Ecological Response to Human Activities in Southwestern Alberta: Scientific Assessment and Synthesis

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1.0 Summary

The Castle region of southwestern Alberta is an ecologically diverse landscape that supports numerous recreational, agricultural, and industrial activities. Understanding the individual and cumulative impacts of these human stressors on species and ecosystems is critical to informing environmental management decisions. We characterized environmental stressors associated with land use and human activities in the region and summarized scientific evidence for ecological responses of soil, vegetation, hydrology, and wildlife. The scientific evidence included in this report was based on over 150 peer-reviewed journal articles and technical reports, including studies in ecologically relevant landscapes elsewhere in North America. We contextualized the reported ecological relationships and projected impacts in these studies to the Castle region and other similar biogeographic regions in Alberta.

Anthropogenic drivers of biological and ecological responses in the Castle region (defined here to include Castle Provincial Park and Castle Wildland Provincial Park) include forestry, cattle grazing, industrial development, recreational activities, human-caused wildfire, and the introduction and spread of invasive species. The area also has profound cultural and sacred value and has been used by Indigenous Peoples for generations. Key natural drivers include wildfire disturbance and extreme weather events such as floods and drought. Topographic variability and climate drive regional patterns of soil, vegetation, and biotic communities. Because long-term monitoring data from the Castle region are sparse, many of these environmental drivers are not well documented. Consequently, we focussed on drivers that could be characterized using available information, primarily derived from public databases and land cover inventories based on aerial photographs and satellite images.

Roads constructed in the Castle region to support forestry, oil and gas, coal mining, and other human uses, plus a network of trails used for motorized and non-motorized recreation, have resulted in over 2,000 kilometers of linear footprints in the region. These linear features are likely the most significant anthropogenic feature in the Castle region because of the human activities and impacts they facilitate. While the use of linear footprints by people (including motorized vehicles) has not been directly quantified in the Castle region, it is likely that off-highway vehicle use has exacerbated soil erosion, loss of vegetation cover, alteration of vegetation communities, and the introduction and spread of invasive plant species.

Roads and trails in the region cross watercourses thousands of times, with each crossing providing a potential pathway for increased sediment inputs, increased runoff rates, and altered magnitude of flow and water quality in headwater streams. Such increased sedimentation may negatively affect threatened populations of westslope cutthroat trout, bull trout, and other aquatic species in the region that have evolved to inhabit clean, cool, complex, and connected watercourses. Native trout species, in particular, are highly valued ecological, socially, culturally, and economically, and have been the subject of substantial research and conservation efforts. In spite of this, a

synthetic understanding of the impacts of linear footprints and other conservation challenges related to human activity in the Castle region is incomplete. Studies throughout the range of both trout species in western North America suggest that their populations in the Castle region are negatively affected by roads and other linear footprints. It is likely that increased sedimentation from bank erosion and streambed disturbance, leading to reduced water and habitat quality for spawning and other life stages, has contributed to observed regional population declines for bull and westslope cutthroat trout. Further, these declines are likely exacerbated by other threats, including competition or hybridization with introduced non-native fish species and angling pressure.

Linear footprints have been linked to increased grizzly bear mortality in Alberta and elsewhere. Most recorded grizzly bear mortalities in Alberta over the past 10 years were caused by people, and evidence suggests that the majority of human-caused grizzly bear mortalities occur within 500 metres of a road or within 200 metres of a trail. Females with young cubs are particularly sensitive to mortality risk near linear footprints, and behavioural studies suggest this sub-population avoids areas with high road densities. Although direct evidence of OHV impacts on grizzly bears is limited, data suggest that some grizzly bears respond with increased movement rates or avoidance of trails used by OHVs. Using new GIS-based modeling analyses that projected grizzly bear mortality risk from human use of linear footprint, bear mortality risk was projected to decline with restrictions on motorized human use of roads and trails in the region.

Evidence suggests that limiting or reducing land use and human activities in the region is expected to decrease vegetation disturbance, lower rates of invasive species infiltration and expansion, improve the condition of headwater streams, increase the viability of westslope cutthroat trout and bull trout populations, and reduce the risk of grizzly bear mortality. Projected changes in hydro-climatic regimes in the region as the result of climate change will also influence some of these biological and ecological responses. Further monitoring and research are required to reduce uncertainty in estimates of impacts, quantify the contribution of natural vs anthropogenic drivers on ecosystems and species in the region, and to inform management options and actions.

2.0 Introduction

This assessment provides an objective review and synthesis of published scientific data and information of relevance in examining the environmental impacts associated with human disturbance in the Castle region of southwestern Alberta, Canada (Figure 1). The Castle region is defined here as Castle Provincial Park and Castle Wildland Provincial Park, a combined area of 1,052 km^2 (Alberta Environment and Parks 2017a). The Castle region is highly valued by a variety of stakeholders with a diversity of interests and priorities, including those related to wilderness conservation and ecological integrity, agricultural and industrial production, and recreational activities such as camping, hiking and off-highway vehicle (OHV) use.

The Castle region has experienced increased levels of human-caused disturbance over the past few decades. Specifically, recreational use of OHVs and the associated network of trails have expanded in recent years, and the impacts of this increase in OHV use will be additional (or in some cases multiplicative) to the impacts of historical forest harvesting, livestock grazing, hunting, fishing, and non-motorized forms of recreation such as hiking, cycling, and equestrian activities.

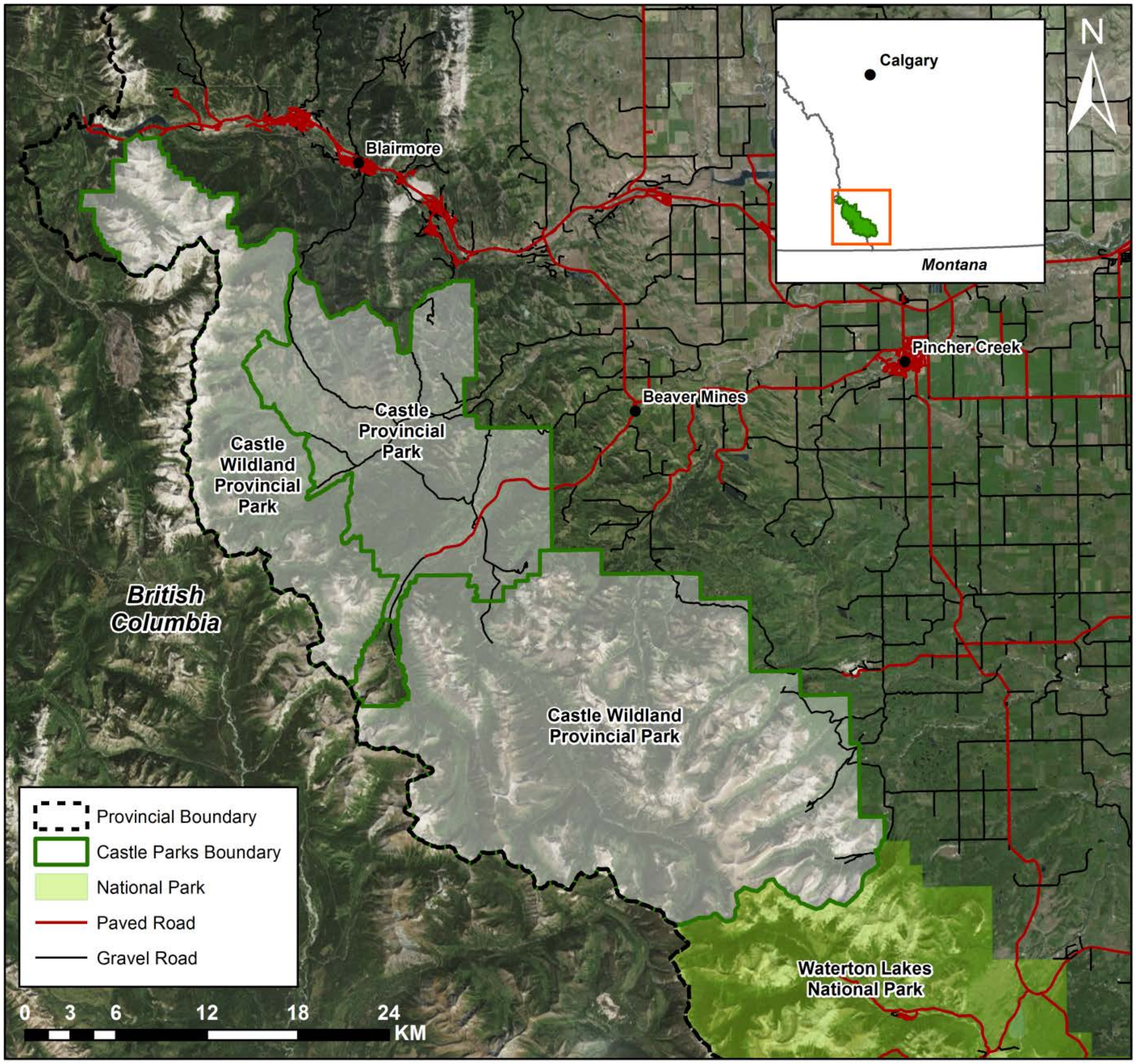

Figure 1 Map of the Castle region. Data sources: Appendix A.

One of the more prominent stakeholder concerns relates to the region's recent designation as a protected area and the prohibition of summertime recreational use of OHVs. The Castle region is a preferred destination for all-terrain vehicle (ATV) recreation in Alberta. For example, the Castle region and areas to the north (collectively, the Crowsnest Pass area) were voted the "Favourite Overall ATVing Area" in Alberta by readers of RidersWest magazine (RidersWest 2017).

This synthesis of environmental data and studies is intended to clarify the state and nature of environmental risks associated with human activities in the Castle region in a manner that may inform current and future decision-making processes in the development and implementation of Alberta's regional management plans and related regulatory regimes.

3.0 Methods

3.1 Review of the scientific literature

Because the predominant human activities and impacts in the Castle region are related to the intensity and extent of linear footprints, a summary of key original research relevant to ecological response to linear footprints is located in Appendix B. A summary of peer-reviewed review articles, which provided a foundation for identifying and exploring further relevant research, is located in Appendix C.

3.2 Spatial data

Data sources used in maps, spatial summaries, and analyses included in this review are listed in the Literature Cited section, and key spatial data sources used in figures are listed in Appendix A. Several sources were used only for visual reference on maps: Provincial Parks (2017); Wildland Provincial Parks (2017); Alberta SPOT Imagery (2016); and ESRI World Imagery WGS84 (2017). All maps were created using ArcMap (Version 10.3.1; ESRI 2015).

The location and type of linear footprints such as roads and trails in the Castle region were based on a combination of four of the most current Government of Alberta and public linear footprint data sources (Linear footprint in the Castle region 2017). The linear footprint dataset created for the Oldman Watershed Headwaters Indicator project (Linear features in the Oldman Watershed 2014, Fiera Biological Consulting Ltd. 2014) was used as the starting dataset, and additional linear footprints, along with attribution, were added from three additional datasets (Linear features in the proposed Castle Parks 2012, Designated trail network for the South Saskatchewan Region 2017, Trail inventory for the Castle Parks 2016). Visual inspection of satellite imagery was used to confirm whether parallel lines in the combined linear footprint dataset were unique or different features, and a topology check was performed to ensure that no duplicate records or overlapping features were included. Lastly, all linear footprints were grouped into one of the following five categories using attribution from input data sources:

- paved roads;
- gravel roads;
- unimproved roads, unclassified roads, and truck trails;
- pipelines and powerlines; and
- cutlines and trails.

The length of each linear footprint category and the area of three feature types (non-linear human footprint, land cover, and historical wildfires) in the Castle region were calculated using ArcMap (Version 10.3.1; ESRI 2015). To calculate the range of linear footprint densities among watersheds within the Castle region, we intersected watersheds (Hydrologic unit code watersheds of Alberta 2017) with linear footprints (Linear footprint in the Castle region 2017) and calculated linear footprint density for the area of each watershed within the Castle region.

We identified stream crossings by intersecting watercourse data (FWMIS Hydrology Arcs 2017) with linear footprint data (Linear footprint in the Castle region 2017). These crossings were then summarized by Strahler stream order (Strahler 1952) and by linear footprint category. Stream crossing density was calculated by dividing the number of stream crossings on each watercourse reach by the total length of the watercourse.

Additional analyses followed a similar approach (i.e., subdivided by Strahler stream order) for summarizing crossings of watercourses occupied by westslope cutthroat trout (Westslope cutthroat trout occurrence within their native range 2017), as well as watercourses that have been identified as federally designated critical habitat in the species recovery strategy (Westslope cutthroat trout federally designated critical habitat 2017). Lastly, crossings of watercourses occupied by bull trout (Bull trout occurrence within their native range 2017), as well as watercourses where bull trout spawning has been observed (Bull trout spawning reaches 2017), were summarized by Strahler stream order.

A digital elevation model (DEM; Alberta provincial digital elevation model 2017) was used to generate percent slope data for the Castle region, which was then grouped into low (0-6%), moderate (6-20%), high (20-40%), very high (40-60%), and extreme (>60%) risk categories (adapted from Meyer 2002). The linear footprint dataset was intersected with these slope risk classes to calculate the length of footprint in each class, which was then used to determine the proportion of total linear footprints that occur in highly, very highly, or extremely sensitive slope classes.

3.3 Uncertainty in spatial data

We assessed the suitability of potential data sources for this review from available documentation, and metadata for sources used in this review are listed in the Literature Cited section. While the accuracy of the locations and attributes of features represented on maps is likely high, a formal accuracy assessment was beyond the scope of this review. Therefore, the maps, summaries, and analyses presented in this review are estimates rather than exact measurements or counts. The accuracy of these estimates could likely be improved by verification via ground-truthing. In the case of linear footprints, for example, not all existing footprints are mapped, particularly if they are not tied to a disposition and they are not visible on satellite imagery (such as low-use ridgetop trails or small trails under dense canopy cover). Additionally, some features may be mapped or categorized incorrectly based on human error during manual tracing of geographic features from aerial or satellite imagery. However, we performed limited randomized checks of the linear footprint dataset against satellite imagery as a verification step, and topology checks were conducted to further reduce error and uncertainty. An additional consideration for interpreting the results of the stream crossing analysis is that most of these stream crossings (i.e., 94%) occur on headwater streams (Strahler stream orders 1 to 3), some of which may be intermittent and difficult to observe depending on the time of year and depending on recent precipitation (Figure 2).

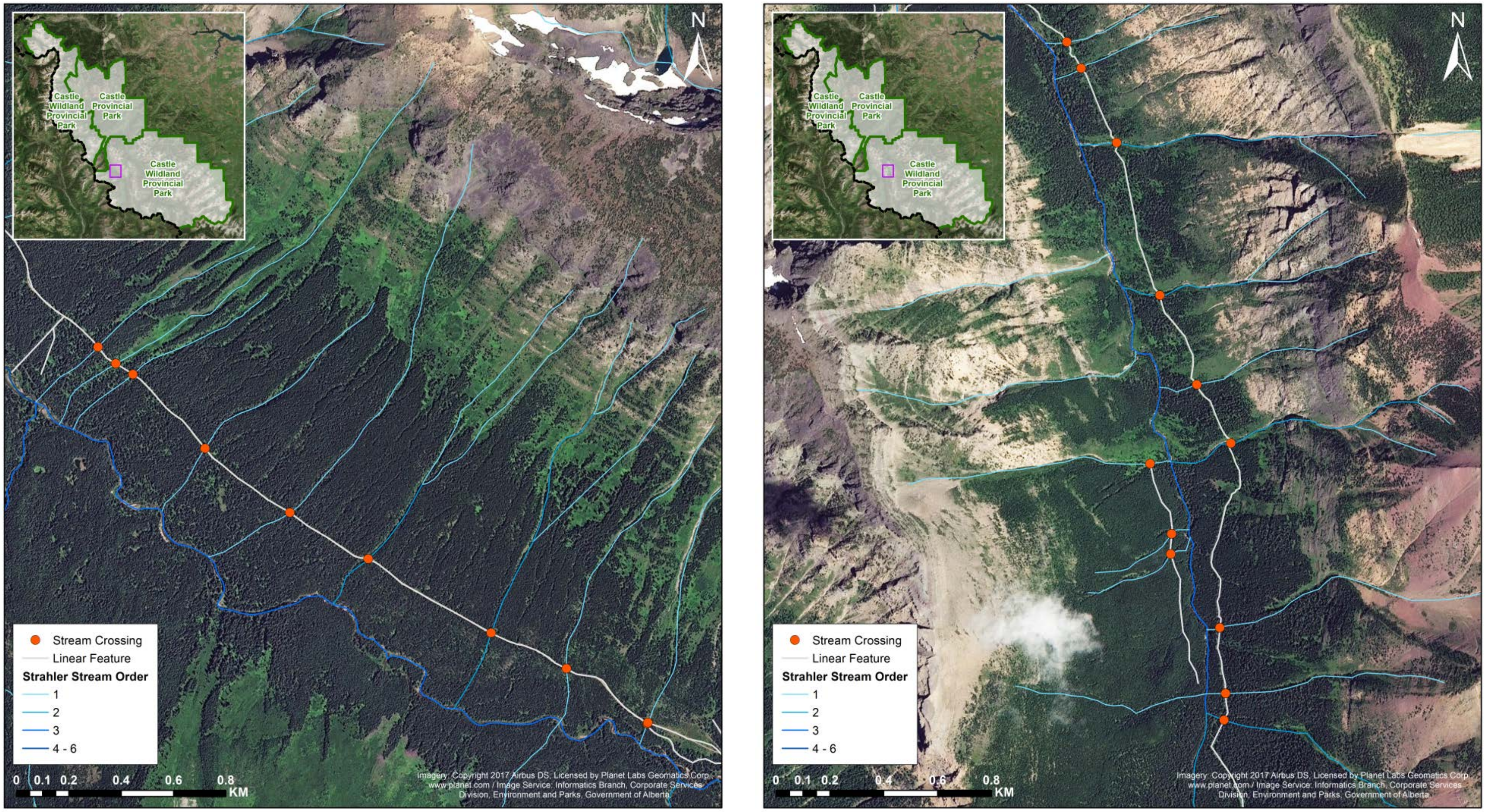

Figure 2 Stream crossings illustration. Data sources: Appendix A.

4.0 Ecosystems and biodiversity

The mountainous topography throughout the Castle region created by tectonic forces 60-70 million years ago has been altered by erosion, transport and deposition of sedimentary materials during multiple ice ages and during the intervening periods. This area of Alberta was probably unglaciated during the Pleistocene epoch that ended approximately 12,000 years ago (Geological Association of Canada 1958*),* thereby providing potential refugia for numerous species of plants and animals that would have colonized newly exposed areas following the previous retreat of glacial ice from the Castle region (Ogilvie 1962). Narrow transition zones unique in Alberta occur abruptly from eastern grasslands of the Foothills Parkland to Montane, Subalpine, and Alpine ecosystems of the Rocky Mountains.

Hydrological processes in the region are driven by complex topography and climatic regimes. Watersheds are dominated by small headwater streams (Strahler stream order[1] 1 to 3; Figure 3), which comprised just under 90% of the approximately 2,000 km of mapped watercourses in the Castle region that ultimately feed major rivers flowing out of the region, including the Carbondale and West Castle rivers (Figure 4).

Headwater systems comprise the vast majority of riverine habitat and water resources, and are fundamental building blocks for numerous ecosystem services (Lowe and Linkens 2005), provide critical habitat for many forms of aquatic biodiversity (Meyer et al. 2007), including imperiled, endemic, and economically valuable species, and will be increasingly important ecological refugia as anticipated long-term climate change is realized (Isaak et al. 2015, Jones et al. in revision). For example, Jones et al. (in revision) project that headwater streams in the Crown Ecosystem, which includes the Castle region, will provide important cold water refuge from future climate change for westslope cutthroat trout and bull trout.

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[1] According to the "top down" system devised by Strahler (1952), rivers of the first order are the outermost tributaries. If two streams of the same order merge, the resulting stream is given a number that is one higher. If two rivers with different stream orders merge, the resulting stream is given the higher of the two numbers.

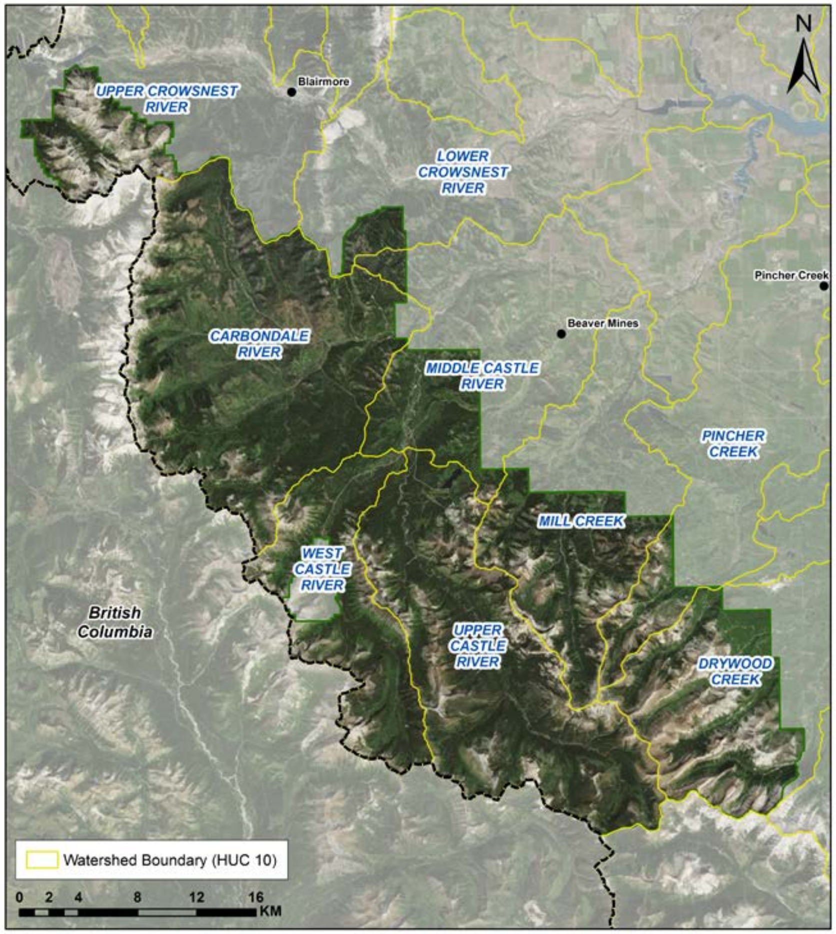

Figure 3 Watersheds in the Castle region. Data sources: Appendix A.

