

We compared Akaike's Information Criterion corrected for small sample size

(AICc) of null models with and without a random intercept for year and season to account

for annual/seasonal variation in habitat use and determined that including a random

effect did not improve model fit. Additionally, we explored the option of creating

separate seasonal models to better approximate habitat selection and intensity of habitat

use throughout the year, but found that limited seasonal sample sizes prevented reliable

modelling (Appendix 2.5). As an alternative, we accounted for seasonal variation in group

size (Appendix 2.5) by including season as a main effect in all intensity of habitat use

models, without directly testing hypotheses related to the effect of season on intensity of

habitat use. We did not consider season as an interaction term to ensure that habitat

selection and intensity of use models could be easily interpreted and applied for

management. Lastly, we calculated AICc and Akaike weights (Burnham and Anderson

2002) for each model in a candidate set. We considered models with ΔAICc < 2.0 as

competitive (Burnham and Anderson 2002) and used AICc to select the most parsimonious top model (i.e. competitive model with fewest parameters).

We validated top habitat use models with 5-fold cross validation (Boyce et al. 2002). For habitat selection models, we divided our data into 5 folds and tested the relationship between area-adjusted frequency of predicted 'used' observations and 10 RSF bin ranks, for each of the 5 folds. We used the mean Spearman's rank correlation coefficient ($\bar{r}_s$) averaged across all folds to determine the predictive capability for each top RSF model. For intensity of habitat use models, we divided our data into 5 folds and calculated a mean Spearman's rank correlation coefficient ($\bar{r}_s$) between observed and predicted intensity of habitat use, averaged across folds. We resolved which landscape features were important in determining bighorn sheep, elk, and mule deer habitat use based on covariate presence in top models.

Strength of response to mining and reclamation

By comparing scaled beta coefficients for landscape covariates, we quantified whether a species had a strong or weak strength of response to mining and reclamation. For ease of discussion, we classified strong selection as $\beta \geq 1$, moderate selection as $0.2 \leq \beta < 1$, and weak selection as $0 < \beta < 0.2$ and classified strong avoidance as $\beta \leq -1$, moderate avoidance as $-0.2 \geq \beta > -1$, and weak avoidance as $0 > \beta > -0.2$. Weak selection and weak avoidance, however, may represent betas where standard errors overlap 0. Finally, to visualize ungulate responses to mining and reclamation spatially, we created predictive maps using top habitat selection models for each species. Predictive maps were created

for the entire study area (i.e. beyond truncation distances) in ArcGIS using unscaled beta coefficients from top habitat selection models. Predictive maps were symbolized using equal interval bins for ease of interpreting the maps. However, equal interval bins may not contain equal sample sizes and do not consider how data are distributed. This approach may limit inference on how many sampling units have high vs. low relative habitat selection.

RESULTS

Habitat modelling

Habitat selection – We observed bighorn sheep using 2,871 sampling units between 2004 and 2017. The top RSF model for bighorn sheep included the base model in addition to disturbed areas, haul roads, main roads, and other grasslands (Model Six, Table 2.5). These results support our hypothesis that the base model alone is unable to fully explain bighorn sheep habitat selection, compared to models that included mining specific features. Bighorn sheep selected habitat closer to high walls, with less edge habitat, and at higher RTP, which supported our predictions (Tables 2.5-2.7; $\bar{r}_s$ = 0.94). Contrary to our predictions, bighorn sheep selected habitat closer to haul roads, disturbed areas, and main roads (Tables 2.5-2.7; $\bar{r}_s$ = 0.94). Bighorn sheep also avoided other grasslands.

We observed elk using 819 different sampling units between 2004-2017. Similar to bighorn sheep, the top RSF model for elk included the base model in addition to mining specific features and other grasslands (Model Five, Table 2.5). Consistent with our predictions, elk selected habitats closer to reclaimed grasslands, farther from disturbed

and riparian areas, and at higher RTPs (Tables 2.5-2.7; $\bar{r}_s$ = 0.89). However, elk selected habitats closer to haul roads and farther from forests, which was contrary to our predictions (Tables 2.5-2.7; $\bar{r}_s$ = 0.89). Elk also selected lower terrain ruggedness and avoided other grasslands.

Lastly, we observed mule deer using 782 different sampling units during the study period. Similar to bighorn sheep and elk, the top RSF model for mule deer also included the base model in addition to mining specific features (Model Seven, Table 2.5). Mule deer selected habitats closer to reclaimed grasslands and disturbed areas, farther from haul roads, at higher RTP, and with less edge habitat, which supported our original expectations (Tables 2.5-2.7; $\bar{r}_s$ = 0.94). In contrast with our expectations, mule deer selected habitat farther from forests and riparian areas (Tables 2.5-2.7; $\bar{r}_s$ = 0.94). Similar to elk, mule deer selected habitats with lower terrain ruggedness and farther from other grasslands.

Intensity of habitat use – We observed 3,680, 943, and 855 different groups of bighorn sheep, elk, and mule deer during the 2004-2017 study period. The breadth of landscape features included in top intensity of habitat use models was narrower than the breadth of landscape features included in top habitat selection models. In the top linear regression model, bighorn sheep used habitats farther from main roads, at higher relative topographic positions, and lower terrain ruggedness more intensely (Tables 2.6, 2.8, 2.9; $\bar{r}_s$ = 0.18). The top model for intensity of elk habitat use included more covariates than the top model for bighorn sheep. Elk used habitat closer to haul roads, farther from disturbed areas, at higher RTP, and with less edge more intensely (Tables 2.6, 2.8, 2.9; $\bar{r}_s$

= 0.31). The top model for intensity of habitat use was the simplest for mule deer and included only one covariate, which may have been due to sample size limitations. Mule deer used habitats with less edge more intensely (Tables 2.6, 2.8, 2.9; $\bar{r}_s$ = 0.31).

Strength of response to mining and reclamation

As we expected, bighorn sheep exhibited strong selection for high walls ($\beta_{dihwall}$ = -8.11, SE = 0.36; Table 2.7). Bighorn sheep also exhibited moderate selection for high RTP, moderate avoidance of other grasslands and edge habitat, and weak selection for disturbed areas, haul roads, and main roads (β_{rtp}= 0.28, SE = 0.02; β_{diogra}= 0.30, SE = 0.03; β_{pedge}= -0.29, SE = 0.03; β_{didist} = -0.13, SE = 0.04; $\beta_{dihroad}$ = -0.10, SE = 0.04; $\beta_{dimroad}$ = -0.13, SE = 0.03; Table 2.7). Similar to bighorn sheep, elk exhibited moderate selection for high RTP and weak avoidance of other grasslands (β_{rtp}= 0.23, SE = 0.04; β_{diogra}= 0.11, SE = 0.05; Table 2.7). In contrast to bighorn sheep, elk exhibited stronger selection for haul roads and avoidance of disturbed areas ($\beta_{dihroad}$= -0.33, SE = 0.07; β_{didist}= 0.33, SE = 0.06; Table 2.7). Elk did, however, strongly select reclaimed grasslands, as we predicted (β_{dirgra} = -1.07, SE = 0.18; Table 2.7). Elk also moderately avoided riparian areas and rugged terrain, and weakly avoided forests (β_{dirip} = 0.30, SE = 0.04; β_{tri} = -0.36, SE = 0.04; $\beta_{diforest}$ = 0.07, SE = 0.04; Table 2.7). Comparable to elk, mule deer strongly selected reclaimed grasslands, moderately avoided riparian areas and rugged terrain, and weakly avoided forests (β_{dirgra}= -1.02, SE = 0.17; β_{dirip} = 0.22, SE = 0.04; β_{tri} = -0.25, SE = 0.05; $\beta_{diforest}$ = 0.14, SE = 0.05; Table 2.7). Mule deer more strongly selected disturbed areas than bighorn sheep and more strongly avoided other

grasslands than elk (β_{didist} = -0.46, SE = 0.05; β_{diogra} = 0.35, SE = 0.05; Table 2.7).

Moreover, mule deer moderately avoided haul roads, which was opposite to both

bighorn sheep and elk ($\beta_{dihroad}$ = 0.63, SE = 0.06; Table 2.7). Lastly, mule deer exhibited

weaker selection for high RTP than both other species, and weak avoidance of edge

habitat (β_{rtp} = 0.20, SE = 0.04; β_{pedge} = 0.14, SE = 0.05; Table 2.7).

DISCUSSION

Our results demonstrate that bighorn sheep, elk, and mule deer selected landscape

features to increase access to quality forage and decrease predation risk. For example, all

study species selected reclaimed grasslands, presumably to forage on high quality

agronomic forbs and grasses, and avoided other grasslands. Secondly, all study species

selected high relative topographic positions, such as hilltops and ridges, which provide

unobstructed visibility, early detection of predators, and an advantageous position for

escaping relative to local topography (Kuck et al. 1985, Cassirer et al. 1992, Kunkel and

Pletscher 2001, Mao et al. 2005).

Bighorn sheep in our study selected high walls, while avoiding forest edges, both

of which are consistent with bighorn sheep anti-predator strategies. Bighorn sheep use

steep, rugged features, such as mining high walls (MacCallum and Geist 1992, Poole et al.

2016), as escape terrain to evade predators (Berger 1991, Bleich et al. 1997). By avoiding

forest edges, bighorn sheep distanced themselves from risky habitat where they are

more vulnerable to predation by cougars (Beier et al. 1995, Laundre and Hernandez 2003,

Laundre and Loxterman 2006, Holmes and Laundre 2006) or wolves (McPhee et al. 2012).

We predicted that elk would select habitats to maximize foraging while reducing vulnerability to predation, and our results support these predictions. Elk selected reclaimed grasslands and ridges on the reclaimed mines, while avoiding rugged terrain, forests, riparian areas, and other grasslands. Because ridges and hills in our study area arose from mining and subsequent reclamation, they are generally flat on top, which is consistent with our finding that elk avoided rugged terrain. Further, elk graze in open habitat to avoid predators that hunt along forest edges (Mao et al. 2005), such as cougars (Beier et al. 1995, Laundre and Hernandez 2003, Laundre and Loxterman 2006, Holmes and Laundre 2006) and wolves (McPhee et al. 2012), but retreat to forests on temporally fine scales when threatened (Creel et al. 2005). Although we did not expect to detect elk avoidance of forests due to the coarse temporal scale of our study, we found that elk weakly avoided forests. This suggests that elk might not seek forest cover when threatened. In other studies, elk fled uphill to ridges before seeking forest cover when disturbed (Kuck et al. 1985, Cassirer et al. 1992). As an alternative to seeking forest cover when threatened, elk likely congregated into large groups to dilute individual risk of predation (Dehn 1990, Jedrezejewski et al. 1992, Hebblewhite and Pletscher 2002) and sought cover using topography rather than forests. Lastly, elk avoided riparian areas, which are travel paths for predators (Kunkel and Pletscher 2001, Dickson and Beier 2002, Bergman et al. 2006, Kauffman et al. 2007) and terrain traps for elk (Kauffman et al. 2007).

Although mule deer selected grasslands that increased forage quality and high relative topographic positions, mule deer also responded to landscape features in ways

that we did not expect. We expected that mule deer would select forests to consume browse but found that they avoided forests. Mule deer strongly select for deciduous-dominated tree stands due to availability of browse and less often select for conifer-dominated tree stands due to limited browse availability in these areas (Serrouya and D'Eon 2008). Forests in our study area are mainly conifer at high elevations with some mixed conifer-deciduous at lower elevations, but we did not differentiate between forest types for our forest covariate. As such, we reason that we might not have been able to detect selection for deciduous stands that provide ample browse for mule deer and, instead, were only able to detect avoidance of conifer-dominated stands with sparse browse. Mule deer also selected habitats with greater exposure to edge habitat. Deer are the most common prey item for cougars (Knopff et al. 2010a, Bacon et al. 2011) and forest edges are the most exploited landscape feature (Beier et al. 1995, Laundre and Hernandez 2003, Laundre and Loxterman 2006, Holmes and Laundre 2006) by cougars, so we expected mule deer to avoid edges. Even though mule deer avoided forests, there may be energetic benefits for mule deer to use edge habitats because these areas allow deer to forage in the open, while remaining close to protective cover. Like elk, mule deer avoided riparian areas, suggesting that predation risk associated with riparian areas outweighed the benefit of foraging for browse. The vast extent of reclaimed grasslands probably provided adequate food resources for mule deer such that they did not have to seek browse to supplement. We also found that mule deer avoided rugged terrain. Lingle (2002) suggests that over short temporal windows, mule deer forage on gentler slopes and move into rugged habitats when predators are close (Lingle 2002, Lingle et al. 2008).

As such, we submit that the temporal scale of our analysis was not fine enough to detect mule deer selecting rugged terrain.

Bighorn sheep, elk, and mule deer responded to mining and reclamation features in ways that we did not anticipate. Jansen et al. (2006) found that bighorn sheep avoided areas of high disturbance on mines, so we originally predicted that bighorn sheep would likewise avoid disturbed areas, haul roads, and main roads to reduce exposure to perceived human threats. However, bighorn sheep selected all of these features. Bighorn sheep are sensitive to unpredictable events (MacArthur et al. 1982, Wiedmann and Bleich 2014) but can habituate to predictable human disturbance (Hicks and Elder 1979, MacArthur et al. 1982). Mining activities in our study area are predictable. In disturbed areas, humans operate heavy equipment to remove coal in a consistent manner, without directly interacting with wildlife. Coal is deposited into haul trucks that travel haul roads at low speeds, routinely throughout the day and night. Furthermore, predators often avoid humans (Hebblewhite et al. 2005, Ordiz et al. 2011, Morrison et al. 2014, Jennings et al. 2016), consequently lowering the predation risk and creating refuges for prey in the areas that they avoid (Hebblewhite and Merrill 2007, Schmidt and Kuijper 2015). We suggest that bighorn sheep sought human refuge in disturbed areas and near haul roads on the mines to exploit lower predation risk closer to humans and mining activities. Similarly, Jansen et al. (2009) proposed that bighorn sheep selected disturbed areas as a source of human refuge on surface mines in Arizona, USA. Furthermore, bighorn sheep in our study selected for main roads, which we did not expect. Main roads have less predictable vehicular traffic patterns than haul roads and we rationalize that bighorn

sheep did not select main roads as human refuge. Instead, we propose that bighorn sheep selected main roads (and haul roads too) as a source of minerals. Specifically, we often observed bighorn sheep licking segments of Highway 40 South and haul roads (Pers. Obs.). Bighorn sheep use mineral licks to access vital metabolic salts (Geist 1971, MacCallum and Geist 1992), required for synthesis of hair, hooves, and horns (Jones and Hanson 1985). Moreover, sides of highways are seeded with agronomic mixes in Alberta (Roever et al. 2008) and are mowed consistently throughout snow-free months (Pers. Obs.), producing roadside grasslands with unobstructed visibility. The spatial scale of our landscape data was not fine enough to adequately represent these roadside grasslands in our 'other grasslands' covariate. Thus, we submit that bighorn sheep were not only drawn to main roads as a mineral lick, but drawn to the adjacent forage available.

Elk also seek human refuge as an anti-predator strategy (Hebblewhite and Merrill 2007, Shannon et al. 2014) and we suggest that elk selected habitat closer to haul roads to exploit such refuges. If elk selected human refuge similarly to bighorn sheep, then we would expect elk to select for both disturbed areas and haul roads. Instead, we found that elk selected for haul roads but avoided disturbed areas, which is a puzzling result. We therefore reason that disturbed areas and haul roads do not provide equal human refuge for elk, whereas these areas may for bighorn sheep. Haul roads receive more frequent and predictable human activity than disturbed areas (Pers. Obs.). For instance, once land is mined, the land is considered 'disturbed' until it is reclaimed, but this also means that land can remain inactive with regards to human activity but still be considered 'disturbed' for our analyses. Therefore, elk may be more sensitive to the

slightly less predictable activity in disturbed areas and, hence, only select habitat near haul roads when seeking human refuge.

Mule deer selected for disturbed areas and against haul roads, and this selection against haul roads was unique to mule deer. For instance, elk selected haul roads but avoided disturbed areas, whereas mule deer responded exactly oppositely. Johnson et al. (2000) found that mule deer selected for roads that elk avoided, which provides potential corroboration for our findings. Mule deer might be displaced to lower-quality habitats to avoid competition with elk (Johnson et al. 2000, Ager et al. 2003, Stewart et al. 2010, Lendrum et al. 2012), which might be occurring in our study. Alternatively, mule deer might not rely on human refuge as an anti-predator strategy and instead rely on topography and rugged terrain to detect and avoid predators, respectively. Both possibilities provide an explanation for why mule deer were the only study species to avoid haul roads.

Cross-validation results indicate that the predictive capability for habitat selection models is much higher than for intensity-of-use models. Intensity-of-use models performed poorly when validated and for this reason, we elected not to thoroughly interpret these results or make predictive maps for intensity of habitat use. We suggest that group size may not be the best proxy for inferring spatial intensity of habitat use. We attempted to model intensity of habitat use using number of groups per sampling unit as the response variable, but we were not successful; please see Appendix 2.3 for further details. Habitat selection models, however, performed extremely well based on cross-validation scores, indicating that we can confidently predict habitat selection, even when

observations are pooled over seasons. In other study systems, seasonal differences in ungulate habitat selection reflect changing disturbances (Rowland et al. 2000, Sawyer et al. 2007) and predation risk (Beck et al. 2013), which supplies a motivation for separate seasonal models. Because most 'used' sampling units in our study arose during fall, landscape features presenting in top habitat selection models are biased to include features that determine where bighorn sheep, elk, and mule deer congregate to mate. Small sample sizes limited our ability to model habitat selection separately for four seasons (Appendix 2.5), which could have helped us to decipher what affect season had as a main effect in habitat-selection models. To avoid issues with sample size, we modeled habitat selection split by two seasons, snow and snow-free periods, but found support that pooled seasonal models are as capable as separate seasonal models at predicting habitat selection (Appendix 2.5). Thus, we feel confident that our interpretations of habitat selection for bighorn sheep, elk, and mule deer are relevant, despite models being slightly biased to reflect fall habitat selection.

Habitat selection models indicated that bighorn sheep, elk, and mule deer exhibited strong responses to two highly correlated features. Bighorn sheep strongly selected high walls while elk and mule deer strongly selected reclaimed grasslands. As we predicted, bighorn sheep exhibited the strongest response to mining and reclamation (Table 2.7). Our results support many previous studies that emphasize how critical escape terrain is to bighorn sheep (Smith et al. 1991, Andrew et al. 1999, Singer et al. 2000, Bleich et al. 2009). Moreover, our results support Hudson et al. (1976), who compared habitat breadth among our three study species in east Kootenay, British Columbia,

Canada. Hudson et al. (1976) concluded that bighorn sheep had the most specific response to environment and subsequently a narrow niche breadth, while elk had the weakest response to environment and subsequently a wide niche breadth; mule deer fell somewhere in the middle of the two other species. Maps of predicted relative habitat selection (Figures 2.2 & 2.3) align with Hudson et al. (1976) and highlight that areas of high relative bighorn sheep habitat selection are tightly clustered on Gregg River and Luscar, surrounding high walls and reclaimed grasslands. In contrast, areas of high relative elk habitat selection are somewhat centralized around reclaimed grasslands, but are also spread widely (Figures 2.2 & 2.4). Lastly, areas of high relative mule deer habitat selection are slightly more dispersed than bighorn sheep (Figures 2.2 & 2.5) and also appear to be more tightly clustered around reclaimed grasslands than elk.

Unfortunately, high correlation between high walls and reclaimed grasslands makes it difficult to discern how ungulates are partitioning resources. Landscapes in our study area were designed so that most high walls had reclaimed grasslands seeded in close proximity, causing high correlation between these features. One drawback of this particular landscape design, albeit its success at supporting large bighorn populations, is that we cannot determine whether our study species are more strongly selecting reclaimed grasslands or high walls. High correlation between these features is a major issue for modelling habitat selection in bighorn sheep because they use both high walls and forage on adjacent reclaimed grasslands (Smith et al. 1991), and we cannot adequately tease apart the contributing effects of each feature in our models. To identify which species are driving overall resource partitioning, niche overlap should be

considered. For example, diet and habitat overlap should be measured, as metrics for inferring potential interspecific competition (Sinclair 1985, Ihl and Klein 2001, Vila et al. 2009, Namgail et al. 2010, Jung et al. 2015).

In conclusion, our results emphasize that ungulates select habitats that either increase access to quality forage or decrease predation risk. Our results demonstrate that ungulates are capable of exploiting human-modified landscapes and may even seek refuge from predators in areas of predictable human disturbance, such as near haul roads. In study systems where many ungulates coexist, such as ours, we must understand how species differentially select for the same landscape features. For example, bighorn sheep are dependent on high walls as escape terrain and all study species depend on the tops of hills and ridges for maintaining vigilance. Understanding these interspecific differences in response to external environment are even more important in industrially and anthropogenically modified landscapes, where humans influence landscape design. Our findings provide a foundation for which future research on landscape design and reclamation can be created. If disentangling the effects of each mining feature on ungulates is integral for future ecological research, then landscape design must include high walls without reclaimed grasslands immediately proximate. Instead, high walls could be surrounded by talus or shrubs as an experimental approach. If creating habitat for wildlife is integral for future ecological reclamation, regardless of the mechanisms of habitat selection, then future landscape design need not consider separating high walls from seeded grasslands and should instead focus on developing high walls and flat ridges seeded with forbs and grasses.

Lastly, our study area is of particular interest to Albertans because Gregg River and Luscar will eventually revert to the Crown in upcoming years. These mines host large bighorn, elk, and mule deer populations, and Figures 2.2-2.5 highlight areas where these species are likely to select. Once these mines revert to the Crown, the land will no longer be managed by mining companies and the Province of Alberta will determine how the land is legally designated and used. For example, the Land Management Plan for Gregg River and Luscar (Government of Alberta 2013) proposed a 1-km buffer on either side of Highway 40 South, where wildlife is protected from hunting (Figure 2.6). If protecting wildlife populations is a main goal of the end land use strategy, then extending the 1-km buffer may be necessary to protect more of the habitat that bighorn sheep and elk are likely to select. If the proposal should be approved, ungulates are likely to be displaced from and eventually abandon habitat on Gregg River and Luscar, in response to the introduction of erratic and unpredictable human recreational activities such as hiking and hunting (Wiedmann and Bleich 2014). Overall, the findings from our study become increasingly relevant as the Alberta government approves management strategies for these reclaimed mines.

Table 2.1 Four seasons that ungulate observations were assigned to for use in intensity of habitat use models.

Season	Bighorn sheep	Elk & mule deer
Spring	March 15 – May 31	
Summer	June 1 – August 11	June 1 – August 31
Fall	August 12 – December 31	September 1 – November 30
Winter	January 1 – March 14	December 1 – March 14

Table 2.2 Summary of truncation distances and subsequent sampling units removed for each species and habitat. A unique truncation distance (w; in meters) is summarized per species and habitat. Overall truncation distances (w_2; in meters) are summarized per species, as well as the resulting number and proportion of sampling units removed by right-truncating data (see Buckland et al. (2001)).

Species	Habitat	Truncation distance (w)	Overall truncation distance (w_2)	Units removed[a] [total[b]]	Proportion units removed[c]
Bighorn sheep	Open	300	300	94 [552]	0.17
	Forest	400			
Elk	Open	400	400	15 [237]	0.06
	Forest	400			
Mule deer	Open	300	300	39 [288]	0.14
	Forest	400			

[a] Sampling units with count data removed by right-truncating species' datasets

[b] Total sampling units with count data

[c] Sampling units with count data removed by truncation ÷ total sampling units with count data

Table 2.3 Summary of landscape features as covariates for modelling ungulate habitat use on reclaimed mines in west-central Alberta. When using "distance to" we mean "distance to nearest" feature. Covariate code, type of variable, and units are displayed.

Covariate	Covariate Code	Type of Variable	Unit
Non-mining specific features			
Distance to forest[a]	diforest	Continuous	Meter
Distance to main road	dimroad	Continuous	Meter
Distance to other grassland	diogra	Continuous	Meter
Distance to riparian area[a]	dirip	Continuous	Meter
Terrain ruggedness index	tri	Continuous	Unitless
Relative topographic position index	rtp	Continuous	Unitless
Mining specific features			
Distance to high wall[b]	dihwall	Continuous	Meter
Distance to haul road	dihroad	Continuous	Meter
Proportion edge habitat	pedge	Continuous	Unitless
Distance to disturbed areas	didist	Continuous	Meter
Distance to reclaimed grassland[a]	dirgra	Continuous	Meter

[a] Distance to forest, riparian area, and reclaimed grassland were included in elk and mule deer habitat use models only

[b] Distance to high wall was included in bighorn sheep habitat use models only

Table 2.4 *A priori* candidate model sets for the analysis of bighorn sheep, elk, and mule deer habitat selection on reclaimed mines. Separate model sets were created for bighorn sheep and elk & mule deer to reflect different biologically relevant covariates that were considered for each species.

Model	Explanatory variables	What does model evaluate?
Bighorn sheep		
Null	Intercept only	Landscape features predict habitat selection
Base	dihwall + pedge + rtp	BS[a] avoid predation by cougars and wolves, and maintain unobstructed views
One	Base + didist	BS avoid only disturbed areas
Two	Base + dihroad	BS avoid only haul roads
Three	Base + didist + dihroad	BS avoid only mining disturbances
Four	Base + didist + dihroad + dimroad	BS avoid all disturbances
Five	Three + diogra	BS avoid other grasslands
Six	Four + diogra	BS avoid other grasslands
Seven	Three + diogra + tri	BS select for rugged terrain
Eight	Four + diogra + tri	BS select for rugged terrain
Elk & Mule deer		
Null	Intercept only	Landscape features predict habitat selection
Base	dirgra + diforest + rtp + tri + dirip	Elk & MD[b] select habitat as a trade-off between acquiring forage and minimizing predation risk
One	Base + didist	Elk & MD avoid only disturbed areas
Two	Base + dihroad	Elk & MD avoid only haul roads
Three	Base + didist + dihroad	Elk & MD avoid only mining disturbances
Four	Base + didist + dihroad + dimroad	Elk & MD avoid all disturbances
Five	Three + diogra	Elk & MD avoid other grasslands
Six	Four + diogra	Elk & MD avoid other grasslands
Seven	Three + diogra + pedge	Elk & MD avoid increased edge habitat on mines
Eight	Four + diogra + pedge	Elk & MD avoid increased edge habitat on mines

[a] BS = bighorn sheep
[b] MD = mule deer

Table 2.5 Top-ranked logistic regression models to estimate habitat selection for bighorn sheep, elk, and mule deer on reclaimed mines in west-central Alberta, Canada. Number of estimated parameters (K), model negative log-likelihood (LL), small sample size corrected Akaike's Information Criterion (AICc), AICc difference (ΔAICc) and Akaike weight (w_i) are displayed. The top model for each species' panel is in bold.

Species	Rank	Model[a]	K	LL	AICc	ΔAICc	w_i
	1	Eight	9	-6625.18	13268.37	0	0.634
	2	**Six**	**8**	**-6626.73**	**13269.46**	**1.097**	**0.366**
	3	Seven	8	-6634.38	13284.77	16.408	0
	4	Five	7	-6636.84	13287.69	19.32	0
Bighorn	5	Four	7	-6693.09	13400.19	131.822	0
sheep	6	One	5	-6733.84	13477.68	209.312	0
	7	Three	6	-6733.56	13479.13	210.761	0
	8	Two	5	-6746.39	13502.78	234.41	0
	9	Base	4	-6753.66	13515.32	246.956	0
	10	Null	1	-7895.72	15793.44	2525.069	0
	1	Six	10	-2035.27	4090.58	0	0.316
	2	Eight	11	-2034.52	4091.1	0.519	0.244
	3	**Five**	**9**	**-2036.69**	**4091.42**	**0.844**	**0.207**
	4	Seven	10	-2035.92	4091.88	1.304	0.165
	5	Three	8	-2039.37	4094.77	4.197	0.039
Elk	6	Four	9	-2038.61	4095.25	4.672	0.031
	7	One	7	-2049.14	4112.31	21.728	0
	8	Base	6	-2057	4126.02	35.448	0
	9	Two	7	-2056.92	4127.86	37.286	0
	10	Null	1	-2247.81	4497.62	407.043	0
	1	Eight	11	-1918.84	3859.72	0	0.624
	2	**Seven**	**10**	**-1920.5**	**3861.04**	**1.311**	**0.324**
	3	Six	10	-1922.64	3865.33	5.608	0.038
	4	Five	9	-1924.67	3867.38	7.658	0.014
Mule	5	Four	9	-1949.49	3917.01	57.288	0
deer	6	Three	8	-1953.94	3923.91	64.183	0
	7	Two	7	-1985.07	3984.16	124.436	0
	8	Base	6	-2026.2	4064.42	204.698	0
	9	One	7	-2026.14	4066.31	206.581	0
	10	Null	1	-2155.57	4313.14	453.417	0

[a] Model structure based on candidate set listed in Table 2.4

Table 2.6 Summary of 5-fold cross validation results for top habitat selection and top intensity of habitat use models for bighorn sheep, elk, and mule deer. A mean Spearman's rank correlation coefficient ($\bar{r}_s$) and p-value are provided. Signs indicate the sign of the beta coefficient estimates.

Species	Top model	$\bar{r}_s$ [a]	p-value[a]
Habitat selection			
Bighorn sheep	− dihwall − pedge + rtp − didist − dihroad − dimroad + diogra	0.94	< 0.001
Elk	− dirgra + diforest + rtp − tri + dirip + didist − dihroad + diogra	0.89	< 0.01
Mule deer	− dirgra + diforest + rtp − tri + dirip − didist + dihroad + diogra + pedge	0.94	< 0.001
Intensity of habitat use			
Bighorn sheep	season + rtp − tri + dimroad	0.18	< 0.0001
Elk	season − pedge + didist − dihroad + rtp	0.31	< 0.001
Mule deer	season − pedge	0.31	< 0.001

[a] $\bar{r}_s$ and p-values presented are means across five folds

Table 2.7 Scaled beta coefficients and 95% confidence intervals (CI) from top habitat selection models for bighorn sheep, elk, and mule deer, comparing used sampling units (bighorn *n* = 2,871, elk *n* = 819, mule deer *n* = 782) to available sampling units. Distance to high wall (dihwall) was included in only bighorn sheep models whereas distance to forest (diforest), riparian areas (dirip), and reclaimed grasslands (dirgra) were included in only elk and mule deer models.

| Covariate | Bighorn sheep | | | Elk | | | Mule deer | | |
| | β | 95% CI | | β | 95% CI | | β | 95% CI | |
		Lower	Upper		Lower	Upper		Lower	Upper
didist	-0.13	-0.20	-0.06	0.33	0.22	0.45	-0.46	-0.56	-0.35
diforest				0.07	-0.01	0.16	0.14	0.05	0.23
dihroad	-0.10	-0.16	-0.03	-0.33	-0.47	-0.20	0.63	0.51	0.76
dihwall	-8.11	-8.81	-7.41						
dimroad	-0.13	-0.19	-0.07						
diogra	0.30	0.25	0.35	0.11	0.02	0.21	0.35	0.26	0.44
dirgra				-0.90	-1.22	-0.58	-1.02	-1.36	-0.69
dirip				0.30	0.21	0.38	0.22	0.13	0.30
pedge	-0.29	-0.34	-0.23				0.14	0.04	0.23
rtp	0.28	0.24	0.33	0.23	0.15	0.30	0.20	0.11	0.28
tri				-0.36	-0.45	-0.27	-0.25	-0.34	-0.16

Table 2.8 The 5 top-ranked linear regression models for determining relative intensity of bighorn sheep, elk, and mule deer habitat use on reclaimed mines in west-central Alberta, Canada. Number of estimated parameters (K), model negative log-likelihood (LL), small sample size corrected Akaike's Information Criterion (AICc), AICc difference (ΔAICc) and Akaike weight (w_i) are displayed. Akaike weights have been readjusted to reflect this candidate set that includes only the top five models and the null model. The top model for each species panel is in bold. Signs indicate the sign of the beta coefficient estimates.

