



RESEARCH ARTICLE

Forestry, fire, and fur: factors influencing densities of endangered fishers (*Pekania pennanti*) in central British Columbia, Canada

Rory D. Fogarty¹  | Richard D. Weir²  | Karl W. Larsen³ 

¹Environmental Science Program, Thompson Rivers University, Kamloops, British Columbia V2C 0C8, Canada

²Ministry of Water, Land and Resource Stewardship, Government of British Columbia, Victoria, British Columbia V8W 9M1, Canada

³Department of Natural Resource Sciences, Thompson Rivers University, Kamloops, British Columbia V2C 0C8, Canada

Correspondence

Rory D. Fogarty, Environmental Science Program, Thompson Rivers University, Kamloops, British Columbia, V2C 0C8, Canada.

Email: fogarty.rory@gmail.com

Funding information

Habitat Conservation Trust Foundation; Ministry of Forests, Lands and Natural Resource Operations; BC Ministry of Environment and Climate Change Strategy; Together for Wildlife Strategy

Abstract

The isolated and endangered Columbian population of fishers (*Pekania pennanti*) is thought to be declining in central interior British Columbia, Canada, yet little is known about the distribution and abundance of the species in several regions. We conducted DNA-based mark-recapture surveys to estimate the density of fishers in 2 spatially distinct ecosystems where current data were lacking, and evaluated which factors best explained the distribution of fishers in each ecosystem. We found the density of fishers varied substantially between study areas, likely because of the level of trapping mortality 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 (21.3 ± 3.8 [SE] fishers/1,000 km²), whereas density in the Enterprise study area southeast of Williams Lake was among the lowest (8.9 ± 2.7 fishers/1,000 km²). Density estimates for both study areas, however, were orders of magnitude lower than those reported for eastern North America and the western United States and thus are at levels worthy of concern. Forested stands with wet soil moisture regimes composed of older deciduous and spruce trees were most strongly related to higher densities in both study areas; these stands also were known to contain higher densities of the denning and resting structures fishers require to survive and reproduce. This study provided wildlife managers with a snapshot of the density of fishers in 2 areas of

central interior British Columbia that will help refine the estimate for the endangered Columbian population. Additionally, this work suggests that reducing trapping mortality within the Columbian population of fishers and increasing the protection of high-value denning and resting habitats may both be necessary to promote the recovery of the species over the longer term.

KEYWORDS

British Columbia, density, fisher, mesocarnivore, Mustelid, *Pekania pennanti*, spatially explicit capture-recapture

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 2020). Rare carnivores also are often highly sensitive to disturbance and at greater risk of extinction, thus being frequently prioritized for conservation (Buskirk and Zielinski 2003, Ripple et al. 2014). Unfortunately, for many threatened and rare species, detailed population data are lacking and not enough is known to develop appropriate conservation measures (Carroll et al. 2001, Laliberte and Ripple 2004).

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). Throughout their range, fishers occupy forests with relatively high levels of overhead canopy 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). Structures used for resting and denning (e.g., cavities, platforms, and other microstructures) are key habitat features that can be rare across the landscape (Weir and Harestad 2003, Aubry and Raley 2006, Purcell et al. 2009). Female fishers appear to have specific 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 an adult female fisher and up to 3 kits (Weir and Corbould 2008) while having an opening small enough to exclude predators, including adult male fishers (Raley et al. 2012, Matthews et al. 2019). In early April, females use multiple den trees for the 3-month period following kit parturition (Weir and Corbould 2008), with previous research suggesting that high local densities of suitable cavity trees can be an important predictor for whether a tree will be used as a den site by female fishers (Green et al. 2019, Berg et al. 2020). The disease and decay processes that form these denning and resting structures can take up to a century to develop (Basham 1958), making fishers potentially susceptible to habitat disturbances, such as rotational forestry, that remove these structures at rates faster than they regenerate (Weir et al. 2012).

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). Since the mid-1900s, fishers east of the Rocky Mountains recolonized many portions of their historical range, partially attributed 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 5 translocation programs, fishers west of the Rocky Mountains remain geographically restricted and occur as smaller and more isolated populations in British Columbia, Canada, and Washington, Oregon, California, Idaho, and Montana, US (Lapoint et al. 2015; Lewis et al. 2012, 2016).

The densities of fishers in parts of British Columbia are among the lowest recorded anywhere within their range (e.g., 8.8 fishers/1,000 km²; Weir and Corbould 2006), with the highest density for the province being reported in the northeast (16.7 fishers/1,000 km²; 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/1,000 km² (Davis and Weir 2021). These estimates are much lower than those documented in eastern North America (49–385 fishers/1,000 km²; as cited in Weir et al. 2003) and California (140 fishers/1,000 km²; Matthews et al. 2011; Table 1). The reasons for these differences in the density of fishers in British Columbia 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), or other factors such as community predator dynamics, of which much is still unknown (Wengert et al. 2014, Gabriel et al. 2015).

Recent research has elucidated there are 2 distinct populations of fishers in British Columbia—Boreal and Columbian population—separated by a high snowpack zone delineated by the Rocky Mountain divide and the mountainous region south of the Spatsizi Plateau (Figure S1). The most recent estimate for the Boreal population was 896–1,519 adult individuals (British Columbia Conservation Data Centre 2020a) with this population presumably still being relatively contiguous with other populations in the boreal forests of Alberta, Canada. The most recent estimate for the Columbian population was much lower at 299–517 adult individuals (British Columbia Conservation Data Centre 2020b); given that fishers have effectively been extirpated from most of southern British Columbia, this new research suggests the Columbian population of fishers is geographically isolated from other populations in North America with demographic rescue through immigration unlikely (Weir et al. 2024). In recent decades, the low-elevation forests these fishers occupy in the central interior of British Columbia have been subject to disturbance from industrial development (e.g., hydroelectric, oil, and gas), timber harvesting activities, large-scale insect infestations, wildfire, and the accelerated salvage harvest of insect- and fire-damaged trees (Eng et al. 2005; Province of British Columbia 2017, 2018). Isolation, habitat loss, and the accompanying decline in population size prompted the British Columbia government to revise the conservation status of the Columbian population to endangered (British Columbia Conservation Data Centre 2020b).

Although recent trends indicate the Columbian population is declining, the true extent of any decline remains unknown because of inadequate monitoring of fishers at the provincial scale. In British Columbia, 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 2,400-km² pilot study plots in areas suitable for fishers 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 British Columbia 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 (British Columbia Conservation Data Centre 2020b).

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 population declines in central British Columbia. Therefore, the main objective of this study was to use non-invasive DNA-based surveys in 2 representative ecosystems within the range of the Columbian population of fishers

TABLE 1 Density estimates for fishers (*Pekania pennanti*) in North America, with 95% confidence intervals (95% CI) included if reported. Density estimates from British Columbia are indicated with an asterisk.

Study area	Estimation method	Density (fishers/1,000 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)
Northern Sierra Nevada, California, USA	Spatial capture-recapture	108*	87.6–128	Green et al. (2022)

coupled with spatially explicit capture-recapture methodologies to estimate density and abundance where current data did not exist. Additionally, we 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. We 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).

STUDY AREA

We conducted DNA-based hair-snagging surveys in 2 study areas within the range of the Columbian population of fishers (Figure 1). The first was a 2,440-km² region on the Chilcotin Plateau west of Williams Lake, where data were collected during winter 2018–2019 (hereafter the Chilcotin study area; Figure S2); the second was located southeast of Williams Lake (2,580 km²) where data were collected during winter 2020–2021 (hereafter the Enterprise study area; Figure S3). Both study areas consisted of level to gently rolling topography and surveys were conducted in non-mountainous areas at low to moderate elevations (750–1,700 m).

The Chilcotin study area was within the driest ecological zone where fishers are found in British Columbia with forests dominated by lodgepole pine (*Pinus contorta*) and a small component of Douglas-fir (*Pseudotsuga menziesii*), trembling aspen (*Populus tremuloides*), and hybrid spruce (*Picea glauca* × *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 study areas see Fogarty (2023).