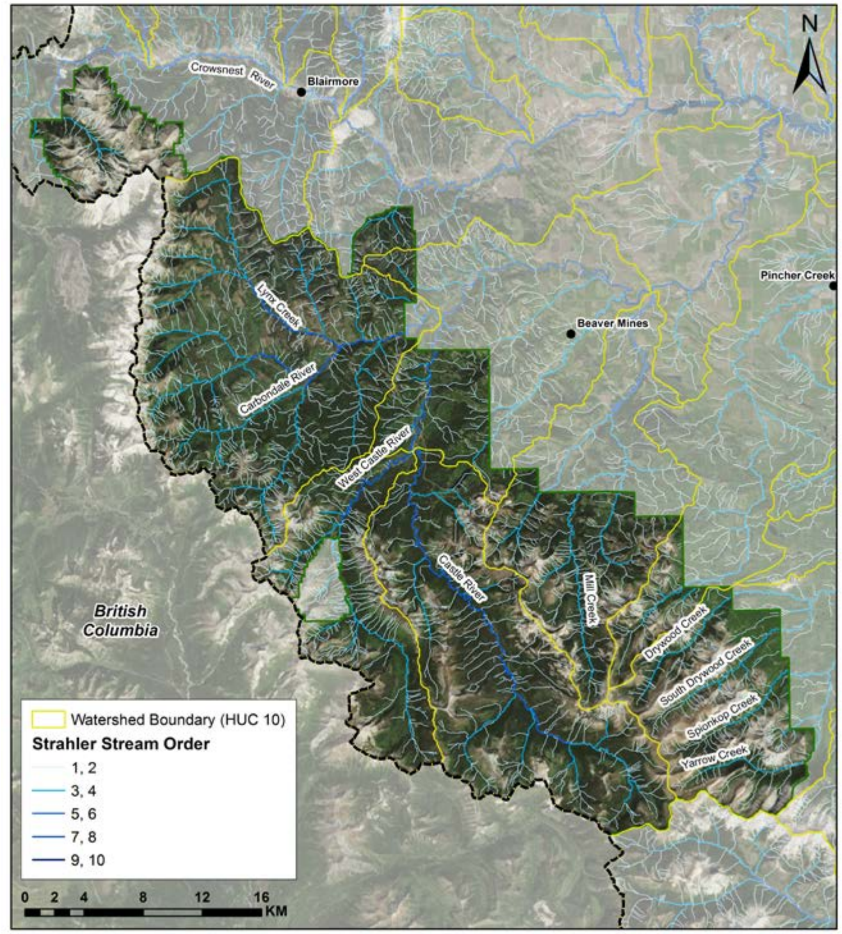

Figure 4 Watercourses in the Castle region. Data sources: Appendix A.

Soils in the Castle region have not been formally surveyed, although some soil information may have been generated locally as a result of regulatory requirements for industrial or commercial approval applications. However, soils in the region have recently been mapped through interpretation of derived ecosite phase data (Figure 5; Derived Ecosite Phase 2017).The derived ecosite phase data were developed based on Alberta Vegetation Inventory (AVI) and LiDAR-derived datasets to provide a framework for grouping ecological sites and ecological site phases in the province.

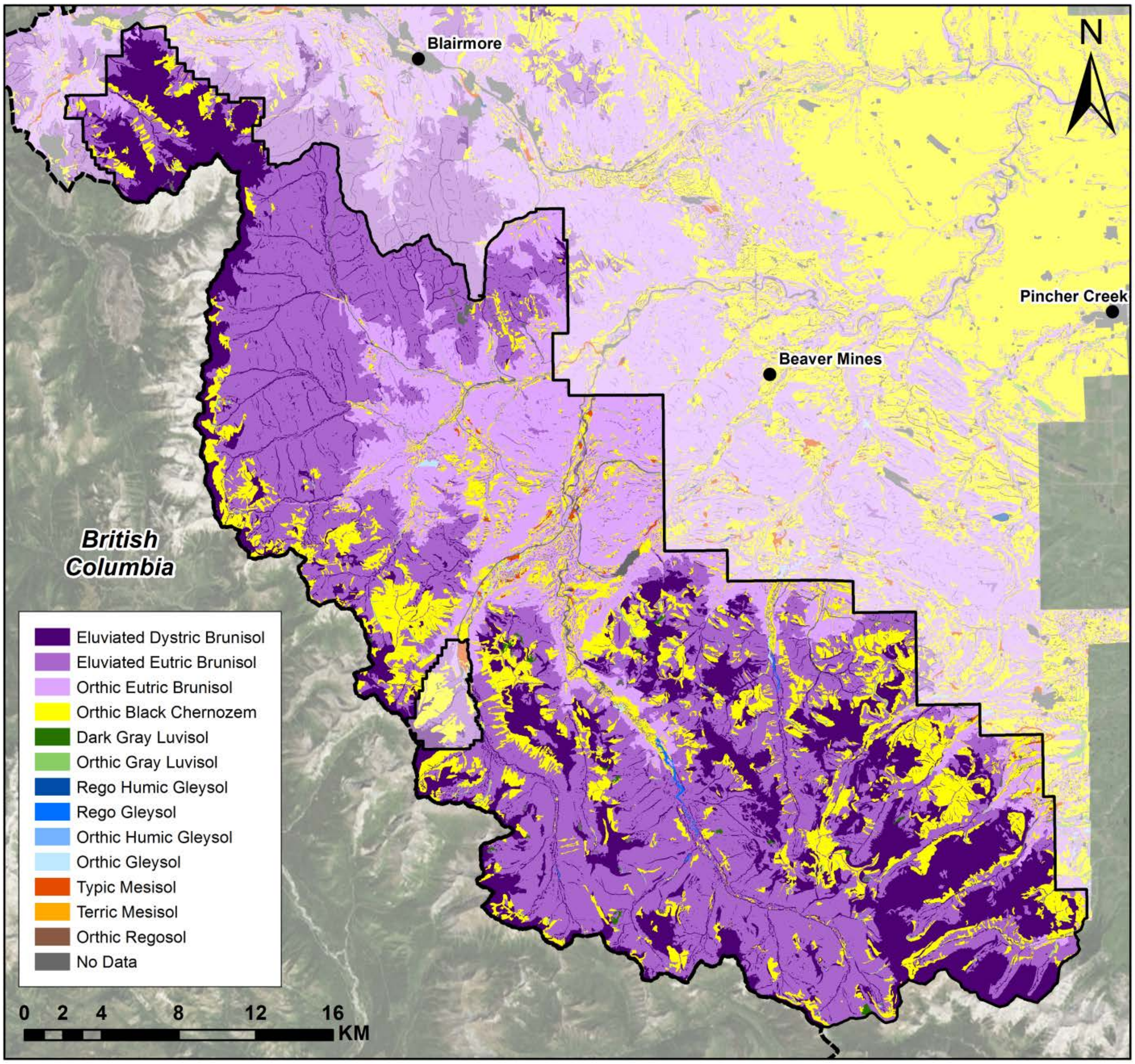

Figure 5 Soil subgroups in the Castle region. Data sources: Appendix A.

Diverse topography and climatic patterns in the Castle region combine to create a complex landscape of soils dominated by those that have undergone relatively little pedogenic alteration. The prevalence of partially developed soils is driven by the lack of conditions necessary for weathering and leaching, as well as the periodic disruption of soil formation by processes such as erosion and deposition. Interpretation of ecosite phase data has classified soils belonging to six out of the ten soil orders defined by the Canadian System of Soil Classification (Soil Classification Working Group 1998). The most dominant soils identified in the Castle region, by area, are Brunisols (83.0%) and Chernozems (14.9%). Remaining soils are classified as Gleysols (0.6%), Luvisols (0.5%), Regosols (0.1%), and Organics (0.1%).

Brunisols represent an intermediate step in the soil development process between undeveloped Regosols and soils orders with more developed, diagnostic horizons such as Chernozems and Luvisols. Regosols are weakly developed mineral soils that occur in areas where significant soil formation and horizonation are not possible because of continuous erosion or deposition. Chernozems are generally associated with grassland and shrubland plant communities, whereas Luvisols (though relatively rare in the Castle region) typically occur under forest communities where sufficient soil formation and horizonation has occurred for diagnostic horizons to develop. Gleysols are found in poorly-drained areas where prolonged water saturation occurs, typically in the valley bottoms. Lastly, Mesisols (a great group within the Organic order) are generally associated with wetlands and are found in low-lying areas where deep accumulations of organic matter occur.

Land cover in the Castle region is dominated by montane and subalpine coniferous forests in valley bottoms, transitioning to alpine shrubland and rock/rubble at higher elevations (Figure 6 and Figure 7). Native grassland areas occur throughout the Castle region with the majority occurring in the southern area. A notable portion of the northern Castle region is composed of relatively young forest recovering from the Lost Creek wildfire (2003), with no other comparable wildfires in the region since the 1930s (See Section 5.4 below). A small proportion of mixed and broadleaf forests occur in the Montane natural subregion.

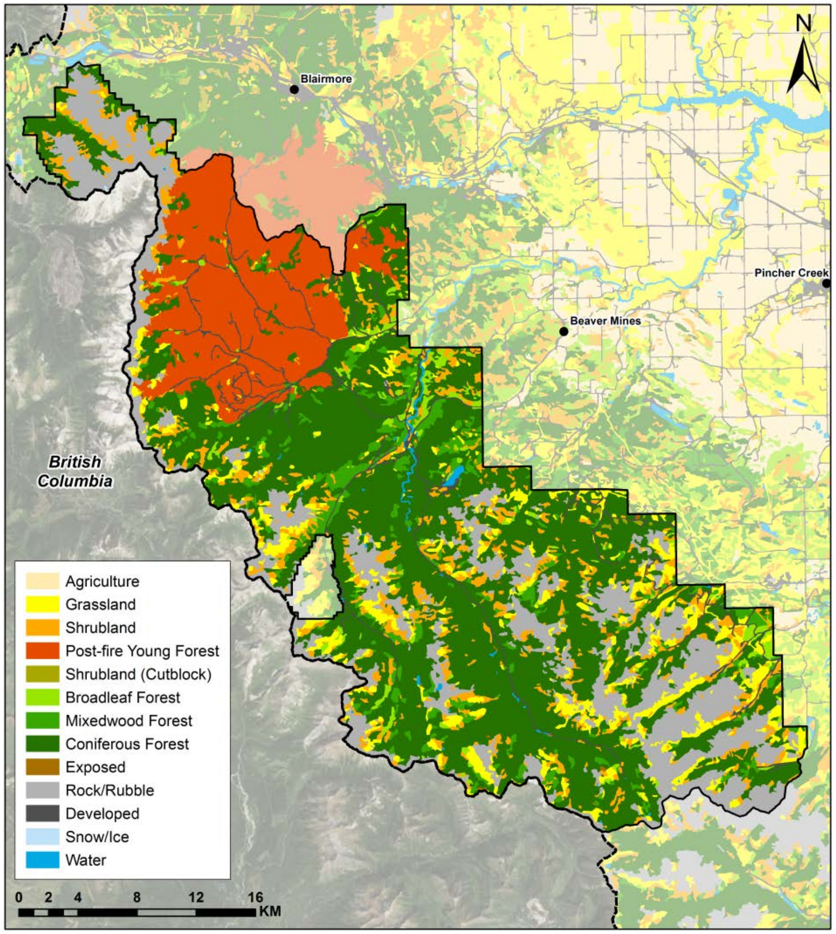

Figure 6 Land cover in the Castle region. Data sources: Appendix A.

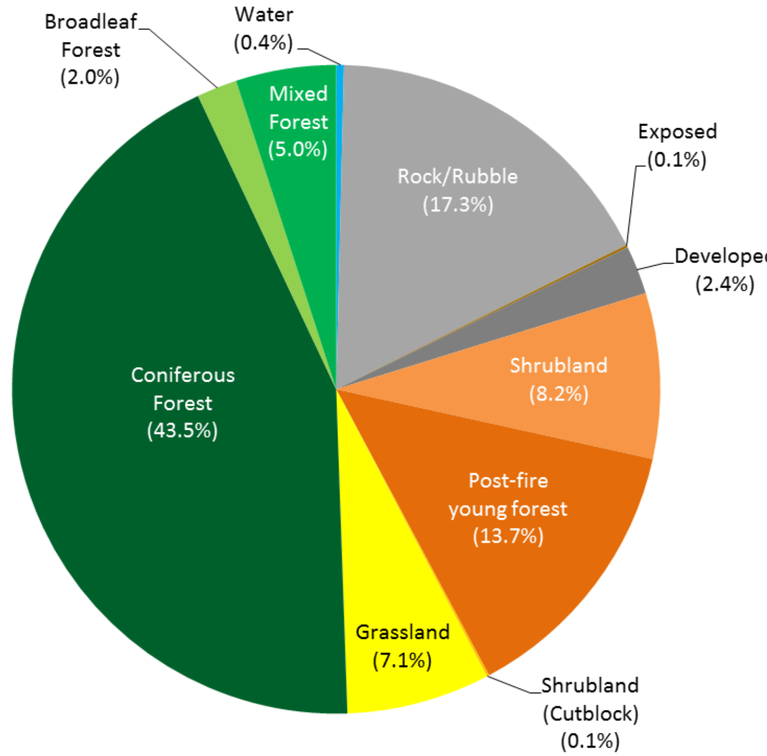

Figure 7 Summary of land cover in the Castle region. Data sources: Appendix A.

The Castle region contains an estimated 15% of Alberta's tracked vascular plant and animal species with non-secure statuses (Alberta Conservation Information Management System 2016; ACIMS) in less than 1% of the province's land area. Records of species of conservation concern (i.e., non-secure status) in the Castle region include 97 vascular plants, 46 non-vascular plants, 14 invertebrates, and 23 other species of fish and wildlife (Alberta Conservation Information Management System 2016; Fish and Wildlife Management Information System 2016, FWMIS; Alberta Environment and Parks 2017a). Eight tracked vegetation communities are present in the region, including whitebark pine / ground juniper - common bearberry woodland (*Pinus albicaulis / Juniperus communis - Arctostaphylos uva-ursi*). According to ACIMS, the number of tracked species and plant communities in the Castle region (189) is comparable to other protected areas along the Eastern slopes, including Banff (155) and Jasper National Parks (201). It should be noted that the Castle region contains twice as many tracked species of vascular plants (106) as both Banff (51) and Jasper (49). While these data serve to illustrate the biodiversity of the Castle region, a key limitation of both ACIMS and FWMIS data is that they are not collected systematically and thus they are biased to areas frequented by people. Few systematic surveys have been completed in the Castle region, and additional tracked species are likely found in the area.

This review addresses the potential impacts of human activities on numerous native plant and animal species, with particular focus on three high-profile species at risk, chosen because they are representative of conservation concerns spanning both the terrestrial and aquatic ecosystems: westslope cutthroat trout (*Oncorhynchus clarki lewisi*), bull trout (*Salvelinus confluentus*), and the grizzly bear (*Ursus arctos horribilis*).

Westslope cutthroat trout: Throughout their Alberta range, westslope cutthroat trout populations have declined by as much as 80% (Cleator et al. 2009). The major causes of declines in distribution, abundance, and genetic diversity of westslope cutthroat trout in Alberta and elsewhere include habitat loss, invasive species, overfishing, and climate change (COSEWIC 2006). Invasive species negatively impact westslope cutthroat trout via introgressive hybridization, competition, and predation. Furthermore, invasive hybridization is exacerbated by climate change (Muhlfeld et al. 2014), and reduces the fitness of westslope cutthroat trout (Muhlfeld et al. 2009a). The subspecies was listed in 2009 as *Threatened* under Alberta's *Wildlife Act* due to the subspecies' small distribution and continuing decline in extent of occurrence, the severely fragmented nature of populations, continuing decline in quality of habitat, and the presence of barriers to dispersal making immigration between watersheds highly unlikely (The Alberta Westslope Cutthroat Trout Recovery Team 2013). Hybridization with non-native rainbow trout (*Oncorhynchus mykiss*) is a primary concern for this subspecies, and cool, higher-elevation streams common in the Castle region may provide refugia for westslope cutthroat from rainbow trout (Rubidge and Taylor 2005, Bennett et al. 2010, Rasmussen et al. 2010, Rasmussen et al. 2011, Yau and Taylor 2013). More recently, however, it has been suggested that high-elevations and cool temperatures alone may be insufficient in preventing hybridization of westslope cutthroat trout with rainbow trout (Muhlfeld et al. 2017). These factors reinforce the importance of preventing habitat degradation, habitat fragmentation, or overfishing for this species, all of which are impacts associated with motorized human access in the Castle region. Currently, Castle region contains 59 km of watercourses identified as federally designated critical habitat for westslope cutthroat trout, i.e., streams considered to be occupied by genetically pure populations of this species (≥99% genetic purity for individual fish sampled) (Fisheries and Oceans Canada 2014; Figure 8).

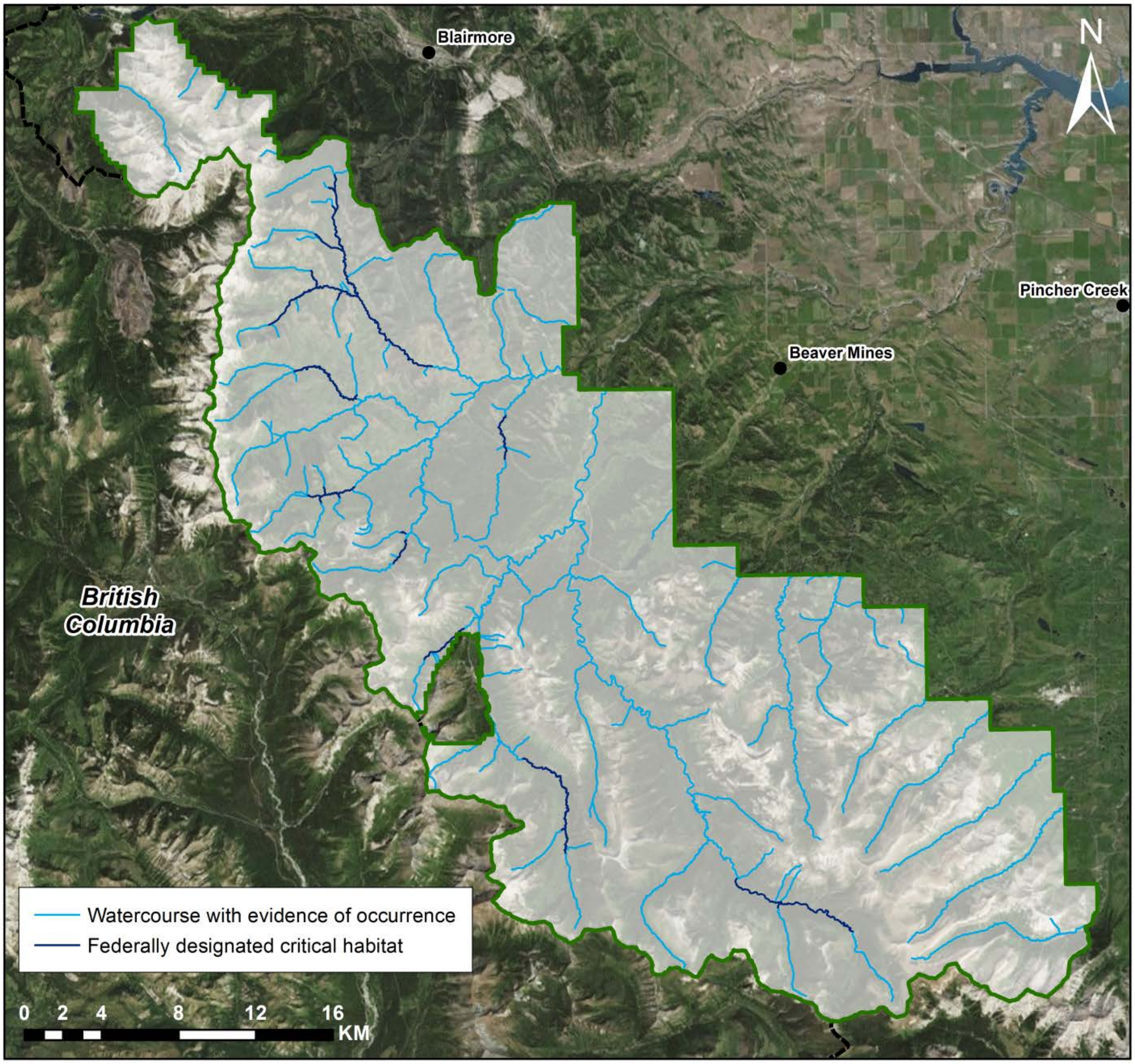

Figure 8 Distribution of westslope cutthroat trout in the Castle region. Data sources: Appendix A.

Bull trout: The Castle region contains important areas of Alberta's remaining bull trout habitat (Figure 9). It is estimated that over half of Alberta's bull trout populations are in decline from a 33% reduction in suitable habitat (Alberta Sustainable Resource

Development 2012). Historical habitat declines have been caused by industrial development, hydrological disruption (and associated loss of connectivity) from dams, roads and culverts causing migration barriers and population isolation (Costello et al. 2003). Bull trout have also been outcompeted by the widespread introduction of non-native fish species including brook trout, brown trout, and rainbow trout (Post and Johnston 2002, Alberta Sustainable Resource Development 2012). Additional population declines can be attributed to overfishing and reduced water quality associated with human activities. Bull trout also have a prolonged egg-incubation period (approximately 7 months), making them particularly vulnerable to increased stream sedimentation caused by the physical disturbance of stream crossings (Alberta Sustainable Resource Development 2012), especially where crossings are impacted by substantial human disturbance.