Species	Rank	Covariates	K	LL	AICc	ΔAICc	w_i
Bighorn sheep	1	season – pedge + dihwall + rtp – tri + dimroad	10	-5594.5	11209.05	0	0.276
	2	season – didist + dihroad + rtp – tri + dimroad	10	-5594.82	11209.69	0.639	0.201
	3	season – pedge + rtp – tri + dimroad	9	-5595.87	11209.78	0.727	0.192
	4	**season + rtp – tri + dimroad**	**8**	**-5596.99**	**11210.03**	**0.971**	**0.17**
	5	season + dihwall + rtp – tri + dimroad	9	-5596.04	11210.13	1.075	0.161
	Null	Season	5	-5641.72	11293.45	84.394	0
Elk	1	season – pedge + didist – dihroad + rtp – dimroad	10	-1511.16	3042.56	0	0.253
	2	**season – pedge + didist – dihroad + rtp**	**9**	**-1512.24**	**3042.68**	**0.113**	**0.239**
	3	season – pedge + didist – dirgra – dihroad + rtp	10	-1511.26	3042.76	0.201	0.229
	4	season – pedge + didist – dihroad + diogra + rtp	10	-1511.5	3043.24	0.678	0.181
	5	season – pedge + didist – dihroad + rtp – tri	10	-1512.12	3044.47	1.909	0.098
	Null	Season	5	-1541.48	3093.02	50.461	0
Mule deer	1	season – pedge + rtp	7	-863.3	1740.72	0	0.318
	2	season – pedge – dirgra + rtp	8	-862.8	1741.77	1.05	0.188
	3	**season – pedge**	**6**	**-864.86**	**1741.81**	**1.087**	**0.185**
	4	season – pedge + didist + rtp	8	-862.98	1742.13	1.404	0.158
	5	season – pedge + diogra + rtp	8	-863.03	1742.24	1.513	0.149
	Null	Season	5	-870.2	1750.47	9.75	0.002

Table 2.9 Scaled beta coefficients and 95% confidence intervals (CI) from top intensity of habitat use models for bighorn sheep, elk, and mule deer, comparing log-transformed group size (bighorn n groups = 3,680, elk n = 943, and mule deer n = 855) as a function of landscape covariates. Only covariates that appeared in at least one of the top models are included.

| Covariate | Bighorn sheep | | | Elk | | | Mule deer | | |
| | β | 95% CI | | β | 95% CI | | β | 95% CI | |
		Lower	Upper		Lower	Upper		Lower	Upper
didist				0.21	0.10	0.32			
dihroad				-0.33	-0.44	-0.22			
dimroad	0.07	0.03	0.11						
pedge				-0.11	-0.19	-0.03	-0.07	-0.12	-0.03
rtp	0.11	0.07	0.14	0.14	0.06	0.22			
tri	-0.11	-0.15	-0.08						
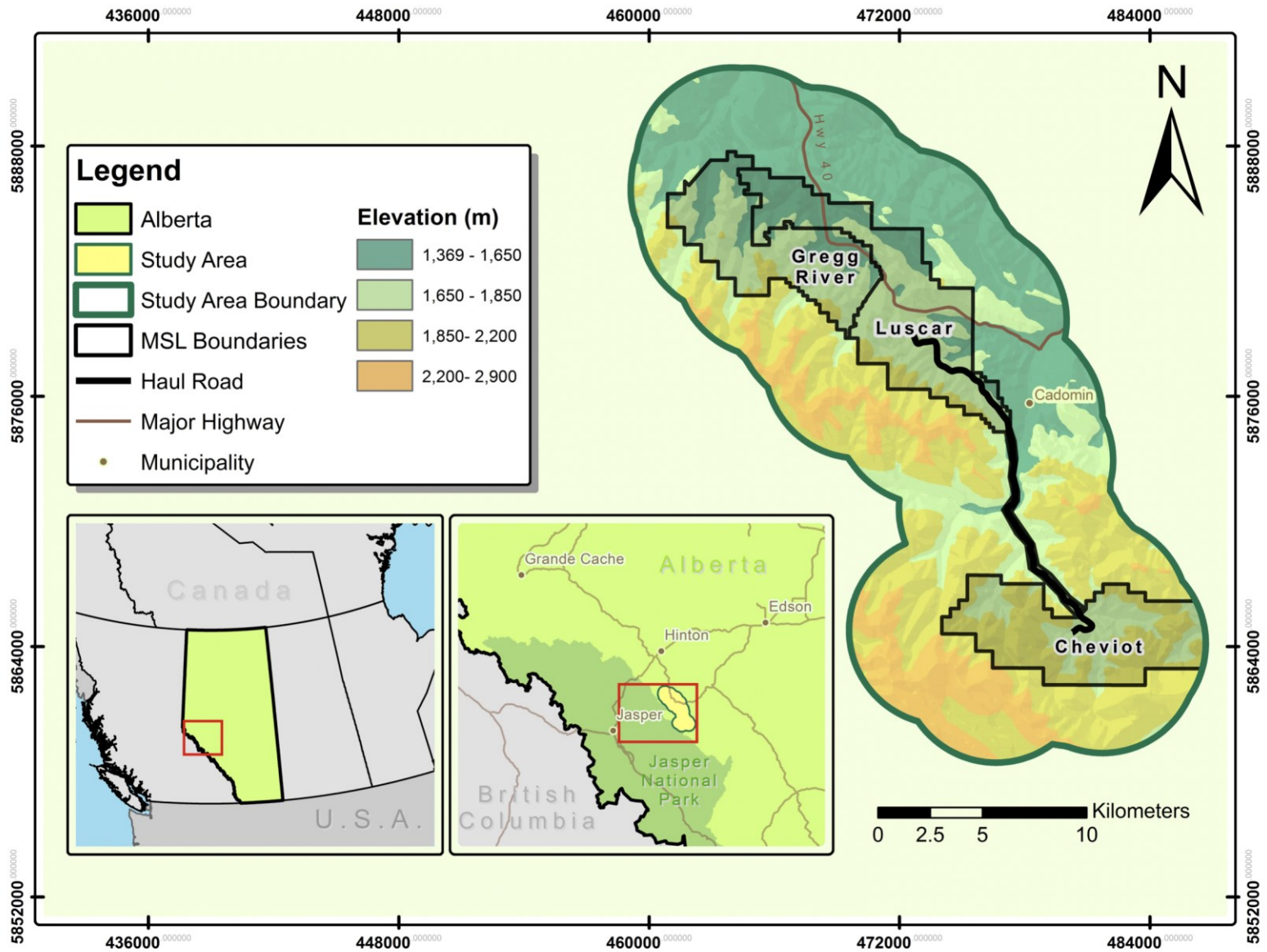
Figure 2.1 Three open pit coal mines, Gregg River, Luscar, and Cheviot, in west-central Alberta, Canada. The mines are located east of Jasper National Park, south-west of Hinton, and west of Cadomin, Alberta.
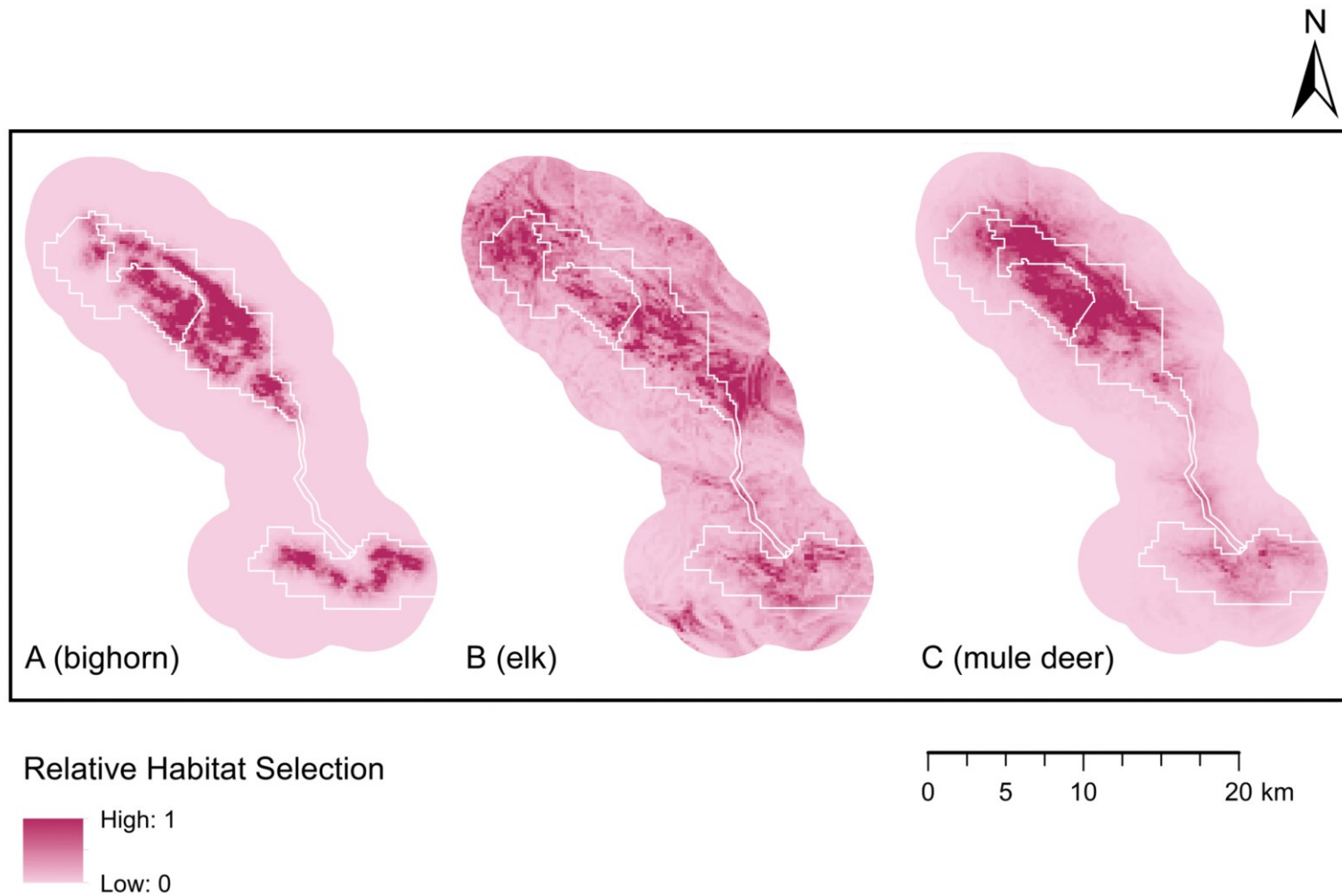
Figure 2.2 Predicted relative habitat selection for bighorn sheep (A), elk (B), and mule deer (C) during 2017 in the west-central Alberta study area, which encompasses three mines (outlined in white). Relative habitat selection was determined by fitting exponential RSFs, selecting a top model, and scaling predictions between 0 (low relative habitat selection; light pink) and 1 (high relative habitat selection; dark pink) for each species.
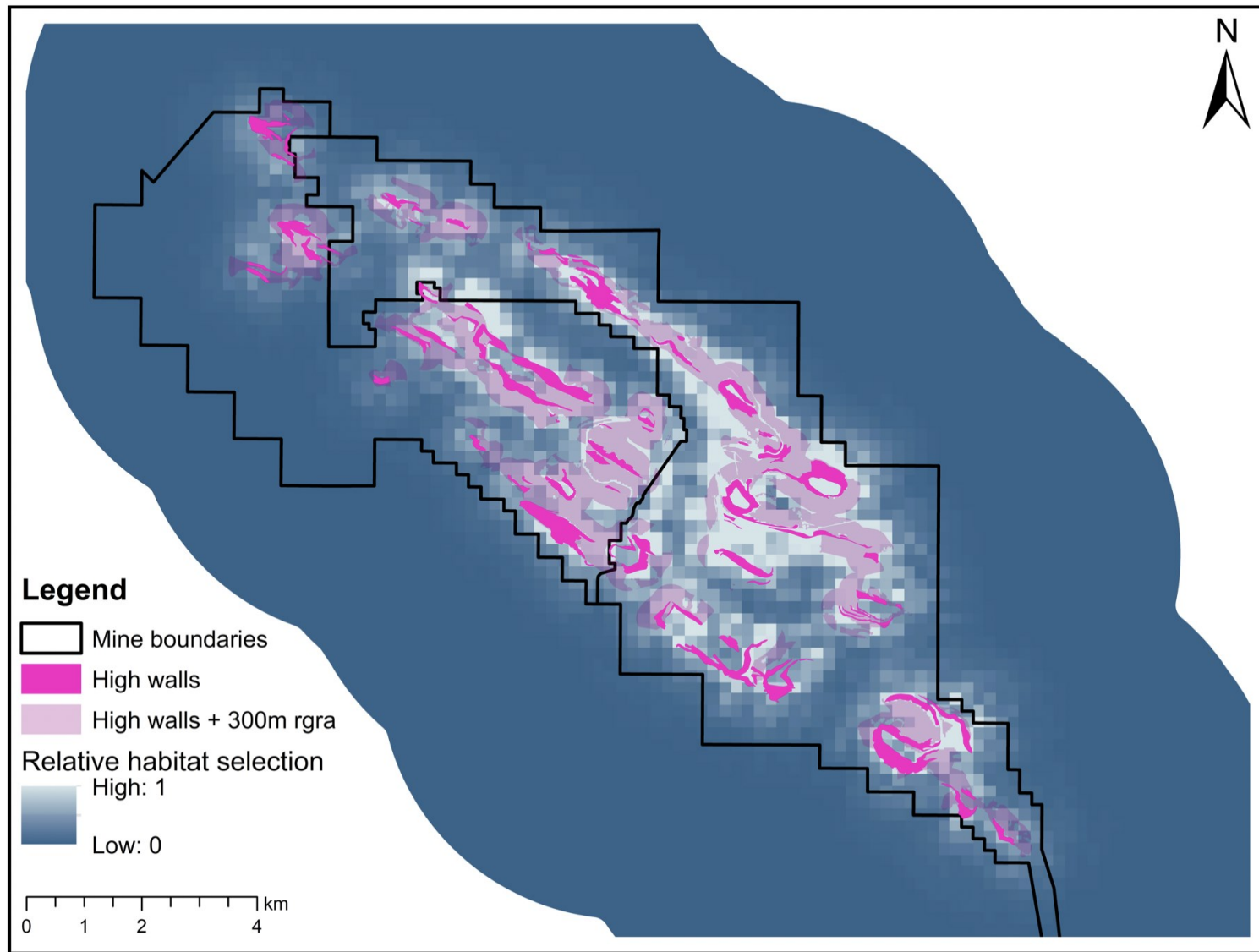
Figure 2.3 Predicted relative habitat selection for bighorn sheep on Gregg River and Luscar mines (outlined in black) in relation to high walls (dark pink). Reclaimed grasslands (rgra) within 300 meters of high walls are symbolized in translucent pink (Smith et al. 1991) (Appendix 2.5). Relative habitat selection is scaled between 0 (low, dark blue) and 1 (high, light blue). Landscape data from 2017 were used to create this map.
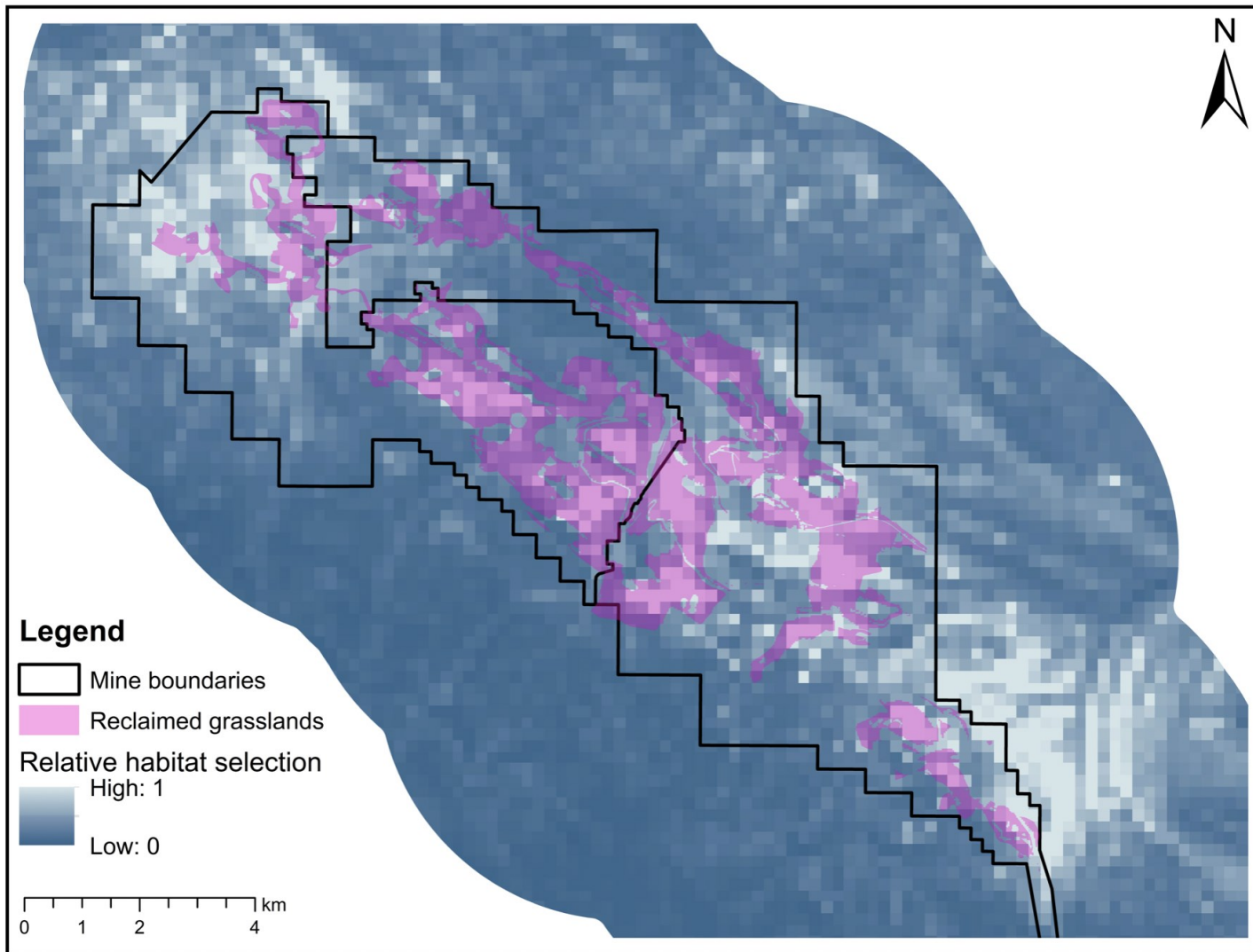
Figure 2.4 Predicted relative habitat selection for elk on Gregg River and Luscar mines (outlined in black) in relation to reclaimed grasslands (translucent pink). Relative habitat selection is scaled between 0 (low, dark blue) and 1 (high, light blue). Landscape data from 2017 were used to create this map.
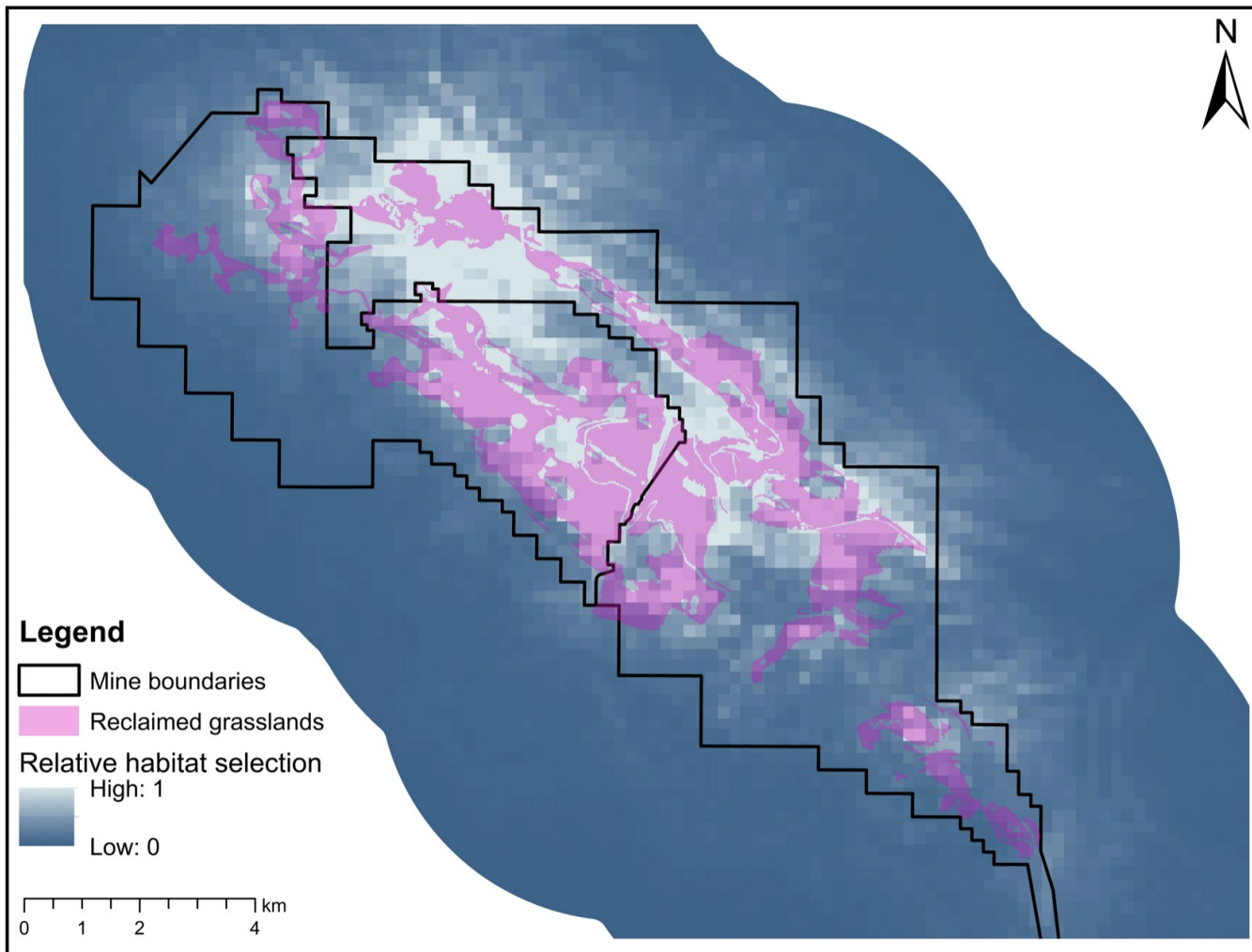
Figure 2.5 Predicted relative habitat selection for mule deer on Gregg River and Luscar mines (outlined in black) in relation to reclaimed grasslands (translucent pink). Relative habitat selection is scaled between 0 (low, dark blue) and 1 (high, light blue). Landscape data from 2017 were used to create this map.
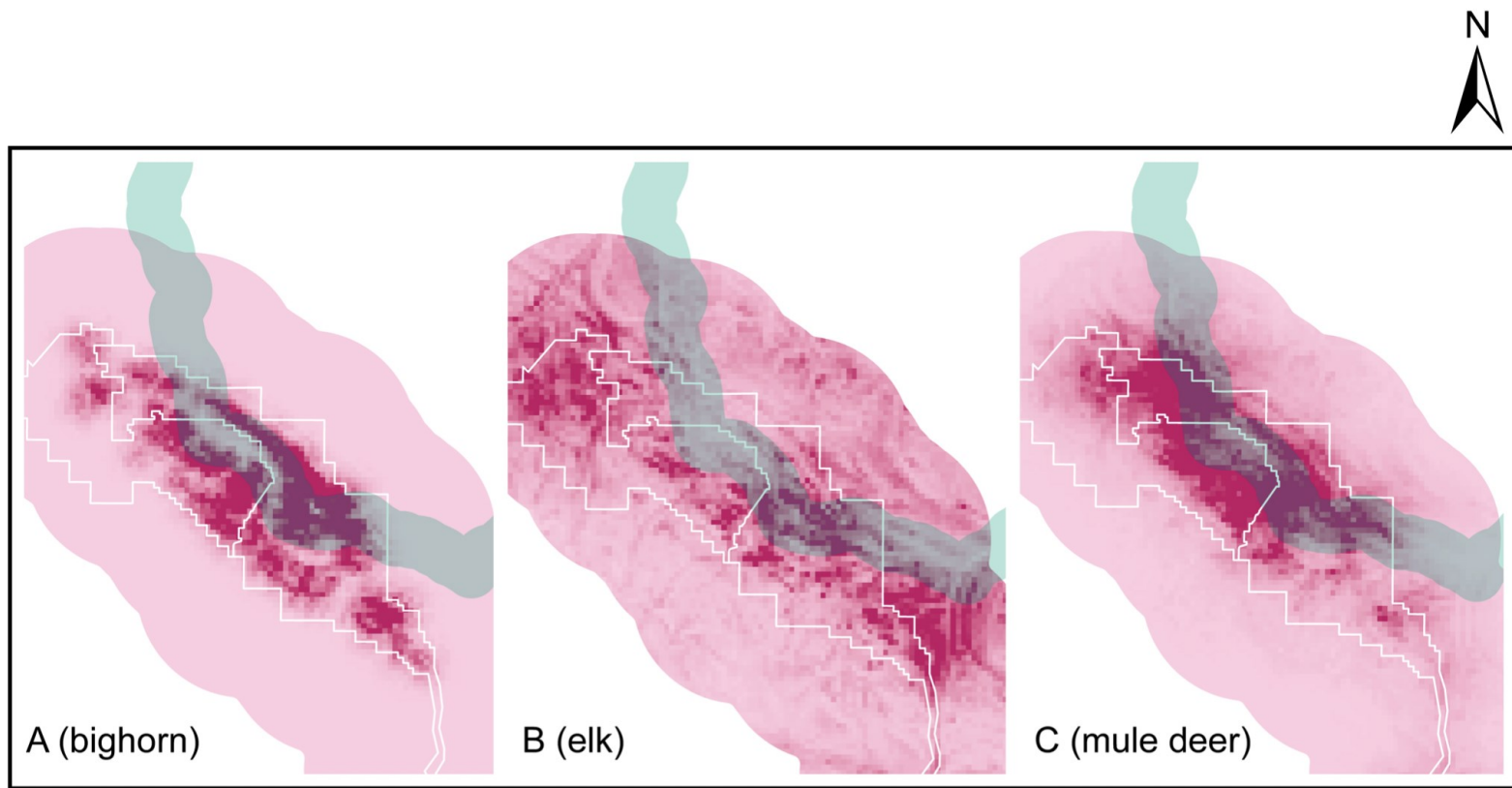
Figure 2.6 Proposed 1-km buffer on either side of Highway 40 South (translucent green), on Gregg River and Luscar mines. Pink maps show predicted relative habitat selection for bighorn sheep (A), elk (B), and mule deer (C) using 2017 landscape data. Dark pink represents high relative habitat selection and light pink represents low relative habitat selection

CHAPTER 3 – LANDSCAPE CONFIGURATION ON RECLAIMED MINES FACILIATES COUGAR PREDATION ON BIGHORN SHEEP

ABSTRACT

Two reclaimed coal mines in west-central Alberta mines host a complex assemblage of large-bodied predator and prey populations, including grizzly bears (*Ursus arctos*), wolves (*Canis lupus*), cougars (*Puma concolor*) and their ungulate prey of bighorn sheep (*Ovis canadensis*), elk (*Cervus elaphus*), and deer (*Odocoileus* spp). Reclamation has been successful at attracting a diverse set of large mammals, but reclamation also might be facilitating increased predation on bighorn sheep. These reclaimed mines have numerous undisturbed forest patches adjacent to open grasslands, which we hypothesized would provide cover for cougars moving through open landscapes. We also hypothesized that cougars would exploit the increased forest edge for hunting bighorn sheep. We captured cougars (*n* = 7) in west-central Alberta and outfitted them with global positioning system (GPS) collars between March 2017 – January 2018. We programmed collars to fix a GPS location every 1.5 hours and employed the cluster method to estimate locations of cougar kills. We visited all accessible clusters (*n* = 455) that were likely to be kills, and if we identified a cougar predation event, we determined species, sex, and age of prey killed. We estimated resource selection functions at the second- and third-order scales to evaluate whether cougars established home ranges and used habitat within home ranges nearer to reclaimed mines. Secondly, we used step selection functions to assess whether landscape configuration and availability of prey influenced cougars as they moved

through the reclaimed mines. Thirdly, we developed a kill occurrence model to evaluate

influence of landscape configuration on the spatial distribution of cougar-made bighorn

sheep kills. Lastly, we assessed diet composition to determine whether cougars

selectively preyed and specialized on bighorn sheep. We found that two cougars selected

reclaimed mines when establishing home ranges. Of these two individuals, one selected

and the other did not select reclaimed mines at the third order. At the third order, mine

and non-mine groups of cougars responded similarly to natural landscape features but

responded differently to anthropogenic features. Non-mine cougars avoided

anthropogenic land and well pads within their home ranges, similar to other cougar

populations in west-central Alberta. At a fine scale, mine cougars selected rocky outcrops,

forest edges, forest, and high relative availability of bighorn sheep availability when on

the reclaimed mines. Cougar predation events on bighorn sheep were closer to forest

edges than randomly expected on the reclaimed mines, based on 17 cougar-made

bighorn sheep kills. These findings support our predictions that forest edges facilitated

predation on bighorn sheep. Lastly, both mine cougars specialized on bighorn sheep, and

selected bighorn sheep disproportionately to their availability (p < 0.05 Fisher's exact

test; by biomass). Ultimately, understanding individual differences in cougar home range

establishment, habitat selection, prey specialization, and prey selection can help to

ensure that cougar and bighorn sheep populations can be managed optimally on the

reclaimed mines. Reclaimed mines may represent an ecological trap for bighorn sheep. If

managing specialist cougars is important to wildlife managers, we recommend that

managers consider configuration of landscapes to relieve predation pressure on bighorn sheep.

INTRODUCTION

Extracting coal through open pit mining alters natural habitats and can result in substantial changes to vegetative (MacCallum and Geist 1992, Holl and Cairns 1994, Wickham et al. 2006, 2013) and topographic structure (Wickham et al. 2013). During mining, forests are logged and overburden is removed, which fragments forests and converts interior forest to edge habitat (Wickham et al. 2006). Blasting and mechanized shoveling exposes coal seams for extraction, which strips the land of its natural topography (Wickham et al. 2013). Reclamation, however, aims to mitigate negative effects of mining on the environment and wildlife by revitalizing ecosystem functioning and returning the landscape to useable habitats. For example, reclaimed mines can successfully support a variety of plant (Strong 2002), bird (Stauffer et al. 2011, Slankard et al. 2018), and mammal species (Yearsley and Samuel 1980, Elliott and McKendrick 1984, Poole et al. 2016, Müller et al. 2017). We focus our research on two neighbouring reclaimed coal mines, Gregg River and Luscar, in west-central Alberta, Canada, where reclamation is legislated. Specifically, Alberta's Environmental Protection and Enhancement Act (Province of Alberta 2000) establishes that mining corporations must consider reclamation as an integral step in the coal extraction process, and must identify premeditated end land use before breaking soil (Government of Alberta 2013). This approach ensures that mined landscapes are returned to an end land use that encompasses the environmental and biological needs of the region. Subsequently, one of

the primary end land uses of reclaimed mines in our study area is habitat for wildlife (Government of Alberta 2013).

