

**An Expanding Meso-Carnivore: Fisher (*Pekania pennanti*)
Occupancy and Coexistence with Native Mustelids in Southeast
Alaska**

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Authorization to Submit Thesis

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Abstract

Around the world, species' distributions are shifting in response to climate change, with many species exhibiting range expansion toward the poles. These range expansions often create new species interactions that are not well understood. The fisher (*Pekania pennanti*) is a mammalian carnivore that has recently expanded its range into the coastal temperate rainforest of Southeast Alaska (SEAK). Little is known about its occupancy and effects on native species in SEAK. The goal of this project was to examine fisher occupancy, density, and co-occurrence with American marten (*Martes americana*) and ermine (*Mustela erminea*) in SEAK. Twenty-five paired camera and hair snag stations (50 unique sites) were deployed north of Juneau, Alaska from January–April 2018, resulting in detection of 15 species and collection of 204 hair samples. A mitochondrial DNA (mtDNA) fragment analysis test conducted on DNA extracted from hair samples showed that 16 of those hair samples were from fisher. From the fisher DNA samples, only one individual fisher was identifiable (a female), making a fisher density analysis unfeasible. However, camera trap data allowed us to conduct single-species occupancy analyses at both local and landscape scales, which showed that fisher detection was negatively impacted by snow density, but positively affected by whether a fisher had been detected during the previous occasion (average site scale $p = 0.24$ (95% CI = 0.12, 0.40); average grid scale $p = 0.15$ (95% CI = 0.07, 0.30)). Fisher occupancy was positively impacted by snow density at the local scale and positively associated with vegetation height at a broader scale (average site scale $psi = 0.30$ (95% CI = 0.08, 0.72); average grid scale $psi = 0.40$ (95% CI = 0.003, 0.83)). Although denser snow could enable fishers, with their high foot-load, to more easily travel on top of the snow, the effect may not be powerful enough to overcome the high energetic costs associated with movement in deep snow and therefore not positively contribute to detectability on a day-to-day basis. The positive correlation between snow depth and snow density in our analysis appears to support this trade-off ($r = 0.43$). Among mustelid species we examined, top models from two-species occupancy analyses indicated that fisher have no effect on marten or ermine occupancy at either local or landscape scales, although there was limited support for fisher negatively impacting marten occupancy at the landscape scale. Analysis of diel activity patterns and temporal overlap indicated that marten and ermine adjust their activity patterns in the presence of fisher. Furthermore, ermine displayed temporal avoidance of marten, whereas marten activity was unaffected by ermine presence. Infrequent detections of fisher made it unfeasible to examine shifts in fisher activity patterns in the presence of marten and ermine. These results show that mustelids in SEAK currently co-exist through temporal partitioning, with smaller-bodied mustelids avoiding larger, more

dominant mustelids. However, if the fisher population continues to expand, these species interactions may shift and increase the potential for competition between fisher, marten, and ermine in SEAK. Overall, this research indicates that fisher, including a female, currently occupy the study area and prefer areas with taller trees and higher snow density, indicating that other areas of Southeast Alaska may be suitable fisher habitat. This study's methodology was unable to measure fisher density, but was able to identify important predictive covariates associated with fisher occupancy in SEAK, and illustrate temporal partitioning patterns among mustelid species in the area. Future research to increase sample size and detect species' trends in SEAK over time will be beneficial, but this research was an important first step in developing a monitoring protocol and management plan for fisher in SEAK.

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Dedication

This thesis is dedicated to my parents, Veronica and Kenneth Kupferman. Their unwavering love and support has driven me through both graduate school and life. Thank you for building me up, encouraging my adventures, and instilling in me the confidence needed to pursue this wild career.

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Introduction

Ongoing climate change is shifting species distributions worldwide, with many species exhibiting range expansion toward the poles (Levinsky et al., 2007; Tylianakis et al., 2008). In North America, mammalian species such as mule deer (*Odocoileus hemionus*), white-tailed deer (*Odocoileus virginianus*), and cougars (*Puma concolor*) have been documented in the Yukon Territory for the first time during recent years (Hoefs, 2001; Jung & Merchant 2005; Knopff et al., 2014). Likewise, the continued northward expansion of red fox (*Vulpes vulpes*) and white-tailed deer in Canada, as well as numerous butterfly species across North America has also been attributed primarily to climate change (Dawe & Boutin, 2016; Walther et al., 2002). These shifting distributions will likely create new species assemblages and community interactions that are not well understood, potentially impacting native species and altering entire ecosystems (Tylianakis et al., 2008).

When new species colonize existing ecological communities, they can impact native, established species via multiple mechanisms. Competition for resources, through both direct and indirect means, is often a primary mechanism driving change (Human & Gordan, 1996; Lang & Benbow, 2013). Competition occurs when organisms vie for a common limited resource (Lang & Benbow, 2013), such that species with the same ecological requirements cannot coexist (Hardin, 1960; Wereszczuk et al., 2015). Consequently, one species may exclude the other, leading to local decline or extinction of the weaker competitor (Hardin, 1960). However, the degree of competition between species can be dictated by many factors including overlap in diet, space, time, predators, and other resources (Manlick et al., 2017). Many ecological studies support the concept of niche partitioning, in which species occupying the same ecological guild will adjust their use of resources to allow for coexistence (Albrecht & Gotelli 2001; Chillo et al., 2010; Wereszczuk & Zalewski, 2015). As animal distributions continue to shift, successful management of novel species and conservation of existing species will require an understanding of the competitive role that novel species may play with native organisms. To this end, determining the population status and requirements of establishing species is a crucial first step towards understanding potential ecological impacts of species introductions.

The fisher (*Pekania pennanti*) is a newly colonizing forest carnivore in Southeast Alaska (SEAK) that may impact native, established species in the region if it continues to expand. Little is known about the abundance, habitat associations, and ecological impacts this novel species may have in SEAK. The goal of this research was to collect baseline data on fisher density, habitat associations,

and competition with established species in the region in order to develop a better understanding of fisher ecology in SEAK and to implement a monitoring protocol for fisher in the region.

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Chapter 1: Assessing Environmental Covariates Associated with Fisher (*Pekania pennanti*) Occupancy in Southeast Alaska

Abstract

Altered environmental conditions associated with climate change are contributing to shifting species' distributions worldwide. Fisher (*Pekania pennanti*) are mammalian carnivores that have been naturally expanding their range in areas across North America, potentially as a result of climate warming in winter. Fishers have been expanding northward within Canada for decades, but have recently expanded their range into the coastal temperate rainforest of Southeast Alaska (SEAK). Fisher range and winter movement is limited by the presence of deep, soft snow, and decreased snow accumulation in SEAK may be contributing to their colonization in the area. I examined the status and ecological requirements of this novel fisher population by investigating environmental covariates associated with fisher occupancy in Juneau, Alaska during the winter of 2018. Using data collected from 25 remote camera stations (50 unique cameras) and Alaska IfSAR datasets, I conducted single-species occupancy analyses at both site (local) and grid (broad) scales using 7-day occasions. There were 17 fisher detections across the study season at 7 clusters (9 unique stations). I found evidence that snow density had a positive effect on fisher occupancy at a local scale, whereas elevation had a negative effect on fisher occupancy. Vegetation height had a positive effect on fisher occupancy at a broad scale in SEAK. Mean fisher detection probability at the site scale was predicted to be 0.24 (95% CI = 0.12, 0.40), and mean occupancy was estimated at 0.30 (95% CI = 0.08, 0.72). Estimated mean fisher detection probability at the grid scale was 0.15 (95% CI = 0.07, 0.30), with an estimated mean fisher occupancy of 0.40 (95% CI = 0.003, 0.83). Results from our research provided an initial assessment of important environmental characteristics associated with fisher occupancy in SEAK. Results further indicate that fisher occupancy is relatively high, and that suitable habitat (i.e., tall trees and dense snowpack) could potentially be found elsewhere in SEAK. Further research to increase sample sizes and to monitor across seasons, years, and a broader area will be beneficial to detect trends in fisher occupancy, along with potential expansion of the current population, and to better understand important predictive covariates in SEAK. This research contributed to a more thorough understanding of an expanding meso-carnivore species that can be used to highlight the effects of shifting species ranges associated with global environmental change.

Introduction

The fisher is a mid-sized mammalian carnivore occupying forest habitats across northern North America. They are a habitat specialist, favoring mature forests with dense canopy cover and high structural diversity (Aubry et al., 2013; Sauder & Rachlow, 2014). Prior to European settlement, the fisher's Northwest range extended from the Sierra Nevada mountains of California to northern British Columbia (Jordan, 2007), but human activities such as over trapping and old-growth logging have led to significant range reductions in the last century (Powell & Zielinski, 1994). Reintroduction efforts began in Nova Scotia in the 1940s as wildlife managers aimed to re-establish populations of fisher as an important regional furbearer and native carnivore (Lewis & Hayes, 2004). Since that time, over 20 successful reintroductions of fisher have taken place across their native range (~77% success rate), leading to regional increases in abundance (Aubry & Lewis, 2003; Lewis & Hayes, 2004; Lewis et al., 2012; Lewis, 2014).

In addition to direct reintroduction by humans, fisher appear to be naturally expanding their range in areas across North America (Lewis, 2014), potentially as a result of climate warming in winter. Fisher are often found in regions of mild winter temperatures and low snowfall levels (Lewis, 2014), and regional increases in temperature and precipitation (falling as rain rather than snow) associated with climate change may be enabling range expansion (IPCC, 2007; Olson et al., 2014).

Fishers have been documented in the Yukon region of Canada for decades but recent accounts from trappers suggest that they have further expanded their range into the coastal temperate rainforest of Southeast Alaska (SEAK) (Jung & Slough, 2011; Parr, 2016), potentially penetrating through the high-altitude coastal mountains via river valleys (ADFG, 2006). In neighboring British Columbia, fishers are associated with late-successional conifer forests, and the high prevalence of conifer forest in SEAK (>71% of vegetation) suggests that the region contains suitable fisher habitat for colonization (Campbell et al., 2004). Forest habitat in SEAK may increase in the future as warmer temperatures are expected to raise the subalpine treeline elevation and assist with the rapid expansion of native conifers (Shanley et al., 2015).

Additionally, fisher range and winter movement is limited by the presence of deep, soft snow because their large body size makes them more likely to sink in areas of soft snow and increases their energetic costs associated with travel (Raine, 1983). Decreased snow accumulation in SEAK and the increased snow density associated with more frequent freeze-thaw events may be contributing to their colonization of the area (Kim et al., 2018; Krohn et al., 1995; Shanley et al., 2015; Zielinski et al.,

2017). Regional snowfall is projected to further decline in the future likely aiding in fisher movement and potential expansion (McAfee et al., 2014; Shanley et al., 2015).

Although fishers have been present at low numbers in SEAK since the late 1990s, little is known about the impacts this newly established fisher population could pose to the region's existing ecological community if it continues to expand. For example, fishers are known to prey on many species native to SEAK, such as snowshoe hare (*Lepus americanus*) and porcupine (*Erethizon dorsatum*), and can also compete with the native American marten (*Martes americana*) (Lewis, 2014; Zielinski et al., 2004; Zielinski et al., 2017). Baseline data informing fisher population status and habitat requirements is needed before a species management plan can be designed. In conjunction with the Alaska Department of Fish and Game and the University of Idaho, the goal of this chapter was to provide an estimate of fisher occupancy in SEAK and to measure the effects of key environmental predictive variables.

Methods and Materials

Study Area

Research was conducted in the forested coastal area north of Juneau, Alaska (Figure 1.1). The study area was approximately 200 km², ranging north of Juneau to the Echo Cove area on the southern edge of Berner's Bay. It was comprised mainly of western hemlock (*Tsuga heterophylla*) and Sitka spruce forest (*Picea sitchensis*), with deciduous trees and herbaceous shrubs dispersed throughout (Boggs et al., 2016). The study area was relatively long and narrow, and was approximately 10 km at its widest point. The shoreline of Favorite Channel delineated the western boundary of the research area and, to the east, conifer forests quickly became steep, mountainous areas consisting of glaciers, snow, and ice (the Juneau Icefield). Due to logistical constraints, research was restricted to the lower elevations (≤ 500 m) between these areas.

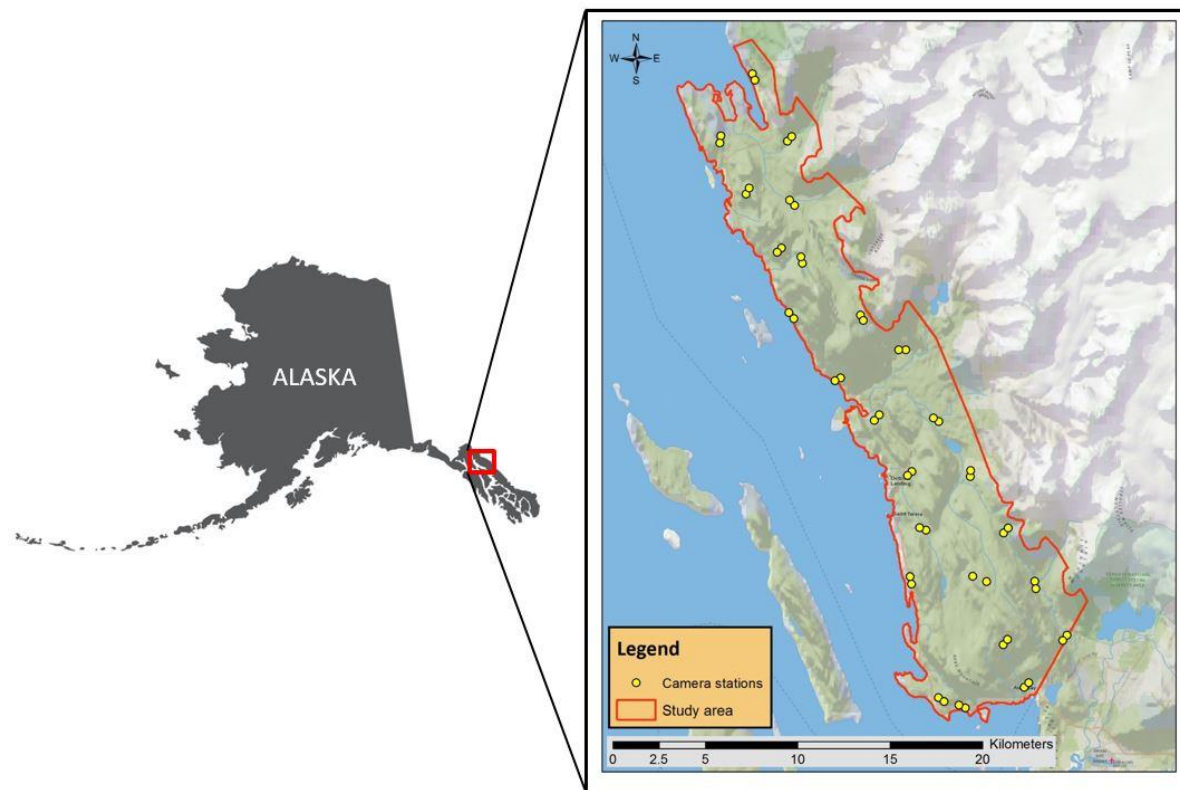


Figure 1.1. The study area, located north of Juneau, Alaska. Locations of the camera stations are indicated by yellow circles.

The study area is part of the coastal temperate rainforest of Southeast Alaska and is characterized by cool summers, mild winters, and high year-round precipitation. The average winter temperature hovers around freezing, and as a result, precipitation in winter can fall as rain or snow, resulting in highly variable annual snowfall (Kirchhoff & Schoen, 1987; Shanley et al., 2015). Climate projections for the region predict an annual decrease in snowfall of 22–40% by the 2080s, despite a projected 3–9% increase in precipitation (Shanley et al., 2015).

