FORESTRY, FIRE AND FUR: FACTORS DRIVING THE DECLINE OF FISHERS (*PEKANIA PENNANTI*) IN CENTRAL INTERIOR BRITISH COLUMBIA

by
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ABSTRACT

Understanding the factors driving the population dynamics of endangered species is critical to effective conservation. Demographic information on how environmental and anthropogenic influences affect a species distribution and abundance provides valuable information to wildlife managers looking to accurately assign conservation status, deploy limited resources efficiently, and set sustainable harvest levels. Unfortunately, obtaining detailed population data for rare and cryptic species that naturally occur at low densities is challenging, and in many cases not enough is known to develop appropriate conservation measures. One such species is the fisher (*Pekania pennanti*), a mid-sized mustelid considered a species-at-risk over much of its North American range.

Using a combination of DNA-based mark-recapture surveys and population viability analysis (PVA), I evaluated the factors driving the declines being seen in the geographically isolated and endangered Columbian population of fishers (*Pekania pennanti*) in central interior British Columbia (BC), Canada. I estimated the density and abundance of fishers in two spatially distinct ecosystems in central interior BC where current data were lacking, and assessed which ecological and anthropogenic factors best explained the distribution of fishers in each ecosystem. Additionally, I constructed population viability models to assess the specific impacts that trapping mortality would have on the persistence of the Columbian population of fishers under different trapping scenarios. I found the density of fishers in my two study areas varied substantially, primarily due to the level of trapping mortality occurring during each survey season. The density of fishers in the Chilcotin study area west of Williams Lake was among the highest ever reported for the province (~21 fishers/1000 km^2), whereas density in the Enterprise study area south and east of Williams Lake was among the lowest (~9 fishers/1000 km^2). Habitat covariates that best explained the density of fishers were similar between both study areas; forested stands with wet soil moisture regimes composed of older deciduous and spruce trees were most strongly related to higher densities in both the Chilcotin and Enterprise study areas. My population modelling suggests that if current mortality rates continue, including deaths from trapping, the Columbian population of fishers will likely be extirpated from central interior BC within two decades. In the absence of additional fur harvest mortality, the Columbian population appeared unlikely to persist beyond 37 years without additional measures being taken to increase female survival and reproductive output.

This study provides valuable information to wildlife managers looking to allocate limited resources to aid in the recovery of an isolated and endangered species in central interior BC. Conservation priorities for the Columbian population of fishers should focus on eliminating mortality from trapping, and protecting the high-value habitats the species requires to survive and reproduce.

Keywords: Fisher, *Pekania pennanti*, density, spatially explicit capture-recapture, trapping mortality, population modelling, British Columbia

Photo credit: Helen Davis and Rich Weir

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CHAPTER 1

INTRODUCTION

The Ecological Importance of Mesocarnivores

Global biodiversity is in decline, primarily due to habitat loss and human-caused mortality (Maxwell et al. 2016). Where these threats to wildlife populations co-occur, they can produce strong complementary effects that are larger than the sum of their parts (Brook et al. 2008). However, these threats do not affect all species equally, and rare species with restricted geographical ranges (Pimm 1991, Gaston 1994), low fecundity (MacArthur and Wilson 1967), poor dispersal abilities (de Vries et al. 1996), and large home range sizes (Woodroffe and Ginsberg 1998) often are the most susceptible to environmental and anthropogenic disturbances. Many mammalian carnivores (principally Order Carnivora) possess some or all of these characteristics and as such are particularly vulnerable to habitat loss and human-caused mortality; unsurprisingly, this means species within the Carnivora continue to be among the most threatened across the globe (Wolf and Ripple 2017).

The ecological importance of large carnivores is well-established. Even a few individuals can strongly influence community structure directly by limiting prey populations, causing cascading effects on vegetation, and providing competition between larger and smaller predators (Ripple and Beschta 2004, Ripple et al. 2014, Sivy et al. 2017). Additionally, carnivores are ideal indicators of ecosystem health because their large home ranges and low population densities make them sensitive to environmental change (Buskirk and Zielinski 2003). As such, conservation planning often focuses on carnivores with the largest home range requirements based on the premise that protecting the habitat necessary to support viable populations also likely protects the habitats of other species with smaller area needs (Noss et al. 1996). To this end, it is usually the large carnivores such as grizzly bears (*Ursus arctos*) and wolves (*Canis lupus*) that are used as ‘umbrella species’ due to their extensive spatial needs as well as their charismatic appeal (Buskirk et al. 2003).

However, scientists increasingly are understanding that smaller carnivores can also play this role due to their similarly large home range requirements (e.g., 405 - 1366 km^2 for wolverines [*Gulo gulo*] - Lofroth 2001, 50 – 500 km^2 for badgers [*Taxidea taxus*] - Weir and

Hoodicoff 2002), and because in many regions they have replaced larger carnivores as the apex predator (Roemer et al. 2009). Small- and mid-sized carnivores (typically <20 kg), collectively referred to as "mesocarnivores", represent an ecologically diverse array of species whose influence on ecosystem structure and function may have previously been underestimated (Roemer et al. 2009). With the decline of large carnivores, many mesocarnivores now play the role of top predator (Laliberte and Ripple 2004), or act as important drivers of ecosystems due to their impact on prey populations or the surrounding environment (Eldridge and Whitford 2009). For example, where wolves have disappeared from certain areas of California, coyotes (*Canis latrans*) have become the top predator, and by consuming smaller predators that typically predate on eggs from songbird nests they have a positive effect on songbird survival rates (Crooks and Soule 1999). Badgers have a substantial influence on rodent populations in grassland ecosystems and the burrows they dig affect water infiltration, soil aeration, organic decomposition, and plant diversity (Eldridge 2004). Ocelots (*Leopardus palus*) in Panama are the main predators of Central American agoutis (*Dasyprocta punctata*; Moreno et al. 2006), and because agoutis are primary consumers of large-bodied seeds, ocelot predation indirectly affects the way forests are structured (Roemer et al. 2009). Taken together, the wide range of roles that mesocarnivores fulfill in a variety of habitats suggest these species may have exceptionally important effects on ecosystem function, structure, or dynamics (Roemer et al. 2009).

Given the importance of mesocarnivores to ecosystems, the best approach to large-scale conservation may be to target multiple focal species, including those species that specialize on rare habitats that have high ecological value (Carroll et al. 2001). Moreover, recent research suggests that the presence of an intact community of specialist species can be used as a reliable indicator of ecosystem health and functionality and the absence of these species suggests an ecosystem out of balance (Clavel et al. 2011). Thus, a high degree of habitat specialization coupled with the considerable spatial requirements of some mesocarnivores may make them more suitable and compelling species on which to focus conservation efforts (Buskirk and Zielinski 2003).

Humans and Furbearing Mesocarnivores

Despite their importance to ecosystems, humans have been responsible for the direct and indirect mortality of carnivores for millennia. In North America, Indigenous peoples hunted and trapped carnivores for food, clothing, cultural reasons, and to protect their families from predators (Proulx 1999). Furbearing species, the majority of which would be considered mesocarnivores, were harvested at low levels by Indigenous people who used fur for clothing, bedding, and shelter (Wright 1987). Prior to European colonization, Indigenous people used primitive trapping techniques such as deadfalls and snares, and population effects were negligible as focus was placed on acquiring the meat and hides of larger animals (Banci and Proulx 1999). Colonists brought with them new trapping technology in the form of steel traps and economic incentives, and the large-scale harvesting of furbearers soon became a keystone enterprise (Gerstell 1985).

The evolution of the fur trade into a commercial entity changed how humans used furbearers as a resource and the harvest and trade of furs resulted in significant declines in many species (Banci and Proulx 1999). In these early Colonial days, the natural resources of North America seemed inexhaustible and unregulated fur harvest quickly resulted in the substantial reduction, extirpation, or extinction of many once-common furbearers like the American beaver (*Castor canadensis*), the sea mink (*Neovison macrodon*), and the sea otter (*Enhydra lutis*)(Ray 1987). In the case of the sea otter, extirpation from the North Pacific resulted in a cascade of effects including excessive growth of their primary prey, sea urchin, and the subsequent decimation of kelp forests upon which a multitude of other species depend (Estes et al. 1998). While fur harvest does not appear to have had a long-term effect on ermine (*Mustela erminea*) populations, American marten (*Martes americana*), Pacific marten (*Martes caurina*), wolverines, and North American river otters (*Lontra canadensis*) had all been extirpated from portions of their former ranges by the early twentieth century (Harrington et al. 2017). This excessive use of resources spawned the first regulations restricting the harvest of various wildlife in North America although more systematic and widespread conservation efforts took much longer to develop (Organ et al. 2012).

By the mid-1900s, the conversion of agricultural lands back to forest, trapping closures, stricter fur harvest regulations, and translocation efforts allowed many furbearing mesocarnivores to recolonize portions of their historical ranges (White et al. 2021). Although increased habitat protections and reduced human-caused mortality has been successful in allowing harvests of several species to once again become sustainable (White et al. 2021), most North American mesocarnivores continue to be killed both directly and indirectly by humans to this day. Forest-dependent mesocarnivores such as the American marten, for example, continue to be impacted by the negative effects of timber harvest removing large areas of late-successional forests, and in most jurisdictions still are trapped for their furs (White et al. 2015). While some species appear to be resilient to high levels of fur harvest (e.g., coyotes; Malhotra 2022), most furbearers fall somewhere between low and intermediate resiliency levels with substantial variation depending on habitat conditions at the regional level (Banci and Proulx 1999). Unfortunately, trapping systems currently approved for use worldwide still are not truly species-specific and it is nearly impossible to ensure that only those species with high resiliency to trapping are harvested (Virgos et al. 2016, Proulx 2022).

Fishers as a North American Furbearing Mesocarnivore

Fishers (*Pekania pennanti*; Figure 1.1) are one example of a low-density and wide-ranging mesocarnivore species that has experienced significant range contractions over the past two centuries due to human-related activities (Powell and Zielinski 1994, Lofroth et al. 2010). Fishers are housecat-sized members of the Mustelid family once widespread and abundant across North America (Gibilisco 1994); following the settlement of Europeans their distribution and abundance declined primarily due to overexploitation in the fur trade and habitat loss (Lofroth et al. 2010). This large-scale habitat loss, along with extremely high pelt prices and a lack of fur harvest regulations in the early 1900s led to the extirpation of fishers from many regions across the southern extent of their range (Lewis et al. 2012). Increased protective measures, trapping restrictions, reintroduction efforts, and habitat recovery programs have allowed fishers to recolonize much of their historic range in eastern North America; however, west of the Rocky Mountains they remain uncommon, existing as fragmented remnant populations in California, Oregon, Idaho, Montana, and British Columbia (BC)(Lofroth et al. 2010).

Throughout North America, fishers inhabit low elevation boreal and temperate coniferous and mixed-wood forests (Lofroth et al. 2010). They are considered a forest-obligate species and typically are associated with increasing amounts of tree canopy cover (Powell 1994, Weir 2003). Like many other forest-dwelling species, fishers use cavities or chambers in live and dead trees as daily refugia for resting, and for reproduction (Zielinski et al. 2004, Lofroth et al. 2010). Resting and denning structures typically are found in the largest diameter standing live trees, snags, or logs (Raley et al. 2012), but other structural features such as platforms, branches or

Figure 1.1. Fishers (*Pekania pennanti*) are an elusive member of the Mustelid family (Photo credit by author).

mistletoe brooms also are important for resting (Davis 2009, Aubry et al. 2013, Green 2017). Although these structures typically occur in higher densities in older forests, fishers are not truly an old-growth dependent species but rather require a mosaic of different stand types within their home range (Lofroth et al. 2010, Raley et al. 2012, Sauder and Rachlow 2015).

Specific habitat requirements for denning and resting make fishers in BC forest obligates, relying on these distinct structural components of forested ecosystems to fulfill their life requisites (Lofroth et al. 2010). Female fishers in BC give birth and den in the cavities of large diameter trees where some form of damage (e.g., frost crack, branch scar, fire-scar) has allowed decay organisms to enter and form internal cavities over time (Weir et al. 2012). When not actively travelling or hunting fishers use specific habitat features to conserve energy, avoid predators, thermoregulate, and to consume prey safely (Raley et al. 2012). These resting structures also are rare and include heart rot cavities, large branches, coarse woody debris, brooms in white spruce (*Picea glauca*) formed by spruce broom rust (*Chrysomyxa arctostaphyli*), among others (Weir 2003, Davis 2009).

The disease and decay processes that create these essential habitat features for fishers take a long time to develop (>100 years) making the animal highly vulnerable to disturbances that remove these structures faster than forests can redevelop them (Weir et al. 2012). In BC, the permanent loss of forested habitats due to land conversion and hydroelectric developments is a threat in some areas of the province (Weir 2003), but forest harvest has the greatest potential to impact fisher habitat negatively in BC due to the prevalence of clear-cut harvesting practices (Weir and Corbould 2010, BC CDC 2020). Over the past two decades, the low-elevation forests that fishers rely on in the province have been significantly impacted by large-scale insect infestations, unprecedented wildfire seasons, and associated salvage logging. Concern over the loss of fisher habitat in our province is nothing new (Weir 2003, Lofroth et al. 2010), yet the current rate and magnitude of forest harvest following these large-scale disturbances has the potential to be highly detrimental to this already low-density species.

Recent genetic work in BC indicates there are two distinct fisher populations within the province, the Boreal and Columbian population units (BC CDC 2020). During its most recent status review of fishers in March of 2020, the BC Conservation Data Centre (BC CDC) assessed these two fisher populations separately and the Columbian population was placed on the province's 'Red-list' as Endangered (BC CDC 2020). This up-listing was due to a low population estimate of 299 – 517 adult individuals, genetic and geographic isolation, and short-term population trends showing declines of 30 – 50% due to loss of forest habitat (BC CDC 2020). Current conditions also suggest this population is declining further as a result of increasing levels of forest harvest causing further habitat loss and degradation, and trapping where the habitat of fishers has already been compromised (BC CDC 2020). In August 2021, the commercial trapping season for fishers was closed within the range of the Columbian population one year after the animals were reclassified as Endangered (Province of British Columbia 2021). However, fishers are regularly taken as bycatch in traps designed and set for other furbearing species (e.g., traps designed for American marten account for 52% of annual fisher bycatch; Province of British Columbia unpubl. data: Compulsory Reporting 2013–2018 [accessed 15 July 2021]) therefore the closure of the trapping season is likely of limited benefit for reducing the number of fishers harvested annually.

Although recent trends indicate that the Columbian population is declining, fishers historically have been poorly monitored across the province and the true extent of any decline remains unknown (R. Weir pers. comm.). As a result, there still are substantial gaps in knowledge about the current distribution and abundance of fishers in several regions of the province, including areas in the central interior where large-scale habitat losses already have occurred (BC CDC 2020). In addition, in areas where the habitat of fishers has already been significantly impacted little is known about the influence that trapping mortality may be having on this small and isolated population.

Objectives and Thesis Organization

The overarching goal of my research was to investigate the key environmental, demographic, and anthropogenic factors driving the dynamics of the Columbian population of fishers. In conducting this research, I sought much-needed information on the current distribution and abundance of the species in central BC while investigating the effect that continuing trapping mortality will have on its sustainability over the longer term. In Chapter 2, I used information from non-invasive DNA-based surveys in two spatially distinct ecosystems within the range of the Columbian population of fishers; this provided estimates of density and abundance where current data do not exist. Additionally, I investigated which anthropogenic and environmental factors best predicted density across these regions, thereby gaining valuable insight into current habitat associations specific to these ecosystems. In Chapter 3, I used survival and reproductive rates from fishers in central BC and Population Viability Analysis (PVA) methodologies to evaluate the effects that continuing mortality from trapping bycatch may be having on the sustainability of the Columbian population of fishers. Finally, in Chapter 4, I briefly summarize my main findings and detail the management implications of this research for the Columbian population of fishers. In conclusion, I provide recommendations for recovery planning efforts and identify areas future research should focus on with respect to fishers in the central interior of BC.

In the remainder of this chapter, I provide a broad description of the range of the Columbian population of fishers in BC, and a more detailed description of the two study areas where DNA-based surveys were conducted.

Study Area

In the central interior of BC, the Columbian population of fishers range throughout forested habitats at low to moderate elevations extending north to the Rocky Mountains and south to Lytton and Lillooet (Figure 1.2; Weir 2003). In this province, ecosystems are classified into ‘biogeoclimatic’, or ‘BEC’, zones based on climate, elevation, and vegetation characteristics. Northern and central portions of the range of the Columbian population consist of the Sub-Boreal Spruce BEC zone (Meidinger et al. 1991), whereas the southerly portion of the range consists of drier ecological zones including the Sub-Boreal Pine-Spruce (Steen & Demarchi 1997), Montane Spruce (Hope et al. 1991a), and Interior Douglas-fir (Hope et al. 1991b) BEC zones.

For my research, I conducted DNA-based hair-snagging surveys in two study areas within the range of the Columbian population of fishers; one on the Chilcotin Plateau west of Williams Lake during the winter of 2018-19 (hereafter named the Chilcotin study area; Figure 1.3), and one on the interior Fraser Plateau south and east of Williams Lake during the winter of 2020-21 (hereafter named the Enterprise study area; Figure 1.4). The 2,440 km^2 Chilcotin study area extended from Chantslar Lake in the west to Alexis Lakes in the east, and Highway 20 in the south to Satah Mountain in the north. The 2,580 km^2 Enterprise study area extended from the grasslands above the Fraser River in the west to Highway 97 in the east, and Williams Lake in the north to Meadow Lake Road in the south. Both study areas consisted of level to gently rolling

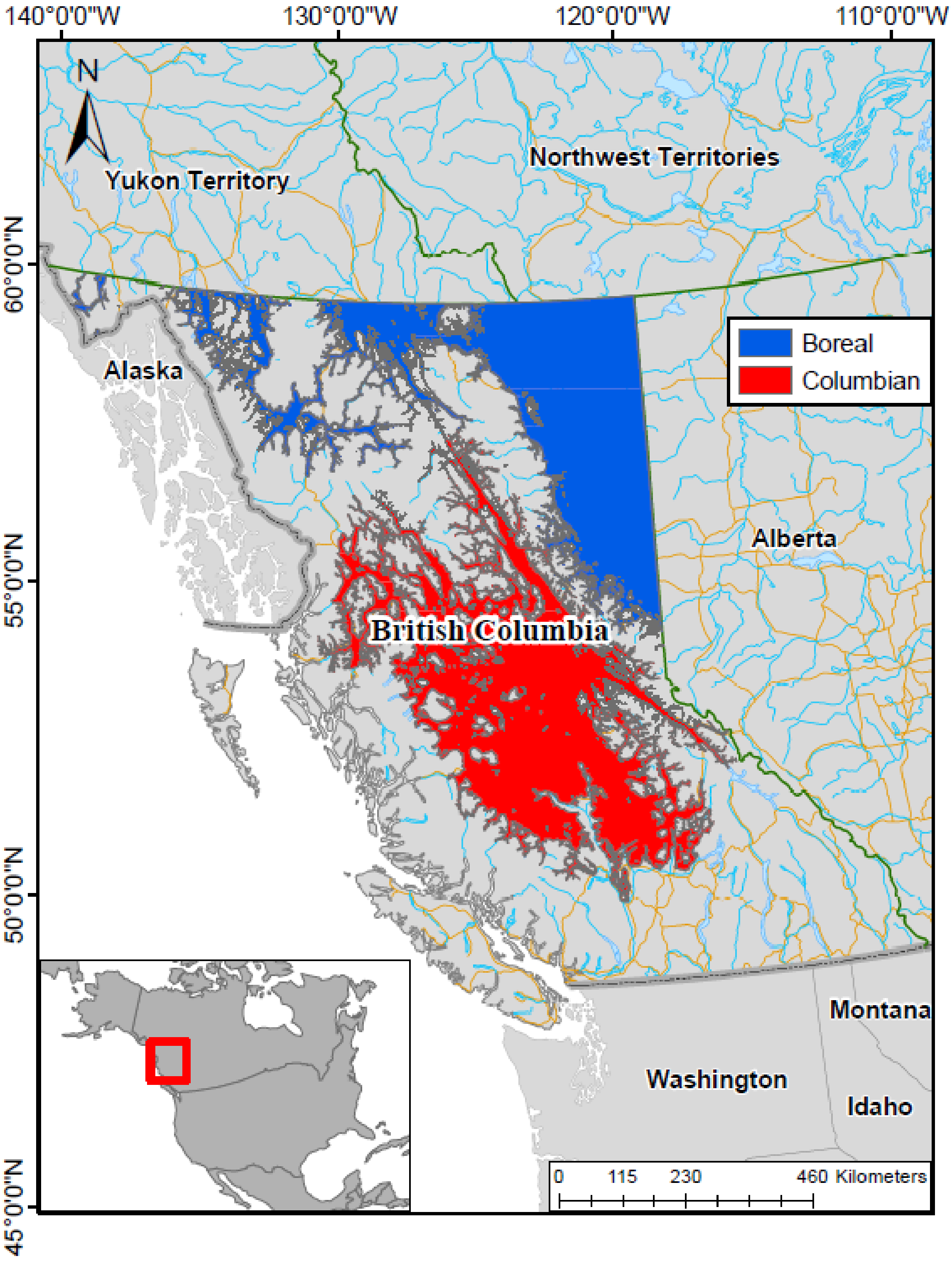

Figure 1.2. Current range of the two fisher (*Pekania pennanti*) populations (Boreal and Columbian, respectively) in British Columbia, Canada (BC CDC 2020).

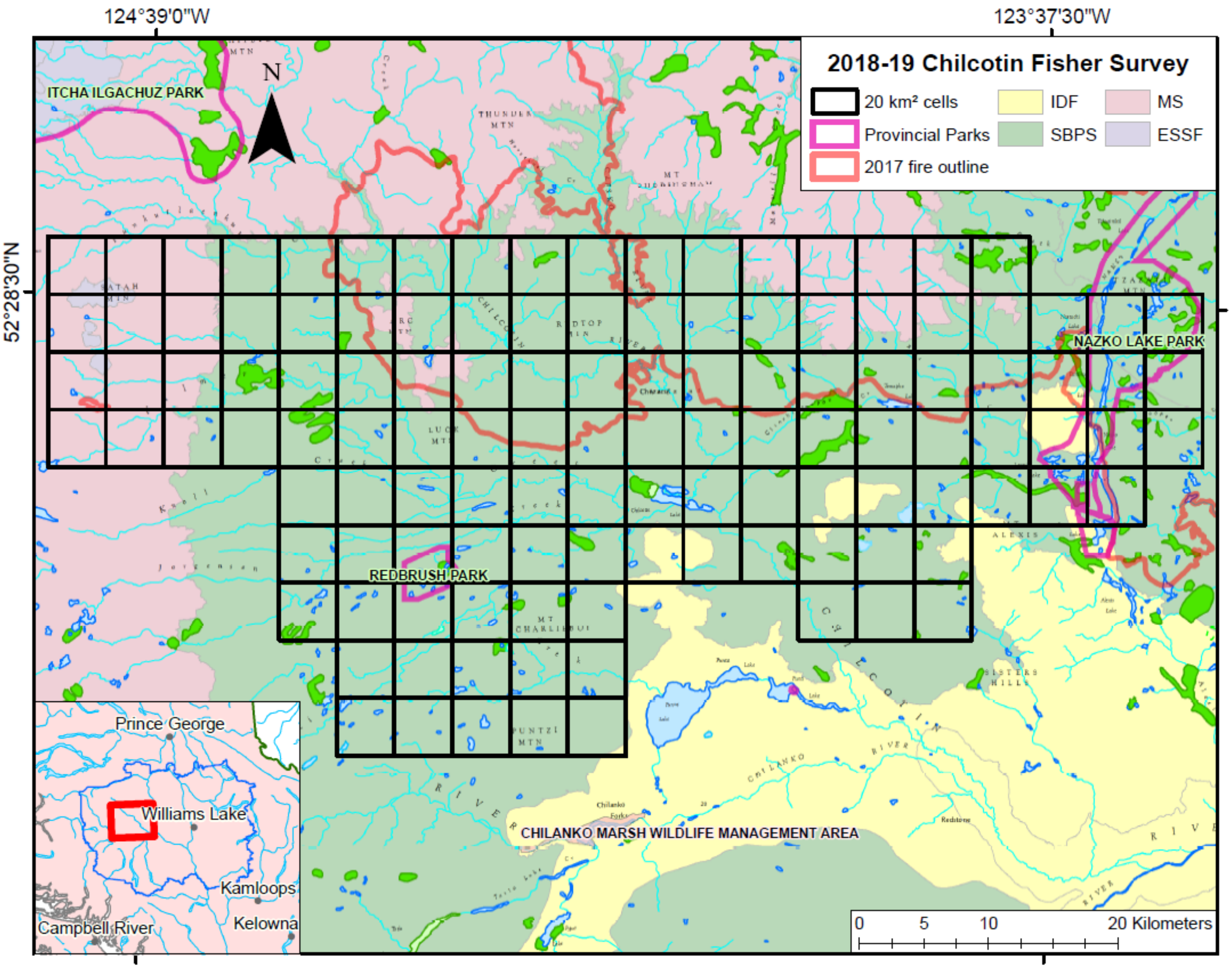

Figure 1.3. Map of the survey area for the 2018-19 Chilcotin Fisher Survey. IDF = Interior Douglas-fir BEC zone; MS = Montane Spruce BEC zone; SBPS = Sub-Boreal Pine-Spruce BEC zone; ESSF = Engelmann-Spruce Subalpine fir BEC Zone.