Reclaimed mines in our study area provide habitat for complex assemblies of predator and prey populations. For instance, the reclaimed mines host a variety of large terrestrial mammals, including grizzly bears (*Ursus arctos*) (Cristescu et al. 2015a, 2015b, Ladle et al. 2019), wolves (*Canis lupus*) (BWT, unpublished data, Pers. Obs.), cougars (*Puma concolor*), bighorn sheep (*Ovis canadensis*) (MacCallum and Geist 1992), elk (*Cervus elaphus*) (Ch 2), and mule deer (*Odocoileus hemionus*) (Ch 2), that occur sympatrically. Mine reclamation has provided a mosaic of valuable habitat features for ungulates and their predators (MacCallum and Geist 1992). For instance, mining pit high walls were reclaimed to function as escape terrain, which is critical for bighorn sheep to avoid predation (Smith et al. 1991, Andrew et al. 1999, Singer et al. 2000, Bleich et al. 2009). Grasslands were seeded in close proximity to high walls, with mixtures of non-native forage species (i.e. red fescue (*Festuca rubra*), alfalfa (*Medicago sativa*), cicer milkvetch (*Astragalus cicer*), and clover (*Trifolium hybridum*)). Reclaimed grasslands adjacent to high walls are especially valuable resources for pregnant and post-parturient sheep (Berger 1991, Bleich et al. 1997). Additionally, we determined in Chapter 2 that elk and mule deer strongly selected reclaimed grasslands. Research also indicates that grizzly bears were less carnivorous and mainly consumed non-native forbs and graminoids on the reclaimed mines (Cristescu et al. 2015a), which demonstrates that bears also might be attracted to reclaimed grasslands. Lastly, undisturbed forest patches were retained, which provide habitats for predators like cougars and wolves.

Although reclamation has been successful at attracting a diverse set of large mammals, reclamation also could be facilitating increased predation on prey populations. Anecdotal evidence suggests that cougars have been frequenting the reclaimed mines and feeding on bighorn sheep. Moreover, cougars are stalk and ambush predators (Beier et al. 1995, Pelletier et al. 2006) that use forest edges (Beier et al. 1995, Laundre and Hernandez 2003, Laundre and Loxterman 2006, Holmes and Laundre 2006) to approach prey while remaining undetected. The reclaimed mines have numerous undisturbed forest patches scattered throughout the landscape, which increases the linear forest edge available to cougars while hunting. Further, many of these forest patches are adjacent to or completely surrounded by high walls and reclaimed grasslands. In addition to providing preferred hunting habitats in close proximity to prey, forest patches may also provide cover for cougars moving through open landscapes. Therefore, our main objective was to evaluate whether landscape configuration on reclaimed mines influenced cougar movement and predation.

Understanding the influence of landscape configuration on cougar predation might help to ensure viable bighorn sheep populations are conserved. Cougars are generalists and employ variable foraging strategies to consume a range of prey types (Elbroch et al. 2013, Elbroch and Wittmer 2013). However, individual cougars can specialize on particular prey species including feral horses (Turner et al. 1992), beavers (*Castor canadensis*) (Lowrey et al. 2016), domestic sheep (Elbroch and Wittmer 2013), European hares (*Lepus europus*) (Elbroch and Wittmer 2013), mule deer (Cooley et al. 2008), and bighorn sheep (Ross et al. 1997, Blake and Gese 2016b). Cougars can be

effective predators on bighorns (Wehausen 1996, Ross et al. 1997, Festa-Bianchet et al. 2006, Knopff et al. 2010a, Rominger 2018) and this predation can reduce bighorn fitness (Bourbeau-Lemieux et al. 2011) and survival (Johnson et al. 2013, Conner et al. 2018), resulting in detrimental effects, especially for small or isolated bighorn populations. Evidence also supports that cougars select vulnerable targets, such as young (Knopff et al. 2010a, Smith et al. 2014), old (Husseman et al. 2003) and shy individuals (Reale 2003). For example, Harrison and Hebert (1988) found that cougars predominantly killed bighorn rams whereas Ross et al. (1997) found that cougars predominantly selected bighorn lambs. Clearly, there is a potential for cougars on the reclaimed mines to target not only bighorn sheep among all ungulates, but rams and lambs among all bighorn sheep.

In addition to direct predation and specializing, cougars also could contribute to declines in bighorn sheep through apparent competition between sheep and deer (Holl et al. 2004, Johnson et al. 2013). Apparent competition entails increased predation on bighorn sheep because cougar populations are high due to the availability of deer as alternate prey. Lastly, bighorn sheep management is a notoriously challenging issue in ecology (Boyce and Krausman 2017), which warrants further scientific research targeting cougar predation on the reclaimed mines.

We evaluated broad-scale cougar habitat selection to determine how cougars incorporated reclaimed mines into and within their home ranges. We also evaluated fine scale cougar habitat selection on reclaimed mines to assess how residual landscape configuration influenced cougar movement and predation. Because cougar habitats

(Dickson and Beier 2002, Pierce et al. 2004, Morrison et al. 2014, Dellinger et al. 2018) and prey selection (Cooley et al. 2008) can vary depending on season, we compared habitat selection patterns during two periods of the year. Lastly, we compared prey composition to determine whether cougars selected certain species, age classes, and sexes proportional to their availability on reclaimed mines.

Cougar numbers have been increasing in Alberta over recent years (Knopff et al. 2014b), so we expected some individuals to incorporate reclaimed mines into their ranges. Additionally, cougar home ranges are large (i.e. $\geq$ 300 km^2) (Neal et al. 1987, Dickson and Beier 2002) , so we did not expect that individual cougars would establish their home range exclusively on the reclaimed mines. Instead, we anticipated that multiple cougars would integrate portions of the reclaimed mines into their individual home ranges. Further, we expected that if cougars established home ranges nearer to reclaimed mines to exploit residual landscape configuration and abundant ungulate prey, then cougars should select reclaimed mines within their home ranges. However, if cougars selected reclaimed mines at the second order, but not the third order, this would suggest that home range establishment was a product of available territory in the geographical range. At fine scales, we predicted that cougars would select edge habitats (Beier et al. 1995, Laundre and Hernandez 2003, Holmes and Laundre 2006), rugged terrain (Beier 2009, Chetkiewicz and Boyce 2009), riparian areas (Dickson and Beier 2002, Burdett et al. 2010), and areas of abundant prey (Pierce et al. 2004, Laundre et al. 2007, Knopff et al. 2014b) on the reclaimed mines. We also expected that forest patches in proximity to high walls would facilitate increased cougar predation on bighorn sheep.

Lastly, cougars are relatively tolerant of minor human disturbance (Dickson and Beier 2002, Kertson et al. 2011, Knopff et al. 2014a) and are successful at inhabiting novel ecosystems (Beier 2009, Knopff et al. 2014b, 2014a, Moss et al. 2016), so we did not anticipate that cougars would avoid disturbed areas, haul roads, or highways on the reclaimed mines.

METHODS

Study area

We studied cougars in a 9,577 km^2 area in Yellowhead County and Jasper National Park, Alberta, Canada (approximately 53°13'N 117°29'W), which includes montane, subalpine, alpine, and boreal foothills ecoregions (Figure 3.1). We selected this study area because it encompasses two neighbouring reclaimed coal mines, Gregg River and Luscar, allowing us to evaluate cougar habitat and prey selection on reclaimed mines. Habitats between Gregg River and Luscar are contiguous; that is, there are no man-made boundaries between the two mines that limit animal movement. The reclaimed mines are situated east of the Nikanassin Mountain Range, include mostly subalpine ecoregions, and span approximately 90 km^2. Luscar is owned by Teck Resources Ltd. (Teck; Vancouver, Canada), and Gregg River is owned by Westmoreland Coal Company (Westmoreland; Colorado Springs, USA). Gregg River is almost entirely reclaimed (99%) and Luscar is partially reclaimed (60%). Coal is extracted from Cheviot, a third and active mine, where it is transported 8 km north to Luscar via a haul road. Because Cheviot is active, we did not consider it a 'reclaimed mine'. We henceforth refer to Gregg River and Luscar as

'reclaimed mines'. For more detailed descriptions of areas immediately surrounding the reclaimed mines, please see Chapter 2.

Anthropogenic development increases on a west-east gradient in the study area. The western portion of the study area includes Miette Road, Ashlar Ridge, and the Fiddle Mountain Range of Jasper National Park (JNP). The west of the study area has low anthropogenic disturbance, including low road densities. Access to the east border of JNP is primarily through non-motorized and remote backpacking trails. East of the JNP border and west of Highway 40, motorized trails exist but do not receive heavy traffic. East of Highway 40, clear-cut areas, logging roads, well-pads, and well-pad service roads are dense. The study area also includes the hamlets of Brule (population 31) and Cadomin (population 40), as well as the town of Hinton (population 9,882). Motorized recreation is common throughout the study area, with density of all-terrain vehicle (ATV), snowmobile, and 4x4 trail networks increasing on a west-east gradient, similar to industrial activities.

The current land cover in the study area is approximately 60% forest, 19% anthropogenic (i.e. human-modified) land, 11% barren/rocky, 5% shrub, 3% grassland, and 2% water. Land cover is continually changing, simultaneous with industrial development. Forests dominate the landscape and are mainly conifer, including lodgepole pine (*Pinus contorta*), Englemann spruce (*Picea engelmannii*), and Douglas fir (*Pseudotsuga menziesii*). Black spruce (*P. mariana*), white spruce (*P. glauca*), and tamarack (*Larix laricina*) are common at lower elevations. Deciduous species, such as trembling aspen (*Populus tremuloides*) and balsam poplar (*P. balsamifera*), are also found at lower elevations in mixed conifer-deciduous stands. Grasslands and shrubs are

distributed patchily throughout the study area, and common shrub species include alder (*Alnus* spp.), willow (*Salix* spp.), and common juniper (*Juniperus communis*). Elevation in the study area varies from 908 – 3,080 m and high elevations consist primarily of barren ground, rocks, snow, ice, as well as alpine meadows. The study area is also characterized by Chinook winds that descend to bring warmer temperatures to the region (Natural Regions Committee 2006). Lastly, a variety of prey species are available to cougars, including white-tailed deer (*Odocoileus virginianus*), mule deer (*Odocoileus hemionus*), elk (*Cervus elaphus*), moose (*Alces alces*), and bighorn sheep (*Ovis canadensis*), among other prey (Knopff et al. 2010a).

Capture and monitoring

Cougars were tracked and treed with the assistance of a houndsman with trained hounds (Hornocker 1970). We used a Daninject rifle and darts to remotely inject chemical immobilization drugs, Telazol® (tiletamine HCl and zolazepam HCl) in combination with medetomidine HCl at a dosage of approximately 2.5 mg/kg Telazol and 0.075 mg/kg medetomidine, intramuscularly. Upon completion of processing, we reversed medetomidine with atipamezole HCl at a dosage of 0.3 mg/kg, and released the cougars. We captured 7 cougars, according to procedures approved by the University of Alberta Animal Care and Use Committee (AUP00002113), between March 2017 and January 2018. Once immobilized, we weighed, measured, sexed, and aged cougars. We estimated age using a combination of tooth colour and wear (Shaw et al. 2007), gingival recession

(Laundre et al. 2000), and pelage spotting (Shaw et al. 2007). We fitted cougars with

Lotek Iridium TrackM 2D GPS collars (Lotek Wireless, Newmarket, Ontario, Canada).

We programmed GPS collars to fix a location every 1.5 hours, 24 hours/day.

Between March 2017 and October 2018, we downloaded GPS data from collars every 2

weeks and used a rule-based Python[TM] (Python Software Foundation, Wilmington,

Delaware, USA) algorithm, developed by Knopff et al. (2009), to predict clusters of GPS

locations. Following Anderson and Lindzey (2003), Knopff et al. (2009) defined a cluster

as ≥ 2 GPS points located within 200 m of each other, within a 6-day temporal window.

Clusters could persist past a 6-day temporal window, given that the time difference

between the first and last GPS point was ≤ 6 days. To apply the Python[TM] algorithm, we

rarified our data to 3-hour intervals, to match the frequency of GPS fixes for which the

algorithm was developed (Knopff et al. 2009). We obtained an output from the Python[TM]

algorithm, which was the geometric centroid for each cluster, among other quantitative

measurements for each cluster (Knopff et al. 2009). Knopff et al. (2009) further

developed a logistic regression model to discriminate kills from non-kills, which we used

to predict the likelihood of a particular cluster being a kill. Knopff et al. (2009) emphasize

the requirement for an appropriate probability cutoff to be enforced to distinguish kills

from non-kills. We programmed geometric centroids of clusters into a handheld GPS and

visited all accessible clusters ≥ 30% likelihood of being a kill. We selected this cutoff to

avoid missing true positives, while still improving field efficiency and eliminating some

non-kill clusters (false positives). Further, the cluster method underestimates small prey

(i.e. prey under < 8 kg) (Anderson and Lindzey 2003, Knopff et al. 2009, Bacon et al. 2011),

which would create a bias against neonate ungulate kills. Because detecting juvenile

ungulates was integral to our research questions, we visited all accessible clusters ≥ 10%

likelihood of being a kill during spring and summer months to reduce missing neonate

kills. We reduced visitation to clusters ≥ 10% and < 30% likelihood of being a kill during

winter and fall to improve field efficiency.

We usually visited cluster centroids within 8 days ($\bar{x}$ = 16, range = 1 – 291 days) of

the cougar's last GPS location, in field crews of ≥ 2 people. If prey remains were found at

the cluster centroid, we assigned predation when prey showed obvious signs of being

killed by a cougar (i.e. localized damage to neck/throat from biting, ambush sequence

visible in snow tracks). If this information was not available, we assigned predation when

the age of remains were consistent with the dates in which the cougar made the cluster,

and if the remains were consistent with cougar feeding behaviour. We considered drag

marks, cougar scrapes, multiple cougar scats, sheared hair or large hair mat, removal of

intestines or stomach, prey cached with moss or twigs, and feeding entry point under the

ribs as signs of cougar feeding behaviour. When prey remains were not found

immediately at a cluster centroid, we systemically searched within a 20-m radius

surrounding each cluster centroid. If prey remains still were not located, we walked 10

transect lines either east-west or west-east to form a 100 x 100-m square, with the

cluster centroid at the center of the square. If we found prey remains within the 100 x

100-m square that were consistent with cougar feeding behaviour, or if the age of

remains were consistent with dates of cluster creation, we assigned predation. If we

assigned predation to a cluster, we also recorded the spatial location of prey remains, as

they often differed from the estimated cluster centroid. We assigned scavenging when prey were clearly not killed by a cougar (i.e. killed by vehicle, human, or wolves) or when age of remains were inconsistent with the dates of cluster creation. We assigned non-kill when we could not locate prey remains.

We examined prey remains to determine species, age, and sex, where possible. To identify species in the field, we used skeletal, anatomical, and pelage characteristics (Elbroch 2006). If we could not identify species in the field, we collected hair and tissue samples, when possible, to send for DNA analysis (Molecular Biology Services Unit, University of Alberta). Species were identified through DNA barcoding, based on the CO1 mitochondrial gene. Because some closely related species share CO1 sequences, DNA analysis could not differentiate between white-tailed and mule deer, or coyotes, wolves, and foxes. Thus, we collapsed both deer species into 'deer spp.' and coyotes, wolves, and foxes into '*Canis* spp.' We attempted to determine age and sex for all ungulate prey. When a skull and/or mandible was present, we used horn anatomy, tooth eruption sequences, and tooth wear and colour to differentiate between adults and juveniles (Feldhamer et al. 2003, Elbroch 2006). We considered prey $\leq$ 2 years of age to be juvenile and prey > 2 years of age to be adult. We used horn morphology, presence of antlers, and reproductive organs to differentiate between males and females. Due to difficulties discerning age and sex for non-ungulate prey, we did not attempt to differentiate between ages and sexes for these prey.

Landscape features

We created a database of biologically relevant landscape features (Table 3.1) to be used as covariates in cougar RSFs, SSFs, and kill-occurrence models. We obtained data from multiple sources, including Foothills Research Institute (FRI; Hinton, Alberta), Teck, Bighorn Wildlife Technologies Ltd. (BWT; Hinton, Alberta), GeoBase Series (Natural Resources Canada), Alberta Biodiversity Monitoring Institute (ABMI; Edmonton, Alberta), and AltaLIS (Calgary, Alberta). We used updated and verified spatial data created for Chapter 2 to ensure changes due to mining and reclamation were accounted for.

We obtained annual land cover classification data for 2017 (grain: 30 x 30 m), which categorized our study area into six categories: forest, shrub, grassland, anthropogenic, barren/rocky, and water (McDermid 2005). Anthropogenic land included a variety of man-made structures such as roads, highways, buildings, parking lots, well pads, clear-cut forests, and construction sites. On reclaimed mines, anthropogenic land included haul roads, mine roads, and disturbed areas. Specifically, disturbed areas were areas of either active or inactive coal mining. For forest, shrub, grassland, anthropogenic, and barren/rocky classes, we created a dummy variable where we coded the class of interest with '1' and all other land cover classes with '0'. We defined forest edge as the linear boundary between forest land cover and another non-forest land cover classification, to represent edge habitat that cougars commonly exploit for hunting (Beier et al. 1995, Laundre and Hernandez 2003, Holmes and Laundre 2006). Lakes and rivers represented large permanent water features, while streams represented smaller and potentially ephemeral water features. We further collapsed all water features into one

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water category, which combined lakes, rivers, and streams. We included two covariates for specific anthropogenic features: divided highways and well pads. Highway 16 was the only divided highway and thus entirely represented divided highways. We attempted to divide anthropogenic features into multiple road classes but high correlation among covariates prevented this (Appendix 3.1).

For each landscape feature, we created a 30 x 30-m raster layer to represent the Euclidean distance from the center of each raster cell to the nearest point of each respective feature. Using this method, we calculated distance to nearest forest, shrub, grassland, barren/rocky outcrop, anthropogenic land, divided highway, well pad, forest edge, lake or river, stream, and water. We also calculated density of forest edge (m/m^2) with a moving window of 300 m, 500 m, 1 km, 2 km, and 4 km to represent five different scales at which cougars may respond to fragmentation of forests.

We calculated a terrain ruggedness index (TRI; (Riley et al. 1999)) and relative topographic position (RTP; (Jenness 2006)) using a digital elevation model (DEM) in ArcGIS (Environmental Systems Research Institute 2017). High RTP represented the tops of hills, mountains, and cliffs, whereas low RTP represented riparian areas, valleys, and ravines. We also calculated aspect as a continuous variable in degrees, and re-categorized aspect into four categories: north (315 − 44°), east (45 − 134°), south (135° − 224°), and west (225° − 314°).

Using ArcGIS, we applied a 500 m buffer to the boundaries of Gregg River and Luscar to spatially represent 'reclaimed mines'. The resulting area of the buffered

reclaimed mines was 130 km^2. All future references to 'reclaimed mines' refers to this spatial designation. We chose this buffer distance to represent how ungulate prey species on reclaimed mines may disperse to use habitat surrounding mine boundaries, but not be directly within mine boundaries. Further, we located all undisturbed patches of forest completely surrounded by non-forest land cover classes on reclaimed mines and titled these 'tree patches'. Although edges of tree patches appear in the forest edges covariate, and tree patches themselves appear in the forest covariate, we wanted to represent how tree patches may have innate qualities and serve as cover for cougars when travelling through open habitat. We then created a 30 x 30-m raster layer to represent the Euclidean distance to reclaimed mines and nearest tree patch. Lastly, we used a 30 x 30-m raster layer for distance to disturbed areas, which was developed for Chapter 2.

Second- and third-order resource selection

We employed a used vs. available design (Manly et al. 2002, Johnson et al. 2006) to create exponential RSFs, fitted with logistic regression:

$$RSF(x) = \exp(\beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \cdots + \beta_n x_n) \qquad \text{(Eq 3.1)}$$

where β_i represents the selection coefficient for covariate x_i, for n total covariates. We assessed habitat selection on two scales. We used second order RSFs to evaluate whether cougars situated home ranges closer to reclaimed mines (Johnson 1980). We used third order RSFs to evaluate whether cougars selected habitats within their home ranges disproportionately to their availability (Johnson 1980). We also used third order

RSFs to determine if cougars that used reclaimed mines responded to landscape features differently than cougars that did not use reclaimed mines. We modelled RSFs for two seasons to assess whether seasonality influenced whether cougars integrated reclaimed mines into their home ranges. We binned observations between November – April inclusive into the snow season and observations between May – October inclusive into the snow-free season. We did not monitor two cougars, M2 and F2, during the snow-free season and thus, we could not estimate RSFs for these individuals during this time.

We used GPS locations collected at 1.5-hour intervals to represent 'used' locations. Because cougars remain close to kills until fully consumed (Knopff et al. 2009), we reduced GPS locations from within 100 m of confirmed kills to one GPS point at the centroid of each cluster. This step helped us to avoid estimating RSFs biased towards where cougars handle prey. We defined 'confirmed kills' as any predation event that we verified with field visits or any non-visited cluster with > 80% likelihood of being a kill. We included non-visited clusters > 80% likelihood of being a kill in 'confirmed kills' to account for the fact that some clusters were inaccessible and thus could not be verified with field visits. All clusters > 80% likelihood of being a kill that we could visit were confirmed to be kills, and we chose this cutoff of 80% to conservatively add inaccessible non-visited kills to 'confirmed kills'.

For second order RSFs, we randomly selected available locations from within the study area at a ratio of 1:4 used to available points, per season and individual. For third order RSFs, we first delineated individual cougar home ranges, per season, as 95% minimum convex polygons (MCPs) using the *adehabitatHR* package in R (R Core Team

2018). We then calculated the density of used points per seasonal home range and randomly selected available points within each seasonal home range at equal densities, per individual. Lastly, we assigned landscape features to used and available points for all individuals, both seasons, and both RSF scales in ArcGIS.

For third order RSFs, we divided individuals into 'mine' and 'non-mine' groups of cougars based on whether cougars' home ranges intersected reclaimed mines. Mine cougars included two individuals, M1 and F2, a 4 year old male and 8 year old female, respectively. Due to age and sex differences, we did not collapse both individuals into one mine group and instead modelled mine cougars individually. Conversely, due to similarities in age and sex, we aggregated F1, F3, F4, and F5 (all female cougars approximately 4-6 years of age) into the non-mine group. We modelled each non-mine cougar individually, then estimated a non-mine group model using mixed-effects models with individual as a random effect. We excluded M2 from the non-mine group due to sex and age differences (M2 was a subadult male).

We designed a candidate set of models *a priori* to evaluate alternative hypotheses of cougar habitat selection in the study area (Table 3.2). Before fitting the candidate set, we fit single-term RSF models for each habitat selection scale and season to compare the best forest edge density scale to use. We compared single-term models with Akaike's Information Criterion corrected for small sample size (AICc) and selected the forest edge density scale with the lowest AICc. We determined that forest edge density calculated with a moving window of 4 km and 2 km best explained variation in data for second and third order RSFs, respectively. Because the non-mine home ranges did not overlap

reclaimed mines, we did not include distance to reclaimed mines for non-mine cougars (Appendix 3.2).

For second order RSFs, third order RSFs, SSFs, and kill occurrence models, we standardized ($\bar{x}$ = 0, SD = 1) all continuous covariates, then tested them for multicollinearity using Pearson's correlation coefficient. We included only biologically relevant and non-highly-correlated ($|r| < 0.67$, determined *a priori*) covariates in our candidate sets of models. We did not consider interaction terms or non-linearities in covariates. We calculated AICc and Akaike weights (Burnham and Anderson 2002) for each model in the candidate set. We considered models with with ΔAICc < 2.0 as competitive (Burnham and Anderson 2002) and used AICc to select the most parsimonious top model (i.e. competitive model with fewest parameters). We validated individual top second and third order RSFs with 5-fold cross validation (Boyce et al. 2002). We divided our data into 5 folds and tested the relationship between area-adjusted frequency of predicted 'used' observation and 10 RSF bin ranks, for each of the 5 folds. We used the mean Spearman's rank correlation coefficient coefficient ($\bar{r}_s$) averaged across all folds.

Step selection on reclaimed mines

To determine fine-scale habitat selection on reclaimed mines, we estimated a step selection function (SSF) (Fortin et al. 2005) for each individual of the mine cougar group. Here, a SSF takes the structural form of an RSF (Manly et al. 2002):

$$SSF(x) = \exp(\beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \cdots + \beta_n x_n) \qquad \text{(Eq 3.2)}$$

where β_i is a coefficient estimated with conditional logistic regression for covariate x_i, for n total covariates. We converted GPS locations acquired at 1.5-hour frequency into steps, or Euclidean distances between the start and end points of two GPS relocations, using the *amt* package in R (Signer et al. 2019). We removed all steps that ended outside reclaimed mines. We also removed steps that ended within a 100-m radius of confirmed kills. We removed steps made at kills because we were interested in modelling fine-scale habitat selection as cougars moved through and hunted on reclaimed mines and did not wish to bias SSFs towards movements made by cougars while handling prey. In doing so, we effectively removed short steps from our SSF dataset, precluding the need to include step length as a covariate in SSFs. Henceforth, when we refer to 'steps', we refer to steps on reclaimed mines, excluding steps made at confirmed kills. We split steps into those made in the snow and snow-free season.

We used the *amt* package in R (Signer et al. 2019) to define the distribution of turning angles and step lengths for each individual and seasonal dataset. We then matched 10 available steps to each used step, sampling turning angles and step lengths randomly from a von Mises and gamma distribution, respectively. Available steps shared the same starting point as used steps but ended in different spatial locations. Because we were interested in understanding how landscape features on reclaimed mines influenced how cougars moved through the landscape, we extracted landscape features at the end of each used and available step (Signer et al. 2019), using ArcGIS. Lastly, we used conditional logistic regression to fit SSFs for each individual and season.

We designed a candidate set of models *a priori* to evaluate alternative hypotheses of cougar fine scale habitat selection on reclaimed mines (Table 3.3). Prior to modelling, we determined that forest-edge density calculated with a moving window of 300 m explained more variation for SSFs. We developed a base model, which included non-mine specific covariates that were important to second- and third-order habitat selection. We developed seven models, built off the base model, that included combinations of mine specific and non-mine specific covariates. We designed these models to evaluate whether cougars selected tree patches and avoided anthropogenic areas (i.e. haul roads, disturbed areas, highways) on reclaimed mines. Lastly, we included bighorn sheep, elk, and mule deer availability to evaluate whether cougars selected habitats on reclaimed mines based on prey availability. We acquired availability data for these ungulates from Chapter 2, where we used exponential logistic regression to predict relative bighorn sheep, elk, and mule deer habitat selection on reclaimed mines, given 2017 landscape data. We did not have access to a metric of white-tailed deer availability due to the small sample size on reclaimed mines (41 groups between 2002-2017; BWT, unpublished data). Thus, we assumed that white-tailed deer presence was negligible to cougars when selecting habitat on reclaimed mines.

Kill occurrence on reclaimed mines

We evaluated whether landscape configuration influenced the relative probability of a cougar-made bighorn kill by comparing locations of bighorn sheep kills with random available locations, on reclaimed mines (Manly et al. 2002, Johnson et al. 2006). We

defined bighorn sheep kills based on confirmed kills that we verified with field visits. We

pooled bighorn sheep kills between the two mine cougars and did not divide between

season to increase sample size. For each bighorn sheep kill on reclaimed mines (n = 17),

we randomly selected 20 available locations from within reclaimed mines in a paired

observed-random design with strata. As such, we modelled kill occurrence using

conditional logistic regression to determine the probability of predation occurring at

bighorn sheep kill locations relative to 20 control locations, similar to methods applied by

Kauffman et al. (2007). We extracted values for landscape covariates at each available

location using ArcGIS.

Cougars may stalk, ambush, kill, consume, and cache prey each in a spatially

distinct location, so we assumed that spatial locations of prey remains (as determined by

field visits) represented cache sites. We buffered all verified cache sites by 95 m because

this distance represents a 95% chance of containing the kill site (Blake and Gese 2016a).

We used ArcGIS to calculate the mean of each landscape covariate within buffered cache

sites. Buffered cache sites represented 'used' locations for kill-occurrence models. We

acknowledge that we could have missed cougar kills during monitoring but note that

unvisited clusters were random in relation to landscape features. Thereby, we believe

that modelling kill occurrence on reclaimed mines is representative.

The spatial distribution of predator and prey influence where cougar-killed

bighorns are found (Kauffman et al. 2007). As such, we accounted for prey distribution by

using relative bighorn sheep availability, developed in Chapter 2. Because we modelled

kill occurrence pooled over seasons, we used annual bighorn availability rather than

seasonal (Ch 2). To account for cougar distribution, we pooled GPS locations from the mine cougars on reclaimed mines (n = 2,835) and constructed a cougar utilization distribution (UD) using a 95% kernel estimation in the *adehabitatHR* package in R. We selected a smoothing factor of 650 m because it best represented GPS relocations, with adequate variation between frequently and infrequently used areas. We excluded GPS locations from confirmed kills when creating the UD to reduce spatial dependence between kill locations and cougar distribution. Lastly, we calculated a relative cougar UD by scaling UD estimates between 0 – 1.

We designed a candidate set of models *a priori* to evaluate alternative hypotheses of kill occurrence on reclaimed mines (Table 3.4). We included two null models to evaluate whether distribution of cougar-killed bighorns was a product of bighorn sheep distribution, or a product of cougar distribution. Following Kauffman et al. (2007), we then built an encounter model to evaluate whether spatial distribution of kills was simply related to where cougars encountered bighorn sheep (i.e. cougars killing prey in proportion to distribution of prey). Lastly, we created eight alternate models, which we built off the encounter model. These models took into account the distributions of cougars and bighorn sheep, while evaluating the influence of landscape configuration and mining features on kill occurrence. We considered forest edge and RTP because cougars might have greater kill success near forest edges and at low RTPs (Dickson and Beier 2002). Distance to tree patch was highly correlated with distance to high wall, so we could only include distance to tree patch in the candidate set (Appendix 3.5). We chose tree patches because they represent forest cover in close proximity to high walls and

grasslands. We considered disturbed areas and TRI because bighorn sheep might reduce

cougar kill success by using human refuge (Hebblewhite and Merrill 2007, Jansen et al.

2009, Shannon et al. 2014) and rugged terrain (Lingle 2002, Lingle et al. 2008). Due to

high correlation between distance to disturbed area and haul road, we used only

disturbed areas in the candidate set.

Prey composition on reclaimed mines

To compare prey composition on and off reclaimed mines, we calculated percent

frequency and percent biomass of prey killed on and off reclaimed mines, using only kills

where species was known (n = 183), pooled across all cougars (n = 7). We considered all

prey species for prey composition analyses. We calculated biomass (Appendix 3.3), then

collapsed prey into broader categories for percent frequency and biomass calculations.