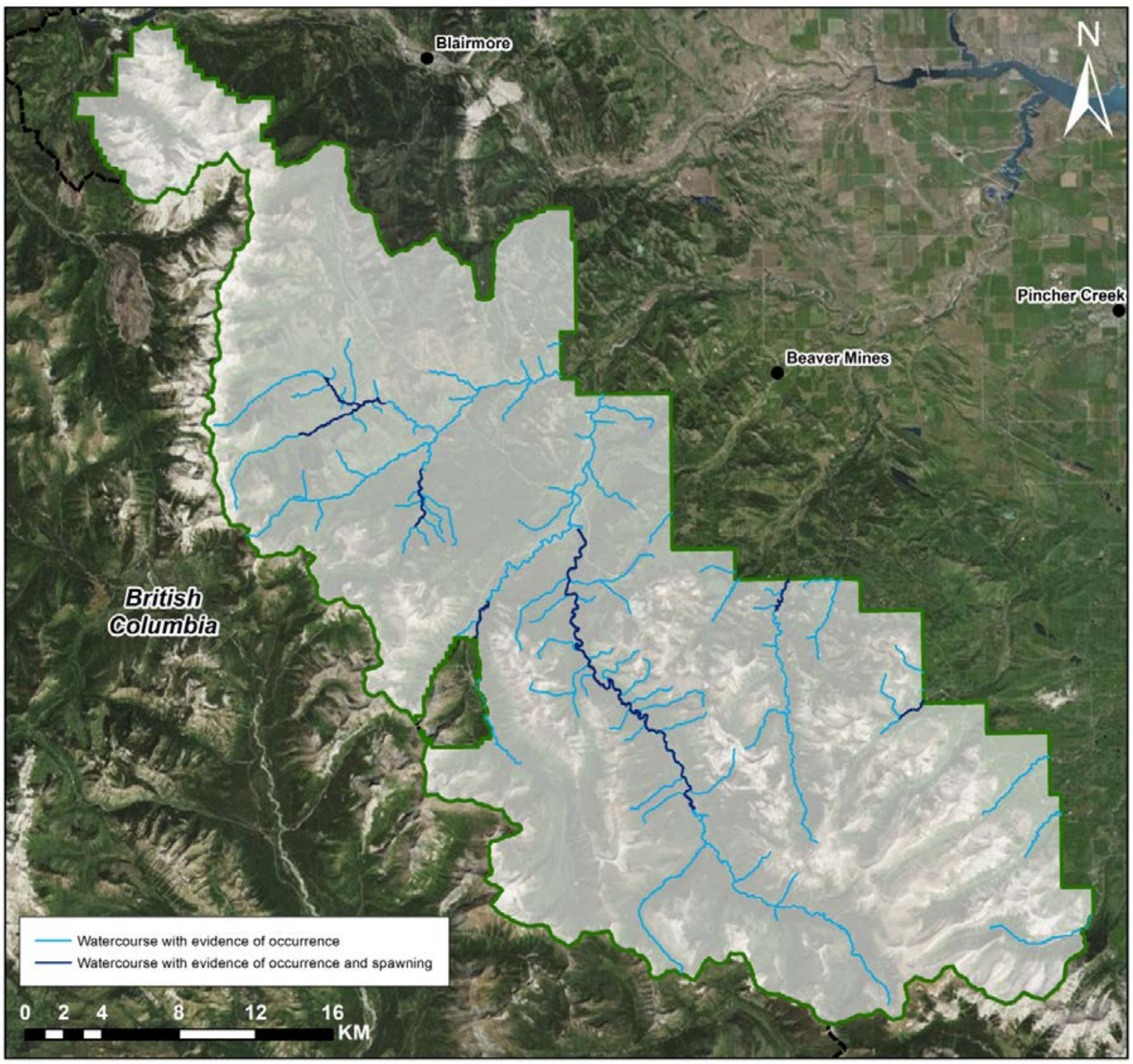

Figure 9 Distribution of bull trout in the Castle region. Data sources: Appendix A.

Grizzly bears: Grizzly bear researchers have identified the Castle region as primarily high-quality bear habitat (Stenhouse and Morehouse in prep; Figure 10). Grizzly bears in the Castle region are a component of a larger Rocky Mountain population that includes grizzly bears in southeastern British Columbia and Montana (Figure 11; Proctor et al. 2012, Morehouse and Boyce 2016), with some individual bears using the Castle region as a core area and other individuals ranging widely (Stenhouse and Morehouse in prep; Figure 11). Human-caused mortality is a key stressor on this species in Alberta, primarily in the form of illegal harvest, which accounted for 40% of known human-caused mortalities between 2006 and 2013 (including 13% mistakenly identified as black bears; Alberta Environment and Parks 2017b). Other significant causes of mortality include vehicle collisions (21%), human-wildlife conflict (20%) and associated agency control (11%), and First Nations harvest (7%). Based on a 6-year average between 2008 and 2013, the Castle bear management area (BMA 6) had the highest grizzly bear mortality rate (including translocations) of any population unit in the province (Alberta Environment and Parks 2017b), and between 2009 and 2013, 38% of all grizzly bears that were relocated due to human-wildlife conflict in Alberta were from the Castle BMA (Alberta Environment and Parks 2017b).

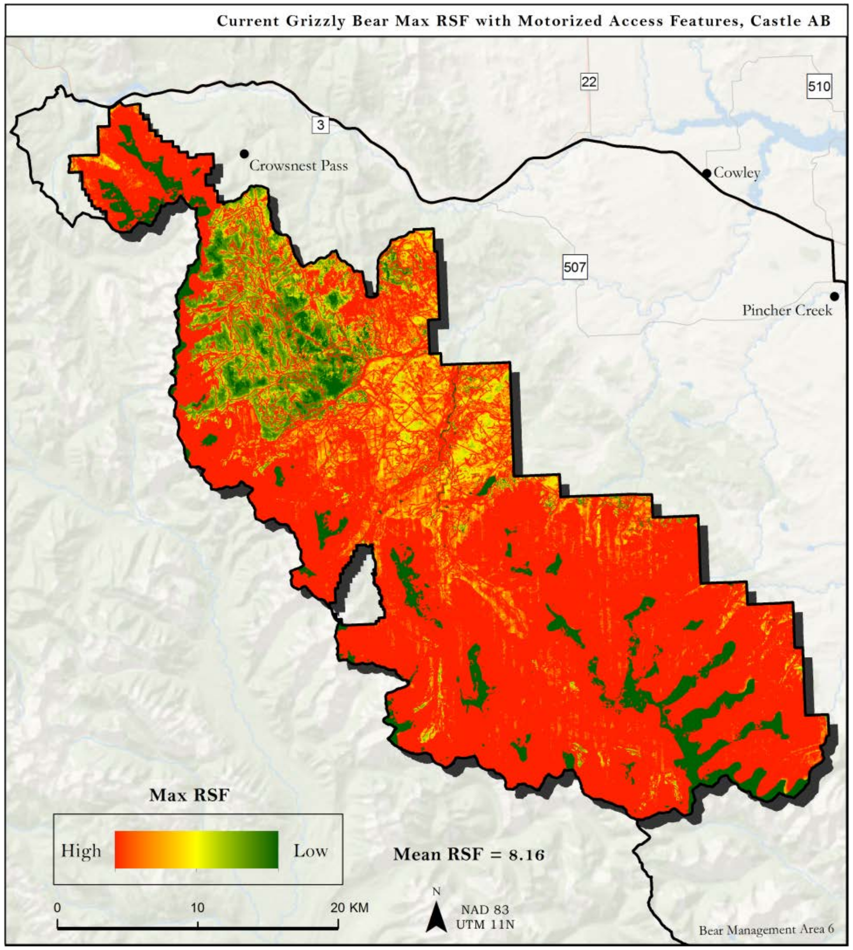

Figure 10 Grizzly bear habitat value in the Castle region (Stenhouse and Morehouse in prep). Data sources: Appendix A.

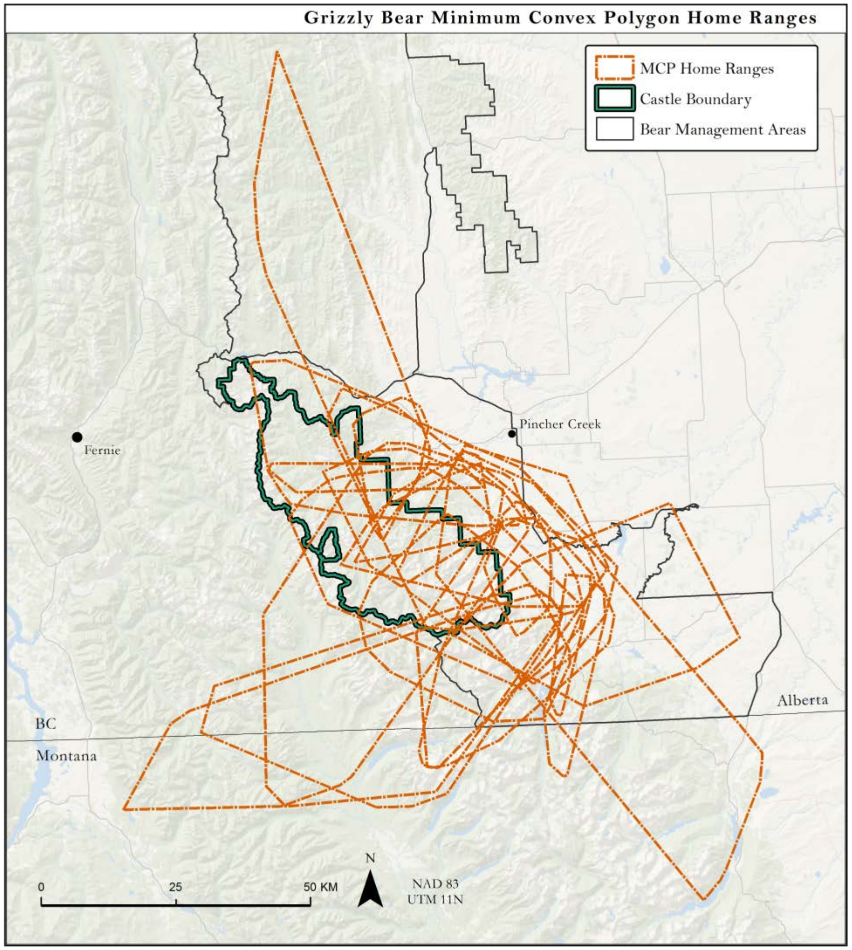

Figure 11 Home range polygons of radio collared grizzly bears in the Castle region, 2005-2009 (Stenhouse and Morehouse in prep). Data sources: Appendix A.

5.0 Human activities and land use

5.1 Linear footprint

Linear footprint consists of human-made linear features caused by vegetation clearing that contrast with the adjacent area alongside. The most common examples of human-made linear footprints on Alberta's landscape are roads, railways, pipelines, seismic-exploration trails, transmission lines, and recreational trails. Expansion of the road network in the Castle region began in 1948 with the onset of focussed seismic exploration for oil and gas southwest of Pincher Creek (Flathead Transboundary Network 1999). Updated inventories indicate a total of 2,125 km of linear footprint in the Castle region (Figure 12) consisting primarily of trails, followed by unimproved roads, pipelines, powerlines, and paved roads (Figure 13).

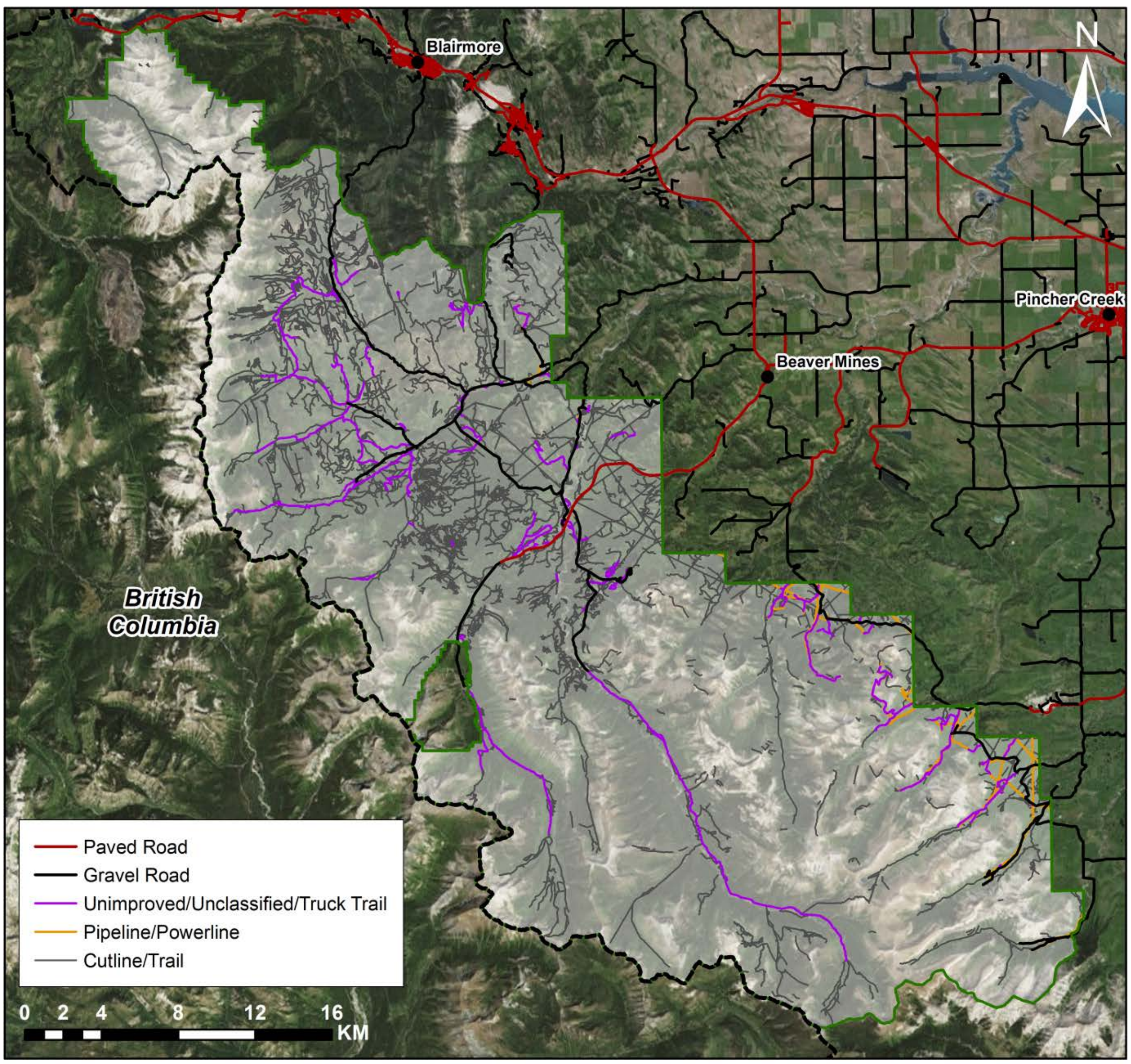

Figure 12 Linear footprint in the Castle region. Data sources: Appendix A.

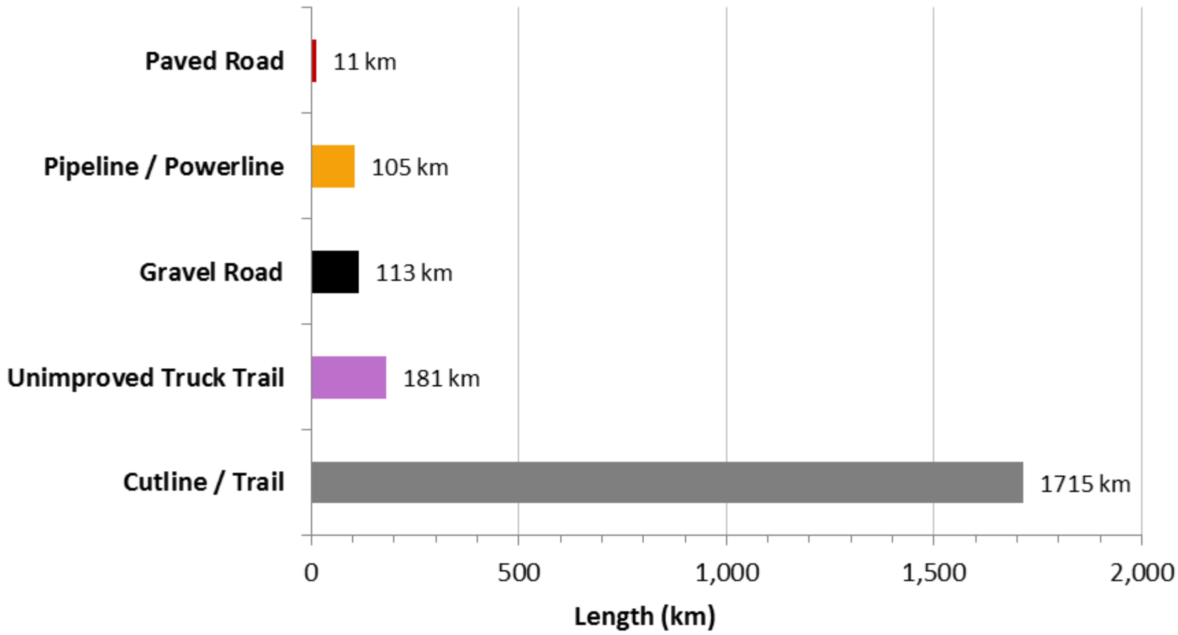

Figure 13 Summary of linear footprint lengths by footprint category in the Castle region. Data sources: Appendix A.

Previous studies have estimated total linear footprints for the Castle region between 1,184 km (Lee and Hanneman 2011) and 1,823 km (Smith and Cheng 2016). These differences are the result of differences in the spatial extent used in each study, including a focus on either the Castle Forest Land Use Zone boundary or the proposed Castle Provincial Park and Wildland Provincial Park boundaries circa 2016. The greater length of linear footprint in this assessment and review (2,125 km) is a function of a larger area and improvements in the resolution, availability, and collation of spatial data for the Castle region, and not necessarily the creation of new linear footprints since previous studies were completed.

At a watershed scale, the average density of all linear footprint in the region is 2.0 km/km^2, ranging from 0.5 km/km^2 to 3.4 km/km^2 (Figure 14, Figure 15). The average density of roads in the Castle region is 0.2 km/km^2 and ranges from 0.0 km/km^2 to 0.5 km/km^2 among the region's watersheds.

In comparison, linear footprint densities for other parks and recreational areas in Alberta are less than half of those found in the Castle region, ranging from 0.19 km/km^2 in Banff National Park to 0.64 km/km^2 in the adjacent Kananaskis Country (Table 1).

Table 1 Density of linear footprint in parks and recreational areas in Alberta.

Protected Area	Linear Footprint Density (km/km^2)
Banff National Park	0.19
Waterton National Park	0.58
Kananaskis Country	0.64
Castle region	1.31 [a]

[a] Densities were calculated using the Government of Alberta base feature layer (Government of Alberta Base Features 2017) which captures a variety of linear footprints but excludes additional trails mapped specifically for the Castle region. For the purpose of comparison, we have used the same data sources for all density calculations summarized in Table 1, which is why the density value for the Castle region is lower than the 2.0 km/km^2 reported elsewhere in this review.

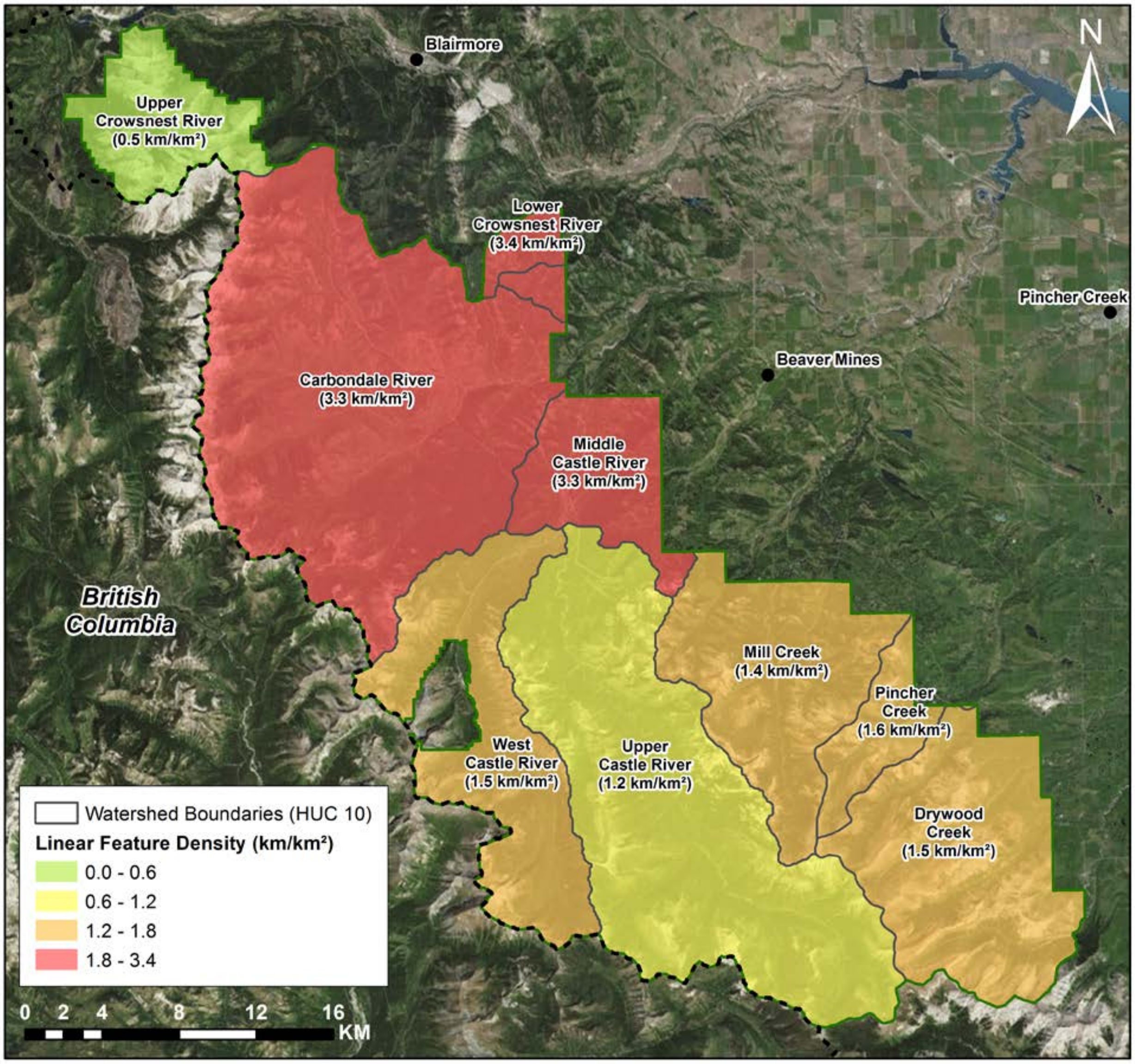

Figure 14 Linear footprint density in the 9 watersheds present in the Castle region. Densities were calculated only for the portion of each watershed in the Castle region. Data sources: Appendix A.