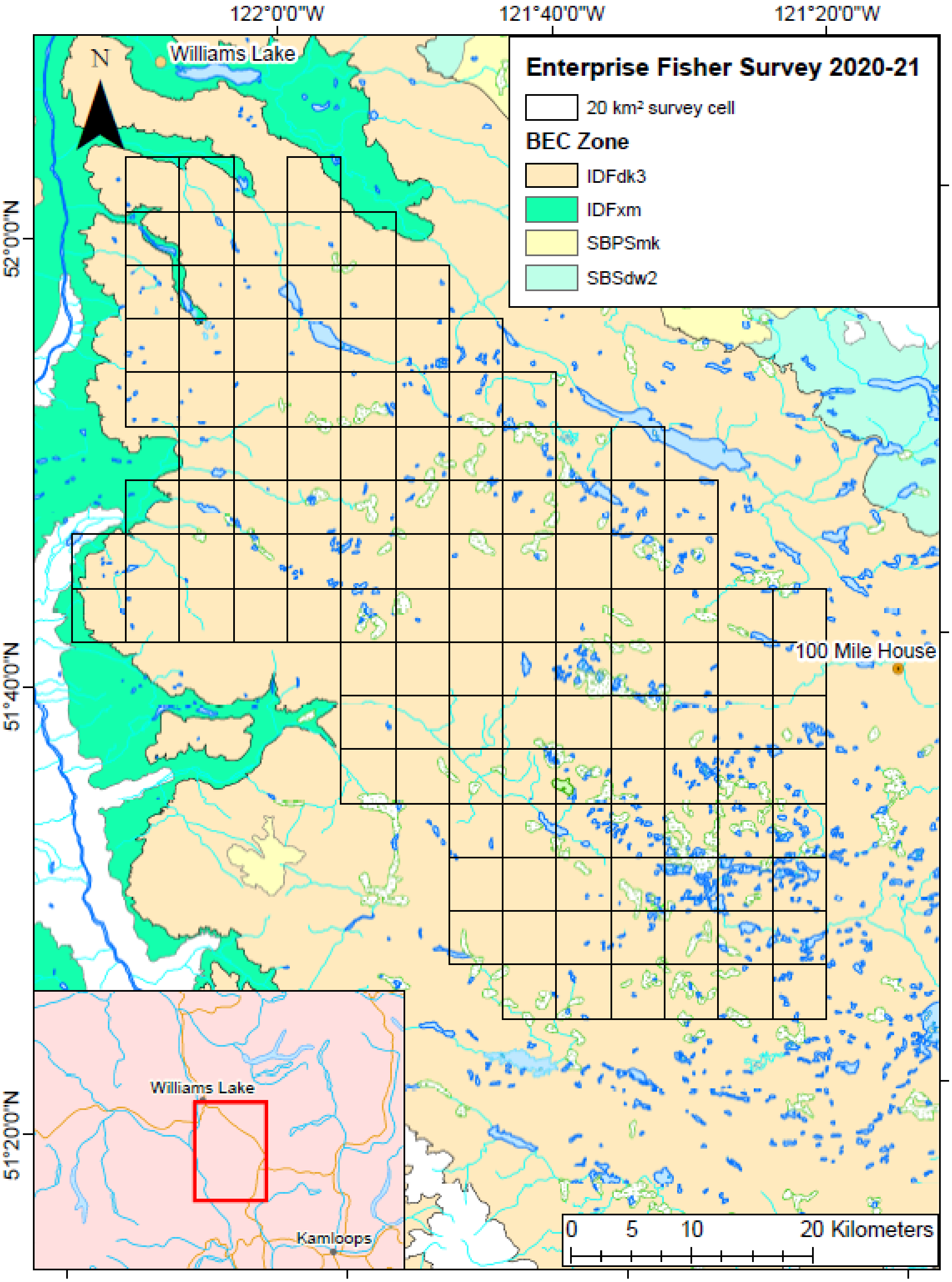

Figure 1.4. Map of the survey area for the 2020-21 Enterprise Fisher Survey. IDFdk3 = Interior Douglas-fir dry cool 3 BEC subzone variant; IDFxm = Interior Douglas-fir very dry mild BEC subzone variant; SBPSmk = Sub-Boreal Pine-Spruce moist cool BEC subzone variant; SBSdw2 = Sub-Boreal Spruce dry warm 2 BEC subzone variant.

topography and surveys were conducted in non-mountainous areas at low to moderate elevations (750 – 1700 m).

The Chilcotin study area was within the driest ecological zone where fishers are found in BC having forests dominated by lodgepole pine (*Pinus contorta*) with smaller components of Douglas-fir (*Pseudotsuga* menziessi), trembling aspen (*Populus tremuloides*), and hybrid spruce (*Picea glauca* x *engelmannii*) concentrated around streams and wetlands (Figures 1.5 and 1.6- Sub-Boreal Pine-Spruce very dry cold [SBPSxc], Montane Spruce very dry very cold [MSxv], and Interior Douglas-fir dry cool 4 [IDFdk4] BEC zones, respectively; Meidinger and Pojar 1991). The Enterprise study area consisted of forests dominated by either multi-aged Douglas-fir or uniform-aged lodgepole pine stands, with small trembling aspen and hybrid spruce stands occurring locally (Figures 1.7 and 1.8 - Interior Douglas-fir dry cool 3 [IDFdk3] BEC zone; Steen and Coupé 1997).

The Chilcotin study area was located on the leeward side of the Coast Range Mountains where the rain shadow effect of the mountains is most pronounced with relatively little precipitation falling (mean annual precipitation 355 – 389 mm) resulting in severely limited vegetation production and soil development in this area (Meidinger and Pojar 1991). The climate in the Chilcotin study area consisted of warm, dry summers (mean warmest month = 12.3 °C; Steen and Coupé 1997) and cold, snowy winters (mean coldest month = -11.8 °C; Steen and Coupé 1997). Mean annual precipitation (433 mm) and mean annual temperature (3.3 °C) were both slightly higher in the Enterprise study area, and the annual average snowfall ranged from 179 cm in the SBPSxc to 231 cm in the IDFdk3 (Steen and Coupé 1997). Approximately 30% of the Chilcotin study area was burnt to varying degrees by the Plateau Fire (Figure 1.9) and ~3% of the Enterprise study area was burnt by the Gustafsen Lake Fire during the 2017 wildfire season, which at the time was the most severe wildfire season on record for BC (Province of BC 2017).

Figure 1.5. Forests in the Chilcotin study area consisted of a mosaic of lodgepole pine stands of various ages with older spruce stands concentrated around streams and wetlands.

Figure 1.6. Typical spruce and pine stand with hair-snagging device set up in the Chilcotin study area.

Figure 1.7. Forests in the Enterprise study area consisted of multi-layered Douglas-fir stands with spruce and aspen localized around riparian areas.

Figure 1.8. Example of a multi-layered Douglas-fir stand in the Enterprise study area.

Figure 1.9. Approximately 30% of the Chilcotin study area was burned during the 2017 Plateau wildfire.

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CHAPTER 2

FACTORS DRIVING POPULATION DENSITIES OF ENDANGERED FISHERS (*PEKANIA PENNANTI*) IN TWO GEOGRAPHICALLY DISTINCT REGIONS OF BRITISH COLUMBIA, CANADA.

Introduction

Accurately and precisely estimating population density is critical for effective conservation and management of wildlife. Population data are crucial for proper evaluation of the conservation status of a species (Jimenez et al. 2017), making decisions about deploying limited resources to protect threatened species (Luo et al. 2020), and setting sustainable harvest levels (Fuller et al. 2016). However, estimating the density of some carnivores can be particularly difficult because they often range widely and tend to occur naturally at low densities (Long et al. 2010, Obbard et al 2010, Krohner 2021). Rare carnivores also are often highly sensitive to disturbance and at greater risk of extinction, thus being frequently prioritized for conservation (Travaini et al. 1997, Thompson 2013). Unfortunately, for many threatened and rare species, detailed population data are lacking and not enough is known to develop appropriate conservation measures (Duggan et al. 2015, White et al. 2020).

Fishers (*Pekania pennanti*) are a rare and elusive member of Family Mustelidae distributed across the boreal and temperate coniferous and mixed-wood forests of North America (Proulx et al. 2004). In the late 1800s, the range of fishers in North America underwent significant contractions stemming from habitat loss and fragmentation, overexploitation in the fur trade, and mortality from predator-control programs (Powell 1993, Krohn et al. 2012, Lewis et al. 2012). The consequence of this large-scale habitat loss coupled with unregulated fur harvest was the disappearance of fishers from many regions across the southern extent of their range by the early 1900s (Lofroth et al. 2010, Lapoint et al. 2015).

Throughout their range, fishers are linked to forests with relatively high overhead cover (Powell 1993, Weir 2003); although not solely dependent on old-growth coniferous forests, the abundance of complex vertical and horizontal structure (e.g., large live trees, snags, logs, and moderate-to-dense-canopy cover) usually found in older temperate forests appears to be an accurate predictor of fisher habitat use and occupancy (Raley et al. 2012). Like other populations west of the Rocky Mountains, fishers in British Columbia (BC) depend on structural components typically found in higher densities in late-successional forests to fulfill their life requisites (Raley et al. 2012, Weir et al. 2012). Structures used for resting and denning (e.g., cavities, platforms, and other microstructures) are critical habitat features that can be rare across the landscape (Weir and Harestad 2003, Aubry and Raley 2006, Purcell et al. 2009). In fact, female fishers in BC appear to have the most stringent habitat requirements for reproduction, with the dens where they birth and raise their kits being found almost exclusively in the internal cavities of large diameter trees (Lofroth et al. 2010). Suitable cavities have highly specific dimensions and must be large enough to accommodate adult female fishers and up to three kits (Weir and Corbould 2008) while having an opening small enough to exclude predators, including adult male fishers (Raley et al. 2012). In early April, females use multiple different den trees continuously for the three-month period following kit parturition (Weir and Corbould 2008); this makes having an adequate supply of den trees critical for successful reproduction (Green et al. 2019, Berg et al. 2020). The disease and decay processes that form these vital denning and resting structures can take up to a century to develop, making fishers highly susceptible to habitat disturbances, such as rotational forestry, that remove these structures at rates faster than forests can redevelop them (Weir et al. 2012).

Since the mid-1900s, fishers east of the Rocky Mountains recolonized many portions of their historic range thanks to enhanced protective measures (e.g., the closing of trapping seasons), translocation efforts, and natural forest succession or reforestation (Bowman et al. 2006, Hapeman et al. 2011, Greenhorn et al. 2018). However, despite some fur harvest bans and five translocation programs, fishers west of the Rocky Mountains remain geographically restricted and occur as smaller and more isolated populations in British Columbia (BC), Canada, and Washington, Oregon, California, Idaho, and Montana (Lewis et al. 2012, Lapoint et al. 2015, Lewis et al. 2016).

The densities of fishers in parts of BC are among the lowest recorded from anywhere within their range (e.g., 8.8 fishers/1000 km^2; Weir and Corbould 2006), with the highest density for the province being reported in the northeast (16.7 fishers/1000 km^2; Weir et al. 2011). The density of fishers at the southwestern extent of the species range in the province recently was estimated at 13.1 fishers/1000 km^2 (Davis and Weir 2021). These estimates, however, are much lower than those documented in eastern North America (49 - 385 fishers/1000 km^2; as cited in Weir et al. 2003) and California (140 fishers/1000 km^2; Matthews et al. 2011). The reasons for these differences in the density of fishers in BC remain unclear but could be related to contemporary habitat loss (Weir 2003), lower prey densities (Weir et al. 2009), an increased cost of locomotion resulting from deeper snow conditions at higher latitudes (Raine 1983), as well as other factors such as community predator dynamics of which much is still unknown. Table 2.1 shows density estimates for fishers from other areas of North America.

Table 2.1. Density estimates for fishers from elsewhere in North America, with 95% confidence intervals (95% CI) included if reported. Density estimates from British Columbia are in bold.

Study Area	Estimation Method	Density (Fishers/1000 km^2)	95% CI	Source
Southcentral Maine, USA	Mark-resight	50 - 120	Not reported	Arthur et al. (1989)
Southern Quebec, Canada	Mark-resight	300	267 - 333	Garant and Crete (1997)
Northcentral Massachusetts, USA	Mark-resight	250	Not reported	Fuller et al. (2001)
Northcentral British Columbia, Canada	**Minimum number alive**	**8.8**	**7.7 - 9.9**	**Weir and Corbould (2006)**
Southern Sierra Nevada, California, USA	Mark-resight	100	67 - 144	Jordan (2007)
Hoopa Valley Indian Reservation, California, USA	Mark-resight	520	430 - 640	Matthews et al. (2011)
Hoopa Valley Indian Reservation, California, USA	Mark-resight	140	130 - 160	Matthews et al. (2011)
Northeastern British Columbia, Canada	**Minimum number alive**	**16.3**	**11.6 - 21**	**Weir et al. (2011)**
Western New York, USA	Spatial capture-recapture	45	20 - 101	Linden et al. (2017)
Southcentral British Columbia, Canada	**Spatial capture-recapture**	**13.1**	**6.3 - 27.4**	**Davis and Weir (2021)**

Recent research has elucidated there are two distinct populations of fishers in BC - Boreal and Columbian population, respectively - separated by a high snowpack zone delineated by the Rocky Mountain divide and the mountainous region south of the Spatsizi Plateau (see Figure 1.2 in Chapter 1). The most recent estimate for the Boreal population is 896 – 1519 adult individuals (BC CDC 2020a) with this population presumably still being relatively contiguous with other populations in the boreal forests of Alberta. The most recent estimate for the Columbian population, however, is exceedingly low at 299 – 517 adult individuals (BC CDC 2020b); given that fishers have effectively been extirpated from most of southern BC this new research suggests the Columbian population of fishers is geographically isolated from other populations in

North America with demographic rescue through immigration unlikely. In recent decades, the low-elevation forests these fishers rely on in the central interior of BC have been subject to extensive disturbance from industrial development (e.g., hydroelectric, oil and gas), ongoing timber harvesting activities, large-scale insect infestations, extraordinary wildfire seasons, and the accelerated salvage harvest of insect- and fire-damaged trees (Eng et al. 2005, Province of BC 2017, Province of BC 2018). Isolation, habitat loss, and the accompanying decline in population size resulted in the BC government recently revising the conservation status of the Columbian population to endangered (BC CDC 2020b).

Although recent trends indicate the Columbian population is declining, the true extent of any declines remains unknown due to inadequate monitoring of fishers at the provincial scale. In BC, ecosystems are classified based on climate, elevation, and vegetation characteristics and thus far surveys for fishers have occurred only in the relatively higher productivity ecosystems in the northcentral (Weir and Corbould 2006) and northeastern portions of the province (Weir et al. 2011), and one smaller scale study at the extreme southwestern periphery of the species provincial range (Davis and Weir 2021). Additionally, naïve density estimates were generated for two 400 km^2 pilot study areas in moderate to high capability fisher habitat using DNA-based methods; however, these did not include any spatial information and were based on small sample sizes (2 and 5 fisher individuals identified; Davis 2004). The range of the Columbian population in BC encompasses a diversity of ecosystems and because the density of fishers can vary widely among regions depending on habitat quality (Davis and Weir 2021), detailed inventory information from different parts of the province is needed to improve the reliability of the population estimate and identify key factors affecting densities in these areas.

As a result of the patchy nature of these monitoring efforts there still are considerable gaps in knowledge about the current distribution and abundance of fishers over expansive areas of BC’s central interior, including where large-scale habitat losses have already occurred. Without representative estimates of the density of fishers across the range of the Columbian population, it will be challenging for wildlife managers to accurately assess population trends and the true extent of the apparent declines being seen in central BC. Therefore, the main objective of this study was to use non-invasive DNA-based surveys in two representative ecosystems within the range of the Columbian population of fishers coupled with spatially explicit capture-recapture methodologies to estimate density and abundance where current data do not exist. Additionally, I investigated habitat factors known from previous research to be important for predicting fisher density across these regions, thus gaining valuable insight into current habitat associations specific to these ecosystems. I predicted that the density of fishers would be positively associated with stands containing more denning and resting structures, overhead cover, mature forest, riparian features, and higher soil moisture levels, and negatively associated with open areas (e.g., wetlands, recent burns, young cut blocks). The information gained from this study can be used by land managers to refine the current population estimate and identify key habitats where fishers are enduring in the central interior of BC, allowing these areas to be prioritized for increased protection and other fisher management strategies.

Materials and Methods

Study area

I conducted DNA-based hair-snagging surveys in two study areas within the range of the Columbian population of fishers. The first was a 2,440 km^2 region on the Chilcotin Plateau west of Williams Lake, where data were collected during winter 2018-19 (hereafter named the Chilcotin study area; see Figure 1.3 in Chapter 1); the second was located southeast of Williams Lake (2,580 km^2) where data were collected during winter 2020-21 (hereafter named the Enterprise study area; see Figure 1.4 in Chapter 1). Both study areas consisted of level to gently rolling topography and surveys were conducted in non-mountainous areas at low to moderate elevations (750 – 1700 m).

The Chilcotin study area was within the driest ecological zone where fishers are found in BC with forests dominated by lodgepole pine (*Pinus contorta*) and a small component of Douglas-fir (*Pseudotsuga* menziessi), and trembling aspen (*Populus tremuloides*), and hybrid spruce (*Picea glauca* x *engelmannii*) concentrated around streams and wetlands (Meidinger and Pojar 1991). The Enterprise study area also was within the drier central interior of the province where forests are dominated by either multi-aged Douglas-fir or uniform-aged lodgepole pine stands, with small trembling aspen and hybrid spruce stands occurring locally (Steen and Coupé 1997). For additional details on my study areas see Chapter 1.

Both study areas have extensive histories of logging and mining, with large numbers of pine trees also killed by mountain pine beetle (*Dendroctonus ponderosae*) and subsequently salvage harvested at accelerated rates during the past two decades (Eng et al. 2005). These industries built and continue to build pervasive road networks in this region, although the mining industry is less active than it was historically. Western portions of the Enterprise study area were previously logged using a partial cutting silvicultural system to maintain mule deer (*Odocoileus hemionus*) winter range habitat and were selectively logged in attempts to suppress several patchily-distributed Douglas-fir beetle (*Dendoctronus pseudotsugae*) infestations. Approximately 30% of the Chilcotin study area was burnt to varying degrees by the Plateau Fire and ~3% of the Enterprise study area was burnt by the Gustafsen Lake Fire during 2017, which at the time was the most severe wildfire season on record for BC (Province of BC 2017).

Registered traplines overlapped both study areas with a legal trapping season for fishers open from November 1st to February 15th during both survey winters (Province of BC 2020). Since the year 2000, a total of 142 fishers were trapped from the 9 registered traplines that overlapped with the Enterprise study area (Province of British Columbia unpubl. data: BC Wild Fur Harvest Database 2000 - 2019 [accessed 15 July 2022]). Within the same time period, a total of 30 fishers were trapped from the 6 traplines overlapping the Chilcotin study area (Province of British Columbia unpubl. data: BC Wild Fur Harvest Database 2000 - 2019 [accessed 15 July 2022]).

Fisher DNA sampling

I conducted DNA-based surveys by remotely collecting hair samples following consistent sampling methodology in the two study areas. I divided each study area into 20 km^2 cells to approximate the smallest expected size of a female home range in the central interior (Weir et al. 2009). Field crews established sampling sites in each cell using detectors that snagged and collected hair and follicle tissue but did not restrain animals. Crews deployed detectors following the design of Foran et al. (1997), with two pieces of wood (2 x 19 x 60 cm) screwed together to form a triangular ‘cubby’ and both ends left open (Figure 2.1). Field personnel baited cubbies with a piece of chicken attached via tie wire to the middle of the inside of the cubby, and smeared a commercial beaver castor (Sharpe’s Beaver Lure)and fisher lure (Sharpe’s Fisher Lure) mixture on the chicken and a woolen jute string hung on branches next to the cubby to act as attractants. Cubbies had four pieces (approximately 1.5 x 5 cm) of adhesive-based mouse-trap paper fastened to the inside (2 at each end) to collect hair from individual fishers when they tried to access the bait inside the cubby. Crews affixed cubbies vertically to the bole of a tree using 4 – 7.5 cm long screws and placed a 19 x 30 cm roof board above the unit to prevent rain or snow from degrading any hair samples present.

Field crews accessed sampling sites by foot, pick up truck, snowmobile, and helicopter, and deployed cubbies at sites in the best available habitat (Mowat and Paetkau 2002). Crews revisited sites at 21-day intervals beginning in mid-December and ending in mid-March for N = 4 sampling sessions each winter. Upon each revisit crews assessed the glue pads and cubby edges for hair and follicle tissues. Glue pads that had collected hair were covered with plastic paper and placed in a paper envelope and stored in a dry environment to be sent for processing at the end of the survey. Field personnel moved sampling sites at least once within a cell (>800 m from the previous site) during the winter to reduce the possibility of habituation, maximize the number of spatial recaptures of the same individuals at different locations, and evenly distribute sampling efforts across the study areas.

Figure 2.1. Typical set up of triangular wooden hair-snagging ‘cubby’ fastened vertically to a tree. Inset picture shows the inside of the device with bait and adhesive glue strips used to collect fisher hair.

I sent all hair samples to Wildlife Genetics International (WGI) in Nelson, BC for microsatellite genotyping. An additional 7 fisher hair and tissue samples collected from a trapper operating in the Enterprise study area at the time of the survey were analyzed to determine if those animals were also detected at our sampling sites. Lab technicians selected the best available samples for analysis, ideally using clipped roots of 10 guard hairs where possible, and/or up to 30 whole under-fur hairs if needed to supplement guard hair. Technicians used QIAGEN DNeasy Blood and Tissue kits (Qiagen, Toronto, ON, Canada) to extract DNA. The lab identified hair samples to species by partially sequencing the mitochondrial 16S rRNA gene (Johnson and O'Brien 1997). For samples identified as fishers, technicians determined the individual identity of each animal using 7 microsatellite markers previously identified in an earlier study on fishers in BC (Lut604, MP0055, Ma-1, MP0247, Mvis072, MP0144, MP0182; Weir et al. 2013). For each individual identified by its multi-locus genotype, the lab was able to determine sex using the ZFX/ZFY/SRY gender marker (Davis and Weir 2021). Error checking followed Paetkau (2003), proven with other species to consistently produce low error rates (Kendall et al. 2009).