Both study areas have histories of logging and mining, with large areas of pine trees also killed by mountain pine beetle (*Dendroctonus ponderosae*) and subsequently salvage harvested at accelerated rates during the past 2 decades (>18 million ha affected; Government of Canada 2024). These industries built and continue to build high-density 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 (*Dendroctonus pseudotsugae*) infestations. Approximately 30% of the Chilcotin study area was burned at varying severities by the Plateau Fire and roughly 3% of the Enterprise study area was burned by the Gustafsen Lake Fire; both of these fires occurred during the summer of 2017, which at the time was the most severe wildfire season on record for British Columbia (Province of British Columbia 2017).

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

METHODS

Fisher DNA sampling

We conducted DNA-based surveys by remotely collecting hair samples following consistent sampling methodology in the 2 study areas. We divided each study area into 20-km² cells to approximate the smallest expected size of a

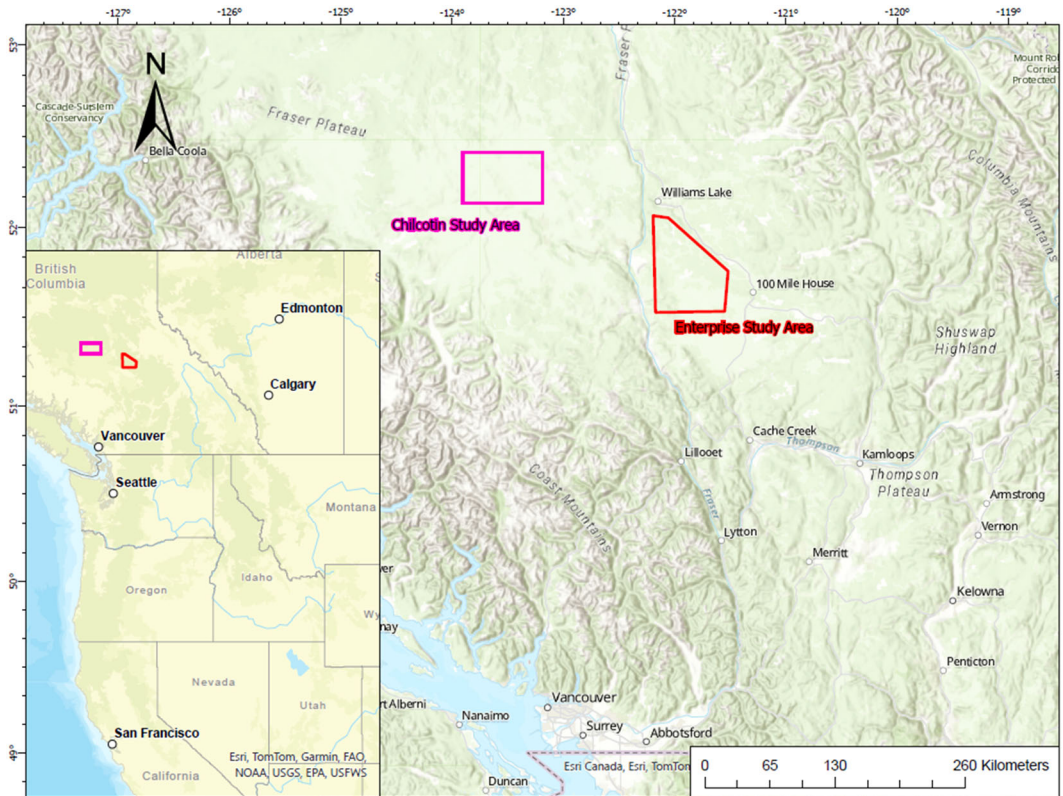


FIGURE 1 The location of the 2 DNA-based hair snagging surveys completed within the range of the Columbian population of fishers during the winters of 2018–2019 and 2020–2021.

female home range in the central interior (Weir et al. 2009). Field crews installed one sampling site in each cell during each session 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 2 pieces of wood ($2 \times 19 \times 60$ cm) screwed together to form a triangular cubby structure and both ends left open. 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 (Hawbaker's Beaver Lure, Hawbaker's Lures, Fort Loudon, PA, USA) and fisher lure (Hawbaker's Fisher Lure, Hawbaker's Lures) mixture on the chicken and a woolen jute string hung on branches next to the cubby to act as attractants. Cubbies had 4 pieces (approximately 1.5×5 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 19×30 -cm roof board above the unit to prevent rain or snow from degrading any hair samples present.

Field crews accessed sampling sites by foot, truck, snowmobile, and helicopter, and deployed cubbies at sites in ideal fisher habitat when possible (e.g., older stands with high amounts of canopy closure and complex woody debris; Mowat and Paetkau 2002). However, given that some portions of each survey area consisted of 10- to 20-year-old monoculture pine stands and other portions were burned, some sampling sites were set up in less ideal habitats and adjacent to open areas. Crews revisited sites at 21-day intervals beginning in mid-December and ending in mid-March for 4 sampling sessions each winter. Upon each revisit crews assessed the glue pads and cubby edges for hair and follicle tissues. For glue pads that had collected hair, crews covered them with plastic paper, placed them in a paper envelope, and stored them 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.

We sent all hair samples to Wildlife Genetics International (WGI) in Nelson, British Columbia for microsatellite genotyping. We analyzed an additional 7 fisher hair and tissue samples collected from a trapper operating in the Enterprise study area at the time of the survey 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 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 British Columbia (Lut604, MP0055, Ma-1, MP0247, Mvis072, MP0144, MP0182; Weir et al. 2013). For each individual identified by its multi-locus genotype, the lab determined sex using the ZFX/ZFY/SRY sex marker (Davis and Weir 2021). Error checking followed Paetkau (2003), proven with other species to consistently produce low error rates (Kendall et al. 2009).

Spatial capture-recapture analysis

We employed spatially explicit capture-recapture (SECR) methods in 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 3 parameters: density (D), detection probability (g_0), 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 animal home range centers in a population initially is treated as a homogeneous Poisson point process where density (i.e., 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 (g_0) with distance from an animal's home range center (Efford 2021). Therefore, g_0 is technically defined as the detectability of an individual at a certain detector if the individual's home range center was at that exact location. The spatial scale parameter σ is related to the movement during sampling occasions and is used along with animal capture histories to estimate centers of individuals in the sampling area and the associated area of integration (Borchers and Efford 2008). Estimation of density, g_0 , and σ is achieved by numerically maximizing the likelihood with respect to the parameters of each of the sub-models (Efford et al. 2009).

We used package `secr` (Efford 2021) in Program R (R Core Team 2020) to estimate the density of fishers in the 2 study areas. We 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 (Efford 2021). Following Efford (2021), we estimated the root pooled spatial variance to generate a preliminary and biased estimate of σ and buffered the detector array for each study area by 4 times this preliminary σ (10,000-m and 18,000-m buffer widths for the Chilcotin and Enterprise study areas, respectively). We 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). We chose a mask point spacing of 2,200 m, which fell within the range of recommended sizes ($<\sigma$; Efford 2021) and yielded a computationally reasonable number of mask points for both study areas (991 for the Chilcotin, 1,220 for the Enterprise). The Interior Douglas-

fir very dry-mild (IDFxm) biogeoclimatic zone is not considered viable fisher habitat in British Columbia (Weir and Almuedo 2010); therefore, we 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 3 modeling parameters (Efford 2021), and we first evaluated the effects of ecologically and behaviorally relevant covariates on g_0 and σ to determine the candidate observation model most supported by the data that was biologically reasonable to be included in the second step of the analysis (i.e., density estimation). For the detection models, we 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, we included automated behavioral 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). We expected that detection probability might change between sampling sessions so we 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 ; Efford 2021). Male fisher home ranges are substantially larger than female home ranges in British Columbia (Weir et al. 2009) and both sexes use space differently; therefore, we also included a sex covariate (h_2). For the Chilcotin study area, we expected that burn severity 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

We 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). We used spatial data from the British Columbia Data Catalogue (Province of British Columbia 2021) and the British Columbia Fisher Habitat and Forestry Web Module (BC Fisher Habitat Working Group 2021) to assess the effects of habitat covariates on fisher density in our study areas (Table S1). Because fishers have been shown to select habitat at multiple scales (Weir and Harestad 2003), we used a moving window analysis to re-scale covariates using 2 different buffer sizes around mask points representing the average size of a female fisher core use area (central 4.84-km² area within a female's home range; Weir 1995) and an average home range size for female fishers (30 km²; Davis 2009, Weir et al. 2009) in central interior British Columbia. We based these buffer sizes on the female portion of the population because they appear to have the most specific habitat requirements in British Columbia (Weir and Corbould 2008, Davis 2009, Lofroth et al. 2010). We calculated the proportion, length, or density of covariates within these buffer areas and appended these data to the habitat mask prior to fitting candidate models. We tested for correlation between habitat covariates using the `corrplot` function in the R package `corrplot` (Wei 2017) and did not include highly correlated covariates in the same model (Spearman's r rank >0.6; Dormann et al. 2013).

