



## Research Article

## The role of environmental variables and sympatric meso-carnivores on the detection and occupancy of American mink during winter

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### Abstract

The spatial distributions of animals generally are affected by the availability of food, competition, predators, mates, and the need to communicate with conspecifics. An understanding of a given species' spatial distribution is essential when considering the ecological requirements of populations as well as the impacts of anthropogenic activities and environmental change. The American mink (*Neovison vison*) is a cryptic, semi-aquatic carnivore that ranges over a large portion of North America yet the ecological role of the species is not well understood. We sought to investigate the linkages between habitat and species co-occurrence on the occupancy patterns of mink within riparian habitats during winter. We monitored mink using remote cameras (n=37) which were deployed in riparian habitat along streams including lakeshore/stream confluences. We found that fish-bearing streams positively affected mink occupancy, while the amount of older (>40 years) coniferous forests had a negative relationship with mink occupancy. We postulate that while mink seem to occur at high densities in altered ecosystems and in areas where they are invasive, in their native range these animals may be limited by environmental and competitive pressures in the system. Future work should explore the interactions between carnivore species in addition to habitat selection in order to develop more robust monitoring and management practices.

## Introduction

The distribution of a species on a landscape includes many factors such as food, competitors, predators, mates, and intraspecific communication (Powell, 2012). Thus, understanding fully the ecology of any population (much less impacts caused by environmental change) requires knowledge of how and why the animal is spatially distributed (Chelgren et al., 2015; Walpole et al., 2012; Poley et al., 2014). However, collecting this type of information can be particularly challenging for cryptic species. Many members of the Order Carnivora fall into this category, and hence they remain poorly understood and their conservation status not well known (Boitani and Powell, 2012). Traditionally, many carnivore population assessments have focussed on understanding population abundance or density (MacKenzie and Reardon, 2012). Alternatively, spatial distributions of these animals have been examined using marked animal locations and associated environmental variables to determine habitat selection patterns (Boyce et al., 2002; Johnson et al., 2006). However, these assessments are expensive and often impractical, particularly for species that have low economic value and/or no threat to populations. Recently, more emphasis has been placed on passive detection-nondetection surveys (i.e., remote cameras, hair traps, snow tracks etc.) to determine areas that are occupied by various carnivore species (MacKenzie et al., 2006; Long et al., 2011; Schooley et al., 2012). Noninvasive survey techniques eliminate the need for animal handling, and occupancy models can provide estimates of habitat use without the estimation of actual population parameters (Long et al., 2011). This approach is empirically-based and uses animal detections and ecological covariates to estimate occupancy patterns. Additionally, when paired with survey methods such as remote cameras that

detect multiple species, likelihood-based models can also explore patterns of species co-occurrence and site occupancy (MacKenzie et al., 2006; Burton et al., 2015). Although the consideration of other species (especially predator-prey relationships) is not new in resource selection models (Johnson et al., 2002; Anderson and Johnson, 2014; DeCesare et al., 2014) the incorporation of carnivore co-occurrence data in occupancy models using passive data collection techniques is relatively novel and has not been applied widely in field ecology studies.

American mink (*Neovison vison*) are cryptic, semi-aquatic carnivores (F. Mustelidae) and poorly understood across their native range (Schooley et al., 2012). Despite a long history of harvest and population management, little is published about mink ecology or population dynamics in North America (Schooley et al., 2012; Larivière, 2003). What is known is that the animals typically have distributions that are linear and associated with water features (Larivière, 2003). Across their range, mink consume a variety of fish, mammals, amphibians, birds and crustaceans but mammals such as muskrat (*Ondatra zibethicus*), mice and voles are reported to be the most important diet items for mink during all seasons (Eagle and Whitman, 1987); and in general, the species' diet reflects whatever is available in the local prey base (Larivière, 2003). Despite its semi-aquatic nature, mink are not particularly agile (in comparison to the more aquatic otter) in water and are limited to foraging for small or slow-moving prey in shallow water (Dunstone and Birks, 1987). In Illinois, however, Wolff et al. (2015) demonstrated that mink occupancy during summer in a largely agricultural landscape was influenced by the availability of preferred prey (crayfish). Other information on mink ecology is derived from research outside of their native range, where there has been substantial research on mink as an introduced species (Bonesi et al., 2004; Santulli et al., 2014; Fasola et al., 2009; Medina, 1997). In an effort to establish a baseline study on mink ecology, we assessed mink winter