Fisher have been documented in the Juneau area since the 1990s, with 26 fisher taken and reported between 1996–2019 (C. Rice, personal communication; Parr, 2016). Of those, 14 were females and 10 were males, with 2 of unknown sex (C. Rice, personal communication). Recent necropsies on harvested fisher have identified 2 juvenile females, 1 juvenile male, 1 adult female, and 1 adult male (A. Crupi, personal communication). Given that females typically disperse shorter distances than males, and that at least one adult female was reported, it seems likely that a breeding population exists in SEAK near Juneau.

The area is already home to many other carnivore species including American marten (*Martes americana*), ermine (*Mustela erminea*), mink (*Neovison vison*), wolverine (*Gulo gulo*), wolf (*Canis lupus*), coyote (*Canis latrans*), and black and brown bear (*Ursus americanus* and *arctos*, respectively) (Parr, 2016). Many of these species are potential competitors, and even predators, of fisher, with previous studies documenting fisher mortalities due to wolverine and coyote (Lofroth et al., 2010; Wengert et al., 2014). Species that fisher likely prey on include snowshoe hare (*Lepus americanus*), porcupine (*Erethizon dorsatum*), and red squirrel (*Sciurus vulgaris*) (Parr, 2016). Additionally, fish have been documented occasionally in the diet of fishers, and the abundance of salmon-bearing streams in the region could provide salmon as a seasonal prey source, particularly as it is a known source of food for the sympatric American marten in the region (Flynn et al., 2004; Giuliano et al., 1989; Weir et al., 2005). The combined prey availability, climate, and habitat characteristics of the study area likely create significant suitable habitat capable of supporting a fisher population (Campbell et al., 2004).

Camera Grid Design

Camera stations (Bushnell® Trophy Cam HD Aggressor 24MP No-Glow, Bushnell Outdoor Products, Overland Park, KS) were deployed across our study area from 15 January through 20 April, 2018. These remote cameras were paired with hair snags to collect genetic samples for use in a future density analysis. Before deploying stations, preliminary simulations were conducted using custom scripts within the ‘secr’ package in program R to design the station layout (Efford, 2017); this was done to ensure adequate spatial sampling and detection rates for the density analysis using spatially-explicit capture–recapture techniques. The simulations took into account both fisher and marten home range size in order to estimate the parameter σ , the spatial scale parameter of an individual, which is related to the declining probability of detecting an animal as the distance from an animals’ activity center to the station increases (Sun et al., 2014). The simulations also set a limit of 60 stations, based on what would be realistic to maintain in the field. Based on the simulations, hexagonal grid cells approximately 10.6 km² were distributed across the study area, with two camera stations clustered 350 m apart in the center of each grid cell. In total, 50 cameras were deployed across the study area for a total of 25 sites (Figure 1.1). More information on the camera grid design and density analysis is available in Appendix A.

Because fisher occupancy and habitat use are unknown for this region, I included a broad array of vegetation types in the grid design, only eliminating regions that were logistically unfeasible to access or consisted of human development, water, rock, and/or ice. The long distance lures Gusto™

and Mega Musk™ (Minnesota Trapline Products, Inc., Penneck, MN) were hung from lure boxes in view of the camera. Lure was replenished and the type of lure was alternated at each station check (approximately every 14 days).

Quantifying Environmental Covariates

Identifying and quantifying relevant landscape and climate characteristics is important in understanding fisher ecology and requirements within SEAK. Snow is a major limiting factor for fisher distribution, with one study finding that fishers did not occupy regions with > 65 cm snowpack due to prohibitive costs of movement through deep, soft snow (Raine, 1983; Zielinski et al., 2017). To measure the effect of snow on fisher occupancy, snowfall was monitored in several ways. Each of the stations contained a PVC pipe with clearly marked height measurements placed within view of the camera. Each camera was set to take a photo at noon each day to obtain snow depth measurements over time. At each station visit, snow depth and density were measured using a measuring tape and snow density gauge (Electronic Snow Density Gauge 250, Brooks-Range Mountaineering Equipment, Co., San Leandro, CA). The remote cameras recorded temperature with each photo taken, and in addition, a photo and corresponding temperature was automatically taken each day at noon, which was used to calculate average noon temperature covariate values for each site. Elevation, aspect, slope, canopy cover, distance to road, and vegetation height variables were obtained from Alaska IfSAR datasets (USGS, 2017).

In addition to the environmental covariates mentioned previously, a prey availability covariate was measured by summing camera detections of fisher prey species at each site. The covariate entitled “behavioral effect” (BK) was added to determine whether a fisher detection at one occasion was related to a detection on the previous occasion (Sweitzer & Furnas, 2016). This was a binary covariate with a 0 value indicating that a fisher was not detected at the previous occasion and a 1 indicating prior detection.

Occupancy Modeling

Before beginning my occupancy analysis, I developed hypotheses regarding which covariates would contribute to fisher detectability and occupancy in SEAK (Tables 1.1 & 1.2, respectively). To determine whether fishers’ abiotic associations differed between local and landscape scales, occupancy analyses were conducted at two scales: site (fine scale) and grid (broad scale). Covariates were tested at both scales when possible. Due to spatial correlation between each cluster of cameras,

site-level covariate values were determined by averaging covariate values between the two camera stations within each cluster to create one site-level value. Because of logistical and accessibility constraints, some of the cameras were not in the exact location set by the simulations. Therefore, to determine covariates at the grid scale, a buffer around each camera cluster was created within ArcGIS, equal in area to the original grid cells (~10.6 km²), rather than using values from the grid cell polygons themselves. Available environmental data was then averaged for the area within each buffer, creating grid level covariate values.

In my initial hypotheses, covariates were separated into site covariates that remained constant throughout the surveys and observation covariates that changed with each occasion. These initial hypotheses were extensive and were gradually eliminated at each step so as not to flood the occupancy models with parameters. Increasing the number of parameters in an occupancy model with a small dataset can increase the likelihood of model overfitting (Burnham & Anderson, 2002), or cause spurious effects such as increasing standard errors of estimates or causing a model to not converge. Elimination of covariates through correlations, using a low correlation cut-off of $r > |0.6|$, helped to reduce the likelihood of over parameterizing models with the sparse fisher dataset. Covariates were standardized for all analyses to have a mean of zero and a standard deviation of 1.

Model Fitting and Selection

Single-species occupancy models are beneficial in determining the probability of a species occupying a given area, and can provide insight into important environmental factors that contribute to species' occupancy, while accounting for imperfect detection (Mackenzie & Bailey, 2004). In this analysis, single-species occupancy models were run using 7-day occasions within the 'unmarked' package in program R. There were a number of factors to consider when determining occasion length. First, the limited detections of fisher made the ideal occasion length one that would increase occasion detection rate to an estimable number but also not condense the detection data so much that the total number of detections was below the sample size needed to analyze models. Additionally, the occasion length chosen needed to coincide with the timeframe in which the covariates were measured. For instance, snow density data was only recorded once every two weeks when a station was visited, making very short occasion lengths, such as one or two days, infeasible without losing the accuracy of snow density measurements. To determine reasonable occasion lengths, I conducted a sensitivity analysis. Occasion length was varied from 1 to 10 days, and corresponding fisher detection histories were created. I then ran null occupancy and detection models for each occasion length, and compared

detection and occupancy outputs and resulting confidence intervals. Ultimately, a 7-day occasion resulted in the highest detection rate while maintaining an appropriate confidence interval ($p = 0.18$, 95% CI = 0.11, 0.29), and all covariates were able to be reasonably estimated within this timeframe. Additionally, the total sample size of fisher detections was not significantly reduced when detection history was condensed into 7-day occasions. Resulting models from the single-species occupancy analyses were compared for best-fit using Akaike's information criterion with a bias-correction term for small sample size (AIC_c) (Burnham & Anderson, 2002; Fiske & Chandler, 2011; Symonds & Moussalli, 2010). AIC_c is widely used in wildlife studies to determine the model that best represents the underlying biological mechanisms of the study in question (Symonds & Moussalli, 2010).

I employed a two-step approach to model selection by: 1) first fitting competing models of detectability dependent on the environment and selecting an optimal detectability model while keeping occupancy constant; and 2) then incorporating occupancy covariates into that model (Fuller et al., 2016). To determine the best-fit detectability model, I ran a separate model for each detectability covariate while keeping occupancy constant, and compared AIC_c scores to determine which singular detectability covariate had the lowest AIC_c (Table 1.6). I then used forward-stepwise selection to sequentially add covariates to the univariate model based on AIC_c rankings (starting with the covariate with the lowest AIC_c score) until I obtained a model with the lowest AIC_c score that had reasonable standard errors and parameter estimates. Twenty-four sites were included in the final analysis after one site was discarded due to a lack of data caused by camera malfunctions.

In order to assess model fit to ensure that biological inferences could be made from the best-fit models, each global model for site and grid scale was assessed using Mackenzie-Bailey goodness-of-fit tests, and \hat{c} values were examined to check for overdispersion of the data (Burnham & Anderson, 2002; Mackenzie & Bailey, 2004). \hat{c} values >1 indicate overdispersed data (where the sampling variability is greater than expected under an assumed distribution), and such problematic overdispersion will yield a small p value (Burnham & Anderson, 2002).

Table 1.1. Predicted effects of covariates on fisher detection probabilities in Southeast Alaska.

Detection covariate	Hypothesis
Snow depth	Probability of detection will decrease with deeper snow due to reduced movement.
Snow density	Probability of detection will decrease as snow density declines due to reduced movement ability in soft snow.
Temperature	Probability of detection will decrease as temperature declines due decreased movement that comes with increased thermogenic costs associated with low temperatures.
Behavioral effect (BK)	Probability of detection will increase when a fisher has been detected on the previous occasion due to site fidelity.
Distance to road	Probability of detection will increase further from roads due to less human disturbance.

Table 1.2. Predicted effects of covariates on fisher occupancy probabilities in Southeast Alaska.

Occupancy covariate	Hypothesis	Scale
Elevation	Probability of occupancy will decrease with increasing elevation.	Site and Grid
Slope	Probability of occupancy will decrease with increasing slope due to increased difficulty to access.	Site
Canopy cover	Probability of occupancy will increase with increasing canopy cover due to canopy cover associations with fisher resting sites (Sauder & Rachlow, 2015).	Site and Grid
Vegetation height	Probability of occupancy will increase with increasing vegetation height due to fishers' association with mature forests (Sauder & Rachlow, 2014).	Site and Grid
Aspect	Probability of occupancy will be higher on east-facing slopes due to the shelter provided from ocean winds traveling from the west.	Site
Distance to road	Probability of occupancy will increase further from roads due to less human disturbance.	Site
Prey	Probability of occupancy will be positively associated with food availability.	Site
Average station snow depth	Probability of occupancy will decrease with increasing snow depth due to decreased movement ability.	Site
Average station snow density	Probability of occupancy will increase with increasing snow density because the denser snow increases movement ability.	Site
Average temperature	Probability of occupancy will decrease with decreasing temperature due to the increased energetic costs associated with thermoregulation.	Site

Results

There were 17 fisher detections across the study season at 7 stations (9 unique cameras). After testing for covariate correlations, the occupancy covariates slope, distance to road, and station snow depth were eliminated at the site scale, and at the grid scale, the covariates canopy cover and aspect were eliminated. The covariates prey, snow depth, snow density, distance to road, and temperature were only available at the site scale and were not included in the grid scale analysis. I also tested for correlations among the detectability covariates and found no significant correlations.

As a result, the detectability covariates I evaluated in model selection were distance to road, snow depth, snow density, temperature, and behavioral effect. The occupancy covariates at the site scale were station elevation, station canopy cover, station vegetation height, station aspect, prey, average snow density and average temperature. At the grid scale, the covariates evaluated were elevation and vegetation height.

Single-Species Occupancy Analysis

Detectability Model

When comparing univariate models of each detectability covariate, snow density had the lowest AIC_c value, followed by behavioral effect, snow depth, temperature, and distance to road, respectively. I ultimately determined that the best detectability model, keeping occupancy constant, included snow density and behavioral effect as covariates (Table 1.3). Snow density had a negative effect on fisher detection (-1.17, 95% CI = -1.88, -0.51) whereas behavioral effect, or fisher detection at the previous occasion, had a positive effect (0.36, 95% CI = 0.04, 0.68) (Figures 1.2 & 1.3). These same detectability covariates were incorporated in both the site and grid scale occupancy analyses because the detectability covariates are localized around each camera site and therefore do not change as the occupancy scale changes.

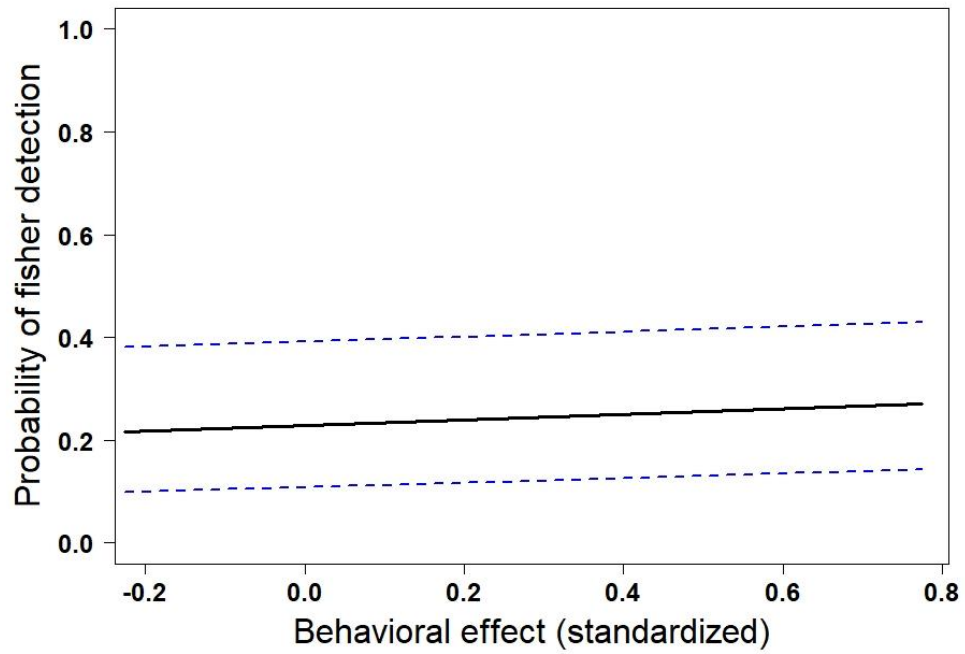


Figure 1.2. Site scale effect of previous detection of fisher at a site (“behavioral effect”) in Southeast Alaska from the top-ranked model of fisher detection. All model covariates were standardized.

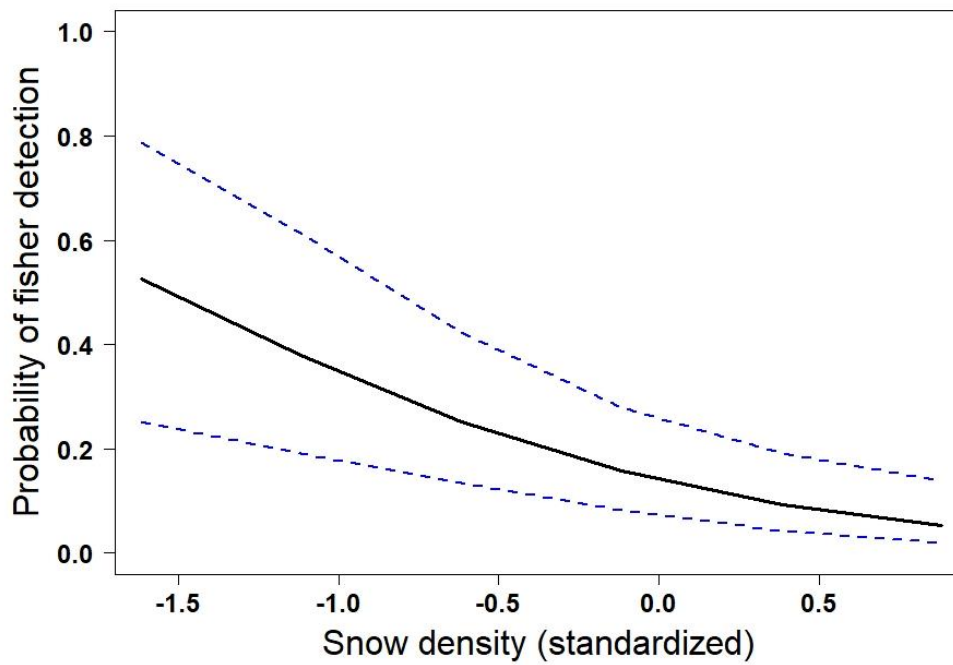


Figure 1.3. Site scale effect of snow density on fisher detection in Southeast Alaska from the top-ranked model of fisher occupancy. All model covariates were standardized.