For detection and density models, we used Akaike's Information Criterion corrected for small sample sizes (AIC_c) to evaluate the support for each model in a candidate set (Tables S2–S3) to identify the model that was best supported by the data (Burnham and Anderson 2002). We first compared detection models from each candidate set to determine the detection variables best supported by the data. We chose detection models based on support from the data and knowledge of fisher biology while also considering model parsimony. We then included this

TABLE 2 Variables used in candidate models known from previous research to influence the density of fishers (*Pekania pennanti*) within the Chilcotin and Enterprise study areas in the central interior of British Columbia, Canada. We report the variable name, variable description, and whether it was used in the candidate model set for each study area (C = Chilcotin, E = Enterprise). We calculated the area, length, or density of each variable for both the 4.84-km² (core use scale) and the 30-km² (home range scale) buffer areas around each habitat mask point.

Variable	Description	Study area
Primary branch resting	Area of spruce-leading, secondary, or tertiary stands aged ≥ 83 years old with crown closure $\geq 40\%$	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 $\geq 20\%$	C + E
Primary coarse wood resting	Area of stands with spruce or aspen content $\geq 25\%$ and aged ≥ 100 years old	C + E
Primary movement	Area of stands with total cover $\geq 50\%$ ($\geq 30\%$ shrub cover and $\geq 20\%$ 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
Stands with wet soil moisture regimes	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 $\geq 30\%$ 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
Crown closure >50%	Area of stands with crown closure $\geq 50\%$ calculated as a proportion of the 4.84 km ² and 30 km ² buffer areas.	E

detection model in the second step of the analysis identifying which density model was best supported by the data. Models within 2 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). We 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 $\sum w_i \geq 0.95$.

As with other capture-recapture modeling, 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) we calculated beta coefficients and their associated 95% confidence intervals and if the 95% confidence intervals spanned zero, we treated those variables as potentially informative but not reliable predictors. We 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. We 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. We 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. We 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.

We initially ran density models using combined data for both sexes, as we 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, we also ran separate models for each sex.

RESULTS

Chilcotin study area

In the Chilcotin study area, we identified 48 individual fishers (15 males, 33 females) through 127 detections across 256 sampling sites over the course of the winter. A total of 3 fishers were harvested from a trapline overlapping part of the Chilcotin study area in the winter of 2018–2019, according to the British Columbia Provincial Wild Fur Harvest database (Province of British Columbia unpublished data: British Columbia 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 behavioral effect, where detection probability depended on whether a fisher was detected at a site on the previous occasion (Table 3). This learned effect also was included in the 3 top-ranked detection models for females only that were well supported by the data ($\Delta AIC_c < 2.0$; Table 3). The top-ranked male-only detection model included a site learned response where site effectiveness changes once any animal was detected; however, the model with detection probability and σ fixed also performed well ($\Delta AIC_c < 1.0$; Table 3). Although multiple detection models were well-supported by the data for the combined sex and female-only models, we retained the top-ranked detection models for future fitting, as we felt they were reasonable based on the ecology of the species. We retained the male-only detection model with detection probability and σ fixed because 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 if there had been any previous fisher detection at the site. This model also estimated that the population had a male-to-female ratio of 31 ± 0.067 males to 69 ± 0.067 females. Detection probability for the female-only model followed a similar pattern

TABLE 3 Model selection for detection for fishers (*Pekania pennanti*) in the Chilcotin (2018–2019) and Enterprise (2020–2021) study areas in the central interior of British Columbia, Canada. We estimated 3 parameters: density (D), detection probability (g_0), and a spatial parameter (σ). No density covariates were included in this analysis. The table shows the number of model parameters (K), model log likelihood ($\log\text{Lik}$), Akaike's Information Criterion corrected for small sample sizes (AIC_c), the difference in AIC_c values (ΔAIC_c), and relative model weight (w_i). Models ≤ 2.0 AIC_c units of the top-ranked models are displayed, with the null model assuming g_0 and σ are fixed shown for reference.

Study area	Sex	Model ^a	K	$\log\text{Lik}$	AIC_c	ΔAIC_c	w_i
Chilcotin	Both	$g_0 \sim B, \sigma \sim 1$	5	-445.1	901.59	0.00	0.37
		$g_0 \sim B + T, \sigma \sim 1$	6	-444.8	903.68	2.09	0.13
		$g_0 \sim 1, \sigma \sim 1$ (null)	4	-454.4	917.79	16.21	0.00
	Female	$g_0 \sim B, \sigma \sim 1$	4	-285.5	580.41	0.00	0.40
		$g_0 \sim B, \sigma \sim B$	5	-284.3	580.79	0.38	0.33
		$g_0 \sim B + T, \sigma \sim 1$	5	-284.6	581.44	1.04	0.24
		$g_0 \sim 1, \sigma \sim 1$ (null)	3	-297.2	601.24	20.83	0.00
	Male	$g_0 \sim k, \sigma \sim 1$	4	-154.8	321.55	0.00	0.22
		$g_0 \sim bk, \sigma \sim 1$	4	-155.1	322.23	0.68	0.16
		$g_0 \sim 1, \sigma \sim 1$ (null)	3	-157.1	322.43	0.88	0.14
		$g_0 \sim \text{Low}, \sigma \sim 1$	4	-155.4	322.84	1.29	0.12
		$g_0 \sim Bk, \sigma \sim 1$	4	-155.8	323.52	1.96	0.08
	Enterprise	Both	$g_0 \sim 1, \sigma \sim B$	5	-149.6	313.27	0.00
$g_0 \sim 1, \sigma \sim 1$ (null)			4	-152.3	315.08	1.81	0.15

^aDetection covariates included whether there was a detection on the previous occasion (B), a trap-specific behavior (bk), a trend over trapping occasion (T), site effectiveness changes once any animal is caught (k), and low intensity burn (Low). A 1 indicates g_0 or σ remained constant.

with the value at the home range center being 0.12 ± 0.045 for naïve fishers and 0.47 ± 0.079 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 . The σ values were similar between all 3 model categories: $2.75 \text{ km} \pm 0.16$ for the model combining both sexes, $2.68 \text{ km} \pm 0.19$ for the female-only model, and $2.83 \text{ km} \pm 0.29$ 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 sets of candidate models contained nearly all the candidate models we considered (Tables S4–S9). 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 4). 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.

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 subhydryc soils) within both the 4.84-km^2 and 30-km^2 buffer areas, with the models including area of primary denning stands and spruce- or aspen-leading stands

TABLE 4 Model selection for the top-ranked density models for fishers (*Pekania pennanti*) in the Chilcotin (2018–2019) study area in the central interior of British Columbia, Canada. We estimated 3 parameters, density (D), detection probability ($g0$), and a spatial parameter (σ), 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 likelihood ($\log\text{Lik}$), Akaike's Information Criterion corrected for small sample sizes (AIC_c), the difference in AIC_c values (ΔAIC_c), and relative model weight (w_i). Models $\leq 2.0 \text{ AIC}_c$ units of the top-ranked models are displayed, along with the null model that assumes all parameters are constant.