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occupancy in a northern part of its native range where ecosystems are relatively undisturbed, flow of natural waterways are not regulated, and the only major landscape disturbance is ongoing forestry activities. We chose winter because cold winter temperatures can be a limiting factor to many endothermic mammals (Marchand, 2013) and home ranges were considered more stable during this season. Overall, we sought to investigate the linkages between habitat and species co-occurrence on the occupancy patterns of mink within riparian habitats. Our objectives were to (1) assess covariates that affect the detectability of American mink in non-invasive surveys, and (2) assess habitat and species co-occurrence covariates that affect mink occupancy patterns. Specifically, we hypothesized that mink would (1) be positively associated with fish-bearing streams, and (2) would be negatively affected by the presence of American marten (*Martes americana*).

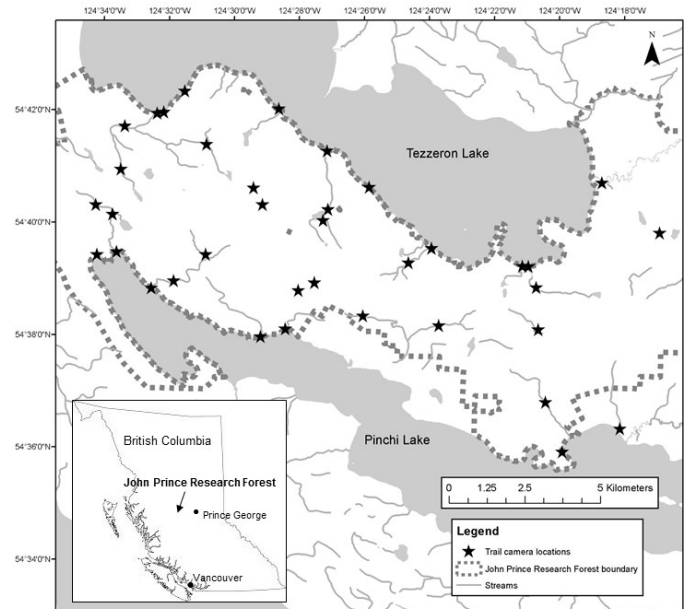
## Methods

### Study area

The research was conducted in the John Prince Research Forest (JPRF; Fig. 1 a 16,500 ha portion of forested public land 45 km northwest of Fort St. James, British Columbia, Canada. The JPRF is characterized by rolling terrain with low mountains (700 m to 1267 m a.s.l) and is within the Sub-Boreal Interior ecoprovince. The JPRF is situated between two large lakes, Tezzeron (8079 ha) and Pinchi (5586 ha), and has a relatively high density of streams. The area has experienced a wide variety of logging activities over the past 70 years and contains a mosaic of old and young coniferous forests with interspersed deciduous stands. The stands have a relatively rich understory of deciduous shrubs and regenerating conifers (see Hodder (2016) for a more detailed study area description).

### Camera Surveys

In winter 2013, we monitored mink with remote cameras which are commonly used to detect wildlife species (Burton et al., 2015). Specifically, we used Bushnell Trophy Cam (Model 119467) and Bushnell Trophy Cam HD Max (Model 119477) passive infrared cameras (Bushnell Outdoor Products, Kansas, USA). Both camera models had identical technical specifications and settings. We deployed 37 cameras in riparian habitat along streams including lakeshore/stream confluences throughout the JPRF. Camera stations were active for three 15 day sessions: January 26–Feb 9, March 5–19, and April 2–16. Riparian corridors were chosen for camera placement due to the semi-aquatic nature of mink and subsequent selection for habitats near riparian features (Hodder, 2016; Burton et al., 2015). We stratified camera locations by streams that were “fish bearing” (n=16) and “non-fish bearing” (n=21) with representation at junctions with lakeshore and upland habitats. The study area has a complete stream classification inventory as per the provincial standards in British Columbia (Forest Practices Code of British Columbia, 1998). Our cameras were not spatially independent in terms of a minimum distance between stations but instead were monitoring different sub-watersheds, an approach we feel appropriate given the linear nature of mink home ranges (Larivière, 2003). This method notwithstanding, 26 of the 37 sites were > 1 km apart. At each



**Figure 1** – Map of study area showing remote camera locations (n=37) in the John Prince Research Forest in central British Columbia, Canada..