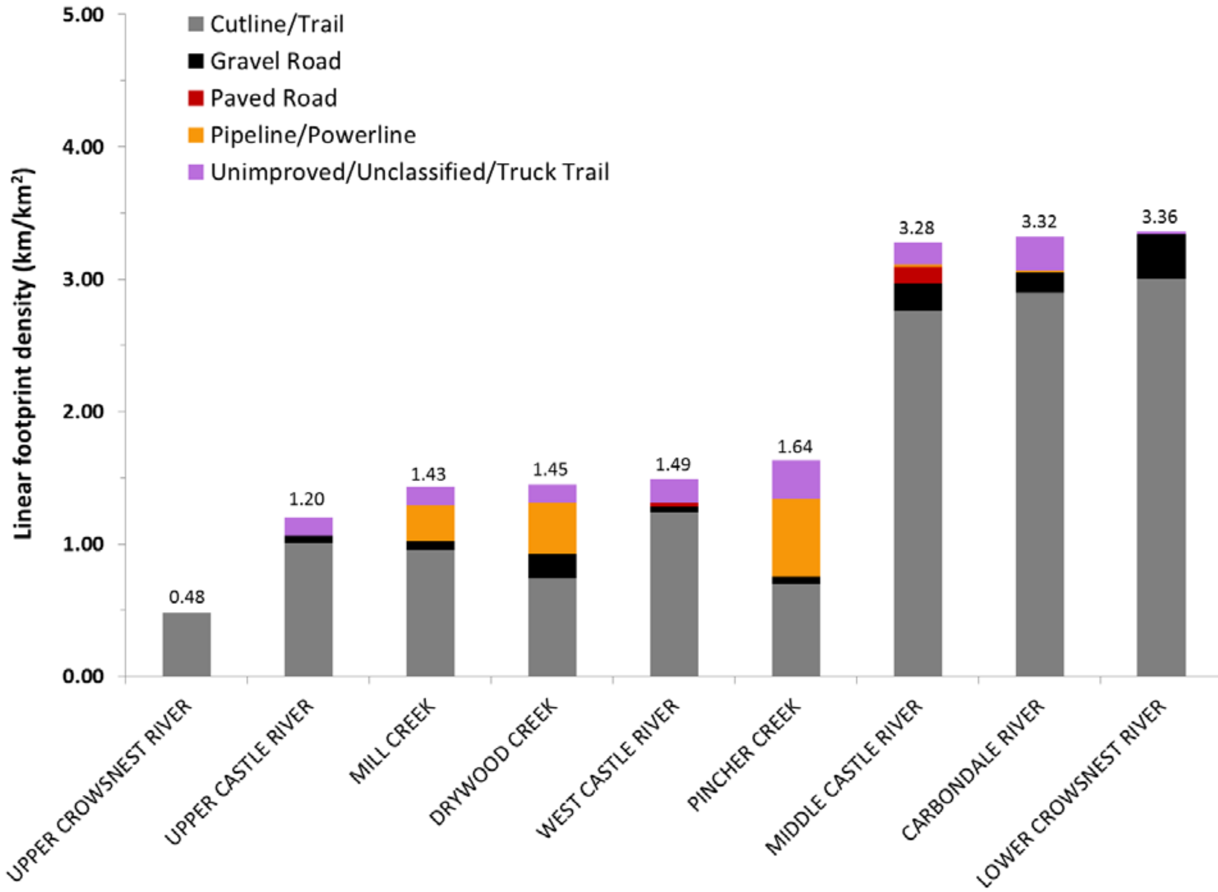

Figure 15 Summary of linear footprint density in the Castle region. Data sources: Appendix A.

Within the Castle region, 43% of the mapped linear footprint is located in areas that would classify as highly sensitive to motorized recreation, according to Meyer's (2002) sensitivity ranking based on slope of terrain (Figure 16, Figure 17).

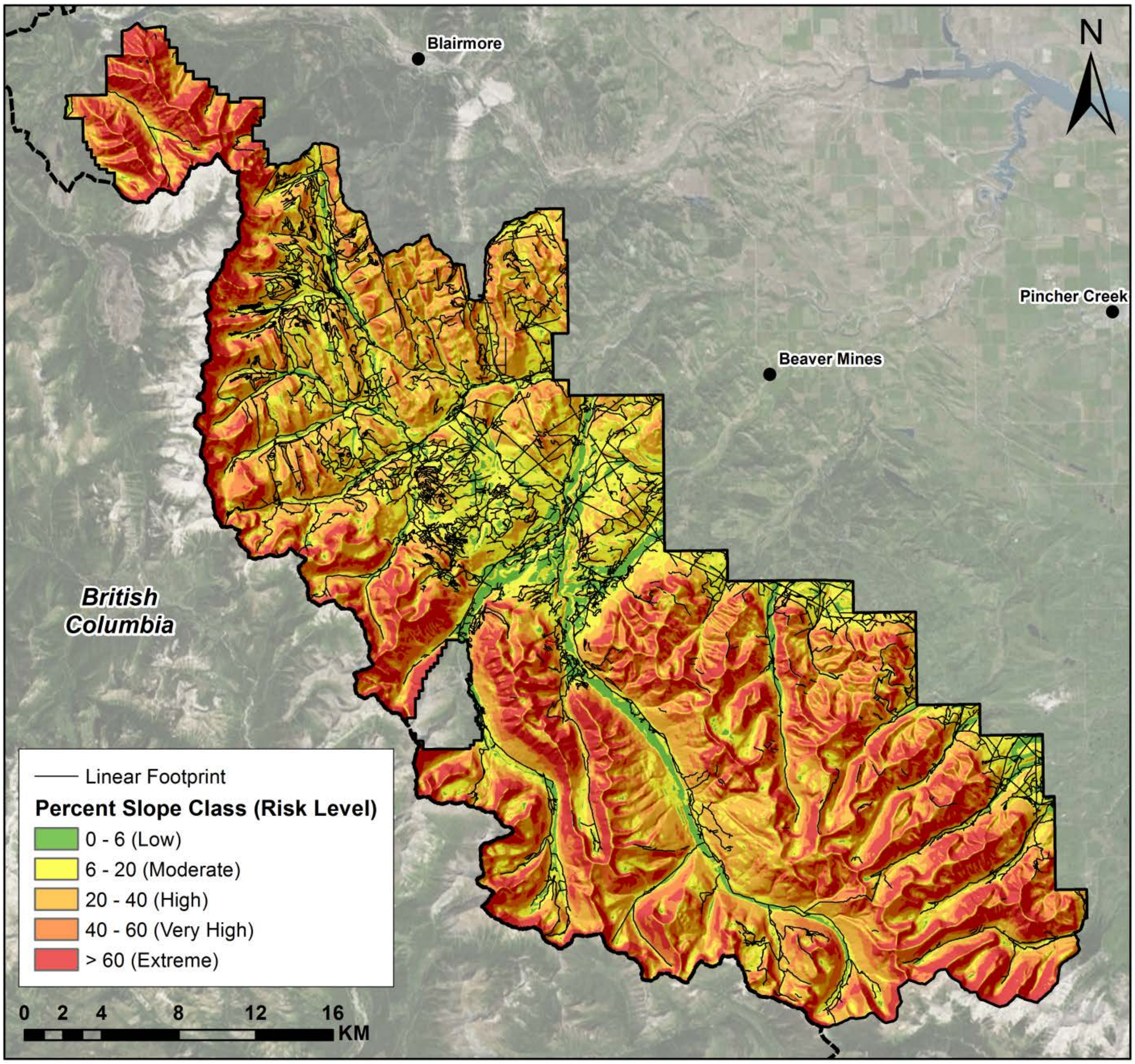

Figure 16 Slope classes in the Castle region. Data sources: Appendix A.

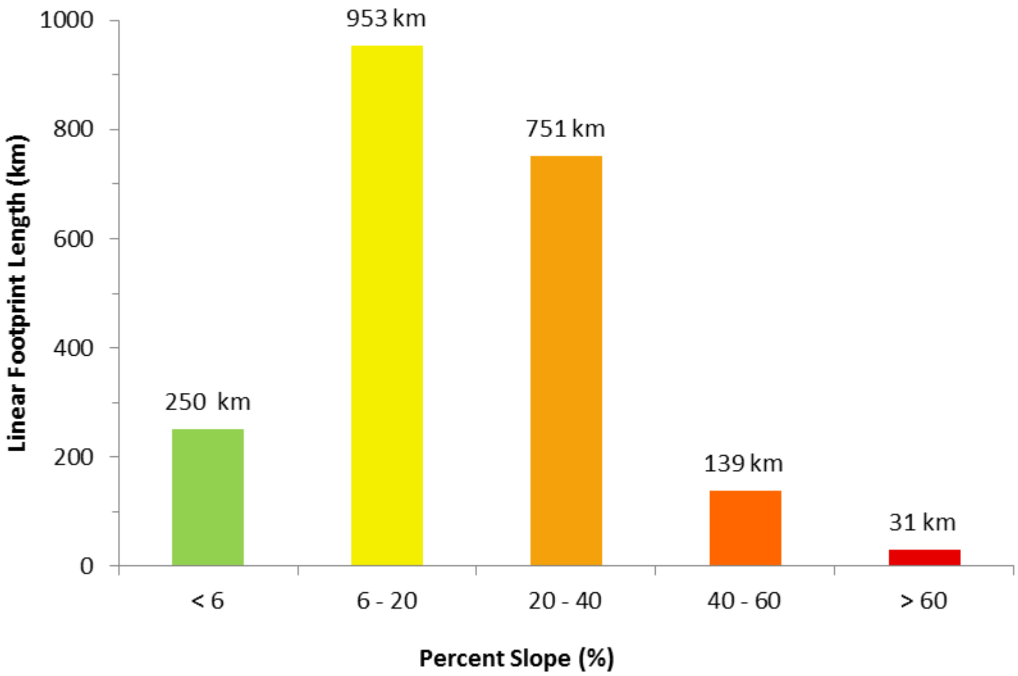

Figure 17 Summary of linear footprint length by slope class in the Castle region. Data sources: Appendix A.

Mapped linear footprints in the Castle region cross watercourses approximately 1,600 times, with 59% of these crossings occurring in headwater streams (Strahler stream order 1, Figure 18), and some streams are crossed by linear footprints more than 10 times per kilometre of stream length (Figure 19). There are approximately 35 bridges constructed for use by OHVs in the Castle region (T. Armitage, pers. comm.); these and other crossings that lack bridges may be characterized by creating rutting, pooling of water, and subsequent erosion and sedimentation (Figure 20). Where there is OHV activity at these stream crossings, even when the streams are not running they may still contribute to in-stream sedimentation when water eventually flows through these ephemeral channels.

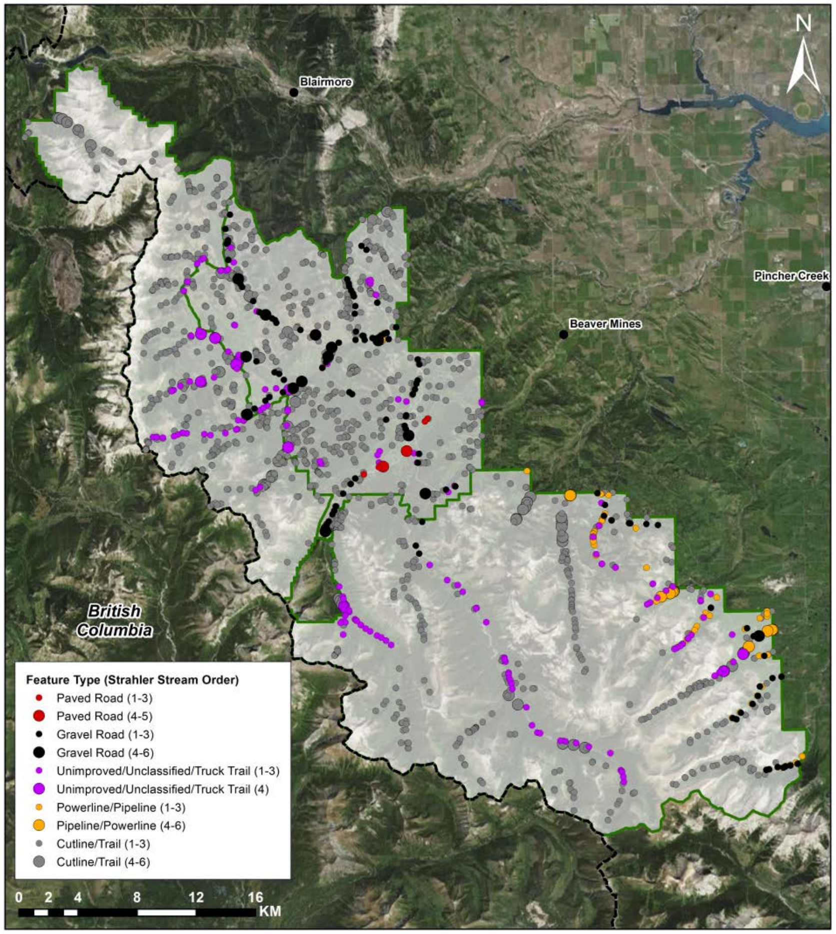

Figure 18 Stream crossings in the Castle region. Data sources: Appendix A.

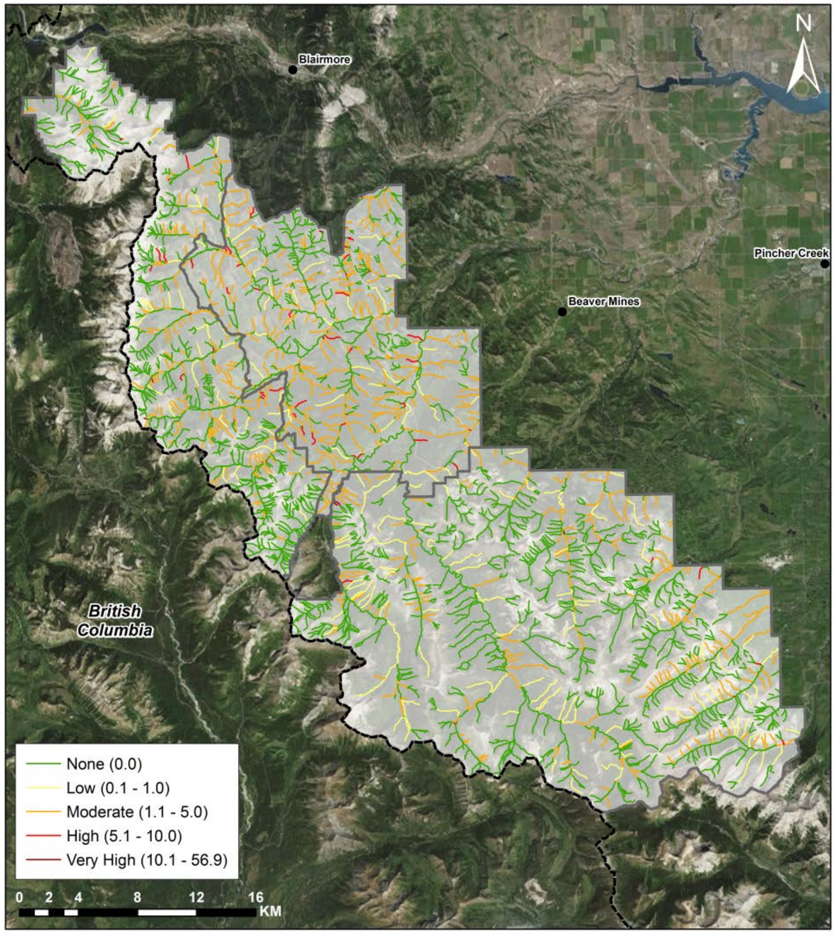

Figure 19 Stream crossing density in the Castle region. Data sources: Appendix A.

Figure 20 Trail-based rutting and water pooling (left) and multiple OHV trails crossing an unbridged stream (right) in the Castle region.

5.2 Other footprint

Non-linear human footprint in the Castle region is dominated by forest harvesting (Figure 21), much of which occurred more than 30 years ago (Figure 22). Roughly 60 well sites occur along the eastern boundary of the Castle region and range in size from 0.5 to 3 ha. Other “disturbed vegetation” consists of abandoned borrow pits (<4 ha in size) and the hamlet of Beaver Mines. In the late 1800s, forest harvesting in the region began as small independent operations, and from 1914 to 1921 much of the Castle region was included within Waterton Lakes National Park. The 1930 *Resources Transfer Act* shifted management of the area from Canada to the Alberta government. Land uses since the late 1800s in the Castle region have included cattle and sheep grazing, forestry, oil and gas development, mining, hunting, fishing, and other recreation activities. Modern forestry began in the 1950s, and was discontinued in the Castle region shortly before the protected area designation in 2016. During that intervening period, approximately 4% of forest stands in the Castle region were harvested. Consequently, on the basis of its legacy footprint, forest harvesting is the dominant form of landscape-scale anthropogenic disturbance (Figure 22), and has altered the structure and composition of forest stands in some parts of the region. The region also has 10 grazing-allotment areas dispersed from valley bottoms to alpine ridges (Grazing allotments - Distribution units 2016). Seismic exploration and subsequent oil and gas development began in the region in 1948 southwest of Pincher Creek (Flathead Transboundary Network 1999), and a total of 60 wells were drilled between 1908 and 2011 (Alberta Energy Regulator 2017).

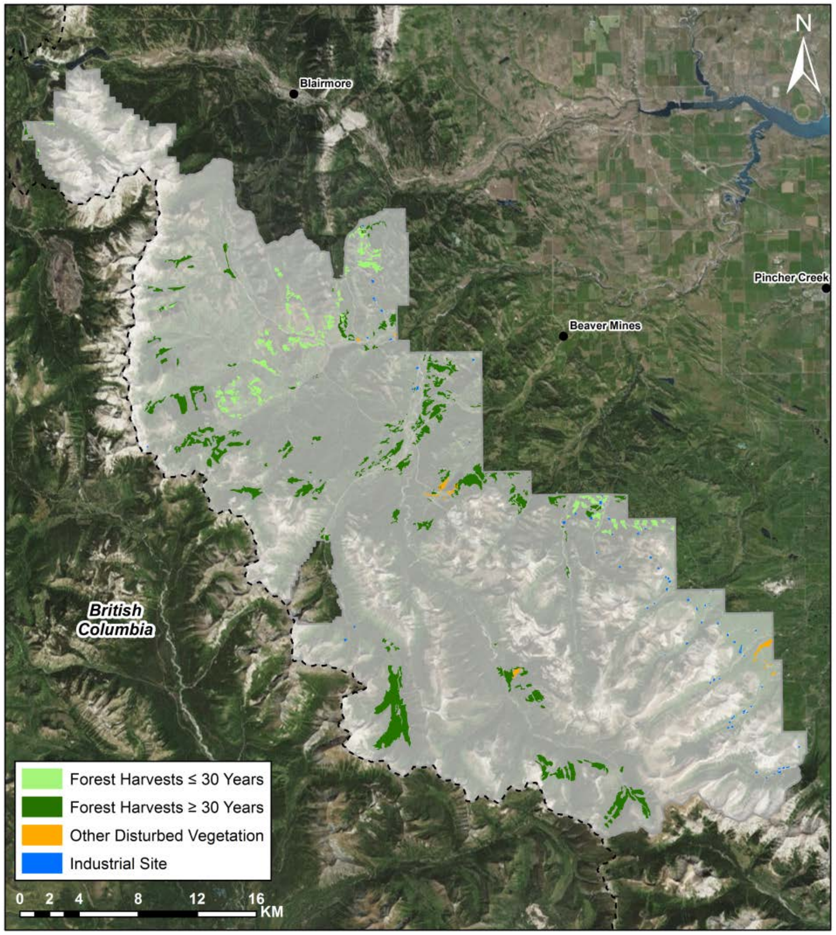

Figure 21 Human footprint in the Castle region. Data sources: Appendix A.

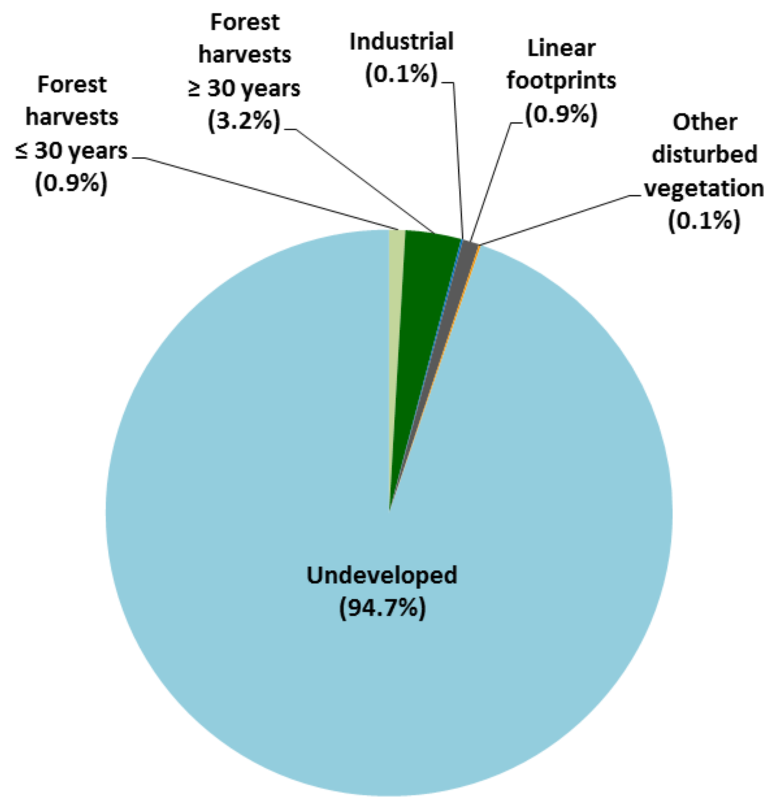

Figure 22 Summary of human footprint in the Castle region. Data sources: Appendix A.

5.3 Recreational activities

Known recreation activities in the region include hunting, fishing, hiking, natural history appreciation, photography, birding, horseback riding, camping, cycling, cross country and downhill skiing, snowboarding, and motorized recreational vehicle use. The growing network of linear footprint in the region, combined with increasingly affordable recreational off-highway vehicles, has made the region a destination for this activity. Conflicts with non-motorized users and wilderness proponents have been ongoing since the 1970s (Kariel 1997).

There is little information on the frequency and type of vehicle use in the Castle region; however, many linear footprints in the region are trails on which the use of on-highway vehicles is not possible (Figure 12). OHV registrations in Alberta increased from 80,614 to 149,804 from 2004 to 2016 (Alberta Transportation 2017), although this is likely an underestimate of OHV use in Alberta because a significant percentage of OHV regulation violations resulting in enforcement actions stem from failure to licence. Conservation officers and park rangers in the Pincher Creek District, in which the Castle region is located, spent 4,144 staff hours on public lands enforcement over a 7-month period in 2016. This resulted in 548 enforcement actions being taken, 16% of which were in relation to violations of OHV regulations (Alberta Parks 2017).

Non-motorized recreation activities associated with the use of roads and trails include hiking, cycling, cross country skiing, and equestrian activities. Other activities that occur in the area and not necessarily on trails are kayaking, canoeing, climbing, and backcountry skiing and snowboarding.

5.4 Altered fire regime

Human activities in the Castle region have altered the area's wildfire regime. In much of North America, forest fires have driven the ecological patterns within which biodiversity has adapted (Guyette et al. 2002; Kernan and Hessl, 2010; Sibold et al. 2006; Syphard et al. 2008; Yang et al. 2008). Changes to the landscape-scale patterning of vegetation caused by fire can have cascading effects on species distribution and abundance (Keane et al. 2002, Perry et al. 2011) and on the spatial distribution of subsequent fires (Hessburg and Agee 2003). Human activity during the past several decades has changed fire activity in the Castle region in two primary ways: first, improved fire suppression has likely decreased the frequency of larger fires; and second, human access and recreation have likely increased the occurrence and location of smaller-scale human-caused fires, relative to historical fire regimes.