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

^aDetection covariates included whether there was a detection on the previous occasion (B). A 1 indicates $g0$ or σ remained constant. Density covariates included stream density and proportion of area in primary branch resting stands, denning stands, high-intensity burn stands, stands with wet soils, and spruce-leading (Sx -leading) or aspen- or cottonwood-leading (At -leading) stands near wetland.

within 100 m of a wetland also being well-supported (Table 4). 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 km²) and home range (30 km²) scales (Table 4). The second-ranked male-only model that included area of primary branch resting stands and length of streams within both the 4.84-km² and 30-km² buffer areas also had good support from the data (i.e., within 1.75 AIC_c units of the 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 for all fishers estimated an average density of 21.3 ± 3.8 fishers/1,000 km², or an abundance of 101.9 ± 18.0 fishers, across the entire 4,790-km² Chilcotin habitat mask area. The top-ranked male-only model estimated an average density of 5.2 ± 1.4 male fishers/1,000 km², or an abundance of 25.1 ± 6.7 male fishers, whereas the female-only model estimated an average density of 15.3 ± 4.3 female fishers/1,000 km², or an abundance of 73.2 ± 20.6 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-km² (Table 5; β = 10.9; 95% CI = 5.7–16.1) and the 30-km² (Figure 2; Table 5; β = 13.8; 95% CI = 4.8–22.9) buffer areas. Density in the female-only model was positively related to the area of stands with wet soil moisture regimes within both the 4.84-km² (Table 5; β = 4.9; 95% CI = 0.9–8.8) and the 30-km² buffer areas (Figure 2; Table 5; β = 4.9; 95% CI = 0.1–9.7). The top-ranked male-only model also was positively related to the area of branch resting stands within both the 4.84-km² (Table 5; β = 15.2; 95% CI = 5.7–16.1) and the 30-km² buffer areas (Figure 2; Table 5; β = 21.6; 95% CI = 10.2–33.0).

We 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,

TABLE 5 Beta coefficients (β) and associated 95% confidence intervals (CI) for the top-ranked density models for fishers (*Pekania pennanti*) at both the core use area scale (4.84 km²) and the home range scale (30 km²) in the Chilcotin (2018–2019) and Enterprise (2020–2021) study areas in the central interior of British Columbia, Canada. We estimated 3 parameters: density (*D*), detection probability (*g0*), and a spatial parameter (*sigma*).

Study area	Sex	Scale	Model ^a	β	95% CI	
					Lower	Upper
Chilcotin	Both	Core use	<i>D</i> ~ primary branch resting, <i>g0</i> ~ <i>B</i> , <i>sigma</i> ~ 1	10.9	5.7	16.1
		Home range	<i>D</i> ~ primary branch resting, <i>g0</i> ~ <i>B</i> , <i>sigma</i> ~ 1	13.8	4.8	22.9
	Female	Core use	<i>D</i> ~ stands with wet soil moisture, <i>g0</i> ~ <i>B</i> , <i>sigma</i> ~ 1	4.9	0.9	8.8
		Home range	<i>D</i> ~ stands with wet soil moisture, <i>g0</i> ~ <i>B</i> , <i>sigma</i> ~ 1	4.9	0.1	9.7
	Male	Core use	<i>D</i> ~ primary branch resting, <i>g0</i> ~ <i>k</i> , <i>sigma</i> ~ 1	15.1	5.7	16.1
		Home range	<i>D</i> ~ primary branch resting, <i>g0</i> ~ <i>k</i> , <i>sigma</i> ~ 1	21.6	10.2	33.0
Enterprise	Both	Core use	<i>D</i> ~ stands with wet soil moisture, <i>g0</i> ~ 1, <i>sigma</i> ~ 1	15.8	6.8	24.7
		Home range	<i>D</i> ~ crown closure >50%, <i>g0</i> ~ 1, <i>sigma</i> ~ 1	5.1	1.5	8.8

^aDetection covariates included whether there was a detection on the previous occasion (*B*) and site effectiveness changes once any animal is caught (*k*). A 1 indicates *g0* or *sigma* remained constant. Density covariates included proportion of area in primary branch resting stands, stands with wet soils, and stands with crown closure >50%.

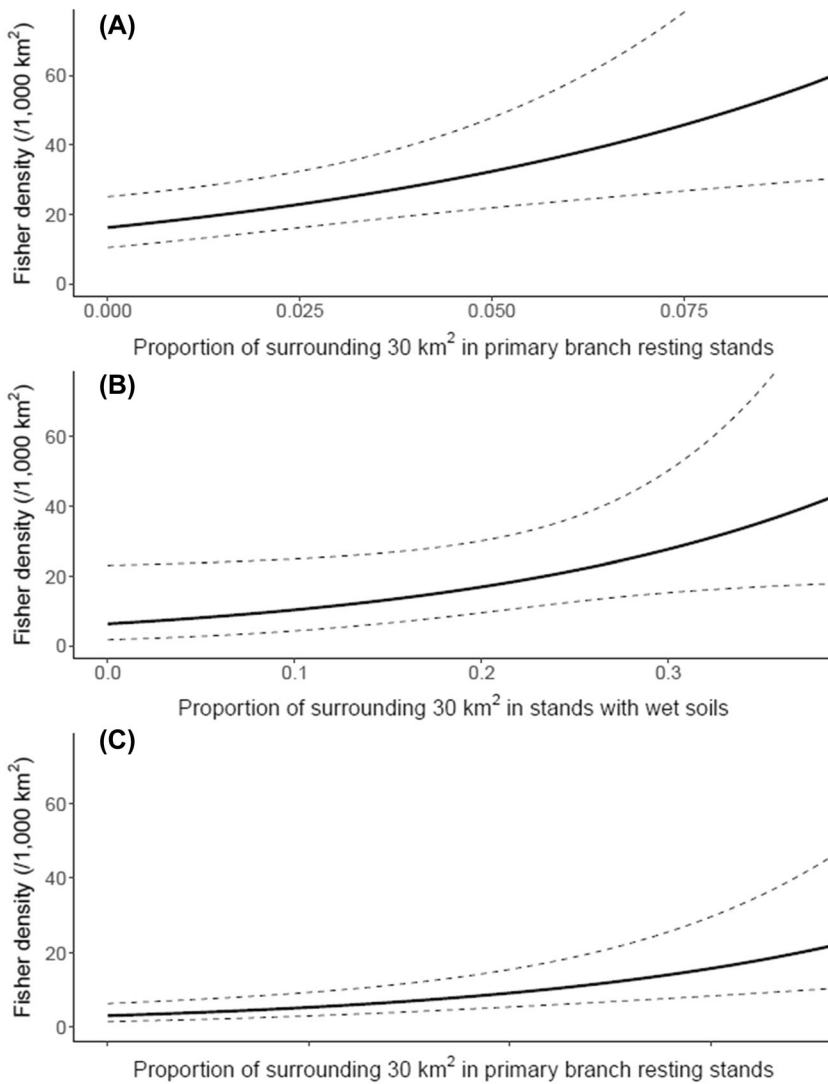


FIGURE 2 Relationship between density of fishers (*Pekania pennanti*) and variables included in the top-ranked models for the Chilcotin study area in 2018–2019 for data pooled across sexes (A), for females only (B), and for males only (C); dashed lines represent the upper and lower 95% confidence intervals.

in the northwest and northeast corners, and a handful of small areas farther south (Tables S4–S5). The top-ranked female-only model also predicted high densities in the study area and in the northwest corner but also with more higher density pockets distributed across the remainder of the area (Table S6).

Enterprise study area

We 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 our survey began, with a total of 9 unique fisher individuals killed by trappers operating within the study area during the 2020–2021 fur harvest season. We were unable to assess whether 4 other trapped fishers were detected in our sampling because we did not have samples for genetic identification. Because we could not determine the identity of all individuals or the timing and location of the fishers killed by trappers and because they were no longer part of the population, we did not include these 13 animals in our density calculations. However, the number of unique individuals we did identify indicated that between 30–38% of the fishers occupying the Enterprise study area during winter 2020–2021 were killed via trapping.

We documented 21 individuals recaptured at different locations only 9 times and therefore did not attempt to model each sex separately and report results from models pooled across sexes. The top-ranked detection model included a behavioral effect where the spatial scale parameter σ depended on whether a fisher was detected on the preceding occasion (Table 3). The null model, which assumed both detection probability and σ remained constant, also performed well ($\Delta AIC_c < 2.0$), and we selected this model to retain for future fitting because 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 and the spatial parameter σ was large at $4.67 \text{ km} \pm 0.0$. This model also estimated that the population had a male-to-female ratio of 29 ± 9.9 males to 71 ± 9.9 females.