S***patial capture-recapture analysis***

I employed spatially explicit capture-recapture (SECR) methods that used a maximum likelihood framework (Borchers and Efford 2008, Efford et al. 2009) to estimate the density of fishers in each study area. This approach uses the detection history of identifiable animals (i.e., location and timing of captures) to typically estimate three parameters: density (D), detection probability (g0), and a spatial parameter σ (Efford et al. 2009). Like many other methods used for estimating animal abundance, SECR combines a state model to describe variation in the data due to an ecological process (i.e., how animals are distributed on the landscape), and an observation model to describe imperfections in the observation of the process (i.e., just because a fisher is present does not mean it will consistently be detected). In the state model, the distribution of home range centres in a population initially is treated as a homogeneous Poisson point process where density (= intensity) is the sole parameter of the process (Borchers and Efford 2008). An inhomogeneous Poisson process also may be fitted to evaluate the effects that different covariates have on the predicted density (Efford 2021). For the observation model, a function is used to describe the decline in detection probability (g0) with distance from an animal's home range center (Efford 2021). Therefore, g0 is technically defined as the detectability of an individual at a certain detector if the individual's activity center was at that exact location. The spatial scale parameter σ is related to the range size during sampling occasions and is used along with animal capture histories to estimate range centers of individuals in the sampling area and the associated area of integration (Borchers and Efford 2008). Estimation of density, g0, and σ is achieved by numerically maximizing the likelihood with respect to the parameters of each of the sub-models (Efford et al. 2009).

I used package *secr* (Efford 2021) in Program R (R Version 4.0.2; R Core Team 2020) to estimate the density of fishers in the two study areas. I first constructed a 'habitat mask', which is a grid of points used to facilitate computation, to delineate the bounds of the area of integration (i.e., the total area over which density was estimated), and to store habitat covariates for spatial models of density. Following Efford (2021), I estimated the root pooled spatial variance to generate a preliminary and biased estimate of σ and buffered the trap array for each study area by four times this preliminary σ (10,000 m and 18,000 m buffer widths for the Chilcotin and Enterprise study areas, respectively). I then evaluated each of these buffer sizes to ensure that the masks were large enough that animals outside the habitat mask would have a negligible probability of being detected within the sampling areas so as not to potentially affect density estimates (Efford 2021). I chose a mask point spacing of 2200 m that fell within the range of recommended sizes (<1*σ; Efford 2021) and yielded a computationally reasonable number of mask points for both study areas (991 for the Chilcotin, 1220 for the Enterprise). The Interior Douglas-fir very dry-mild (IDFxm) BEC zone is not considered viable fisher habitat in BC (Weir and Almuedo 2010) therefore I excluded any mask points falling in this zone, and those within large bodies of water, from the final habitat masks.

Detection covariates

The SECR framework allows for both standard and user-defined covariates to be assigned to all three modelling parameters (Efford 2021), and I first evaluated the effects of ecologically and behaviourally relevant covariates on g0 and σ to determine the candidate observation model most supported by the data to be included in the second step of the analysis (i.e., density estimation). For the detection models, I used a half-normal function to model the shape of decline in detection probability with distance from home range center.

Previous research has indicated that mustelids may become ‘trap-happy’ or ‘trap-shy’ (Royle et al. 2011, Mowat et al. 2019) therefore I included automated behavioural response covariates (Efford 2021) where parameters may depend on detection at the preceding occasion (B), there is a step change after first detection (b), site effectiveness changes once any animal is caught (k), and site effectiveness changes if there was a detection on the preceding occasion (K). I expected that detection parameters might change over the course of the winter, so I evaluated this potential effect by including a time covariate with one level for each occasion (t) and a time trend factor where there is a linear trend over occasions on the link scale (T). Male fisher home ranges are substantially larger than females in BC (Weir et al. 2009) and both sexes use space differently: I also therefore included a sex covariate (h2). For the Chilcotin study area, I expected that burn severity intensity also may influence detection parameters and included it as separate user-defined covariates (i.e., whether a detector was in an unburned, low burn intensity, medium burn intensity, or high burn intensity polygon).

Density covariates

I evaluated habitat factors that could affect fisher density by creating several candidate models containing covariates that previous studies have shown to be related to fisher use (Table 2.2). I used spatial data from the BC Data Catalogue (Province of BC 2021) and the BC Fisher Habitat Forestry Web module (BC Fisher Habitat Forestry Web Module 2021) to assess the effect of habitat covariates on fisher density in our study areas (see Appendix A for a complete list of references and variable descriptions). Because fishers have been shown to select habitat at multiple scales (Weir and Harestad 2003), I used a moving window analysis to re-scale covariates using two different buffer sizes around mask points representing the average size of a female fisher core use area (4.84 km^2; Weir 1995) and an average home range size for female fishers (30 km^2, Davis 2009: Weir et al. 2009) in central interior BC. I based these buffer sizes on the female portion of the population because they appear to have the most specific habitat requirements in BC (Weir and Corbould 2008, Davis 2009, Lofroth et al. 2010). I calculated the proportion, length, or density of covariates within these buffer areas and these data were appended to the habitat mask prior to fitting candidate models. I tested for correlation between habitat covariates using the *corrplot* function in the R package *corrplot* (Wei 2021) and did not include highly correlated covariates in the same model (Spearman's r Rank >0.6); Dormann et al. 2013).

For both detection and density models, I used Akaike's Information Criterion corrected for small samples sizes (AIC_c) to evaluate the support for each model in a candidate set (see Appendix B for a full list of model candidate sets) to identify the model that was best supported by the data (Burnham and Anderson 2002). I first compared detection models from each candidate set to determine the detection variables best supported by the data. I chose detection models based on support from the data and knowledge of fisher biology while also considering model parsimony. I then included this detection model in the second step of the analysis identifying which density model was best supported by the data. Models within two AIC_c of each other were considered equivalent and those >4 AIC_c units away from the top model were considered to have negligible support from the data (Burnham and Anderson 2002). I quantified the strength of evidence for top models using Akaike weights (w_i; Burnham and Anderson 2002) and identified the 95% confidence set of best models where $\sum w_i \geq 0.95$.

Table 2.2. Variables used in candidate models known from previous research to influence the density of fishers within the Chilcotin and Enterprise study areas in the central interior of British Columbia, Canada. I report the variable name, variable description, and whether it was used in the candidate model set for each study area (C = Chilcotin, E = Enterprise). The area, length, or density of each variable was calculated for both the 4.84 km^2 (core use scale) and the 30 km^2 (home range scale) buffer areas around each habitat mask point. A full description of variables with their applicable reference can be found in Appendix A.

Variable	**Description**	**Study Area**
Primary branch resting	Area of spruce-leading, secondary, or tertiary stands aged ≥83 years old with crown closure ≥40%	C + E
Primary denning	Area of aspen- or cottonwood-leading stands aged ≥135 years old or Douglas-fir-leading stands aged ≥207 years old with crown closure ≥20%	C + E
Primary coarse wood resting	Area of stands with spruce or aspen content ≥25% and aged ≥100 years old	C + E
Primary movement	Area of stands with total cover ≥50% (≥30% shrub cover and ≥20% tree cover)	C + E
High intensity burn	Area of stands with trees dead, needles, twigs, and understory consumed	C
Medium or high intensity burn	High intensity description or, stands with trees dead, scorched needles remain on trees, understory burned	C
Stream density	Density of streams in km/km^2	C + E
Stream length	Total length of all streams in km	C + E
Prey density	Estimated total relative density of digestible energy based on kcal/g provided by snowshoe hares and squirrels	C + E
Wet soil moisture regime	Area of stands with subhygric, hygric, or subhydric soil moisture regimes. Located primarily along wetlands and streams, contain spruce, and a relatively well-developed shrub layer	C + E
Spruce-aspen riparian forest	Area of spruce-or aspen-leading stands within 100 m of a wetland	C + E
Riparian closed canopy forest	Area of stands with ≥30% crown closure within 100 m of wetland	C + E
Open areas	Area of recently logged or burned stands (<12 years old), wetlands, and agricultural fields	C + E
Old riparian forest	Area of stands >100 years old within 100 m of wetland or stream	C + E
Cut blocks harvested <20 years ago	Area of cut blocks harvested <20 years ago	C + E
At-leading stands	Area of aspen- or cottonwood-leading stands	E
Sx-dominated riparian forest	Area of spruce-leading stands within 100 m of a stream or wetland	E
Road density	Density of roads in km/km^2	C + E
Active trap line	Area of one active registered trap line during the Enterprise survey	E
Mature and old stands	Area of stands >80 years old	E
Old Douglas-fir forest	Area of Douglas-fir-leading or secondary stands ≥100 years old	E

As with other capture-recapture modelling, the SECR framework uses model formulae that define variation in each parameter as a function of covariates that is linear on a 'link' scale (Efford 2021). Fitting a model provides estimates of its 'beta coefficients' which can then be used to make predictions (Efford 2021). For each model parameter (i.e., D, g0, and σ) I calculated beta coefficients and their associated 95% confidence intervals (CI) and if the 95% CI spanned zero we treated those variables as potentially informative but not reliable predictors. I used the top-ranked density models (which included the top-ranked detection sub-model) and the *region.N* function in *secr* (Efford 2021) to estimate population size of fishers across habitat mask areas. I used the *predictDsurface* function included in *secr* (Efford 2021) which takes the beta coefficients from the top-ranked density models on the default log-link scale to predict the density of fishers across habitat mask areas. I then created another habitat mask encompassing the larger ecologically similar areas immediately adjacent to both study areas and used the same top-ranked model coefficients to predict the density of fishers at each mask point. I created a raster layer of the predicted density at each mask point using the *raster* function in *secr* (Efford 2021) and developed final maps for a visual representation of the density of fishers across the larger surrounding ecologically similar areas as predicted by the top models.

I initially ran density models using combined data for both sexes as I had relatively small sample sizes for the Enterprise study area and combined sex models often produce nearly identical density estimates (Efford and Mowat 2014, Mowat et al. 2019) For the Chilcotin study area, I also ran separate models for each sex.

Results

Chilcotin study area

In the Chilcotin study area I identified 48 individual fishers (15 males, 33 females) through 127 detections across 256 sampling sites over the course of the winter. A total of three fishers were harvested from a trapline overlapping part of the Chilcotin study area in the winter of 2018-19, according to the BC Provincial Wild Fur Harvest database (Province of British Columbia unpubl. data: BC Wild Fur Harvest Database 2018 - 2019 [accessed 15 July 2022]), however, the individual identity of these animals went unconfirmed as hair samples were not shared with us by the trapper. The top-ranked detection model (pooled across sexes) included a behavioural effect, where detection probability depended on whether a fisher was detected at a site on the previous occasion. This best detection model was >16 AIC_c units lower than the null model (Table 2.3). This learned effect also was included in the three top-ranked detection models for females only that were well supported by the data (ΔAIC_c <2.0), and the top model ranked >19 AIC_c units lower than the null female-only model (Table 2.3). The top-ranked male-only detection model included a site learned response where site effectiveness changes once any animal is caught, however, the model with detection probability and σ fixed also performed well (ΔAIC_c <1.0; Table 2.3). Although multiple detection models were well-supported by the data for the combined sex and female-only models, I retained the top-ranked detection models for future fitting as I felt they were reasonable based on the ecology of the species. I retained the male-only detection model with detection probability and σ fixed since this minimized the overall number of model parameters.

For the top-ranked detection model pooling both sexes, detection probability at the home range center was 0.17 ± 0.039 *SE* for naïve fishers detected at the trap for the first time, and 0.42 ± 0.060 *SE* if there had been any previous fisher detection at the site. This model also predicted a ratio of 31 ± 0.067 *SE* ♂♂ to 69 ± 0.067 ♀♀. Detection probability for the female-only model followed a similar pattern with the value at the home range center being 0.12 ± 0.045 *SE* for naïve fishers and 0.47 ± 0.079 *SE* if there was a detection on the previous occasion. Detection probability at the home range center for the male-only model (i.e., null model) was 0.28 ± 0.067 *SE*. σ values were similar between all three model categories: for the model combining both sexes σ was 2.75 km ± 0.16 *SE*, was 2.68 km ± 0.19 *SE* for the female-only model, and was 2.83 km ± 0.29 *SE* for the male-only model.

After selecting the detection models, the top-ranked models involving habitat variables showed consistent patterns at both spatial scales (i.e., core use area and home range), producing nearly identical density and abundance estimates. Several of the 95% confidence set of candidate models contained nearly all of the candidate models I considered (Appendix C). Density for the combined sex model was most strongly related to area of primary branch resting stands at both the core use area (4.84 km^2) and home range area (30 km^2) scales (Table 2.4). The top-ranked combined sex model at the core use scale was 16 times more likely and scored 5.5 AIC_c units lower than the null model that predicted no effect of habitat on density, and the top-ranked combined sex model at the home range scale was 5 times more likely and scored 3.1 AIC_c units lower than the null model. Several other combined sex models that included other habitat covariates in addition to area of primary branch resting stands were well-supported by the data, however, the confidence intervals of the beta coefficients for the second variable in each of these models spanned zero, suggesting they were potentially informative but no longer a reliable predictor.

Table 2.3. Model selection table to evaluate detection for fishers in the Chilcotin and Enterprise study areas in the central interior of British Columbia, Canada. I estimated 3 parameters: density (D), detection probabiity (g0), and a spatial parameter (sigma). No density covariates were included in this analysis. The table shows the number of model parameters (K), model log likelhood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (wi). Models ≤2.0 AICc units of the top-ranked models are displayed here, with the null model assuming g0 and sigma are fixed shown for reference.

Study Area	**Sex**	**Model**	**K**	**logLik**	**AIC$_c$**	**ΔAIC$_c$**	***w*$_i$**
Chilcotin	Both	g0~B sigma~1	5	-445.1	901.59	0.00	0.37
		g0~B + T sigma~1	6	-444.8	903.68	2.09	0.13
		g0~1 sigma~1 (null)	4	-454.4	917.79	16.21	0.00
	Female	g0~B sigma~1	4	-285.5	580.41	0.00	0.40
		g0~B sigma~B	5	-284.3	580.79	0.38	0.33
		g0~B + T sigma~1	5	-284.6	581.44	1.04	0.24
		g0~1 sigma~1 (null)	3	-297.2	601.24	20.83	0.00
	Male	g0~k sigma~1	4	-154.8	321.55	0.00	0.22
		g0~bk sigma~1	4	-155.1	322.23	0.68	0.16
		g0~1 sigma~1 (null)	3	-157.1	322.43	0.88	0.14
		g0~Low sigma~1	4	-155.4	322.84	1.29	0.12
		g0~Bk sigma~1	4	-155.8	323.52	1.96	0.08
Enterprise	Both	g0~1 sigma~B	5	-149.6	313.27	0.00	0.37
		g0~1 sigma~1 (null)	4	-152.3	315.08	1.81	0.15

*****Detection covariates included whether there was a detection on the previous occasion (B), a trap-specific behaviour (bk), a trend over trapping occasion (T), site effectiveness changes once any animal is caught (k), and low intensity burn (Low). A ‘1’ indicates g0 or sigma remained constant.

Table 2.4. Model selection table showing the top-ranked density models for fishers in the Chilcotin study area in the central interior of British Columbia, Canada. I estimated 3 parameters: density (D), detection probabiity (g0), and a spatial parameter (sigma) at the female fisher core use area scale (4.84 km^2) and the home range scale (30 km^2). The table shows the number of model parameters (K), model log likelhood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (w_i). Models ≤2.0 AICc units of the top-ranked models are displayed here, along with the null model that assumes all parameters are constant.

Sex	**Scale**	**Model**	**K**	**logLik**	**AIC_c**	**ΔAIC_c**	**w_i**
Both	Core use	D ~ Primary branch resting, g0 ~ B, sigma ~1	6	-441.0	896.09	0.00	0.26
		D ~ Primary branch resting + stream density, g0 ~ B, sigma ~1	7	-440.5	897.83	1.75	0.11
		Null	5	-445.1	901.59	5.50	0.02
	Home range	D ~ Primary branch resting, g0 ~ B, sigma ~1	6	-442.2	898.49	0.00	0.17
		D ~ Primary denning, g0 ~ B, sigma ~1	6	-442.9	899.91	1.41	0.08
		D ~ Primary denning + high intensity burn, g0 ~ B, sigma ~1	7	-441.7	900.22	1.72	0.07
		D ~ wet soil moisture, g0 ~ B, sigma ~1	6	-443.2	900.40	1.90	0.07
		Null	5	-445.1	901.59	3.09	0.04
Female	Core use	D ~ wet soil moisture, g0 ~ B sigma ~ 1	5	-282.7	577.56	0.00	0.24
		D ~ Primary denning, g0 ~ B sigma ~ 1	5	-283.2	578.70	1.14	0.13
		D ~ Sx- or At-leading stands near wetland, g0 ~ B sigma ~ 1	5	-283.4	578.95	1.39	0.12
		Null	4	-285.5	580.41	2.85	0.06
	Home range	D ~ wet soil moisture, g0 ~ B sigma ~ 1	5	-283.2	578.72	0.00	0.22
		D ~ Primary denning, g0 ~ B sigma ~ 1	5	-283.9	580.06	1.34	0.11
		D ~ Sx- or At-leading stands near wetland, g0 ~ B sigma ~ 1	5	-284.0	580.21	1.49	0.10
		Null	4	-285.5	580.41	1.69	0.09
Male	Core use	D ~ Primary branch resting, g0 ~ 1 sigma ~ 1	4	-152.8	317.60	0.00	0.44
		D ~ Primary branch resting + stream density, g0 ~ 1 sigma ~ 1	5	-151.1	318.89	1.29	0.23
		Null	3	-157.1	322.43	4.83	0.04
	Home range	D ~ Primary branch resting, g0 ~ 1 sigma ~ 1	4	-153.6	319.21	0.00	0.42
		D ~ Primary branch resting + stream density, g0 ~ 1 sigma ~ 1	5	-152.7	322.10	2.89	0.10
		Null	3	-157.1	322.43	3.23	0.08

covariates in addition to area of primary branch resting stands were well-supported by the data, however, the confidence intervals of the beta coefficients for the second variable in each of these models spanned zero, suggesting they were potentially informative but no longer a reliable predictor.

Density in the top male-only model also was most strongly related to area of primary branch resting stands at both the core use (4.84 km^2) and home range (30 km^2) scales (Table 2.4).

The top-ranked male-only model at the core use scale was 11 times more likely and scored 4.8 AIC_c units lower than the null model that predicted no effect of habitat on density, and the top-ranked male-only model at the home range scale was 5 times more likely and scored 3.3 AIC_c units lower than the null model. The second-ranked male-only model that included area of primary branch resting stands and length of streams within both the 4.84 km^2 and 30 km^2 buffer areas also had good support from the data (i.e., within 1.75 AIC_c units of top model), but the beta coefficient of the stream covariate had a 95% confidence interval that spanned zero, making it no longer a reliable predictor.

The top-ranked model from the candidate set for females predicted that density was related to the area of stands with wet soil moisture regimes (i.e., subhygric, hygric, or subhydric soils) within both the 4.84 and 30 km^2 buffer areas, with the models including area of primary denning stands and spruce- or aspen-leading stands within 100 m of a wetland also being well-supported (Table 2.4). Correlation analysis indicated these three habitat variables were also weakly correlated (Spearman's r Rank <0.4) which aligned with the modelling results. The top-ranked female-only model at the core use scale was 4 times more likely and scored 2.8 AIC_c units lower than the null model that predicted no effect of habitat on density. The top-ranked female-only model at the home range scale was twice as likely and scored 1.7 AIC_c units lower than the null model that predicted no effect of habitat on density.

The top-ranked model for all fishers estimated an average density of 21.3 ± 3.8 *SE* fishers/1000 km^2, or an abundance of 101.9 ± 18.0 *SE* fishers, across the entire 4,790 km^2 Chilcotin habitat mask area. The top-ranked male-only model estimated an average density of 5.2 ± 1.4 *SE* male fishers/1000 km^2, or an abundance of 25.1 ± 6.7 *SE* male fishers, whereas the female-only model estimated an average density of 15.3 ± 4.3 *SE* female fishers/1000 km^2, or an abundance of 73.2 ± 20.6 *SE* female fishers. Density in the top-ranked combined sex model was positively related to the area of branch resting stands within both the 4.84 km^2 (Table 2.5; $\beta = 10.9$; 95% CI: 5.7 – 16.1) and the 30 km^2 (Figure 2.2a, Table 2.5; $\beta = 13.8$; 95% CI: 4.8 – 22.9) buffer areas. The top-ranked male-only model also was positively related to the area of branch resting stands within both the 4.84 km^2 (Table 2.5; $\beta = 15.2$; 95% CI: 5.7 – 16.1) and the 30 km^2 buffer areas (Figure 2.2b, Table 2.5; $\beta = 21.6$; 95% CI: 10.2 – 33.0). Density in the female-only model was positively related to the area of stands with wet soil moisture regimes within both the

4.84 km^2 (Table 2.5; β = 4.9; 95% CI: 0.9 – 8.8) and the 30 km² buffer areas (Figure 2.2c, Table 2.5; β = 4.9; 95% CI: 0.1 – 9.7).

I used the top-ranked models at the home range scale to predict the density of fishers over a larger area that was ecologically similar to the survey grid. The combined sex and male-only models predicted a relatively uniform density over the majority of this zone, with localized areas of higher densities of fishers overlapping our study area, in the northwest and northeast corners, and a handful of small areas further south (Figures 2.3 and 2.4). The top-ranked female-only model also predicted high densities in the study area and in the northwest corner, but also with higher density pockets more well-distributed across the remainder of the area (Figure 2.5).

Table 2.5. Beta coefficients (β) and associated 95% confidence intervals (CI) for the top-ranked density models for fishers at both the core use area scale (4.84 km^2) and the home range scale (30 km^2) in the Chilcotin and Enterprise study areas in the central interior of British Columbia, Canada.