Fire suppression is known to have altered the age and complexity of Alberta forests (Cumming 2005). The Castle region is naturally a wildfire-dominated ecosystem; however, because of fire suppression, evidence of large natural fires is relatively old, dating back to 1931-36 with the Castle Watershed, Castle River, and the Crowsnest fires (Figure 24). In the interests of commercial forest preservation, fire suppression efforts since 1950 have altered historical fire-return intervals over large areas throughout southern Alberta (Rogeau et al. 2016b). The provincial wildfire database indicates a marked absence of area burned in the Castle region since the introduction of fire suppression (Figure 25). The ecological implications of this could be profound. For instance, historically in the subalpine regions of southern Alberta (i.e., 70% of the Castle region by area), forest fires typically occurred approximately every 65-85 years, but in recent decades this has almost doubled to roughly every 149 years, notably decreasing the extent of natural wildfires (Rogeau et al. 2016a, Rogeau et al. 2016b). The resulting older, fire-suppressed forests effectively become unnaturally large fuel loads susceptible to large, intense, stand-replacing fires where mineral soils are unnaturally exposed, slopes are eroded, and few green patches remain within the fire perimeter (Rogeau 2012, Keane et al. 2002). Evidence suggests that the risk of these kinds of large fires increases with human access.

In addition to the effects of fire suppression, any fires that do occur in the Castle are now notably driven by human activities, resulting in smaller fires occurring in non-natural spatial locations. Historically, fires in the Castle region (similar to much of Alberta's Eastern Slopes) were ignited by lightning strikes, a phenomenon that has effectively now been replaced by human-caused ignitions (e.g., Figure 26). Over the past 50 years provincial data indicate that 89% of the 458 reported wildfires in the Castle region were human-caused. Moreover, because these ignitions are human-caused, the spatial location of these ignitions has changed, thereby altering not only the frequency and size of fires, but also their location on the landscape. Human-caused ignitions are now occurring within lightning strike shadows where fires were historically rare (e.g., the western portions of the Castle region; Figure 26). In addition, areas prone to lightning-induced fire now have the additive ignition frequency of human-caused fires. Although many of these human-caused fires are relatively small (less than 1 ha), recreation-caused fires (a subset of human-caused fires in the provincial records) now account for over 90% of the ignitions in the Castle region (Historical wildfire database 2017). In comparison, province-wide data between 1961 and 2014 indicate that the top 3 causes of wildfires were lightning (45%), recreation (17%) and residential (15%), although it is important to bear in mind that the "recreation" category was only introduced in 1996 (Alberta Agriculture and Forestry 2016). Accordingly, human-caused wildfires were identified as a key emerging issue in the Wildfire Management Plan released by the Calgary Forest Area in 2016 (Alberta Agriculture and Forestry 2016). Human-caused fires require mitigation in the Castle region, particularly since they are associated with access routes.

Key predictors of the occurrence of wildfires include distance from roads or trails, proximity to infrastructure or rail lines, and both human population and road densities (Brosofske et al. 2007; Syphard et al. 2007). Increased access (e.g., road density) into forested areas often increases levels of successful, accidental (e.g., campfires and debris burning), or deliberate (i.e., arson) fire ignition (Cardille et al. 2001, Prestemon et al. 2002, Guyette and Spetich 2003). Although evidence is not common that OHVs themselves are ignition sources for forest fires (however, see Baxter 2002 and Baxter 2004), most human-caused fires from 1961-2016 in the Castle region were within 100 m of a road-accessible linear footprint (Historical wildfire database 2017). This suggests that increased access into remote backcountry areas via linear footprints likely contributes to the observed increase in human-caused fires in the Castle region.

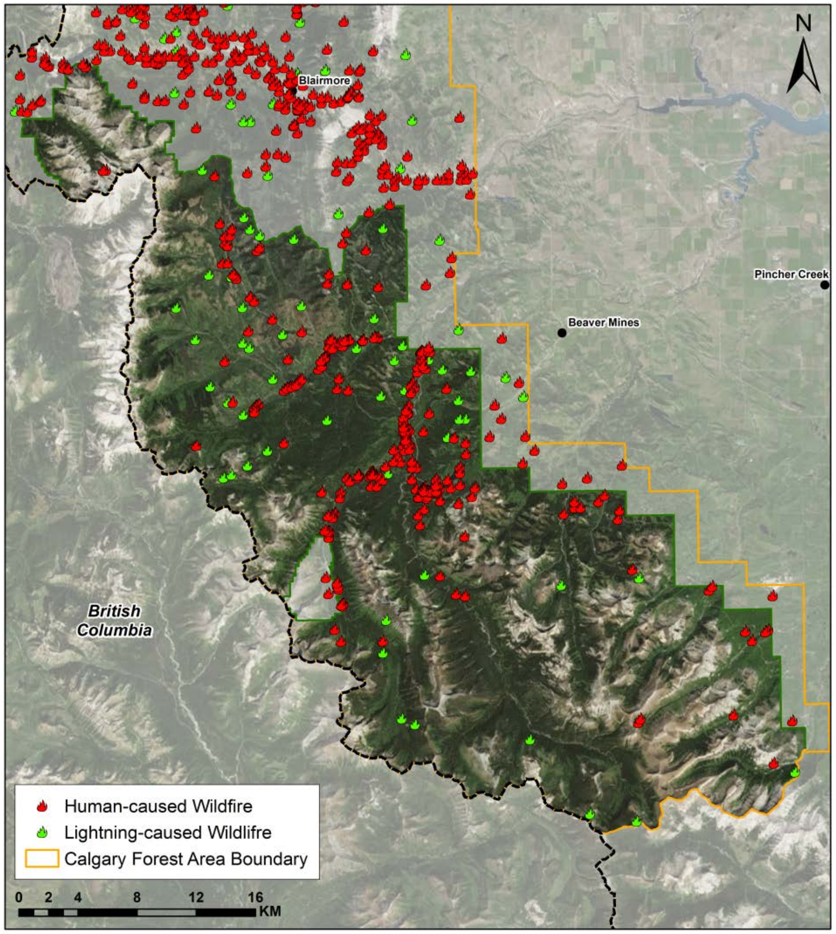

Figure 23 Reported wildfires in the Castle region, 1961 - 2016. Data sources: Appendix A.

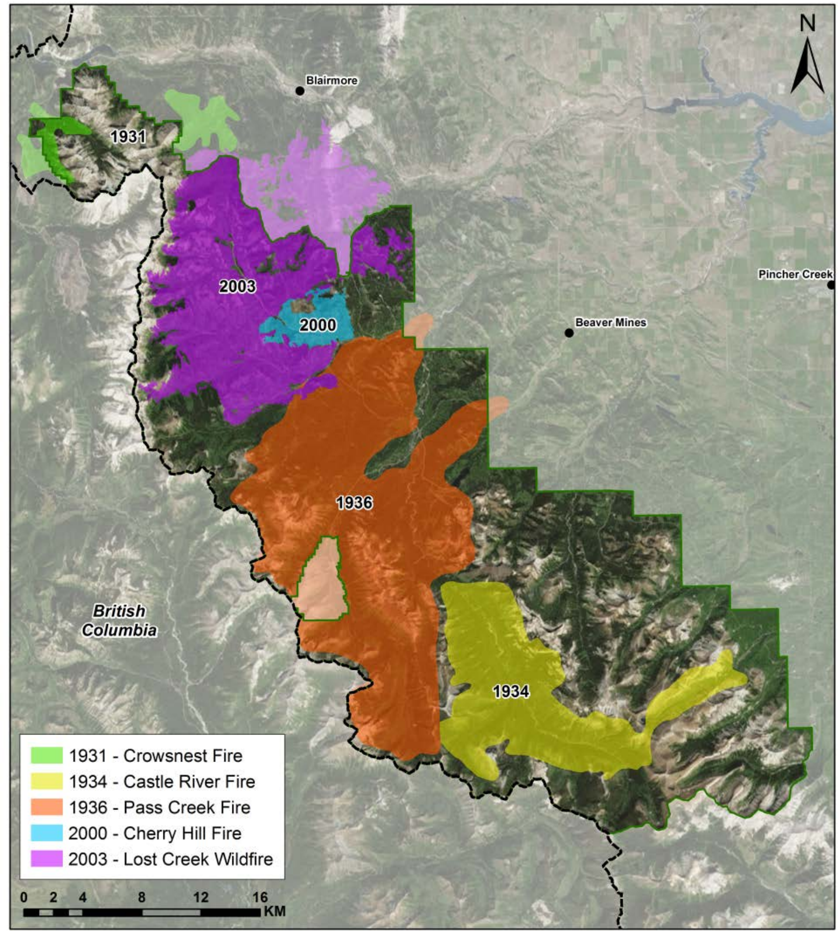

Figure 24 Historical extent of wildfires > 200 ha in the Castle region 1931 - 2016. Data sources: Appendix A.

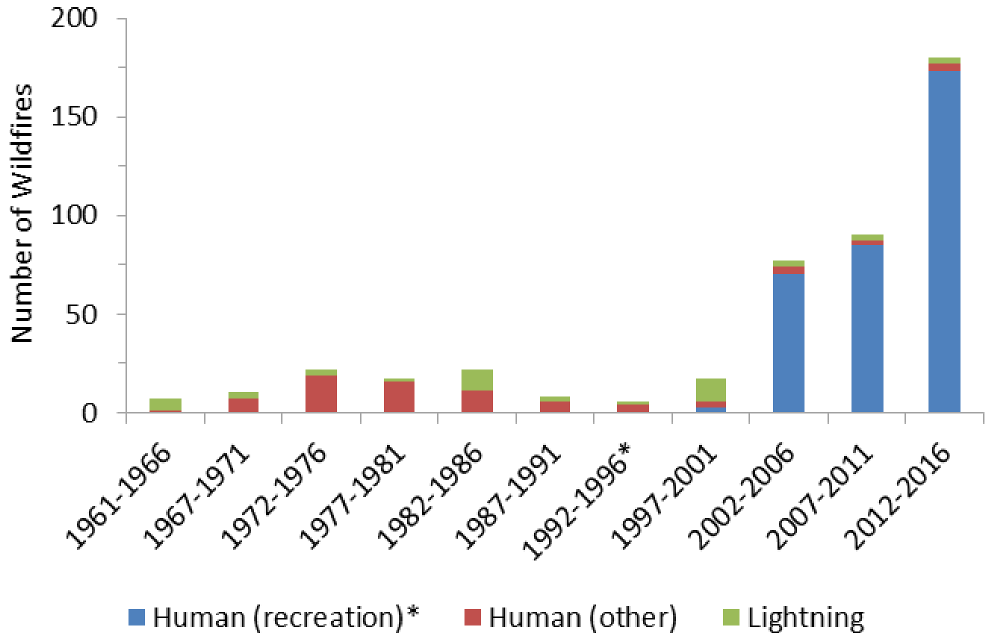

Figure 25 The number of wildfires in the Castle region, 1961 - 2016. Data sources: Appendix A.

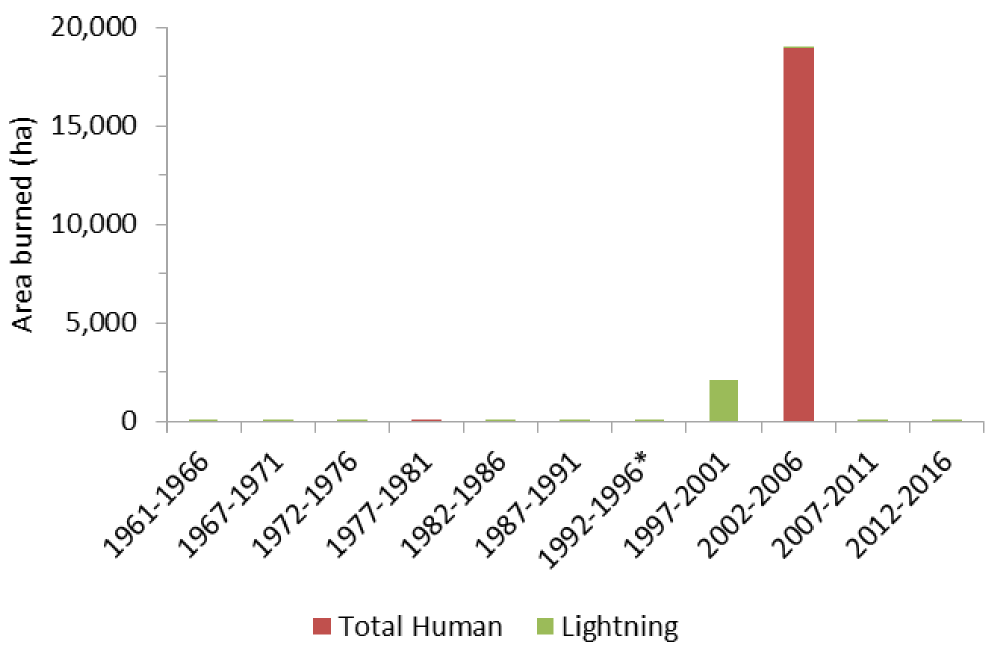

Figure 26 Area burned by wildfire in the Castle region, 1961 - 2016. Data sources: Appendix A.

5.5 Harvesting populations of fish and wildlife

Historical hunting by Indigenous Peoples dates back approximately 11,000 years in the Castle region and the area continues to be used for the practice of Aboriginal and Treaty rights by members of the Blackfoot Confederacy and other Indigenous groups (Berry and Brink 2004, Yanicki 2012). European settlers arrived sometime after 1750, and fur traders from the east arrived in the 1800s looking to trade with local trappers for beaver and other fur bearing animals. Fish and wildlife harvesting is now regulated by the provincial government, and the impacts of these activities are managed to maintain sustainable wildlife populations in the region. Recent research suggests hunting is likely the largest cause of mortality of elk populations in the region (Ciuti et al. 2012a, Thurfjell et al. 2017). Sport-fishing of both stocked fish and wild populations occurs in both lake and flowing-stream environments throughout the region. Overfishing was a primary driver of regional population declines of westslope cutthroat trout (Alberta Environment and Sustainable Resource Development 2013) and bull trout (Alberta Sustainable Resource Development 2012).

5.6 Grazing

The elimination of free-ranging bison in North America during the 1800's dramatically altered a key ecological driver: large herbivore grazing. Anthropological and paleontological evidence, along with historical records, indicate that bison populations occurred in southwestern Alberta until their extirpation in the latter half of the 19th century (Quigg 1978, Reeves 1978, Morgan 1980). Bison herds in the northern Great Plains (including the Castle region) may have been migratory or non-migratory, depending on spatial variability of forage conditions within their ranges (Epp 1988). In southwestern Alberta, some resident herds may have occupied intermountain valleys year-round (Reeves 1978), while others seasonally migrated from the plains to the foothills and intermountain valleys during the winter months (Reeves 1978, Quigg 1978). The former, referred to by some as "mountain herds", likely moved up-valley in the spring and summer months from their wintering grounds following green-up (Reeves 1978). The latter may have migrated into the foothills and intermountain valleys to overwinter in sheltered wooded areas and to graze in areas kept relatively free of snow cover by high winds and frequent chinooks (Reeves 1978, Quigg 1978).

Grazing by livestock – though not ecologically analogous to that of bison (Fleischner 1994, Hartnett et al. 1997) – has occurred in the eastern slopes of the Rocky Mountains, including the Castle region, since 1879 (soon after the extirpation of bison; Weerstra 1986). Stocking rates set in 1914 were intended to utilize 80% of the above ground biomass to reduce fire hazard; and evidence of overgrazing and soil erosion was recorded in parts of the region during the 1930's (Alberta Environment and Parks 2015). Since 1947, annual (June to October) recorded stocking rates in the largest allotment (Castle River; see Figure 27) have declined from 5,796 Animal Unit Months (AUM; a measure of grazing intensity) in 1947 to 2,320 AUMs in 1977. Stocking rates since 1977 have ranged from 2,000 to 3,000 AUMs (Alberta Environment and Parks 2015).

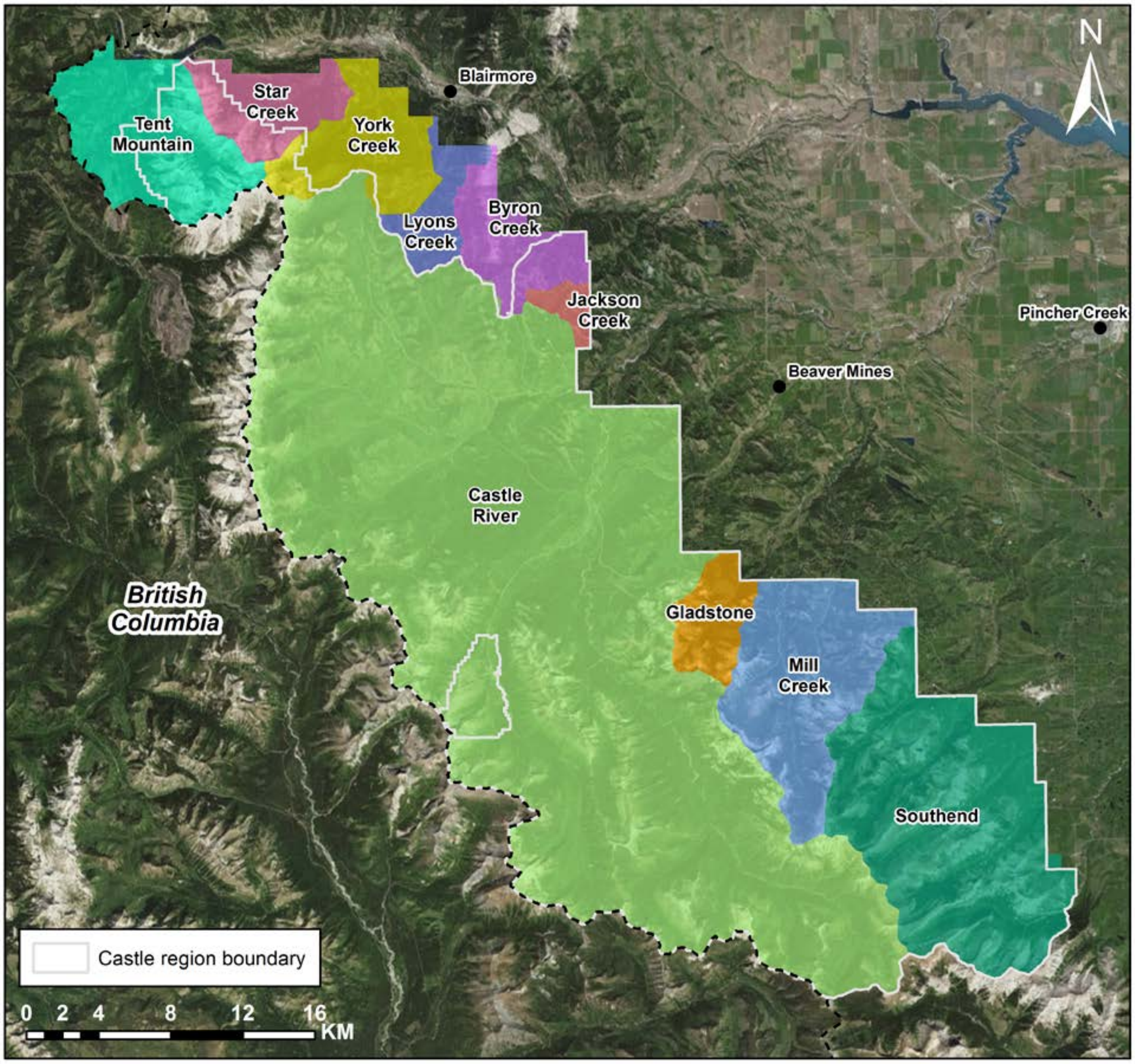

Figure 27 Grazing allotments in the Castle region. Data Sources: Appendix A.

6.0 Ecological response to human use of linear footprints

6.1 Soil and vegetation

The general ecological impacts of linear features, trails, and roads on soil and vegetation are well-documented, and in general include increased rates of soil erosion and compaction, destruction and loss of vegetation cover, loss of species richness, shifts in species composition, and habitat fragmentation (Weaver and Dale 1978, Kay 1981, Forman and Alexander 1998, Stokowski and LaPointe 2000, Kelleway 2005, Arocena et al. 2006, Nepal and Way 2007, Ouren et al. 2007, Pickering and Hill 2007, Dickson et al. 2008, Geneletti and Dawa 2009, Burgin and Hardiman 2012). While all types of human activities can be expected to have some level of negative impact, OHV use across all seasons causes a disproportionate level of impact and damage compared to non-motorized recreational activities, such as hiking, biking, and horse riding (Adams 1998, Stokowski and LaPointe 2000, Sack and da Luz 2003, Foltz 2006, Marzano and Dandy 2012, Switalski and Jones 2012, Switalski 2016). For example, soils on trails used by off-highway vehicles were found to be more compacted than comparably used and similarly located equestrian and hiking trails (Sack and da Luz 2003), and winter trail-use by snowmobiles has been shown to delay spring thaw via snow compaction and to alter plant community composition (Keddy et al. 1979, Switalski 2016).

Even when OHV trail use is limited, effects on soils and vegetation are considerable (Liddle 1997, Foltz 2006). Impacts are often irreversible (Olive and Marion 2009), and any natural recovery is either slow or nonexistent (da Luz 1999, Kinugasa and Oda 2014, Kinugasa et al. 2015, van Vierssen Trip 2015). Once OHVs remove vegetation along linear footprints, exposed soils are vulnerable to exposure, making them susceptible to erosion via wind and water (Weaver and Dale 1978, Sack and da Luz 2003, Foltz 2006, Olive and Marion 2009). With each successive pass along a trail by an OHV, the subsurface soil layer becomes increasingly compacted (Nortjé et al. 2012), and as use continues, increasing local compaction and loss of soil-anchoring vegetation leads to reduced ground permeability and increases in the frequency and intensity of surface runoff (Wilshire et al. 1978, Kay 1981, Brown 1994, Sack and da Luz 2003, Buckley 2004, Foltz 2006, Olive and Marion 2009, Switalski and Jones 2012). The shear forces from spinning tires on OHVs further contribute to and intensify erosion by creating mud holes, gullies, mounds, ruts, and erosion channels on OHV trails (see Photo 1) (Stokowski and LaPointe 2000, Hermanutz and Stavne 2009, Arp and Simmons 2012). Damage caused by OHV use is greater in turning and maneuvering areas (Liu et al. 2010, Burgin and Hardiman 2012), at stream crossings (Chin et al. 2004, Kidd et al. 2014, Marion et al. 2014), and in water-logged or poorly drained areas (Arp and Simmons 2012).

Grassland ruts and erosion channels from OHV use in the Castle region.