Site Scale Occupancy

At the site (local) scale, mean weekly occupancy was estimated to be 0.30 (95% CI = 0.08, 0.72). Mean weekly fisher detection was predicted to be 0.24 (95% CI = 0.12, 0.40). Stage 2 of model selection, determining best univariate models, yielded eleven models (Table 1.4). I eliminated vegetation height from further consideration because the univariate model containing it failed to converge. Snow density was the covariate with the lowest AIC_c score that still had acceptable standard errors and parameter estimates, followed by station elevation. In the final stage of model selection, these two best-supported occupancy covariates were combined with the detection covariates. The top-ranked model based on AIC_c score incorporated the effects of both average snow density and station elevation on occupancy at the site scale. Snow density had a positive effect on fisher occupancy (1.91, 95% CI = 0.03, 3.78) (Figure 1.4a) whereas elevation had a negative effect (-2.98, 95% CI = -6.91, 0.95) (Figure 1.4b). Aspect was then incorporated into the model, but yielded a higher AIC_c score by > 4 units. At the site scale, the global model produced a \hat{c} value of 0.17 (p value = 0.81), showing no evidence of lack-of-fit or overdispersion of the data (Burnham & Anderson, 2002).

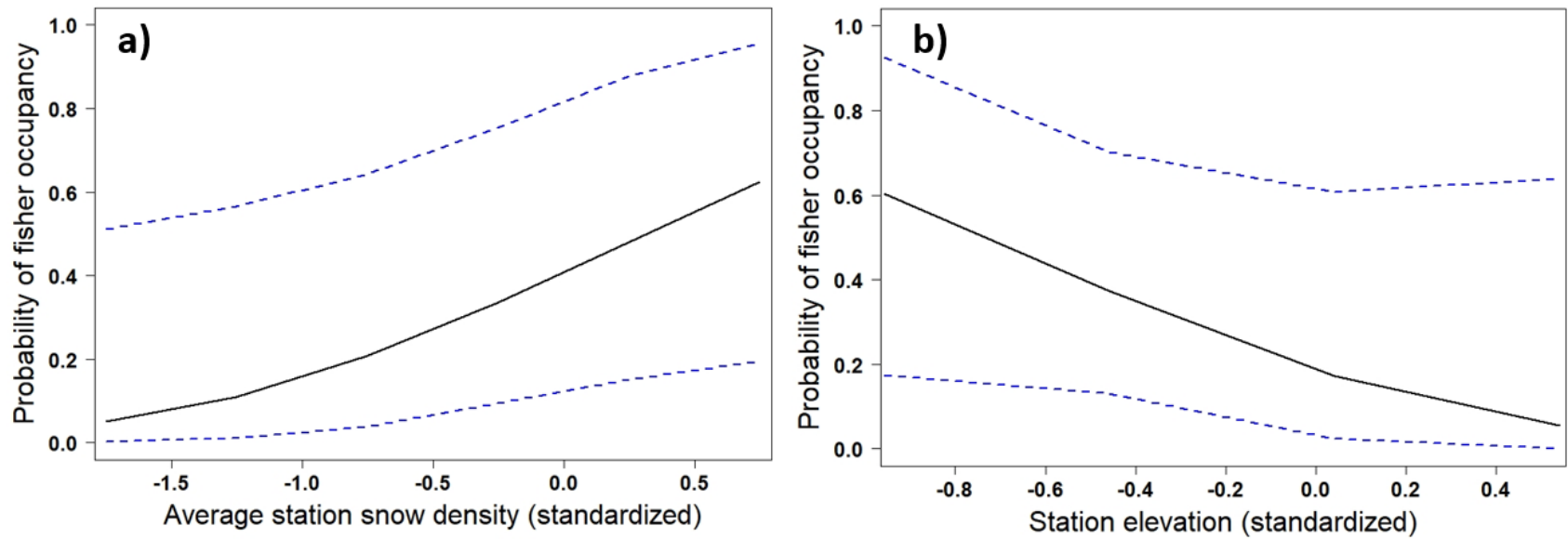


Figure 1.4. Site scale effect of average station snow density (a) and station elevation (b) in Southeast Alaska from the top-ranked model of fisher occupancy. All model covariates were standardized.

Grid Scale Occupancy

At the larger scale of grid cells, the estimated mean for weekly detection probability for fisher was 0.15 (95% CI = 0.07, 0.30). Estimated weekly mean fisher occupancy was 0.40 (95% CI = 0.003, 0.83). To assess fisher occupancy at this scale, I repeated the same model building process as above and substituted grid scale covariates in place of site scale covariates (Table 1.5). Univariate models for the two grid scale occupancy covariates each converged successfully, and the model incorporating vegetation height had the lowest AIC_c score. However, when elevation was added, the model produced a low AIC_c score, but did not converge and yielded high standard errors and parameters, so was not included in model selection moving forward. Vegetation height had a positive impact on fisher occupancy at the grid scale (2.97, 95% CI = -1.31, 7.25) (Figure 1.5). At the grid scale, the global model c-hat was 0.14 (p value = 0.76), showing no evidence of lack-of-fit or overdispersion of the data (Burnham & Anderson, 2002).

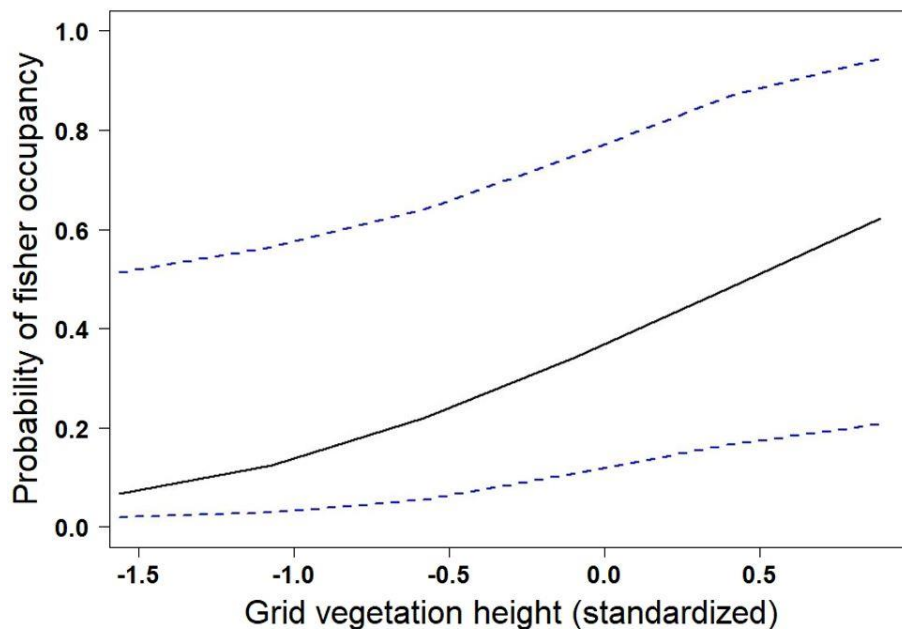


Figure 1.5. Broad scale (grid level) effect of vegetation height in Southeast Alaska from the top-ranked model of fisher occupancy. All model covariates were standardized.

Table 1.3. Detectability model results for fisher in Southeast Alaska, with constant occupancy. Intercept values for both detection (p) and occupancy (psi) are reported as well as the detectability covariates of road (rd), snow depth, snow density, temperature (temp), and behavioral effect (BK). AIC_c, delta AIC_c, and weight of the models is also reported. AIC_c = Akaike information criterion for small sample sizes.

	p(Int)	psi(Int)	p(distance to road)	p(snow depth)	p(snow density)	p(temp)	p(BK)	AIC _c	delta AIC _c	weight
p(snow density + BK)	-1.95	-0.55			-1.17		0.36	101.50	0.00	0.61
p(snow density)	-1.59	-0.67			-1.29			103.49	1.99	0.23
p(snow depth + snow density + BK)	-1.98	-0.53		-0.25	-1.07		0.36	104.41	2.90	0.14
p(BK)	-2.17	-0.31					0.49	109.61	8.10	0.01
p(rd + snow depth + snow density + temp + BK)	-1.98	-0.54	0.12	-0.16	-1.10	0.24	0.36	111.01	9.51	0.01
p(snow depth)	-1.70	-0.71		-1.07				114.39	12.89	0.00
p(temp)	-1.60	-0.73				0.42		117.22	15.72	0.00
p(rd)	-1.51	-0.75	-0.07					119.65	18.15	0.00

Table 1.4. Occupancy model results for fisher at the site scale in Southeast Alaska. Intercept values for both detection (p) and occupancy (psi) are reported as well as the detectability covariates of behavioral effect (BK) and snow density (SnD), and the occupancy covariates of prey, elevation, canopy cover (CC), vegetation height (VH), aspect, snow density (SD), and temperature (temp). AIC_c, delta AIC_c (delta), and weight of the models is also reported. AIC_c = Akaike information criterion for small sample sizes.

	p(Int)	psi(Int)	p(BK)	p(SD)	psi(pre)	psi(elevation)	psi(CC)	psi(VH)	psi(aspect)	psi(SnD)	psi(temp)	AIC _c	delta	weight
p(BK + SnD) psi(VH)	-2.45	-0.43	0.49	-0.97				27.64				93.55	0.00	0.84
p(BK + SnD) psi(elevation + SnD)	-1.92	-1.23	0.35	-1.19		-2.98				1.91		98.55	5.00	0.07
p(BK + SnD) psi(SnD)	-2.24	0.03	0.42	-1.31						1.75		100.33	6.78	0.03
p(BK + SnD) psi(elevation)	-1.83	-1.37	0.34	-1.13		-2.00						101.05	7.50	0.02
p(BK + SnD) psi(.)	-1.95	-0.55	0.36	-1.17								101.50	7.95	0.02
p(BK + SnD) psi(aspect)	-1.99	-0.57	0.37	-1.19					-0.77			102.53	8.98	0.01
p(BK + SnD) psi(elevation + aspect + SnD)	-1.92	-1.22	0.35	-1.19		-3.06			0.13	2.02		102.59	9.04	0.01
p(BK + SnD) psi(CC)	-1.98	-0.71	0.37	-1.17			1.24					103.70	10.15	0.01
p(BK + SnD) psi(pre)	-1.92	-0.61	0.36	-1.16							0.24	104.50	10.95	0.00
p(BK + SnD) psi(temp)	-1.98	-0.53	0.37	-1.18	-0.20							104.61	11.06	0.00
p(BK + SnD) psi(pre + elevation + CC + VH + aspect + SnD + temp)	-1.78	-9.09	0.32	-1.18	-12.89	-6.79	-14.86	33.00	-2.33	18.64	30.70	110.28	16.73	0.00

Table 1.5. Occupancy model results for fisher at the grid scale in Southeast Alaska. Intercept values for both detection (p) and occupancy (psi) are reported as well as the detectability covariates of behavioral effect (BK) and snow density (SnD) and the occupancy covariates of vegetation height (VH) and elevation. AIC_c, delta AIC_c (delta), and weight of the models is also reported. AIC_c = Akaike information criterion for small sample sizes.

	p(Int)	psi(Int)	p(BK)	p(SnD)	p(temp)	psi(VH)	psi(elevation)	AIC_c	delta	weight
p(BK + SnD) psi(VH + elevation)	-2.56	-1.68	0.51	-1.04		38.63	-29.93	98.67	0.00	0.50
p(BK + SnD) psi(VH)	-2.44	0.07	0.48	-0.99		2.97		99.60	0.94	0.32
p(BK + SnD) psi(.)	-1.95	-0.55	0.36	-1.17				101.50	2.83	0.12

Discussion

This study sought to identify the effects of environmental and behavioral variables on occupancy and detectability of a novel forest carnivore, the fisher, that has recently colonized Southeast Alaska, likely as a result of climate change-driven reductions in snow depth. Average fisher occupancy ranged from 0.30 (site scale) to 0.40 (broad scale), which is similar to the estimated average yearly fisher occupancy in the Sierra Nevada mountain portion of its range ($\psi = 0.38$) but lower than estimates in New York ($\psi = 0.69$, estimated from 3-week occasions) (Linden et al., 2017; Zielinski et al., 2013). My occupancy estimates are within the range of previous studies' estimates from established fisher populations, indicating that fisher also likely have an established population in Southeast Alaska.

Average snow density across the season had a positive effect on fisher site scale occupancy in my study area, which lent support to my occupancy hypothesis. However, increased snow density had a negative impact on fisher detectability, which was in contrast to my expectations. Many northern carnivore species, such as the Canada lynx (*Lynx canadensis*) and arctic fox (*Vulpes lagopus*), possess disproportionately large feet relative to their body size (i.e. low foot-load (animal mass (g)/area of four paws (cm²)) (Murray & Lariviere, 2002). This low foot-load aids in movement, making it easier for animals to travel on top of deep, soft snow (Murray & Lariviere, 2002). However, the fisher has a relatively large body size for a mustelid, with a high foot-load (Krohn et al., 2003). This makes it difficult for fisher to traverse areas of deep snow (Krohn et al., 2003). Although high snow density has the potential to help fisher travel more easily on top of the snow, the effect may not be powerful enough to overcome the high energetic costs associated with movement in deep snow and therefore not positively contribute to detectability on a day-to-day basis (Krohn et al., 1993; Manlick et al., 2017). The positive correlation between snow depth and snow density in my analysis may corroborate this ($r = 0.43$). However, these results suggest that although high snow density may not increase localized fisher movement rates (i.e. detectability), its effects increase fisher site occupancy. The projected decrease in snow fall in SEAK coupled with an increase in freeze-thaw events leading to denser snow may lift movement limitations imposed on fisher in the region and enable the fisher population to continue to expand its population size and distribution (Kim et al., 2018; Raine, 1983).

Also consistent with my expectations, fisher site scale occupancy was found to decrease with increasing elevation. Again, this can most likely be explained by the movement limitations imposed by deep snow on fisher because elevation was positively correlated with snow depth in this study

area. Finally, the increase in fisher occupancy relative to vegetation height supports my grid scale occupancy hypothesis and aligns with previous studies that have illustrated the positive association between fisher occupancy and mature forests (Kilpatrick & Rego, 1994).

Fisher occupancy was driven by different environmental variables at the local versus broad scales, suggesting that fisher habitat selection is a hierarchical process. Fisher habitat selection is often examined at local scales, as researchers aim to identify specific areas associated with resting sites important for survival and denning (Sauder & Rachlow, 2014). However, it is also important to understand fisher habitat selection at broader scales in order to assess the potential impacts of large scale forest management practices on the species, and to better predict the likelihood of fisher occupancy when fine scale habitat data is not available (Sauder & Rachlow, 2014). Assessing fisher occupancy covariates at multiple scales was also important for this study as fisher home range in SEAK was unknown. Previous studies have shown fisher home range size to vary widely across geographic locations and between the sexes, with an average home range size of 40 km² for males and 15 km² for females in the Western United States (Powell & Zielinski, 1994; Zielinski et al., 2004). It is likely that fisher within SEAK have similar home ranges to those in the Western United States, making the grid scale (~10.6 km²) analysis from this study more likely an estimation of probability of use rather than true occupancy due to the fact that the grid cells were smaller than the average home range size of fisher. However, this study found that different covariates contributed to fisher occupancy between site and grid scales, suggesting that fisher requirements may shift gradually even among scales smaller than a home range. This shows that intermediary measurements of fisher habitat associations can also be of value in predicting occupancy. Additional data assessing important covariates contributing to fisher occupancy at a known home range scale would be beneficial to further assess how fisher requirements change with scale.