We 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., we included the null detection model in the second step of the analysis). At both spatial scales we investigated (i.e., female core use area and home range, 4.84 km^2 and 30 km^2 , respectively), the top 4 models contained the same covariates and were well-supported ($\Delta AIC_c < 2.0$), but their ranking order slightly differed between scales (Tables S10–S11). 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 subhydryc soils; Table 6; $\beta = 10.9$; 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%.

TABLE 6 Model selection for the top-ranked density models for fishers (*Pekania pennanti*; both sexes combined) in the Enterprise (2020–2021) study area in the central interior of British Columbia, Canada. We estimated 3 parameters, density (D), detection probability ($g0$), 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 likelihood ($\log\text{Lik}$), Akaike's Information Criterion corrected for small sample sizes (AIC_c), the difference in AIC_c values (ΔAIC_c), and relative model weight (w_i). Models $\leq 2.0 AIC_c$ units of the top-ranked models are displayed, along with the null model that assumes all parameters are constant.

Scale	Model ^a	K	$\log\text{Lik}$	AIC_c	ΔAIC_c	w_i
Core use	$D \sim$ stands with wet soil moisture, $g0 \sim 1$, sigma ~ 1	5	-147.8	309.64	0.00	0.50
	Null	4	-152.3	315.08	5.44	0.03
Home range	$D \sim$ crown closure >50%, $g0 \sim 1$, sigma ~ 1	5	-149.2	312.45	0.00	0.22
	$D \sim$ open areas, $g0 \sim 1$, sigma ~ 1	5	-149.3	312.52	0.06	0.21
	$D \sim$ stands with wet soil moisture, $g0 \sim 1$, sigma ~ 1	5	-149.8	313.60	1.15	0.12
	$D \sim$ primary movement, $g0 \sim 1$, sigma ~ 1	5	-150.0	314.02	1.57	0.10
	Null	4	-152.3	315.08	2.63	0.06

^aA 1 indicates $g0$ or sigma remained constant. Density covariates included proportion of area in stands with wet soils, stands with crown closure >50%, open areas, and primary movement stands.

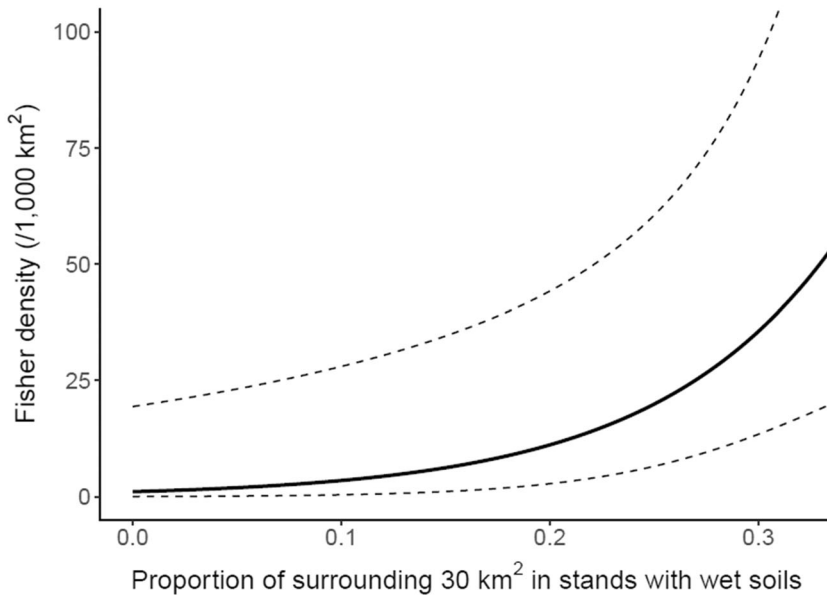


FIGURE 3 Modeled fisher (*Pekania pennanti*) density (pooled across sexes) in stands with wet soil moisture regimes (i.e., wetter than mesic) within Interior Douglas-fir forest types of the Enterprise study site in central interior British Columbia in 2020–2021.

We 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 2 scales with an average of 8.9 ± 2.7 fishers/1,000 km² estimated at the core use scale and 8.5 ± 2.7 fishers/1,000 km² estimated at the home range scale. These density estimates translated into abundance estimates of 52.8 ± 16.1 fishers and 50.5 ± 15.8 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 3). This model predicted a relatively low density (<10 fishers/1,000 km²) over most of the extrapolated area, with a few localized areas of higher densities in the center and along the southern boundary (Figure S7).

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 (~ 16.3 fishers/1,000 km²; Weir et al. 2011), whereas the estimates from the Enterprise study area were among the lowest (~ 8.8 fishers/1,000 km²; Weir and Corbould 2006). Yet these density estimates were substantially lower than those reported from eastern North America (50–327 fishers/1,000 km²; Powell and Zielinski 1994) and California (140 fishers/1,000 km², Matthews et al. 2011; 108 fishers/1,000 km², Green et al. 2022). The reason behind these differences in the density of fishers within and beyond British Columbia is not clear but has been hypothesized to relate to mesocarnivore community dynamics (D. Hodder, Director of Research and Education, John Prince Research Forest, personal communication), differences in prey availability (Weir et al. 2011), and snow conditions (Raine 1983), which differ regionally within British Columbia and 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 2 study areas. Both 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 a mountain pine beetle epidemic that resulted in millions of hectares of pine trees being killed and salvage harvested at accelerated rates (Eng et al. 2005). Nearly a 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 British Columbia (Province of British Columbia 2017), yet fishers were detected making forays into the more heavily burned areas and along the fringes of the fire boundary (R. D. Weir, Artemis Wildlife Consultants, unpublished data). We 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 observed lack of response to recently burned areas may be related to how soon after the fire we surveyed (i.e., within 1 year) and effects from this disturbance may be lagging, similar to the 1-year lag in post-fire population decline of fishers seen in California (Green et al. 2022).

One explanation behind the difference in the density of fishers estimated for the 2 study areas involved the relative levels of human-caused mortality during each survey season. The only trapline overlapping the Chilcotin study area that documented fisher harvest during the 2018–2019 season reported 3 animals being trapped (Province of British Columbia unpublished data: British Columbia Wild Fur Harvest Database 2000–2019, accessed 15 July 2022) with no other fishers reported from the Chilcotin study area, suggesting overall trapping pressure was low at the time of our survey. This aligns with the findings of Davis (2009) who stated that fishers in the Chilcotin area generally experience low levels of trapping pressure and was further supported by the low numbers of fishers trapped on traplines overlapping the Chilcotin study area since 2000 (Province of British Columbia unpublished data: British Columbia Wild Fur Harvest Database 2000–2019, accessed 15 July 2022). Conversely, in the Enterprise study area, we were able to confirm via genetic analysis or timing of capture that 9 unique fisher individuals were killed by trappers during the winter of our survey, which affected our density and abundance estimates.

New research from Weir et al. (2024) has shown that within British Columbia there are not only 2 genetically distinct populations, but within the Columbian population there also are 3 genetically clustered subpopulations. These subpopulations are separated by 2 large rivers (Fraser and Nechako Rivers, respectively), with low gene flow across these 2 barriers (Weir et al. 2024). The Chilcotin subpopulation is located on the west side of the Fraser River and the Cariboo subpopulation is located on the east side of the Fraser River, meaning fishers in our 2 study areas belonged to different subpopulations. There may be genetic differences between fishers in the Chilcotin and Enterprise study areas that are affecting population density, and future research should explore this topic further.

Overall, we found that the density of fishers in the Chilcotin study area was positively associated with area of primary branch resting habitat. When the proportion of primary branch resting stands in the surrounding 30 km² increased from 0 to 0.05, the density of fishers was predicted to double from 16 fishers/1,000 km² to 32 fishers/1,000 km². When not actively hunting or traveling, fishers use protected resting sites to conserve energy, avoid predation, thermoregulate, and safely consume prey (Lofroth et al. 2010, Raley et al. 2012). Primary branch resting habitat in the drier ecological regions of British Columbia consists of older (>83 years), large-diameter spruce stands with relatively high amounts (>40%) of crown closure (Weir and Almuedo 2010). These results were supported by numerous other studies in British Columbia and western North America that show fishers use large-diameter spruce trees relatively often for resting (Weir and Harestad 2003, 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. 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), with density models including primary denning habitat and spruce- or aspen-leading stands within 100 m of a wetland also well-supported. 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 by the presence of hybrid spruce, a more well-developed shrub layer, and a lesser abundance of lichens (Meidinger and

Pojar 1991), 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, 2004). Protecting and reserving these older, large-diameter spruce and aspen stands adjacent to riparian features will contribute to the retention of 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 British Columbia.