We collapsed elk and moose into 'large ungulates'. We collapsed lynx (*Lynx canadensis*),

snowshoe hares (*Lepus americanus*), and red squirrels (*Tamiasciurus hudsonicus*) into

'other'. We defined specialization based on the prey item that comprises the majority of

an individual's diet (Elbroch et al. 2013) and assessed specialization for non-mine and

mine cougars.

Selection is the consumption of prey disproportionately to its availability (Estes et

al. 2003, Knopff and Boyce 2007, Elbroch and Wittmer 2013). To assess whether cougars

selected ungulate prey relative to availability on reclaimed mines, we compared

proportion of observed ungulate kills per species and age-sex class with proportion

available per species and age-sex class. We acquired data on availability of ungulate

species and age-sex classes from BWT (BWT, unpublished data; Appendix 3.4). For each of the four surveys conducted in 2017, we summed the total number of bighorn sheep, elk, and mule deer observed, then calculated the proportional availability of each species based on frequency and biomass. We averaged proportions from each survey to acquire an estimated annual proportional availability per species. Similarly, we summed the total number of adult female, adult male, and juvenile bighorn sheep per survey, then calculated the proportional availability of each age-sex class, based on frequency and biomass. We averaged proportions from each survey to acquire an estimate annual proportional availability per age-sex class. We did not calculate proportional availability of elk and mule deer age-sex classes due to the small number of these species killed on reclaimed mines. For observed kills, we summed only kills where species and age-sex class were known, which limited sample sizes. Lastly, we compared proportion killed and proportion available per species and age-sex classes, by both frequency and biomass, using Fisher's exact test of independence. We chose a Fisher's exact test of independence because it can handle small counts in a contingency table. One of the assumptions of a Fisher's exact test of independence is that observed and expected outcomes are conditioned (i.e. totals are fixed) (McDonald 2015). In our case, both observed and expected outcomes are unconditioned. When this assumption is not met, the Fisher's test is no longer 'exact' but can still be used because it provides a conservative evaluation of the null hypothesis (McDonald 2015).

RESULTS

Second-order resource selection

We collected 752 − 5,546 ($\bar{x}$ = 3,989) GPS locations per cougar and GPS collars remained

on cougars for 48 − 357 days ($\bar{x}$ = 256). On average, collars had 97% fix success. The

average number of used points was 1,161 (range = 405 − 1,437) during the snow season,

and 1,671 (range = 1,281 − 2,051) during the snow-free season, per individual. The

average number of available points was 4,644 (range = 1,620 − 7,012) during the snow

season, and 6,686 (range = 5,124 − 8,204) during the snow-free season, per individual.

Individual second-order habitat selection varied in the snow season, with 6 of 19

candidate models selected as a top model for at least one cougar (Table 3.2). Only 4 of 19

candidate models were selected in the snow-free season, but two cougars were not

monitored during this time and therefore did not contribute a model (Table 3.2). Because

the domain of availability for second-order RSFs was the same for all cougars, we

compared strength of selection for the covariates. We validated second-order models

with 5-fold cross validation finding strong predictive ability with mean $\bar{r}_s > 0.90$.

Mine and non-mine cougars responded similarly to anthropogenic features, but

responded differently to certain natural features. Specifically, both groups selected

habitats nearer to anthropogenic land, highways, and well pads, in both seasons (Table

3.5). Selection for anthropogenic land and highways was strongest for the non-mine

cougars in both seasons, but selection for well pads was highest for F2 in the snow

season (Table 3.5). Both groups responded similarly to northerly and southerly aspects,

forests, terrain ruggedness, and streams. For instance, both groups avoided northerly

and selected southerly aspects, compared to easterly aspects, for both seasons. Further,

both groups selected habitat nearer to forests in both seasons; this selection was

strongest for M1 in the snow and snow-free seasons ($\beta_{\text{M1 snow}}$= -1.485, SE =

0.147; $\beta_{\text{M1 snow-free}}$= -0.981, SE = 0.108; Table 3.5). Both groups selected more rugged

terrain, and both groups (with the exception of M1 in the snow season) selected habitat

nearer to streams (Table 3.5). The mine cougars did not exhibit strong selection for high

or low relative topographic positions. However, the non-mine cougars selected higher

RTPs during the snow season ($\beta_{\text{nm snow}}$= 0.204, SE = 0.023; Table 3.5). Grassland was not

included by the mine cougars in the snow season, but when it did appear in a top model

for either group, cougars selected habitats nearer to grasslands (Table 3.5).

Selection for barren/rocky outcrops, forest edge, lakes or rivers, and density of

forest edge differed between the mine and non-mine cougars, and within the mine

cougars. During the snow season, M1 and F2 both selected habitat closer to forest edges

and lake or rivers, whereas the non-mine cougars selected habitat farther from forest

edges and lakes or rivers (Table 3.5). Opposite to F2 and the non-mine cougars, M1

selected habitats closer to barren/rocky outcrops and at lower densities of forest edge in

the snow and snow-free seasons (Table 3.5). Selection for reclaimed mines differed

between individuals. Although both individuals of the mine group selected habitat closer

to reclaimed mines ($\beta_{\text{M1 snow}}$ = -2.533, SE = 0.093; $\beta_{\text{M1 snow-free}}$ = -3.335, SE = 0.103;

$\beta_{\text{F2 snow}}$ = -12.358, SE = 0.764; Table 3.5), strength of selection for reclaimed mines was

approximately $3.7 - 4.8$x stronger for F2 than M1.

Lastly, habitat selection at the second order varied between individuals for the mine group of cougars. During the snow season, M1 selected lower density of forest edge, and selected higher density of forest edge during the snow-free season. Further, M1 selected habitats closer to shrubs, streams, and grasslands during the snow-free season ($\beta_{dishrub}$ = -0.072, SE = 0.053; $\beta_{distream}$ = -0.232, SE = 0.067; $\beta_{digrassland}$ = -0.154, SE = 0.065; Table 3.5). These covariates were not included in M1's top model during the snow season.

Third-order resource selection

Mean MCP area was 328 km^2 (range = 129 – 1,257 km^2) during the snow season and 421 km^2 (range = 123 – 1,303 km^2) during the snow-free season, per individual. Average density of used points for the snow season was 5.8 pts/km^2 (range = 1.1 – 9.2 pts/km^2) and 7.4 pts/km^2 for the snow-free season (range = 1.4 – 10.3 pts/km^2). Similar to second-order habitat selection, third-order habitat selection varied by cougar group in the snow and snow-free seasons. For example, 6 of 19 and 4 of 19 candidate models were selected as a top model for at least one cougar, in the snow and snow-free seasons, respectively (Table 3.2). We validated third-order models with 5-fold cross validation and all mean $\bar{r}_s >$ 0.90, with the exception of F3 ($\bar{r}_s$ = 0.83) and F5 ($\bar{r}_s$ = 0.87) during the snow-free season.

Cougars in both groups selected southerly aspects and avoided northerly aspects, as compared to easterly aspects, within home ranges (Table 3.6). Further, both groups selected more rugged terrain and this selection was consistently stronger in the snow season than snow-free season for M1 and non-mine cougars ($\beta_{tri\ snow}$= 0.669 to 0.824;

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$\beta_{\text{tri snow-free}}$ = 0.071 to 0.220; Table 3.6). Both groups selected habitats nearer to

barren/rocky outcrops, forests, and forest edges ($\beta_{\text{dibarrenrocky}}$ = -0.079 to -0.561;

β_{diforest} = -0.045 to -0.508; β_{diedge} = -0.147 to -0.722; Table 3.6), compared to available

habitats within home ranges. Additionally, both groups selected habitats with lower

density of forest edge ($\beta_{\text{dstyedge2km}}$ = -0.124 to -0.208; Table 3.6). With the exception of

the non-mine cougars in the snow-free season, both groups selected lakes or rivers

within home ranges ($\beta_{\text{dilakeriv}}$ = -0.135 to -0.760; Table 3.6). Distance to stream was not

included in the top model for M1 during the snow season, nor was it included in the top

model for the non-mine cougars in the snow-free season. When it did appear in a top

model, cougars selected habitats nearer to streams within their home ranges (β_{distream} =

-0.026 to 0.169; Table 3.6). Lastly, when grassland appeared in two top models, cougars

selected habitats farther from grasslands within home ranges (Table 3.6).

Selection for anthropogenic land and well pads differed between mine and non-

mine cougars, and within the mine group of cougars. M1 selected habitat closer to

anthropogenic land in both seasons ($\beta_{\text{dianthro snow}}$ = -0.209, SE = 0.042;

$\beta_{\text{dianthro snow-free}}$ = -0.128, SE = 0.030), whereas the non-mine cougars and F2 selected

habitat farther from anthropogenic land in the snow season ($\beta_{\text{nm dianthro}}$ = 0.395, SE =

0.070; $\beta_{\text{F2 dianthro}}$ = 0.316, SE = 0.071; Table 3.6). Secondly, F2 selected habitat closer to

well pads in the snow season ($\beta_{\text{diwellpad snow}}$ = -0.261, SE = 0.065), whereas M1 and the

non-mine cougars selected habitat farther from well pads in both seasons (Table 3.6).

Both groups selected highways, within home ranges (β_{dihwy} = -0.413 to -1.063; Table 3.6).

Lastly, M1 and the non-mine cougars selected habitat closer to shrubs in the snow-free

season ($\beta_{\text{M1 dishrub}}$ = -0.141, SE = 0.053; $\beta_{\text{nm dishrub}}$ = -0.060, SE = 0.023; Table 3.6), but

the non-mine cougars and F2 selected habitat farther from shrubs in the snow season

($\beta_{\text{nm dishrub}}$ = 0.060, SE = 0.029; $\beta_{\text{F2 dishrub}}$ = 0.451, SE = 0.077; Table 3.6).

Third-order selection for reclaimed mines differed between individuals within the

mine group of cougars. M1 selected reclaimed mines, relative to random available

locations within seasonal home ranges ($\beta_{\text{dimines snow}}$ = -0.347, SE = 0.082;

$\beta_{\text{dimines snow-free}}$ = -0.591, SE = 0.068; Table 3.6), and F2 avoided reclaimed mines,

within her snow-season home range ($\beta_{\text{dimines snow}}$ = 2.457, SE = 0.459; Table 3.6).

Step selection on reclaimed mines

During the snow season, M1 and F2 made 496 and 741 steps, respectively. During the

snow-free season, M1 made 526 steps. Mean step lengths for M1 and F2 during the

snow season were 395 and 195 m, respectively. The mean step length for M1 during the

snow-free season was 415 m. Fine-scale step selection on reclaimed mines varied by

individual and season. The top SSF model for M1 in the snow season was Model 5,

whereas the top SSF model for F2 in the snow season was Model 15 (Table 3.7). Further,

the top model differed seasonally for M1 (Table 3.7).

At a fine scale, the mine cougars took steps on the reclaimed mines that ended in

habitats closer to barren/rocky outcrops, forest edges, forests, and high relative high

bighorn sheep availability (Figure 3.2). However, selection for these landscape features

varied seasonally for M1. M1 more strongly selected for barren/rocky outcrops, forest

edges, and forests when moving on reclaimed mines in the snow season, as compared to

snow-free (Figure 3.2). In contrast, M1 did not select for high relative bighorn sheep

availability when moving on reclaimed mines in the snow season but did during the snow-

free season (Figure 3.2).

Additionally, M1 selected for fewer landscape features when moving on

reclaimed mines in the snow season, as compared to the snow-free season. M1 took

steps on the reclaimed mines that ended in habitats with higher RTPs, southerly or

westerly aspects, more rugged terrain, less dense forest edge, greater distance to shrubs,

and shorter distance to tree patches, during the snow season (Figure 3.2). M1 took steps

that ended in habitats with higher RTPs, easterly aspects, and shorter distances to

grasslands and anthropogenic land, during the snow-free season (Figure 3.2).

Step selection also varied between individuals in the mine group of cougars. For

instance, during the snow season, F2 took steps that ended in habitats farther from

streams, grasslands, and high relative elk availability (Figure 3.2). Selection against these

features was unique to F2.

Kill occurrence on reclaimed mines

The top model to estimate cougar kill occurrence on reclaimed mines was Model 2 (Table

3.8), which included the encounter model and distance to edge (Table 3.4). Thus,

occurrence of cougar-made bighorn kills on the reclaimed mines was influenced by

where cougars encountered bighorn sheep ($\beta_{\mathbf{kud.cougar}}$ = 0.840, SE = 0.189; $\beta_{\mathbf{bh.avail}}$ =

0.907, SE = 0.266), as well as by landscape configuration. In particular, cougars killed

bighorn sheep on reclaimed mines closer to forest edges (β_{diedge} = -1.092, SE = 0.486; Appendix 3.5).

Prey composition on reclaimed mines

Bighorn sheep comprised roughly 89% of prey killed on reclaimed mines, and other ungulates comprised the remainder, pooled among all study cougars (Table 3.9). Deer and large ungulates comprised the majority of prey killed off reclaimed mines (63 and 26% by biomass, respectively; Table 3.9). Prey composition was more variable off the reclaimed mines and included beaver (1.83%), bighorn sheep (5.39%), canids (3.13%), and other (0.58%) prey (Table 3.9).

We detected a high degree of prey specialization among study cougars (Figure 3.3). Most non-mine cougars (F1, F3, F5, and M2) specialized in deer, with one individual (F4) specializing in large ungulates (Figure 3.3). Mine cougars (M1 and F2) both specialized in bighorn sheep, but deer and beaver were also prevalent in M1's diet (Figure 3.3). Bighorn sheep was the primary prey for both M1 and F2 while on reclaimed mines (Figure 3.4). Deer and bighorn sheep were the primary prey for M1 and F2, respectively, while off reclaimed mines (Figure 3.4).

Of the ungulates available on the reclaimed mines (bighorn sheep, elk, and mule deer), cougars selected bighorn sheep disproportionately to their availability by frequency ($p < 0.05$, Fisher's exact test; Table 3.10) and by biomass ($p < 0.001$, Fisher's exact test; Table 3.10). When we conducted post-hoc pairwise Fisher's exact tests with a Bonferroni correction between the three prey species, we did not find evidence that

cougars selected bighorn sheep disproportionately to their availability by frequency (bighorn sheep vs. elk $p = 0.06$; elk vs. mule deer $p > 0.50$; bighorn sheep vs. mule deer $p > 0.50$; Table 3.10). However, cougars selected bighorn sheep and did not select elk, when availability was defined by biomass (bighorn sheep vs. elk $p < 0.05$; elk vs. mule deer $p > 0.05$; bighorn sheep vs. mule deer $p > 0.50$; Table 3.10). Cougars selected adult female, adult male, and juvenile bighorn sheep on the reclaimed mines relative to their availability, by frequency ($p > 0.50$, Fisher's exact test) and by biomass ($p > 0.50$, Fisher's exact test).

DISCUSSION

Two cougars, an adult male (M1) and an adult female (F2), selected reclaimed mines when establishing home ranges. We predicted that individuals would integrate portions of the reclaimed mines into their home ranges, instead of exclusively using the 130-km^2 mines. Our results corroborate this prediction. During the snow season, F2's home range encompassed 67% of the reclaimed mines, yet the reclaimed mines comprised only 36% of her entire home range. Similarly, M1's snow and snow-free home ranges encompassed 80 and 92% of the reclaimed mines, respectively, yet the reclaimed mines comprised only 8 and 9% of his snow and snow-free home ranges, respectively. Adult cougar home ranges often overlap one another (Neal et al. 1987, Pierce et al. 1999, Sawyer and Lindzey 2002), and the average home range size for an adult cougar in our study was 328 km^2. The 130-km^2 reclaimed mines represent a portion of the average cougar's home range and are likely too small to exclusively support one individual or more than two resident cougars. We do acknowledge, however, that we might have missed resident

cougars that incorporated portions of the reclaimed mines into their home ranges. During the monitoring period, we found one incidental kill on Gregg River that exhibited signs of cougar feeding behaviour but was not made by any of the cougars we collared. Further, BWT collected camera trap images of an uncollared cougar during the monitoring period, approximately 10 km south of Luscar. We concentrated intense search effort trying to catch a cougar south-east of the reclaimed mines, yet never observed a single cougar track in this region, other than those made by collared cougars. Thus, it is possible that we missed collaring a resident cougar that also established its home range on the reclaimed mines during our study.

Multiple studies highlight the potential for cougar habitat selection to vary individually in response to human development (Kertson et al. 2011, Wilmers et al. 2013, Knopff et al. 2014a, Benson et al. 2016, Buderman et al. 2018), but we did not observe this at the second-order scale. Instead, both the mine and non-mine cougars established home ranges nearer to anthropogenic land, highways, and well pads, in both seasons. Therefore, it stands to reason that non-mine cougars would respond to the reclaimed mines similarly to the mine group of cougars. At the third-order scale, we observed variability in response to anthropogenic land, highways, and well pads among the two cougar groups, which supports previous findings (Kertson et al. 2011, Wilmers et al. 2013, Knopff et al. 2014a, Benson et al. 2016, Buderman et al. 2018). The non-mine cougars selected habitat further from anthropogenic land and well pads, which was similar to other cougar populations in west-central Alberta (Knopff et al. 2014a).

Selection for natural landscape features within home ranges was similar for the mine and non-mine groups of cougars. For example, both groups selected habitat with rugged terrain, southerly aspects, and lower density of forest edges, and selected habitat closer to barren/rocky outcrops, forests, forest edges, and water drainages, within home ranges. Cougars select habitats in prey-rich areas (Kertson et al. 2011), such as southerly aspects (Telfer 1978). Cougars also select forests (Holmes and Laundre 2006, Kertson et al. 2011) using vegetative cover while hunting (Logan and Irwin 1985). Rugged terrain and water drainages also provide cover for cougars while hunting (Logan and Irwin 1985, Burdett et al. 2010). Interestingly, both groups of cougars in our study area avoided high density of forest edge when selecting habitats within home ranges. We propose that avoidance of areas with dense forest edge likely represented avoidance of landscapes that were too highly fragmented for cougars. However, cougars selected habitats near forest edges within home ranges, representing a fine-scale preference for habitats that facilitate hunting (Laundre and Hernandez 2003, Lehman et al. 2017).

We expected that if cougars selected home ranges near reclaimed mines to exploit residual landscape configuration and abundant ungulate prey that cougars would select reclaimed mines at both second- and third-order scales. However, we found partial support for this hypothesis. We did not find evidence that F2 selected reclaimed mines within her home range, despite situating her home range nearer to reclaimed mines. In fact, F2 avoided habitats near reclaimed mines within her home range. We submit that by removing locations from within 100 m of kill sites, we may have created a bias against the reclaimed mines for any individual that spent a disproportionate amount of time on

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the reclaimed mines handling prey. F2 may also have selected her home range based on where there was available territory. In contrast, M1 selected habitat nearer to reclaimed mines at the second and third order, in both seasons, which supports our original prediction.

Cougars did not exploit residual landscape configuration as expected when selecting habitat at fine scales, or when depredating bighorn sheep. We observed weak selection for tree patches by M1 during the snow season, but otherwise, cougars did not select tree patches as we anticipated. Kertson et al. (2011) found that cougars used forest patches and reserves, within residential areas, but these patches were much larger than the tree patches in our study area. We submit that tree patches on the reclaimed mines may be too small to provide adequate habitat for cougars, while moving through open landscapes. The surrounding land cover type may also influence whether cougars limit themselves to moving within and among forest patches. As such, residential areas may correspond to riskier habitat than open grasslands for cougars, and may explain why our results differed from Kertson et al. (2011). We did, however, find evidence that cougars selected forest edges at a fine scale, which supports our prediction that cougars would exploit residual landscape configuration on the reclaimed mines. Forest edges included the edges of tree patches. Selection for forest edges, and lack of selection for tree patches, provides additional evidence that tree patches did not provide any innate benefit to cougars at a fine scale. Finally, we found that cougars selected habitats near forest edges for killing bighorn sheep. Ultimately, these results support that edges,

regardless of whether they encompassed a tree patch, provided preferred hunting habitats and facilitated cougar predation on bighorn sheep.

Cougars selected landscape features that increased the likelihood of encountering bighorn sheep, during the snow season. Cougars commonly select habitat where prey are abundant (Pierce et al. 2004, Laundre et al. 2007, Knopff et al. 2010b, Kertson et al. 2011, Lowrey et al. 2016), so we postulate that selection of rocky outcrops and high relative bighorn sheep availability was to increase the likelihood of encountering bighorn sheep. However, we found that M1 did not select habitat with high relative bighorn sheep availability during the snow-free season, which may reflect changes in distribution of prey, or may reflect changes in his hunting behaviour.

Further, we found support that cougars selectively preyed and specialized on bighorn sheep on the reclaimed mines, suggesting cougars used the reclaimed mines to hunt sheep. Overall, we detected a high degree of specialization among the seven study cougars, with most non-mine cougars specializing in deer. Both mine cougars were bighorn sheep specialists. When we assessed selection by biomass, we found that cougars selected bighorn sheep more than was expected and selected elk less than expected, as compared to the availability of each species. However, when we assessed selection by frequency of individuals, we found evidence that cougars selected bighorn sheep and elk relative to their availabilities. If we assume that cougars hunt prey based on biomass, we can conclude that cougars selected bighorn sheep over elk on the reclaimed mines. Elk are larger than sheep and result in a bigger meal for cougars, so selecting elk might allow cougars to maximize meat consumption per predation event.

Alternatively, because elk are larger, depredating them might present a greater challenge to cougars. We did not find evidence that cougars selected a particular age-sex class of bighorn sheep, contrary to other studies (Harrison and Hebert 1988, Ross et al. 1997, Knopff et al. 2010a). However, we acknowledge that our sample size of classified bighorns was small and might have limited our ability to adequately test whether cougars selected age-sex classes on the reclaimed mines.

We did not observe differences in prey selection for bighorn sheep and mule deer on reclaimed mines. Cougars can cause declines in bighorn sheep populations through apparent competition with deer (Holl et al. 2004, Johnson et al. 2013). M1 consumed mainly bighorn sheep on the reclaimed mines and consumed mainly bighorn sheep and deer off the reclaimed mines (Figure 3.4). Thus, both prey likely contributed to the maintenance of this individual. If subsisting primarily on deer off the reclaimed mines allowed M1 to almost exclusively target bighorn sheep on the reclaimed mines, then asymmetric apparent competition might have negative effects on selected prey (i.e. bighorn sheep) (Holl et al. 2004, Johnson et al. 2013). Further, cougars can prey-switch when preferred prey are not abundant (Soria-Diaz et al. 2018). M1 may have prey-switched from deer to more abundant prey (i.e. bighorn sheep) while hunting the reclaimed mines. If multiple cougars within our study were to have similar diet preferences to M1, then it is possible bighorn sheep on the reclaimed mines may experience declines.

We evaluated fine-scale habitat selection, kill occurrence, and prey selection using an effective sample size of two individual cougars. We recognize that our sample

size might limit our ability to make population-level inferences. However, we emphasize the potential for one cougar that specializes on bighorn sheep to have substantial effects (Harrison and Hebert 1988, Ross et al. 1997, Festa-Bianchet et al. 2006, Blake and Gese 2016b). For example, Ross et al. (1997) and Festa-Bianchet et al. (2006) both observed a specialist female cougar learn to hunt, then subsequently prey heavily on, bighorn sheep in south-western Alberta. Cougars have a high capacity to acquire hunting techniques and foraging behaviours that are required to specialize on target prey species (Knopff et al. 2010a, Elbroch and Wittmer 2013, Lowrey et al. 2016). Spatial foraging behaviours, such as spending disproportionate amounts of time in the target prey species' preferred habitat, can result in dietary specialization for cougars (Lowrey et al. 2016). Similar to Lowrey et al. (2016), we observed that both M1 and F2 selected areas of the reclaimed mines where bighorn sheep were abundant, resulting in diets that were mainly composed bighorn sheep (Figures 3.3 & 3.4). Thus, the effect that one cougar can have on a bighorn sheep population is demonstrable despite small sample sizes, when considering the substantial evidence we have of sheep specialization in our study system.

Our results can help to understand whether the reclaimed mines are an ecological trap or ecological sanctuary for bighorn sheep. Some local biologists hypothesize that bighorn sheep are enticed to the reclaimed mines by non-native forage, engineered escape terrain, and protection from hunting, but experience higher rates of predation due to residual landscape configuration (Kneteman 2016)[19]. Others hypothesize that engineered escape terrain does not parallel natural escape terrain and instead, high walls provide accessible congregations of sheep that cougars can easily stalk and ambush. We

found that two cougars specialized on bighorn sheep and contributed to 17 bighorn sheep kills on the reclaimed mines, during a period of 19 months. And, we found that forest edges facilitated cougar predation on bighorn sheep. When we included high walls in kill occurrence models (Appendix 3.5), we found support that bighorn sheep kills occurred closer to high walls than random locations. These results support that engineered escape terrain may not provide bighorn sheep adequate protection from cougars. Cougar attacks on bighorn sheep might disproportionately occur near natural escape terrain as well, but spatial cougar predation risk has never been modelled before for bighorn sheep. We recognize that bighorn populations on the reclaimed mines are likely representative of recruitment due to nutritious forage and mild winters. However, if bighorn sheep also experience disproportionately high predation at forest edges adjacent to forage and high walls, this mismatch may represent an ecological trap for bighorn sheep.

The population of bighorn sheep found on and surrounding the reclaimed mines are of interest to hunters and wildlife enthusiasts in Alberta. Luscar and Gregg River will revert to Crown land in upcoming years, and the Province of Alberta will determine how resident wildlife populations are managed. Provincially, both bighorn sheep and cougars are managed mainly as trophy species (Government of Alberta 2012, 2015). A strong-held belief of many Albertan sheep hunters is that cougars that specialize in bighorn sheep reduce the availability of trophy rams for hunters. As such, a fall cougar-hunting season was implemented in Alberta, in 2016, to allow sheep hunters to opportunistically kill cougars, in the hopes of targeting sheep-killing cougars. Targeting individual cougars

that inhabit the reclaimed mines and specialize on bighorn sheep might not be an efficient way to ensure optimal management of predator and prey populations. For instance, both mine cougars that we studied died during our study period, so we are lacking information on the dietary preferences of any cougars that currently inhabit the reclaimed mines. Instead of eliminating individual cougars, we suggest an alternative approach. We recommend that managers consider how configuring landscapes might mitigate predation pressure on bighorn sheep, especially because this strategy can apply to all future cougars that integrate the reclaimed mines into their home ranges. Ensuring that forest edges are trimmed back from high walls is an example of landscape configuration that might help to reduce this ecological trap for bighorn sheep.

Table 3.1 Summary of landscape features (non-mine and mine specific) as covariates for modelling cougar resource selection in west-central Alberta. Models are broken into second and third order resource selection functions (RSF), step selection functions (SSF), and kill occurrence model (kill). When using "distance to", we mean "distance to nearest" feature. Covariate code, type of variable, units, and whether the covariate was considered in each respective candidate set of models are displayed.

Covariate	Covariate code	Type of variable	Unit	Inclusion of covariate			
				Second	Third	SSF	Kill
Non-mine specific							
Relative topographic position index	rtp	Continuous	Unitless	X	X	X	X
Terrain ruggedness index	tri	Continuous	Unitless	X	X	X	X
Aspect	aspect	Categorical	Unitless	X	X	X	
Distance to forest	diforest	Continuous	Metre	X	X	X	
Distance to shrub	dishrub	Continuous	Metre	X	X	X	
Distance to grassland	digrassland	Continuous	Metre	X	X	X	
Distance to lake or river	dilakeriv	Continuous	Metre	X	X	X	
Distance to stream	distream	Continuous	Metre	X	X	X	
Distance to barren/rocky outcrop	dibarrenrocky	Continuous	Metre	X	X	X	
Distance to divided highway	dihwy	Continuous	Metre	X	X		
Distance to well pad	diwellpad	Continuous	Metre	X	X		
Distance to forest edge	diedge	Continuous	Metre	X	X	X	X
Distance to anthropogenic land	dianthro	Continuous	Metre	X	X	X	
Density of forest edge[a]	dstyedge	Continuous	m/m^2	4 km	2 km	300 m	
Mine specific							
Distance to reclaimed mines	dimines	Continuous	Metre	X	X		
Distance to tree patch	ditreepatch	Continuous	Metre			X	X
Distance to disturbed area	didist	Continuous	Metre				X
Relative bighorn sheep availability[b,c]	bh.avail	Continuous	Unitless			X	X
Relative elk availability[b]	ek.avail	Continuous	Unitless			X	
Relative mule deer availability[b]	dr.avail	Continuous	Unitless			X	
Kernel UD cougar	kud.cougar	Continuous	Unitless				X

[a] best scale for dstyedge in each model is specified
[b] relative prey availability separated into snow and snow-free seasons for SSFs
[c] annual relative bighorn sheep availability used in kill occurrence model

Table 3.2 *A priori* candidate models and top model results per individual cougar for second and third order resource selection functions, during the snow and snow-free seasons. Non-mine cougar group models (nm) pooled data from F1, F3, F4, and F5, and included a random effect for individual cougar (1|ind).

Model name	Covariate structure	Second order Snow	Second order Snow-free[c]	Third order Snow	Third order Snow-free[c]
Null	1				
Base	aspect + tri + rtp + dilakeriv + diforest + dibarrenrocky				
1	Base + dishrub + digrassland + distream				
2	Base + dishrub + dstyedge*				
3	Base + dianthro + dstyedge*				
4	Base + diwellpad + dihwy + dianthro				
5	Base + dianthro + dstyedge* + dishrub + distream			F4	
6	Base + dihwy + dianthro + dishrub + dstyedge*				F3, nm
7	Base + diwellpad + dihwy + dstyedge* + diedge				F1
8	Base + diwellpad + dianthro + dstyedge* + distream + digrassland				
9	Model 4 + dstyedge* + diedge				
10	Model 4 + dstyedge* + diedge + digrassland[a]	F1		F1, F5	
11	Model 4 + dstyedge* + diedge + distream	F4	F5	M2	
12	Model 4 + dstyedge* + diedge + distream + dishrub	F5	F4		F4, F5
13[b]	Model 4 + dstyedge* + diedge + distream + dishrub + digrassland	M2, F3, nm	F1, F3, nm	F3, nm	--[b]
14	Model 8 + dimines				
15	Model 9 + dimines	M1		M1	
16	Model 10 + dimines				
17	Model 11 + dimines				
18	Model 12 + dimines	F2			M1
19[b]	Model 13 + dimines		M1	F2	--[b]

*dstyedge4km was used in 2nd order models, dstyedge2km was used in 3rd order models

[a] digrassland was not included in the candidate set for 3rd order snow-free RSFs; thus, dishrub replaced digrassland in Model 10

[b] because dishrub replaced digrassland in 3rd order snow-free RSFs, Models 13 and 19 are replicates and were not included in the candidate set for 3rd order snow-free RSFs

[c] Individual cougars M2 and F2 do not have snow-free seasonal models

Table 3.3 *A priori* candidate models for step selection functions of two individual cougars, M1 and F2, on reclaimed coal mines in west-central Alberta, Canada.