This study faced several limitations. I found that fisher detection probability increased if fisher had been detected during the previous occasion, illustrating that fisher exhibit site fidelity and suggesting that the lure and bait used may have biased the detection and occupancy results. However, given that colonizing fisher are such a rare and cryptic carnivore, lure and bait were necessary to maximize detections over a short timeframe to provide initial estimates for this novel species. An alternative explanation for this site fidelity may be that fisher use areas of their home range for weeks at a time due to the presence of resources and would therefore be detected at a site repeatedly over several occasions.

It is also important to note that there were wide confidence intervals regarding the estimated covariate effects (Figures 1.2–1.5), almost certainly due to the relatively small sample size of fisher

detections in this single-season study (Guillera-Arroita et al., 2010). The small sample size also limited the number of covariates that could be incorporated into each model, and further research to increase sample sizes and seasonal monitoring across years will be beneficial to both detect trends in fisher occupancy and to better understand important predictive covariates in SEAK.

This research provided important baseline data that will aid in the development of a management plan and monitoring protocol for the fisher population in SEAK. Results from our research provided an assessment of important environmental characteristics associated with fisher occupancy and detectability, although our ability to make ecological inference was limited by sample size. This is a first step towards the eventual development of a habitat suitability map and monitoring protocol for fishers within SEAK, making it possible to identify areas most likely to exhibit fisher colonization in the future should the species continue to expand its range.

More broadly, as environmental conditions associated with climate change continue to transform species' distributions, the resulting alterations to community composition and interactions have the potential to significantly alter ecosystem dynamics as native flora and fauna adjust to direct and indirect effects of novel species (Tylianakis et al., 2008). Managers must understand and adapt to shifting species' assemblages in Southeast Alaska and elsewhere, making baseline studies such as this increasingly valuable, particularly when coupled with ongoing monitoring to track ecological community change through time.

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Chapter 2: Spatial and Temporal Partitioning of Mustelids in Southeast Alaska

Abstract

As species' distributions shift in response to changing environmental conditions, novel species interactions are emerging that are not yet well understood. As novel species enter existing ecological communities, their effects on fellow guild members are difficult to predict. Competition between species within the same guild can regulate populations and influence community structure, but niche partitioning is a way for sympatric species to coexist through differential use of space, time, and resources. The fisher (*Pekania pennanti*) is a newly colonizing carnivore species in Southeast Alaska (SEAK), and may impact the native SEAK forest carnivore guild through competitive interactions. I examined the potential for competition between fisher, American marten (*Martes americana*), and ermine (*Mustela erminea*) in SEAK during the winter of 2018 by investigating patterns of spatial and temporal overlap between the species at both local and broad scales, while accounting for important biotic and abiotic factors. Using data collected from 25 remote camera stations (50 unique cameras), and single and two-species occupancy models, I found evidence of temporal, rather than spatial, avoidance patterns among the SEAK mustelid species investigated. Best-supported two-species occupancy models did not support spatial avoidance hypotheses. Marten occupancy and detection were not impacted by fisher presence. Similarly, ermine detection and occupancy were not affected by fisher or marten presence, and fisher occupancy and detection was not affected by either marten or ermine presence. Analysis of diel activity patterns showed that ermine and marten adjusted their activity in the presence of fisher to reduce temporal overlap. Limited fisher detections made it unfeasible to examine shifts in fisher activity patterns in the presence of marten and ermine. Overall, these results show that mustelids in SEAK during winter display a pattern of hierarchical temporal partitioning, with smaller-bodied mustelids avoiding larger, more dominant mustelids. However, if the fisher population continues to expand, these species interactions may shift and increase the potential for competition between fisher, marten, and ermine in SEAK.

Introduction

Competition among species within the same ecological guild can regulate populations and influence community structure (Palomares & Caro, 1999). Both direct and indirect competitive interactions have been documented in many carnivore communities, and can impact species at lower trophic levels through top-down control (Caro & Stoner, 2002; Donadio & Buskirk, 2006). These interactions can change the behavior of lower-rank competitors as well as regulate their population size and distribution (Prugh et al., 2009). The outcomes of these competitive interactions are often hierarchical in nature, with the larger-bodied competitors outcompeting smaller species for resources such as food and space (Donadio & Buskirk, 2006; Ripple et al., 2013).

To alleviate competition and enable coexistence, sympatric species that share similar geographic distributions may differentiate their use of resources at smaller scales (i.e., niche partitioning) (Chillo et al., 2010). This concept of niche partitioning has been exhibited through species' differential use of habitat, diet, and time periods during the daily cycle (Chiang et al., 2012). The degree of competition and niche partitioning in a community often depends on the scarcity of the resources in a system, where limited resources increases the likelihood of competitive interactions and the subordinate competitor must alter its behavior and/or resource utilization or else face population decline (Hayward & Slotow, 2009). The addition of a new species to an area can alter the established coexistence mechanisms within the native ecological community and result in changes in the dynamics of competition and niche partitioning among species in the system. The recent colonization of fisher in Southeast Alaska presents an excellent opportunity to learn about competition and niche partitioning among a guild when a new guild member joins the current community.

Fisher are a mid-sized forest carnivore, and a member of the mustelid family. The Mustelidae are a diverse family of mammalian carnivores that are often characterized by their long, tubular body shape, narrow necks, and short legs, although body mass may vary by orders of magnitude (King, 1989). Competitive interactions between mustelids have been documented, resulting in resource exclusion of the weaker competitor and even intraguild predation (Zielinski, 2000). The fisher (*Pekania pennanti*), American marten (*Martes pennanti*), and ermine (*Mustela erminea*) are three mustelid species that may compete in areas of distributional overlap, such as Southeast Alaska, and which can all occupy similar forested habitat types (Zielinski et al., 2004; Zielinski et al., 2017).

The fisher and marten, in particular, share similar morphology, diet, and spatial distributions (Manlick et al., 2017). The fisher is significantly larger than the marten (2–6 kg vs. 0.5 kg–1.4 kg) and can compete with marten for resources such as food and space (Ruggiero et al., 1994; Zielinski & Duncan, 2004). However, at fine scales, niche partitioning in both diet and spatial distribution has been exhibited between the two species which may allow for their coexistence (Fisher et al., 2013). Although fisher and marten can co-exist, Krohn et al. (1995) suggested that interference competition from fisher can potentially reduce the viability of sympatric marten populations. The fishers' large body size relative to marten allows it to prey on mid-sized mammals such as snowshoe hare (*Lepus americanus*) and porcupine (*Erethizon dorsatum*) (Lewis, 2014; Ruggiero et al., 1994). However, both fisher and marten opportunistically prey on a variety of taxa, ranging from small mammals to birds and insects (Raine, 1987; Zielinski et al., 1999).

The ermine is significantly smaller than both fisher and marten, weighing only 1/10th of the average mass of a marten (Woodford, 2008). Its small body size makes it susceptible to many natural predators such as raptors, coyotes, and foxes (Gilbert, 1970). Additionally, both fisher and marten prey on ermine (Gilbert, 1970; Thompson & Colgan, 1990; Weir & Harestad, 2005).

In terms of dietary niches, the body size of ermine makes it more suited to eating smaller prey items than the fisher and marten. Ermine mainly consume small mammals such as voles and mice, although larger prey including rabbits, birds, and even weasels have also been documented in their diets (Erlinge & Sandell, 1988; Piontek et al., 2015; Simms, 1979). This specialization in smaller mammals may reduce competitive interactions with other mustelids at times when larger prey is abundant for fisher and marten (Simms, 1979). However, overlap in diet between mustelid species has been shown to increase significantly in regions where larger prey is less available, and decreased prey availability in winter may lead to a seasonal increase in competition (Powell et al., 1997; Zielinski & Duncan, 2004). The effects of competition between fisher, American marten, and ermine could have unexpected effects on the ecological communities in their regions of overlap.

Temporally, mustelid activity patterns can shift with the seasons, and their activities are often closely tied with that of their prey (Zielinski, 2000). For example, marten increase their nocturnal activity in the winter in response to the shifting activity of squirrels and mice, although they still remain active during crepuscular hours (Clark et al., 1987). Similarly, ermine are typically most active at night during the main activity times of their prey, although these nocturnal activity patterns may also be a tactic to avoid competition by taking advantage of the nightly decrease in competitive pressures as marten and fisher activity patterns may peak at crepuscular and diurnal hours, respectively (Zielinski, 2000).

Spatially, fisher, marten, and ermine share similar geographic distributions (Simms, 1979; Zielinski & Duncan, 2004). However, the degree of finer scale distributional overlap between these species can vary based on the landscape in question. For instance, in the mountainous Sierra Nevada region, the fisher is found at low-to-mid elevations in areas of mixed conifer-deciduous forest, whereas marten and ermine are found at higher elevations, with marten associated with conifer forests and ermine associated with a wide variety of wooded areas with thick understory (Fagerstone, 1987; Zielinski et al., 2017). In such heterogeneous landscapes, overlap and the potential for competition between the species may be limited. However, homogeneous landscapes with less habitat variation and lower elevations can lead to a more significant overlap in distribution (Zielinski et al., 2017). A study conducted by Zielinski et al. (2017) examined niche overlap between marten and fisher across a climatic gradient in California. The study found that in sympatric zones, marten exhibited both temporal and spatial avoidance of fisher, where they were more prone to avoid areas of their home range that overlapped with fisher and more likely to utilize uncharacteristic habitats at lower elevations (Zielinski et al., 2017). Ermine may also display spatial and temporal avoidance of both marten and fisher (Halle & Stenseth, 2000; St-Pierre et al., 2006). The hierarchical, body size-dictated dominance structure exhibited by the mustelid guild makes it likely that ermine are subordinate to fisher and marten and would therefore display these similar avoidance patterns to these species in areas of sympatry. In this study, the region of interest consists of few habitat types and is restricted to low elevations, making significant distributional overlap likely and increasing the opportunity for both direct and indirect interactions between fisher, marten, and ermine.

Changes in marten and ermine abundance and distribution as a result of fisher colonization could impact Southeast Alaska ecologically, economically, and culturally. As a predator, marten and ermine can regulate prey abundance and play a role in structuring ecosystems through top-down effects (Molsher et al., 2017; Ripple et al., 2013). Ermine, in particular, have been shown to play a major role in controlling vole population cycles, and any changes in ermine and marten predation patterns due to fisher colonization could have detrimental effects on prey species and alter ecosystem dynamics (Fagerstone, 1987; Ripple et al., 2013). Culturally and economically, the marten plays a significant role in the Alaskan fur trade with over 17,000 marten harvested from 2010–2015 (Parr, 2016). Should the fisher become more abundant, a portion of marten trapping efforts could be reallocated to fisher trapping as their average fur price is similar (Parr, 2016). Ermine are also trapped in Alaska to a lesser extent, with their pelts selling for a fraction of what fisher and marten furs are currently worth (Parr, 2016). However, many mustelid species are susceptible to by-catch, and may be accidentally killed in traps not intended for them (Proulx, 2000). Increased trapping efforts in SEAK associated with fisher colonization may unintentionally increase ermine trap-related

mortalities and alter their population structure. Overall, little is known about the ecological and economic impacts this novel fisher population have on the region and to the native SEAK mustelids if expansion continues.

The goal of this chapter was to determine the potential for fisher, American marten, and ermine competition in the region by examining temporal and spatial aspects of mustelid niche overlap. I accomplished this by 1) examining spatial co-occurrence patterns of fisher, marten, and ermine, taking into account important biotic and abiotic factors, 2) quantifying temporal overlap of the species, and 3) exploring whether individual species' activity patterns shifted in the presence of the other mustelids.

Methods and Materials

Study Area

The study area is described in Chapter 1: Materials and Methods, *Study Area*.

Camera Grid Design

The camera grid design is outlined in Chapter 1: Materials and Methods, *Camera Grid Design*.

Quantifying Environmental Covariates

The process detailing further data collection and the determination of occupancy and detection covariates is outlined in Chapter 1: Materials and Methods, *Quantifying Environmental Covariates & Occupancy Modeling*.

Single-Species Occupancy Analysis

Before beginning to evaluate two-species occupancy models, I conducted single-species occupancy analyses for fisher, marten, and ermine. This step was used to identify key environmental covariates important for each species that could then be incorporated into the two-species occupancy models. Environmental characteristics can influence species' co-occurrence independently from competition, and the addition of key environmental covariates into two-species occupancy models

enables disentanglement of avoidance patterns from species' responses to environmental conditions (Scully et al., 2018). Important covariates for fisher detection and occupancy were determined in Chapter 1: Materials and Methods, *Occupancy Modeling & Model Fitting and Selection*; I repeated the same two-step approach to model selection (first fitting a detection model, then adding the occupancy model) for marten and ermine at both the site and grid scale, and fitted models from the same initial set of covariates used in fisher model selection, with all covariates standardized to have a mean of zero and a standard deviation of 1. As in the fisher models, 24 sites (48 cameras) were included in the final analysis after one site was discarded due to a lack of data caused by camera malfunctions.

Single-species occupancy models were run using 7-day occasions within the 'unmarked' package in program R, and resulting models were compared for best-fit using Akaike's information criterion with a bias-correction for small sample sizes (AIC_c) (Burnham & Anderson, 2002; Fiske & Chandler, 2011; Symonds & Moussalli, 2010). In model selection, models within 2 AIC_c units of the top model ($\Delta_i < 2$) are considered to have substantial support (Burnham & Anderson, 2002). However, increasing the number of parameters in an occupancy model with a small dataset can increase the likelihood of model overfitting (Burnham & Anderson, 2002), or cause spurious effects such as increased standard errors of estimates or prevention of model convergence. In order to mitigate the effects of over parameterizing the upcoming two-species models with covariates, in instances where models were within 2 $\Delta_i AIC_c$ of the top model, only top-model covariates were included in subsequent steps (e.g., the top detectability model was subsequently used in the single-species occupancy model; likewise, the top single-species environmental covariates were subsequently used in co-occupancy models). Models were assessed for goodness-of-fit following the protocol outlined in Chapter 1: Materials and Methods, *Model Fitting and Selection*.

Co-occurrence Analyses

Prior to running the two-species occupancy models, I developed hypotheses regarding the potential outcomes of each species interaction (Table 2.1). Single-season, two-species occupancy models were run using program PRESENCE 2.12.24 (Hines, 2006). Conditional two-species occupancy models allow the detection and occupancy of one species to be modeled against the presence of another species, while incorporating covariates that contribute to each species' occupancy and detection (Richmond et al., 2010). In conditional models, one species is considered dominant and the other subordinate, where the probability of occupancy and detection of the subordinate species is conditional on the presence of the dominant species (Richmond et al., 2010). I ran separate two-

species occupancy analyses to evaluate fisher–marten, fisher–ermine, and ermine–marten interactions. Due to the small number of fisher detections, AIC_c was used to rank models in all pairwise occupancy analyses. I ran each of these interactions at both the site and grid scale, for a total of six separate occupancy analyses. I assigned each species as dominant or subordinate based on published evidence regarding species interactions. Fisher was assigned as the dominant species in both combinations of species pairs in which it was included (fisher–marten and fisher–ermine) (McCann et al., 2010; Rosenzweig, 1966), whereas ermine was assumed to be the subordinate species in both (fisher–ermine and marten–ermine) (Gilbert, 1970; Thompson & Colgan, 1987). Marten was assigned as the subordinate species in the fisher–marten model and the dominant species in the marten–ermine model (McCann et al., 2010; Thompson & Colgan, 1987) (Figure 2.1).