Risk of soil erosion within the Castle region is primarily medium (33%) or medium to high (63%), and these areas overlap extensively with areas accessed for human recreation (Figure 28; Derived Ecosite Phase 2017). Based on what has been observed elsewhere, the Castle region can be expected to experience higher rates of soil erosion in the presence of OHV use than if access were restricted to non-motorized use. However, a comprehensive assessment of the extent and intensity of human-caused soil erosion in the Castle region is lacking.

Valley floors in the Castle region are at high risk of soil rutting and compaction (Figure 29; Derived Ecosite Phase 2017). Therefore, limiting motorized access by OHVs would reduce rutting and compaction impacts associated with their use. Additionally, impacts of OHVs are increasingly being documented in less-developed and undisturbed areas. This is likely because of technological advancements that have increased the ability of OHVs to traverse difficult terrain, and increases in the number, vehicle footprint size, and use of OHVs on designated and non-designated trails (Adams 1998, Maxell and Hokit 1999, Stokowski and LaPointe 2000, Foltz 2006). If primary access points into the Castle region are not restricted in relation to OHV use, any associated or attached linear footprint may facilitate OHV access into more remote areas and into non-designated areas or trails. Particularly in treeless environments, such as at higher elevations, impacts of OHV use on soils and vegetation can be expected to be the most severe and persist for far longer (Crisfield et al. 2012).

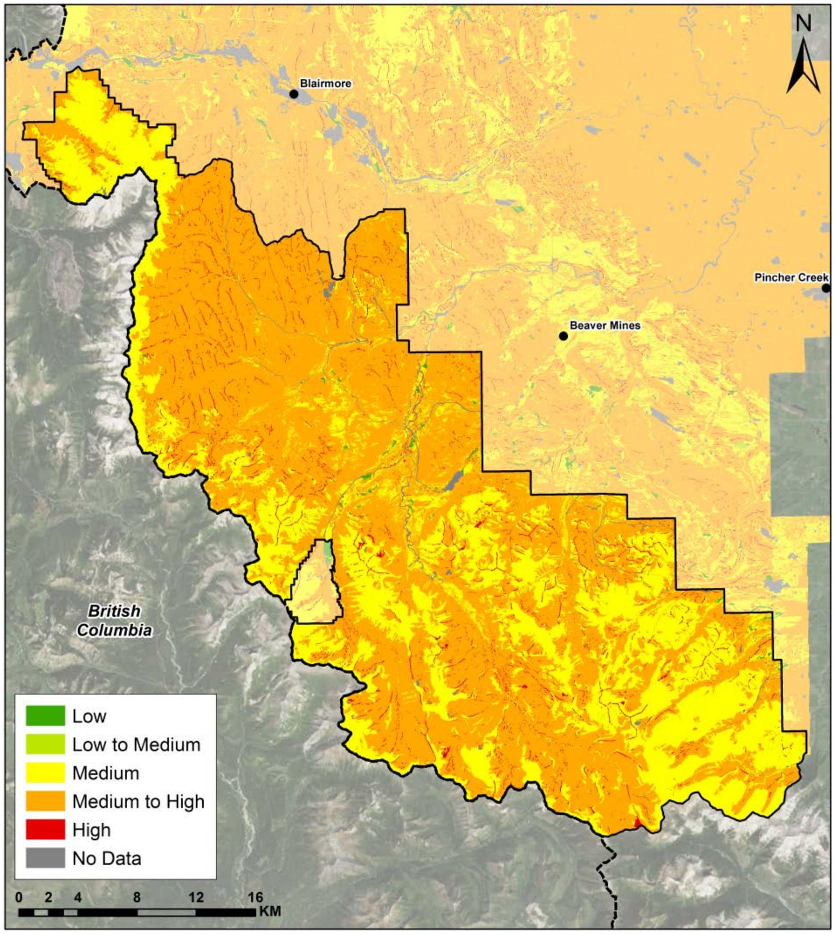

Figure 28 **Soil erosion hazard in the Castle region. Data Sources: Appendix A.**

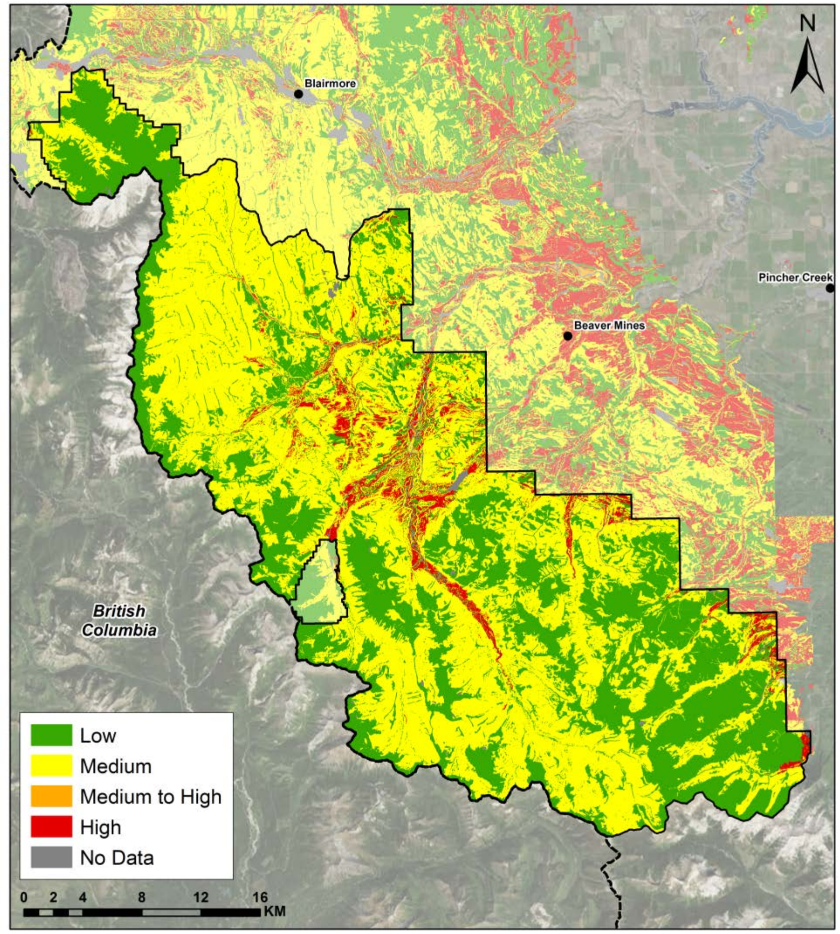

Figure 29 **Soil rutting and compaction hazard in the Castle region. Data Sources: Appendix A.**

Once soils are affected, impacts of human trail use extend directly to affect vegetation communities. Native plants along OHV trails may be weakened, malformed, and more susceptible to disease and insect predation, and any germinating seeds can be damaged by trampling (Switalski and Jones 2012). Wind-borne soil and other surface material from passing OHVs can settle on vegetation up to 100 m away from trails (Padgett et al. 2008), causing reduced productivity within plant communities adjacent to trails as a result of impaired photosynthesis, decreased reproductive capacity, and reductions in plant size, density, and diversity (Groom et al. 2007). Disturbance of vegetation within and adjacent to linear footprints also alters plant community composition and species richness (Kelleway 2005, Nepal and Way 2007, Pickering and Hill 2007, Dickson et al. 2008).

Vegetation loss and soil compaction associated with OHV use contributes to conditions that favour invasive species (Stokowski and LaPointe 2000, Havlick 2002, Foltz 2006, Goossens and Buck 2009, Hermanutz and Stavne 2009), a leading direct and indirect cause of biodiversity loss (Didham et al. 2005). There is often increased establishment of invasive and/or non-native plant species near linear footprints (Hansen and Clevenger 2005, Rooney 2005, Bella 2011) and at human recreation sites (Anderson et al. 2015), which is likely facilitated by the ecological traits of many invasive species that enable establishment and rapid growth on disturbed soils (Parendes and Jones 2000). Spread and colonization of invasive or non-native species is facilitated further via transportation of seeds and vegetative structures by off-highway vehicles (Adams 1998, Rooney 2005, Hermanutz and Stavne 2009). Aquatic invasive species (both plants and invertebrates) may also be transported into wetlands, watercourses, and lakes via seeds or eggs that are in mud attached to OHVs (Waterkeyn et al. 2010, Banha et al. 2014).

Conditions currently exist in the Castle region for the establishment and spread of invasive and non-native species due to OHV use. The Castle region contains rare, unique, and sensitive vegetation communities, such as old growth forest, big sagebrush, and several distinctive plant communities where vegetation zones overlap. Many of these communities and the species that comprise them are inherently more vulnerable and ecologically fragile, and have lower or longer rates of recovery when disturbed, thereby increasing their susceptibility to impacts of OHV use (Geneletti and Dawa 2009, Tommervik et al. 2012).

Importantly, the processes of vegetation loss, soil compaction and erosion, and invasive species establishment do not occur in isolation, but operate as a series of positive feedbacks that lead to further degradation and loss of natural vegetation (Crisfield et al. 2012, van Vierssen Trip 2015). Ultimately, while intensity of trail use and sensitivity of the affected environment determine the level of impact, it should be understood that the mere presence of OHV is a greater determinant of the degree of associated negative environmental effects than varying levels of OHV use (Foltz 2006, Olive and Marion 2009). Therefore, when considered along with the high risks associated with erosion, soil rutting and compaction in the Castle region, impacts from OHV use can be expected to be severe and long-lasting in sensitive areas.

6.2 Water quality

As described in section 4, headwater streams are a dominant feature of the hydrological system in the Castle region. There are 2,029 km of streams and rivers in the Castle region, most of which (91%) are headwater systems (i.e., Strahler stream order 1 to 3). While water quality parameters are not monitored in the region, the quantity of sediment deposited and suspended in mountain streams is generally driven by the composition of surface materials in the stream bed and surrounding watershed, and landscape slope, water infiltration capacity, and precipitation. Rates of sediment input are highly variable over time, with increased streambed scouring during high flow conditions and increased runoff of sediment during precipitation events with overland flow.

Linear footprints cross watercourses a total of 1,614 times in the Castle region, with 94% of these crossings on headwater streams (Strahler order 1 to 3; Section 3.1). Increased use of roads and trails in the Castle region will likely increase the input of surficial materials to streams, primarily via disturbed surface materials and damaged soil-holding vegetation, thereby altering water quality by increasing the concentration of suspended sediment and nutrients. Linear footprints cross some streams in the study area more than 10 times per stream-kilometer, with each crossing creating a potential source or pathway for sediment transport and input to streams. For perspective, in a montane watershed in Colorado with a trail density of 0.2 km/km^2 (significantly lower than the Castle region's trail density of 0.9 km/km^2), sediment production from OHV trails was three times greater than from forest roads, with sediment production positively correlated with trail slope, traffic load, and the amount of unconsolidated material on the surface of OHV trails (Welsh 2008). Given the higher trail density in the Castle region, particularly in the Middle Castle River, Carbondale River, and Lower Crowsnest River watersheds, it is highly likely that sediment production is significantly higher than observed in the Colorado study.

Increased sediment concentrations have been demonstrated in streams with OHV crossings in both forested watersheds and non-forested areas (Chin et al. 2004, Marion et al. 2014). In some landscapes, OHV trails may also alter flow regimes and invertebrate communities in streams they cross (Arp and Simmons 2012, Kidd et al. 2014). In areas with high OHV use, trail usage can change the overall hydrology of the area by creating new flow pathways and, therefore also result in increased sediment movement (Ouren et al. 2007). The creation of drainage pathways has been found to increase drainage density in headwater streams (Arp and Simmons 2012) and may also increase the delivery of contaminants to streams (Ouren et al. 2007, Trombulak and Frissell 2000).

Of particular concern for water quality is the dynamic nature of OHV trails, especially at river crossings (Arp and Simmons 2012, Marion et al. 2014). At these crossings, there is commonly rutting and/or pooling of water that create trail obstacles and lead to subsequent trail braiding, where one trail becomes many (Arp and Simmons 2012). The resulting multiple creek crossings exacerbate erosion and sedimentation. This is especially important at stream crossings where water quality is most directly affected by sediment mobilization and where changes to channel morphology can occur rapidly (Marion et al. 2014, Trombulak and Frissell 2000).

6.3 Stream trout

Land use and human activity, including use of linear footprints, can increase sediment input into streams (Opperman et al. 2005). Such sediment inputs may limit the recruitment of salmonids, including westslope cutthroat trout and bull trout (Al-Chokhachy et al. 2016). Furthermore, use of linear footprints that cross streams tends to widen them, reduce water depths, and homogenize habitats, and the number of upstream road crossings is positively correlated with presence of hybridization between westslope cutthroat trout and rainbow trout, possibly because changes in availability and quality of habitat that can follow land use disturbance may contribute to successful invasion and establishment of non-native competitors (Muhlfeld et al. 2009b).

In the Castle region, there are an estimated 390 crossings of the 519 km of streams in which there is evidence of occupation by westlope cutthroat trout, and 45 of which occur on reaches of federally designated critical westslope cutthroat trout habitat (Figure 30). It must be noted, however, that the current extent of federally designated critical habitat is based on data from genetic surveys conducted in 2013, and does not represent all areas occupied by genetically pure populations of westslope cutthroat trout. Nor does it represent all areas necessary to recover the species. Fisheries and Oceans Canada has proposed an expansion of critical habitat upstream and possibly downstream of existing critical habitat. Thus, the number of crossings of critical habitat will increase as the extent of identified critical habitat increases. There are also approximately 268 crossings of the 330 km of streams occupied by bull trout, including 39 crossings of reaches with evidence of bull trout spawning (Figure 31). Similar to westslope cutthroat trout, surveys of bull trout spawning have not been exhaustive, and spawning could occur in all areas occupied by the species.

Stream crossings are problematic for these trout species primarily because they are sensitive to sedimentation. Field experiments have shown that both bull and cutthroat trout embryo survival was negatively related to the amount of fine sediment in their spawning excavations (Bowerman et al. 2014, Magee et al. 1996). Similarly, an increase in trout physiological stress has been linked to periods of high sediment loads (Reid et al. 2003). Finer silts and clays from disturbed or eroded soils may be transported much greater distances overland through riparian buffers (Cooper et al. 1987, Lowrance et al. 1984) and it is this finer component of sediment that is most detrimental to salmonids such as westslope cutthroat trout. Fine sediment loading reduces hyporheic exchange by reducing the interstitial pore size of streambed sediment or by cementing larger particles together, and may interfere with spawning site selection and development of embryos, or cause entombment of emerging alevins (Sear et al. 2008). Alternatively, silt and clay particles may adhere to membranes of eggs, effectively sealing pores that must be permeable to supply oxygen to developing embryos (Greig et al. 2005, Julien and Bergeron 2006).

Beyond the spawning, developmental and hatching portion of the salmonid life cycle, fine sediment can have further negative impacts on later life stages. Juvenile salmonids often use large substrate as cover. Practices that increase sedimentation may result in a reduction of such critical nursery habitat used through the first years of life for these species (Watson and Hillman 1997). Furthermore, sediment may have trophic impacts by reducing hyporheic zone habitat for macroinvertebrates (Weigelhofer and Waringer 2003). These organisms serve a vital trophic role in the stream food web and a reduction in their abundance may limit fish production. Sedimentation and turbidity can also contribute to decreases in primary production at the base of the local food chain (Henley et al. 2010).

Increased sedimentation associated with linear footprints has been linked to population reductions of stream trout. Valdal and Quinn (2011) found a negative relationship between the abundance of westslope cutthroat trout and road density (which ranged from 0.0 to 1.5 km/km^2; Valdal 2006) within 100 m of headwater streams in southeastern BC. Declines in westslope cutthroat trout abundance were also positively associated with areas of high soil erodibility in near-stream zones. Elsewhere, road density negatively affected westslope cutthroat trout populations in US Forest Service and Bureau of Land Management lands (Quigley and Arbelbide 1997), and the abundance of bull trout spawning excavations in Montana was negatively correlated with both road and stream-crossing density (Baxter and McPhail 1999). Mayhood (2000) concluded that most watersheds containing westslope cutthroat trout habitat along the Rocky Mountain Eastern slopes are at risk of damage because of land use changes and resource extraction.

Similar negative responses to increasing road densities have been observed for bull trout. For instance, Ripley et al. (2005) reported negative responses of bull trout populations to roads across a range of road densities from 0.0 to 1.6 km/km^2 in the 3,475 km^2 Kakwa River basin of west-central Alberta from 1994 to 2001. Bull trout were 50% less likely to be found where road density was greater than 0.4 km/km^2, when compared to roadless areas, and were predicted to be absent where road density exceeds 1.6 km/km^2. Similar results have been found in Idaho (Dunham and Rieman 1999), and in the Columbia and Klamath River basins in the western United States (Rieman et al. 1997). Similarly, Quigley and Arbelbide (1997) found that Columbia Basin watersheds with evidence of bull trout spawning and rearing had considerably fewer roads (0.3 km/km^2) compared to watersheds with less vigorous bull trout populations (0.9 km/km^2). Bull trout typically were absent at an average road density of 1.1 km/km^2. More recently, Kovach et al. (2016) found that the interaction of roads and non-native salmonid fish species contributed to lower bull trout abundances and declining populations. While density of linear footprints in the 9 Castle region watersheds (including those used by OHVs) ranges from 0.5 to 3.4 km/km^2, the density of roads ranges from 0.0 to 0.5 km/km^2, well within the range at which negative responses of bull trout were observed elsewhere.

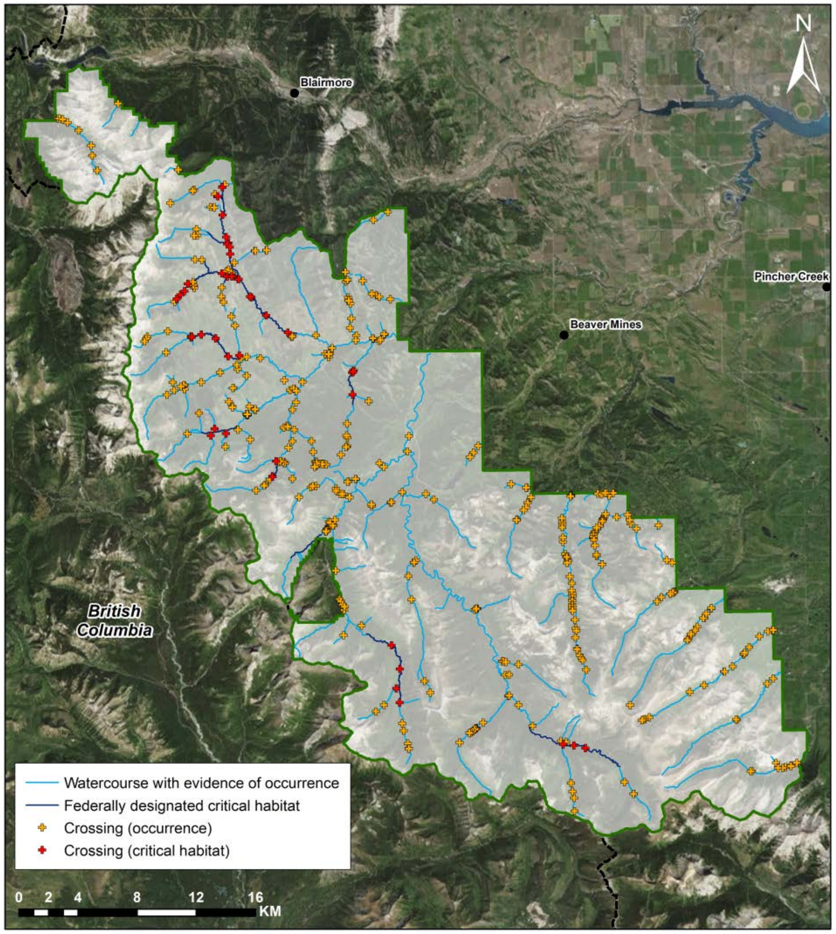

Figure 30 **Crossings of linear footprints on stream reaches with confirmed westslope cutthroat trout occurrence, and federally designated critical habitat. Data sources: Appendix A.**

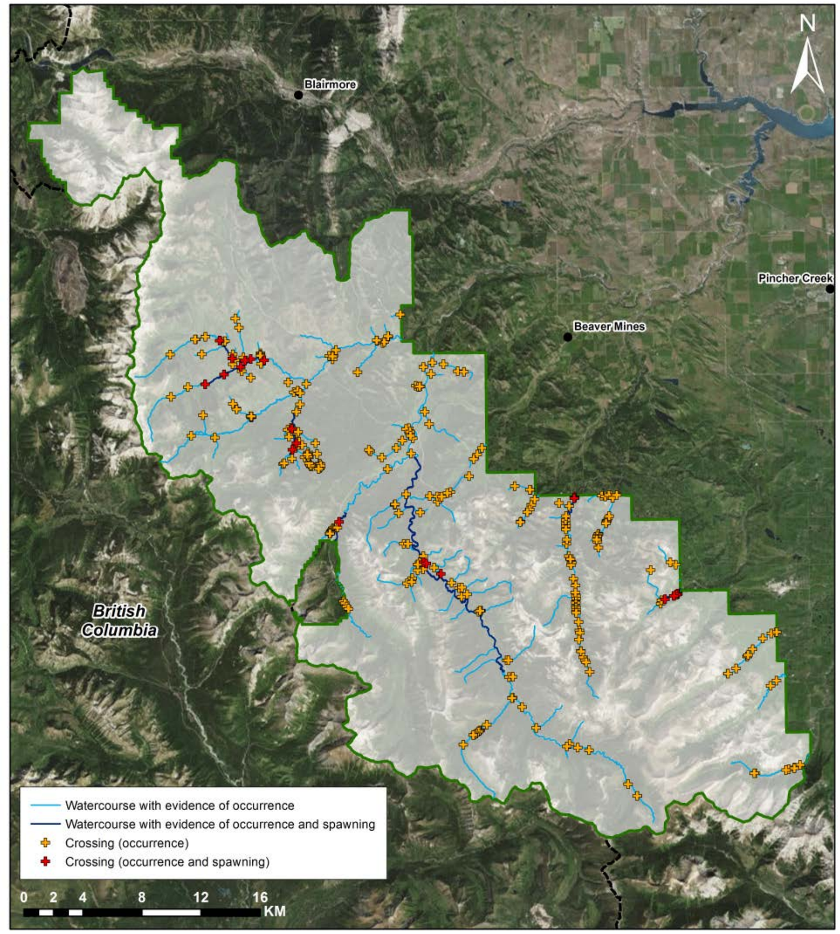

Figure 31 **Crossings of linear footprints on stream reaches with confirmed bull trout occurrence and spawning areas. Data sources: Appendix A.**

6.4 Grizzly bears

Direct effects of access on grizzly bears

There is a large body of literature describing the relationship between human access and grizzly bear survival. Roads can provide humans access into grizzly bear habitat (Nielsen et al. 2004, Schwartz et al. 2006), and human-related causes are the primary source of grizzly bear mortality across North America, including areas where the hunting of bears is not allowed (Peek et al. 1987, McLellan et al. 1999, Benn and Herrero 2002, Garshelis et al. 2005, Schwartz et al. 2006, McLellan 2015). Most human-caused grizzly bear mortalities in Alberta and British Columbia are less than 500 metres from a road (Benn and Herrero 2002, Boulanger and Stenhouse 2014, McLellan 2015), or within 200 metres of a trail (Benn and Herrero 2002).