Model name	Covariate structure
Base	aspect + diforest + dibarrenrocky + diedge
1	Base + dilakeriv
2	Base + dishrub + rtp + ditreepatch
3	Base + distream + tri
4	Base + dilakeriv + distream + diedge
5	Base + dishrub + rtp + tri + ditreepatch + dstyedge300m
6	Base + digrassland + distream
7	Base + digrassland + rtp + dianthro
8	Model 5 + bh.avail[a]
9	Model 6 + bh.avail
10	Model 7 + bh.avail
11	Model 5 + bh.avail + dr.avail[a]
12	Model 6 + bh.avail + dr.avail
13	Model 7 + bh.avail + dr.avail
14	Model 5 + bh.avail + ek.avail[a]
15	Model 6 + bh.avail + ek.avail
16	Model 7 + bh.avail + ek.avail

[a] prey availability split into snow and snow-free seasons for SSFs

Table 3.4 *A priori* candidate models for kill occurrence on reclaimed mines in west-central Alberta. Data for these models were pooled between mine cougars (n = 17 kills pooled over n = 2 cougars; M1 and F2) and seasons.

Model name	Covariate structure
Null 1	kud.cougar
Null 2	bh.avail
Encounter	kud.cougar + bh.avail
1	Encounter + ditreepatch
2	Encounter + diedge
3	Encounter + diedge + ditreepatch
4	Encounter + rtp
5	Encounter + tri
6	Encounter + diedge + didist
7	Encounter + rtp + tri
8	Encounter + diedge + ditreepatch + didist + rtp + tri

Table 3.5 Seasonally stratified scaled beta coefficients from second order resource selection functions, comparing two individual cougars that used reclaimed mines, M1 and F2, with non-mine cougar group model in west-central Alberta, Canada. Scaled beta coefficients (Coeff.) that overlap zero are indicated with an asterisk.

	Second order									
Covariate	M1				Non-mine group				F2	
	Snow		Snow-free		Snow		Snow-free		Snow	
	Coeff.	SE	Coeff.	SE	Coeff.	SE	Coeff.	SE	Coeff.	SE
rtp	0.050	0.049	-0.054	0.042	0.204	0.023	-0.065	0.020	-0.044*	0.106
dibarrenrocky	-0.103	0.052	-0.269	0.053	0.713	0.025	0.535	0.021	0.052*	0.087
diedge	-0.162	0.055	-0.384	0.085	0.541	0.035	0.600	0.030	-0.532	0.183
diforest	-1.485	0.147	-0.981	0.108	-0.461	0.054	-0.353	0.044	-0.206	0.186
dilakeriv	-0.776	0.054	-0.398	0.046	0.043	0.035	0.553	0.027	-0.807	0.124
dishrub			-0.072	0.053	0.214	0.028	-0.017	0.022	0.448	0.114
distream			-0.232	0.067	-0.379	0.024	-0.435	0.021	-0.242	0.186
tri	0.721	0.078	0.213	0.064	1.393	0.048	0.818	0.040	0.596	0.157
dstyedge4km	-0.260	0.074	0.109	0.068	1.339	0.040	0.454	0.028	1.592	0.233
digrassland			-0.154	0.065	-0.075	0.028	-0.064	0.023		
dianthro	-0.538	0.193	-0.895	0.134	-3.163	0.254	-5.577	0.264	-0.202*	0.476
dihwy	-1.468	0.074	-1.252	0.058	-3.223	0.073	-3.359	0.066	-0.415	0.154
diwellpad	-1.649	0.158	-0.804	0.109	-0.564	0.060	-0.719	0.068	-4.334	0.387
dimines[a]	-2.533	0.093	-3.335	0.103					-12.358	0.764
aspectnorth[b]	-0.337	0.144	-0.283	0.111	-0.388	0.066	-0.176	0.054	-0.442	0.214
aspectsouth[b]	1.081	0.131	0.337	0.112	0.545	0.061	0.623	0.050	0.960	0.229
aspectwest[b]	0.724	0.136	0.241	0.118	-0.279	0.071	0.021	0.059	-0.320	0.248

[a] dimines was not included in the candidate set for non-mine cougars
[b] contrasted with easterly aspects

Table 3.6 Seasonally stratified scaled beta coefficients from third order resource selection functions, comparing two individual cougars that used reclaimed mines, M1 and F2, with non-mine cougar group model in west-central Alberta, Canada. Scaled beta coefficients (Coeff.) that overlap zero are indicated with an asterisk.

	Third order									
	M1				Non-mine group				F2	
Covariate	Snow		Snow-free		Snow		Snow-free		Snow	
	Coeff.	SE	Coeff.	SE	Coeff.	SE	Coeff.	SE	Coeff.	SE
rtp	-0.038*	0.047	-0.050	0.038	0.115	0.021	-0.050	0.019	0.133	0.057
dibarrenrocky	-0.290	0.060	-0.398	0.059	-0.177	0.030	-0.079	0.025	-0.561	0.085
diedge	-0.147	0.047	-0.594	0.082	-0.168	0.033			-0.722	0.105
diforest	-0.508	0.067	-0.373	0.051	0.021*	0.026	-0.045	0.022	-0.179	0.052
dilakeriv	-0.625	0.042	-0.509	0.041	-0.135	0.032	0.150	0.026	-0.760	0.065
dishrub			-0.141	0.053	0.060	0.029	-0.060	0.023	0.451	0.077
distream			-0.169	0.050	-0.026	0.021			-0.030	0.077
tri	0.669	0.057	0.071	0.039	0.824	0.035	0.220	0.029	0.522	0.065
dstyedge2km	-0.208	0.062	-0.124	0.065	-0.203	0.030	-0.194	0.027	0.092*	0.123
digrassland					0.165	0.027			0.213	0.073
dianthro	-0.209	0.042	-0.128	0.030	0.395	0.070	-0.008*	0.056	0.316	0.071
dihwy	-0.661	0.075	-0.413	0.043	-0.794	0.077	-1.063	0.056	-0.027*	0.106
diwellpad	0.269	0.058	0.208	0.038	0.312	0.031			-0.261	0.065
dimines[a]	-0.347	0.082	-0.591	0.068					2.457	0.459
aspectnorth[b]	-0.362	0.141	-0.362	0.105	-0.584	0.065	-0.163	0.055	-0.883	0.131
aspectsouth[b]	0.863	0.130	0.234	0.109	0.296	0.058	0.264	0.050	-0.134*	0.138
aspectwest[b]	0.603	0.136	0.003*	0.115	-0.357	0.071	-0.164	0.060	-0.501	0.145

[a] dimines was not included in the candidate set for non-mine cougars
[b] contrasted with easterly aspects

Table 3.7 Top-ranked conditional logistic regression models to estimate step selection for two individual cougars, M1 and F2, on reclaimed mines in west-central Alberta, Canada. Snow and snow-free seasons are presented for M1, and snow season is presented for F2. Number of estimated parameters (K), model log-likelihood (LL), small sample size corrected Akaike's Information Criterion (ΔAICc), and Akaike weight (w_i) are displayed. The top model for each individual and season is in bold.

Individual & season	Rank	Model[a]	K	LL	AICc	ΔAICc	w_i
M1 snow	1	**Model 5**	11	-1111.51	2245.57	0	0.56
	2	Model 14	13	-1110.17	2247.1	1.531	0.26
	3	Model 11	13	-1110.87	2248.5	2.932	0.129
	4	Model 2	9	-1116.86	2252.09	6.519	0.021
	5	Model 16	11	-1115.22	2252.98	7.417	0.014
M1 snow-free	1	**Model 10**	10	-1210.39	2441.2	0	0.562
	2	Model 16	11	-1210.22	2442.96	1.763	0.233
	3	Model 13	11	-1210.35	2443.22	2.021	0.205
	4	Model 7	9	-1217.79	2453.94	12.74	0.001
	5	Model 9	9	-1222.16	2462.67	21.467	0
F2 snow[b]	1	**Model 15**	10	-1725.56	3471.43	0	0.934
	2	Model 9	9	-1730.43	3479.11	7.684	0.02
	3	Model 16	11	-1728.61	3479.59	8.166	0.016
	4	Model 12	10	-1729.81	3479.93	8.504	0.013
	5	Model 6	8	-1732.14	3480.48	9.05	0.01

[a] Model structure based on candidate set listed in Table 3.3
[b] F2 does not have a SSF for snow-free season

Table 3.8 Top-ranked conditional logistic regression models to estimate cougar kill occurrence of bighorn sheep on reclaimed mines in west-central Alberta, Canada. Kills (n = 17) were pooled over cougars (n = 2; M1 and F2) for reclaimed mines. Number of estimated parameters (K), model log-likelihood (LL), small sample size corrected Akaike's Information Criterion (ΔAICc), and Akaike weight (w_i) are displayed. The top model is in bold.

Rank	Model[a]	K	LL	AICc	ΔAICc	w_i
1	**Model 2**	**3**	**-37.81**	**83.04**	**0**	**0.532**
2	Model 3	4	-37.59	85.68	2.646	0.142
3	Model 6	4	-37.81	86.12	3.084	0.114
4	Encounter	2	-41.08	86.83	3.796	0.08
5	Model 1	3	-39.91	87.24	4.202	0.065

[a] Model structure based on candidate set listed in Table 3.4

Table 3.9 Comparison of the proportional frequency and biomass of prey killed by cougars on and off mines in west-central Alberta, Canada. Results are presented for condensed prey types using data from 183 predation incidents for 7 cougars. Age-sex classes are further summarized for bighorn sheep, deer spp., and large ungulates, where % Freq. and % Biomass values add up to 100% within each focal ungulate group (symbolized by shaded rows). Table totals were calculated from summing all non-shaded rows.

Prey	On mine				Off mine			
	Freq.	% Freq.	Biomass (kg)[c]	% Biomass	Freq.	% Freq.	Biomass (kg)[c]	% Biomass
Beaver	0	0.00	0	0.00	6	3.66	159	1.83
Bighorn sheep	17	89.47	1181	89.91	6	3.66	469	5.39
Adult female	3	17.65	195	16.52	1	16.67	65	13.87
Adult male	4	23.53	468	39.64	2	33.32	234	49.95
Unknown adult[a]	2	11.76	182	15.42	1	16.67	91	19.42
Juvenile	3	17.65	86	7.24	1	16.67	29	6.08
Unknown[b]	5	29.41	250	21.18	1	16.67	50	10.68
Canis spp.	0	0.00	0	0.00	17	10.37	272	3.13
Deer spp.	1	5.26	31	2.36	110	67.07	5493	63.21
Adult female	0	0.00	0	0.00	7	6.36	490	8.92
Adult male	0	0.00	0	0.00	3	2.73	285	5.19
Unknown adult[a]	0	0.00	0	0.00	14	12.73	1155	21.03
Juvenile	1	100.00	31	100.00	47	42.73	1457	26.52
Unknown[b]	0	0.00	0	0.00	39	35.45	2106	38.34
Large ungulate	1	5.26	102	7.73	15	9.15	2247	25.86
Adult female elk	0	0.00	0	0.00	1	6.67	230	10.24
Adult male elk	0	0.00	0	0.00	0	0.00	0	0.00
Unknown adult elk[a]	0	0.00	0	0.00	0	0.00	0	0.00
Juvenile elk	1	100.00	102	100.00	10	66.67	1015	45.17
Unknown elk[b]	0	0.00	0	0.00	3	20.00	356	15.84
Moose	0	0.00	0	0.00	2	13.33	646	28.75
Other	0	0.00	0	0.00	10	6.10	51	0.58
Total	19	100	1314	100	164	100	8691	100

[a] Unknown adult represents predation incidents where age class could be confirmed, but sex could not be
[b] Unknown represents predation incidents where age class could not be confirmed as either adult or juvenile
[c] Biomass was calculated using table in Appendix 3.3

Table 3.10 Summary of proportion bighorn sheep, elk, and mule deer available on reclaimed mines in 2017 by frequency and biomass (kg). Proportions available represent the means from four surveys, collected by Bighorn Wildlife Technologies Ltd. Observed kills (n = 19) were pooled across mine cougars (n = 2; M1 and F2) and represent kills made in 2017. Observed and expected kills were compared using Fisher's exact tests.

Prey species	Proportion available by frequency	Proportion available by biomass[a]	Observed kills	Expected kills by frequency[b]	Expected kills by biomass[a,b]
Bighorn sheep	0.58	0.35	17	11	7
Elk	0.41	0.64	1	8	12
Mule deer	0.01	0.01	1	0	0
Total	1.00	1.00	19	19	19

[a] See Appendix 3.3 for biomass reference table
[b] Expected number of kills were calculated by multiplying total observed kills by proportion available, by frequency and biomass
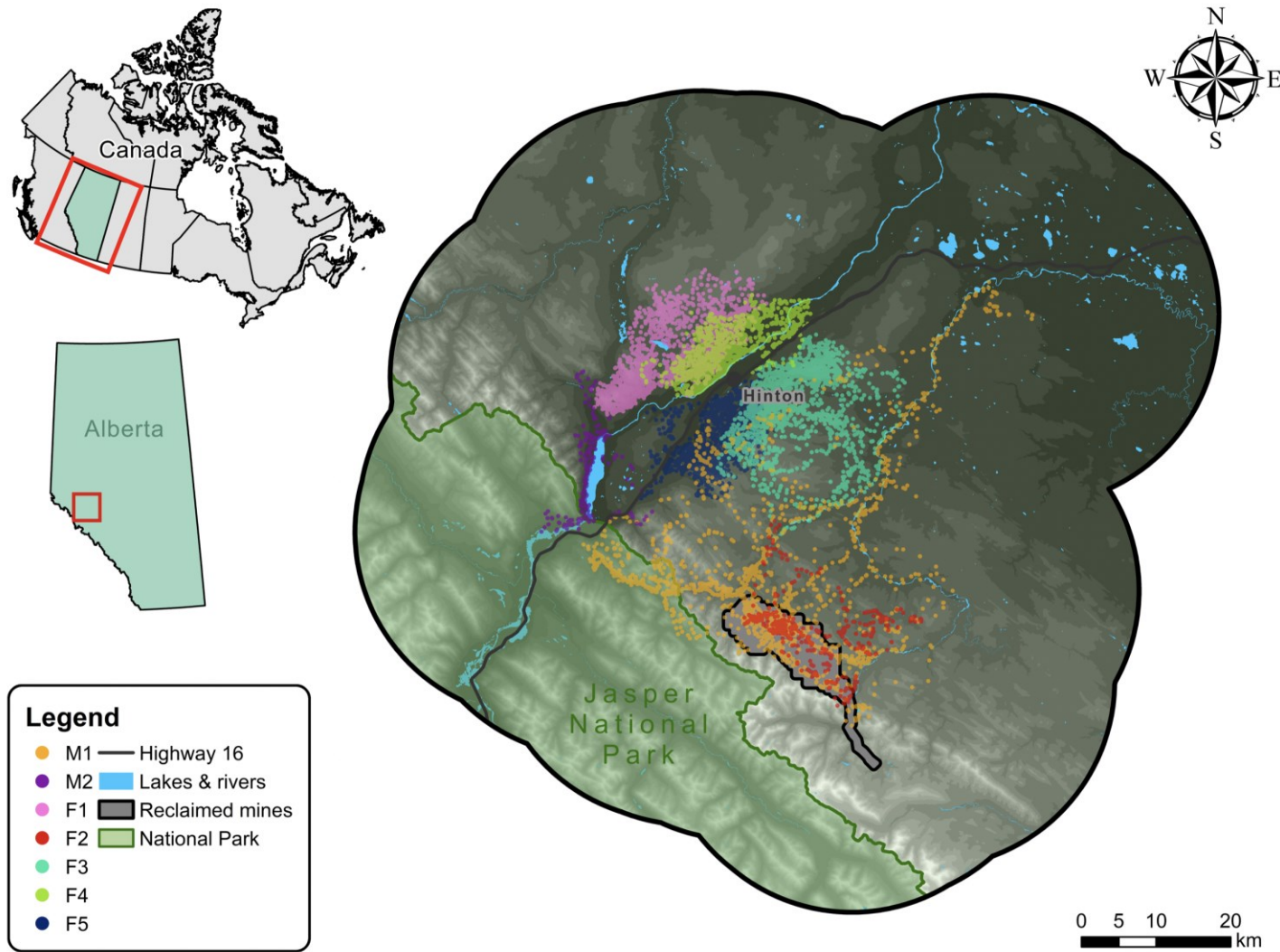
Figure 3.1 Locations of cougars (*n* = 7; each individual symbolized by a different colour), acquired by GPS collars between March 2017 – October 2018, in west-central Alberta, Canada. Males are indicated by M and females are indicated by F. Study area elevation is symbolized using a hillshade effect, with lighter colours representing higher elevations and darker colours representing lower elevations.
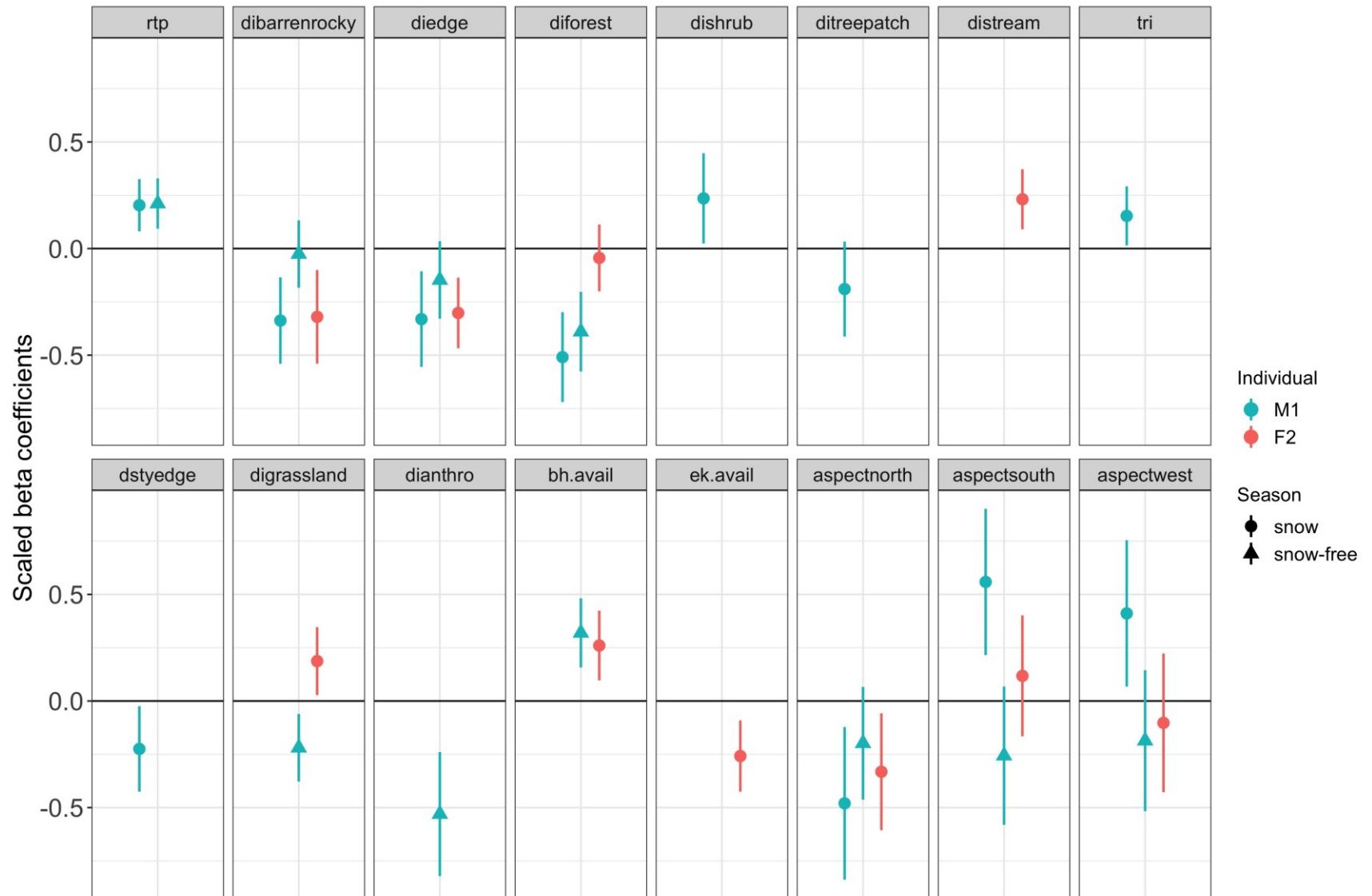
Figure 3.2 Scaled beta coefficients from top step selection functions for M1 and F2, comparing used steps (M1 snow = 496, M1 snow-free = 526, F2 snow = 741) to available steps during snow and snow-free seasons on reclaimed mines in west-central Alberta, Canada. Vertical lines represent 95% confidence intervals for scaled beta coefficient estimates. North, south, and west aspects are contrasted against easterly aspects.
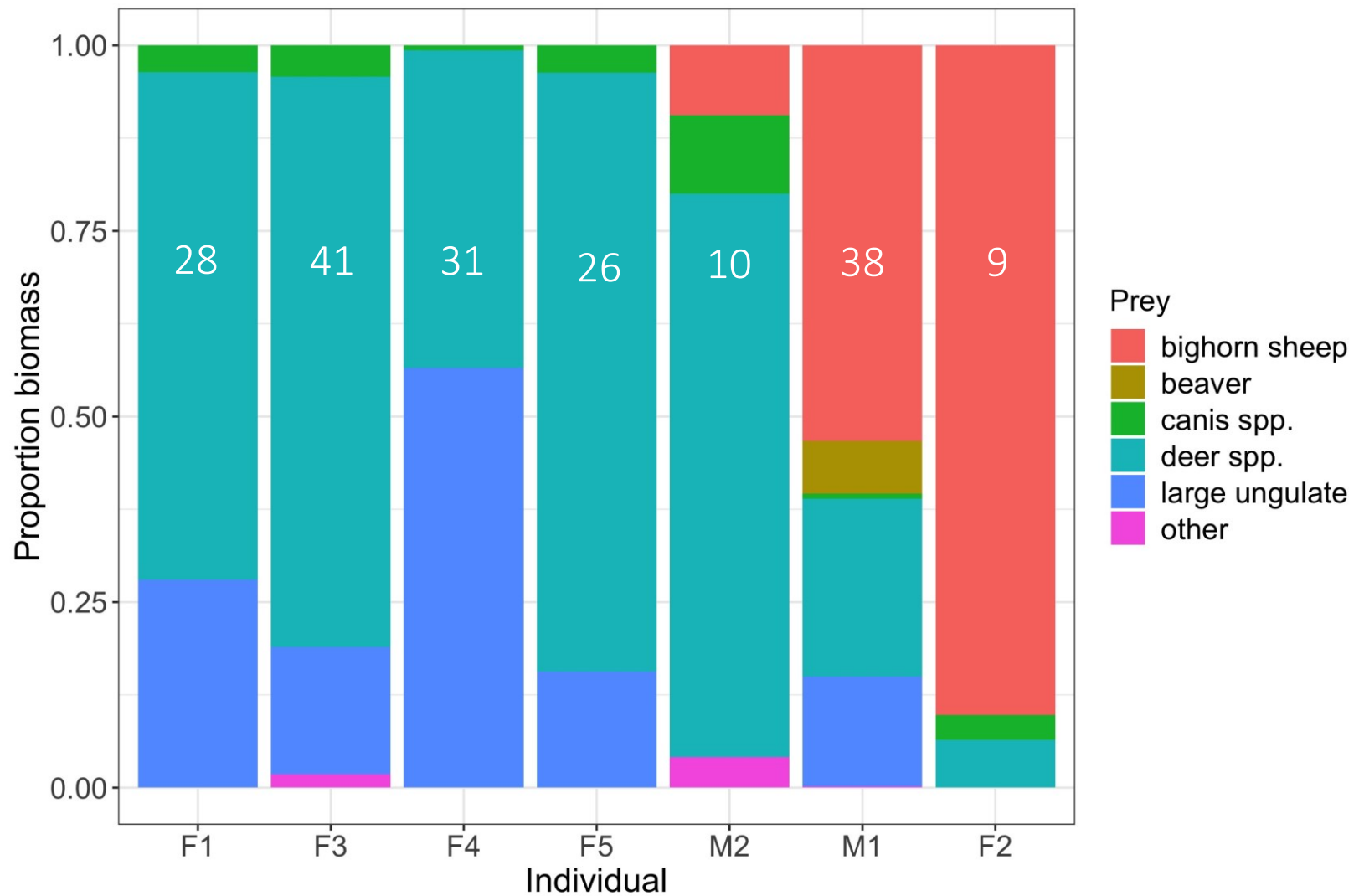
Figure 3.3 Diet composition, by proportion biomass, for non-mine and mine cougars (sample size of kills per individual shown in white). *Canis* spp. includes coyotes, wolves, and foxes; deer spp. includes white-tailed and mule deer; large ungulate includes moose and elk; and other includes lynx, red squirrel, and snowshoe hare.
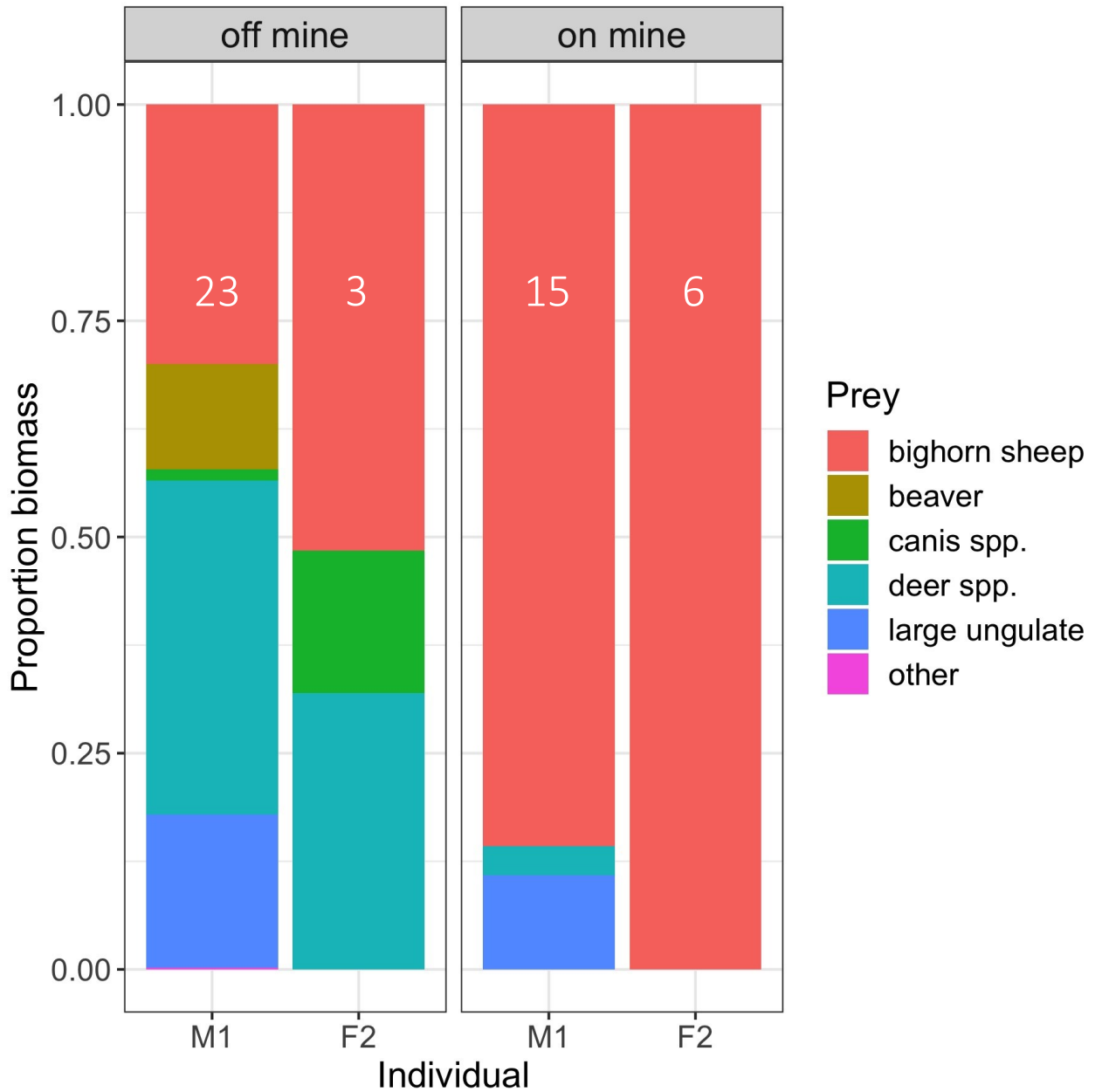
Figure 3.4 Diet composition, by proportion biomass, for M1 and F2 on and off reclaimed mines (sample size of kills shown in white). *Canis* spp. includes coyotes, wolves, and foxes; deer spp. includes white-tailed and mule deer; large ungulate includes moose and elk; and other includes lynx, red squirrel, and snowshoe hare.

CHAPTER 4 – CONCLUSIONS: WILDLIFE AS A TARGET FOR EVALUATING RECLAMATION

In this thesis, we examined how landscape configuration and features influenced habitat selection by a predator and prey population on two reclaimed coal mines. One of the primary end land uses of the reclaimed mines is habitat for wildlife (Government of Alberta 2013) and based on our findings, the reclaimed mines have provided habitats for bighorn sheep, elk, mule deer, and cougars.

In Chapter 2, we found that bighorn sheep most strongly responded to reclamation, compared to elk and mule deer. In particular, bighorn sheep strongly selected for escape terrain on reclaimed mines, and this was similar to what others have found on mines (Jansen et al. 2006, Bleich et al. 2009, Poole et al. 2016) and in natural areas (Smith et al. 1991, Andrew et al. 1999, Singer et al. 2000). Our findings support that retaining and engineering mining pit high walls provided critical habitat for bighorn sheep and ensured that the reclaimed mines would be attractive habitat within geographical ranges of sheep. Additionally, we found in Chapter 2 that elk and mule deer strongly selected for reclaimed grasslands, which supports that non-native legumes and forbs also attracted ungulates to the reclaimed mines. Ultimately, we provided the first resource selection functions for ungulates on Luscar and Gregg River, and we successfully predicted habitat selection based on residual landscape features.

In Chapter 3, we found that cougars established home ranges near reclaimed mines, within their geographical range. Further, we highlighted that at a fine scale, cougars selected habitats where they were likely to encounter bighorn sheep, as well as where encounters were more likely to be successful predation events. We found support

for our hypothesis that cougars exploited residual landscape configuration when hunting bighorn sheep, and also found that cougars both selectively preyed and specialized on bighorn sheep. Numerous other studies (Kertson et al. 2011, Wilmers et al. 2013, Knopff et al. 2014a, Benson et al. 2016, Buderman et al. 2018) have highlighted the flexibility in which cougars can integrate human-dominated landscapes into their home ranges; the cougars in our study were no exception.

We established that residual landscape features, including high walls, reclaimed grasslands, and forest edges, facilitated the dynamic relationship between cougars and their prey. Experimental manipulations of landscape may also help to understand the relationship between forest edges in close proximity to high walls, and cougar predation on sheep. For example, trimming forest edges back from high walls might help to determine whether cougar-caused bighorn mortality can be reduced, and may help to assess whether the mines represent an ecological trap. Additionally, future studies should examine aspects of the ecological trap hypothesis that were beyond the breadth of this study. For example, Bighorn Wildlife Technologies Ltd. (BWT) recorded adult to juvenile ratios for ungulates between 2004-2017, which could be used to infer population metrics like recruitment. Further, examining alternative population metrics such as immigration and mortality could help to elucidate whether bighorn sheep are attracted to the reclaimed mines from adjacent ranges, and whether cougar predation may be limiting population recruitment.