					95% CI	
Study Area	**Sex**	**Scale**	**Model**	**β**	**Lower**	**Upper**
Chilcotin	Both	Core Use	D ~ Primary branch resting, g0 ~ B, sigma ~1	10.9	5.7	16.1
		Home Range	D ~ Primary branch resting, g0 ~ B, sigma ~1	13.8	4.8	22.9
	Female	Core Use	D ~ wet soil moisture, g0 ~ B sigma ~ 1	4.9	0.9	8.8
		Home Range	D ~ wet soil moisture, g0 ~ B sigma ~ 1	4.9	0.1	9.7
	Male	Core Use	D ~ Primary branch resting, g0 ~ k sigma ~ 1	15.1	5.7	16.1
		Home range	D ~ Primary branch resting, g0 ~ k sigma ~ 1	21.6	10.2	33.0
Enterprise	Both	Core Use	D ~ wet soil moisture, g0 ~ 1 sigma ~ 1	15.8	6.8	24.7
		Home Range	D ~ Crown closure >50%, g0 ~ 1 sigma ~ 1	5.1	1.5	8.8

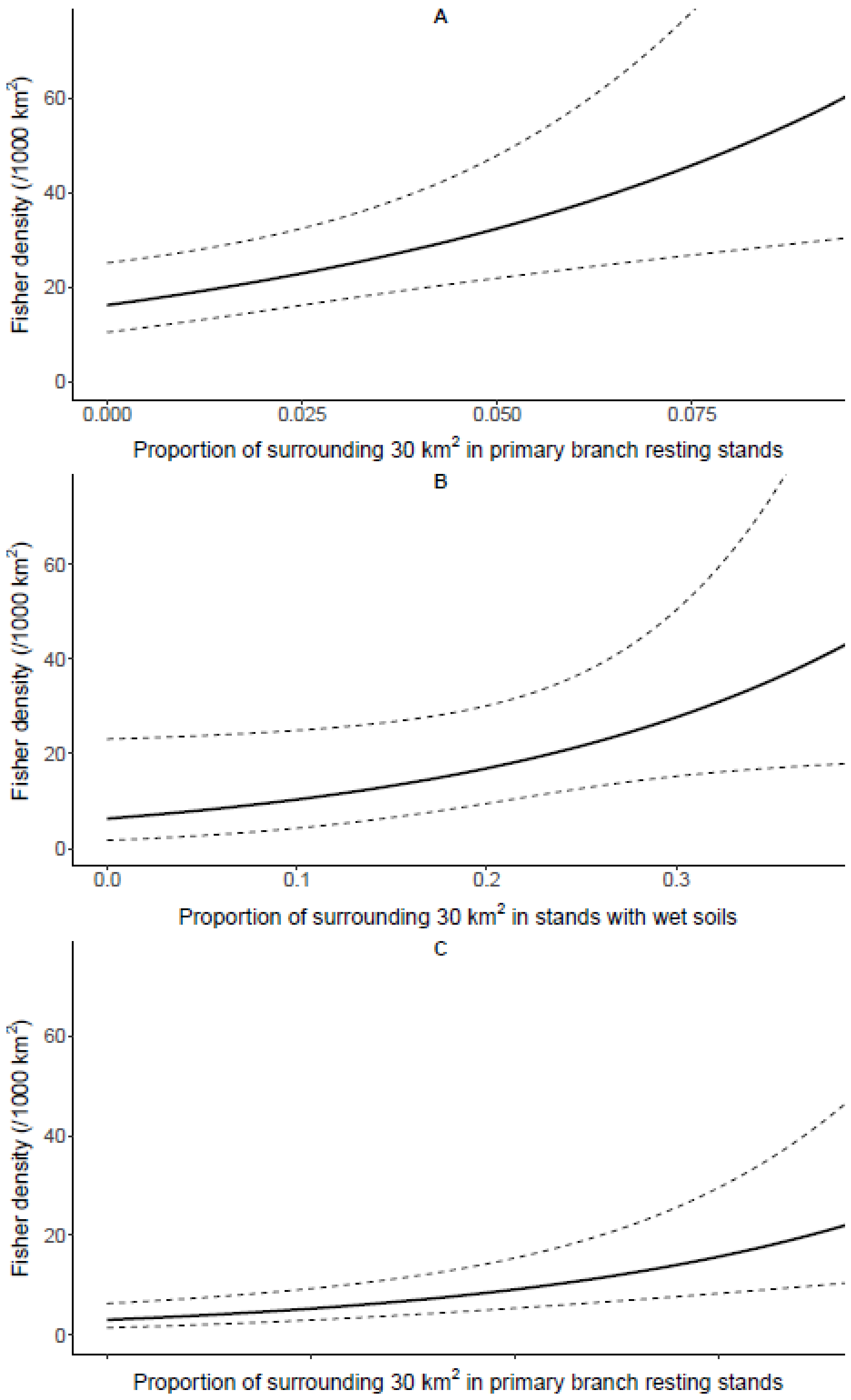

Figure 2.2. Relationship between density of fishers and variables included in the top-ranked models for the Chilcotin study area. Panel A shows the top-ranked model pooled across sexes, Panel B is for females only, and Panel C is for males only; dashed lines represent the upper and lower 95% confidence intervals.

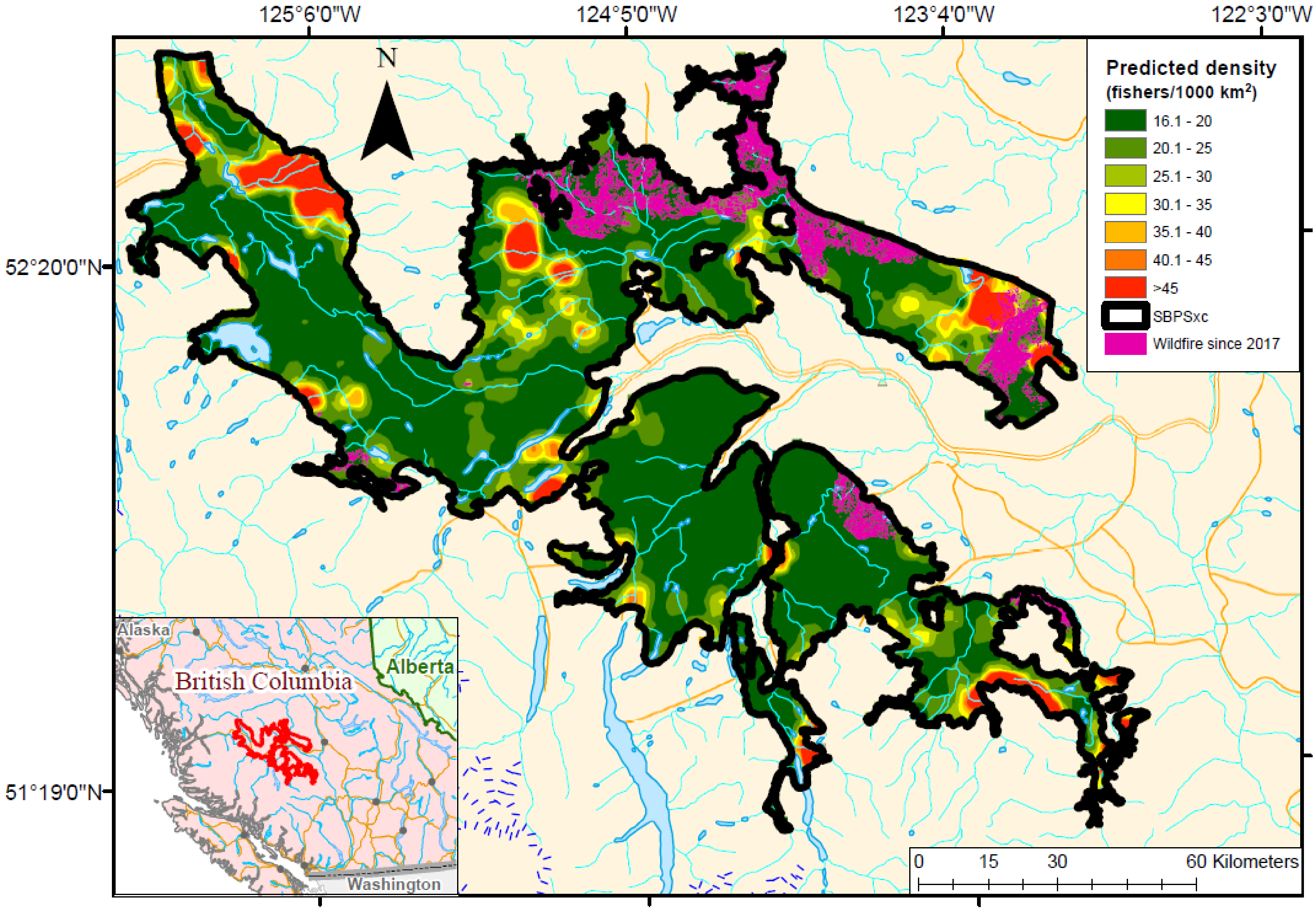

Figure 2.3. Predicted density of fishers (fishers/1000 km^2), pooled across sexes, in Sub-Boreal Pine-Spruce forests in central BC estimated from spatial capture-recapture analysis of genetically identified fishers sampled during the winter of 2018-19 as predicted by the proportion of surrounding 30 km^2 containing primary branch resting stands. SBPSxc = Sub-Boreal Pine-Spruce very dry cold BEC subzone variant.

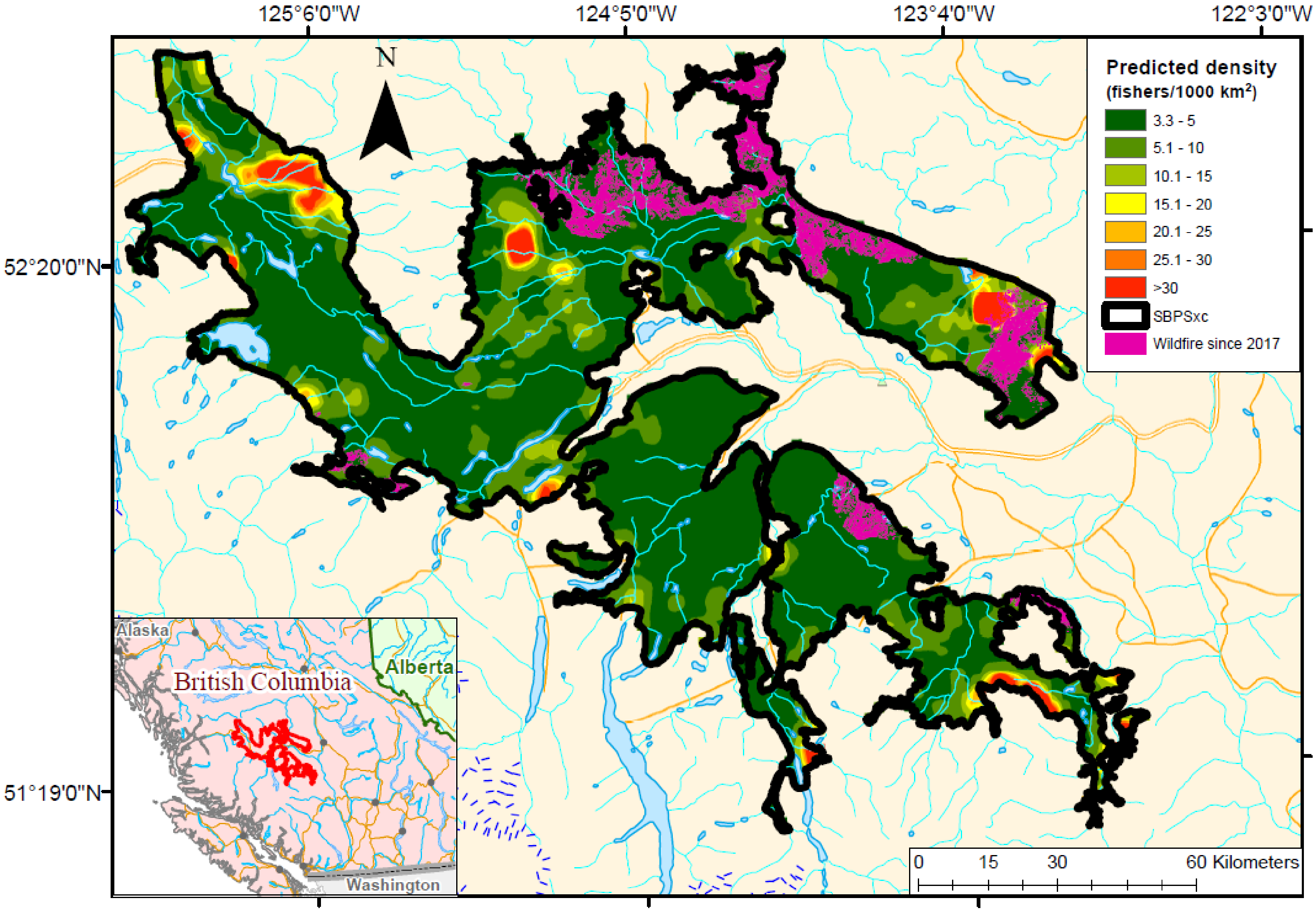

Figure 2.4. Predicted density of male fishers (fishers/1000 km^2) in the Sub-Boreal Pine-Spruce forests in central BC estimated from spatial capture-recapture analysis of genetically identified fishers sampled during the winter of 2018-19 as predicted by the proportion of surrounding 30 km^2 containing primary branch resting stands. SBPSxc = Sub-Boreal Pine-Spruce very dry cold BEC subzone variant.

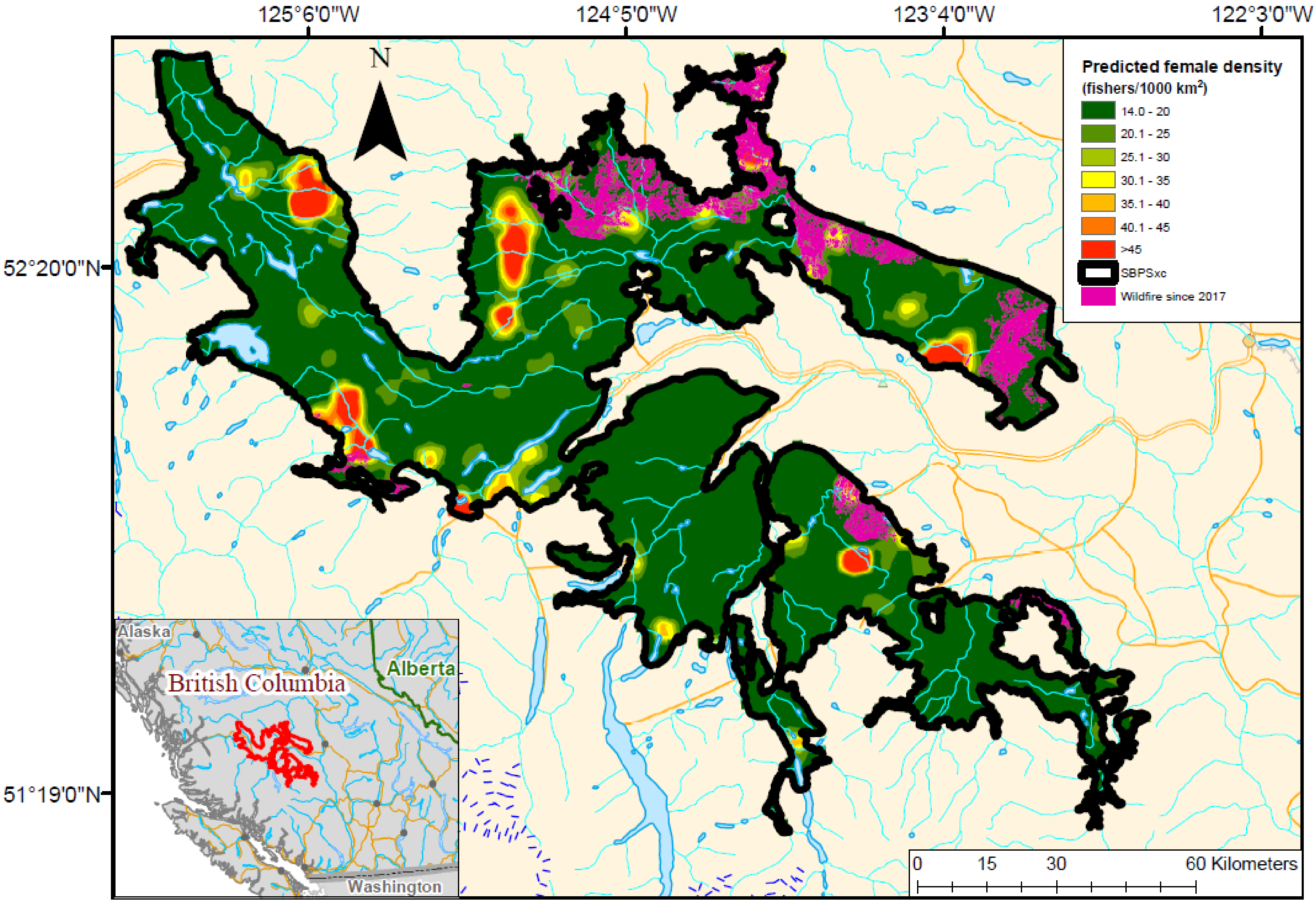

Figure 2.5. Predicted density of female (fishers/1000 km^2) in the Sub-Boreal Pine-Spruce forests in central BC estimated from spatial capture-recapture analysis of genetically identified fishers sampled during the winter of 2018-19 as predicted by the proportion of surrounding 30 km^2 containing primary branch resting stands. SBPSxc = Sub-Boreal Pine-Spruce very dry cold BEC subzone variant.

Enterprise study area

I sampled fishers between December 2020 and March 2021 in the Enterprise study area at 304 unique sampling sites and identified 22 individual fishers (6 males, 16 females) that were detected 31 times over the course of the winter. One female fisher was confirmed by genetic analysis to have been killed by a trapper operating in the study area. Five fishers from the study area were killed by trappers before my survey began, with a total of 9 unique fisher individuals killed by trappers operating within the study area during the 2020-21 fur harvest season. I was unable to assess whether 4 other trapped fishers were detected in our sampling because I did not have samples for genetic identification. Because I could not determine the identity of all individuals nor the timing and location of the fishers killed by trappers and because they were no longer part of the population, I did not include these 13 animals in my density calculations.

I documented 21 individuals recaptured at different locations only nine times and therefore did not attempt to model each sex separately and report here results from models pooled across sexes. The top-ranked detection model included a behavioural effect where the spatial scale parameter σ depended on whether a fisher was detected on the preceding occasion (Table 2.3). The null model which assumed both detection probability and σ remained constant also performed well ($\Delta AIC_c < 2.0$), and I selected this model to retain for future fitting since it reduced the number of overall model parameters. Detection probability at the home range center was very low for the Enterprise study area at 0.039 ± 0.0 *SE* and the spatial parameter σ was large at 4.67 km ± 0.0 *SE*. This model also estimated that the population had a male-to-female ratio of 29 ± 9.9 *SE* males to 71 ± 9.9 *SE* females.

I evaluated a similar suite of variables as the Chilcotin study area thought to potentially influence the density of fishers, with no variation in detection probability and σ (i.e., I included the null detection model in the second step of the analysis). At both spatial scales I investigated (i.e., female core use area and home range, 4.84 km^2 and 30 km^2, respectively) the top four models contained the same covariates and were well-supported ($\Delta AIC_c < 2.0$) but at the home range scale their ranking order was slightly different than at the core use area scale (Table 2.6). The top-ranked model at the core use area scale predicted that density was most strongly related to the area of stands with wetter than mesic soil moisture regimes (i.e., subhygric, hygric, or subhydric soils - Table 2.6; β = 10.9; 95% CI: 5.7 – 16.1). At the home range scale this model was also well-supported, but the top-ranked model predicted that the density of fishers was related to the area of stands with crown closure greater than 50%. At the core use scale, the top-ranked model was 15 times more likely and scored 5.4 AIC_c units lower than the null model with no predicted effect of habitat on density. At the home range scale, the top-ranked model was 4 times more likely and scored 2.6 AIC_c units lower than the null model with no predicted effect of habitat on density.

Table 2.6. Model selection table showing the top-ranked density models for fishers in the Enterprise study area in the central interior of British Columbia, Canada. I estimated 3 parameters: density (D), detection probabiity (g0), and a spatial parameter (sigma) at the female fisher core use area scale (4.84 km^2) and the home range scale (30 km^2). The table shows the number of model parameters (K), model log likelhood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (w_i). Models ≤2.0 AICc units of the top-ranked models are displayed here, along with the null model that assumes all parameters are constant.

Sex	Scale	Model	K	logLik	AIC_c	ΔAIC_c	w_i
Both	Core use	D ~ wet soil moisture, g0 ~ 1 sigma ~ 1	5	-147.8	309.64	0.00	0.50
		Null	4	-152.3	315.08	5.44	0.03
	Home range	D ~ Crown closure >50%, g0 ~ 1 sigma ~ 1	5	-149.2	312.45	0.00	0.22
		D ~ Open areas analysis, g0 ~ 1 sigma ~ 1	5	-149.3	312.52	0.06	0.21
		D ~ wet soil moisture, g0 ~ 1 sigma ~ 1	5	-149.8	313.60	1.15	0.12
		D ~ Primary movement, g0 ~ 1 sigma ~ 1	5	-150.0	314.02	1.57	0.10
		Null	4	-152.3	315.08	2.63	0.06

I used the model including stands with wet soil moisture regimes to estimate the density and abundance of fishers in the Enterprise study area and to predict the density of fishers over the larger ecologically similar surrounding areas as it had strong support from the data at both spatial scales. The density estimates generated from this model were largely consistent between the two scales with an average of 8.9 ± 2.7 *SE* fishers/1000 km^2 estimated at the core use scale and 8.5 ± 2.7 *SE* fishers/1000 km^2 estimated at the home range scale. These density estimates translated into abundance estimates of 52.8 ± 16.1 *SE* fishers and 50.5 ± 15.8 *SE* fishers at each spatial scale, respectively, for the Enterprise study area. The density of fishers was positively associated with the area of stands with wet soil moisture regimes (Figure 2.6, Table 2.5 - β = 13.7; 95% CI: 4.1 – 23.3). This model predicted a relatively low density (<10 fishers/1000 km^2) over most of the extrapolated area, with a few localized areas of higher densities in the center and along the southern boundary (Figure 2.7).

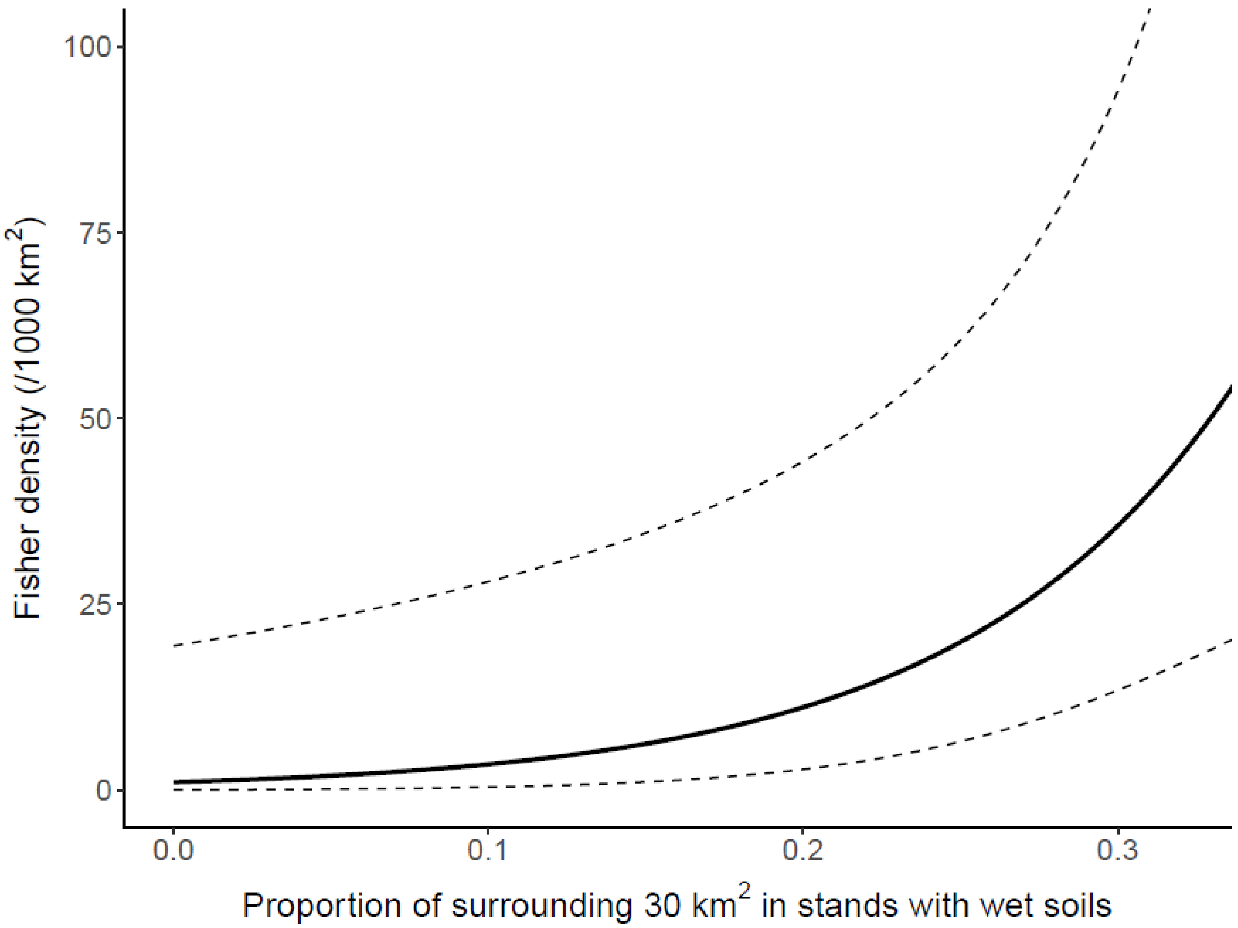

Figure 2.6. Modelled fisher density (pooled across sexes) in stands with wet soil moisture regimes (i.e., wetter than mesic) within Interior Douglas-fir forest types of the Enterprise study site in central interior BC.