These results demonstrate that throughout the drier ecological zones of interior British Columbia, wetter stand types are associated with higher densities of fishers. Similar to the Chilcotin results, our 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-km² area increased from 0.1 to 0.3, the predicted density of fishers increased from 1 fisher/1,000 km² to 21 fishers/1,000 km². Although the Enterprise study area was not quite as dry as the Chilcotin, it is also considered one of the drier ecosystems where fishers are found in British Columbia (BC Fisher Habitat Working Group 2021). Stands with these higher soil moisture levels in the Enterprise study area were 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 provided higher densities of denning and resting structures and prey similar to their counterparts in other drier ecological zones of interior British Columbia (e.g., the Chilcotin Plateau), making the positive association we found with higher density of fishers consistent with other areas of the province. Protecting riparian stands from disturbance likely is important to help ensure the structures fishers need to fulfill their life requisites remain available in sufficient quantities on the landscape.

While our model selection results generally supported habitat associations suggested by previous research, one 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 our study areas. Throughout their range, fishers consistently occupy areas with overhead cover and avoid more open areas (Powell and Zielinski 1994, Weir 2003, Lofroth et al. 2010, Raley et al. 2012). In north-central British Columbia, 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, USA, found nearly the same relationship (Sauder and Rachlow 2015). When we included open areas as a covariate in our density models, the relationship between the density of fishers was slightly positive in the Enterprise study area, and this model ranked within the top 4 from the candidate set. We hypothesize that in the drier ecological zones of British Columbia, the Weir and Corbould (2010) model may be confounded because it includes area of wetlands: the density of fishers in these areas appeared to be strongly related to forested stands that were often found along the fringes of wetlands, and these stands likely also represent the highest value fisher habitat within these regions.

Extrapolating our density models to areas larger than the study areas that were similar ecologically was informative but also comes with important limitations, and we caution against using these models to estimate the total abundance of fishers for these larger areas. Our trapping arrays only covered approximately 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 our 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. However, these density models built on and supported previous research regarding the habitat associations of fishers in central interior British Columbia and could be used to identify priority areas for habitat protections, areas to direct future population monitoring efforts, and areas of the province where fishers may be enduring at higher densities.

CONSERVATION IMPLICATIONS

This study has provided land managers with important information regarding the distribution and abundance of fishers within the endangered Columbian population in the central interior of British Columbia, Canada, and the methods we used can be applied to other small and isolated populations facing the same cumulative threats of habitat loss, fragmentation, and degradation, human-caused mortality, and climate change. Population density estimates reported for these 2 study areas were among the highest and lowest ever documented in British Columbia, which has important implications for the future conservation and management of this population. Our results suggest that different levels of trapping mortality may have substantially affected the density and abundance estimates in our 2 study areas and future research should look at using empirical data on other sources of mortality to specifically evaluate these potential effects. Species such as fishers with low reproductive rates, large home ranges, and limited distributions can rarely sustain additive mortality from trapping (Banci and Proulx 1999), and it may be necessary to adjust trapping regulations to decrease the incidental take of fishers within the range of the Columbian population to prevent local extirpation (Fogarty et al. 2022). The Chilcotin region of central British Columbia appeared 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 subpopulation persists into the future should be a priority for wildlife managers.

Density estimates from our 2 study areas were 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. It can be very challenging for wildlife managers to ensure that small and isolated populations existing at low densities like this one that also face multiple threats (e.g., habitat loss, human-caused mortality, industrial development) continue to remain viable. The density of fishers had strong positive associations to similar habitat characteristics in both study areas, which are identifiable using existing spatial data layers and information from the interior of British Columbia. We 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 of the state of 2 fisher population clusters in representative ecosystems from central British Columbia, 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.

ACKNOWLEDGMENTS

Funding for this study was generously provided by the British Columbia Ministry of Environment & Climate Change Strategy and the British Columbia Ministry of Forests, Lands, Natural Resource Operations & Rural Development through the Conservation Economic Stimulus Initiative administered by the Habitat Conservation Trust Foundation, and the Together for Wildlife Strategy.

CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

ETHICS STATEMENT

All field sampling procedures used in this work were approved by the Thompson Rivers University Animal Care Committee (file 102619).

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

ORCID

Rory D. Fogarty  <http://orcid.org/0009-0007-5354-9530>

Richard D. Weir  <http://orcid.org/0000-0002-5381-8514>

Karl W. Larsen  <http://orcid.org/0000-0003-3013-1653>

REFERENCES

- Arthur, S. M., W. B. Krohn, and J. R. Gilbert. 1989. Home range characteristics of adult fishers. *Journal of Wildlife Management* 53:674-679.
- Aubry, K. B., and C. M. Raley. 2006. Ecological Characteristics of Fishers (*Martes pennanti*) in the southern Oregon Cascade Range. USDA Forest Service, Olympia, Washington, D.C., USA.
- Banci, V., and G. Proulx. 1999. Resiliency of furbearers to trapping in Canada. Pages 175-203 in G. Proulx, editor. *Mammal trapping*. Alpha Wildlife Research & Management, Sherwood Park, Alberta, Canada.
- Basham, J. T. 1958. Decay of trembling aspen. *Canadian Journal of Botany* 36:491-505.
- BC Fisher Habitat Working Group. 2021. British Columbia Fisher Habitat and Forestry Web Module. BC Fisher Habitat Working Group, Victoria, Canada. <https://www.bcfisherhabitat.ca/>
- Berg, S. S., J. D. Erb, V. T. Spaid, D. L. Dewey, P. L. Coy, B. A. Sampson, J. R. Fieberg, T. W. Arnold, and J. D. Forester. 2020. The role of local cavity tree density in the selection of den sites by female fishers (*Pekania pennanti*) in northern Minnesota. *Canadian Journal of Forest Research* 50:742-750.
- Borchers, D. L., and M. G. Efford. 2008. Spatially explicit maximum likelihood methods for capture-recapture studies. *Biometrics* 64:377-385.
- Bowman, J., D. Donovan, and R. C. Rosatte. 2006. Numerical response of fishers to synchronous prey dynamics. *Journal of Mammalogy* 87:480-484.
- British Columbia Conservation Data Centre. 2020a. Conservation status report: *Pekania pennanti* pop. 4. <https://a100.gov.bc.ca/pub/eswp/>. Accessed 2 Jun 2021.
- British Columbia Conservation Data Centre. 2020b. Conservation status report: *Pekania pennanti* pop. 5. <https://a100.gov.bc.ca/pub/eswp/>. Accessed 6 Nov 2020.
- Burnham, K., and D. Anderson. 2002. *Model selection and multimodel inference - a practical information-theoretic approach*, second edition. Springer-Verlag, New York, New York, USA.
- Carroll, C., R. F. Noss, and P. C. Paquet. 2001. Carnivores as focal species for conservation planning in the Rocky Mountain region. *Ecological Applications* 11:961-980.
- Davis, R. L. 2003. Stand level habitat use by furbearer species in the Anahim Lake area of British Columbia. Yun Ka Whu'ten Holdings Ltd., Williams Lake, British Columbia, Canada.
- Davis, R. L. 2004. Yun Ka Whu'ten Furbearer Track Transect Project - 2003/2004 Summary Report. Yun Ka Whu'ten Holdings Ltd., Williams Lake, British Columbia, Canada.
- Davis, R. L. 2009. Denning ecology and habitat use by fisher (*Martes pennanti*) in pine dominated ecosystems of the Chilcotin Plateau. Simon Fraser University, Burnaby, British Columbia, Canada.
- Davis, R. L., and R. D. Weir. 2021. Density of fishers (*Pekania pennanti*) at the southwestern edge of the species' range in British Columbia. *Northwestern Naturalist* 102:232-238.
- Dormann, C. F., J. Elith, S. Bacher, C. Buchmann, G. Carl, G. Carré, J. R. G. Marquéz, B. Gruber, B. Lafourcade, P. J. Leitão, et al. 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography* 36:27-46.
- Efford, M. G. 2021. Secr 4.4 - spatially explicit capture-recapture in R. <https://www.otago.ac.nz/density/pdfs/secr-overview.pdf>. Accessed 5 Jul 2021.
- Efford, M. G., D. L. Borchers, and A. E. Byrom. 2009. Density estimation by spatially explicit capture-recapture: likelihood-based methods. Pages 255-269 in D. L. Thompson, E. G. Cooch, and M. J. Conroy, editors. *Modeling demographic processes in marked populations*. Springer, New York, New York, USA.
- Efford, M. G., and G. Mowat. 2014. Compensatory heterogeneity in spatially explicit capture-recapture data. *Ecology* 95: 1341-1348.
- Eng, M., A. Fall, J. Hughes, T. Shore, B. Riel, P. Hall, and A. Walton. 2005. Provincial level projection of the current mountain pine beetle outbreak: an overview of the model and results of year 3 of the project. Natural Resources Canada, Ottawa, Canada.