site, a camera was set 0.5–1 m above the snow on a tree or fallen log. Bait and lure were set near the ground 2–3 meters from the camera. Bait was a combination of salmon paired with either beaver or moose meat and hung 0.5–1 m from the ground. In an effort to limit potential bias with bait we kept portions small (~ 80 grams) and low to the ground to minimize any broadcast effect. During the last two surveys, a small diameter log (<15 cm diameter) was added to the set and secured in the snow with one end pointing out directly below the bait (~ 30–40 cm below bait). The addition of this log served as a platform for mink to use as they approached the bait, allowing for better video captures and in turn, better verification of species and even individual markings. Commercial mink lure and beaver castor were placed directly above the bait as well as on the log or ground below the bait. Bait was replaced and additional lure added approximately mid-way through each session. Cameras were set to take 30 seconds of video with a 1 second delay between videos. This video schedule allowed for near continuous recording for the time the animal was in view. Sensor level was set to normal, LED control for night vision was set to medium, and video sound recording was turned on.

### Statistical Methods

We used likelihood-based occupancy modelling to evaluate the influence of covariates (Tab. 1) on detection and occupancy of American mink (MacKenzie et al., 2006; Long et. al, 2011; Shannon et al., 2014). Specifically, we used single-season occupancy models in PRESENCE (Version 7.1 USGS-PWRC. <http://www.mbr-rc.usgs.gov/software/presence.html>; Hines, 2006) to estimate detection rates (P;

**Table 1** – Independent variables used for detection and occupancy models for American mink in the John Prince Research Forest, central British Columbia, Canada.

Variable Name	Coding	Description
temperature	continuous	Mean temperature during 15 day trapping session.
Julian	continuous	Julian date to mid-point of survey
conifer	continuous	Percentage of forests in 100 m radius of camera trap location that was conifer leading and >40 yrs old.
deciduous	continuous	Percentage of forests in 100 m radius of camera trap location that was deciduous leading.
riparian	continuous	Amount of linear riparian habitat (m) in 100 m radius of camera trap location.
beaverlodge	continuous	Distance to the nearest beaver lodge.
fishbearing	categorical	Stream at camera trap location was fish bearing.
American marten	categorical	Marten detected at site during all three trap sessions.
weasel	categorical	Weasel detected at site during a trap session.
Canada lynx	categorical	Lynx detected at site during a trap session.

**Table 2** – Overall QAIC<sub>c</sub> model rankings with K (number of parameters), QAIC<sub>cΔi</sub> (difference from top model score), QAIC<sub>cWi</sub>, -2Log(L) (negative 2 log likelihood),  $\chi^2$  (Chi square value) and *p* ( $\chi^2$  associated *p*-value) for detection models of American mink in the John Prince Research Forest, central British Columbia, Canada.

Model	K	QAIC <sub>cΔi</sub>	QAIC <sub>cWi</sub>	-2Log(L)	$\chi^2$	<i>p</i>
$\psi(\cdot)$ , P(survey)	4	0	0,86	129,13	6,7	0,36
$\psi(\cdot)$ , P(julian+survey)	5	3,9	0,12	129,13	6,7	0,33
$\psi(\cdot)$ , P(temp+survey)	5	9,19	0,01	129,13	6,7	0,37
$\psi(\cdot)$ , P(marten+julian+survey)	6	9,54	0,01	129,13	6,42	0,37
$\psi(\cdot)$ , P(marten+survey)	5	12,6	<0,01	129,06	6,43	0,39
$\psi(\cdot)$ , P(temp+julian+survey)	6	13,7	<0,01	129,06	6,7	0,36
$\psi(\cdot)$ , P(marten+temp+survey)	6	17,5	<0,01	129,06	6,43	0,39

probability that a mink was detected if present) and site occupancy ( $\psi$ ; probability that a mink occupied the site) for multiple surveys of the same site (MacKenzie et al., 2006; Schooley et al., 2012). Estimation in PRESENCE assumes that there are no false positives (unlikely with camera data) and effectively copes with missing data (MacKenzie et al., 2006). We used single-season, single-species models because the original study design was stratified to survey mink habitat (i.e., riparian areas) and would not necessarily be representative of other species' habitats (e.g., marten). In addition, we considered our sample size (n=37) as insufficient to accommodate multi-species models with higher numbers of parameters.