We wish to shift focus and demonstrate that wildlife can serve as a target for evaluating ecological reclamation. Traditionally, ecological reclamation focuses on the re-

establishment of natural habitats (Peipoch et al. 2015), including aquatic systems (Bernhardt and Palmer 2011), physical properties (Wickham et al. 2013), and vegetation communities (Holl and Cairns 1994, Strong 2002, Wickham et al. 2006, Swab et al. 2017). It is often assumed that if vegetation communities re-establish, then wildlife will follow soon after. However, this might not always be the case, as reclamation can fail at re-establishing wildlife populations (Larkin et al. 2008, Bennett et al. 2013, Peipoch et al. 2015). Failure occurs for a number of reasons, including lack of landscape connectivity (Larkin et al. 2008, Peipoch et al. 2015), heterogeneity (Larkin et al. 2008, Peipoch et al. 2015), quality (Bennett et al. 2013), and complexity (Manning et al. 2013). If one of the components of a functioning ecosystem is successful recolonization of wildlife populations, we have not truly restored ecosystem function until we have restored wildlife (Jones and Davidson 2016).

Before Luscar and Gregg River were mined for coal, the land was forested and supported a multitude of regional fauna including grizzly bears, wolves, cougars, and deer. Reclamation at Luscar and Gregg River aimed to restore mountain wildlife habitat in an area altered by open pit mining for coal (MacCallum and Geist 1992) and previous work exemplified that this reclamation provided forage-rich habitat for grizzly bears (Cristescu et al. 2015a, 2015b). We demonstrated in Chapters 2 and 3 that wildlife occupy and respond to spatial patterns in the reclaimed landscape, which is important if we are going to take an animal-centric approach to evaluating reclamation (Jones and Davidson 2016). Understanding whether wildlife select particular landscape features on the reclaimed mines is paramount to ensuring wildlife continue to recolonize and thrive in reclaimed

habitats. Further, assessing whether wildlife avoid particular landscape features on the reclaimed mines can help ensure that negative effects of mining and reclamation are mitigated. Ultimately, we, as ecologists, must make a shift to consider not only vegetation communities as a metric for evaluating successful ecological reclamation, but also recruitment of diverse wildlife populations (Jones and Davidson 2016).

WORKS CITED

Ager, A. A., B. K. Johnson, J. W. Kern, and J. G. Kie. 2003. Daily and seasonal movements and habitat use by female rocky mountain elk and mule deer. Journal of Mammalogy 84:1076–1088.

Alberta Agriculture and Forestry. 2017. Interpolated weather data since 1961 for Alberta townships.

Anderson, C. R. J., and F. G. Lindzey. 2003. Estimating cougar predation rates from GPS location clusters. Journal of Wildlife Management 67:307–316.

Anderson, E. D., and R. A. Long. 2012. Winter resource selection by female mule deer *Odocoileus hemionus*: Functional response to spatio-temporal changes in habitat. Wildlife Biology 18:153–163.

Andrew, N. G., V. C. Bleich, and P. V. August. 1999. Habitat selection by mountain sheep in the Sonoran Desert: Implications for conservation in the United States and Mexico. California Wildlife Conservation Bulletin 12:1–30.

Ardente, N. C., Á. C. Ferreguetti, D. Gettinger, P. Leal, A. C. Mendes-Oliveira, F. Martins-Hatano, and H. G. Bergallo. 2016. Diversity and impacts of mining on the non-volant small mammal communities of two vegetation types in the Brazilian Amazon. PLoS ONE 11:1–16.

Bacon, M. M., G. M. Becic, M. T. Epp, and M. S. Boyce. 2011. Do GPS clusters really work? Carnivore diet from scat analysis and GPS telemetry methods. Wildlife Society Bulletin 35:409–415.

Beck, J. L., K. T. Smith, J. T. Flinders, and C. L. Clyde. 2013. Seasonal habitat selection by elk in north central Utah. Western North American Naturalist 73:442–456.

Beier, P. 2009. A focal species for conservation planning. Pages 177–189 *in* M. G. Hornocker and S. Negri, editors. Cougar: Ecology and Evolution. Chicago, Illinois.

Beier, P., D. Choate, and R. H. Barrett. 1995. Movement patterns of mountain lions during different behaviors. Journal of Mammalogy 76:1056–1070.

Bennett, V. A., V. A. J. Doerr, E. D. Doerr, A. D. Manning, D. B. Lindenmayer, and H. J. Yoon. 2013. Causes of reintroduction failure of the brown treecreeper: Implications for ecosystem restoration. Austral Ecology 38:700–712.

Benson, J. F., J. A. Sikich, and S. P. D. Riley. 2016. Individual and population level resource selection patterns of mountain lions preying on mule deer along an urban-wildland gradient. PLoS ONE 11:1–16.

Berger, J. 1991. Pregnancy incentives, predation constraints and habitat shifts: Experimental and field evidence for wild bighorn sheep. Animal Behaviour 41:61–77.

Bergman, E. J., R. A. Garrott, S. Creel, J. J. Borkowski, R. Jaffe, and E. Watson. 2006. Assessment of prey vulnerability through analysis of wolf movements and kill sites. Ecological Applications 16:273–284.

Bernhardt, E. S., and M. A. Palmer. 2011. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the Central Appalachians. Annals of the New york Academy of … 1223:39–57.

Blake, L. W., and E. M. Gese. 2016a. Resource selection by cougars: Influence of behavioral state and season. The Journal of Wildlife Management 80:1205–1217.

Blake, L. W., and E. M. Gese. 2016b. Cougar predation rates and prey composition in the Pryor mountains of Wyoming and Montana. Northwest Science 90:394–410.

Bleich, V. C., J. H. Davis, J. P. Marshal, S. G. Torres, and Ben J Gonzales. 2009. Mining activity and habitat use by mountain sheep (*Ovis canadensis*). European Journal of Wildlife Research 55:183–191.

Bleich, V. C., R. T. Bowyer, and J. D. Wehausen. 1997. Sexual segregation in mountain sheep: Resources or predation? Wildlife Monographs 134:3–50.

Blum, M. E., K. M. Stewart, and C. Schroeder. 2015. Effects of large-scale gold mining on migratory behavior of a large herbivore. Ecosphere 6:1–18.

Bourbeau-Lemieux, A., M. Festa-Bianchet, J.-M. Gaillard, and F. Pelletier. 2011. Predator-driven component Allee effects in a wild ungulate. Ecology Letters 14:358–363.

Boyce, M. S., P. R. Vernier, S. E. Nielsen, and F. K. A. Schmiegelow. 2002. Evaluating resource selection functions. Ecological Modelling 157:281–300.

Buchanan, C. B., J. L. Beck, T. E. Bills, and S. N. Miller. 2014. Seasonal resource selection and distributional response by elk to development of a natural gas field. Rangeland Ecology & Management 67:369–379.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling. Oxford University Press Inc., New York.

Buderman, F. E., M. B. Hooten, M. W. Alldredge, E. M. Hanks, and J. S. Ivan. 2018. Time-varying predatory behavior is primary predictor of fine-scale movement of wildland-urban cougars. Movement Ecology 6:1–16.

Burdett, C. L., K. R. Crooks, D. M. Theobald, K. R. Wilson, E. E. Boydston, L. M. Lyren, R. N. Fisher, T. W. Vickers, S. A. Morrison, and W. M. Boyce. 2010. Interfacing models of wildlife habitat and human development to predict the future distribution of puma habitat. Ecosphere 1:1–21.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: A practical information-theoretic approach. Second Edition. Springer-Verlag, New York, USA.

Carson, R. G., and J. M. Peek. 1987. Mule deer habitat selection patterns in northcentral Washington. Journal of Wildlife Management 51:46–51.

Cassirer, E. F., D. J. Freddy, and E. D. Ables. 1992. Elk responses to disturbance by cross-country skiers in Yellowstone-National-Park. Wildlife Society Bulletin 20:375–381.

Chetkiewicz, C.-L. B., and M. S. Boyce. 2009. Use of resource selection functions to identify conservation corridors. Journal of Applied Ecology 46:1036–1047.

Collins, W. B., and P. J. Urness. 1983. Feeding behavior and habitat selection of mule deer and elk on northern Utah summer range. The Journal of Wildlife Management 47:646–663.

Conner, M. M., T. R. Stephenson, D. W. German, K. L. Monteith, A. P. Few, and E. H. Bair. 2018. Survival analysis: Informing recovery of Sierra Nevada bighorn sheep. Journal of Wildlife Management 82:1442–1458.

Cooley, H. S., H. S. Robinson, R. B. Wielgus, and C. S. Lambert. 2008. Cougar prey selection in a white-tailed deer and mule deer community. Journal of Wildlife Management 72:99–106.

Creel, S., J. Winnie, B. Maxwell, K. Hamlin, and M. Creel. 2005. Elk alter habitat selection as an antipredator response to wolves. Ecology 86:3387–3397.

Cristescu, B., G. B. Stenhouse, and M. S. Boyce. 2015a. Grizzly bear diet shifting on reclaimed mines. Global Ecology and Conservation 4:207–220.

Cristescu, B., G. B. Stenhouse, and M. S. Boyce. 2015b. Large omnivore movements in response to surface mining and mine reclamation. Scientific Reports 6:1–10.

Cristescu, B., G. B. Stenhouse, M. Symbaluk, S. E. Nielsen, and M. S. Boyce. 2016. Wildlife habitat selection on landscapes with industrial disturbance. Environmental Conservation 43:327–336.

Dehn, M. M. 1990. Vigilance for predators: detection and dilution effects. Behavioral Ecology and Sociobiology 26:337–342.

Dellinger, J. A., E. R. Loft, R. C. Bertram, D. L. Neal, M. W. Kenyon, and S. G. Torres. 2018. Seasonal spatial ecology of mountain lions (*Puma concolor*) in the central Sierra Nevada. Western North American Naturalist 78:143–156.

DeVore, R. M., M. J. Butler, and M. C. Wallace. 2016. Elk resource selection patterns in a semiarid riparian corridor. The Journal of Wildlife Management 80:479–489.

Dickson, B. G., and P. Beier. 2002. Home-range and habitat selection by adult cougars in southern California. Journal of Wildlife Management 66:1235–1245.

Elbroch, L. M. 2006. Animal skulls: A guide to North American species. Stackpole Books, Mechanicsburg, PA, USA.

Elbroch, L. M., and H. U. Wittmer. 2013. The effects of puma prey selection and specialization on less abundant prey in Patagonia. Journal of Mammalogy 94:259–268.

Elbroch, L. M., P. E. Lendrum, J. Newby, H. Quigley, and D. Craighead. 2013. Seasonal foraging ecology of non-migratory cougars in a system with migrating prey. PLoS ONE 8:1–14.

Elliott, C. L., and J. D. McKendrick. 1984. Food habits of Dall sheep on revegetated coal stripmine spoils in Alaska. Biennial Symposium of the Northern Wild Sheep and Goat Council 3:241–251.

Environmental Systems Research Institute. 2017. ArcGIS Desktop: Release 10.5. Redlands, California.

Estes, J. A., M. L. Riedman, M. M. Staedler, M. T. Tinker, and B. E. Lyon. 2003. Individual variation in prey selection by sea otters: Patterns, causes and implications. Journal of Animal Ecology 72:144–155.

Feldhamer, G. A., B. C. Thompson, and J. A. Chapman. 2003. Wild mammals of North America: Biology, management, and conservation. Second edition. John Hopkins University Press, Baltimore, Maryland, USA.

Festa-Bianchet, M., T. Coulson, J.-M. Gaillard, J. T. Hogg, and F. Pelletier. 2006. Stochastic predation events and population persistence in bighorn sheep. Proceedings of the Royal Society B: Biological Sciences 273:1537–1543.

Fortin, D., H. L. Beyer, M. S. Boyce, D. W. Smith, T. Duchesne, and J. S. Mao. 2005. Wolves influence elk movements: Behavior shapes a trophic cascade in Yellowstone National Park. Ecology 86:1320–1330.

Geist, V. 1971. Mountain sheep: A study in behaviour and evolution. University Chicago Press, Chicago, IL.

Geist, V., and R. G. Petocz. 1977. Bighorn sheep in winter: Do rams maximize reproductive fitness by spatial and habitat segregation from ewes? Canadian Journal of Zoology 55:1802–1810.

Government of Alberta. 2012. Management plan for cougars in Alberta. Pages 1–73. Wildlife Management Branch, Edmonton, Alberta.

Government of Alberta. 2013. Luscar & Gregg River Mines land management plan. Pages 1–22. Alberta Environment and Sustainable Resource Development, Edmonton, Alberta.

Government of Alberta. 2015. Management plan for bighorn sheep in Alberta (draft). Pages 1–137. Wildlife Management Branch, Edmonton, Alberta.

Harper, K. A., S. E. MacDonald, P. J. Burton, J. Chen, K. D. Brosofske, S. C. Saunders, E. S. Euskirchen, D. Roberts, M. S. Jaiteh, and P.-A. Esseen. 2005. Edge influence on forest structure and composition in fragmented landscapes. Conservation Biology 19:768–782.

Harrison, S., and D. M. Hebert. 1988. Selective bighorn predation by cougar within the Junction Wildlife Management Area. Pages 292–306 *in*.

Hayes, C. L., E. S. Rubin, M. C. Jorgensen, R. A. Botta, and W. M. Boyce. 2000. Mountain lion predation of bighorn sheep in the Peninsular Ranges, California. Journal of Wildlife Management 64:954–959.

Heath, D. R., D. A. Saugey, and G. A. Heidt. 1986. Abandoned mine fauna of the Ouachita Mountains, Arkansas: Vertebrate taxa. Journal of the Arkansas Academy of Science 40:33–36.

Hebblewhite, M., and D. Pletscher. 2002. Effects of elk group size on predation by wolves. Canadian Journal of Zoology 80:800–809.

Hebblewhite, M., and E. H. Merrill. 2007. Multiscale wolf predation risk for elk: Does migration reduce risk? Oecologia 152:377–387.

Hebblewhite, M., C. A. White, C. G. Nietvelt, J. A. McKenzie, T. E. Hurd, J. M. Fryxell, S. E. Bayley, and P. C. Paquet. 2005. Human activity mediates a trophic cascade caused by wolves. Ecology 86:2135–2144.

Hicks, L. L., and J. M. Elder. 1979. Human disturbance of Sierra-Nevada bighorn sheep. Journal of Wildlife Management 43:909–915.

Hingtgen, T. M., and W. R. Clark. 1984. Small mammal recolonization of reclaimed coal surface-mined land in Wyoming. The Journal of Wildlife Management 48:1255–1261.

Hobbs, N. T., D. L. Baker, and R. B. Gill. 1983. Comparative nutritional ecology of montane ungulates during winter. The Journal of Wildlife Management 47:1.

Holl, K. D., and J. Cairns. 1994. Vegetational community development on reclaimed coal surface mines in Virginia. Bulletin of the Torrey Botanical Club 121:327–337.

Holl, S. A., V. C. Bleich, and S. G. Torres. 2004. Population dynamics of bighorn sheep in the San Gabriel Mountains, California, 1967-2002. Wildlife Society Bulletin 32:412–426.

Holmes, B. R., and J. W. Laundre. 2006. Use of open, edge and forest areas by pumas *Puma concolor* in winter: Are pumas foraging optimally? Wildlife Biology 12:201–209.

Hornocker, M. G. 1970. An analysis of mountain lion predation upon mule deer and elk in the Idaho Primitive Area. Wildlife Monographs 21:3–39.

Hudson, R. J., D. M. Hebert, and V. C. Brink. 1976. Occupational patterns of wildlife on a major east Kootenay winter-spring range. Journal of Range Management 29:38.

Husseman, J. S., D. L. Murray, G. Power, C. Mack, C. R. Wenger, and H. Quigley. 2003. Assessing differential prey selection patterns between two sympatric large carnivores. Oikos 101:591–601.

Ihl, C., and D. R. Klein. 2001. Habitat and diet selection by muskoxen and reindeer in western Alaska. The Journal of Wildlife Management 65:964.

Jackman, S. 2017. pscl: classes and methods for R developed in the Political Science Computational Laboratory. Sydney, New South Wales, Australia.

Jansen, B. D., P. R. Krausman, J. R. Heffelfinger, and J. C. deVos Jr. 2006. Bighorn sheep selection of landscape features in an active copper mine. Wildlife Society Bulletin 34:1121–1126.

Jansen, B. D., P. R. Krausman, K. D. Bristow, J. R. Heffelfinger, and J. C. deVos Jr. 2009. Surface mining and ecology of desert bighorn sheep. The Southwestern Naturalist 54:430–438.

Javed, A., and I. Khan. 2012. Land use/land cover change due to mining activities in Singrauli Industrial Belt, Madhya Pradesh using remote sensing and GIS. Journal of Environmental Research and Development 6:834–843.

Jedrezejewski, W., B. Jedrezejewska, H. Okarma, and A. L. Ruprecht. 1992. Wolf predation and snow cover as mortality factors in the ungulate community of the Bialowieza-National-Park, Poland. Oecologia 90:27–36.

Jenness, J. 2006. Topographic position index (tpi_jen.avx) extension for ArcView 3.x, v. 1.2. Jenness Enterprises.

Jennings, M. K., R. L. Lewison, T. W. Vickers, and W. M. Boyce. 2016. Puma response to the effects of fire and urbanization. Journal of Wildlife Management 80:221–234.

Johnson, B. K., J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of mule deer and elk during spring. Journal of Wildlife Management 64:685–697.

Johnson, C. J., S. E. Nielsen, E. H. Merrill, T. L. McDonald, and M. S. Boyce. 2006. Resource selection functions based on use–availability data: Theoretical motivation and evaluation methods. Journal of Wildlife Management 70:347–357.

Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65–71.

Johnson, H. E., M. Hebblewhite, T. R. Stephenson, D. W. German, B. M. Pierce, and V. C. Bleich. 2013. Evaluating apparent competition in limiting the recovery of an endangered ungulate. Oecologia 171:295–307.

Johnson, M. S., P. D. Putwain, and R. J. Holliday. 1978. Wildlife conservation value of derelict metalliferous mine workings in Wales. Biological Conservation 14:131–148.

Jones, M. E., and N. Davidson. 2016. Applying an animal-centric approach to improve ecological restoration. Restoration Ecology 24:836–842.

Jones, R. L., and H. C. Hanson. 1985. Mineral licks, geophagy, and biogeochemistry of North American ungulates. The Iowa State University Press, Iowa City.

Jung, T. S., S. A. Stotyn, and S. M. Czetwertynski. 2015. Dietary overlap and potential competition in a dynamic ungulate community in Northwestern Canada. The Journal of Wildlife Management 79:1277–1285.

Kauffman, M. J., N. Varley, D. W. Smith, D. R. Stahler, D. R. MacNulty, and M. S. Boyce. 2007. Landscape heterogeneity shapes predation in a newly restored predator-prey system. Ecology Letters 10:690–700.

Kertson, B. N., R. D. Spencer, J. M. Marzluff, J. Hepinstall-Cymerman, and C. E. Grue. 2011. Cougar space use and movements in the wildland–urban landscape of western Washington. Ecological Applications 21:2866–2881.

Kneteman, J. 2016. Resilient space: Bighorn sheep (*Ovis canadensis*) ecological resilience in the Northern Rocky Mountains. Pages 1–156 MSc, University of Alberta.

Knopff, A. A., K. H. Knopff, M. S. Boyce, and C. C. St Clair. 2014a. Flexible habitat selection by cougars in response to anthropogenic development. Biological Conservation 178:136–145.

Knopff, K. H., A. A. Knopff, A. Kortello, and M. S. Boyce. 2010a. Cougar kill rate and prey composition in a multiprey system. Journal of Wildlife Management 74:1435–1447.

Knopff, K. H., A. A. Knopff, and M. S. Boyce. 2010b. Scavenging makes cougars susceptible to snaring at wolf bait stations. Journal of Wildlife Management 74:644–653.

Knopff, K. H., A. A. Knopff, M. B. Warren, and M. S. Boyce. 2009. Evaluating global positioning system telemetry techniques for estimating cougar predation parameters. Journal of Wildlife Management 73:586–597.

Knopff, K. H., and M. S. Boyce. 2007. Prey specialization by individual cougars in multiprey systems. Pp. 194-210 of Transactions of the 72nd North American Wildlife and Natural Resources Conference, Portland, Oregon.

Knopff, K. H., N. F. Webb, and M. S. Boyce. 2014b. Cougar population status and range expansion in Alberta during 1991-2010. Wildlife Society Bulletin 38:116–121.

Kuck, L., G. L. Hompland, and E. H. Merrill. 1985. Elk calf response to simulated mine disturbance in southeast Idaho. Journal of Wildlife Management 49:751–757.

Kunkel, K. E., and D. H. Pletscher. 2000. Habitat factors affecting vulnerability of moose to predation by wolves in southeastern British Columbia. Canadian Journal of Zoology 78:150–157.

Kunkel, K., and D. H. Pletscher. 2001. Winter hunting patterns of wolves in and near Glacier National Park, Montana. Journal of Wildlife Management 65:520–530.

Ladle, A., T. Avgar, M. Wheatley, G. B. Stenhouse, S. E. Nielsen, and M. S. Boyce. 2019. Grizzly bear response to spatio-temporal variability in human recreational activity. Journal of Applied Ecology 56:375–386.

Larkin, J. L., D. S. Maehr, J. J. Krupa, J. J. Cox, K. Alexy, D. E. Unger, and C. Barton. 2008. Small mammal response to vegetation and spoil conditions on a reclaimed surface mine in eastern Kentucky. Southeastern Naturalist 7:401–412.

Laundre, J. W., and J. Loxterman. 2006. Impact of edge habitat on summer home range size in female pumas. The American Midland Naturalist 157:221–229.

Laundre, J. W., and L. Hernandez. 2003. Winter hunting habitat of pumas *Puma concolor* in northwestern Utah and southern Idaho, USA. Wildlife Biology 9:123–129.

Laundre, J. W., L. Hernandez, and S. G. Clark. 2007. Numerical and demographic responses of pumas to changes in prey abundance: Testing current predictions. Journal of Wildlife Management 71:345–355.

Laundre, J. W., L. Hernandez, D. Streubel, K. Altendorf, and C. L. Gonzalez. 2000. Aging mountain lions using gum-line recession. Wildlife Society Bulletin 28:963–966.

Lehman, C. P., C. T. Rota, M. A. Rumble, and J. J. Millspaugh. 2017. Characteristics of successful puma kill sites of elk in the Black Hills, South Dakota. Wildlife Biology 1:1–10.

Lendrum, P. E., C. R. J. Anderson, R. A. Long, J. G. Kie, and R. T. Bowyer. 2012. Habitat selection by mule deer during migration: Effects of landscape structure and natural-gas development. Ecosphere 3:–19.

Lingle, S. 2002. Coyote predation and habitat segregation of white-tailed deer and mule deer. Ecology 83:2037–2048.

Lingle, S., A. Feldman, M. S. Boyce, and W. F. Wilson. 2008. Prey behavior, age-dependent vulnerability, and predation rates. American Naturalist 172:712–725.

Logan, K. A., and L. L. Irwin. 1985. Mountain lion habitats in the Big Horn Mountains, Wyoming. Wildlife Society Bulletin 13:257–262.

Lowrey, B. H., L. M. Elbroch, and L. Broberg. 2016. Is individual prey selection driven by chance or choice? A case study in cougars (*Puma concolor*). Mammal Research 61:1–9.

MacArthur, R. A., V. Geist, and R. H. Johnston. 1982. Physiological correlates of social behaviour in bighorn sheep: A field study using electrocardiogram telemetry. Journal of Zoology 196:401–415.

MacCallum, B. N. 1989. Seasonal and spatial distribution of bighorn sheep at an open pit coal mine in the Alberta foothills. Journal American Society of Mining and Reclamation 1989:141–150.

MacCallum, B. N., and V. Geist. 1992. Mountain restoration: Soil and surface wildlife habitat. GeoJournal 27:23–46.

Manly, B. F., L. McDonald, D. Thomas, T. L. Mcdonald, and W. P. Erickson. 2002. Resource selection by animals: Statistical design and analysis for field studies. Kluwer, Dordrecht, The Netherlands.

Manning, A. D., R. B. Cunningham, and D. B. Lindenmayer. 2013. Bringing forward the benefits of coarse woody debris in ecosystem recovery under different levels of grazing and vegetation density. Biological Conservation 157:204–214.

Mao, J. S., M. S. Boyce, D. W. Smith, F. J. Singer, D. J. Vales, J. M. Vore, and E. H. Merrill. 2005. Habitat selection by elk before and after wolf reintroduction in Yellowstone National Park. Journal of Wildlife Management 69:1691–1707.

McDermid, G. J. 2005. Remote sensing for large-area, multi-jurisdictional habitat mapping. PhD Dissertation, University of Waterloo, Waterloo, Ontario.

McDonald, J. H. 2015. Handbook of biological statistics. Third Edition. Sparky House Publishing, Baltimore, Maryland.

McPhee, H. M., N. F. Webb, and E. H. Merrill. 2012. Hierarchical predation: Wolf (*Canis lupus*) selection along hunt paths and at kill sites. Canadian Journal of Zoology 90:555–563.

Merkle, J. A., K. L. Monteith, E. O. Aikens, M. M. Hayes, K. R. Hersey, A. D. Middleton, B. A. Oates, H. Sawyer, B. M. Scurlock, and M. J. Kauffman. 2016. Large herbivores surf waves of green-up during spring. Proceedings of the Royal Society B: Biological Sciences 283:1–8.

Morrison, C. D., M. S. Boyce, S. E. Nielsen, and M. M. Bacon. 2014. Habitat selection of a re-colonized cougar population in response to seasonal fluctuations of human activity. Journal of Wildlife Management 78:1394–1403.

Moss, W. E., M. W. Alldredge, K. A. Logan, and J. N. Pauli. 2016. Human expansion precipitates niche expansion for an opportunistic apex predator (*Puma concolor*). Scientific Reports 6:1–5.

Müller, A., M. Dahm, P. K. Bøcher, M. Root-Bernstein, and J.-C. Svenning. 2017. Large herbivores in novel ecosystems - habitat selection by red deer (*Cervus elaphus*) in a former brown-coal mining area. PLoS ONE 12:1–20.

Namgail, T., S. E. Van Wieren, C. Mishra, and H. H. T. Prins. 2010. Multi-spatial co-distribution of the endangered Ladakh urial and blue sheep in the arid Trans-Himalayan mountains. Journal of Arid Environments 74:1162–1169.

Natural Regions Committee. 2006. Natural regions and subregions of Alberta. Pages Pub. No. T–852. Government of Alberta.

Neal, D. L., G. N. Steger, and R. C. Bertram. 1987. Mountain lions: Preliminary findings on home-range use and density in central Sierra Nevada. Pages 1–6. United States Department of Agriculture Forest Service, Berkeley, California.

Oehler, M. W., V. C. Bleich, and R. T. Bowyer. 2005. Mountain sheep and mining: Implications for conservation and management. California Fish and Game 91:149–178.

Ordiz, A., O.-G. Stoen, M. Delibes, and J. E. Swenson. 2011. Predators or prey? Spatio-temporal discrimination of human-derived risk by brown bears. Oecologia 166:59–67.

Peipoch, M., M. Brauns, F. R. Hauer, M. Weitere, and H. M. Valett. 2015. Ecological simplification: Human influences on riverscape complexity. BioScience 65:1057–1065.

Pelletier, F., Y. Gendreau, and C. Feder. 2006. Behavioural reactions of bighorn sheep (*Ovis canadensis*) to cougar (*Puma concolor*) attacks. Mammalia 70:160–162.

Pierce, B. M., R. T. Bowyer, and V. C. Bleich. 2004. Habitat selection by mule deer: Forage benefits or risk of predation? Journal of Wildlife Management 68:533–541.

Pierce, B. M., V. C. Bleich, J. D. Wehausen, and R. Terry Bowyer. 1999. Migratory patterns of mountain lions: Implications for social regulation and conservation. Journal of Mammalogy 80:986–992.

Poole, K. G., R. Serrouya, I. E. Teske, and K. Podrasky. 2016. Rocky Mountain bighorn sheep (*Ovis canadensis canadensis*) winter habitat selection and seasonal movements in an area of active coal mining. Canadian Journal of Zoology 94:733–745.

Province of Alberta. 2000. Environmental Protection and Enhancement Act, RSA 2000, c E-12.

R Core Team. 2018. R: A language and environment for statistical computing. Vienna, Austria.

Reale, D. 2003. Predator-induced natural selection on temperament in bighorn ewes. Animal Behaviour 65:463–470.

Riley, S. J., S. D. DeGloria, and R. Elliot. 1999. A terrain ruggedness index that quantifies topographic heterogeneity. Intermountain Journal of Sciences 5:23–27.

Roever, C. L., M. S. Boyce, and G. B. Stenhouse. 2008. Grizzly bears and forestry - I: Road vegetation and placement as an attractant to grizzly bears. Forest Ecology and Management 256:1253–1261.

Rominger, E. M. 2018. The Gordian knot of mountain lion predation and bighorn sheep. The Journal of Wildlife Management 82:19–31.

Rominger, E. M., H. A. Whitlaw, D. L. Weybright, W. C. Dunn, and W. B. Ballard. 2004. The influence of mountain lion predation on bighorn sheep translocations. Journal of Wildlife Management 68:993–999.

Ross, P. I., M. G. Jalkotzy, and M. Festa-Bianchet. 1997. Cougar predation on bighorn sheep in southwestern Alberta during winter. Canadian Journal of Zoology 75:771–775.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. The Journal of Wildlife Management 64:672–684.

Sawyer, H., and F. Lindzey. 2002. A review of predation on bighorn sheep (*Ovis canadensis*). Wyoming Cooperative Fish and Wildlife Research Unit, Laramie, WY.

Sawyer, H., M. J. Kauffman, and R. M. Nielson. 2009. Influence of well pad activity on winter habitat selection patterns of mule deer. Journal of Wildlife Management 73:1052–1061.

Sawyer, H., R. M. Nielson, F. G. Lindzey, L. Keith, J. H. Powell, and A. A. Abraham. 2007. Habitat selection of Rocky Mountain elk in a nonforested environment. Journal of Wildlife Management 71:868–874.

Sawyer, H., R. M. Nielson, F. Lindzey, and L. L. McDonald. 2006. Winter habitat selection of mule deer before and during development of a natural gas field. Journal of Wildlife Management 70:396–403.

Schmidt, K., and D. P. J. Kuijper. 2015. A "death trap" in the landscape of fear. Mammal Research 60:275–284.

Schueler, V., T. Kuemmerle, and H. Schroder. 2011. Impacts of surface gold mining on land use systems in western Ghana. AMBIO 40:528–539.

Serrouya, R., and R. G. D'Eon. 2008. The influence of forest cover on mule deer habitat selection, diet, and nutrition during winter in a deep-snow ecosystem. Forest Ecology and Management 256:452–461.

Shannon, G., L. S. Cordes, A. R. Hardy, L. M. Angeloni, and K. R. Crooks. 2014. Behavioral responses associated with a human-mediated predator shelter. PLoS ONE 9:1–8.

Shaw, H. G., P. Beier, M. Culver, and M. Grigione. 2007. Puma field guide: A guide covering the biological considerations, general life history, identification, assessment, and management of *Puma concolor*. Pages 1–129. The Cougar Network.

Sheehy, D. P., and M. Vavra. 1996. Ungulate foraging areas on seasonal rangeland in northeastern Oregon. Journal of Range Management 49:16–23.

Sherwin, R. E., D. Stricklan, and D. S. Rogers. 2000. Roosting affinities of Townsend's big-eared bat (*Corynorhinus townsendii*) in northern Utah. Journal of Mammalogy 81:939–947.