- Fogarty, R. D. 2023. Forestry, fire and fur: factors driving the decline of fishers (*Pekania pennanti*) in central interior British Columbia. Thesis, Thompson Rivers University, Kamloops, British Columbia, Canada.
- Fogarty, R. D., R. D. Weir, E. C. Lofroth, and K. W. Larsen. 2022. Trapping mortality accelerates the decline of the fisher, an endangered mesocarnivore, in British Columbia, Canada. *Endangered Species Research* 49:1-12.
- Foran, D. R., S. C. Minta, and K. S. Heinemayer. 1997. DNA-based analysis of hair to identify species and individuals for population research and monitoring. *Wildlife Society Bulletin* 25:840-847.
- Fuller, A. K., D. W. Linden, and J. A. Royle. 2016. Management decision making for fisher populations informed by occupancy modeling. *Journal of Wildlife Management* 80:794-802.
- Fuller, T. K., E. C. York, S. M. Powell, T. A. Decker, and R. M. DeGraaf. 2001. An evaluation of territory mapping to estimate fisher density. *Canadian Journal of Zoology* 79:1691-1696.
- Gabriel, M. W., L. W. Woods, G. M. Wengert, N. Stephenson, J. M. Higley, C. Thompson, S. M. Matthews, R. A. Sweitzer, K. Purcell, R. H. Barrett, et al. 2015. Patterns of natural and human-caused mortality factors of a rare forest carnivore, the fisher (*Pekania pennanti*) in California. *PLoS One* 10:1-19.
- Garant, Y., and M. Crete. 1997. Fisher, *Martes pennanti*, home range characteristics in a high density untrapped population in southern Quebec. *Canadian Field-Naturalist* 111:359-364.
- Government of Canada. 2024. Mountain pine beetle (factsheet). Natural Resources Canada, Ottawa, Canada. <https://natural-resources.canada.ca/forests/fire-insects-disturbances/top-insects/13397>
- Green, D. S., A. N. Facka, K. P. Smith, S. M. Matthews, and R. A. Powell. 2022. Evaluating the efficacy of reintroducing fishers (*Pekania pennanti*) to a landscape managed for timber production. *Forest Ecology and Management* 511:120089.
- Green, R. E., K. L. Purcell, C. M. Thompson, D. A. Kelt, and H. U. Wittmer. 2019. Microsites and structures used by fishers (*Pekania pennanti*) in the southern Sierra Nevada: a comparison of forest elements used for daily resting relative to reproduction. *Forest Ecology and Management* 440:131-146.
- Greenhorn, J. E., J. Bowman, and P. J. Wilson. 2018. Genetic monitoring suggests increasing structure following recolonization by fishers. *Journal of Wildlife Management* 82:1403-1416.
- Hapeman, P., E. K. Latch, J. A. Fike, O. E. Rhodes, and C. W. Kilpatrick. 2011. Landscape genetics of fishers (*Martes pennanti*) in the Northeast: dispersal barriers and historical influences. *Journal of Heredity* 102:1-9.
- Hope, G. D., W. R. Mitchell, D. A. Lloyd, W. R. Erickson, W. L. Harper, and B. M. Wikeem. 1991. Interior Douglas-fir Zone. Pages 153-166 in D. V. Meidinger, and J. Pojar, editors. *Ecosystems of British Columbia*. British Columbia Ministry of Forests Special Report Series 6, Victoria, Canada.
- Jimenez, J., J. C. Nunez-Arjona, C. Rueda, L. M. Gonzalez, F. Garcia-Dominguez, J. Munoz-Igualada, and J. V. Lopez-Bao. 2017. Estimating carnivore community structures. *Scientific Reports* 7:41036.
- Johnson, W. E., and S. J. O'Brien. 1997. Phylogenetic reconstruction of the Felidae using 16S rRNA and NADH-5 mitochondrial genes. *Journal of Molecular Evolution* 44:S98-S116.
- Jordan, M. J., J. M. Higley, S. M. Matthews, O. E. Rhodes, M. K. Schwartz, R. H. Barrett, and P. J. Palsbøll. 2007. Development of 22 new microsatellite loci for fishers (*Martes pennanti*) with variability results from across their range. *Molecular Ecology Notes* 7:797-801.
- Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography of a recovering grizzly bear population. *Journal of Wildlife Management* 73:3-16.
- Krohn, W. B. 2012. Distributional changes of American martens and fishers in eastern North America, 1699-2001. Pages 58-73 in K. B. Aubry, W. J. Zielinski, M. G. Raphael, G. Proulx, and S. W. Buskirk, editors. *Biology and conservation of martens, sables, and fishers: a new synthesis*. Cornell University Press, Ithaca, New York, USA.
- Krohner, J. M. 2020. Finding fishers: determining the distribution of a rare forest mesocarnivore in the northern Rocky Mountains. Thesis, University of Montana, Missoula, USA.
- Laliberte, A. S., and W. J. Ripple. 2004. Range contractions of the North American carnivores and ungulates. *Bioscience* 54: 123-138.
- LaPoint, S. D., J. L. Belant, and R. W. Kays. 2015. Mesopredator release facilitates range expansion in fisher. *Animal Conservation* 18:50-61.
- Lewis, J. C., K. J. Jenkins, P. J. Happe, D. J. Manson, and M. McCalmon. 2016. Landscape-scale habitat selection by fishers translocated to the Olympic Peninsula of Washington. *Forest Ecology and Management* 369:170-183.
- Lewis, J. C., R. A. Powell, and W. J. Zielinski. 2012. Carnivore translocations and conservation: insights from population models and field data for fishers (*Martes pennanti*). *PLoS One* 7:1-15.
- Linden, D. W., A. K. Fuller, J. A. Royle, M. P. Hare, and J. Frair. 2017. Examining the occupancy-density relationship for a low-density carnivore. *Journal of Applied Ecology* 54:2043-2052.
- Lofroth, E. C. 2004. Fisher (*Martes pennanti*) British Columbia Population Science Review. Ministry of Water, Land and Air Protection, Victoria, British Columbia, Canada.