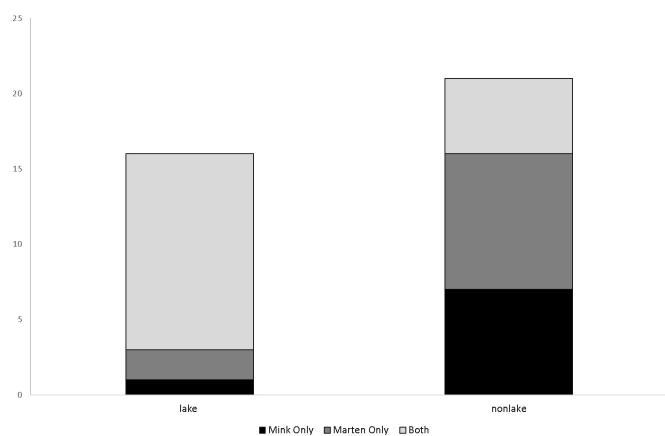
We chose variables (Tab. 1) that we considered important to detection and occupancy patterns of mink. Temperature can influence animal movements and may affect the detection of tracks and other signs (Kenda et al., 2005; Long et. al, 2011; Zimmerling, 2005) and Julian day can show within season effects. The amount of older (>40 years) conifer leading forests around each station was used to assess associations with old, complex forests and potentially other species that use those habitats (e.g., marten; Powell et al., 2003). The amount of riparian habitat available was used as mink have been shown to have strong riparian associations (Larivière, 2003). Areas with beaver ponds are often rich sources of biomass because of the complex habitat created by beaver structures (Gard, 1961; McDowell and Naiman, 1986). The status of a stream as being fishbearing or not was used because of food availability for mink (Eagle and Whitman, 1987) as well as the larger amount of riparian habitat available around fishbearing streams (Forest Practices Code of British Columbia, 1998). Species (American marten, weasels, Canada lynx) co-occurrence variables were used as the presence of potentially competing species could affect both detection and occupancy of mink (Burton et al., 2015). We developed sets of biologically plausible models for detection (n=7) and occupancy (n=15; Burnham and Anderson, 2002). For detection models, we used a constant occupancy probability while varying detection covariates paired with differences in survey period ( $\psi(\cdot)$ ,P(survey)). For occupancy models, we let the probability of detection (P) differ between

surveys but otherwise remain constant (without covariates) while varying habitat and carnivore co-occurrence covariates that may influence species occupancy ( $\psi$ ). We used the Akaike information criterion difference (AIC<sub>Δi</sub>) and associated weight (AIC<sub>cWi</sub>) for small sample sizes to rank the most parsimonious models (Anderson et al., 2000). We used parametric bootstrapping with 1000 permutations in PRESENCE to assess goodness-of-fit for detection and occupancy models (Kaiser and O'Keefe, 2015). We used  $\hat{c}$  as a measure of overdispersion and considered a value less than 1 as having acceptable model fit (MacKenzie and Bailey, 2004; Kaiser and O'Keefe, 2015). For those models with poor fit we used QAIC<sub>c</sub> (Quasi-AIC) as a correction to more accurately portray covariates in model rankings (MacKenzie and Bailey, 2004). We used model averaging (Anderson et al., 2000) to help represent the uncertainty inherent in the model selection process. We considered the 85% confidence intervals as a measure of significance for averaged beta coefficients of all models in the a priori set (Arnold, 2010). While our sample size was insufficient to withhold portions of the data to further investigate trends using occupancy models, we compared percent overlap between mink and marten at "lake" and "non-lake" sites as a way to further explore the potential relationship between mink and marten co-occurrence.

## Results

During the winter of 2013, there was a naïve occupancy rate (i.e., proportion of sites with detections of mink) of 0.65 based on remote camera surveys of riparian habitats in the study area. Mink were active during all times of day with detections being classed as "night" and "day" (based on light conditions observed in videos) during 49 and 51 percent of detections, respectively. We used variables from Tab. 1 to construct 22 models in PRESENCE. Models were constructed for detection (n=7; Tab. 2) and occupancy using habitat and species occurrence data (n=15; Tab. 3). For the detection models, all assessed models were deemed to have poor fit with  $\hat{c}$  values all greater than 1, suggesting some degree of overdispersion. As a result, we corrected the model rankings by using the QAIC scores. After this correction, the best model of detection probability (QAIC<sub>cWi</sub>=0.86) had no covariates and suggested that detection varied among survey periods. Overall, the detection probabilities for mink during the study were 0.61 (95% CI=0.37–0.81), 0.24 (95% CI=0.12–0.44), and 0.48 (95% CI=0.29–0.69) for Sessions 1, 2, and 3, respectively. The average detection rate for mink throughout the study was 0.44.