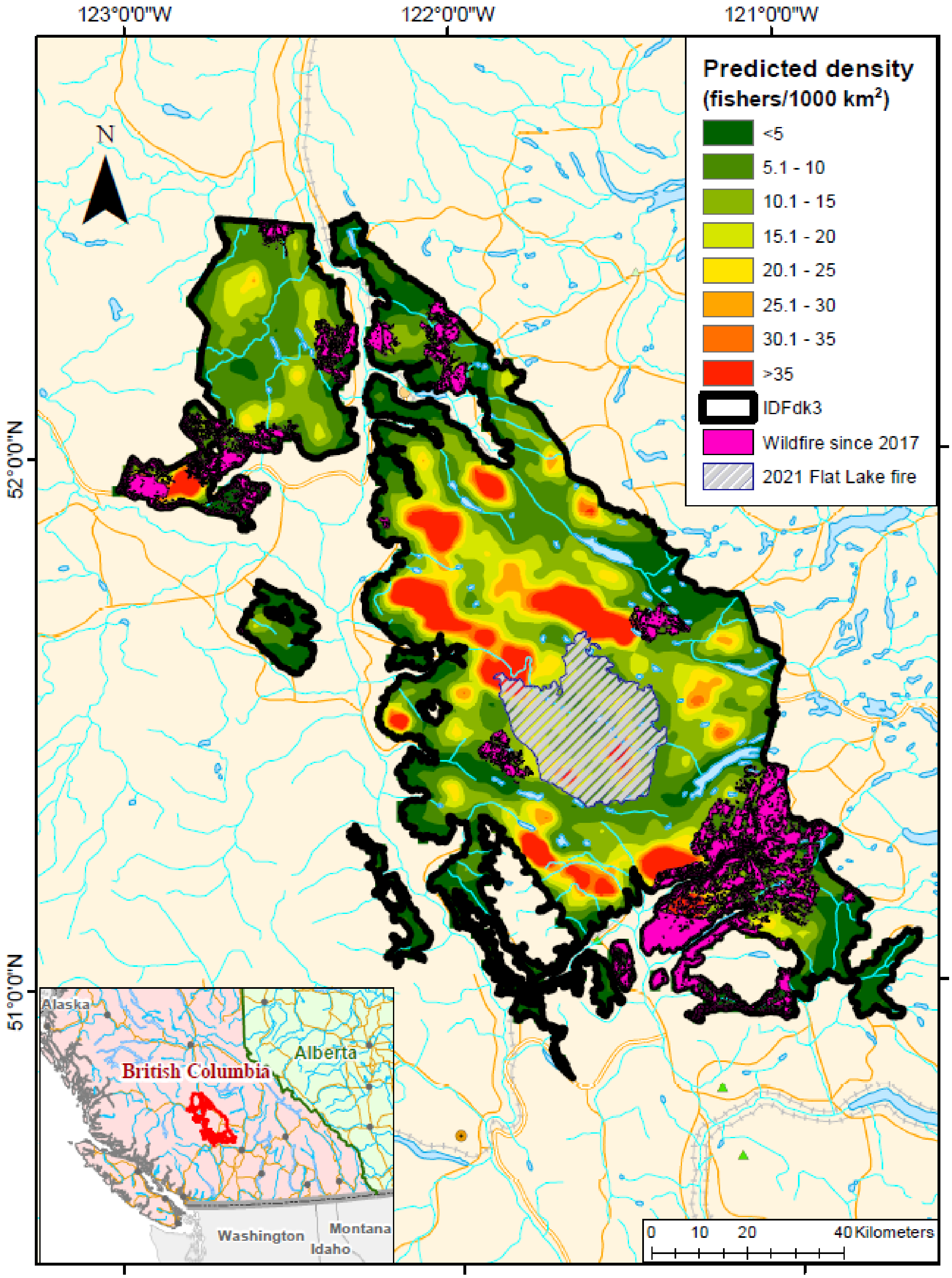

Figure 2.7. Predicted density of fishers (fishers/1000 km^2) in the Interior Douglas-fir forests of central interior BC estimated from spatial capture-recapture analysis of genetically identified fishers sampled during the winter of 2020-21 as predicted by the proportion of surrounding 30 km^2 containing stands with soil moisture regimes wetter than mesic. IDFdk3 = Interior Douglas-fir dry cool 3 BEC subzone variant.

Discussion

These results affirm the notion that the density of fishers varies markedly within the Pacific Northwest, but they also suggest populations in this study region are at levels worthy of concern. For example, the density estimates for the Chilcotin study area were among the highest reported across the province (*cf.* 16.3 fishers/1000km^2; Weir et al. 2011), whereas the estimates from the Enterprise study area were among the lowest (*cf.* 8.8 fishers/1000 km^2 - Weir and Corbould 2006). Yet, these density estimates are substantially lower than that reported from eastern North America (50 to 327 fishers/1000 km^2; Powell and Zielinski 1994) and California (140 fishers/1000 km^2; Matthews et al. 2011). The reason behind these differences in the density of fishers within and beyond BC is not clear, but have been hypothesized to relate to mesocarnivore community dynamics (D. Hodder pers. comm.), differences in prey availability (Weir et al. 2011), and snow conditions (Raine 1983) which differ regionally within BC, as well as across the eastern and southern portions of the species' entire range.

Differences in biological, abiotic, and anthropogenic factors may help explain such notable differences in the density of fishers between the two study areas. Both of these two study areas encompass zones predicted by Lofroth (2004) to support moderate to high capabilities of fishers based on climate, vegetation, and prey communities. Recently, however, forests in both study areas were highly affected by the Mountain Pine Beetle Epidemic which resulted in massive swaths of pine trees being killed and salvage harvested at unprecedented rates (Eng et al. 2005). Interestingly, nearly one-third of the Chilcotin study area was also burned by the Plateau Wildfire in 2017 which at the time was the largest wildfire on record for BC (Province of BC 2017) yet fishers were detected making forays into the more heavily burned areas as well as along the fringes of the fire boundary. I would have expected that having such a large area recently disturbed by wildfire would have negatively affected the density of fishers in the Chilcotin study area, however, the number of animals detected was much higher there than in the less disturbed Enterprise study area.

Perhaps the most direct explanation behind the difference in the density of fishers estimated for the two study areas involves the relative levels of human-caused mortality. The only trapline overlapping the Chilcotin study area that documented fisher harvest during the 2018-19 season reported 3 animals being trapped (Province of British Columbia unpubl. data: BC Wild Fur Harvest Database 2000 - 2019 [accessed 15 July 2022]) but I was unable to confirm whether I had also detected these animals during the survey. At the time of these surveys in BC, trappers were required to report any fisher trapped in an open season to the government within 15 days of the season closing date of February 15 (Province of BC 2020) and aside from the 3 individuals mentioned above, no other fishers were reported from the Chilcotin study area, suggesting overall trapping pressure was very low at the time of our survey. This aligns with the findings of Davis (2009) who stated that fishers in the Chilcotin generally experience low levels of trapping pressure, and is further supported by the low numbers of fishers trapped on traplines overlapping the Chilcotin study area since 2000 (Province of British Columbia unpubl. data: BC Wild Fur Harvest Database 2000 - 2019 [accessed 15 July 2022]). Conversely, in the Enterprise study area I was able to confirm via genetic analysis or timing of capture that 9 unique fisher individuals were killed by trappers during the winter of my survey. Four other fishers were reported as harvested by trappers from within the Enterprise study area, however, I was unable to confirm these animals as unique individuals without hair or tissue samples. All told, this suggests between 30 and 38% of the fishers that occurred in the Enterprise population prior to the trapping season were removed via fur harvest, substantially affecting density and abundance estimates for this study area.

Overall, I found that the density of fishers in the Chilcotin study area was positively associated with area of primary branch resting habitat. In fact, when the proportion of primary branch resting stands in the surrounding 30 km^2 increased from 0 to 0.05, the density of fishers was predicted to double from 16 fishers/1000 km^2 to 32 fishers/1000 km^2. When not actively hunting or traveling, fishers use protected resting sites to conserve energy, avoid predation, thermoregulate, and to safely consume prey (Lofroth et al. 2010, Raley et al. 2012). Primary branch resting habitat in the drier ecological regions of BC consists of older (>83 years), large-diameter spruce stands with relatively high amounts (>40 %) of crown closure (Weir and Almuedo 2010). These results are supported by numerous other studies in BC and western North America that show fishers use large-diameter spruce trees relatively often for resting (Weir and Harestad 2003, Weir et al. 2004, Davis 2009). Davis (2009) found the most frequently used arboreal rest sites in their West Chilcotin study area were in hybrid spruce (*Picea glauca* x *engelmannii*) trees, with fishers primarily resting on platforms created by spruce broom rust (*Chrysomyxa arctostaphyli*), on squirrel nests, or on branches. Spruce stands in the Chilcotin study area typically were concentrated around streams and wetlands (Meidinger and Pojar 1991) that may contain older trees found within this landscape due to historical fire patterns and logging practices that preserved riparian vegetation. The fact that stream density was directly related to fisher density in the Chilcotin also may be a reflection of this past timber harvesting practice. Protecting these older spruce stands from future forest harvest should help ensure fishers have an adequate amount of high-value resting habitat available on the landscape, which will in turn help the species continue to persist at higher densities.

The male-only model for the Chilcotin also predicted that density was most strongly related to primary branch resting habitat, but the top-ranked female-only model indicated the density of female fishers was best predicted by stands with wet soil moisture regimes (i.e., subhygric, hygric, or subhydric soils). Density models including primary denning habitat and spruce- or aspen-leading stands within 100 m of a wetland also were well-supported by our female detection data, and the correlation analysis indicated some weak association between these three variables. Stands with wet soil moisture regimes in the Chilcotin are small and primarily occur at the fringe of wetlands and along stream channels and are differentiated from drier stands due to the presence of hybrid spruce, a more well-developed shrub layer, and a lesser abundance of lichens (Meidinger and Pojar 1991), and these distinctions help explain the correlation between the variables included in the top three female-only density models. The Chilcotin plateau is one of the driest regions where fishers are found in BC, and these locally wetter stand types have been shown to provide higher densities of both denning and resting structures (Davis 2009) while also supporting higher amounts of important prey species (Davis 2003, Davis 2004). Across the range of fishers, deciduous trees provide important denning structures critical to reproduction (Powell 1993, Weir and Harestad 2003, Weir 2003) and in the Chilcotin large diameter trembling aspen are most abundant in these wetter, more productive stands around riparian features (Davis 2009). Spruce and aspen stands are important predictors of fisher denning and resting habitats in BC (Weir and Harestad 2003, Davis 2009, Weir et al. 2012), and my modelling results support these other studies by showing the density of female fishers also is positively associated with these types of habitats. Protecting and reserving these older, large diameter spruce and aspen stands adjacent to riparian features should help ensure there are more of the crucial denning and resting structures that fishers require for both survival and reproduction available on the landscape, thereby increasing the chances the species will continue to persist at higher densities in the Chilcotin region of BC.

These results demonstrate that throughout the drier ecological zones of interior BC, wetter stand types are associated with higher densities of fishers. Similar to the Chilcotin results, my top-ranked density models for the Enterprise study area predicted the density of fishers was strongly related to stands with wet soil moisture regimes. When the proportion of stands with wet soil moisture regimes in the surrounding 30 km^2 area increased from 0.1 to 0.3, the predicted density of fishers increased from 1 fisher/1000 km^2 to 21 fishers/1000 km^2. Although the Enterprise study area is not quite as dry as the Chilcotin, it is also considered one of the drier ecosystems where fishers are found in BC (BC Fisher Habitat Forestry Web Module 2021). Stands with these higher soil moisture levels in the Enterprise study area are generally found near the base of north-facing slopes, and adjacent to streams and wetlands (Meidinger and Pojar 1991). These stands typically have closed forest canopies and primarily consist of a mixture of hybrid white spruce and Douglas-fir, with greater shrub cover than other drier stands (Hope et al. 1991). Although no specific data exists for the Enterprise study area, given their characteristics these stands likely provide higher densities of denning and resting structures and prey similar to their counterparts in other drier ecological zones of interior BC (e.g., the Chilcotin Plateau), making the positive association I found with higher density of fishers consistent with other areas of the province. Protecting these high-value riparian stands from disturbance likely is important to ensure the structures fishers need to fulfill their life requisites remain available in sufficient quantities on the landscape.

While my model selection results generally supported habitat associations suggested by previous research, one interesting departure from this trend was the link between fisher densities and the amount of 'open' areas (e.g., wetlands, recent burns, young cutblocks) in one of my two study areas. Throughout their range, fishers have consistently been tied to areas with overhead cover, being reported to avoid more open habitats (Powell and Zielinski 1994, Weir 2003), Lofroth et al. 2010, Raley et al. 2012). In north-central BC, Weir and Corbould (2010) found a strongly inverse relationship between the probability of a fisher home range being occupied and increasing amounts of 'open' areas within that home range, and an independent study from Idaho found nearly the exact same relationship (Sauder and Rachlow 2014). When I included 'open' areas as a covariate in our density models the relationship between the density of fishers was only slightly positive in the Enterprise study area, but this model did rank within the top four from the candidate set. In the Chilcotin study area the 'open' areas model ranked last and areas that were burned at moderate or higher intensities also did not appear to be strongly related to density, although the exact influence these covariates may be having on the density of fishers is somewhat unclear (95% confidence intervals of beta coefficients spanning zero). The observed lack of response to recently burned areas may be related to how soon after the fire I surveyed (i.e., within one year), and habitat-related effects from this disturbance may be lagging behind and will be more apparent over the longer term in the Chilcotin study area. I hypothesize that in the drier ecological zones of BC the Weir and Corbould (2010) model is primarily confounded because the density of fishers here appears to be strongly related to forested stands that are often found along the fringes of wetlands, and these stands likely also represent the highest value fisher habitat within these regions.

Extrapolating my density models to areas larger than my study areas but similar ecologically was informative but also comes with important limitations, and I caution against using these models to estimate the total abundance of fishers for these larger areas. My trapping arrays only covered ~20% and ~30% of these larger ecologically similar areas and estimates are based on only one winter of survey data. Even within the areas immediately adjacent to my survey grids there is a high amount of ecosystem diversity, and assuming models based on one year of detection data have obvious limitations, especially for the Enterprise study area where a limited number of recaptures resulted in less precise density estimates. Alternatively, these density models built on and supported previous research regarding the habitat associations of fishers in central interior BC and could be used to identify priority areas for habitat protections, areas to direct future population monitoring efforts, and to help identify other areas of the province where fishers may be enduring at higher densities.

Conservation implications

This study provides land managers with important information regarding the distribution and abundance of fishers within the endangered Columbian population in the central interior of British Columbia (BC), Canada, and the methods I used can be applied to other small and isolated populations facing the same cumulative threats of habitat loss, fragmentation, and degradation, and human-caused mortality. Population density estimates reported for these two study areas are among the highest and lowest ever documented in BC, which has important implications for the future conservation and management of this population. The difference in density estimates between the two study areas can largely be attributed to the different levels of trapping mortality in these areas. Based on my findings, the negative effect of trapping on the density of fishers is quite clear and unless trapping regulations and methods are adjusted to substantially reduce this source of mortality it is likely that the density of fishers in the Enterprise study area will continue to decline and ultimately may never recover. In chapter 3, I explore in further detail the effects that trapping mortality may have on the persistence of the Columbian population of fishers. The Chilcotin region of central BC appears to have one of the highest densities of fishers in the province and may be an important source population should translocation programs become necessary; therefore, protective measures aimed at ensuring this population cluster persists into the future should be a priority.

Density estimates from these two study areas were substantially lower than elsewhere in North America and given the current status of the Columbian population the importance of maintaining viable clusters of fishers where they continue to persist cannot be understated. The density of fishers had strong positive associations to similar habitat types in both my study areas which are identifiable using existing spatial data layers and information from the interior of BC. I suggest that land managers make use of this existing information and the management levers at their fingertips to identify and protect these high-value habitats from further industrial disturbances, improving the chances that fishers will continue to persist in this region.

This study has provided a snapshot look at the state of two fisher population clusters in representative ecosystems from central BC, but long term and continuous monitoring of these population clusters and others should be built into the management of the Columbian population of fishers going forward. Without being able to accurately assess population trends across years from rigorous and repeatable monitoring at regular intervals, it will be challenging to determine whether conservation measures being implemented are improving the long-term sustainability of this endangered population.

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CHAPTER 3

TRAPPING MORTALITY ACCELERATES THE DECLINE OF THE FISHER (*PEKANIA PENNANTI*), AN ENDANGERED MESOCARNIVORE IN BRITISH COLUMBIA, CANADA.†

Introduction

Carnivores of all sizes play influential roles in regulating ecosystems through trophic interactions influencing prey abundance, the subsequent effects on vegetation dynamics, and intraguild competition between larger and smaller predators (Prugh et al. 2009, Ritchie et al. 2012). However, mammalian carnivores worldwide face a multitude of threats including habitat loss and fragmentation, predator control programs, and unsustainable levels of hunting and trapping (Prugh et al. 2009, Wolf & Ripple 2017). In North America, several carnivore species have contracted their ranges over the past two centuries following Euro-American colonization of the continent and the ensuing development and demand for resources necessary to support an expanding human population (Laliberte & Ripple 2004). Although some species have re-established in portions of their historical ranges, many have not and continue to decline in abundance (Gittleman et al. 2001, Wolf & Ripple 2017). Understanding the key environmental, demographic, and anthropogenic factors driving the dynamics of endangered carnivore populations is crucial to ensuring the persistence of these important species over time (Yackulic et al. 2011, Wolf & Ripple 2017).

Fishers (*Pekania pennanti*) are a medium-sized member of Family Mustelidae inhabiting the boreal and temperate forests of North America (Powell 1993) and, like many other carnivores, the species has experienced significant range contractions since the late 1800s (Laliberte & Ripple 2004, Lofroth et al. 2010). Following European settlement, fisher range contracted northward primarily due to habitat loss via commercial timber harvest and agricultural land clearing, and overexploitation in the fur trade (Lofroth et al. 2010, Lewis et al. 2012, Lapoint et al. 2015).

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By the early 1900s, large-scale habitat loss, coupled with high pelt prices and minimal fur harvest regulations, led to the decimation of many fisher populations across the southern extent of their range (Lewis et al. 2012). Subsequent protective measures, translocation efforts, and the reversion of agricultural lands back to forest have enabled fishers to recolonize some of their historical range in eastern North America, yet fishers continue to exist west of the Rocky Mountains as smaller and more isolated populations in British Columbia (BC), Canada, and a handful of western states in the United States of America (Lofroth et al. 2010).

As elsewhere within their range, fishers in BC depend on unique structural elements typically found in late-successional forests for both survival and reproduction (Raley et al. 2012, Weir et al. 2012). Structures used for denning and resting are essential habitat features primarily associated with deformed and decaying live trees and include cavities, platforms, and other microstructures (Weir & Harestad 2003, Aubry & Raley 2006, Purcell et al. 2009). Because these structures are rare and can take up to a century to develop, fishers are highly susceptible to habitat alterations that remove these structures at rates faster than forests can redevelop them (Weir et al. 2012). Over the past three decades, the low elevation forests of central BC have undergone widespread habitat change through ongoing forest harvest, large-scale insect infestations, unprecedented wildfire seasons, and the accelerated rates of salvage logging that followed these disturbances (Eng et al. 2005, Province of BC 2017, Province of BC 2018b). Fisher densities in BC are some of the lowest documented from anywhere within their range (e.g., 8.8 fishers/1000 km^2; Weir & Corbould 2006), prompting concerns among wildlife managers, trappers, stakeholders, First Nations, and conservation groups over the decline of these animals and their habitat. Moreover, recent research indicates there are two distinct fisher populations in BC (Boreal population and Columbian population, respectively) separated by a high snowpack zone encompassing the Rocky Mountain divide and the mountainous region south of the Spatsizi Plateau (see Figure 1.2 in Chapter 1). Given that fishers have already been extirpated from most of southern BC, this work suggests that the Columbian fisher population effectively has been isolated from any other population on the continent.

As a result of patterns in habitat loss and the accompanying decline in population size, the status of the Columbian fisher population was revised to endangered by the BC government (Province of BC 2020). This was accompanied by the cessation of commercial fisher trapping within the range of the Columbian fisher population (Province of BC 2021b). However, fishers continue to be taken as bycatch in traps designed and set for other furbearing species (e.g., American marten (*Martes americana*), 52% of annual fisher mortality; Province of BC 2018a). Therefore, the objective of this study was to specifically evaluate the effects this mortality from trapping bycatch may be having on the sustainability of the Columbian fisher population. I used field-collected data on fisher reproduction and survival from central BC along with data from the provincial fur harvest database in population viability models to project the population response to different levels of trapping mortality. I predicted that low reproductive output and high rates of natural female mortality in Columbian fishers would affect this population's ability to endure additional, additive mortality from trapping. The knowledge gained from this study will allow wildlife managers to craft more effective population management measures to help ensure that this population persists over the long term.

Materials & Methods

Study area

The range of fishers (*Pekania pennanti*) within the Columbian population encompasses forested habitats at low to moderate elevations in central BC (Weir 2003). Northern and central portions of this range consist of the Sub-Boreal Spruce biogeoclimatic (BEC) zone (Meidinger et al. 1991), whereas the southerly portion of the range primarily consists of drier ecological zones including the Sub-Boreal Pine-Spruce (Steen & Demarchi 1997), Montane Spruce (Hope et al. 1991a), and Interior Douglas-fir (Hope et al. 1991b) BEC zones.

Population viability analysis software

I built population models using Vortex version 10.5.5.0 (Lacy & Pollak 2020), a software program shown to be suitable for populations with low reproductive rates (Lacy 1993, Kim et al. 2016, Winton et al. 2020). Vortex is an individual-based modelling process that simulates population outcomes by sequentially stepping through a series of events describing the annual cycle of a typical sexually reproducing organism (Lacy et al. 2020). It incorporates demographic, environmental, and genetic stochasticity as it follows the fate of individuals from birth to death based on probabilities and user-defined parameters (Lacy et al. 2020). Model outputs included probability of extinction, mean stochastic population growth rate (r), and mean time to extinction.