I ran four models for each two-species combination to determine whether significant species interactions occurred, formulated to test if: 1) detection and occupancy of the subordinate species was dependent on the presence of the dominant species, 2) detection was dependent but occupancy was independent, 3) detection was independent but occupancy was dependent, and 4) both detection and occupancy of the subordinate species was independent from the presence of the dominant species. Species interactions were evaluated based on the species interaction factor (SIF, $\hat{\gamma}$) of the top models. A value of $\hat{\gamma} > 1$ implies that two species co-occur more often than expected by chance, $\hat{\gamma} < 1$ suggests species avoidance, and $\hat{\gamma} = 1$ indicates that the two species occur independently (Robinson et al., 2014).

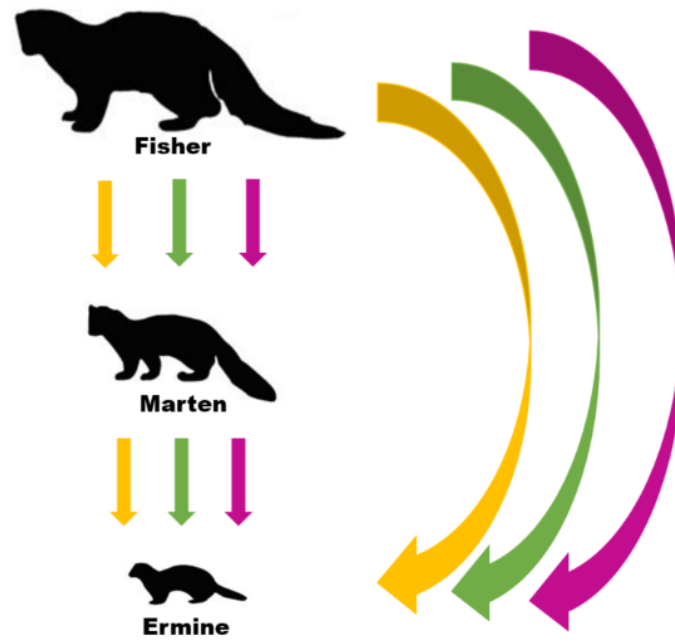


Figure 2.1. Schematic of predicted dominance structure between fisher, marten, and ermine with regards to effects on detection (yellow), occupancy (green), and temporal avoidance (purple). Arrows indicate the direction of effect (i.e. an arrow pointing to marten from fisher indicates that marten will be affected by fisher).

Table 2.1. Predicted effects of spatial co-occurrence between fisher, marten, and ermine in Southeast Alaska. $\hat{\gamma}$ = species interaction factor (SIF).

Fisher	Occupancy is independent from marten and ermine occupancy ($\hat{\gamma}=1$).
Marten	Occupancy is dependent on fisher occupancy, with marten avoiding areas where fisher are present ($\hat{\gamma} < 1$), but independent from ermine occupancy ($\hat{\gamma} = 1$).
Ermine	Occupancy is dependent on fisher and marten occupancy, with ermine avoiding areas where fisher and marten are present ($\hat{\gamma} < 1$).

Table 2.2. Predicted temporal avoidance patterns between fisher, marten, and ermine in Southeast Alaska.

Temporal Patterns
Fisher will not exhibit temporal avoidance, with activity patterns not differing between sites with vs. without ermine and marten co-occurrence.
Marten will exhibit temporal avoidance of fisher, with activity patterns changing significantly between sites with vs. without fisher co-occurrence.
Marten will not exhibit temporal avoidance of ermine, with activity patterns not differing between sites with vs. without ermine co-occurrence.
Ermine will exhibit temporal avoidance of both fisher and marten, with activity patterns changing significantly between sites with vs. without co-occurrence with fisher and marten.

Diel Activity Patterns

Time-stamped camera data can be used to examine species' activity patterns and to identify temporal overlap between species. Such information can be useful in identifying species' avoidance patterns and the potential for competitive interactions (Frey et al., 2017). Data from 50 camera stations were compiled for use in examining species' daily activity patterns. Due to the close proximity of cameras within each station (350 m), detections from the two cameras within a station cluster were combined for analyses to reduce the likelihood of capturing the same animal movement in a short time frame (Tambling et al., 2015). Many studies only consider detections > 30 minutes apart to be separate detections (Kelly & Holub, 2008; Monterroso et al., 2013), but in order to maximize small sample sizes with regards to species' camera detections while maintaining adequate temporal independence between detections at individual cameras, camera detections were considered separate observations when an animal completely left the field of view and was observed > 15 minutes apart. To account for daylight savings time, all detection times were standardized to Alaskan Standard Time (Coordinated Universal Time - 9 hours).

In order to identify the primary activity times of each species, the diel cycle was categorized into three periods: night (20:00–05:00), day (08:00–17:00), and crepuscular (05:00–08:00 & 17:00–20:00), and the proportion of detections within each diel category was determined for each species.

The coefficient of overlap (Δ) was calculated for each species pair. This value ranges from 0 to 1, with a value of zero indicating no overlap in activity patterns and a value of 1 indicating complete overlap (Monterroso et al., 2014). The value is taken from the minimum density function at each point in time of the activity curves of the two species being compared (Bu et al., 2016; Monterroso et al., 2014). Three distinct nonparametric estimators of the coefficient of overlap exist for circular data (Δ_1 , Δ_4 , Δ_5), with Ridout and Linkie (2009) determining that Δ_1 reduced the root mean square error (RMSE) in small sample sizes compared to the other two metrics (Ridout & Linkie, 2009). Because of this, Δ_1 was the estimator used in this analysis. Detections at all stations where the species pairs co-occurred were aggregated to produce a total density curve for each species at sites of overlap. The coefficient of overlap was designated as “low” if the coefficient value fell below 50%, “moderate” if the value ranged from 50–75%, and “high” if the overlap coefficient was > 75% (Monterroso et al., 2014). The 95% confidence interval for each overlap calculation was determined by bootstrapping 10,000 samples from the estimated density curves of each species (Bu et al., 2016). All activity patterns and overlap estimates were determined using the ‘overlap’ package in program R (Meredith & Ridout, 2018).

To determine whether or not fisher, marten, and ermine activity patterns shifted in each others' presence, I compared single species' activity patterns between stations where it co-occurred with the other mustelids versus stations where it did not. This was done by calculating the Watson-Wheeler test statistic (W , p value ≤ 0.05) within the 'circular' package in program R (Agostinelli & Lung, 2017). The Watson-Wheeler test assesses differences between nonparametric circular datasets (Zar, 2010), and as suggested by Fisher (1993), only distributions of ≥ 10 detections were considered. Prior to completing these comparisons, I developed hypotheses regarding the expected shifts in each species' activity pattern in the presence of the other mustelid species (Table 2.2).

Results

Approximately 84,800 remote camera photos were collected from January 15 through April 20, 2018 across 25 stations (50 unique cameras). In total, 15 species were detected, with fisher detected 26 times at 7 stations (9 unique cameras). Marten were detected 204 times at 14 stations (24 unique cameras), and ermine were detected 507 times at 24 stations (43 unique cameras). Total species detections are summarized in Table 2.3.

Table 2.3. Summary of species detections from January 2018–April 2018 in Southeast Alaska.

Focal Species	Detections
Fisher (<i>Pekania pennanti</i>)	26
American Marten (<i>Martes americana</i>)	204
Ermine (<i>Mustela erminea</i>)	507
Non-Focal Species	
Wolverine (<i>Gulo gulo</i>)	11
American Mink (<i>Neovison vison</i>)	23
Wolf (<i>Canis lupus</i>)	2
Sitka Black-Tailed Deer (<i>Odocoileus hemionus sitkensis</i>)	8
Porcupine (<i>Erethizon dorsatum</i>)	78
River Otter (<i>Lontra canadensis</i>)	2
Red Squirrel (<i>Sciurus vulgaris</i>)	988
Bird (multiple sp.)	35
Mouse/Vole (multiple sp.)	32
Domestic Dog (<i>Canis lupus familiaris</i>)	45
Domestic Cat (<i>Felis catus</i>)	1
Human (<i>Homo sapien</i>)	6

Single-Species Occupancy Analyses

Fisher

Detection and Site Scale Occupancy

As reported in Chapter 1, snow density was shown to have a negative effect on fisher detection, whereas the behavioral effect (BK) covariate was found to have a positive effect on fisher detection. Average snow density and elevation were both found to have positive effects on fisher occupancy.

Grid Scale Occupancy

As reported in Chapter 1, vegetation height was found to have a positive effect on fisher occupancy.

Marten

Detectability Model

Mean marten detection was predicted to be 0.32 (95% CI = 0.22, 0.45). Using the same model selection process employed for fisher, I found that behavioral effect and snow density were the top detectability covariates and both had positive effects on marten detection (BK = 0.74, 95% CI = 0.44, 1.04; snow density = 0.47, 95% CI = 0.04, 0.89) (Table 2.4). When comparing univariate model rankings of each covariate, the behavioral effect model had the lowest AIC_c score, followed by the snow density and temperature models. Building both behavioral effect and snow density into a new model yielded a lower AIC_c score. Adding temperature to that model increased the AIC_c score slightly, but Δ_i AIC_c remained within 2 units of the top model. In an effort to minimize the number of parameters to be incorporated into the two-species models, only the top model parameters, behavioral effect and snow density, were included in the best detectability model moving forward. Goodness-of-fit tests indicated that there was no evidence of overdispersion in this dataset (site scale: $\hat{c} = 0.75$ (p value = 0.67); grid scale: $\hat{c} = 0.68$ (p value = 0.82)) (Burnham & Anderson, 2002).

Table 2.4. Model rankings for detectability models with constant occupancy for fisher, marten, and ermine in Southeast Alaska. Intercept values for both detection (p) and occupancy (psi) are reported as well as the top detectability covariates of snow depth, snow density (SnD), temperature (temp), and behavioral effect (BK). AIC_c, delta AIC_c, and weight of the top models are also reported. AIC_c = Akaike information criterion corrected for small sample sizes.

Fisher	p(Int)	psi(Int)	p(snow depth)	p(SnD)	p(temp)	p(BK)	AIC_c	delta AIC_c	weight
p(SnD + BK)	-1.95	-0.55		-1.17		0.36	101.50	0.00	0.61
p(SnD)	-1.59	-0.67		-1.29			103.49	1.99	0.23
p(snow depth + SnD + BK)	-1.98	-0.53	-0.25	-1.07		0.36	104.41	2.90	0.14
Marten									
p(SnD + BK)	-0.81	0.53		0.47		0.74	237.86	0.00	0.44
p(SnD + BK + temp)	-0.78	0.50		0.46	0.31	0.71	238.03	0.17	0.40
p(BK)	-0.66	0.40				0.75	239.81	1.95	0.16
Ermine									
p(temp + BK)	0.09	3.58			0.51	0.87	325.34	0.00	0.56
p(snow depth + BK + temp)	0.05	9.59		-0.27	0.48	0.89	325.90	0.55	0.43
p(BK)	0.09	3.60				1.04	333.97	8.63	0.01

Site Scale Occupancy

Mean marten occupancy at the site scale was predicted to be 0.63 (95% CI= 0.40, 0.81). The top-ranked model for marten occupancy included an effect of snow density and previous detection on current detectability, and no covariate effects on occupancy (Table 2.5). This top-ranked model was subsequently incorporated into the two-species models. Models incorporating average station snow density and station elevation effects on occupancy ranked second and third, respectively, in AIC_c rankings (Table 2.5). While the AIC_c values of individual covariate models for snow depth and elevation were each within 2 Δ_i AIC_c units of the top model, when incorporated together, Δ_i AIC_c increased to > 2 from the top model. These results show that station snow density and station elevation potentially have a positive effect on marten occupancy at the site scale even though they were not included in the top AIC_c model (snow density = 0.70, 95% CI = 0.29, 1.69; elevation = 0.81, 95% CI = -0.71, 2.32).

Grid Scale Occupancy

Estimated mean marten occupancy at the grid scale was 0.53 (95% CI= 0.23, 0.82). Both vegetation height and elevation have a positive effect on marten occupancy (vegetation height = 1.25, 95% CI = 0.007, 2.49; elevation = 1.85, 95% CI = 0.01, 3.67) (Table 2.6). When the two grid scale occupancy covariates of vegetation height and elevation were separately incorporated into occupancy models, the model incorporating elevation had the lowest AIC_c score. When vegetation height was added to the elevation model, the model produced the lowest AIC_c score, making it the top grid scale marten occupancy model.

Ermine

Detectability Model

Mean detection was predicted to be 0.39 (95% CI = 0.29, 0.50). Both temperature and behavioral effect had positive effects on ermine detection (temperature = 0.51, 95% CI = 0.21, 0.82; BK = 0.87, 95% CI = 0.57, 1.16) (Table 2.4). Using the same process as above, I found that the univariate model for behavioral effect had the lowest AIC_c score, followed by temperature, snow depth, and snow density, respectively. Incorporating both behavioral effect and temperature into the model yielded a lower AIC_c score. Building a model that added in snow depth with temperature and behavioral effect decreased the model fit, with Δ_i AIC_c 0.55 units from the top model. As above, even though this model incorporating snow depth with temperature and behavioral effect was within 2 Δ_i

AIC_c units of the top model (which incorporates only behavioral effect and temperature), it was not included in the models moving forward to minimize the number of parameters in the two-species models.

Site and Grid Scale Occupancy

Mean ermine occupancy was predicted to be 0.97 (95% CI = 0.60, 1.00). Incorporating occupancy covariates into the ermine models did not improve model fit over the null occupancy model at either scale (Tables 2.5 & 2.6). At the site scale, the model incorporating average snow density was 1.02 AIC_c units higher than the top (null) model, and showed snow density to have a negative effect on ermine occupancy (-2.60, 95% CI = -6.89, 1.69). However, the 95% upper confidence limit was positive, showing that this negative effect is not significant. At the grid scale, the model incorporating elevation was within 0.95 AIC_c units of the top (null) model, indicating that elevation may have a negative effect on ermine occupancy (-1.52, 95% CI = -3.73, 0.70). However, once again, the 95% confidence limit overlapped zero. Goodness-of-fit tests indicated that there was no evidence of overdispersion in this dataset (site scale: $\hat{c} = 0.67$ (p value = 0.07); grid scale: $\hat{c} = 0.64$ (p value = 0.91)) (Burnham & Anderson, 2002).

Co-occurrence Analyses

Fisher–Marten

At the site scale, the best-supported model indicated that marten detection and occupancy were independent from fisher presence (Table 2.7). This model had a substantial amount of support with a model weight of 0.92. The second-highest ranking model suggested that marten occupancy was dependent on fisher occupancy, but ranked $> 4 \Delta_i$ AIC_c units higher than top model.

Similarly, at the grid scale, the best-supported model indicated that marten detection and occupancy were independent from fisher presence (Table 2.7). The second-highest ranking model indicated that marten occupancy was negatively affected by fisher presence ($\hat{\gamma} = 0.52$, 95% CI = 0.24, 0.81). This model was only Δ_i 3.17 units higher than the best-fit model, indicating that there is an appreciable amount of support for this candidate model, although it had significantly less AIC_c weight (0.16) than the top model (0.80).

Table 2.5. Model rankings for occupancy models for fisher, marten, and ermine at the site scale in Southeast Alaska. Intercept values for both detection (p) and occupancy (psi) are reported. The parameter estimates for the top detectability covariates (behavioral effect (BK), snow density (SnD), and temperature (temp)) and the top occupancy covariates (station elevation, canopy cover (CC), vegetation height (VH), and snow density (SnD)) are also reported. AIC_c, delta AIC_c, and weight of the top models are also reported. AIC_c = Akaike information criterion corrected for small sample sizes.