Recent studies have shown that grizzly bear survival is reduced in areas of high road density (Schwartz et al. 2010a, Boulanger and Stenhouse 2014). Boulanger and Stenhouse (2014) showed that females with cubs have lower survival rates in areas of high road density relative to females without cubs. They observed an ecological threshold at a road density of 0.75 km/km^2, above which they predicted population decline. The density of roads in Castle region watersheds is below this ecological threshold (with a maximum road density 0.5 km/km^2. However, the density of all linear footprints in the Castle region, including those used by OHV's, is above this threshold in most watersheds (range 0.5 to 3.4 km/km^2, average 2.0 km/km^2, Figure 15).

Further, grizzly bear survival is positively associated with the availability of habitat distant from roads (Schwartz et al. 2010a). The area of roadless habitat has been used as a measure of habitat security for grizzly bears in some jurisdictions (Interagency Grizzly Bear Committee 1998, United States Fish and Wildlife Service 2003, Schwartz et al. 2010a). Numerous researchers also have suggested that closing roads or restricting motorized access in high-quality bear habitat is favourable for grizzly bear survival (Mace et al. 1996, Roever et al. 2010, Schwartz et al. 2010a, Northrup et al. 2012, Boulanger and Stenhouse 2014).

Indirect effects of access on grizzly bears

Aside from contributing to direct mortality of grizzly bears, indirect effects of roads on grizzly bears, including behavioural and distributional responses, have been well documented. (McLellan and Shackleton 1988, Mace and Manley 1993, Mace et al. 1996, Gibeau et al. 2002, Roever et al. 2008, Graham et al. 2010, Northrup et al. 2012, Fortin et al. 2016). Grizzly bears may avoid or select areas near roads depending on a variety of demographic (age and sex class) and environmental factors (including habitat quality, season, traffic volumes, time of day, etc.). Roadside vegetation can be of high nutritional value for bears (Roever et al. 2008, Graham et al. 2010, Boulanger et al. 2013), but these nutritional gains may be offset by associated decreases in survival and reproduction in areas of high open road density (Boulanger et al. 2013). The spatiotemporal displacement of bears by people may decrease nutritional intake and increase energetic costs (Fortin et al. 2016). Grizzly bears have been shown to move faster near roads, and to be more nocturnal in response to human activity (Gibeau et al. 2002, Roever et al. 2010, Schwartz et al. 2010b, Northrup et al. 2012).

McLellan and Shackleton (1988) documented grizzly bear avoidance of areas within 250 m of roads and estimated a 58% reduction in functional habitat within 100 m of roads. Avoidance of roads by grizzly bears in their study was independent of traffic volume, suggesting that even a few vehicles can displace bears, particularly in remote areas where they may be less habituated to human activity. McLellan and Shackleton (1988) also found that in spring females with cubs selected areas near roads more frequently than other demographic cohorts, possibly to avoid adult males, which used areas close to roads less frequently. Mace and Manley (1993) observed declines in habitat use by grizzly bears when open road density exceeded 0.5 km/km^2, and found that closing roads positively affected grizzly bear use once open road densities were reduced below 0.6 km/km^2.

Subsequent analyses by Mace et al. (1996) showed that road density (both open and closed) within female grizzly bear home ranges was significantly less than outside of home ranges (0.6 km/km^2 vs. 1.1 km/km^2), suggesting that females avoided areas of higher road density. Furthermore, they found that bears avoided habitats close to roads (within 500 m) when traffic volumes exceeded 10 vehicles per day.

A study by Gibeau et al. (2002) showed that adult female grizzly bears were more impacted by human activity and development, and remained further away from roads relative to other cohorts, regardless of habitat quality. They also found that grizzly bears were less likely to use habitats close to trails unless habitat quality was high, and that they preferred to utilize high quality habitats close to trails during the human inactive period, rather than when human activity was high.

Research by Graham et al. (2010) showed that females with cubs selected areas close to roads more than expected in the spring (similar to the findings of McLellan and Shackleton 1988), and found that females were more likely to cross roads than males. Increased use of habitat near roads may increase the likelihood of negative interactions with people, and thus a higher risk of grizzly bear mortality. Furthermore, analysis of survival data suggested that mortality rates were highest for young bears that used areas close to roads (Graham et al. 2010). High mortality rates of young females would have negative long-term effects on population viability, particularly when population density is low.

In another example, in southwestern Alberta, grizzly bears selected habitats on private agricultural lands with high road density but low human use relative to the multi-use public lands (Northrup et al. 2012). Habitat near roads with fewer than 20 vehicles per day was more likely to be used by grizzly bears, while bears avoided roads receiving moderate traffic (20-100 vehicles per day) and strongly avoided high-use roads (more than 100 vehicles per day; Northrup et al. 2012).

Effects of off-highway vehicles on grizzly bears

The literature to date is sparse on the effects of OHV use on grizzly bears. Recently completed work suggests that some bears respond negatively to high levels of OHV use on trails; however, there was substantial variation between individual bears and not all age/sex groups responded similarly (Ladle 2017). Though the sample size of bears was small, a study in Montana demonstrated that grizzly bears used areas near OHV trails less than expected when compared to reference sites (Graves 2002).

Modelling grizzly bear habitat value and mortality risk in the Castle region

A suite of analysis tools developed from published research are currently used by the Government of Alberta to evaluate grizzly bear habitat and inform decision making to support grizzly bear conservation in the province (GBTools 2016; fRI Research Grizzly Bear Program, 2016). These tools include resource selection function (RSF) models (from Nielsen et al. 2002) that are specifically tailored to each grizzly bear population unit in Alberta and can be used to estimate habitat value for bears. Additionally, they include a mortality risk model from Nielsen et al. (2004) that can be used to evaluate the possible impact of motorized access restrictions on grizzly bear populations. Together, these models were used to evaluate habitat value and mortality risk for the Castle region (Stenhouse and Morehouse, in prep).

Modelling grizzly bear habitat value in the Castle region

The RSF models originally developed by Nielsen et al. (2002) represent the relative probability of grizzly bear use of habitats and landscapes and are generally used as surrogates for habitat value (though selection may not always equate with habitat quality). The RSF models, when applied to the Castle region using the best available spatial datasets, indicated that both areas contain a significant amount of high value grizzly bear habitat (i.e., a high RSF value). A large portion (62%) of the Castle region north of Highway 774 was identified as high value grizzly bear habitat, whereas 87% of the area south of the highway was identified as high value habitat (Figure 10). Realized habitat quality for grizzly bears, however, also depends on the security of an area, which is strongly related to motorized access.

Modelling current and future grizzly bear mortality risk in the Castle region

The grizzly bear mortality risk model was originally developed by Nielsen et al. (2004) based on the relationships between grizzly bear mortality data and a variety of other environmental and anthropogenic variables in Alberta's Yellowhead grizzly bear population unit. The model predicts the relative probability of human-caused grizzly bear mortality as a function of landscape variables such as terrain, proximity to roads and trails, and land-use. Nielsen et al. (2004) found that grizzly bear mortalities were positively associated with human access, which is consistent with the scientific literature on the relationship between motorized access and grizzly bear mortality. To evaluate the possible effects of motorized access restrictions in the Castle region, mortality risk was first evaluated under current access conditions. Mortality risk was then assessed for future conditions using proposed access networks for the Castle Provincial and Wildland Provincial Parks. Comparisons of current and future scenarios indicated that motorized access restrictions would reduce grizzly bear mortality risk in both study areas.

6.5 Other wildlife

Previous comprehensive reviews have been conducted examining linear footprint impacts on wildlife. In a review of over 187 peer-reviewed studies, Trombulak and Frissell, (2000) concluded that impacts of roads on wildlife are generally negative, with impacts that include an increase in mortality rates, vehicle collisions, human-wildlife conflict, and hunting and fishing pressure, as well as alterations of animal behaviours and chemical and physical environments. Likewise, Gaines et al. (2003) reviewed 238 sources and found the most common impacts on wildlife species were altered habitat use from human-caused displacement and avoidance, and disturbance at a specific site during a critical life-history stage (e.g., nesting, breeding grounds, etc.). They found negative impacts arose from both motorized and non-motorized activities, the severity of which was contingent on the wildlife species. Similarly, Ouren et al. (2007) reviewed over 700 peer-reviewed studies focussed on the impacts of OHVs to ecological systems. They found that both the noise and the physical presence of OHVs in wildlife areas effectively reduced habitat connectivity, changed animal movements, and altered population and recolonization dynamics. They concluded that disturbance effects can range from physiological impacts to altered behaviors, which can lead to declines in local population size, survivorship, and productivity.

Overall, the effects of human activity on wildlife varies depending on the area, type of human activity, wildlife species, seasonality, and life history of the species, but the scientific consensus is that human activities along linear footprints generally translate into negative impacts on wildlife (Quinn and Chernoff 2010, Ciuti et al 2012b, Hebblewhite and Merrill 2008, Muhly et al 2013, Gaines at al 2003). Furthermore, many of these impacts may increase as people venture further off-trail (Taylor and Knight 2003, Stankowich 2008, Borkowski et al 2006). These include:

- Physical disturbance and habitat degradation;
- Access-facilitated over-harvesting;
- Altered behaviour and the associated energy demands and stress; and
- Lower productivity and population densities.

Species-specific examples of these impacts range from altered animal behavior to nest abandonment, among others. Several studies have shown increased vigilance and flight response by birds, ungulates, and other wildlife when approached by people engaged in various forms of recreational activities (Taylor and Knight 2003, Canfield et al. 1999, Papouchis et al 2001, González et al. 2006). This subsequent human avoidance is known to negatively alter space use in kit foxes (Jones et al 2017), moose (Harris et al. 2014), elk (Brown et al. 2012, Preisler et al. 2006, Frair et al. 2008), pronghorn (Brown et al. 2012), and waterbirds (McLeod et al. 2013). Preliminary observations from camera traps in the adjacent Livingstone-Oldman region suggest that large carnivores may preferentially select roads and trails with little or no OHV use (Garrow 2008). Similarly, Muhly et al. (2011) found high-human activity on roads and trails displaced predator species, effectively altering predator-prey interactions and creating spatial refuge from predation. In nearby Banff, Kootenay and Yoho National Parks, Rogala et al. (2011) found both wolves and elk avoided trails and also adjusted their proximity to trails to alter natural predator-prey dynamics. Direct vehicle collisions on roads and rail lines are a well-known negative impact of linear footprints on many wildlife species (Maxell and Hokit 1999, Benn and Herrero 2002). Measured negative impacts from altered space use include higher energy costs due to distance moved, avoidance of key habitats, reduced use of high-quality habitat, and higher mortality rates (Boyle and Samson 1985, Wisdom et al. 2005, Snetsinger and White 2009, Stankowich 2008, Neumann et al. 2010, Wiedmann and Bleich 2014).

Reproductive productivity for many species is also known to be negatively impacted by human activity. For example, songbirds nesting close to recreational trails may abandon active nests (Barton and Holmes 2007). Golden eagles nesting near trails used by off-highway vehicles experience decreased reproduction (Steenhof et al. 2014, Spaul and Heath 2016) as did hooded plovers, for which nest losses from human disturbance can exceed 80% (Buick and Paton 1989). And American oystercatchers' egg incubation behaviour has been shown to be directly affected by OHV traffic (McGowan and Simons 2006). Ultimately, these impacts may accumulate and translate into decreased animal densities linked to human presence, as has been suggested for wolverines (Carroll et al. 2001, Fisher et al. 2013), fishers (Carroll et al. 2001), and even wide-ranging species like woodland caribou, recently extirpated from the adjacent Banff National Park (Hebblewhite et al. 2010).

7.0 Ecological response to grazing

As summarized in numerous reviews, responses to cattle grazing are complex, due to variability in grazer movements and behaviour, and ecological variability within and among grazed ecosystems (Kauffman and Krueger 1984, Belsky 1986, Milchunas and Lauenroth 1993, Belsky et al. 1999). Studies have found that grazing may alter soil structure, vegetation stratification, plant community composition, and wildlife populations in upland habitats (Jones 2000, Lyseng 2016), riparian areas (Roath and Krueger 1982, Gillen et al. 1984, Scrimgeour and Kendall 2003), and adjacent streams (Kauffman et al. 1983, 1984, Harding et al. 1998, Belsky et al. 1999, Clary 1999, Freilich et al. 2003). Because the behaviour and seasonal movements of cattle differ from those of bison populations that historically occupied the Castle region, the loss of bison and the introduction of cattle have likely impacted a wide range of species and ecosystem functions (Milchunas and Lauenroth 1993, Fleishner 1994).

Evidence from rangeland health assessments at several reference areas in the Castle River allotment suggests that plant communities in parts of the Castle region have shifted over the past several decades as these communities recover from historically high levels of cattle grazing (Alberta Environment and Parks 2015, Adams et al. 2016). Among the plant community changes observed, agronomic species such as Kentucky bluegrass and timothy has displaced native species in some areas (Alberta Environment and Parks 2015).

8.0 Conclusions and scientific recommendations

This report provides an overview of human activities and land use in the Castle region, along with evidence from the published scientific literature for their potential impacts on species and ecosystems (Figure 32).

Figure 32 Summary diagram of ecological response to human activities in the Castle region.

While the footprints of historical disturbances from forestry and other industrial activities are a significant part of the landscape of the Castle region, linear footprints used primarily for recreation have been a dominant feature of land use in recent years. The average density of mapped linear footprints in the Castle region is 2.0 km/km^2 and ranges from 0.5 to 3.4 km/km^2 among the 9 watersheds in the region (Figure 33). Most of these features are trails that are inaccessible by on-highway vehicles. The average density of roads alone is 0.2 km/km^2 and ranges from 0.0 to 0.5 km/km^2 (Figure 34).

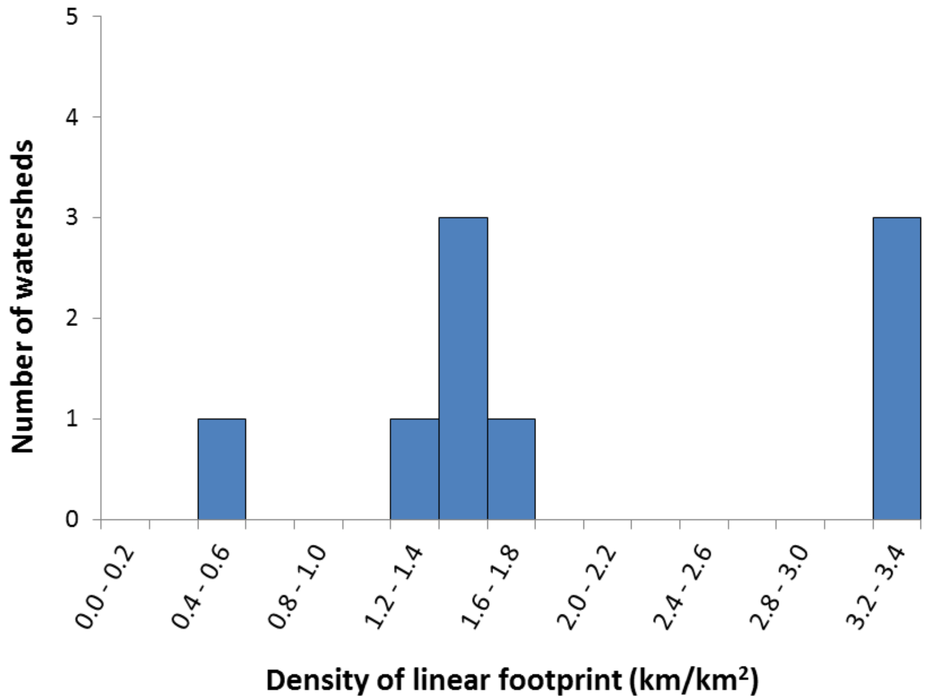

Figure 33 Count of watersheds by linear footprint density class. Data sources: Appendix A.

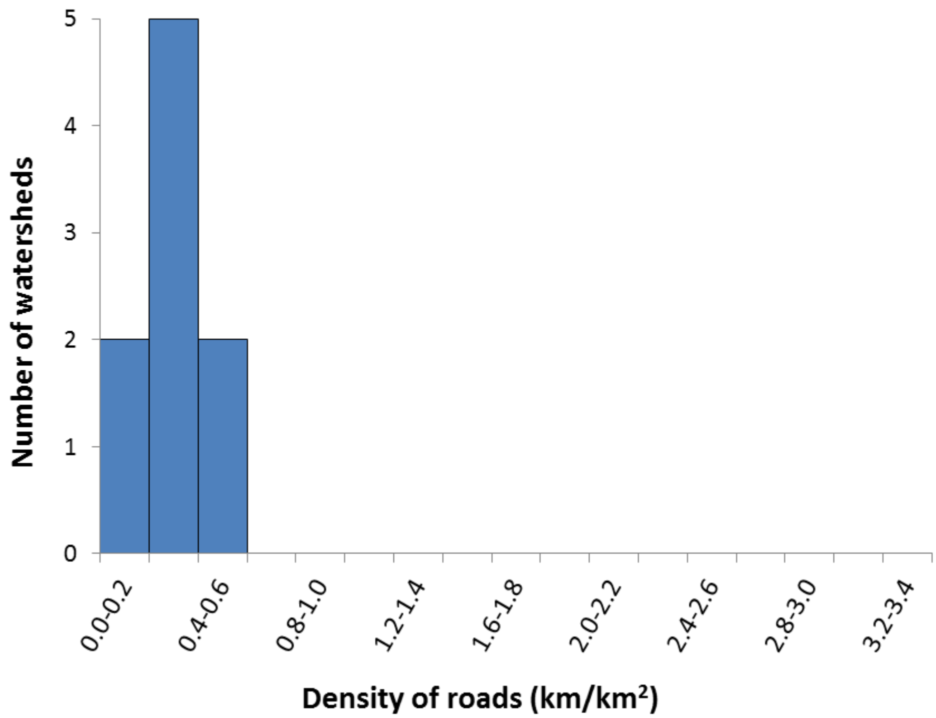

Figure 34 Count of watersheds by road density class in the Castle region. Data sources: Appendix A.

The type and frequency of human activity on linear footprints in the region needs to be better quantified, including the use of trails by recreational OHVs. Recreational use of the Castle region by people may have contributed to increases in the frequency of human-caused fires and the expansion of invasive plants.

Studies conducted elsewhere have documented biological and ecological responses to linear footprints such as those found in the Castle region and their use by people. Use of trails by OHVs can cause increased soil erosion and loss of vegetation. Trails close to streams have been shown to provide a potential pathway for increased sediment input into aquatic ecosystems, including those used by fish for spawning and rearing. Erosion and deposition of dust released by OHVs on surrounding vegetation have been shown to alter the composition of vegetation communities. Finally, the presence of people in remote areas used by grizzly bears increases the likelihood of conflict with people, representing an increased risk of bear mortality. Behavioural responses to human activity along trails have been observed in grizzly bears and numerous other wildlife species.

Several large-scale studies have demonstrated a negative relationship between linear footprints and the biotic endpoints of interest in the Castle region. Because those studies were conducted elsewhere, their relevance to the Castle region is inferential.

However, many of the ecological responses to human activity and land use observed elsewhere are likely to also apply in the Castle region.

In particular, because the type and frequency of motorized use in back-country regions is rarely documented, it is challenging to understand the influence of these factors on observed ecological responses. While many studies report biotic responses to the density of linear footprints accessible to on-highway vehicles (i.e., roads), few studies have focussed on trails that are generally accessible only by off-highway vehicles. Because the recreational use of OHVs is expanding in many regions of North America, this is an important area for future ecological research. Further monitoring and research are required to assess the ecological significance of type and frequency of motorized use of linear footprints on westslope cutthroat trout, bull trout, and grizzly bears in Castle region and elsewhere in Alberta.

The specific threshold of critical impact related to density of linear footprint and frequency of human use varies by species and footprint type. However, in general increased human land use and disturbance, including trails and OHVs, are associated with ecological disruption and impairment. If the goal is to enhance conservation outcomes in the Castle region and create conditions that are better able to sustain healthy ecosystems, then restricting human activities, including OHV use, will likely increase the chances of achieving those objectives. Positive ecological outcomes could include decreased vegetation disturbance, lower rates of invasive species infiltration and expansion, improved condition of headwater streams, increased viability of westslope cutthroat trout and bull trout populations, and reduced risk of human-caused grizzly bear mortalities.

Projected changes in hydro-climatic regimes in the region as the result of climate change will also influence some of these biological and ecological endpoints. Biologically diverse areas of high topographic and environmental diversity such as the Castle region will likely serve as refugia for many species as communities shift in response to projected warming and associated environmental changes.

Further monitoring and research are required to reduce uncertainty and determine the respective impacts of natural and anthropogenic drivers on species and ecosystems in the Castle region, and to better inform management options and actions. EMSD's provincial ambient monitoring programs will aim to fill many of the scientific and monitoring gaps and uncertainties identified in this review.

9.0 Acknowledgments

The Environmental Monitoring and Science Division could not have completed this review without the contributions from other authors, expertise from the Policy and Planning Division, Operations Division, Wildfire Management Branch, and countless personal communications from inside and outside government. The foundation built by the Alberta Parks Division was paramount, and we thank everyone involved for their assistance. The reviews provided by the external scientific experts are also appreciated.

All photos from the Government of Alberta.

10.0 Glossary

Animal unit month (AUM): a measure of grazing intensity equivalent to the average quantity of forage consumed by a 1,000 lb mature cow and her suckling calf in a one month period.

Anthropogenic driver: Human-caused activities that can affect living (biotic) and non-living (abiotic) components of an ecosystem (Nelitz et al. 2015).

Bear management area (BMA): Defined areas of land in Alberta where grizzly bears are managed for conservation. The 'Recovery Zone' is where the government intends to recover the population. The 'Support Zone' is intended to allow for grizzly bears whose home ranges are not fully centered in the Recovery Zone. (Alberta Environment and Parks 2017b).

Biological endpoint: A final point at which an effect of a stressor or disturbance on an individual can be measured. In an ecosystem context, ecologically relevant endpoints can potentially be used to infer the effects of the stressor or disturbance on populations, communities and ecosystems (Maltby 1999).

Cumulative effects: The synergistic, interactive, or unpredictable outcomes of multiple land-use practices. (Ross 1998, Johnson 2013)

Ecological response: How an ecosystem or other ecological feature responds after natural or anthropogenic disturbance.