Input parameters

I parameterized the initial population model primarily using the survival and reproduction information from Lofroth et al. (2022) who analyzed radio-telemetry and other data from the studies of 60 free-ranging fishers in central BC by Weir (1995), Weir & Corbould (2008), and Davis (2009), along with published literature from a well-studied fisher population in California (Table 3.1; Sweitzer et al. 2015). I ran 1000 simulations of each model to predict population outcomes over a period of 100 years using a one-year time step and did not include density dependence, catastrophes, inbreeding, or dispersal effects. I set the quasi-extinction threshold as the point when only one sex remained and considered this to be the point at which the extirpation of Columbian fishers occurred in the Vortex programming language; hereafter ‘extinction’ refers to reaching this threshold. I did not assign any correlation between reproduction and survival due to a lack of empirical data needed to inform this relationship. The current estimate for the Columbian fisher population is 299 – 517 adult individuals (adult fishers are those animals >2 years old; BC CDC 2020). Therefore, I used an initial population size scaled upwards from the midpoint of this adult population estimate to include animals in all age cohorts for each simulation (i.e., initial population size of 571 individuals). I used a proportional age distribution for the initial population based upon the most recent data available from central interior BC (Weir & Corbould 2008). I modelled three different age cohorts for each sex: Adults aged >2 years old, subadults aged 1 – 2 years old, and kits aged 0 – 1 years old.

Vortex defines mortality rates in the language of matrix life-table analysis as the percentage of animals alive at age x that die before reaching age $x + 1$ (Lacy et al. 2020). Survival and subsequent mortality rates were calculated following methods detailed in Lofroth et al. (in press) using monitoring data collected from the three radio-telemetry studies from central BC. Because these survival calculations included mortalities from trapping, I reran their analyses excluding fishers that died in traps using the *survival* package in the R programming language (R Version 4.0.2; R Core Team 2020). This ensured that only non-trapping sources of mortality

Table 3.1. Input parameters and associated data sources for population viability analysis of the Columbian population of fishers (*Pekania pennanti*) in British Columbia (BC), Canada. Most parameters were derived from three radio-telemetry studies on fishers in central BC and one small, well-studied population in California. Environmental Variation (EV) refers to the annual variation in reproduction and survival due to random changes in the environment and is modelled using the standard deviation for each applicable parameter (Lacy et al. 2020).

Parameter	Value (± EV)	Source	Location
Population parameters			
Initial Population Size	571	BC CDC (2020)	British Columbia, CA
Carrying Capacity	10000		
Reproductive parameters			
Breeding system	Polygynous	Lofroth et al. (2010), Smith et al. (2020)	Western North America
Age at first litter (female)	2	Weir (2003), Lofroth et al. (2010)	British Columbia, CA
Age at sexual maturity (male)	2	Weir (2003), Lofroth et al. (2010)	British Columbia, CA
Maximum age of breeding	8	Lofroth et al. (2010)	Western North America
Maximum number of progeny per brood	4	Lofroth et al. (2010)	Western North America
Mean number of kits per litter	1.7 (± 0.69)	Lofroth et al. (in press)	British Columbia, CA
Sex ratio at birth (in % males)	50	Frost & Krohn (1997), Matthews et al. (2019)	North America
% Females breeding annually	54 (± 41)	Lofroth et al. (in press)	British Columbia, CA
% Males breeding annually	100		
Mortality rates (%)			
Kits (age 0 - 1; female)	43 (± 19)	Sweitzer et al. (2015)	California, USA
Kits (age 0 - 1; male)	43 (± 19)	Sweitzer et al. (2015)	California, USA
Subadult (age 1 - 2; female)	40 (± 22)	Lofroth et al. (in press), this study	British Columbia, CA
Subadult (age 1 - 2; male)	14 (± 13)	Lofroth et al. (2022), this study	British Columbia, CA
Adult (age >2 years; female)	21 (± 9)	Lofroth et al. (2022), this study	British Columbia, CA
Adult (age >2 years; male)	10 (± 10)	Lofroth et al. (2022), this study	British Columbia, CA

were included in the initial population progression since the effect of annual fur harvest mortality was simulated in Vortex in a separate step. I used kit mortality rates from birth to the age of 1 from an untrapped, well-studied population in California, as it was the most proximate published study that included data on fisher kit survival (Sweitzer et al. 2015).

I used a polygynous breeding system for our models as both female and male fishers have been documented breeding with multiple partners in the same year (Smith et al. 2020). I set the age at first breeding to be 2 years old for both males and females. Although female fishers can breed at 1 year of age, because they exhibit delayed implantation they will not give birth to kits until they are at least 2 years old (Lofroth et al. 2010). Male fishers 1-year-old in age can produce sperm, but it is believed their baculum is not developed enough to cause females to ovulate until they are two years old (Douglas & Strickland 1987, Frost et al. 1997). I used a sex ratio of 50:50 at birth (Frost et al. 1997, Matthews et al. 2019). I set the maximum number of litters per year to be one and maximum litter size to be 4 kits per litter (Paragi et al. 1994, Aubry & Raley 2006). Not all female fishers successfully breed every year (Powell 1993, Lofroth et al. 2010), and I used the proportion of females breeding annually and average litter size as calculated from the three radio-telemetry studies from central BC (Lofroth et al. in press). I assumed that 100% of extant males of reproductive age would be in the breeding pool each year.

I incorporated environmental variation (EV), or the annual variation in reproduction and survival due to random changes in the environment, by incorporating the standard deviation or error observed from the empirical data with each demographic parameter in our modelling inputs. For each annual iteration, fluctuations in the annual probabilities of mortality and reproduction were modelled as binomial processes based on the mean and standard deviation specified for each parameter (Lacy et al. 2020).

Model manipulation and sensitivity testing

I initially tested the effect of fur harvest levels on the probability of future extinction using the annual average number of fishers harvested from the Columbian fisher population between 2009 and 2017 (mean = 169 ± 37 fishers, or 29.6% of the initial population; Province of BC 2021a). More recent data were unavailable since there was a time lag between harvest reporting and entry in the BC Wild Fur Harvest database. Although the commercial fisher trapping season was recently closed, killing traps certified for trapping other furbearing species consistently caught a substantial number of fishers every year; therefore, I used provincial fur harvest data to calculate the proportion of fishers caught in different trap types to inform our alternative modelling scenarios (Table 3.2; Province of BC 2018a).

Table 3.2. The proportion of fishers (*Pekania pennanti*) harvested from the Columbian population in British Columbia, Canada by trap class as calculated from the provincial fur harvest database.

Trap Class	Species typically set for:	Overall percentage of fisher harvest
120[a]	American marten *Martes americana*, fisher[b] *Pekania pennanti*	52%
160	American marten, fisher	7%
220	Fisher, lynx *Lynx canadensis*, bobcat *Lynx rufus*,	8%
280	Lynx, bobcat	6%
330	Lynx, bobcat, wolverine *Gulo gulo*	16%
Foothold	Lynx, bobcat, coyote *Canis latrans*, wolf *Canis lupus*	3%
Snare	Lynx, bobcat, coyote, wolf	8%

[a] 120-class killing traps are currently the only trap type that can be modified to specifically exclude fishers.

[b] The commercial fisher trapping season was closed in August of 2021.

Both before and after the trapping season closure, most trappers in BC did not specifically set traps to target fishers but typically set traps for the equally valuable and more abundant American marten (*Martes americana*), as either species could be harvested using the same trap set. As such, 120-class traps (e.g., Belisle SUPER X 120, LDL B120 Magnum), which are primarily used, and certified, for trapping American martens and occasionally American mink (*Neovison vison*), accounted for a substantial proportion of fisher harvest during each trapping season (52% of fisher harvest; Province of BC 2018a). Fishers also were killed in slightly larger traps set specifically for them, but which also captured American martens (e.g., 160- and 220-class traps; 15% of fisher harvest), as well as 280- and 330-class traps set primarily for Canada lynx (*Lynx canadensis*), bobcats (*Lynx rufus*), and wolverines (*Gulo gulo*)(22% of fisher harvest). Leghold traps and snares commonly used for trapping Canada lynx, bobcats, coyotes (*Canis latrans*), and grey wolves (*Canis lupus*) accounted for 11% of fisher harvest each year.

I simulated the effects of varying levels of trapping mortality on population viability based upon the mortality attributed to these different classes of traps. I considered 4 different scenarios and modelled the effect of mortality under these scenarios on population persistence (Table 3.3). In Trapping Scenario 1, the harvest of fishers in any trap type continued unabated which, because fishers are killed in all trap types certified and set for other furbearing species in

BC, I expected to remain very similar to the current annual average harvest rate of 169 fishers per year (29.6% of the initial population). In trapping Scenario 2, I considered the possibility that mortality from 120-class traps could be eliminated since this is currently the only trap class that can be modified to specifically exclude fishers (I-J Hansen pers. comm.) but kills from other traps continued. Trapping Scenario 2 simulated the harvest of 81 fishers (14.2% of the initial population) from the population annually and was equal to the annual average number of fishers trapped in non-120-class traps. In Trapping Scenario 3, I considered the possibility that mortality from traps previously certified for use on fishers could be removed from the harvest but kills from other traps continued. Trapping Scenario 3 simulated the harvest of 56 fishers (9.8% of the initial population) which is the annual average number of fishers caught in traps not certified for use on fishers (i.e., 280- and 330-class kill traps, foothold, and snare traps). I assumed the number of fishers harvested each year under these scenarios would decline in concert with overall population declines, therefore I applied a constant trapping mortality rate using the percentage of the initial population harvested annually for each of the three trapping scenarios (e.g., 29.6% of the population was harvested annually for Trapping Scenario 1). The percentage of animals harvested from each age-sex class was based on the proportional distribution of carcasses submitted by trappers across BC (Weir 2003). I also considered a No Trapping Scenario (Scenario 0) where no fishers were removed from the population through fur harvest.

I used sensitivity testing to assess the relative influence of each individual input parameter on the predicted model outcomes. I simulated the effects of altering model parameters one at a time while holding all other values constant to better understand the relative sensitivity of each input parameter on the projected stochastic population growth rate. Given the uncertainty associated with the current estimate for the Columbian fisher population, I varied the initial population size in Scenario 0 from 300 to 2000 fishers to evaluate how this factor would change the predicted fate of the population. The age structures of mustelid populations are thought to be inherently unstable (Powell 1994), therefore I used different starting age distributions from published sources (Weir 1995, Weir & Corbould 2008, Buskirk et al. 2012, Lacy & Pollak 2020) to determine if this would change the projected outcomes. Following similar methodologies used in other studies on fisher demographics in the western United States (Lewis et al. 2012, Sweitzer et al. 2015), I then manipulated additional model parameters by ± 10% around initial values to evaluate changes to stochastic population growth rate. These parameters included male and female mortality rates for each age-sex class, litter size, and the proportion of females breeding each year, as well as each parameter's associated value of environmental variation.

Table 3.3. Fur harvest scenarios considered in the projection of population outcomes for fishers (*Pekania pennanti*) in the Columbian population, British Columbia, Canada. The percentage of the total population removed under each scenario was based upon the annual average reported harvest of fishers in the Columbian population between 2009 and 2017 (shown in brackets) and a projected initial population size of 571 fishers. We applied a constant trapping mortality rate using this percentage of the initial population harvested annually to account for the decline in harvest we expected to see each year as the overall population declines (i.e., as the population declines over time so does the number of fishers harvested).

Trapping Scenario	**Trap types that harvest fishers**	**Typical target species**	**Percentage of population harvested**[a]
0: No fishers harvested in any trap type	None	N/A	0% (0 fishers)
1: Fishers harvested in all trap types (status quo)	All trap types	See list below	29.6% (169 fishers)
2: Fishers harvested in non-marten sets only	160-, 220-, 280-, 330- class killing, foothold, and snare traps	Fisher *Pekania pennanti*, lynx *Lynx canadensis*, bobcat *Lynx rufus*, wolverine *Gulo Gulo*, coyote *Canis latrans*, wolf *Canis lupus*	14.2% (81 fishers)
3: Fishers harvested in non-fisher-certified sets only	280-, 330- class killing, foothold, and snare traps	lynx *Lynx canadensis*, bobcat *Lynx rufus*, wolverine *Gulo Gulo*, coyote *Canis latrans*, wolf *Canis lupus*	9.8% (56 fishers)

[a] Based on an initial population size of 571 fishers.

[b] 120-class killing traps are currently the only trap type that can be modified to specifically exclude fishers.

Results

No Trapping Scenario (Scenario 0 – no annual fur harvest mortality)

Scenario 0, which used known rates of natural mortality and reproduction from free-ranging fishers and included no mortality from fur harvest, predicted that the Columbian fisher population in the central interior of BC would become extirpated in 36.9 years (SE = 0.43), on average (Table 3.4). The results of this modelling scenario showed a steadily declining

Columbian fisher population with a mean population growth rate of -0.1261 (SE = 0.0015) across all years, and a probability of extinction of 100% in 100 years.

Table 3.4. Stochastic growth rate, probability of extinction, and mean time to extinction (± standard error) for the Columbian population of fishers (*Pekania pennanti*) in British Columbia (BC), Canada under different levels of trapping mortality. Simulations were run 1000 times over a period of 100 years using Vortex software (version 10.5.5.0). Both males and females were available for trapping in these scenarios with an age distribution used from fisher carcass data from BC (Weir 2003). Extinction was considered when only one sex remained in the population.

Trapping Scenario	**Percentage of population trapped annually (%)**	**Rationale**	**Stochastic population growth rate (r)**	**Probability of extinction (%)**	**Mean time to extinction (yrs)**
0	0	No fishers trapped in any trap types	-0.13 ± 0.0015	100	36.9 ± 0.43
1	29.6	Average number of fishers trapped annually between 2009-2017	-0.27 ± 0.0023	100	10.7 ± 0.03
2	14.2	Average number of fishers trapped annually between 2009-2017 in non-120 class traps[a]	-0.23 ± 0.0023	100	16.5 ± 0.12
3	9.8	Average number of fishers trapped annually between 2009-2017 in non-fisher-certified traps	-0.21 ± 0.0021	100	20.6 ± 0.19

[a] 120 class traps typically used for American Marten *Martes americana* can be modified to specifically exclude fishers - other trap classes currently cannot.

Alternative scenarios

I modelled a suite of alternative population scenarios based on status quo annual fur harvest rates to evaluate and predict the probability of extinction for the Columbian fisher population. When fur harvest removed any animals from the population, the probability of extinction was 100% over 100 years and extinction predicted within 21 years for all three scenarios that we considered (Figure 3.1; Table 3.4). Generally, the predicted population growth rate for the Columbian population decreased substantially with increasing levels of fur harvest.

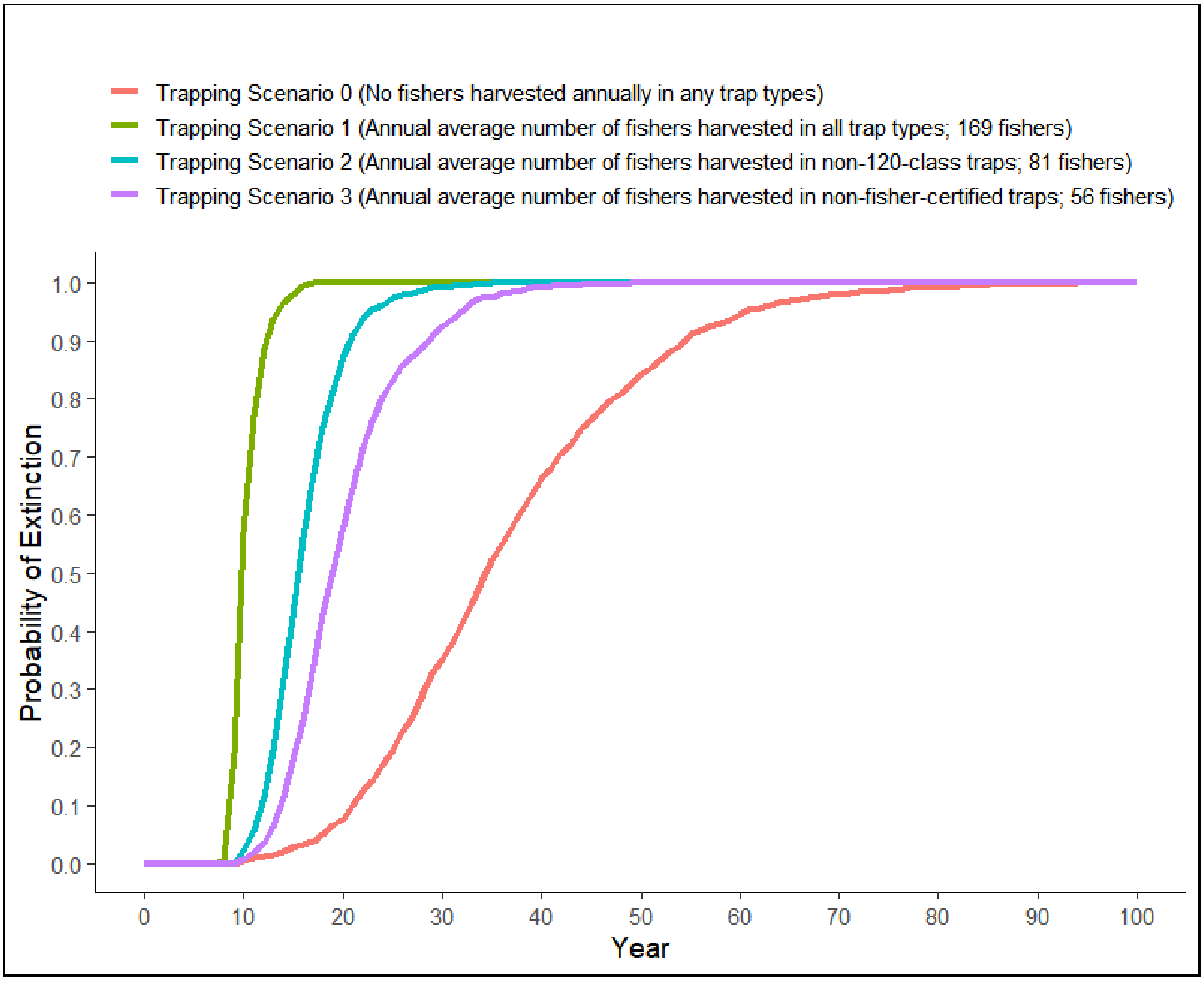

Figure 3.1. Influence of alternative levels of mortality from trapping bycatch on the probability of extinction over 100 years for the Columbian population of fishers (*Pekania pennanti*) in British Columbia, Canada, using parameterizations listed in Table 3.1 and Trapping Scenarios in Table 3.3. Modelled using Vortex software (version 10.5.5.0).

Specifically, under the status quo scenario (Scenario 1), which assumed the average annual fur harvest rate of 29.6% of the population continued unabated, the population was predicted to become extinct within 10.7 years (SE = 0.03). Trapping Scenario 2, which assumed that fishers were excluded and not killed in 120-class American marten (*Martes americana*) traps but continued to be caught in all other trap types (48% of annual trapping mortality; Province of BC 2018a), predicted a population growth rate of -0.2272 (SE = 0.0023) and a mean time to extinction of 16.5 years (SE = 0.12). Trapping Scenario 3, which assumed that mortality from traps certified for use on fishers could be removed from the harvest but kills from other traps continued, predicted a population growth rate of -0.2050 (SE = 0.0021) and a mean time to extinction of 20.6 years (SE = 0.19).

Sensitivity testing

I used sensitivity testing to explore the effects that altering other model parameters had on the stochastic population growth rate. Age structures of mustelid populations are thought to be inherently unstable (Powell 1994), however, changing the age distribution used for the initial population had negligible effects on probability of extinction and mean time to extinction, and a moderate effect on the population growth rate (range: -0.1513 to -0.1261; Table 3.5). Altering the initial population size from 300 to 2000 fishers had very little effect on the predicted population growth rate (range: -0.1336 to -0.1189) or probability of extinction (>99.8% in all scenarios), however it did result in differences in mean time to extinction (range: 34.2 – 44.3 years; Table 3.6). Population models were most sensitive to adult female mortality followed by the percentage of females breeding annually, subadult female mortality, and female kit mortality, in order of decreasing significance (Figure 3.2). Varying male mortality rates and changing the environmental variation associated with each parameter had little effect on the stochastic population growth rate. Therefore, although there was some uncertainty associated with the values used for the environmental variation in each demographic rate, refining these values would not change the conclusions drawn about the viability of the population.

To further evaluate the effects that varying female mortality would have on the predicted probability of extinction for the Columbian fisher population, I decreased mortality rates for female fishers in increments of 2% (i.e., mortality rate decreased from 20% to 18%, 18% to 16%, etc.) from the initial values used in Scenario 0, with no additional mortality from trapping (Figure 3.3). Decreasing female mortality rates by 10% resulted in less than half of the 1000 populations simulated going extinct within 100 years. Decreasing female mortality rates by 12% resulted in a slightly positive stochastic population growth rate and a probability of extinction of less than 25% within 100 years. When female mortality rates were decreased by 10% and the percentage of females successfully breeding was increased by 10%, the population growth rate became positive at 0.0227 (SE = 0.0008) with a probability of extinction less than 6% within 100 years.

Table 3.5. Effect of using different age structures on the predicted stochastic population growth rate, probability of extinction, and mean time to extinction (± standard error) for the Columbian population of fishers (*Pekania pennanti*) in British Columbia, Canada, over 100 years. Extinction was considered when only one sex remained in the population. Modelled using Vortex software (version 10.5.5.0).

Age Distribution	**Source**	**Stochastic population growth rate (r)**	**Probability of extinction (%)**	**Mean time to extinction (yrs)**
Proportional based on radiotelemetry research in BC	Weir & Corbould (2008)	-0.13 ± 0.0015	100	36.9 ± 0.43
Proportional based on fisher carcass data from 1988-1993 in BC	Weir (1995)	-0.13 ± 0.0015	100	36.6 ± 0.40
Proportional based on population model	Buskirk et al. (2012)	-0.14 ± 0.0015	100	34.4 ± 0.39
Stable age distribution (as calculated by Vortex 10.5.0.0)	Lacy & Pollak (2020)	-0.15 ± 0.0016	100	31.5 ± 0.39

Table 3.6. Stochastic population growth rate, probability of extinction, and mean time to extinction (± standard error) for the Columbian population of fishers (*Pekania pennanti*) in British Columbia, Canada over 100 years with different initial population sizes. Extinction was considered when only one sex remained in the population. Modelled using Vortex software (version 10.5.5.0).

Initial population size	**Stochastic population growth rate (r)**	**Probability of extinction (%)**	**Mean time to extinction (yrs)**
300	-0.12 ± 0.0015	100	34.2 ± 0.41
400	-0.12 ± 0.0014	100	35.3 ± 0.39
571[a]	-0.13 ± 0.0015	100	36.9 ± 0.43
700	-0.13 ± 0.0015	99.9	38.4 ± 0.42
1000	-0.13 ± 0.0014	99.9	40.1 ± 0.43
2000	-0.13 ± 0.0013	99.8	44.3 ± 0.45

[a] Approximate midpoint of the current estimate for the Columbian fisher population, British Columbia, Canada. This was the value used for initial population size in all Trapping Scenarios.