For the set of models testing the influence of habitat covariates and species co-occurrence on mink occupancy, all assessed models were deemed to have acceptable fit with  $\hat{c}$  values less than 1. There was considerable uncertainty in model selection with the top model having a low model weight (AIC<sub>cWi</sub>=0.21). Another 3 additional models had an AIC<sub>cΔi</sub><2 and therefore were considered equivalent models (Tab. 3). Of these top models, the conifer variable was included in all, with the variables fish-bearing, beaverlodge, and marten also exhibiting influence. After model averaging, the coefficient values suggested that mink occupancy had a negative relationship with conifer habitat and was positively associated with fish-bearing streams (Tab. 4). However, only the fish-bearing variable was significant with confidence intervals not overlapping zero. Using the model-averaged results the mean occupancy within riparian habitat across the study area for mink was 0.77



**Figure 2** – Spatial overlap of American marten and American mink detections at camera stations representing "lake" and "non-lake" locations in the John Prince Research Forest, central British Columbia, Canada.

**Table 3** – Overall QAIC<sub>c</sub> model rankings with K (number of parameters), QAIC<sub>cΔi</sub> (difference from top model score), QAIC<sub>cWi</sub> (model weight), -2Log(L) (negative 2 log likelihood),  $\chi^2$  (Chi square value) and  $p$  ( $\chi^2$  associated p-value) for occupancy models of American mink in the John Prince Research Forest, central British Columbia, Canada.

Model	K	QAIC <sub>cΔi</sub>	QAIC <sub>cWi</sub>	-2Log(L)	$\chi^2$	$p$
$\psi$ (conifer+beaver), P(survey)	6	0	0,21	117,65	6,29	0,74
$\psi$ (conifer), P(survey)	5	0,79	0,14	120,78	6,22	0,5
$\psi$ (fish+conifer), P(survey)	6	1,5	0,1	119,15	6,09	0,49
$\psi$ (marten+fish+conifer), P(survey)	7	1,92	0,08	117,17	6,01	0,53
$\psi$ (conifer+marten), P(survey)	6	2,07	0,07	119,72	6,17	0,48
$\psi$ (lynx+conifer+beaver), P(survey)	7	2,33	0,06	117,58	6,2	0,47
$\psi$ (marten+beaver+conifer), P(survey)	7	2,38	0,06	117,63	6,27	0,47
$\psi$ (weasel+beaver+conifer), P(survey)	7	2,38	0,06	117,63	6,3	0,45
$\psi$ (fish+beaver+conifer), P(survey)	7	2,4	0,06	117,65	6,29	0,76
$\psi$ (riparian+conifer), P(survey)	6	3,23	0,04	120,88	6,19	0,69
$\psi$ (fish+weasel+conifer), P(survey)	7	3,47	0,04	118,72	6,12	0,5
$\psi$ (lynx+fish+conifer), P(survey)	7	3,77	0,03	119,02	6,1	0,5
$\psi$ (marten+beaver+conifer+fish), P(survey)	8	3,95	0,03	116,72	5,96	0,52
$\psi$ (beaver), P(survey)	5	6,09	0,01	126,08	6,18	0,75
$\psi$ (riparian+beaver), P(survey)	6	6,38	0,01	124,03	6,21	0,71

(95% CI=0.73–0.80). When comparing “lake” versus “non-lake” sites, we found that 81.25% of sites in the lakeshore zone had both marten and mink detections, but the sites that were non-lakeshore detected both species at only 19.05% of sites (Fig. 2).