Ecological risk: The likelihood that the environment or features of an ecosystem will be affected by an event or activity.

Human footprint: The Alberta Biodiversity Monitoring Institute's definition is as follows: "The temporary or permanent transformation of native ecosystems to support residential, recreational or industrial land uses. Under this definition, human footprint includes the geographic extent of areas under human use that either have lost their natural cover for extended periods of time (e.g., cities, roads, agricultural land, and surface mines) or whose natural cover is periodically reset to earlier successional conditions by industrial activities (e.g., cut blocks and seismic lines)." (Alberta Biodiversity Monitoring Institute 2014)

Hydrological unit code (HUCs): A classification system for watersheds or drainage basins. Feature classes for the HUC watersheds of Alberta range from HUC 2 (larger watersheds) to HUC 8 (finest level).

Invasive species: Plants, animals or other organisms are non-native species that are introduced to an area outside of their natural habitat and threaten the native species or ecosystems where they become established (Alberta Environment and Parks 2016)

Linear footprint: Landscape features that are somewhat straight and similar, and act as a corridor since they are different than the surrounding area. Common examples of human-made linear features on a landscape are roads, railways, pipelines and transmission lines.

Mortality risk: An output from a statistical model estimating a species chance of death in a regional spatial area. (Nielsen et al. 2004)

Natural driver: Forces of nature that affects living (biotic) and non-living (abiotic) components of an ecosystem. (Nelitz et al. 2015)

Regulatory limit: In the context of motorized trail use, defined areas where certain road or trail densities are not exceeded.

Strahler stream order: A watercourse classification system based on a hierarchy of tributaries, first proposed by A. Strahler (1952) and R. Horton (1945). (United States Geographical Society 2016)

Stressor: An event or activity in an ecosystem that can have potential impacts on an organism, population or on the ecosystem as a whole. (Nelitz et al. 2015)

Threshold: In ecological terms, the point at which there is a large response to ecosystem quality or components because of changes in natural or anthropogenic drivers. (Groffman et al. 2006)

Tracked species (ACIMS): Species that are placed on tracking lists because currently available occurrence data suggest that they may be rare.

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Appendix A Summary of data sources used in report figures. See Literature Cited for full reference.

Figure	Title	Data sources
1	Map of the Castle region.	Provincial Parks (2017) Wildland Provincial Parks (2017) Roads of Alberta (2016)
2	Stream crossings illustration.	Linear footprint in the Castle region (2017) FWMIS Hydrology Arcs (2017) Alberta SPOT Imagery (2016)
3	Watersheds in the Castle region.	Hydrologic unit code watersheds of Alberta (2017)
4	Watercourses in the Castle region.	Hydrologic unit code watersheds of Alberta (2017) Fish and Wildlife Management Information System (FWMIS) – Hydrology Arcs (2017).
5	Soil subgroups in the Castle region.	Derived Ecosite Phase (2017)
6	Land cover in the Castle region.	ABMI Wall-to-Wall Land Cover 2010 (2012) Castilla et al. (2014)
7	Summary of land cover in the Castle region.	ABMI Wall-to-Wall Land Cover 2010 (2012) Castilla et al. (2014)
8	Distribution of westslope cutthroat trout in the Castle region.	Westslope cutthroat trout occurrence within their native range (2017) Westslope cutthroat trout federally designated critical habitat (2017)
9	Distribution of bull trout in the Castle region.	Bull trout occurrence within their native range (2017) Bull trout spawning reaches (2017)
10	Grizzly bear habitat value in the Castle region	Stenhouse and Morehouse (in prep.)
11	Home range polygons of radio collared grizzly bears in the Castle region, 2005-2009.	Stenhouse and Morehouse (in prep.)
12	Linear footprint in the Castle region.	Linear footprint in the Castle region (2017)
13	Summary of linear footprint lengths by footprint category in the Castle region.	Provincial Parks (2017) Wildland Provincial Parks (2017) Linear footprint in the Castle region (2017)
14	Linear footprint density in the Castle region. Densities were calculated for only the portion of each watershed in the Castle region.	Hydrologic unit code watersheds of Alberta (2017) Linear footprint in the Castle region (2017)
15	Summary of linear footprint density in the Castle region.	Hydrologic unit code watersheds of Alberta (2017) Linear footprint in the Castle region (2017)

Figure	Title	Data sources
16	Slope classes in the Castle region.	Linear footprint in the Castle region (2017) Alberta provincial digital elevation model (2017) Meyer (2002)
17	Summary of linear footprint length by slope class in the Castle region.	Linear footprint in the Castle region (2017) Alberta provincial digital elevation model (2017) Meyer (2002)
18	Stream crossings in the Castle region.	Linear footprint in the Castle region (2017) FWMIS Hydrology Arcs (2017).
19	Stream crossing density in the Castle region.	Linear footprint in the Castle region (2017) FWMIS Hydrology Arcs (2017)
20	Trail-based rutting and water pooling (left) and multiple OHV trails crossing an unabridged stream (right) in the Castle region.	N/A
21	Human footprint in the Castle region.	ABMI Wall-to-Wall Human Footprint Inventory 2014 (2017) Alberta Vegetation Inventory Extended (AVIE; 2017) Post Inventory Cutblocks (2017)
22	Summary of human footprint in the Castle region.	ABMI Wall-to-Wall Human Footprint Inventory 2014 (2017) Alberta Vegetation Inventory Extended (AVIE; 2017) Post Inventory Cutblocks (2017)
23	Reported wildfires in the Castle region, 1961 - 2016.	Historical wildfire database (2017)
24	Historical extent of wildfires > 200 ha in the Castle region 1931 - 2016.	Spatial wildfire data (2017)
25	The number of wildfires in the Castle region, 1961 - 2016.	Historical wildfire database (2017)
26	Area burned by wildfire in the Castle region, 1961 - 2016.	Spatial wildfire data (2017)
27	Grazing allotments in the Castle region.	Grazing allotments - Distribution units (2016)
28	Soil erosion hazard in the Castle region.	Derived Ecosite Phase (2017)
29	Soil rutting and compaction hazard in the Castle region.	Derived Ecosite Phase (2017)

Figure	Title	Data sources
30	Crossings of linear footprints on stream reaches with confirmed westlope cutthroat trout occurrence, and federally designated critical habitat.	Linear footprint in the Castle region (2017) Westslope cutthroat trout occurrence within their native range (2017) Westslope cutthroat trout federally designated critical habitat (2017)
31	Crossings of linear footprints on stream reaches with confirmed bull trout occurrence and spawning areas.	Linear footprint in the Castle region (2017) Bull trout occurrence within their native range (2017) Bull trout spawning reaches (2017)
32	Summary diagram of ecological responses to human activities in the Castle region.	N/A
33	Density of roads and trails in the Castle region.	Hydrologic unit code watersheds of Alberta (2017) Linear footprints in the Castle region (2017)
34	Density of roads in the Castle region.	Hydrologic unit code watersheds of Alberta (2017) Linear footprints in the Castle region (2017).

Appendix B Summary of key literature: ecological response to linear footprint.

Linear footprint Type / Use / Density	**Stressor** Soil removal / vegetation loss / sediment runoff / human -caused mortality / animal disturbance / other	**Biotic endpoint** Soil parameter / vegetation parameter / water quality / population parameter (reproduction / mortality / occurrence / abundance), animal behaviour	**Key finding**	**Location**	**Source**
Trails Motorized	Animal disturbance	Kit fox space use	Negative correlation between winter OHV trail density and kit fox space use.	Sonoran Desert, Arizona, USA	Jones et al. 2017
Multi-use trails Motorized, non-motorized 0.7-7.8 km/km^2	Animal disturbance	Golden eagle territory occupancy and productivity	Negative relationship between occupancy and off-road vehicle use; negative relationship between productivity and off-road vehicle use.	Idaho, USA	Spaul and Heath 2016
Road Motorized 0 – 2.5 km/km^2	Human-caused mortality	Grizzly bear mortality rate	Positive relationship between grizzly bear mortality and road density from 1999 to 2012.	Central Rockies of Alberta	Boulanger and Stenhouse 2014
Winter snowmobile trails Snowmobiles 0 -100% coverage by snowmobile tracks	Animal disturbance	Moose habitat use measured from radiotelemetry	Avoidance by moose of areas of high snowmobile trail density.	Kenai Peninsula, south-central Alaska, USA	Harris et al. 2014
Trails Motorized	Soil removal and sediment runoff	Water quality	All stream crossings evaluated had soil loss; downstream of OHV stream crossing, river substrate has increased mud coating.	Western Arkansas, USA	Marion et al. 2014
Off-road Motorized, non-motorized	Animal disturbance	Waterbird flight response	Flight response to non-motorized use greater than motorized use.	Victoria, Australia	McLeod et al. 2013

Linear footprint	Stressor	Biotic endpoint	Key finding	Location	Source
Human disturbance Motorized, non-motorized Distance to disturbance		Westslope cutthroat/ Rainbow trout Hybridization	Human disturbance may have direct and indirect effects on the spread of non-native species and hybridization in stream fish.	SW Alberta, SE British Columbia	Yau and Taylor 2013
Trails Motorized	Permafrost alteration / soil removal / hydrological changes	Vegetation and organic soil loss	OHVs are altering headwater hydrology (drainage density); increased mean active layer depth.	Wrangell-St. Elias National Park, USA	Arp and Simmons 2012
Paved road Motorized, non-motorized	Animal disturbance	Ungulate behaviour (vigilance, defensive, flee, travel)	Behavioural response was variable to motorized and non-motorized use along paved road.	Grand Teton National Park, Wyoming, USA	Brown et al. 2012
Road Motorized Vehicle use	Animal disturbance	Grizzly bear behaviour (movement)	Negative relationship between habitat use and moderate and high traffic roads.	Southwestern Alberta	Northrup et al. 2012
Human disturbance (Roads/Settlements)	Animal Disturbance	Grizzly bear population fragmentation	Females have lower habitat disturbance thresholds.	Western Canada, Northwest USA and SE Alaska, USA	Proctor et al. 2012
Roads Motorized 0-1.05 km/km^2 (E. Valdal 2006)	Uncertain (sediment runoff / human-caused mortality)	Westslope cutthroat trout abundance	Negative relationship between trout abundance and road density within 100 m of streams.	Southeastern British Columbia	Valdal 2006
Road Motorized	Human-caused mortality	Grizzly bear mortality rate	Positive relationship between grizzly bear mortality and road density.	Greater Yellowstone Ecosystem, USA	Schwartz et al. 2010a
Trails Motorized	Soil removal	Soil parameter	Greater dust throw measured in silt soils.	Nevada, USA	Goossens and Buck 2009

Linear footprint	Stressor	Biotic endpoint	Key finding	Location	Source
Trails Motorized, non-motorized	Soil removal / sediment runoff	Soil parameter / water quality	Mean soil loss on OHV trails was significantly higher than other forms of recreational use; horse trails still had large amount of soil loss whereas hiking and biking trails had minimal.	North Central Tennessee, USA	Olive and Marion 2009
Roads Motorized 0.0 – 2.5 km/km^2	Animal disturbance	Elk habitat use and mortality risk	Negative relationship between habitat use and road density; Positive relationship between mortality risk and road density. Road densities ≤ 0.5 km/km^2 yielded the high probability of elk occurrence.	West-central Alberta	Frair et al. 2008
Trails Motorized	Soil removal	Soil parameter / vegetation parameter	OHVs are a significant source of soil erosion from aeolian displacement.	Western Kentucky, USA	Padgett et al. 2008
Road and trail Motorized Road: 0.6 km/km^2 Trail: 0.2 km/km^2	Sediment runoff	Water quality	OHV trails produced 6 times sediment amount than greater number of road segments; 24% of OHV trails connected to and influencing stream water quality.	South Platte River Watershed, Colorado, USA	Welsh 2008
Trails Motorized	Animal disturbance	Animal behaviour	Increased nest desertion and abandonment rates by songbirds <100m from an OHV trail than those >100m from trail.	Northeast California, USA	Barton and Holmes 2007
Other – pipeline crossing Non-motorized	Sediment runoff	Aquatic invertebrate community composition and fish abundance	Changes in invertebrate community composition and decrease in fish abundance downstream of pipeline creek crossing.	N/A - Review article	Lévesque and Dubé 2007

Linear footprint	Stressor	Biotic endpoint	Key finding	Location	Source
Trails Non-motorized Trail use	Vegetation loss, species richness, non-native plants	Vegetation community	Compared high use and low use hiking trails. Showed significant differences for vegetation cover, exposed soil, species richness etc. Non-native plants only present on high use trail.	Mount Robson Provincial Park, BC	Nepal and Way 2007
Roads/Trails Motorized	OHV effects on soils, vegetation, wildlife and habitats, water quality and air quality		Soil compaction negatively impacts vegetation growth which leads to a decrease in plant diversity. OHV use also increases sedimentation and pollutants in water and air.	Bureau of Land Management areas, USA	Ouren et al. 2007
Roads, trails and off-road/trail Motorized, non-motorized	Animal disturbance	Spanish imperial eagle flight response	No response to passing surface vehicles; flight response to overhead aircraft and passing pedestrians.	Central Spain	González et al. 2006
Off-trail Motorized, non-motorized Vehicle use	Animal disturbance	American oystercatcher incubation behaviour	Negative relationship between incubation duration and all-terrain vehicle traffic.	Coastal North Carolina, USA	McGowan and Simons 2006
Road/trail Motorized, non-motorized OHV use	Animal disturbance	Ungulate behaviour (flight/avoidance)	Elk respond negatively (with flight/avoidance) to OHV use (> 1 km).	Starkey Experimental Forest and Range, Oregon, USA	Preisler et al. 2006
Road Motorized 0 – 1.6 km/km^2	Animal disturbance	Bull trout occurrence	Negative relationship between bull trout occurrence and road density. Compared to roadless areas, bull trout were 50% less likely to be found where road density was greater than 0.4 km/km2, and were predicted to be absent where road density exceeded 1.6 km/km^2.	Kakwa River basin, west-central Alberta	Ripley et al. 2005

Linear footprint	Stressor	Biotic endpoint	Key finding	Location	Source
Road and trail Motorized Distance to disturbance	Human-caused mortality	Grizzly bear mortality rate	Greater mortality near linear footprints.	Southwestern Alberta	Nielsen et al. 2004
Trails Motorized, non-motorized Trail use	Sediment dynamics, soil compaction	Soils	OHVs compact soil and have increase erosion rates compared to hiking and equestrian trails.	Wayne National Forest, Ohio, USA	Sack and da Luz 2003
Roads Motorized 0.0 – 2.5 km/km^2	Animal disturbance	Carnivore occurrence (fisher, lynx, wolverine, grizzly bear)	Varying response to road density. Higher road density at sites with fisher (1.3 km/km^2); lower road density at sites with wolverine (0.7 km/km^2).	Rocky Mountains, northern USA and southern Canada	Carroll et al. 2001
Roads All		Terrestrial and aquatic communities	Review the ecological effects of roads on terrestrial and aquatic communities.	N/A - Review synthesis	Trombulak and Frissell 2000
Road Motorized 0 – 1.2 km/km^2	Erosion and sedimentation	Bull trout reproduction	Negative relationship between bull trout redd abundance and road density.	Swan Basin Montana, USA	Baxter et al. 1999
Roads Motorized	Erosion and sedimentation	Bull trout occurrence	Negative relationship between bull trout occurrence and density of roads within stream basins.	Boise River Basin, Idaho, USA	Dunham and Rieman 1999
Trails Motorized, non-motorized Trail use	Species richness, non-native plants	Vegetation community	OHV use results in more areas being accessed, and spread of non-native plants 4x the amount when compared with non-motorized users.	Montana and northern Idaho, USA	Adams 1998
Roads Motorized	Animal disturbance	Bull trout occurrence	Negative relationship between bull trout occurrence and road density.	Columbia River and Klamath River Basins: Idaho, Montana, Nevada, Oregon, Washington and Wyoming, USA	Rieman et al. 1997

Linear footprint	Stressor	Biotic endpoint	Key finding	Location	Source
Road Motorized 0 – 6.3 km/km^2	Human-caused mortality	Grizzly bear habitat use	When traffic, road densities and human access increases, bears avoid these areas and survival rates decline.	Swan Mountain Range, Montana, USA	Mace et al. 1996
Road Motorized	Animal disturbance	Salmonid spawning and rearing success	Negative relationship between the proportion of a watershed supporting strong salmonid populations (spawning and rearing) and road density.	Columbia River and Klamath River Basins: Idaho, Montana, Nevada, Oregon, Washington and Wyoming, USA	Quigley and Arbelbide 1997
Off-trail Motorized	Human-caused mortality (nests)	Hooded plover nesting success	Loss of 81% of nests (average 6% per day).	South Australia	Buick and Paton 1989
Road Motorized	Animal disturbance	Grizzly bear habitat use	Most of the 23 grizzly bears used areas within 250 m of open roads significantly less than expected; equivalent to 8% loss of available habitat.	Flathead Valley, southeastern BC, northern Montana, USA	McLellan and Shackleton 1988
Road Motorized	Sediment runoff	Water quality	Road construction increased sediment load in drainage basin by 7 times.	Missoula, Montana, USA	Anderson and Potts 1987

Appendix C Summary of literature reviews: ecological response to linear footprint.

Source	Title	Studies Reviewed	Key Findings
Switalski 2016	Snowmobile best management practices for Forest Service travel planning: A comprehensive literature review and recommendations for management. Four articles: 1. Introduction 2. winter recreation use conflict 3. Wildlife and 4. water quality, soils, vegetation	Series of articles – 90 references	Alpine environments are particularly sensitive to disturbance, snowmobiles can pollute waterways, cause soil erosion, damage vegetation. Snowmobiles can impact sensitive and hunted wildlife species, from energy expenditures, denning disruption, to physiological and behavioural responses.
Marzano and Dandy 2012	Recreational use of forest and disturbance of wildlife. A literature review.	450 references	Review the impact of recreational activities on the flora, fauna and habitat in UK forests. Non-motorized trail use such as hiking, biking, and horse riding have been shown to cause less impact on soil and vegetation than motorized uses by off-highway vehicles.
Switalski and Jones 2012	Off-road vehicle best management practices for forestlands: A review of scientific literature and guidance for managers.	70 references	Document how compaction from OHV traffic increases surface flow, soil erosion and sedimentation. Loss of vegetation following OHV use, leaves plants that do survive along trails weakened, malformed, more susceptible to disease/insect predation. Vegetation trampling by OHVs can damage germinating seeds, and OHVs are a major vector for non-native invasive plant species.
Backcountry Hunters and Anglers 2011	Cumulative and universal: ATV impacts on the landscape and wildlife	92 references	Impacts of OHV use are cumulative, universal and can be achieved by low intensity traffic over short time periods. OHV use affects soil and hydrologic function. OHV travel can disproportionately alter animal behaviour relative to traditional forms of recreation due to the distances motorized vehicles can travel in a day.
Daigle 2010	A summary of the environmental impacts of roads, management responses, and research gaps: A literature review.	160 references	Overview of potential environmental impacts of resource roads including effects on terrestrial and aquatic wildlife, plant communities, and physical elements found across landscapes in British Columbia. Effects may be local or may apply to large areas. Road effects can occur during construction or with subsequent road presence, upkeep, and use.

Source	Title	Studies Reviewed	Key Findings
Stankowich 2008	Ungulate flight responses to human disturbance: A review and meta-analysis.	59 studies used for meta-analysis	Evidence shows ungulates pay attention to approacher behavior, have greater perceptions of risk when disturbed in open habitats. Females or groups with young offspring show greater flight responses than adult groups. Populations in areas with higher human traffic showed reduced wariness. Hunted populations showed significantly greater flight responses than non-hunted populations.
Ouren et al. 2007	Environmental effects of off-highway vehicles on Bureau of Land Management lands: A literature synthesis, annotated bibliographies, extensive bibliographies, and internet resources.	700 references	Summary of OHV effects: Soils and watersheds (loss of soil structure, soil compaction, runoff); Vegetation (size and abundance of native plants reduced, dust effects, photosynthetic processes); Wildlife and habitats (both noise and presence of OHVs effectively reduced habitat connectivity, changed animal movements, altered population, recolonization dynamics); Water quality (increased sedimentation, turbidity, pollutants); Air quality (fugitive dust, by product of combustion).
Gaines et al. 2003	Assessing the cumulative effects of linear recreation routes on wildlife habitats on the Okanogan and Wenatchee National Forests.	238 references	Common impacts on wildlife species include altered habitat use from human caused displacement and avoidance and disturbance at a specific site during a critical life history stage. They found negative impacts arose from both motorized and non-motorized activities, the severity of which was contingent on the wildlife species.
Stokowski and LaPointe 2000	Environmental and social effects of ATVs and ORVs: an annotated bibliography and research assessment.	59 references	Concluded that soil compaction caused by OHVs, and shear forces of wheel acceleration create channeling that alters water flow, which intensifies soil erosion and compaction. In turn, this compaction exacerbates runoff and reduces water infiltration, causing a reduction of soil moisture and organic carbon content, which both prevent surface revegetation.

Source	Title	Studies Reviewed	Key Findings
Trombulak and Frissell 2000	Review of ecological effects of roads on terrestrial and aquatic communities.	179 references	Concluded that linear footprint impacts on terrestrial ecosystems are generally negative, with impacts that include an increase in mortality rates, vehicle collisions, human-wildlife conflict, hunting and fishing pressure, as well as alterations of animal behaviours and chemical and physical environments.
Canfield et al. 1999	Ungulates. Effects of recreation on Rocky Mountain wildlife: A review for Montana	205 references	Suggest recreational activities have the potential to displace ungulates to private land and have negative direct and indirect effects on the populations. Big game hunting has more immediate effects on population densities and structures than any other recreational activity.
Forman and Alexander 1998	Roads and their major ecological effects.	139 references	Concluded that increased runoff associated with roads results in increased rate and extent of soil erosion, a reduction in soil percolation, aquifer recharge rates, and alteration of stream-channel morphology. Report road densities of approximately 0.6 km/km^2 appear to be the maximum for a naturally functioning landscape containing sustained populations of large predators (wolf, cougar).
Reid 1993	Research and Cumulative Watershed Effects	800+ references	Cumulative watershed effects (CWEs) include changes that involve watershed processes and influenced by multiple land-use activities. Land-use activities can directly affect vegetation, soil properties, topography, and can import or remove water, chemicals, pathogens, and fauna. Land-use activities reviewed include: roads, dams, forestry, grazing, mining, agriculture, urbanization, recreation and fishing.
Boyle and Samson 1985	Effects of nonconsumptive recreation on wildlife: A review	166 references	A rapid increase in recreation is increasing impacts on wildlife and wildlife habitat. Recreationists can affect wildlife through habitat alteration, disturbance, or direct mortality. Mechanized forms of recreation present the greatest impacts. Important to recognize that individuals, populations and species vary in their sensitivity to disturbance.