Fisher	p(Int)	psi(Int)	p(BK)	p(SnD)	p(temp)	psi(elevation)	psi(CC)	psi(VH)	psi(SnD)	AIC_c	delta AIC_c	weight
p(BK + SnD) psi(VH)	-2.45	-0.43	0.49	-0.97				27.64		93.55	0.00	0.84
p(BK + SnD) psi(elevation + SnD)	-1.92	-1.23	0.35	-1.19		-2.98			1.91	98.55	5.00	0.07
p(BK + SnD) psi(SnD)	-2.24	0.03	0.42	-1.31					1.75	100.33	6.78	0.03
Marten												
p(BK + SnD) psi(.)	-0.81	0.53	0.74	0.47						237.86	0.00	0.29
p(BK + SnD) psi(SnD)	-0.76	0.46	0.72	0.41					0.70	239.09	1.23	0.16
p(BK + SnD) psi(elevation)	-0.81	0.63	0.74	0.46		0.81				239.18	1.32	0.15
Ermine												
p(BK + temp) psi(.)	0.09	3.58	0.87		0.51					325.34	0.00	0.28
p(BK + temp) psi(SnD)	0.10	5.28	0.85		0.52				-2.60	326.36	1.02	0.17
p(BK + temp) psi(CC)	0.10	23.29	0.85		0.52		-64.49			326.52	1.18	0.16

Table 2.6. Model rankings for occupancy models for fisher, marten, and ermine at the grid scale in Southeast Alaska. Intercept values for both detection (p) and occupancy (psi) are reported. The parameter estimates for the top detectability covariates (behavioral effect (BK), snow density (SnD), and temperature (temp)) and the top occupancy covariates (station vegetation height (VH) and elevation) are also reported. AIC_c, delta AIC_c, and weight of the top models are also reported. AIC_c = Akaike information criterion corrected for small sample sizes.

Fisher	p(Int)	psi(Int)	p(BK)	p(SnD)	p(temp)	psi(VH)	psi(elevation)	AIC_c	delta AIC_c	weight
p(BK + SnD) psi(VH + elevation)	-2.56	-1.68	0.51	-1.04		38.63	-29.93	98.67	0.00	0.50
p(BK + SnD) psi(VH)	-2.44	0.07	0.48	-0.99		2.97		99.60	0.94	0.32
p(BK + SnD) psi(.)	-1.95	-0.55	0.36	-1.17				101.50	2.83	0.12
Marten										
p(BK + SnD) psi(VH + elevation)	-0.74	0.64	0.71	0.40		1.25	1.85	234.25	0.00	0.57
p(BK + SnD) psi(elevation)	-0.78	0.79	0.72	0.44			1.53	235.72	1.47	0.27
p(BK + SnD) psi(.)	-0.81	0.53	0.74	0.47				237.86	3.61	0.09
Ermine										
p(BK+ temp) psi(.)	0.09	3.58	0.87		0.51			325.34	0.00	0.51
p(BK + temp) psi(elevation)	0.10	4.50	0.85		0.52		-1.52	326.30	0.95	0.32
p(BK + temp) psi(VH)	0.09	3.71	0.86		0.51	0.80		328.24	2.89	0.12

Fisher–Ermine

At the site scale, the best-fit model indicated that ermine detection and occupancy were independent from fisher presence, with a model weight of 0.93 (Table 2.8). The detection-independent, occupancy-dependent model was $> 5 \Delta_i AIC_c$ units from the top model, and had significantly less support (model weight = 0.07), indicating that there was not substantial evidence that ermine occupancy was affected by fisher presence.

Grid scale results were similar to those above, with the top model indicating ermine independence from fisher presence for both occupancy and detection (Table 2.8). The next-best model suggested that ermine occupancy was dependent on fisher presence and was within $2 \Delta_i AIC_c$ units. However, the SIF value was 1.04 (95% CI = 0.95, 1.13), and the confidence interval included 1, indicating that there is not significant support that ermine occupancy is dependent on fisher presence.

Marten–Ermine

The top model at the site scale for marten–ermine interactions indicated that ermine detection and occupancy were independent from marten presence. The next-highest ranking model indicated that ermine detection was dependent on marten presence whereas ermine occupancy was independent. This model had an average detection SIF of 0.67 (95% CI = 0.45, 0.89) which suggests ermine avoidance of marten (Table 2.9). This model ranked $2.77 \Delta_i AIC_c$ units more than the top performing model, indicating that there is support for this interaction, although it was significantly less weighted (0.17) than the top model (0.70).

At the grid scale, the best-supported model indicated that ermine detection and occupancy were independent from marten presence. The next-highest ranking model, indicating ermine detection independence but occupancy dependence, was $> 4 \Delta_i AIC_c$ units from the top model, indicating that there is considerably less support for this model than the top model.

Table 2.7. AIC_c output table ranking best-fit models for marten responses to co-occupancy with fisher at both site and grid scales in Southeast Alaska. Delta AIC_c, and model weight of each model are also reported. AIC_c = Akaike information criterion corrected for small sample sizes.

Fisher Effects on Marten Detection and Occupancy

Site Scale	AIC_c	delta AIC_c	weight
detection independent, occupancy independent	385.53	0	0.92
detection independent, occupancy dependent	390.36	4.83	0.08
detection dependent, occupancy independent	402.67	17.14	<0.01
detection dependent, occupancy dependent	410.9	25.37	0

Grid Scale	AIC_c	delta AIC_c	weight
detection independent, occupancy independent	375.96	0	0.80
detection independent, occupancy dependent	379.13	3.17	0.16
detection dependent, occupancy independent	381.95	5.99	0.04
detection dependent, occupancy dependent	389.47	13.51	<0.01

Table 2.8. AIC_c output table ranking best-fit models for ermine responses to co-occupancy with fisher at both site and grid scales in Southeast Alaska. Delta AIC_c, and model weight of each model are also reported. AIC_c = Akaike information criterion corrected for small sample sizes.

Fisher Effects on Ermine Detection and Occupancy

Site Scale	AIC_c	delta AIC_c	weight
detection independent, occupancy independent	465.96	0	0.93
detection independent, occupancy dependent	471.2	5.24	0.07
detection dependent, occupancy independent	486.16	20.2	0
detection dependent, occupancy dependent	495.49	29.53	0

Grid Scale	AIC_c	delta AIC_c	weight
detection independent, occupancy independent	460.81	0	0.55
detection independent, occupancy dependent	462.52	1.71	0.23
detection dependent, occupancy independent	463.23	2.42	0.16
detection dependent, occupancy dependent	465.23	4.42	0.06

Table 2.9. AIC_c output table ranking best-fit models for ermine responses to co-occupancy with marten at both site and grid scales in Southeast Alaska. Delta AIC_c, and model weight of each model are also reported. AIC_c = Akaike information criterion corrected for small sample sizes.

Marten Effects on Ermine Detection and Occupancy

Site Scale	AIC_c	delta AIC_c	weight
detection independent, occupancy independent	624.56	0	0.70
detection dependent, occupancy independent	627.33	2.77	0.17
detection independent, occupancy dependent	628.09	3.53	0.12
detection dependent, occupancy dependent	633.35	8.79	<0.01

Grid Scale	AIC_c	delta AIC_c	weight
detection independent, occupancy independent	623.13	0	0.91
detection independent, occupancy dependent	628.09	4.96	0.08
detection dependent, occupancy independent	631.53	8.4	0.01
detection dependent, occupancy dependent	640.48	17.35	<0.01

Diel Activity Patterns

Across all stations, the three species exhibited significantly different activity patterns ($W = 19.69$, p value < 0.001) (Figure 2.2). Fisher, marten, and ermine all co-occurred with each other at four stations, although I was unable to determine whether these activity patterns in areas of overlap were significantly different due to the low number of fisher detections at these stations (8 detections). 53.8% of fisher detections occurred during the day, 38.5% occurred at night, and 7.7% occurred at dawn, with no detections at dusk. In contrast, 62.7% of marten detections occurred at night whereas only 12.7% of detections occurred during daylight hours. 24.5% of marten detections occurred during dawn and dusk. 63.3% of ermine detections were at night, 17.0% occurred during the day, and 19.7% occurred at dawn and dusk.

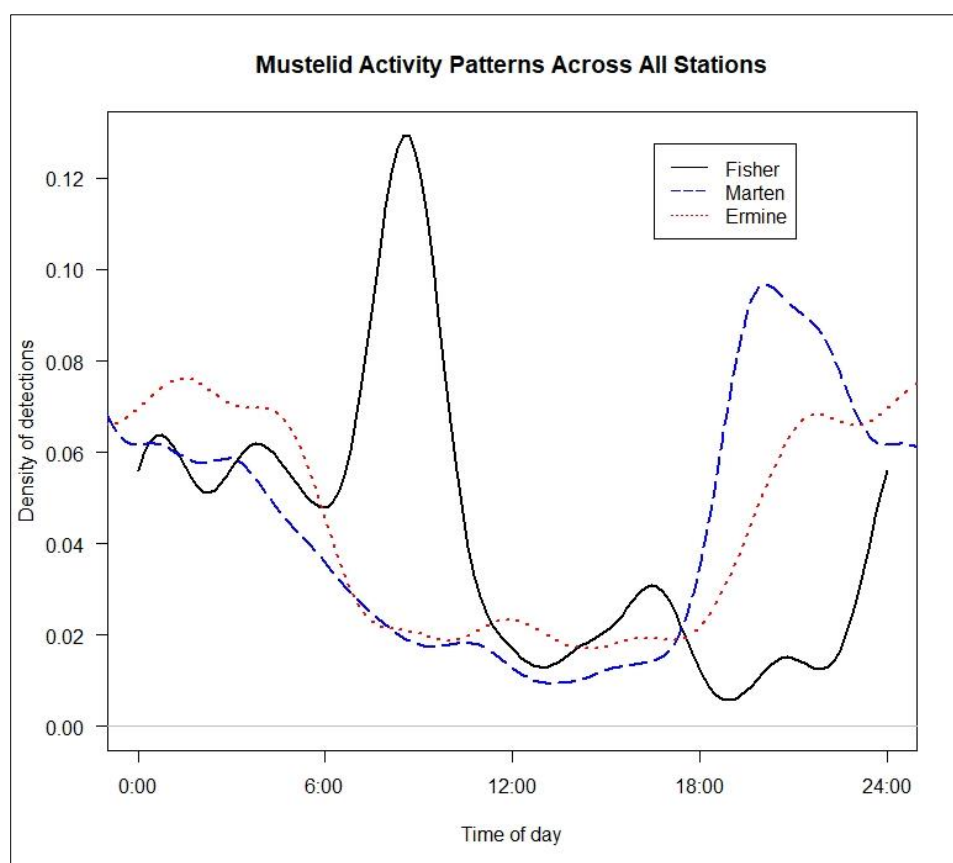


Figure 2.2. Fisher, marten, and ermine diel activity patterns across all stations in Southeast Alaska.

Fisher–Marten Temporal Overlap

Fisher and marten co-occurred at 4 stations. At those stations, fisher were detected 8 times and marten were detected 11 times. The overlap estimate for activity patterns at stations of co-occurrence was deemed low at 0.44 (95% CI = 0.18, 0.69) (Figure 2.3). The Watson-Wheeler test requires a sample size of at least 10 to determine significant differences in datasets, and as a result I was unable to determine whether fisher and marten activity patterns were significantly different at stations of co-occurrence. However, activity patterns of the two species across all stations, regardless of co-occurrence, was significantly different ($W = 12.71$, p value = 0.002). The coefficient of overlap for fisher and marten activity patterns across all stations, regardless of co-occurrence, was moderate at 0.61 (95% CI = 0.47, 0.75).

Fisher occurred at 3 stations where no marten were detected. At these stations, fisher were detected 18 times. Because of the low sample size mentioned above, I was also unable to determine whether fisher activity patterns differed significantly between areas of co-occurrence with marten and areas where no marten were present. Activity pattern plots indicate that marten are present at stations where fisher are most active during mid-morning and afternoon hours and least active during the early morning (~3:00) and evening hours (~18:00–21:00) (Figure 2.6a).

Marten were detected a total of 141 times at 11 different stations where fisher were not detected. Marten activity patterns differed significantly ($W = 9.88$, p value = 0.01) between stations with fisher presence versus stations without. Activity patterns indicate that marten adjust their activity to be more active during dawn and midday in areas of co-occurrence, with activity peaks ~06:00 and ~13:00 (Figure 2.6a). Fisher activity patterns peak during the late morning (~10:00), and although martens adjust their activity patterns in areas of co-occurrence to be more active during the day, they exhibit two peaks on either side of prime fisher activity (~06:00 and ~13:00), becoming markedly less active during peak fisher activity. The inverse peaks and lulls of marten activity patterns in relation to fisher activity, as well as the shifting activity of marten in the presence of fisher, show that marten exhibit temporal avoidance of fisher on the SEAK landscape.

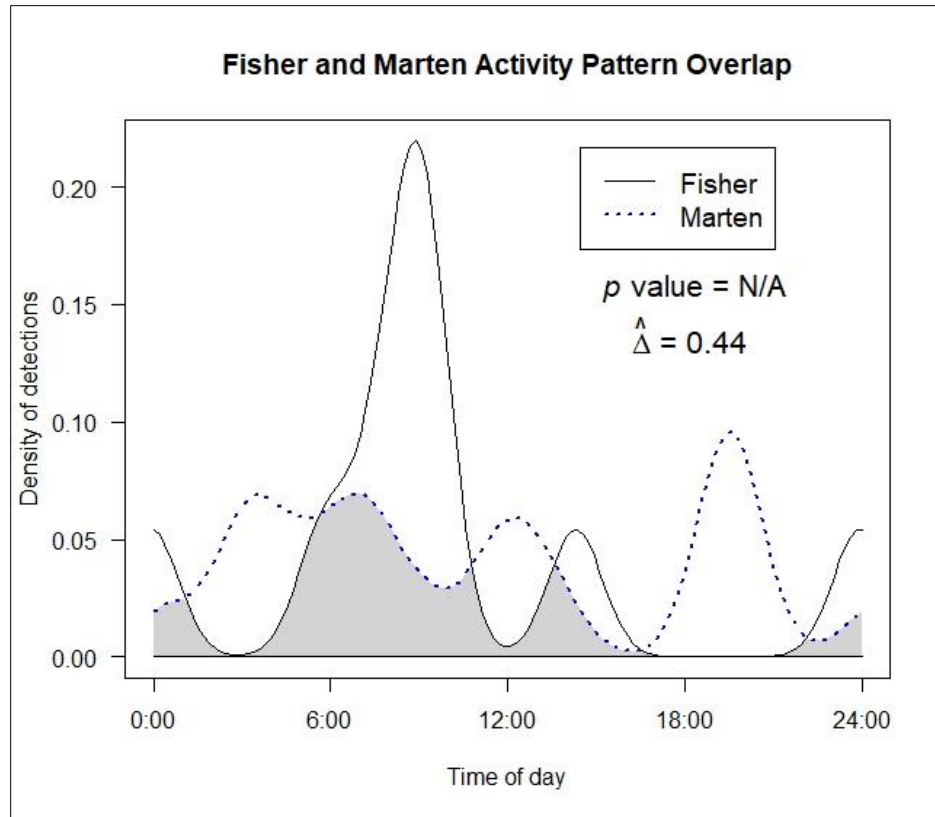


Figure 2.3. Fisher and marten activity pattern overlap plots at stations of co-occurrence in Southeast Alaska. The coefficient of overlap ($\hat{\Delta}$) is reported, as well as the p value representing whether significant differences in activity patterns exist between the two species, as determined through the Watson-Wheeler test. Note: There were not enough fisher detections in areas of overlap with marten to determine whether significant differences between fisher and marten activity patterns existed (p value = N/A).

Fisher–Ermine Temporal Overlap

Fisher and ermine co-occurred at 7 stations. At those stations, fisher were detected 26 times and ermine were detected 180 times. There was moderate activity pattern overlap between the two species at these stations ($\hat{\Delta}_1 = 0.63$, 95% CI = 0.47, 0.79), and fisher and ermine activity patterns were significantly different ($W = 10.20$, p value = 0.01) (Figure 2.4). Fisher were not detected at any stations where ermine were not present, so I was unable to determine whether fisher activity patterns differed significantly in the presence of ermine (Figure 2.6b). However, ermine were detected 327 times at 16 stations where fisher were not detected. Ermine activity patterns differed significantly between areas of fisher co-occurrence and areas without ($W = 6.55$, p value = 0.04). Ermine activity in the presence of fisher increased in the early morning hours (~03:00) and decreased during daylight hours when compared to its activity patterns when fisher were absent (Figure 2.8a).