Signer, J., J. Fieberg, and T. Avgar. 2019. Animal movement tools (amt): R package for managing tracking data and conducting habitat selection analyses. Ecology and Evolution 9:880–890.

Sinclair, A. R. E. 1985. Does interspecific competition or predation shape the African ungulate community? The Journal of Animal Ecology 54:899.

Singer, F. J., V. C. Bleich, and M. A. Gudorf. 2000. Restoration of bighorn sheep metapopulations in and near western national parks. Restoration Ecology 8:14–24.

Slankard, K. G., D. L. Baxley, and G. L. Sprandel. 2018. The impacts of native-grassland restoration on raptors and their prey on a reclaimed surface mine in Kentucky. Northeastern Naturalist 25:277–290.

Smith, J. B., J. A. Jenks, T. W. Grovenburg, and R. W. Klaver. 2014. Disease and predation: Sorting out causes of a bighorn sheep (*Ovis canadensis*) decline. PLoS ONE 9:1–9.

Smith, T. S., J. T. Flinders, and D. S. Winn. 1991. Habitat evaluation procedure for Rocky Mountain bighorn sheep in the Intermountain West. Great Basin Naturalist 51:205–225.

Soria-Diaz, L., M. S. Fowler, O. Monroy-Vilchis, and D. Oro. 2018. Functional responses of cougars (*Puma concolor*) in a multiple prey-species system. Integrative Zoology 13:84–93.

Stauffer, G. E., D. R. Diefenbach, M. R. Marshall, and D. W. Brauning. 2011. Nest success of grassland sparrows on reclaimed surface mines. Journal of Wildlife Management 75:548–557.

Stewart, K. M., R. T. Bowyer, J. G. Kie, and M. A. Hurley. 2010. Spatial distributions of mule deer and North American elk: Resource partitioning in a sage-steppe environment. The American Midland Naturalist 163:400–412.

Stewart, K. M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. Journal of Mammalogy 83:229–244.

Strong, W. L. 2002. Enhancing botanical diversity on minesoils: An *a posteriori* assessment. International Journal of Surface Mining, Reclamation and Environment 16:85–96.

Strong, W. L., and K. R. Leggat. 1992. Ecoregions of Alberta.

Swab, R. M., N. Lorenz, S. Byrd, and R. Dick. 2017. Native vegetation in reclamation: Improving habitat and ecosystem function through using prairie species in mine land reclamation. Ecological Engineering 108:525–536.

Telfer, E. S. 1978. Cervid distribution, browse and snow cover in Alberta. The Journal of Wildlife Management 42:352.

Thomas, L., S. T. Buckland, E. A. Rexstad, J. L. Laake, S. S, S. L. Hedley, J. R. B. Bishop, T. A. Marques, and K. P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. The Journal of Applied Ecology:5–14.

Torstenson, W., J. C. Mosley, T. K. Brewer, M. W. Tess, and J. E. Knight. 2006. Elk, mule deer, and cattle foraging relationships on foothill and mountain rangeland. Rangeland Ecology & Management 59:80–87.

Turner, J. W. J., M. L. Wolfe, and J. F. Kirkpatrick. 1992. Seasonal mountain lion predation on a feral horse population. Canadian Journal of Zoology 70:929–934.

Unsworth, J. W., L. Kuck, E. O. Garton, and B. R. Butterfield. 1998. Elk habitat selection on the Clearwater National Forest, Idaho. Journal of Wildlife Management 62:1255–1263.

Vila, A. R., L. Borrelli, and L. Martínez. 2009. Dietary overlap between huemul and livestock in Los Alerces National Park, Argentina. Journal of Wildlife Management 73:368–373.

Webb, S. L., M. R. Dzialak, K. L. Kosciuch, and J. B. Winstead. 2013. Winter resource selection by mule deer on the Wyoming–Colorado border prior to wind energy development. Rangeland Ecology & Management 66:419–427.

Wehausen, J. D. 1996. Effects of mountain lion predation on bighorn sheep in the Sierra Nevada and Granite Mountains of California. Wildlife Society Bulletin 24:471–479.

Weir, J. N., S. P. Mahoney, B. McLaren, and S. H. Ferguson. 2009. Effects of mine development on woodland caribou *Rangifer tarandus* distribution. Wildlife Biology 13:66–74.

Whitaker, J. O., and L. J. Rissler. 1992. Winter activity of bats at a mine entrance in Vermillion County, Indiana. American Midland Naturalist 127:52–59.

Wickham, J., K. H. Riitters, T. G. Wade, M. Coan, and C. Homer. 2006. The effect of Appalachian mountaintop mining on interior forest. Landscape Ecology 22:179–187.

Wickham, J., P. Bohall Wood, M. C. Nicholson, W. Jenkins, D. Druckenbrod, G. W. Suter, M. P. Strager, C. Mazzarella, W. Galloway, and J. Amos. 2013. The overlooked terrestrial impacts of mountaintop mining. BioScience 63:335–348.

Wiedmann, B. P., and V. C. Bleich. 2014. Demographic responses of bighorn sheep to recreational activities: A trial of a trail. Wildlife Society Bulletin 38:773–782.

Wilmers, C. C., Y. Wang, B. Nickel, P. Houghtaling, Y. Shakeri, M. L. Allen, J. Kermish-Wells, V. Yovovich, and T. Williams. 2013. Scale dependent behavioral responses to human development by a large predator, the puma. PLoS ONE 8:1–11.

Yearsley, E. F., and D. E. Samuel. 1980. Use of reclaimed surface mines by foxes in west Virginia. The Journal of Wildlife Management 44:729–734.

APPENDIX 2.1 – FURTHER DETECTABILITY ANALYSES

Rationale

Our original goal was to not only examine detectability of ungulate groups and select an appropriate right-truncation distance at which to remove outlying observations, but to inflate ungulate group counts such that group counts accounted for detectability biases that exists when observing animals at large distances. Appendix 2.1 summarizes the rationale, methods, findings, and a brief discussion of results from this original goal.

Methods

Following the methods described in Chapter 2, we proceeded to fit several potential models to each of the six datasets, after selecting an appropriate right-truncation distance. We examined model outputs, histograms, and quantile-quantile plots for model robustness, shape, and estimator efficiency criteria. We selected a top model based on small sampled size corrected Akaike's Information Criterion (AICc) (Burnham and Anderson 2002) and goodness-of-fit (GOF) results, using χ^2 GOF test and Kolmogorov-Smirnov GOF test (Appendix 2.1 – Tables 1-6, Appendix 2.1 – Figure 1). When several competing models were possible (ΔAICc < 4.0) (Burnham and Anderson 2002), we selected a top model with the lowest AICc that also met both GOF requirements and produced biologically reasonable inflated ungulate counts. Using the distance assigned to each sampling unit, we calculated the probability of detection $g(y)$, at a given distance y:

$$g(y) = \frac{\text{key}(y)[1 + \text{series}(y_s)]}{\text{key}(0)[1 + \text{series}(0)]}$$

where $key(y)$ is the key function and $series(y_i)$ is the series expansion. We inflated

ungulate counts within sampling units using calculated probabilities of detection $g(y)$

and mean distances y assigned to sampling units:

$$\text{Inflated \# individuals} = \frac{\text{Observed \# individuals}}{g(y)}$$

We chose not to extrapolate detection functions past truncation distances, as we felt this

created less reliable model estimates and biologically unreasonable inflation to ungulate

count data. When a species' detection function in open habitat was truncated at a

different distance than it's detection function in forest habitat, we inflated the species'

overall counts with the smaller of the two truncation distances (Ch 2 – Table 2.2). Within

remaining sampling units, we summed each species' counts inflated by the open

detectability analysis with the counts inflated by the forest detectability analysis. The sum

of counts represents the final inflated number of individuals, per species, using a

particular sampling unit over the span of a season.

We assessed the role of topography, using digital elevation models (DEMs), in

ungulate detectability using the Viewshed tool in ArcGIS Spatial Analyst (Environmental

Systems Research Institute 2017). The Viewshed tool uses a DEM and an observer survey

route as input to calculate areas in a specific landscape that are visible and non-visible to

an observer from the survey route, based on topography. Due to substantial

topographical changes in our study area during the 14-year study period, we completed a

Viewshed analysis using a separate DEM for each year. Using Viewshed results, we

calculated the proportion of each sampling unit that was visible, based on the total area

of each sampling unit (40 000 m^2). We then calculated Pearson's correlation coefficient to compare the relationship between proportion of sampling unit visible and sampling unit probability of detection, from detectability analyses, using data pooled over years. Because we only applied detection functions within truncation distances to calculate probability of detection, we examined the role of topography within truncation distances only. We hypothesized that topography would affect visibility, and, therefore, influence detectability as observers cannot detect ungulate groups behind hills, cliffs, and mountains. The results of these correlations will provide insight on the role topography plays in predicting detectability or whether alternative factors, such as habitat type, group size, and species may influence detectability of ungulates in our study area.

Results

Between 2004 and 2017, we completed a 159-km fixed survey route with 89 transects, 57 times ($\bar{x}$ = 4.1 surveys per year). The observed number of individuals in a group ranged from 1 − 181 ($\bar{x}$ = 12.1, SD = 15.3) bighorn sheep, 1 − 293 ($\bar{x}$ = 14.0, SD = 24.5) elk, and 1 − 32 ($\bar{x}$ = 2.9, SD = 2.5) mule deer. Inflated sums of individuals per sampling unit ranged from 0 − 1 148 bighorn sheep, 0 − 1 313 elk, and 0 − 101 mule deer, within set truncation distances.

Overall, the estimated average detection probability varied depending on the species and habitat (Appendix 2.1 − Figure 2). Average detection probability was highest for elk (open habitat: 0.5267, 95% CI = 0.5010 − 0.5537; forest habitat: 0.5523, 95% CI = 0.4771 − 0.6393) and lowest for mule deer (open habitat: 0.3815, 95% CI = 0.3472 − 0.4193; forest habitat: 0.3059, 95% CI = 0.2567 − 0.3644). Average detection probability

for bighorn sheep was 0.4831 (95% CI = 0.4634 – 0.5035) in open habitat and 0.3743

(95% CI = 0.2881 – 0.4863) in forest habitat. Detection was higher in open than forest

habitat for both bighorn sheep and mule deer. In contrast, detection was higher for elk in

forest than open habitat.

We conducted Pearson's product-moment correlations in R (R Core Team 2018)

between visibility and probability of detection, for each species in open and forest

habitat. Visibility and probability of detection were weakly positively correlated for all

species, in both open and forest habitats (Appendix 2.1 – Table 7). Strength of correlation

was similar between species and observed habitats (Appendix 2.1 – Table 7).

We modelled intensity of habitat use using inflated, then log-transformed,

number of individuals per group as our continuous response variable and landscape

covariates as our predictor variables, pooled over all sampling units and years. We

included season as a covariate in all models, including the null model, to account for

known seasonal variation in group size (Appendix 2.5), but we did not directly test

hypotheses related to seasonal influence on group size. The top model for bighorn sheep

intensity of habitat use using non-inflated counts included relative topographic position,

terrain ruggedness, and distance to haul road (Appendix 2.1 – Table 8). The top model

using inflated counts presented an entirely different suite of covariates: proportion edge

habitat, distance to other grassland, and distance to haul road (Appendix 2.1 – Table 8).

Both models were equally parsimonious based on number of parameters, but the model

using inflated counts produced a lower r^2 value and larger p-value, indicating lower

predictive capability. The top model for elk intensity of habitat use using non-inflated

counts included distance to disturbed areas, distance to haul road, relative topographic position, and proportion edge habitat (Appendix 2.1 – Table 8). Similarly, the top model using inflated counts contained these four covariates and one additional covariate: distance to forest. As such, the model using inflated counts was slightly more complex and also had the same predictive capability as a model using non-inflated counts. Lastly, the top model for mule deer intensity of habitat use using non-inflated counts included only one covariate: proportion of edge habitat (Appendix 2.1 – Table 8). The top model using inflated counts was slightly more complex and contained distance to disturbed areas and terrain ruggedness, in addition to proportion of edge habitat. Modelling with non-inflated and inflated counts provided essentially equal predictive capabilities for mule deer.

Discussion

Our goal was to adjust the number of individuals per ungulate group based on the pattern of detectability for each species in open and forest habitats, within reliable observation distances (i.e. inside the truncation distance). Despite selecting a top detection function per detectability analysis, with the foremost goal being biologically reasonable inflated counts, we found that the inflated number of individuals per group was still biologically unreasonable. For example, we set the truncation distance for bighorn sheep at 300 meters and all bighorn groups observed in sampling units > 300 meters from the survey route were discarded prior to selecting a detection function to inflate counts by, and prior to any habitat modelling. However, the inflated bighorn sheep counts ranged from 1 individual up to 1,148 individuals per group. Upon further

reflection, we are inclined to suggest that a group of nearly 1,200 bighorn sheep is not biologically reasonable. Inflated counts for elk and mule deer are just as biologically unreasonable: 1,313 elk in the largest group, and 101 mule deer in the largest group.

However unreasonable these counts seem, conceptualizing inflated counts as a *relative* measure of abundance could still help to amplify ungulate responses to certain landscape features that might otherwise be masked. Thus, we modelled intensity of habitat use with inflated counts and compared these results with models using non-inflated counts. The results for bighorn sheep support that inflating counts provides a different estimate of the landscape features driving intensity of habitat use, as distance to main road is the only common covariate among both models. Nonetheless, we can be fairly certain that bighorn sheep congregate into larger groups, and hence use habitat more intensely, closer to main roads, as beta coefficient estimates are positive using non-inflated and inflated counts (Appendix 2.1 – Table 8). We cannot, however, be certain of the influence of edge habitat, RTP, or TRI. The predictive capability of models created with non-inflated was higher for bighorn sheep (r^2 = 0.18 for non-inflated model vs. r^2 = 0.14 for inflated model).

The results for elk and mule deer support our original reasoning that modelling with inflated counts may help to amplify responses to certain landscape features. For example, inflating elk counts amplifies an effect of distance to forest that was not present when modelling with non-inflated counts, but all other covariates remain the same between the two models (Appendix 2.1 – Table 8). Similarly, inflating mule deer counts amplifies effects of distance to disturbed areas and terrain ruggedness when estimating

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intensity of habitat use, while proportion of edge remains in both models (Appendix 2.1 –

Table 8). Moreover, mean r^2 and p-values from validating non-inflated and inflated elk

and mule deer models are the same (Appendix 2.1 – Table 8), indicating that inflating

counts provides equally reliable predictions as using original observations.

Overall, inflating counts may be a reasonable approach for some species, but not

others. For bighorn sheep, results are inconclusive. For elk and mule deer, however, our

results suggest that inflating counts may be a reasonable approach to elucidate species'

responses that may not be present otherwise. As a solution to these differing approaches

for each species, we opted not to adjust counts for our habitat modelling, as we feel that

inflating counts depends on weakly supported assumptions. Firstly, we argue that

truncating data at 300, 400, and 300 meters for bighorn sheep, elk, and mule deer,

respectively, ensures that we can rely on the accuracy of our original group counts. We

are confident that our defined truncation distances are conservative enough that

observers were able to count individuals consistently and reliably within these distances.

Secondly, predictive capability of models is equal, if not better, when using non-inflated

counts. Thirdly, we are confident that we sampled almost equally in open and forest

habitat types, which removes the possibility of our observations being biased to only one

habitat type. Lastly, by correlating probability of detection and visibility, we were able to

determine that visibility is weakly related to detection for all species. As such, we can

assume that detection in our study area is based on more than just visibility due to

topography. This particular result reduces the bias that ungulates could not be observed

during our study due to topography. Detection differences were more likely related to

observer bias and external factors, such as weather.

Appendix 2.1 – Table 1 Conventional distance sampling analyses results for correcting probability of bighorn sheep detection in open habitat (n = 4,282 observations post-truncation, survey length = 159 km, number of transects = 89, area sampled = 240 km^2, w = 300 m). Number of parameters, number of adjustment parameters, small sampled size corrected Akaike's Information Criterion (AICc), AICc difference (ΔAICc) and goodness-of-fit metrics are displayed. The top model is shaded. Results for top model are included in the second pane.

Competing models	No. of parameters	No. of adjustment parameters	AICc	ΔAICc	χ² GOF [a] p-value	K-S [b] GOF p-value
uni+cos	3	3	47128.00	0.00	0.490	0.789
hn+cos	4	3	47130.29	2.29	0.389	0.891
hr+simpoly	5	3	47132.57	4.57	0.346	0.848
hr	2	0	47139.91	11.91	0.059	0.065
hn	1	0	47176.17	48.17	0.000	0.001
uni+simpoly	2	2	47191.70	63.70	0.000	0.000

Parameter	Point Estimate	Standard Error	% Coefficient of Variation	95 % Confidence Interval	
A1 [c]	0.8728	0.0181			
A2 [d]	0.1628	0.0213			
A3 [d]	0.0345	0.0191			
f(0) [e]	0.6901 x 10^{-2}	0.1459 x 10^{-3}	2.11	0.6620 x 10^{-2}	0.7193 x 10^{-2}
p [f]	0.4831	0.0102	2.11	0.4634	0.5035
ESW [g]	144.9	3.065	2.11	139.0	151.1

[a] GOF = goodness-of-fit test
[b] Kolmogorov-Smirnov
[c] A1 = shape parameter for detection function
[d] A2, A3 = scale parameters for detection function
[e] f(0) = probability density function of detected distances, evaluated at distance 0
[f] p = average estimated probability of detection
[g] ESW = effective strip width, measured in meters

Appendix 2.1 – Table 2 Conventional distance sampling analyses results for correcting probability of bighorn sheep detection in forest habitat (n = 84 observations post-truncation, survey length = 159 km, number of transects = 89, area sampled = 243 km^2, w = 400 m). Number of parameters, number of adjustment parameters small sampled size corrected Akaike's Information Criterion (AICc), AICc difference (ΔAICc) and goodness-of-fit metrics are displayed. The top model is shaded. Results for top model are included in the second pane.

Competing models	No. of parameters	No. of adjustment parameters	AICc	ΔAICc	χ^2 GOF[a] p-value	K-S[b] GOF p-value
hr	2	0	942.59	0.00	0.454	0.953
hn+cos	3	2	943.64	1.05	0.484	0.863
uni+cos	2	2	944.13	1.53	0.437	0.388
hn	1	0	946.13	3.54	0.222	0.255
uni+simpoly	2	2	953.58	10.99	0.057	0.013

Parameter	Point Estimate	Standard Error	% Coefficient of Variation	95 % Confidence Interval	
A1[c]	101.4	21.51			
A2[d]	2.174	0.4940			
f(0)[e]	0.6679 x 10^{-2}	0.8828 x 10^{-3}	13.22	0.5104 x 10^{-2}	0.8678 x 10^{-2}
p[f]	0.3743	0.0495	13.22	0.2881	0.4863
ESW[g]	149.7	19.79	13.22	115.2	194.5

[a] GOF = goodness-of-fit test
[b] Kolmogorov-Smirnov
[c] A1 = shape parameter for detection function
[d] A2, A3 = scale parameters for detection function
[e] f(0) = probability density function of detected distances, evaluated at distance 0
[f] p = average estimated probability of detection
[g] ESW = effective strip width, measured in meters

Appendix 2.1 – Table 3 Conventional distance sampling analyses results for correcting probability of elk detection in open habitat (n = 958 observations post-truncation, survey length = 159 km, number of transects = 89, total area sampled = 240 km^2, w = 400 m). Number of parameters, number of adjustment parameters, small sampled size corrected Akaike's Information Criterion (AICc), AICc difference (ΔAICc) and goodness-of-fit metrics are displayed. The top model is shaded. Results for top model are included in the second pane.

Competing models	No. of parameters	No. of adjustment parameters	AICc	ΔAICc	χ^2 GOF [a] p-value	K-S [b] GOF p-value
hn	1	0	11091.55	0.00	0.736	0.890
uni+cos	2	2	11092.19	0.64	0.783	0.990
uni+simpoly	3	3	11093.79	2.24	0.740	0.989
hr+simpoly	4	2	11097.07	5.52	0.646	0.984
hr+cos	2	0	11098.54	6.99	0.419	0.762

Parameter	Point Estimate	Standard Error	% Coefficient of Variation	95 % Confidence Interval	
A1[c]	171.5	4.992			
f(0)[d]	0.4747 x 10^{-2}	0.1209 x 10^{-3}	2.55	0.4515 x 10^{-2}	0.4990 x 10^{-2}
p[e]	0.5267	0.0134	2.55	0.5010	0.5537
ESW[f]	210.7	5.368	2.55	200.4	221.5

[a] GOF = goodness-of-fit test

[b] Kolmogorov-Smirnov

[c] A1 = shape parameter for detection function

[d] f(0) = probability density function of detected distances, evaluated at distance 0

[e] p = average estimated probability of detection

[f] ESW = effective strip width, measured in meters

Appendix 2.1 – Table 4 Conventional distance sampling analyses results for correcting probability of elk detection in forest habitat (n = 55 observations post-truncation, survey length = 159 km, number of transects = 89, total area sampled = 243 km^2, w = 400 m). Number of parameters, number of adjustment parameters small sampled size corrected Akaike's Information Criterion (AICc), AICc difference (ΔAICc) and goodness-of-fit metrics are displayed. The top model is shaded. Results for top model are included in the second pane.

Competing models	No. of parameters	No. of adjustment parameters	AICc	ΔAICc	χ^2 GOF [a] p-value	K-S [b] GOF p-value
uni+cos	1	1	642.44	0.00	0.212	0.386
hn	1	0	642.79	0.35	0.197	0.405
hr	2	0	644.24	1.80	0.089	0.628
uni+simpoly	2	2	644.26	1.82	0.157	0.404

Parameter	Point Estimate	Standard Error	% Coefficient of Variation	95 % Confidence Interval	
A1 [c]	0.8108	0.1324			
f(0) [d]	0.4527 x 10^{-2}	0.3309 x 10^{-3}	7.31	0.3911 x 10^{-2}	0.5240 x 10^{-2}
p [e]	0.5523	0.0404	7.31	0.4771	0.6393
ESW [f]	220.9	16.15	7.31	190.8	255.7

[a] GOF = goodness-of-fit test
[b] Kolmogorov-Smirnov
[c] A1 = shape parameter for detection function
[d] f(0) = probability density function of detected distances, evaluated at distance 0
[e] p = average estimated probability of detection
[f] ESW = effective strip width, measured in meters

Appendix 2.1 – Table 5 Conventional distance sampling analyses results for correcting probability of mule deer detection in open habitat (n = 1,002 observations post-truncation, survey length = 159 km, number of transects = 89, area sampled = 243 km^2, w = 300 m). Number of parameters, number of adjustment parameters, small sampled size corrected Akaike's Information Criterion (AICc), AICc difference (ΔAICc) and goodness-of-fit metrics are displayed. The top model is shaded. Results for top model are included in the second pane.

Competing models	No. of parameters	No. of adjustment parameters	AICc	ΔAICc	χ^2 GOF [a] p-value	K-S [b] GOF p-value
hn+cos	4	3	11016.92	0.00	0.492	0.991
hr+cos	3	1	11017.20	0.28	0.446	0.852
uni+cos	5	5	11018.12	1.20	0.467	0.998
hr+simpoly	3	1	11018.25	1.33	0.353	0.634
uni+simpoly	5	5	11028.46	11.54	0.032	0.217
hn+hermite	1	0	11064.95	48.03	0.000	0.000

Parameter	Point Estimate	Standard Error	% Coefficient of Variation	95 % Confidence Interval	
A1 [c]	139.8	4.423			
A2 [d]	0.2707	0.0522			
A3 [d]	0.0410	0.0499			
A4 [d]	0.1695	0.0473			
f(0) [e]	0.8737 x 10^{-2}	0.4206 x 10^{-3}	4.81	0.7950 x 10^{-2}	0.9602 x 10^{-2}
p [f]	0.3815	0.0184	4.81	0.3472	0.4193
ESW [g]	114.5	5.510	4.81	104.1	125.8

[a] GOF = goodness-of-fit test
[b] Kolmogorov-Smirnov
[c] A1 = shape parameter for detection function
[d] A2, A3, A4 = scale parameters for detection function
[e] f(0) = probability density function of detected distances, evaluated at distance 0
[f] p = average estimated probability of detection
[g] ESW = effective strip width, measured in meters

Appendix 2.1 – Table 6 Conventional distance sampling analyses results for correcting probability of mule deer detection in forest habitat (n = 88 observations post-truncation, survey length = 159 km, number of transects = 89, area sampled = 240 km^2, w = 400 m). Number of parameters, number of adjustment parameters, small sampled size corrected Akaike's Information Criterion (AICc), AICc difference (ΔAICc) and goodness-of-fit metrics are displayed. The top model is shaded. Results for top model are included in the second pane.

Competing models	No. of parameters	No. of adjustment parameters	AICc	ΔAICc	χ^2 GOF [a] p-value	K-S [b] GOF p-value
hn+cos	2	1	966.77	0.00	0.060	0.682
hr	2	0	967.86	1.08	0.017	0.771
uni+cos	3	3	968.36	1.59	0.077	0.548
uni+simpoly	4	4	974.79	7.68	0.013	0.042
hn	1	0	980.13	13.45	0.004	0.006

Parameter	Point Estimate	Standard Error	% Coefficient of Variation	95 % Confidence Interval	
A1 [c]	1.296	0.1015			
A2 [d]	0.6348	0.1386			
A3 [d]	0.3389	0.1158			
f(0) [e]	0.8174 x 10^{-2}	0.7214 x 10^{-3}	8.83	0.6861 x 10^{-2}	0.9738 x 10^{-2}
p [f]	0.3059	0.0270	8.83	0.2567	0.3644
ESW [g]	122.3	10.80	8.83	102.7	145.8

[a] GOF = goodness-of-fit test
[b] Kolmogorov-Smirnov
[c] A1 = shape parameter for detection function
[d] A2, A3 = scale parameters for detection function
[e] f(0) = probability density function of detected distances, evaluated at distance 0
[f] p = average estimated probability of detection
[g] ESW = effective strip width, measured in meters

Appendix 2.1 – Table 7 Summary of Pearson's correlation coefficients between visibility and probability of detection, for bighorn sheep, elk, and mule deer, in open and forest habitat. 95% confidence intervals (95% CI) for Pearson's correlation coefficient are displayed.

Species	Habitat	Pearson's correlation coefficient	95% CI Lower	Upper
Bighorn sheep	Forest	0.164	0.152	0.176
	Open	0.172	0.161	0.183
Elk	Forest	0.163	0.152	0.174
	Open	0.165	0.154	0.176
Mule deer	Forest	0.169	0.158	0.180
	Open	0.166	0.154	0.178

Appendix 2.1 – Table 8 Top-ranked linear regression models for determining bighorn sheep, elk, and mule deer intensity of habitat use, modelled with non-inflated and inflated ungulate counts. Covariates, number of model parameters (K), 5-fold cross-validation r^2 values, and 5-fold cross-validation p-values are presented. Signs indicate the sign of the beta coefficient estimates.

Species	Count type	Covariates	K	$r^{2\ a}$	p-value[a]
Bighorn sheep	Non-inflated	season + dimroad + rtp – tri	8	0.18	< 0.0001
	Inflated	season + dimroad – pedge	7	0.14	< 0.01
Elk	Non-inflated	season – pedge + didist – dihroad + rtp	9	0.31	< 0.001
	Inflated	season – pedge + didist – dihroad + rtp – diforest	10	0.30	< 0.001
Mule deer	Non-inflated	season – pedge	6	0.31	< 0.001
	Inflated	season – pedge + didist + tri	8	0.33	< 0.001

[a] r^2 and p-values presented are means across five folds
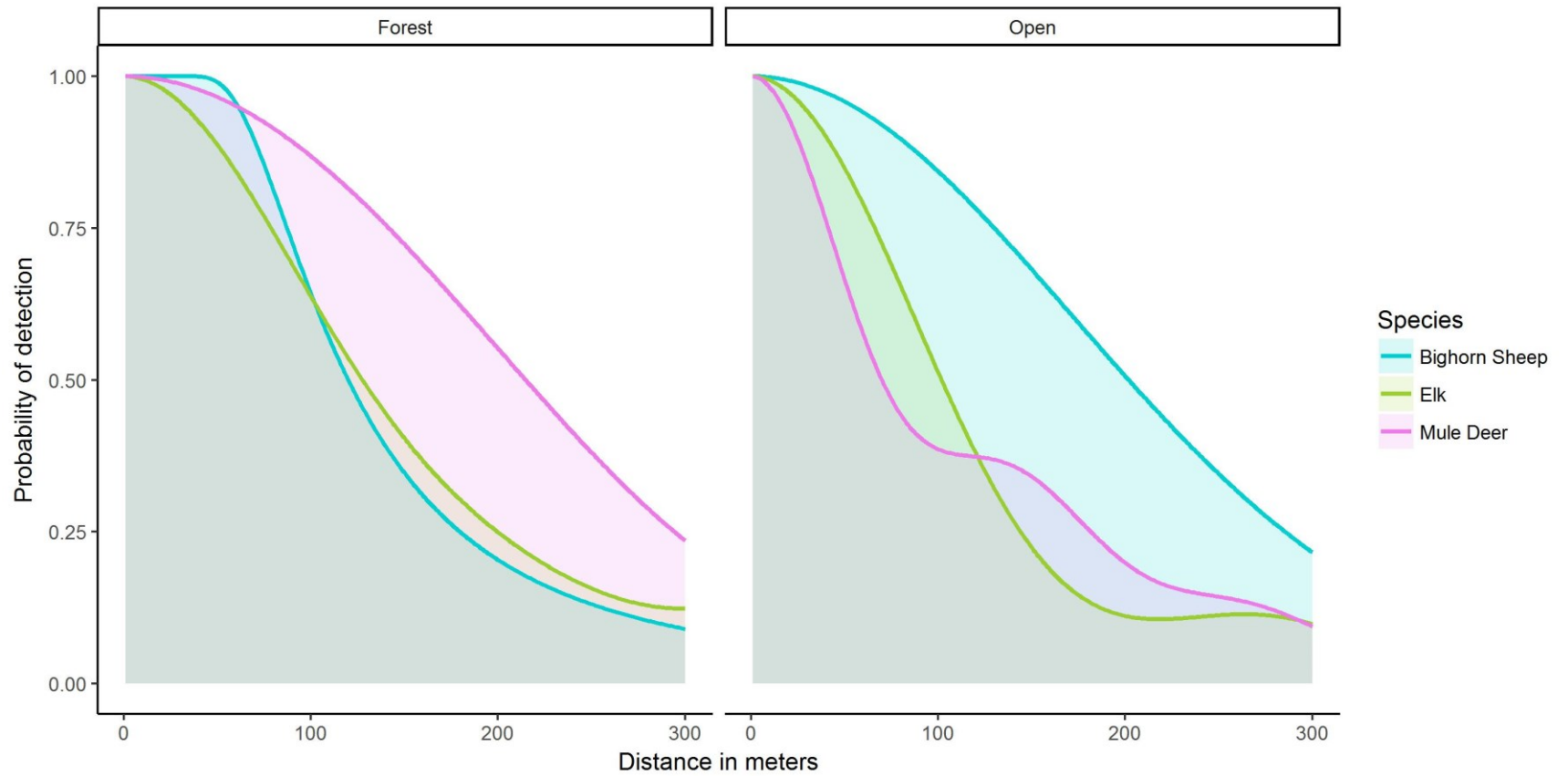
Appendix 2.1 – Figure 1 Comparison of fitted detection functions for bighorn sheep, elk, and mule deer in open and forest habitat on reclaimed mines in west-central Alberta. Detection functions are based on top models summarized in Appendix 2.1– Tables 1-6.
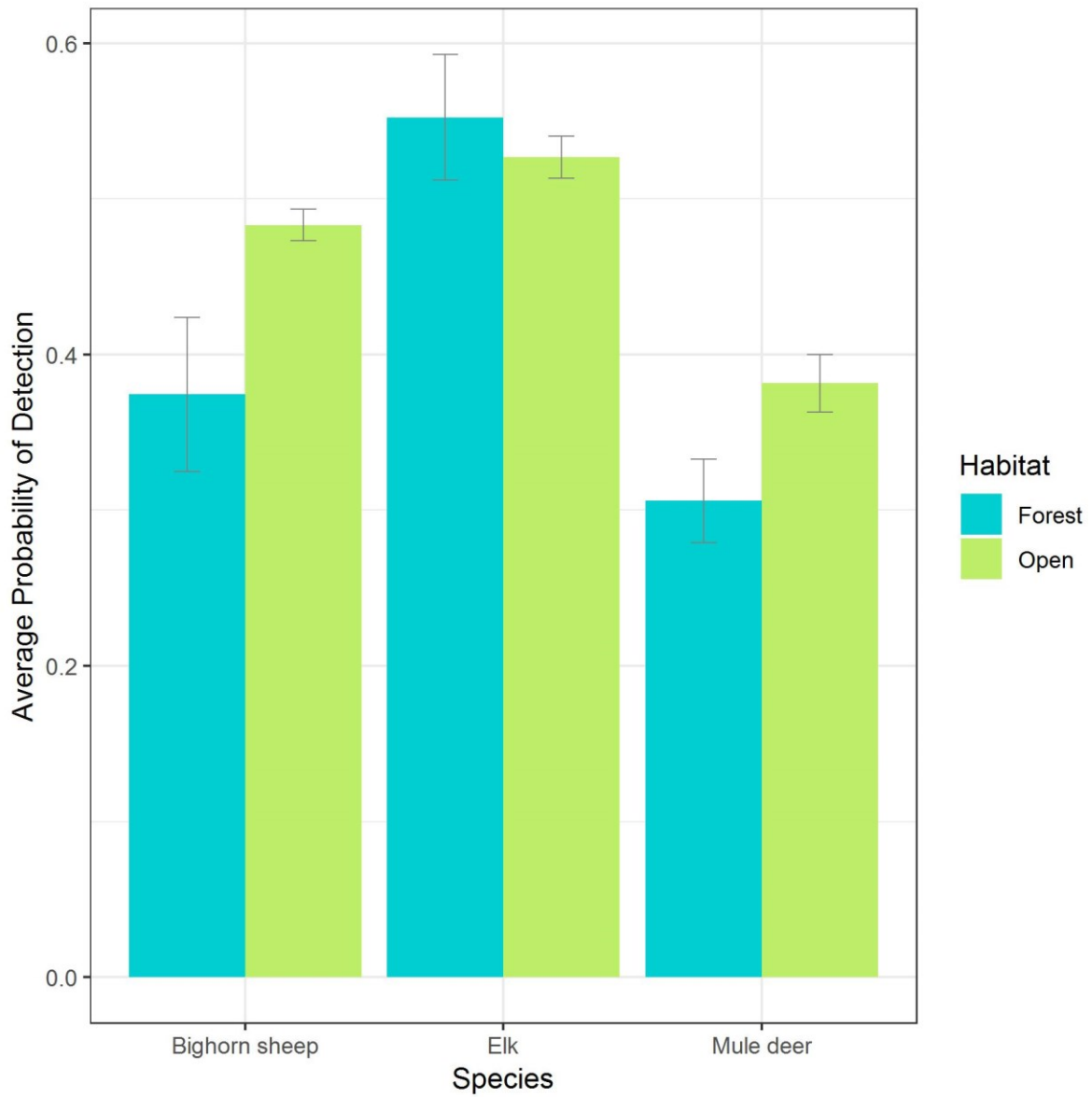
Appendix 2.1 – Figure 2 Estimated average probability of detection for bighorn sheep, elk, and mule deer in forest (blue) and open (green) habitat in the west-central Alberta study area. Average probabilities of detection were obtained from the top model for each conventional distance sampling analysis.