## Discussion

Our surveys of riparian areas for the presence of mink yielded relatively modest detection rates that varied between survey periods, with higher rates occurring earlier and later in winter. Naïve occupancy within riparian habitats was relatively high, which is consistent with findings of similar work using track surveys for mink in Illinois during summer (Schooley et al., 2012). There are no comparable data available that explore mink detection and occupancy during winter in its native range. In terms of occupancy patterns, we found cameras located at fish-bearing streams were more likely to detect mink. These results are comparable to the results of Hodder (2016), where telemetered mink were monitored in the same study area. It must be noted, however, that many of the streams in the study area are completely frozen during winter, so it may not be fish per se that are bringing mink into this riparian area, but rather the habitat structure or other characteristics. However, we also observed a negative trend (though not significant) between mink occupancy and the amount of older (>40 years) coniferous forests which could represent high-value marten habitat (Powell et al., 2003). Considering the differences in percent overlap between the lakeshore and non-lakeshore zones, this could support the negative association detected between marten presence and conifer habitat in the models. Males appear to preferentially select areas around lakeshores, whereas females favour areas near forest streams (see Hodder, 2016), suggesting a potential avoidance of marten habitat by female mink. To our knowledge there is no literature investigating the potential for competitive interactions between mink and marten. Despite our focus on monitoring mink activities in riparian habitats, there was still higher naïve occupancy for forest dwelling marten (0.77) than for the more riparian mink (0.65), suggesting considerable spatial overlap. Given that marten and mink are of similar size (400–1400 g vs. 500–1500 g, respectively), the potential for interactions between these two species is high, a potential effect that should be considered when studying either species in areas of sympatry. While no reliable data exist for mink diet in this region, an overlap with marten possibly could explain some of the indirect association between the two species, given that the latter primarily feeds on small mammals such as voles (Powell et al., 2003) while mink diet also can be dominated by mice and voles during all seasons (Eagle and Whitman, 1987). Additionally, mink (as with several other mustelids) are poorly adapted for extreme temperatures due to their elongated, fusiform body shape (Kruuk et al., 1994). This likely further restricts the diet and habitats accessible to mink in winter due to thermoregulatory limits on their ability to forage in aquatic environ-

ments (Kruuk et al., 1994). Hodder (2016) suggested this may be particularly critical for females due to their smaller body size. In addition to this constraint, much of the aquatic habitat available to mink in summer is unavailable during winter as a result of complete ice cover. We postulate that while mink seem to occur at high densities in altered ecosystems (Larivière, 2003) and in areas where they are invasive (Bonesi et al., 2004; Bonesi and Palazon, 2007), in their native range these animals are restricted by environmental extremes (low winter temperatures) and competitive pressures in the system.

Interspecific interactions have been documented between American mink and other mustelid species. Ben-David (1996) demonstrated that there was niche partitioning between mink and the North American River Otter (*Lontra canadensis*) in Alaska while mink appeared to be negatively affected by competition with Eurasian Otter (*Lutra lutra*) in Europe (Erlinge, 1972; Bonesi et al., 2004). It also has been well documented that introduced American mink have had serious impacts on their European counterpart (Maran and Henttonen, 1995; Santulli et al., 2014). However, Lodé (1993) reported no competitive overlap in diets between mink and the more terrestrial European Polecats (*Mustela putorius*) while Harrington and Macdonald (2008) found that mink and polecats overlapped home ranges but avoided simultaneous use of areas. Unfortunately, we had insufficient detections of river otter at our camera stations to use in our analyses, despite the species occurring in relatively abundant numbers in the study area (Johnson et al., 2013).

There are limitations to this study that should be recognized. Our sample size of camera stations (n=37) was relatively small and may have produced larger coefficients and associated confidence intervals. A single-season analysis is a snapshot in time and may be influenced by conditions in that single year. Also, other species (e.g., river otter) that are abundant and could influence mink occupancy may not have been attracted to the bait and thus would be under-represented. Lastly, there may be spatial autocorrelation between sites that cause marten

**Table 4** – Model averaged beta coefficients ( $\beta$ ) and associated confidence intervals (85%) for covariates included in occupancy models of American mink in the John Prince Research Forest, central British Columbia, Canada.

Covariate	$\beta$	Lower 85% CI	Upper 85% CI
Riparian	0,03	-0,1	0,16
Conifer	-8,46	-21,4	4,43
Fishbearing	0,57	0,42	1,1
Beaverlodge	-0,13	-0,44	0,18
Weasel	0,02	-0,04	0,09
Lynx	-0,05	-0,39	0,28
Marten	-0,54	-1,57	0,54

detections to be over-represented in the models. However, this is less of a concern in light of the associations found with habitat variables (e.g., conifer), and the relatively small scale of measurement (100 m radius) used around the camera sites. One or a combination of these factors may possibly account for some of the model uncertainty found in this study. Notwithstanding, this study provides an important initial data set on a relatively unstudied furbearer in its native habitat.

Large-scale alterations to forest landscapes, including commercial forestry or the recent unprecedented changes caused by forest pests (e.g. pine beetle outbreaks in western Canada) will undoubtedly impact carnivore communities in various ways, including alterations to the coexistence and interaction of species such as those included in this study. Knowledge of these interactions, and how changes to habitat influences the community, are needed to augment more traditional habitat selection studies. Ultimately, understanding the long-term spatial distribution and population dynamics of carnivore communities will be required to craft meaningful management and conservation programs for the taxa.

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