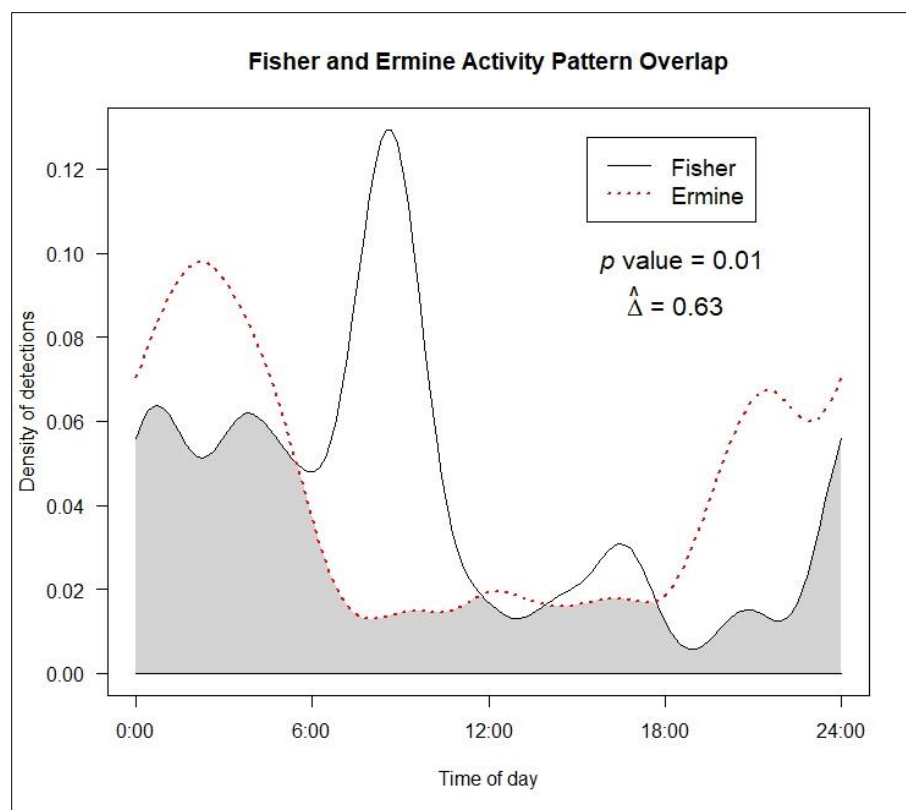


Figure 2.4. Fisher and ermine activity pattern overlap plots at stations of co-occurrence in Southeast Alaska. The coefficient of overlap ($\hat{\Delta}$) is reported, as well as the p value representing whether significant differences in activity patterns exist between the two species, as determined through the Watson-Wheeler test.

Marten–Ermine Temporal Overlap

Marten and ermine co-occurred at 11 stations. At those stations, marten were detected 120 times and ermine were detected 217 times. The coefficient of overlap at stations of co-occurrence was high ($\hat{\Delta}_1 = 0.79$, (95% CI = 0.70, 0.87), although activity patterns between marten and ermine were significantly different ($W = 9.92$, p value = 0.01) (Figure 2.5). Marten occurred at 2 stations where no ermine were detected, and were detected at those stations 39 times. Marten activity patterns did not differ significantly between stations with and without ermine ($W = 1.10$, p value = 0.58) (Figure 2.7b). Ermine occurred at 10 stations where no marten were detected and were detected at those stations 250 times. Ermine activity patterns differed significantly between stations with and without marten ($W = 6.84$, p value = 0.03) (Figure 2.8b). Ermine activity patterns showed a slight increase in activity in the early morning hours (peaking at ~03:00) when marten were present and a slight decrease during the day, when compared with ermine activity patterns in the absence of marten.

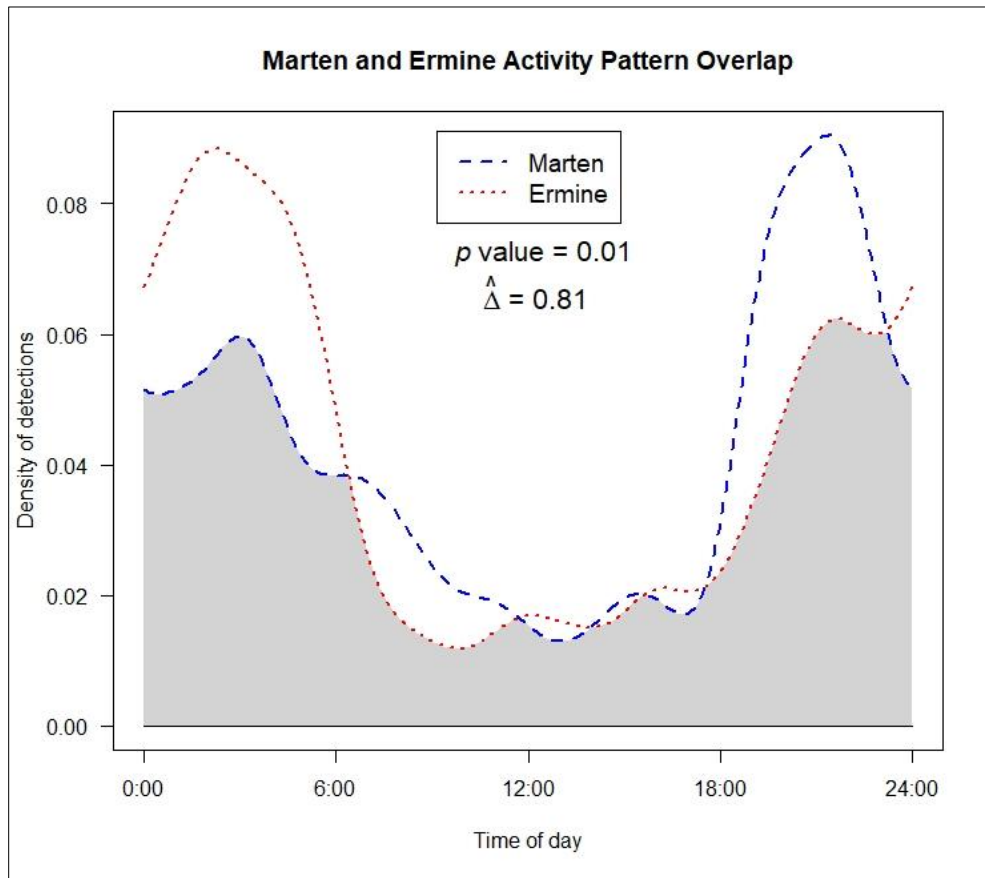


Figure 2.5. Marten and ermine activity pattern overlap plots at stations of co-occurrence in Southeast Alaska. The coefficient of overlap ($\hat{\Delta}$) is reported, as well as the p value representing whether significant differences in activity patterns exist between the two species, as determined through the Watson-Wheeler test.

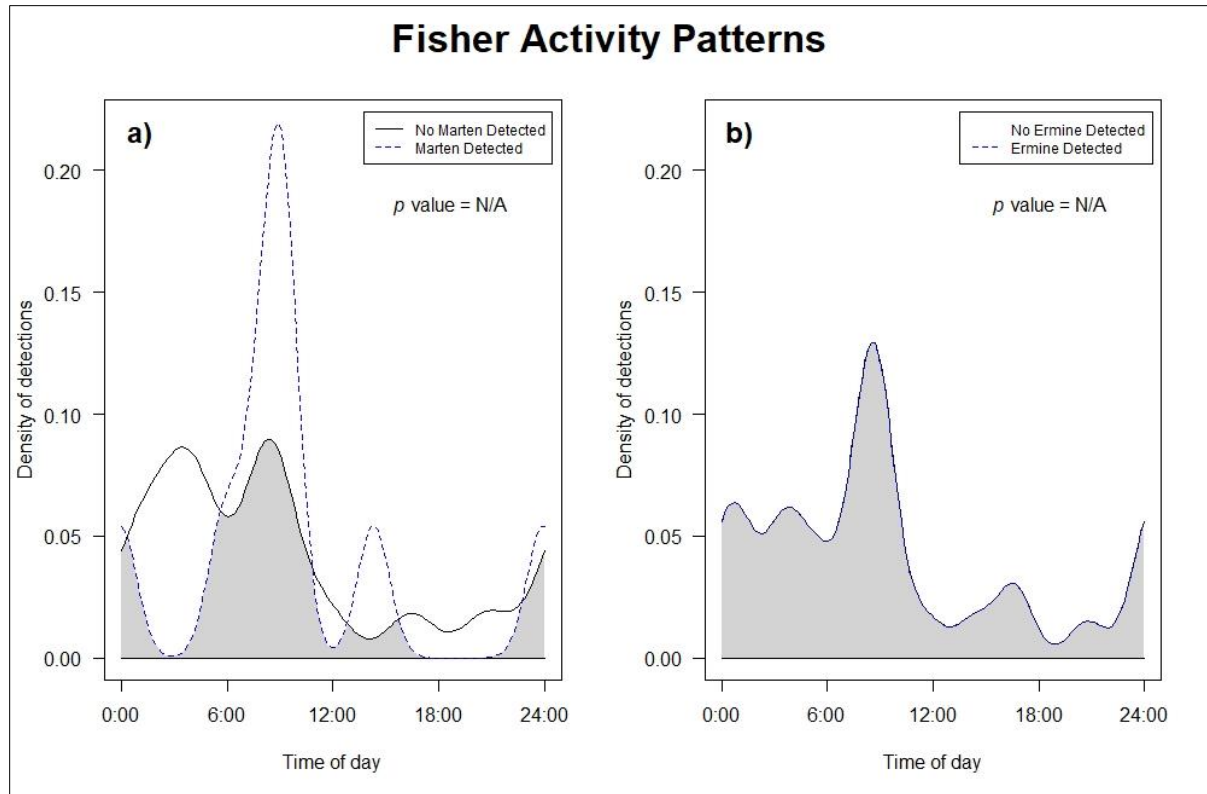


Figure 2.6. Differences in fisher activity patterns between areas where they co-occur with marten in Southeast Alaska and areas where they do not (a), and areas where they co-occur with ermine in Southeast Alaska and areas where they do not (b). p values represent whether or not significant differences in fisher activity patterns exist between areas of co-occurrence with marten and ermine and areas without, as determined through the Watson-Wheeler test. Note: Fisher detections (< 10) in areas of co-occurrence with marten were insufficient to determine if significant differences in fisher activity patterns occurred (p value = N/A). In addition, ermine occurred at all stations where fisher were detected, making it impossible to test for significant differences in fisher activity patterns (p value = N/A).

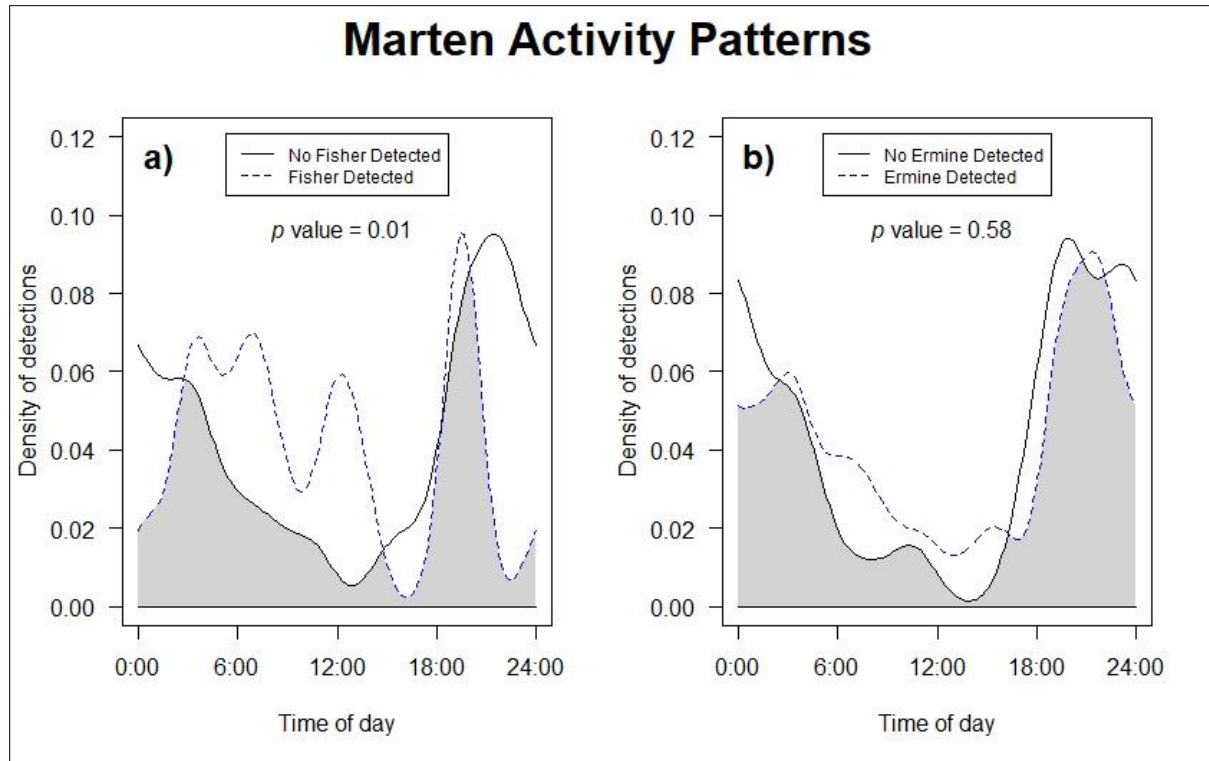


Figure 2.7. Differences in marten activity patterns between areas where they co-occur with fisher in Southeast Alaska and areas where they do not (a), and areas where they co-occur with ermine in Southeast Alaska and areas where they do not (b). p values represent whether or not significant differences in marten activity patterns exist between areas of co-occurrence with fisher and ermine and areas without, as determined through the Watson-Wheeler test.

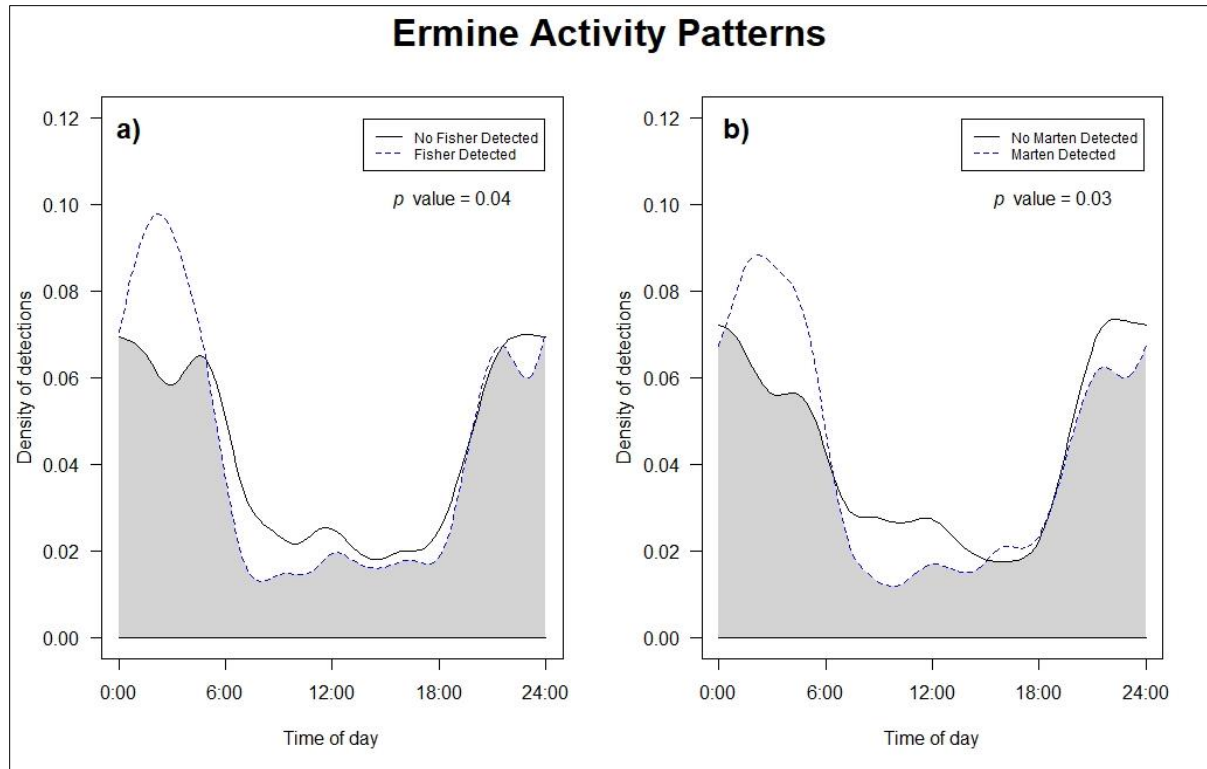


Figure 2.8. Differences in ermine activity patterns between areas where they co-occur with fisher in Southeast Alaska and areas where they do not (a), and areas where they co-occur with marten in Southeast Alaska and areas where they do not (b). p values represent whether or not significant differences in ermine activity patterns exist between areas of co-occurrence with fisher and marten and areas without, as determined through the Watson-Wheeler test.