- Lofroth, E. C., C. M. Raley, J. M. Higley, R. L. Truex, J. S. Yaeger, J. C. Lewis, P. J. Happe, and L. L. Finley. 2010. Conservation of fishers (*Martes pennanti*) in south-central British Columbia, Western Washington, Western Oregon, and California. Volume I: Conservation Assessment. Volume I. USDI Bureau of Land Management, Denver, Colorado, USA.
- Long, R. A., T. M. Donovan, P. MacKay, W. J. Zielinski, and J. S. Buzas. 2010. Predicting carnivore occurrence with noninvasive surveys and occupancy modeling. *Landscape Ecology* 26:327-340.
- Luo, G., W. Wei, Q. Dai, and J. Ran. 2020. Density estimation of unmarked populations using camera traps in heterogeneous space. *Wildlife Society Bulletin* 44:173-181.
- Matthews, S. M., D. S. Green, J. M. Higley, K. M. Rennie, C. M. Kelsey, and R. E. Green. 2019. Reproductive den selection and its consequences for fisher neonates, a cavity-obligate mustelid. *Journal of Mammalogy* 100:1305-1316.
- Matthews, S. M., J. M. Higley, J. S. Yaeger, and T. K. Fuller. 2011. Density of fishers and the efficacy of relative abundance indices and small-scale occupancy estimation to detect a population decline on the Hoopa Valley Indian Reservation, California. *Wildlife Society Bulletin* 35:69-75.
- Meidinger, D. V., and J. Pojar. 1991. *Ecosystems of British Columbia*. British Columbia Ministry of Forests Special Report Series 6, Victoria, Canada.
- Mowat, G., A. P. Clevenger, A. D. Kortello, D. Hausleitner, M. Barrueto, L. Smit, C. Lamb, B. DorsEy, and P. K. Ott. 2019. The sustainability of wolverine trapping mortality in southern Canada. *Journal of Wildlife Management* 84:213-226.
- Mowat, G., and D. Paetkau. 2002. Estimating marten (*Martes americana*) population size using hair capture and genetic tagging. *Wildlife Biology* 8:201-209.
- Obbard, M. E., E. J. Howe, and C. J. Kyle. 2010. Empirical comparison of density estimators for large carnivores. *Journal of Applied Ecology* 47:76-84.
- Paetkau, D. 2003. An empirical exploration of data quality in DNA-based population inventories. *Molecular Ecology* 12: 1375-1387.
- Powell, R. A. 1993. *The fisher - life history, ecology & behavior*. Second edition. University of Minnesota Press, Minneapolis, USA.
- Powell, R. A., and W. J. Zielinski. 1994. Fisher. Pages 38–73 in L. F. Ruggiero, K. B. Aubry, S. W. Buskirk, L. J. Lyon, and W. J. Zielinski, editors. *The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the western United States*. US Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, USA.
- Proulx, G., K. Aubry, J. Birks, S. Buskirk, C. Fortin, H. Frost, W. Krohn, L. Mayo, V. Monakhov, and D. Payer. 2004. World distribution and status of the genus *Martes* in 2000. Pages 21-76 in D. J. Harrison, A. K. Fuller, and G. Proulx, editors. *Martens and fishers (Martes) in human-altered environments*. Springer, New York, New York, USA.
- Province of British Columbia. 2017. *Wildfire season summary 2017*. Province of British Columbia, Victoria, Canada.
- Province of British Columbia. 2018. *Wildfire season summary 2018*. Province of British Columbia, Victoria, Canada.
- Province of British Columbia. 2020. *Hunting and trapping regulations synopsis 2020 - 2022*. Province of British Columbia, Victoria, Canada.
- Province of British Columbia. 2021. *BC data catalogue*. Province of British Columbia, Victoria, Canada.
- Purcell, K. L., A. K. Mazzoni, S. R. Mori, and B. B. Boroski. 2009. Resting structures and resting habitat of fishers in the southern Sierra Nevada, California. *Forest Ecology and Management* 258:2696-2706.
- R Core Team. 2020. *R: a language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria.
- Raine, R. M. 1983. Winter habitat use and responses to snow cover of fisher (*Martes pennanti*) and marten (*Martes americana*) in southeastern Manitoba. *Canadian Journal of Zoology* 61:25-34.
- Raley, C. M., E. C. Lofroth, R. L. Truex, J. S. Yaeger, and J. M. Higley. 2012. Habitat ecology of fishers in North America. Pages 231–254 in K. B. Aubry, W. J. Zielinski, M. G. Raphael, G. Proulx, and S. W. Buskirk, editors. *Biology and conservation of martens, sables and fishers: a new synthesis*. Cornell University Press, Ithaca, New York, USA.
- Ripple, W. J., J. A. Estes, R. L. Beschta, C. C. Wilmers, E. G. Ritchie, M. Hebblewhite, J. Berger, B. Elmhagen, M. Letnic, M. P. Nelson, et al. 2014. Status and ecological effects of the world's largest carnivores. *Science* 343:1241484.
- Royle, J. A., A. J. Magoun, B. Gardner, P. Valkenburg, and R. E. Lowell. 2011. Density estimation in a wolverine population using spatial capture-recapture models. *Journal of Wildlife Management* 75:604-611.
- Sauder, J. D., and J. L. Rachlow. 2015. Forest heterogeneity influences habitat selection by fishers (*Pekania pennanti*) within home ranges. *Forest Ecology and Management* 347:49-56.
- Steen, O. A., and R. A. Coupe. 1997. *A field guide to forest site identification and interpretation for the Cariboo Forest Region*. British Columbia Ministry of Forests, Land Management Handbook No. 39, Victoria, Canada.
- Wei, T., V. Simko, M. Levy, Y. Xie, Y. Jin, and J. Zemla. 2017. *Corrplot: visualization of a correlation matrix*. <https://github.com/taiyun/corrplot>
- Weir, R. D. 2003. *Status of the fisher in British Columbia*. British Columbia Ministry of Sustainable Resource Management, Conservation Data Centre, and Ministry of Water, Land and Air Protection, Biodiversity Branch, Victoria, Canada.

- Weir, R. D., and P. L. Almuedo. 2010. British Columbia's Interior: Fisher Wildlife Decision Aid. *BC Journal of Ecosystems & Management* 10:35-41.
- Weir, R. D., and F. B. Corbould. 2006. Density of fishers in the Sub-Boreal Spruce biogeoclimatic zone of British Columbia. *Northwestern Naturalist* 87:118-127.
- Weir, R. D., and F. B. Corbould. 2008. Ecology of fishers in the Sub-boreal forests of north-central British Columbia, Final Report. Peace/Williston Fish and Wildlife Compensation Program, Prince George, British Columbia, Canada.
- Weir, R. D., and F. B. Corbould. 2010. Factors affecting landscape occupancy by fishers in north-central British Columbia. *Journal of Wildlife Management* 74:405-410.
- Weir, R. D., and A. H. Harestad. 2003. Scale-dependent habitat selectivity by fishers in south-central British Columbia. *Journal of Wildlife Management* 67:73-82.
- Weir, R. D., A. H. Harestad, and F. B. Corbould. 2009. Home ranges and spatial organization of fishers, *Martes pennanti*, in central British Columbia. *Canadian Field-Naturalist* 123:126-132.
- Weir, R. D., E. C. Lofroth, and M. Phinney. 2011. Density of fishers in boreal mixed wood forests of northeastern British Columbia. *Northwestern Naturalist* 92:65-68.
- Weir, R. D., E. C. Lofroth, M. Phinney, and L. R. Harris. 2013. Spatial and genetic relationships of fishers in boreal mixed-wood forests of northeastern British Columbia. *Northwest Science* 87:114-125.
- Weir, R. D., M. Phinney, and E. C. Lofroth. 2012. Big, sick, and rotting: Why tree size, damage, and decay are important to fisher reproductive habitat. *Forest Ecology and Management* 265:230-240.
- Weir, R. D., A. M. Rankin, L. Robinson, K. L. Pilgrim, M. K. Schwartz, and M. K. Lucid. 2024. Genetic structuring of fishers in British Columbia: implications for population conservation and management. *Journal of Mammalogy* 105:465-480.
- Wengert, G. M., M. W. Gabriel, S. M. Matthews, J. M. Higley, R. A. Sweitzer, C. M. Thompson, K. L. Purcell, R. H. Barrett, L. W. Woods, R. E. Green, et al. 2014. Using DNA to describe and quantify interspecific killing of fishers in California. *Journal of Wildlife Management* 78:603-611.

Associate Editor: Jonathan Gilbert.

SUPPORTING INFORMATION

Additional supporting material may be found in the online version of this article at the publisher's website.

How to cite this article: Fogarty, R. D., R. D. Weir, and K. W. Larsen. 2025. Forestry, fire, and fur: factors influencing densities of endangered fishers (*Pekania pennanti*) in central British Columbia, Canada. *Journal of Wildlife Management* e70010. <https://doi.org/10.1002/jwmg.70010>