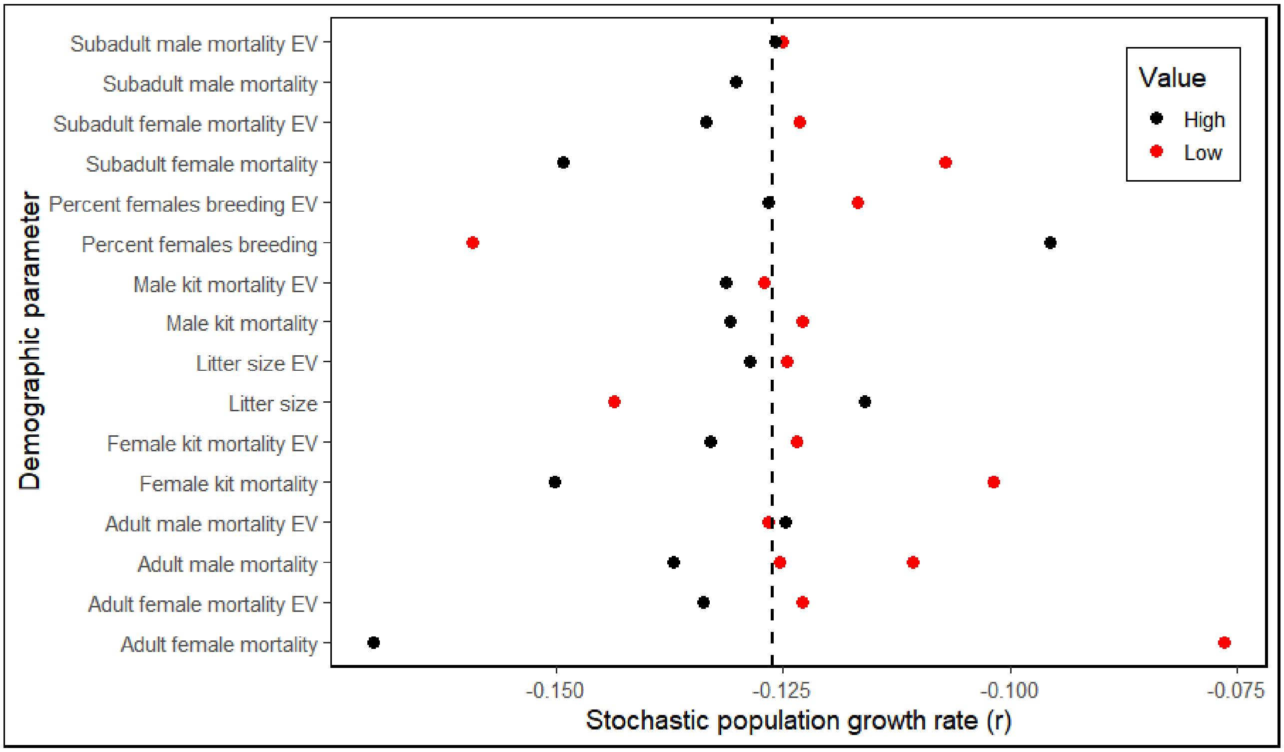

Figure 3.2. The effects of varying input parameters for breeding and mortality rates and their associated values of Environmental Variation (EV) by ±10% on the stochastic population growth rate for the Columbian population of fishers (*Pekania pennanti*) in British Columbia, Canada, under Scenario 0 – no trapping mortality. The black dashed line is set at the stochastic population growth rate value for Scenario 0. Red indicates the parameter value was decreased by 10% and black indicates the parameter value was increased by 10%. Each parameter was varied while holding all other parameters constant. Modelled using Vortex software (version 10.5.5.0).

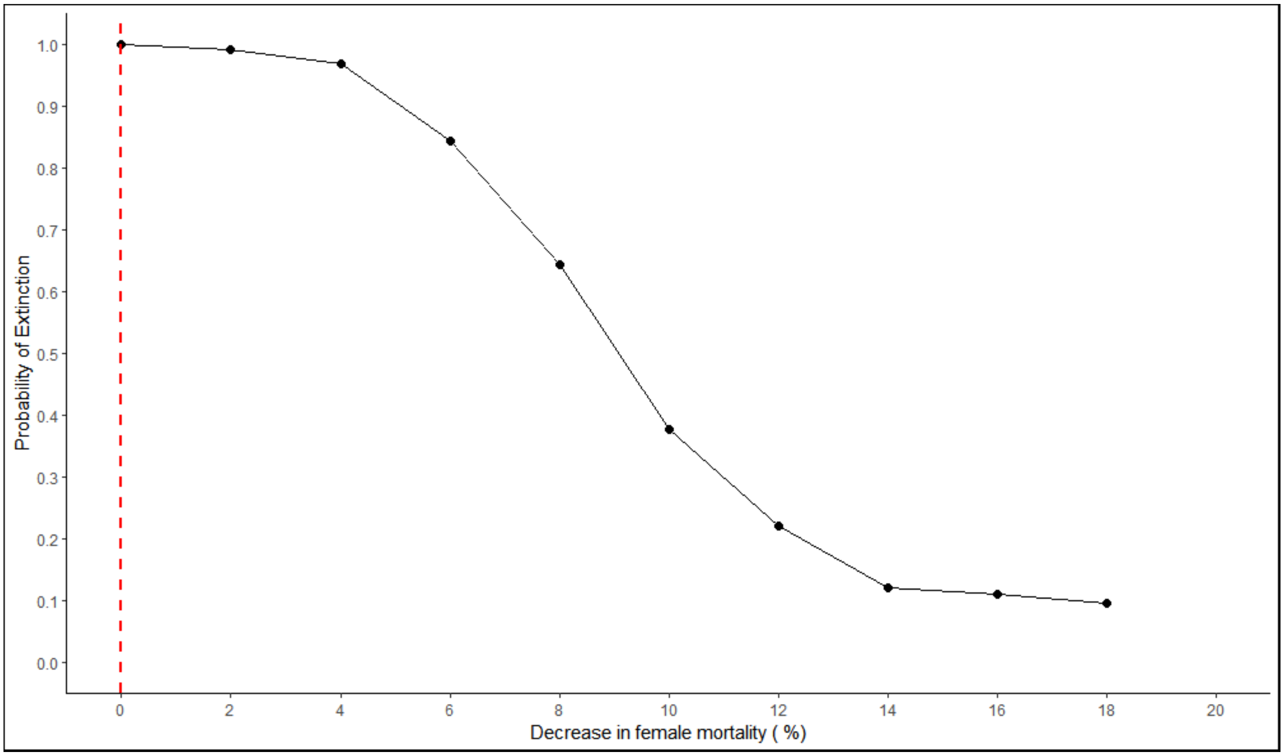

Figure 3.3. The effects of decreasing natural female mortality in increments of 2% on the probability of extinction over 100 years for the Columbian population of fishers (*Pekania pennanti*) in British Columbia, Canada. The red dashed line at 0 represents natural female mortality rates used for initial population projections, detailed in Table 1, that were derived from three radio-telemetry studies on fishers in the central interior of BC between 1995 – 2009. These initial rates were subsequently decreased in increments of 2% to evaluate the effect on probability of extinction, in the absence of any mortality from trapping. Modelled using Vortex software (version 10.5.5.0).

Discussion

Population simulations indicated that, under current reproductive and natural mortality rates, the Columbian population of fishers appears unable to sustain any additional mortality in the form of fur harvest. Because fishers are easily trapped (Powell 1979, Powell 1994), small populations can be put at further risk by the trapping of other furbearers in these same areas (Douglas & Strickland 1987, Powell & Zielinski 1994). The commercial trapping season was discontinued in 2021 within the range of the Columbian population (Province of BC 2021b), however fishers in central BC continue to be regularly killed through the legal trapping of other furbearing species every year (Province of BC 2021a). During the two years that the fisher trapping season was previously closed in BC in the early 1990’s, the number of fishers incidentally caught exceeded the number legally caught the preceding year (Powell & Zielinski 1994), which suggests the closure of the fisher trapping season alone will have minimal benefit for this population. Unless trapping regulations are changed in central BC to substantially reduce the number of fishers trapped as bycatch each year, my modelling indicates the Columbian population will become extirpated from the region in just over a decade.

Closing the commercial trapping season or eliminating the use of body-gripping traps (i.e., killing traps) for other terrestrial furbearers within Columbian fisher range are likely the most effective ways to eliminate the significant threat that bycatch mortality represents to the persistence of this fisher population. In central BC most fishers are caught in 120-class killing traps designed for American martens (*Martes americana*)(52%; Province of BC 2018a). In 2019 - 2020, the BC Fisher Habitat Working Group initiated a pilot project to design and construct a "fisher exclusion box" which can be used to modify 120-class traps to specifically exclude fishers (I-J Hansen pers. comm.). Legally requiring these "fisher exclusion boxes" to be used by trappers would eliminate most of the risk of fishers being caught in this class of traps and decrease considerably the fisher bycatch in central BC while still allowing some level of marten trapping. However, steadily increasing lumber prices makes the cost of producing these "exclusion boxes" not insignificant and is currently limiting the voluntary adoption of their use by trappers in the province (I-J Hansen pers. comm.). Trap classes designed for other larger furbearing animals, leghold traps, and snares also account for a substantial proportion of the fisher harvest in central BC (48%; Province of BC 2018a), and at this time these other trap classes and types do not have modifications designed to exclude fishers. When I simulated a scenario where only those fishers caught in non-120 class traps were harvested from the population, the mean time to extinction for the Columbian population increased from 10.7 (status quo) to 16.5 years. These results further demonstrate that without eliminating the bycatch of fishers in traps set for other furbearers through increased trapping regulations, it is highly unlikely that the Columbian fisher population will persist in central BC.

Even when fur harvest mortality was removed, projections still indicated a steadily declining Columbian fisher population. While mortality from trapping bycatch may be the proximate threat towards the fisher population in the interior of BC, habitat loss, fragmentation, and degradation are reported to be the primary drivers behind the long-term population declines being seen in the province (Weir 2003, Lofroth et al. 2010, BC CDC 2020).

Throughout their range fishers depend on structures associated with late-successional forests for both survival and reproduction (Lofroth et al. 2010), and reproductive output has been linked to habitat quality (Raley et al. 2012). The dens where fishers birth and raise their kits in BC are exclusively found in the cavities of large diameter trees (Lofroth et al. 2010, Weir et al. 2012), which are atypical and uncommon on the landscape. Fishers use protected resting sites when not actively hunting or traveling to conserve energy, avoid predation, for thermoregulatory purposes, and to consume prey safely (Lofroth et al. 2010, Raley et al. 2012). In BC, fishers rest in cavities in large diameter trees, on platforms formed in spruce and subalpine fir trees from abnormal growths caused by spruce broom rust (*Chrysomyxa arctostaphyli*) or fir broom rust (*Melampsorella caryophyllacearum*), and on large branches (Weir 2003, Davis 2009). Forest management has the greatest potential to negatively impact fisher habitat in BC due to the prevalence of clear-cut harvesting with short cutting rotations which removes these important structures at rates faster than they develop under current forest harvesting and secondary successional timelines (Weir & Corbould 2010, BC CDC 2020). Limiting further disturbance to high-value fisher denning and resting habitats in central interior BC will also likely be necessary to promote the recovery of the species in the region.

My population models were highly sensitive to the loss of reproductive-aged females and the percentage of females successfully producing offspring each year. These results align with other studies from a small, untrapped, and isolated fisher population in the southern Sierra Nevada mountains of California (Lamberson 2000, Sweitzer et al. 2015) as well as published literature on other mammalian carnivores (Hebblewhite et al. 2003, Mills et al. 2018, Hooker et al. 2020), which all found female demographic response important to their study populations. Simulations also were sensitive to subadult female and female kit mortality, indicative of an overall vulnerability of the population to all forms of female mortality. Furthermore, these modelling results demonstrate that in the absence of any additional mortality from trapping, moderate decreases of 10% in female mortality and increases of 10% in breeding success would have considerably positive impacts on the persistence of the Columbian fisher population. Taken together, the observations that populations are sensitive to female survival and reproductive success and that the loss of fisher denning and resting habitats is likely detrimental to both are concerning, and these results suggests that without identifying and protecting areas where fishers are actively reproducing it will be challenging to recover the Columbian fisher population to sustainable levels over the longer term.

As with all modelling approaches, there are assumptions and limitations to these predictions. I assumed that reproductive and mortality rates remained constant through time, which may not be the case. I did not include dispersal effects in our models and our analysis was aspatial and applied to the entirety of Columbian fisher range. Suitable fisher habitat is highly fragmented in BC and there are likely negative effects on the population due to increased dispersal distances for both kits leaving their mothers to establish their own territories, and males and females seeking breeding opportunities. I did not include any effects due to inbreeding depression, which would also likely exert detrimental effects on the population if it continued to decline. Catastrophes such as the large-scale wildfires that are becoming more common were also not included, and I expect the destruction of such extensive areas of the forested habitats that fishers require may have further adverse effects on the population. The potential effects that prey availability may have on fisher populations also went unmodelled. Importantly, it is likely the factors not included in the modelling (e.g., dispersal effects) would have additional impacts and not improve the outlook for this fisher population. I provide a more detailed discussion on the limitations of this model in Chapter 4.

Conclusions

Like many other low-density and wide-ranging carnivores, fishers (*Pekania pennanti*) in the Columbian population in central BC face numerous threats, most notably habitat loss and fragmentation, and human-caused mortality. As a result, this population is declining and without intervention by management authorities is at risk of extirpation from the region in the very near future. My modelling strongly suggests that the annual level of mortality from trapping within the Columbian population of fishers is currently unsustainable and accelerating the declines being seen in this population. My analysis suggests that in addition to the recent closure of the trapping season for fishers in central BC, it is likely necessary to modify the trapping seasons and regulations (including restricting the use of kill traps) for other furbearers found within the range of the Columbian population to reduce mortality from bycatch to give fishers in this region a chance to persist long enough to allow for active habitat recruitment. Additionally, identifying areas where fishers are currently successfully breeding and protecting these habitats from further disturbances will be needed to help increase survival and reproductive rates to levels high enough to reverse population declines over the longer term. Future research should focus on investigating other management tools aside from regulating trapping and habitat protection that also may be effective in helping to increase survival rates and reproductive outputs within the Columbian population of fishers in BC.

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CHAPTER 4

CONCLUSION

Summary of Thesis

The overall objective of this thesis was to investigate the environmental, demographic, and anthropogenic factors influencing the decline of an isolated and endangered carnivore population to provide science-based information for conservation and management. More specifically, I intended to fill the considerable knowledge gaps regarding the current distribution and abundance of the Columbian population of fishers in central BC, and assess the effect that continuing trapping mortality will have on its sustainability over the long term. To achieve this, I (1) used information from non-invasive DNA-based surveys at two study sites within the range of the Columbian population of fishers to estimate density and abundance where current data are lacking, (2) investigated which anthropogenic and environmental factors best predicted fisher density across these areas to gain insight into current habitat associations within these ecosystems, and (3) conducted a Population Viability Analysis (PVA) to assess the impacts that trapping mortality would have on the persistence of the Columbian population of fishers under alternative management scenarios.

The primary findings from my thesis were:

- The density of fishers in the Chilcotin and the Enterprise study areas differed markedly. The former was among the highest reported for BC at ~21 fishers/1000 km^2, whereas the estimate for the latter was among the lowest at ~9 fishers/1000 km^2. Both of these density estimates are substantially lower than those reported for fishers in other areas of North America.
- Trappers harvested 13 fishers from the Enterprise study area during the time of the survey. Although I was unable to confirm the individual identity of 4 of these animals, this catch suggested 30-38% of the fishers present within the Enterprise study area were removed via fur harvest having a significant impact on the density and abundance estimates of the animals in this study area.

- Covariates that best predicted the density of fishers were similar between the two study areas. Older, large-diameter spruce stands with high amounts of crown closure best predicted the density of fishers in the Chilcotin study area, and stands with wet soil moisture regimes best predicted the density of fishers in the Enterprise study area. Both stand types are typically found adjacent to streams and wetlands, and these stands are known to contain higher densities of the critical habitat features fishers require for denning and resting, as well as higher numbers of available prey species.
- Population modelling suggested that if current fur harvest mortality rates continue the Columbian population of fishers may become extirpated within 11 years. When even greatly reduced levels of fur harvest mortality were included in the models, extirpation of this population still was predicted within 21 years.
- When fur harvest mortality was removed from my modelling, the Columbian population appeared unlikely to persist beyond 37 years. This suggests that other factors such as habitat loss, fragmentation, and degradation also are contributing to the negative population growth rate predicted by the models.
- Population growth rates were most sensitive to the loss of reproductive-aged females and the percentage of females successfully producing offspring each year. In the absence of additional fur harvest mortality, modest increases of 10% in both female survival and breeding success resulted in a positive population growth rate, suggesting that identifying and protecting critical denning and resting habitat also may be worth considering to help reverse population declines.

Overall, these findings demonstrate the density of fishers in the central interior of BC is highly variable among regions but continues to be among the lowest from anywhere within the species' range. Given the current status of the Columbian population, the importance of identifying and protecting existing clusters of fishers that are actively reproducing should not be understated. These results also support the idea that fishers in central BC are associated with older spruce stands and increasing amounts of overhead cover, and riparian forests adjacent to streams and wetlands. Protecting these stand types from further disturbance will likely be important to ensure the structures fishers require to fulfill their life requisites remain available in sufficient numbers on the landscape. As with other small and isolated populations, my analysis showed that the Columbian population of fishers is highly susceptible to the removal of even a few individuals, particularly reproductive-aged females. Not only did fur harvest mortality result in substantial differences in density estimates between my two study areas, but population modelling showed that if fur harvest rates of fishers continue at current levels this population will likely disappear from the region in just over a decade. This study was based on the most current and accurate population data for fishers in central BC, and as such these results have significant implications for the conservation and persistence of the species in the region.

Management Recommendations

My study provides important information regarding the conservation of fishers in central interior BC, and the methods I used can be applied to other small and isolated populations facing the same cumulative threats of habitat loss, fragmentation, and degradation, and human-caused mortality. During its most recent status assessment, the BC Conservation Data Centre identified habitat loss due to forest harvesting as the primary threat to the Columbian population of fishers in central BC (BC CDC 2020). Secondary threats include loss or degradation of habitat from agriculture, transportation, and wildfire, and trapping mortality where habitats have been compromised (BC CDC 2020). Although the International Union for the Conservation of Nature (IUCN) lists the status of fishers as “Least Concern” globally (Helgen and Reid 2018), this organization still recommends as the primary conservation measure for the species ‘to prevent excessive harvest’ (Helgen and Reid 2018). Here I present management recommendations based on the findings of this thesis, along with historical information, to provide decision-makers with ways to reduce or eliminate these threats to the Columbian population of fishers in central BC to help prevent extirpation of the species from the region in the very near future.

Population Management

Until very recently, the monitoring of fisher populations in British Columbia has been sporadic at best. Fur harvest records have been used the longest as a surrogate for rigorous population monitoring, however, trends in harvest rarely reflect actual population trends and estimates of population size because of differences in trapper effort and pelt prices (Banci 1989, Banci and Proulx 1999). Population estimates for fishers in BC have largely been based on two radio-telemetry studies, one in the Williston region of north-central BC (8.8 fishers/1000 km^2; Weir and Corbould 2006) and one northeast of the Rocky Mountains (16.3 fishers/1000 km^2; Weir et al. 2011). Estimates of fisher abundance for the province were derived using a combination of these estimates, along with habitat capability mapping (Lofroth 2004). This study provides much needed current density and abundance estimates for fishers in two distinct ecosystems in central BC, however, the lack of long-term monitoring data over the entire range of the Columbian population makes it challenging to accurately determine the true magnitude of fisher population declines in central BC. To address this issue going forward, I recommend:

- Updating the current population estimate for the Columbian population using the density and abundance estimates from this study.
- Repeating the survey work for both the Chilcotin and Enterprise study areas within the next 5 years to begin to build the dataset necessary to accurately determine trends in fisher densities in these two regions.
- Developing rigorous and repeatable monitoring plans for fishers throughout the province and implementing them at regular intervals (e.g., every 5 years) across different regions. These may include DNA-based surveys using methods similar to this study, camera trapping surveys, or further radio-telemetry work to help focus management efforts on those population clusters in most need of immediate action.

Habitat Management

Fishers have been described as one of the most habitat-specialized mammals in North America (Buskirk and Powell 1994), relying on unique and uncommon structures in forests for both survival and reproduction (Raley et al. 2012). The processes that form these structures can take centuries to develop them, therefore fishers are highly susceptible to forest disturbances such as rotational logging and wildfire (Lofroth et al. 2010, BC CDC 2020). The results of this study showed that the density of fishers was most strongly related to stand types containing higher densities of the critical denning and resting structures required to fulfill their life requisites. These stand types have been spatially mapped across the province and are freely available to any individual via the BC Data Catalogue (Province of BC 2022). The BC Fisher Habitat Forestry Web Module also provides a free GIS-based tool that allows users to input proposed forest harvest unit shapes which it uses to calculates the density of denning and resting stands that remain in the surrounding landscape. Based on this information, users are given the number of denning and resting features to retain within their cut block, as well as the number of coarse woody debris piles and logs to create. When the surrounding landscape is depauperate in these crucial denning and resting stands, the user is given a warning to avoid the harvest of certain stands. Currently this guidance is only legally required to be used by forest licensees in the Cariboo Region, but with flexibility to be implemented such that a harvest warning does not preclude that polygon from still being harvested. In other words, currently forest licensees and others proposing timber harvest still can harvest stands with high numbers of denning and resting structures regardless of how many might remain in the surrounding areas. To ensure that these high-value denning and resting stands remain available in sufficient quantities on the landscape, I recommend:

- Forest licensees and others proposing timber harvesting activities in all regions of the province where fishers are found should be legally required to use the GIS-based tool prior to harvest and adhere to the guidelines for habitat retention as specified.
- When proposed cut blocks contain areas with a harvest warning these polygons should be ground-truthed to confirm they contain denning and resting structures that fishers require for survival and reproduction. If these polygons have the appropriate characteristics, they should be removed from the cut block and reserved from harvest until the surrounding forests in the landscape can provide the number of denning and resting stands necessary for a fisher to occupy that area.

Harvest Management

Furbearing carnivores throughout Canada and the United States, including British Columbia, continue to be killed every year through targeted and/or incidental trapping. While some species appear to be resilient to higher levels of fur harvest mortality (e.g., coyotes, red fox; Banci and Proulx 1999), others are not and the effect of trapping on these species has the potential to cause population declines and extirpations in a short period of time (Harrington et al. 2017). In his extensive review of mammal trapping, Proulx (2021) outlines 3 conditions that should always be met to allow fur trapping to occur: 1) when it employs species-specific trapping systems, 2) when it will not impact species at risk, and 3) when it uses only trapping systems that meet the highest standards of animal welfare. In the case of fishers and other furbearing species in BC, none of these three conditions are met. Traps certified for use in this province routinely catch multiple different species (see Chapter 3), and not one trapping system is truly species-specific. Given that there are multiple species at risk in BC that are also furbearers (e.g., wolverine, fisher), current circumstances in this province simply do not justify fur trapping until systems can be improved to specifically target individual species. California has completely banned commercial and recreational fur trapping for animal pelts aside from rodents (Newsome 2019), and Washington no longer allows the use of foot-hold traps, body-gripping traps (including all conibear types) and snares, except by special permit (Washington Department of Fish and Game 2022). It would not be unreasonable for the government of BC to implement similar restrictions until trapping systems can be employed that specifically exclude those species with low resiliency to trapping. With this in mind, I recommend:

- The trapping season be closed for all terrestrial furbearing species (aside from rodents) within the range of the Columbian population of fishers until the population has recovered to levels where a harvest season does not result in population declines.
- Notwithstanding the above, 120-class traps could continue to be used to trap mink and marten if they have been retrofitted to specifically exclude fishers as per the specifications and designs provided by the BC Fisher Habitat Working Group.
- The trapping season should remain closed within the range of the Columbian population until trapping systems that are truly species-specific have been designed and approved for use such that only species with a high resiliency to trapping mortality may be captured.