APPENDIX 2.2 – ASSIGNING DISTANCES TO SAMPLING UNITS

If mean distances were assigned to sampling units based on the distance from sampling unit centroid to nearest point on survey route (henceforth called Method 2), the distances assigned to each sampling unit above would be approx. 76 m, 49 m, and 3 m for panels A, B, and C, respectively (Appendix 2.2 – Figure 1). However, Method 2 does not take into account the tortuosity and configuration of the survey route within each sampling unit and, instead, considers only one measurement. By using the approach we described in our methods (we'll call this approach Method 1), the distances assigned to each sampling unit are 80 m, 50 m, and 17 m, for panels A, B, and C, respectively. Method 1 estimates similar values to Method 2 for panels A and B. However, panel C exemplifies the issues with using Method 2. In reality, the survey route is tortuous in this sampling unit and there are many opportunities to observe ungulate groups. It so happens that the survey route passes very near (3 meters) to the sampling unit centroid, which likely is an under-estimate of distance between survey route and ungulate group. By taking the mean of many pixel distances, as we do in Method 1, the unique survey configuration per sampling unit, and therefore unique opportunities to view ungulate groups within the sampling unit, are better accounted for.
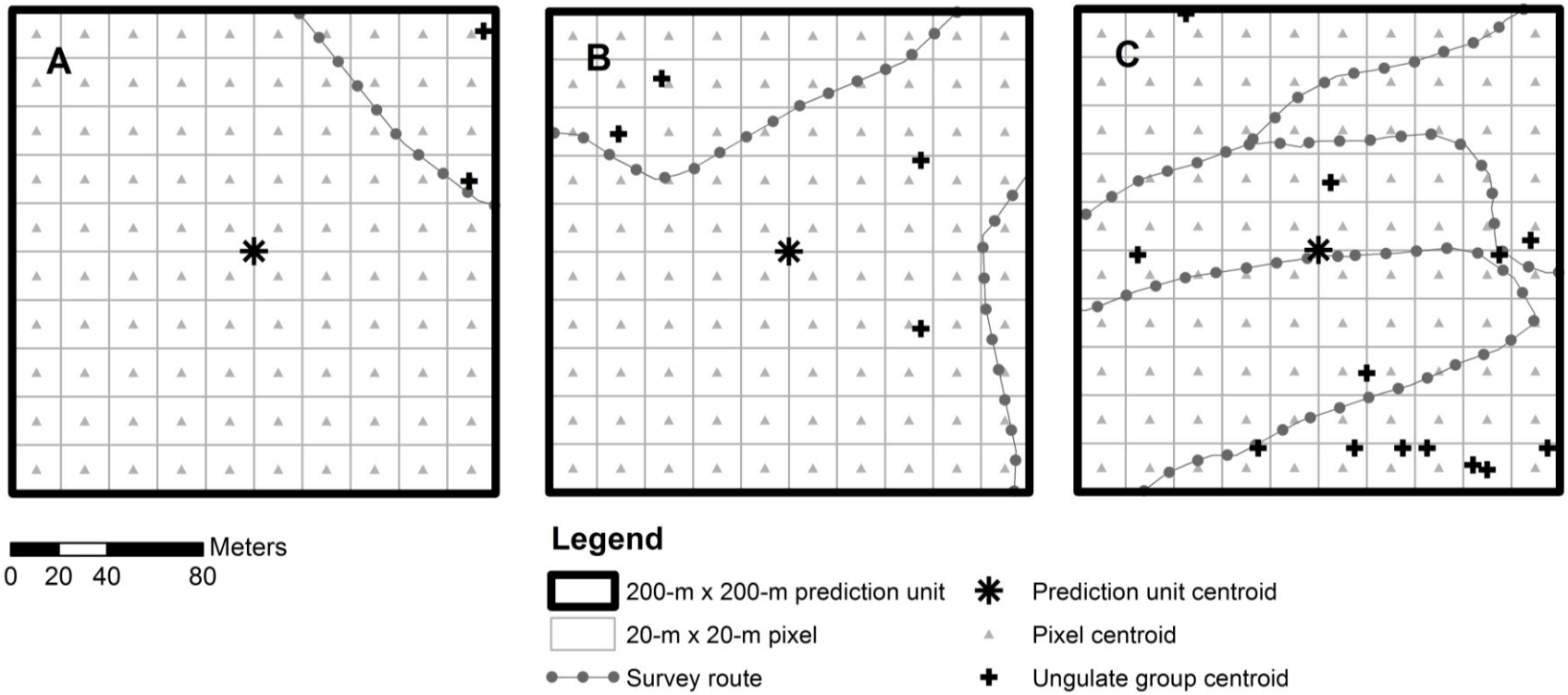
Appendix 2.2 – Figure 1 Examples of variation in survey route tortuosity and configuration within 200-meter x 200-meter sampling units. Panel A (far left) shows the least tortuous survey route, panel B (middle) shows a somewhat tortuous survey route, and panel C (far right) shows the most tortuous survey route.

APPENDIX 2.3 – ATTEMPTING TO MODEL INTENSITY OF HABITAT USE WITH HURDLE

MODELS

We originally planned to model ungulate habitat use using three sequential models. The

first model would answer: "where are ungulates selecting habitats?" and estimate

models of habitat selection. The second model would answer: "given that ungulates

selected habitats, how many groups occurred in selected resource units?" and then

estimate group size variation associated with selected habitats. The third model would

answer: "given that ungulates select habitat here and there are ungulate groups here,

how many individuals are in each group?" and this can be interpreted as a measure of

the intensity of habitat use. We chose to attempt to answer the first two questions with a

hurdle model, which I will explain in greater detail below.

Hurdle models (also known as zero-truncated models) are highly applicable in

ecology, especially for sampling designs with a tendency to inflate zero-counts, such as

ours. For example, we surveyed a large spatial extent of available habitat but only

detected ungulates using habitat in a small portion, which led to many 0's in our dataset

and some counts > 0. Hurdle models can be conceptualized as two separate generalized

linear models (GLMs), with the second GLM ('count' portion) conditional on the first

'conditional' portion. Using our analyses as a working example, we estimated whether

habitat was selected or not (a binary response variable) with logistic regression and a

binomial distribution (see Ch 2 – Methods). We estimated exponential RSFs for each

species from the logistic regression results (Ch 2 – Results). The count portion of hurdle

models takes the form of a count response variable (i.e. number of groups per sampling

unit per season), truncated at 0 to only include positive counts. In our case, we planned to use the count portion to determine how many groups used each sampling unit, given the sampling unit was selected. The count response variable is modelled using either a Poisson, negative binomial, or geometric distribution.

Multiple packages are available in R (R Core Team 2018) to run hurdle models, with varying computational efficiencies. We employed the *pscl* package in R (Jackman 2017), which allows the user to specify separate covariate combinations for each of the conditional and count portions of the model. We did not have issues determining and validating top models for the conditional portion of the hurdle (see Ch 2 – Results for logistic regression summaries). We did, however, have issues modelling the number of groups per sampling unit. The distribution of our response variable appeared visually like it should fit a negative binomial distribution, but the group data were so sparse (i.e. ranging from 1 – 4 groups) that the underlying distributions were difficult to test. Thus, we tested all distributions for each species and determined that a negative binomial fit better than a Poisson, based on AICc. However, negative binomial null models were consistently better than models with any landscape covariates. Ultimately, this led us to reason that the landscape covariates we had access to were not good predictors of number of groups using each sampling unit. We opted to model the number of individuals per group using a simple linear regression (Ch 2 – Methods) as an estimate of intensity of habitat use, as this was all our data allowed.

APPENDIX 2.4 – HIGHLY CORRELATED COVARIATES

Most high walls in our study area were reclaimed such that grasslands were seeded in close proximity. Thus, these landscape features were highly correlated (Pearson's r = 0.86; Appendix 2.4 – Figure 1), limiting our ability to include both covariates in one candidate model set. To remedy this issue, we attempted to create a covariate to represent high walls and only their adjacent reclaimed grasslands. According to Smith et al. (1991), 95% of bighorn sheep activity occurs within 300 meters from high walls. As such, we buffered high walls to include all reclaimed grasslands within 300 meters and termed this covariate 'high walls + proximate reclaimed grasslands'. We removed 'high walls + proximate reclaimed grasslands' from reclaimed grasslands so that 'remaining reclaimed grasslands' did not include any reclaimed grasslands within 300 m from high walls. In doing this, we hoped that we could tease apart the selection for high walls and reclaimed grasslands among our study species. For instance, we predicted that bighorn sheep would select for 'high walls + proximate reclaimed grasslands' but elk and mule deer would select for 'remaining reclaimed grasslands', as the latter two species have no biological use for high walls. However, the two new covariates were also highly correlated (Pearson's r = 0.81; Appendix 2.4 – Figure 1).

We attempted to fit highly correlated covariates in the same candidate set of models to observe changes in beta coefficient estimates. For candidate sets where we considered both distance to high wall (dihwall) and distance to reclaimed grassland (dirgra), beta estimates for dihwall ranged from -6.5 to -9.5 with more negative beta estimates consistently occurring when dirgra was removed from the model. We created

candidate sets where we considered both distance to 'high walls + proximate reclaimed grasslands' (dihwrg) and distance to 'remaining reclaimed grasslands' (dirgrx). Beta estimates for dirgrx were positive when dihwrg was not included in the same model, but were negative when dihwrg was introduced to the model. As such, we did not include highly correlated covariates in our final candidate set of models to ensure that we could rely on our beta coefficient estimates and make inferences for each covariate.
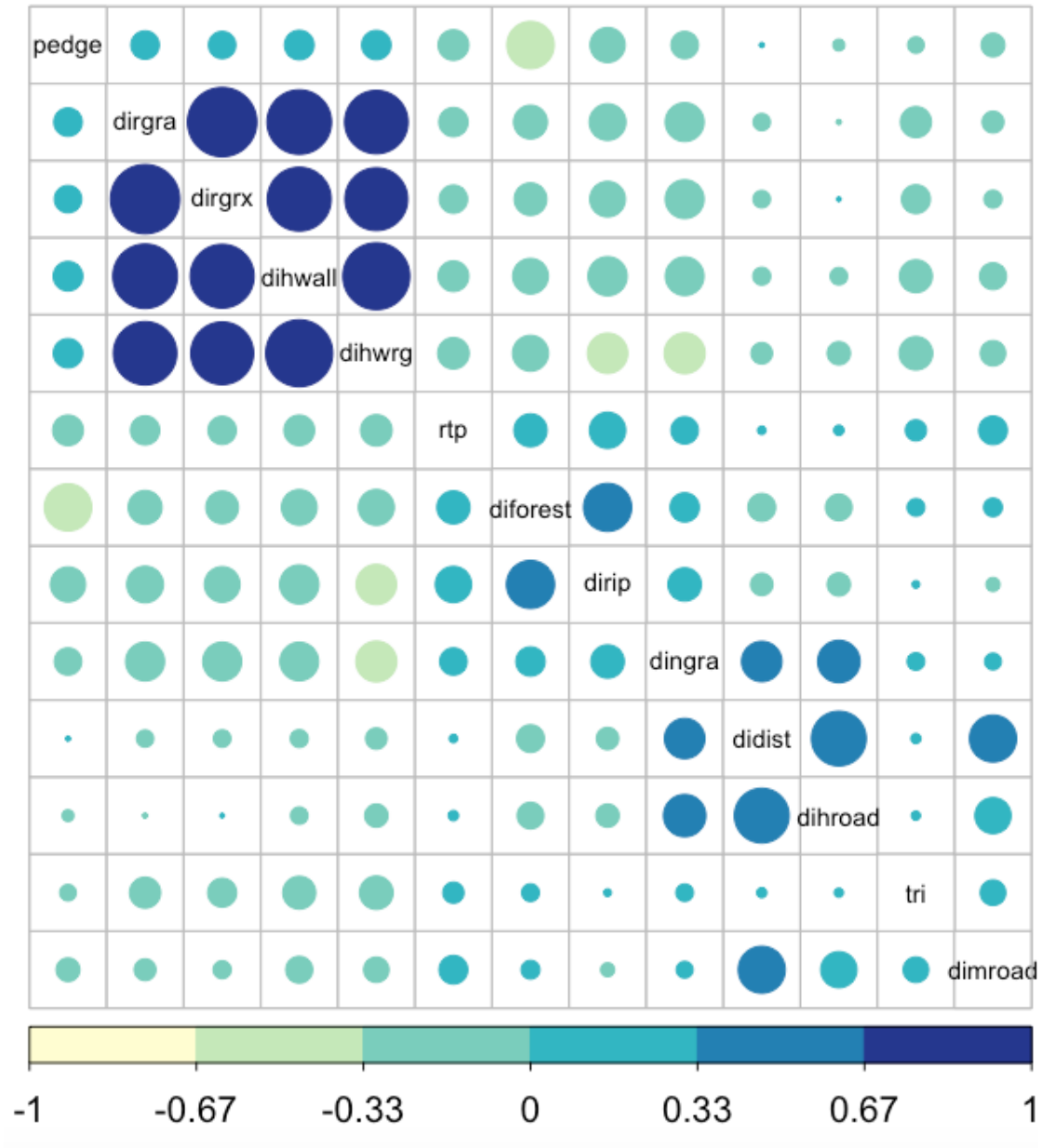
Appendix 2.4 – Figure 1 Correlation plot showing the relationships between continuous covariates. Circle size indicates magnitude of correlation, with larger circles showing larger magnitudes of correlation. Circle colour indicates direction of correlation, with blues indicating a positive correlation and greens/yellows indicating a negative correlation. 'dihwrg' is the abbreviation for 'high walls + proximate reclaimed grasslands' covariate. 'dirgrx' is the abbreviation for 'remaining reclaimed grasslands' covariate. All other abbreviations are detailed in Chapter 2 – Table 1.
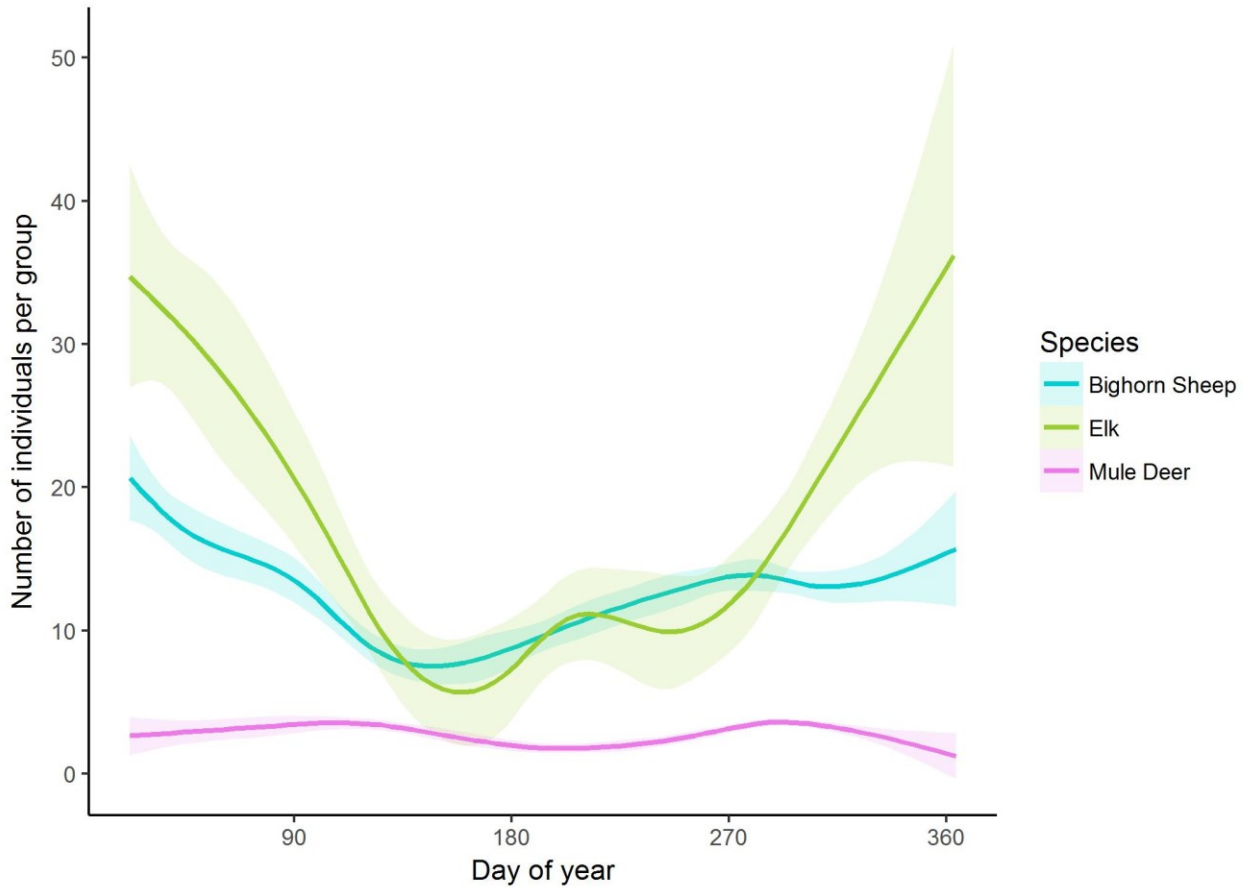
APPENDIX 2.5 – SEASONS

Seasonal differences in ungulate habitat selection are sometimes substantial (Rowland et al. 2000, Sawyer et al. 2007, Beck et al. 2013). As mentioned in Chapter 2 – Methods, we assigned a season to each ungulate centroid. For intensity-of-use models, we included season as a covariate in all models, including the null model, to account for seasonal variation in group size (Appendix 2.5 – Figure 1). For habitat-selection models, we averaged the number of 'used' sampling units per season to determine how many 'available' sampling units we should randomly select per season. We also attempted to model habitat selection for each season. However, due to small sample sizes, especially for elk and mule deer, we were not able to test the *a priori* candidate set of models for each season (Appendix 2.5 – Table 1).

Instead, we assigned each ungulate centroid to two seasons, instead of four. We determined that surveys between May – October (inclusive) would be during the snow-free season, and surveys between November – April (inclusive) would be during the snow season. We calculated the mean number of sampling units 'used' per season and randomly selected 'available' sampling units per season at a 1:5 ratio of used:available. We then replicated the same suite of 'available' sampling units each year for the 14-year study period. Logically, we found that sample sizes were larger when splitting data into only two seasons and then continued to evaluate the *a priori* candidate set of models for snow and snow-free seasons, for all study species (Appendix 2.5 – Table 2).

Model results for snow and snow-free seasons indicate there might be slight differences in selection based on season. However, top models were very similar when

comparing pooled seasons with snow and snow-free seasons. Overall, beta coefficient estimates were generally in the same direction. Further, k-fold cross validation scores indicated that pooling seasons did not detract from the predictive capability of the models. In fact, models with pooled seasons had better predictive capabilities for elk and mule deer. This could be due to the smaller sample size during snow and snow-free seasons causing high variability for these species. In conclusion, we retained models with pooled seasons in our manuscript but present additional seasonal models here if predictions are desired at finer temporal scales (i.e. Ch 3).

Appendix 2.5 – Figure 1 Seasonal variation in number of individual bighorn sheep, elk, and mule deer per species group, near mines in west-central Alberta. Day 90, 180, 270, and 360 roughly correspond to the end of March, June, September, and December, respectively. Lightly coloured shading represents 95% confidence intervals of group size, created by the smoothing function in ggplot2 (R Core Team 2018).

Appendix 2.5 – Table 1 Sample size of 'used' sampling units for bighorn sheep, elk, and mule deer for logistic regression, when considering data split into four seasons, two seasons, and when seasons are pooled. Seasons pooled represents the pooling of winter, spring, summer, and fall seasons, in which the data were originally separated into.

Species	Four seasons				Two seasons		Seasons pooled
	Winter	Spring	Summer	Fall	Snow	Snow-free	
Bighorn sheep	265	646	660	1300	826	1775	2871
Elk	68	225	252	274	231	621	819
Mule deer	28	124	222	408	153	602	782

Appendix 2.5 – Table 2 Top-ranked logistic regression models for determining bighorn sheep, elk, and mule deer habitat selection, modelled with snow and snow-free seasons, as well as pooled. Covariates, number of model parameters (K), 5-fold cross-validation $\bar{r}^2$ values, and 5-fold cross-validation p-values are presented. Signs indicate the sign of the beta coefficient estimates.

Species	Count type	Covariates	K	$\bar{r}_s$[a]	p-value[a]
Bighorn sheep	Pooled	− dihwall − pedge + rtp − didist − dihroad − dimroad + diogra	8	0.94	< 0.001
	Snow	− dihwall − pedge + rtp + didist* − dihroad* − dimroad + diogra − tri	9	0.89	< 0.01
	Snow-free	− dihwall − pedge + rtp − didist − dihroad* + diogra + tri	8	0.96	≈ 0
Elk	Pooled	− dirgra + diforest + rtp − tri + dirip + didist − dihroad + diogra	9	0.89	< 0.01
	Snow	− dirgra + diforest + rtp* − tri + dirip + didist − dihroad − dimroad	9	0.83	< 0.01
	Snow-free	− dirgra + diforest + rtp − tri + dirip + didist − dihroad + dimroad + diogra + pedge	11	0.79	0.02
Mule deer	Pooled	− dirgra + diforest + rtp − tri + dirip − didist + dihroad + diogra + pedge	10	0.94	< 0.001
	Snow	− dirgra + diforest* + rtp − tri* + dirip − didist + dihroad − dimroad	9	0.81	< 0.01
	Snow-free	− dirgra − diforest + rtp − tri + dirip − didist + dihroad + diogra	9	0.86	< 0.01

[a] r^2 and p-values presented are means across five folds

* indicates any covariate where the beta coefficient estimate +/- SE overlaps 0

APPENDIX 3.1 – ROADS IN STUDY AREA

We attempted to create multiple road covariates to represent more specific anthropogenic features. For instance, roads in our study area differ by width, speed limit, frequency of use, type of use, and composition (i.e. gravel or paved). Examples of the variety of roads in our study area include paved undivided highways, paved/gravel municipal roads, gravel logging roads, gravel well-pad service roads, gravel mining roads, and gravel recreational roads. We attempted to represent similar types of roads in collapsed groups: level 1 for high-speed divided highways; level 2 for undivided highways, municipal roads, major logging roads, major well-pad service roads, and major recreational roads; and level 3 for mine roads, minor logging roads, minor well-pad service roads, and minor recreational roads. However, level 2 and 3 roads were both highly correlated with anthropogenic land, which prevented us from including level 2 or 3 roads separately as covariates.

APPENDIX 3.2 – INCLUSION OF MINES IN NON-MINE GROUP MODELS

Because the non-mine group of cougars selected home ranges that did not overlap reclaimed mines, we did not include distance to reclaimed mines as a covariate in second or third order RSFs. Clearly, this presents an issue when comparing selection for mines between mine and non-mine cougar groups in the study area. However, we felt that including mines in the non-mine cougar group model was biologically uninformative. Inclusion of mines in top models may actually represent apparent selection for features highly correlated with reclaimed mines. For example, elevation and reclaimed mines were highly positively correlated, and distance to highway and reclaimed mines were highly negatively correlated. Thus, if reclaimed mines appeared in the non-mine group top model, it would more likely reflect a selection for increasing elevation or further distances to highway, than it would reflect selection for a mine that the cougars are presumably unaware of. Although we were not able to obtain an estimate of selection for/against reclaimed mines for the non-mine cougars, we can use estimates from the mine cougars (M1 and F2) to inform us about how cougars near reclaimed mines may integrate these areas into their geographic and home ranges.

APPENDIX 3.3 – PREY BIOMASS

Appendix 3.3 – Table 1 Prey weights used to calculate prey composition for cougars in west-central Alberta, Canada, between March 2017 and October 2018.

Age & sex class	Prey weights (kg)								
	Deer spp.	Elk	Moose	Bighorn sheep	Beaver[d]	*Canis* spp.[d]	Lynx[d]	Red squirrel[d]	Snowshoe hare[d]
Adult M[a]	95.0	320.0	450.0	117.0					
Adult F[a]	70.0	230.0	418.0	65.0					
Adult unknown[b]	82.5	275.0	434.0	91.0					
Juvenile[c]	31.0	101.5	184.75	28.5					
Unknown[a]	54.0	178.0	323.0	50.0	26.5	16.0	11.0	0.23	1.50

[a] We obtained estimates from Knopff et al. (2010a)
[b] Adult unknown is the average of adult male and adult female weights per species and was used when age category was known, but sex was unknown
[c] Juvenile is the average of yearling and young of year age classes from Knopff et al. (2010b) and was used, irrespective of sex
[d] We could not differentiate male prey from female prey for these species and therefore did not differentiate biomass based on sex

APPENDIX 3.4 – RAW UNGULATE COUNTS FOR 2017

Appendix 3.4 – Table 1 Raw counts of bighorn sheep, elk, and mule deer observed by Bighorn Wildlife Technologies Ltd. on reclaimed mines, during four surveys in 2017. See Chapter 2 for survey methods.

Species	Number observed per age-sex class				
	Adult female	Adult male	Juvenile	Unknown	Total
April survey					
Bighorn	110	62	45	49	266
Elk	356	2	24	0	382
Mule deer	4	1	1	0	6
July survey					
Bighorn	43	130	4	0	177
Elk	214	7	34	0	255
Mule deer	2	2	0	0	4
September survey					
Bighorn	156	142	94	0	392
Elk	48	11	5	0	64
Mule deer	0	5	0	0	5
November survey					
Bighorn	252	141	110	0	503
Elk	175	70	27	5	277
Mule deer	7	1	2	1	11

APPENDIX 3.5 – INCLUDING HIGH WALLS IN KILL OCCURRENCE MODELS

Distance to tree patches and high walls were highly correlated for kill occurrence models. We chose to include tree patches as a covariate because our research questions directly related to understanding the influence of landscape configuration (i.e. tree patches) on cougar kill occurrence. However, we experimentally included high walls in kill occurrence models instead of tree patches and found that this resulted in a different top model (Appendix 3.5 – Table 1). Interestingly, when high wall was included, the top model did not include forest edge (Appendix 3.5 – Table 2). These results may help to understand whether high walls on the reclaimed mines are an ecological trap for bighorn sheep.

Appendix 3.5 – Table 1 *A priori* candidate models for kill occurrence, using high walls instead of tree patch as a covariate. Resulting top model is bolded.

Model name	Covariate structure
Null 1	kud.cougar
Null 2	bh.avail
Encounter	kud.cougar + bh.avail
1	**Encounter + dihwall**
2	Encounter + diedge
3	Encounter + diedge + dihwall
4	Encounter + rtp
5	Encounter + tri
6	Encounter + diedge + didist
7	Encounter + rtp + tri
8	Encounter + diedge + dihwall + didist + rtp + tri

Appendix 3.5 – Table 2 Comparison of beta coefficient estimates for top kill occurrence model when including distance to tree patch and when including distance to high wall. An asterisk indicates that standard error overlaps zero.

Candidate set	Top model name	Scaled beta coefficient estimates (SE)			
		kud.cougar	bh.avail	diedge	dihwall
ditreepatch	Model 2	0.840 (0.189)	0.907 (0.267)	-1.092 (0.486)	
dihwall	Model 1	0.804 (0.210)	-0.123* (0.313)		-5.949 (2.011)