Limitations and Future Research

My study provided much needed information on the current density and abundance of fishers in two geographically distinct ecosystems in central interior BC, and identified which habitat types best predicted the density of fishers in those areas. However, given the variability within these ecosystems and that estimates are based on one year of surveys for each study site, I caution against extrapolating the density models over larger areas until more detection data can be collected. Additionally, the low recapture rates in the Enterprise study site prevented me from determining whether there were sex-based differences affecting male and female fisher densities, which is certainly a possibility. Future research should focus on not only re-surveying these same study areas but also surveying additional areas within the same ecosystem types to help refine the predictive density models to make them applicable to a larger area.

Population Viability Analysis (PVA) is a powerful tool and, when used appropriately, can provide critical information to inform the management of endangered species. My PVA for the Columbian population of fishers utilized survival and reproduction data from studies within central interior BC as well as a small and isolated population of fishers in California, and therefore was informed by the best available empirical information. However, this work also highlights the fact that much of what is known about fishers in BC is based on a few localized studies with limited sample sizes. For example, the litter sizes I used in this model were based on one study area along with data from an artificial den box study (Davis unpubl. data), which may not be representative for the entire Columbian population. This work has shown there are substantial differences in the density of fishers in different regions of BC, and future research should focus on determining whether there are specific differences in the survival and reproductive output of fishers on a regional basis.

The effect that trapping mortality is having on this population appears to be quite significant, however, there were assumptions that needed to be made due to a lack of data about certain aspects of the population including dispersal rates and inbreeding depression. I also did not include a spatial component to my analysis and assumed that demographic rates were consistent throughout the range of the Columbian population. An updated version of this model that includes a spatial component is currently being worked on (Burgar unpubl. data), and my PVA provided the framework for this work. The density estimates I derived for both the Enterprise and Chilcotin study areas also are being incorporated into this new model, and continuing to collect detection data from other areas of the province will help improve this new model going forward. Implementing rigorous and repeatable surveys for fishers on a region-by-region basis would allow changes in population trends to be tracked more effectively and could then be used to test the predictions from the PVA model. Future work looking at whether there are regional differences in survival and reproductive rates for fishers in different areas of the province (e.g., Chilcotin versus Enterprise study areas) would also be highly beneficial. This would then allow for a more detailed metapopulation analysis to determine if fishers in different regions are at greater risk of extirpation so that management efforts could be focused most effectively.

Conclusion

Management strategies that attempt to conserve populations of endangered carnivores usually consist of protecting and/or restoring critical habitats to promote increased recruitment, and reducing the risk of mortality to increase survival rates. These ideas are not new, but they continue to be implemented for a multitude of species quite simply for one reason – they typically are the most successful. In the case of the fisher, these strategies already have been proven to work, with populations in eastern Canada and the United States having recolonized much of their historic range. This study provides further evidence these strategies would be successful in helping to recover the endangered Columbian population of fishers in central BC, and the population monitoring and modelling methods used here can be applied to identify where wildlife managers could focus efforts to protect and recover other at-risk carnivore populations. The extirpation of carnivores by humans continues to repeat itself either directly through shooting, trapping, and poisoning, or indirectly through impacts on prey species or habitat loss. Without revisiting the conservation measures proven to effectively manage these species we may once again lose the important ecosystem services they provide.

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APPENDIX A. DESCRIPTION OF VARIABLES USED IN DENSITY MODELS.

Table A.1. Variables used in candidate models known from previous research to influence the density of fishers within the Chilcotin and Enterprise study areas in the central interior of British Columbia, Canada, along with their associated references. A + symbol indicates a fisher would typically select for this attribute and a – symbol indicates a fisher would typically select against this variable.

Variable Name	Description	Select (+) or Avoid (-)	Reference
Primary branch resting stands	Area of spruce-leading, secondary, or tertiary stands aged ≥83 years old with crown closure ≥40% calculated as a proportion of the 4.84 km^2 and 30 km^2 buffer areas.	+	Weir and Almuedo 2010
Primary denning stands	Area of aspen- or cottonwood-leading stands aged ≥135 years old or Douglas-fir-leading stands aged ≥207 years old with crown closure ≥20% calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Weir and Almuedo 2010
Primary coarse wood resting stands	Area of stands with spruce or aspen content ≥25% and aged ≥100 years old calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Weir and Almuedo 2010
Primary movement stands	Area of stands with total cover ≥50% (≥30% shrub cover and ≥20% tree cover) calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Weir and Almuedo 2010
High intensity burn	Area of stands burned at a high intensity with trees dead, needles, twigs, and understory consumed, calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	-	Hope et al. 2015
Medium or high intensity burn	Area of stands burned at either a high or medium intensity with trees dead, scorched needles remain on trees, understory burned, calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	-	Hope et al. 2015
Stream density	Density of streams in km/km^2 within the 4.84 km2 and 30 km2 buffer areas.	+	Davis 2009
Stream length	Total length of all streams in km within the 4.84 km2 and 30 km2 buffer areas.	+	Davis 2009
Prey density	Estimated total relative density of digestible energy based on kcal/g provided by snowshoe hares and squirrels within the 4.84 km2 and 30 km2 buffer areas.	+	Weir unpubl. data
Wet soil moisture regime	Area of stands with subhygric, hygric, or subhydric soil moisture regimes calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas. Located primarily along wetlands and streams, contain spruce, and a relatively well-developed shrub layer	+	Weir and Corbould 2010
Spruce-aspen riparian forest	Area of spruce-or aspen-leading stands within 100 m of a wetland calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Davis 2009
Riparian closed canopy forest	Area of stands with ≥30% crown closure within 100 m of wetland calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Weir and Corbould 2010

Table continued on next page…

Variable Name	Description	Select (+) or Avoid (-)	Reference
Open areas	Recently logged or burned areas (<12 years old), wetlands, and agricultural fields calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	-	Weir and Corbould 2010
Old riparian forest	Area of stands >100 years old within 100 m of wetland or stream calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Weir and Corbould 2010
Cut blocks harvested <20 years ago	Area of cut blocks harvested <20 years ago calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.		Weir and Corbould 2010
At-leading stands	Area of aspen- or cottonwood-leading stands calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Davis 2009
Sx-dominated riparian forest	Are of spruce-leading stands within 100 m of a stream or wetland calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Davis 2009
Road density	Density of roads in km/km^2 within the 4.84 km2 and 30 km2 buffer areas.	-	Kordosky et al. 2021
Active trap line	Area of one active registered trap line during the Enterprise survey where 8 fishers were trapped, calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	-	Province of BC 2020
Old Douglas-fir forest	Area of Douglas-fir-leading or secondary stands ≥100 years old calculated as a proportion of the 4.84 km2 and 30 km2 buffer areas.	+	Weir and Almuedo 2010

APPENDIX B. SET OF CANDIDATE MODELS CONSIDERED FOR PREDICTING DENSITY OF FISHERS IN THE CHILCOTIN AND ENTERPRISE STUDY AREAS.

Table B.1. Set of candidate models considered for predicting the density of fishers in the Chilcotin study area in the central interior of British Columbia, Canada, during the winter of 2018-19. All models also included a detection sub-model (not shown here) depending on whether I predicted density for both sexes combined, or males and females separately, and candidate sets were considered at both the core use (4.84 km^2) and home range (30km^2) scales.

Model Name	Model
C_1	D ~ Primary branch resting stands
C_2	D ~ Primary denning stands
C_3	D ~ Coarse woody debris resting stands
C_4	D ~ Primary branch resting stands + stands burned at high intensity
C_5	D ~ Primary denning stands + stands burned at high intensity
C_6	D ~ Primary branch resting stands + stands burned at either medium or high intensity
C_7	D ~ Primary denning stands + stands burned at either medium or high intensity
C_8	D ~ Primary denning stands + primary branch resting stands + primary movement stands
C_9	D ~ Primary denning stands + primary branch resting stands
C_10	D ~ Primary branch resting stands + primary movement stands
C_11	D ~ Primary denning stands + stream density
C_12	D ~ Primary denning stands + total length of streams
C_13	D ~ Primary branch resting stands + stream density
C_14	D ~ Primary denning stands + prey density
C_15	D ~ Stands with wet soil moisture regimes
C_16	D ~ Sx or At-leading stands within 100 m of wetland
C_17	D ~ Sx or At-leading stands within 100 m of wetland + stands with >30% crown closure within 100 m of wetland
C_18	D ~ Open areas
C_19	D ~ Primary movement stands
C_20	D ~ Stands >100 years old within 100 m of wetland
C_21	D ~ Cut blocks harvested <20 years ago
Null	D ~ 1 (Density remained uniform across the study area)

Table B.2. Set of candidate models considered for predicting the density of fishers in the Enterprise study area in the central interior of British Columbia, Canada, during the winter of 2020-21. All models also included a detection sub-model with no variation in g0 or σ and were considered at both the core use (4.84 km^2) and home range (30km^2) scales.

Model Name	Model
E_1	D ~ Primary branch resting stands
E_2	D ~ Primary denning stands
E_3	D ~ Coarse woody debris resting stands
E_4	D ~ Primary branch resting stands + primary movement stands
E_5	D ~ Primary denning stands + primary branch resting stands
E_6	D ~ Primary branch resting stands + aspen-leading stands
E_7	D ~ Primary denning stands + Stands with wet soil moisture regimes
E_8	D ~ Primary denning stands + stream density
E_9	D ~ Primary denning stands + total length of streams
E_10	D ~ Sx-dominated riparian forest
E_11	D ~ Open areas
E_12	D ~ Density of roads
E_13	D ~ Active trapline
E_14	D ~ Cut blocks harvested <20 years ago
E_15	D ~ Primary movement stands
E_16	D ~ Crown closure >50%
E_17	D ~ Old Douglas-fir forest
E_18	D ~ Mature and old stands
E_19	D ~ Stands >100 years old within 100 m of wetland
E_20	D ~ Stands >100 years old within 100 m of a stream
Null	D ~ 1 (Density remained uniform across the study area)

APPENDIX C. 95% CONFIDENCE SET OF CANDIDATE MODELS FOR PREDICTING THE DENSITY OF FISHERS IN THE CHILCOTIN AND ENTERPRISE STUDY AREAS.

Table C.1. Comparing model fit between the 95% confidence set of candidate density models for fishers (both sexes) from the Chilcotin study area in the central interior of British Columbia, Canada, during the winter of 2018-19. I estimated 3 parameters: density (D), detection probability (g0), and a spatial parameter (sigma) at the home range scale (30 km^2). All models included a behavioural effect where g0 depends on detection at the preceding occasion and no variation in sigma. I report here the number of model parameters (K), model log likelihood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (w_i).

Model	K	logLik	AIC$_c$	ΔAIC$_c$	w_i
Primary branch resting stands	6	-442.22277	898.494	0	0.1699
Primary denning stands	6	-442.92899	899.907	1.413	0.0838
Primary denning stands + high intensity burn	7	-441.70911	900.218	1.724	0.0717
Stands with wet soil moisture regimes	6	-443.17355	900.396	1.902	0.0656
Primary branch resting stands + high intensity burned stands	7	-441.86828	900.537	2.043	0.0612
Primary branch resting stands + primary denning stands	7	-441.91671	900.633	2.139	0.0583
Sx- or At-leading stands within 100 m of wetland	6	-443.37728	900.803	2.309	0.0535
Primary branch resting stands + stream density	7	-442.01885	900.838	2.344	0.0526
Primary denning stands + medium or high intensity burned stands	7	-442.0941	900.988	2.494	0.0488
Primary branch resting stands + primary movement stands	7	-442.17371	901.147	2.653	0.0451
Primary branch resting stands + medium or high intensity burn stands	7	-442.21686	901.234	2.74	0.0432
Null	5	-445.07928	901.587	3.093	0.0362
Sx or At-leading stands within 100 m of wetland + stands with >30% crown closure within 100 m of wetland	7	-442.45834	901.717	3.223	0.0339
Stands >100 years old within 100 m of wetland	6	-443.83987	901.729	3.235	0.0337
Primary denning + prey density	7	-442.70218	902.204	3.71	0.0266
Cut blocks harvested <20 years ago	6	-444.29887	902.647	4.153	0.0213
Primary denning + stream density	7	-442.92854	902.657	4.163	0.0212
Primary denning + total length of streams	7	-442.92854	902.657	4.163	0.0212
Primary coarse wood resting habitat	6	-444.6061	903.261	4.767	0.0157

Table C.2. Comparing model fit between the 95% confidence set of candidate density models for fishers (both sexes) from the Chilcotin study area in the central interior of British Columbia, Canada, during the winter of 2018-19. I estimated 3 parameters: density (D), detection probability (g0), and a spatial parameter (sigma) at the core use scale (4 km^2). All models included a behavioural effect where g0 depends on detection at the preceding occasion and no variation in sigma. I report here the number of model parameters (K), model log likelihood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (w_i).

Model	K	logLik	AIC$_c$	ΔAIC$_c$	w_i
Primary branch resting stands	6	-441.01821	896.085	0	0.2569
Primary branch resting stands + stream density	7	-440.51565	897.831	1.746	0.1073
Primary branch resting stands + Primary denning stands	7	-440.73139	898.263	2.178	0.0865
Primary branch resting stands + stands burned at a high intensity	7	-440.77092	898.342	2.257	0.0831
Primary branch resting stands + primary movement stands	7	-440.99884	898.798	2.713	0.0662
Primary branch resting stands + stands burned at a high or medium intensity	7	-441.01742	898.835	2.75	0.065
Sx- or At-leading stands within 100 m of wetland	6	-442.62004	899.289	3.204	0.0518
Stands with wet soil moisture regimes	6	-442.89244	899.834	3.749	0.0394
Primary denning stands	6	-442.90056	899.85	3.765	0.0391
Stands >100 years old within 100 m of wetland	6	-442.98999	900.029	3.944	0.0358
Sx or At-leading stands within 100 m of wetland + stands with >30% crown closure within 100 m of wetland	7	-441.62669	900.053	3.968	0.0353
Primary branch resting stands + primary denning stands + primary movement stands	8	-440.66814	901.029	4.944	0.0217
Primary density stands + stands burned at a high or medium intensity	7	-442.14425	901.088	5.003	0.0211
Primary denning stands + stands burned at a high intensity	7	-442.22734	901.255	5.17	0.0194
Null	5	-445.07928	901.587	5.502	0.0164

Table C.3. Comparing model fit between the 95% confidence set of candidate density models for fishers (females only) from the Chilcotin study area in the central interior of British Columbia, Canada, during the winter of 2018-19. I estimated 3 parameters: density (D), detection probability (g0), and a spatial parameter (sigma) at the home range scale (30 km^2). All models included a behavioural effect where g0 depends on detection at the preceding occasion and no variation in sigma. I report here the number of model parameters (K), model log likelihood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (w_i).

Model	**K**	**logLik**	**AIC$_c$**	**ΔAIC$_c$**	**w_i**
Stands with wet soil moisture regimes	5	-283.24974	578.722	0	0.2197
Primary denning stands	5	-283.9211	580.064	1.342	0.1123
Sx- or At-leading stands within 100 m of wetland	5	-283.99545	580.213	1.491	0.1042
Null	4	-285.48957	580.408	1.686	0.0945
Stands >100 years old within 100 m of wetland	5	-284.37415	580.971	2.249	0.0713
Primary denning stands + prey density	6	-283.1026	581.436	2.714	0.0565
Primary denning stands + high intensity burn	6	-283.41174	582.054	3.332	0.0415
Primary branch resting stands	5	-285.05834	582.339	3.617	0.036
Sx or At-leading stands within 100 m of wetland + stands with >30% crown closure within 100 m of wetland	6	-283.61939	582.47	3.748	0.0337
Cut blocks harvested <20 years ago	5	-285.14261	582.507	3.785	0.0331
Primary denning stands + medium or high intensity burn	6	-283.80833	582.847	4.125	0.0279
Primary denning stands + stream density	6	-283.83379	582.898	4.176	0.0272
Primary coarse wood resting stands	5	-285.35081	582.924	4.202	0.0269
Open areas	5	-285.39535	583.013	4.291	0.0257
Primary branch resting stands + primary denning	6	-283.91947	583.07	4.348	0.025
Primary movement stands	5	-285.48286	583.188	4.466	0.0235

Table C.4. Comparing model fit between the 95% confidence set of candidate density models for fishers (females only) from the Chilcotin study area in the central interior of British Columbia, Canada, during the winter of 2018-19. I estimated 3 parameters: density (D), detection probability (g0), and a spatial parameter (sigma) at the core use area scale (4.84 km^2). All models included a behavioural effect where g0 depends on detection at the preceding occasion and no variation in sigma. I report here the number of model parameters (K), model log likelhood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (w_i).

Model	**K**	**logLik**	**AIC$_c$**	**ΔAIC$_c$**	w_i
Stands with wet soil moisture regimes	5	-282.66874	577.56	0	0.2353
Primary denning	5	-283.23751	578.697	1.137	0.1333
Sx- or At-leading stands within 100 m of wetland	5	-283.36233	578.947	1.387	0.1176
Stands >100 years old within 100 m of wetland	5	-283.76019	579.743	2.183	0.079
Primary denning + prey density score	6	-282.51393	580.259	2.699	0.061
Null	4	-285.48957	580.408	2.848	0.0566
Primary denning + high intensity burn	6	-282.94761	581.126	3.566	0.0396
Sx or At-leading stands within 100 m of wetland + stands with >30% crown closure within 100 m of wetland	6	-282.97175	581.174	3.614	0.0386
Primary denning + stream density	6	-283.11155	581.454	3.894	0.0336
Primary denning + medium or high intensity burn	6	-283.12796	581.487	3.927	0.033
Primary branch resting + primary denning	6	-283.14642	581.524	3.964	0.0324
Primary branch resting	5	-284.71097	581.644	4.084	0.0305
Primary coarse wood resting	5	-285.00629	582.235	4.675	0.0227
Cut blocks harvested <20 years ago	5	-285.19081	582.604	5.044	0.0189
Open areas	5	-285.3736	582.969	5.409	0.0157
Primary movement	5	-285.48869	583.2	5.64	0.014

Table C.5. Comparing model fit between the 95% confidence set of candidate density models for fishers (males only) from the Chilcotin study area in the central interior of British Columbia, Canada, during the winter of 2018-19. I estimated 3 parameters: density (D), detection probability (g0), and a spatial parameter (sigma) at the home range scale (30 km^2). All models included no variation in g0 or sigma. I report here the number of model parameters (K), model log likelhood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (w_i).

Model	K	logLik	AIC$_c$	ΔAIC$_c$	w_i
Primary branch resting stands	4	-153.60432	319.209	0	0.4205
Primary branch resting stands + stream density	5	-152.71768	322.102	2.893	0.099
Null	3	-157.12617	322.434	3.225	0.0838
Primary branch resting stands + primary denning stands	5	-153.18098	323.029	3.82	0.0623
Primary branch resting stands + high intensity burn	5	-153.41982	323.506	4.297	0.0491
Primary branch resting stands + stands burned at high or medium intensity	5	-153.42511	323.517	4.308	0.0488
Primary branch resting + primary movement stands	5	-153.5893	323.845	4.636	0.0414
Open areas	4	-156.11872	324.237	5.028	0.034
Primary denning stands	4	-156.44751	324.895	5.686	0.0245
Primary movement stands	4	-156.46024	324.92	5.711	0.0242
Primary coarse wood resting stands	4	-156.60483	325.21	6.001	0.0209
Cut blocks harvested <20 years ago	4	-156.62557	325.251	6.042	0.0205
Sx- or At-leading stands within 100 m of wetland	4	-156.82312	325.646	6.437	0.0168
Stands >100 years old within 100 m of wetland	4	-156.90104	325.802	6.593	0.0156

Table C.6. Comparing model fit between the 95% confidence set of candidate density models for fishers (males only) from the Chilcotin study area in the central interior of British Columbia, Canada, during the winter of 2018-19. I estimated 3 parameters: density (D), detection probability (g0), and a spatial parameter (sigma) at the core area use scale (4.84 km^2). All models included no variation in g0 or sigma. I report here the number of model parameters (K), model log likelihood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (w_i).

Model	**K**	**logLik**	**AIC$_c$**	**ΔAIC$_c$**	**w_i**
Primary branch resting stands	4	-152.7998	317.6	0	0.4402
Primary branch resting stands + stream density stands	5	-151.11257	318.892	1.292	0.2307
Primary branch resting stands + primary denning stands	5	-152.48264	321.632	4.032	0.0586
Primary branch resting stands + primary movement stands	5	-152.61673	321.9	4.3	0.0513
Primary branch resting stands + medium or high intensity burn	5	-152.61774	321.902	4.302	0.0512
Primary branch resting stands + high intensity burn	5	-152.74792	322.163	4.563	0.045
Null	3	-157.12617	322.434	4.834	0.0393
Cut blocks harvested <20 years ago	4	-156.37194	324.744	7.144	0.0124
Primary movement stands	4	-156.41179	324.824	7.224	0.0119
Open areas	4	-156.41889	324.838	7.238	0.0118

Table C.7. Comparing model fit between the 95% confidence set of candidate density models for fishers (both sexes) from the Enterprise study area in the central interior of British Columbia, Canada, during the winter of 2020-21. I estimated 3 parameters: density (D), detection probability (g0), and a spatial parameter (sigma) at the home range scale (30 km^2). All models included no variation in g0 or sigma. I report here the number of model parameters (K), model log likelihood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (wi).

Model	**K**	**logLik**	**AIC$_c$**	**ΔAIC$_c$**	***w*$_i$**
Crown closure >50%	5	-149.22637	312.453	0	0.2195
Open areas	5	-149.259	312.518	0.065	0.2125
Stands with wet soil moisture regimes	5	-149.79926	313.599	1.146	0.1238
Primary movement stands	5	-150.01018	314.02	1.567	0.1003
Sx-leading stands >100 years old within 100 m of a wetland	5	-150.48161	314.963	2.51	0.0626
Null	4	-152.29054	315.081	2.628	0.059
Road density	5	-150.76034	315.521	3.068	0.0473
Cut blocks harvested <20 years ago	5	-151.12867	316.257	3.804	0.0328
Primary coarse wood resting stands	5	-151.41791	316.836	4.383	0.0245
Mature and old structural stages	5	-151.42813	316.856	4.403	0.0243
Active trap line	5	-151.49095	316.982	4.529	0.0228
Primary denning stands	5	-151.74062	317.481	5.028	0.0178
Fd-leading stands >100 years old	5	-151.84659	317.693	5.24	0.016

Table C.8. Comparing model fit between the 95% confidence set of candidate density models for fishers (both sexes) from the Enterprise study area in the central interior of British Columbia, Canada, during the winter of 2020-21. I estimated 3 parameters: density (D), detection probability (g0), and a spatial parameter (sigma) at the fisher core use area scale (4.84 km^2). All models included no variation in g0 or sigma. I report here the number of model parameters (K), model log likelihood (logLik), Akaike's Information Criterion corrected for small sample sizes (AICc), the difference in AICc values (ΔAICc), and relative model weight (wi).

Model	**K**	**logLik**	**AIC$_c$**	**ΔAIC$_c$**	***w*$_i$**
Stands with wet soil moisture regimes	5	-147.819535	309.639	0	0.5005
Crown closure >50%	5	-148.994511	311.989	2.35	0.1546
Open areas	5	-149.513937	313.028	3.389	0.0919
Primary movement stands	5	-149.772111	313.544	3.905	0.071
Null	4	-152.290541	315.081	5.442	0.0329
Road density	5	-150.621095	315.242	5.603	0.0304
Sx-leading stands >100 years old within 100 m of a wetland	5	-150.876009	315.752	6.113	0.0235
Cut blocks harvested <20 years ago	5	-151.312198	316.624	6.985	0.0152
Mature and old structural stages	5	-151.338915	316.678	7.039	0.0148
Active trap line	5	-151.488459	316.977	7.338	0.0128
Primary denning stands	5	-151.571582	317.143	7.504	0.0117