Discussion

This analysis demonstrated how mustelid species in SEAK use space and time, and how these patterns may be contributing to their coexistence. Fisher, marten, and ermine displayed hierarchical temporal avoidance patterns, with the smaller-bodied mustelids shifting activity patterns in the presence of the larger mustelids. However, I found no evidence of spatial partitioning, with the best-supported co-occupancy models showing that both detection and occupancy of the subordinate species were independent from the presence of the dominant species in all pairwise interactions (Figure 2.9).

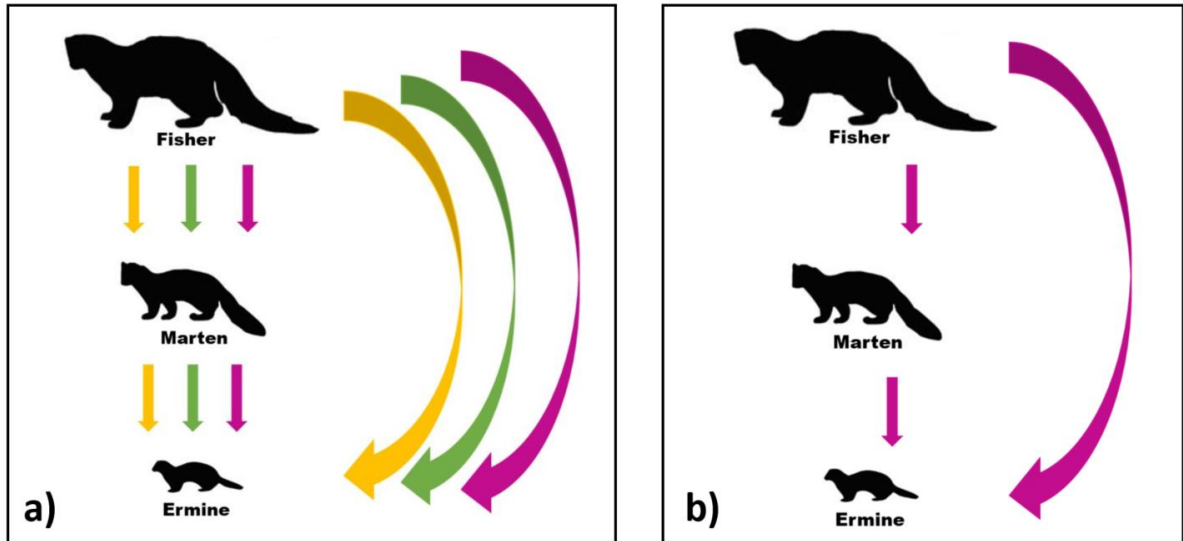


Figure 2.9. Schematic of predicted (a) versus observed (b) dominance structure between fisher, marten, and ermine with regards to effects on detection (yellow), occupancy (green), and temporal avoidance (pink). Arrows indicate the direction of effect (i.e. an arrow pointing to marten from fisher indicates that marten will be affected by fisher).

Although the top fisher–marten co-occupancy models suggest that marten occupancy and detection are unlikely to be strongly impacted by fisher presence, there was a minor indication that marten occupancy was negatively impacted by fisher occupancy at broad scales. One potential explanation is that fisher density in SEAK was not high enough to have a strong impact on marten occupancy. A study conducted in Wisconsin found that recovery of a reintroduced marten population was likely limited by competition with fisher (Manlick et al., 2017). After initial reintroduction efforts of both species, fisher in the study became well-established while marten remained endangered in the region, even after a subsequent augmentation (via translocations) of the initial reintroduced marten population (Manlick, et al., 2017). The study found that there was high fisher occupancy in the region but low marten occupancy (Manlick et al., 2017). Additionally, the fisher population had been reintroduced significantly earlier than the marten population (1966 vs. 1987) (Williams et al., 2007); this likely allowed ample time for fisher to increase in abundance on the landscape before the marten reintroduction effort, potentially putting marten at a large competitive disadvantage when suddenly added to the area as the new, smaller-bodied sympatric mustelid. Should the fisher population in SEAK increase in abundance, it could potentially put more competitive pressure on the native marten in the area.

Similarly, the lack of an effect of fisher on ermine detectability or occupancy, despite being identified as the likely dominant species, can perhaps be explained by the discrepancies in the abundances of the two species. Ermine are well-dispersed throughout the study area with many more detections than fisher, which had few detections and are sparsely distributed on the landscape in comparison. Again, it is possible that fisher abundance is not large enough to have a notable effect on the subordinate species' occupancy. However, increasing the abundance of guild members has been shown to alter ermine activity and fine-scale habitat selection as a way to avoid competition (St-Pierre et al., 2006); if the fisher population continues to establish in the area, any effects of fisher presence on marten and ermine occupancy may become more pronounced.

Alternately, it is possible that fisher may have a greater impact on marten than ermine due to the higher degree of niche overlap between the two species; ermine may then experience trickle-down effects of fisher presence through their interaction with the intermediate marten, rather than through direct interaction with fisher (e.g., if marten spatio-temporally avoid fisher, ermine may even benefit from fisher presence). A similar among-guild cascade occurs between wolves (*Canis lupus*), coyotes (*Canis latrans*), and red foxes (*Vulpes vulpes*) (Levi & Wilmers, 2012). Interference competition takes place between wolves and coyotes as a result of their similar body size and dietary preferences, and wolves can limit coyote populations in areas of sympatry (Levi & Wilmers, 2012). Similarly, the coyotes' intermediate body size also makes it a competitor with red fox, where red fox populations can be negatively impacted by coyote presence (Levi & Wilmers, 2012). Wolf-coyote competition that results in declines in coyote occupancy and abundance can lead to increases in red fox populations because red fox are released from the competitive pressures imposed by coyotes (Levi & Wilmers, 2012). Although I did not find fisher to have a positive effect on ermine in space or time, there was some support for marten presence reducing ermine detectability. This shows that ermine may avoid a site at times when marten are present while not completely discontinuing their use of the area. If the fisher population continues to increase in abundance and distribution, evidence of subsequent cascading benefits may become more apparent.

Analysis of the overlap patterns between species provided further evidence of mustelid temporal avoidance patterns in SEAK. There was not significant evidence showing that fisher avoided either marten or ermine, which was consistent with my hypothesis and lends support to the idea that fisher are dominant over marten and ermine. Results also corroborated my hypothesis that marten temporally avoid fisher by not visiting sites when fisher are active. However, contrary to my expectations, marten occupancy was not affected by fisher presence. Marten may be displaying more pronounced temporal, rather than spatial, avoidance of fisher due to their need to acquire similar prey

resources. This need for common prey resources may result in marten choosing to alter their activity rather than completely surrendering certain resource areas. The contrasting activity patterns that occur between fisher and ermine, and the shifts in ermine activity relative to both fisher and marten activity, were also consistent with my hypothesis that ermine exhibit temporal avoidance patterns of the larger mustelids in SEAK.

In this study, inference was limited by sample size, and a larger sample size may have enabled a clearer assessment of species' interaction patterns in SEAK. However, both marten and ermine had large sample sizes (i.e., total detections) relative to fisher and are well-established species in SEAK. The interactions between marten and ermine in this study therefore provide a strong example of one outcome of mustelid species interactions in the region.

Overall, this research demonstrated that temporal partitioning was likely playing a larger role than spatial partitioning in mustelid coexistence in SEAK. However, in the coming years, fisher will likely continue to expand their population in the area. Suitable habitat within mainland SEAK is confined to a relatively small area due to the glaciation of the region, and an increase in fisher population density could change species' competitive interactions should the resource requirements of the sympatric mustelids exceed SEAK's mainland resource capacity. This could potentially increase the likelihood of competitive interactions and change the current temporal partitioning patterns that were detected by this research. A larger sample size compiled across multiple seasons and years would provide more insight into these initial discoveries and present the opportunity to detect trends in species' interactions across time. Although currently fisher have not been documented on any islands within SEAK, many of the islands are comprised of forested habitat that is potentially suitable for fisher. These islands are already populated with marten and ermine, and should fisher expand its range to islands via swimming or traversing over frozen water channels, it could alter the ecological dynamics of islands in SEAK, as well. Only time will tell how this novel meso-carnivore will continue its expansion in the region.

This research was a first step in quantifying outcomes regarding the competition and niche partitioning of mustelids in the face of a climate-driven range expansion by a potentially dominant competitor. As climate change continues to shift species' ranges, more novel species interactions will likely occur across the globe. Understanding the nature of these interactions and the capacity for species' coexistence on a landscape will be crucial to successfully predict ecological community responses to changing species' compositions and to employ adequate management strategies to alleviate potential deleterious impacts associated with worldwide environmental change.

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Conclusion

This research provided important baseline data that will aid in the development of a management plan and monitoring protocol for the fisher population in SEAK. Coupling fisher occupancy with co-occupancy and activity patterns of American marten and ermine allowed me to make inferences regarding potential impacts of fisher on these established species in the region. Fisher occupancy was relatively high, with an average occupancy of 0.30 and 0.40 at the site and grid scales, respectively, and fisher occupancy was associated with different environmental covariates at local versus broad scales. Although there was no evidence of spatial partitioning among the mustelid species in SEAK, this research indicated that temporal partitioning is taking place, with marten and ermine adjusting their activity patterns in the presence of fisher.

Should this study design be adopted as a monitoring protocol for SEAK fishers, this data could provide valuable insight into trends in fisher occupancy and species interactions over time. The results from the fisher occupancy analysis provided important insight into habitat requirements of fisher in SEAK, and when combined with monitoring will potentially enable the development of a habitat suitability map for fishers in SEAK, making it possible to identify areas most likely to exhibit fisher colonization in the future should the species continues to expand its range. Finally, modification of genetic sampling to collect more fisher DNA will allow for an investigation into fisher density in SEAK and into the level of genetic connectivity between SEAK fishers and the British Columbia population.

Altered environmental conditions associated with climate change will continue to transform species' distributions (Tylianakis et al., 2008). These changes have the potential to significantly alter ecosystem dynamics as native flora and fauna adjust to direct and indirect effects of novel species. Interactions that result in the displacement or decline in native species could decrease biodiversity in a system, and moving forward, it will be important for biologists to monitor for these deleterious changes (Tylianakis et al., 2008; Prop et al., 2015). This research contributed to a more thorough understanding of an expanding mesocarnivore species that can be used to highlight the effects of shifting species interactions associated with global environmental change.

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Appendix A - Determining Fisher (*Pekania pennanti*) Density in Southeast Alaska: Limitations and Suggestions for the Future

Introduction

Designing effective species management plans often relies on accurately estimating the abundance of the target species on a landscape, which can be particularly challenging when surveying for elusive or rare carnivores (Long et al., 2008; Karanth et al., 2011). For these surveys, noninvasive methods are often implemented because they do not require direct observations of animals (Long et al., 2008).

The abundance of the Southeast Alaska (SEAK) fisher population is unknown, and initial estimates of species abundance are beneficial when designing practical species management plans. An original objective of this project was to determine fisher density in Southeast Alaska through the use of genetic sampling and spatially explicit capture-recapture (SECR) methods. Unfortunately, there were not enough genetic samples collected to conduct this density analysis, and certain aspects of our protocol proved challenging in the difficult SEAK winter environment, but what follows are the methods, genetic results, and a discussion of the limitations and potential protocol modifications for future fisher abundance research in SEAK.

Materials and Methods

Camera/Hair Snag Grid Design

When designing this study, it was important to determine a sampling method and station layout that would allow density analyses to be conducted for fisher and marten. Coupled hair snag and camera stations have been used by previous fisher studies to obtain estimates of abundance (Furnas et al., 2017; Long et al., 2008; Schwartz et al., 2006). Adult fisher home range size can vary significantly across geographic locations and between sexes, with an average home range size of 40 km² for males and 15 km² for females in the Western United States (Powell & Zielinski, 1994).

Marten home range size is much smaller than fisher, and varies considerably based on location and analytical methods. Studies conducted in southcentral Alaska and the Yukon Territory estimated average female marten home ranges of 3.71 km² and 4.7 km², respectively (Buskirk, 1983; Archibald & Jessup, 1984). Many previous studies have based grid cell size and station spacing on the smallest home range size of the study species, or delineated study grids based on cells established for other purposes, such as the US Forest Service Public Land Survey System (Zielinski & Kucera, 1995; Linden et al., 2017). This study intended to use spatial capture-recapture methods to estimate density, which are sensitive to grid cell extent and station spacing (Sun et al., 2014). Because of this, we developed a customized station layout for our landscape rather than relying on previous methods. Capture-recapture (CR) models are well-established methods for obtaining species abundance estimates (Sun et al., 2014) and spatially explicit capture-recapture modelling is an emerging field within CR that takes into account the spatial distribution of animals on the landscape and the resulting uneven detection probabilities at sample stations (Royle et al., 2017).

In order to ensure adequate spatial sampling and detection rates across our study area, we conducted preliminary simulations using custom scripts within the ‘secr’ package in program R to design our station layout (Efford, 2017). A simulation study conducted by Sun et al. (2014) found that clustered spacing resulted in the most accurate determination of abundance when compared with regular and sequential spacing configurations (regular spacing consists of evenly spaced stations across a study area and sequential spacing is when stations are clustered and regularly moved to new locations, gradually sampling the entire study area). Because of this, we developed a hexagonal grid cell template with two stations clustered within each grid cell (Sun et al., 2014; Zielinski et al., 2013). We set an initial total trap limit of 60 traps, based on what would be realistic to maintain in the field. For SECR modelling, it is recommended that trap spacing does not exceed 2σ , where σ is the spatial scale parameter of an individual and is related to the probability of detecting an animal as the distance from the animal’s activity center to the trap increases (Sun et al., 2014). It has been suggested that a minimum of two traps be placed within an individual’s home range, and we designated σ at 1,000 m (a distance slightly smaller than the approximate radius of a small female marten home range) in our simulations to ensure a sampling design capable of detecting individuals with small home ranges (Sun et al., 2014). Our simulations took into account detection probability and added a buffer and mask around the landscape to incorporate animals whose home ranges may be only partially included in the study area, and to exclude areas of non-habitat from the spatial integration to calculate density. We ran each combination of cluster and trap spacing through our simulations 100 times and compared the simulated results of detection probability and density to the set initial values in order to evaluate. We

then applied the most appropriate trap and cluster spacing results from our simulations to our study polygon template to create our final study area design.



Figure A.1. A map of the study area illustrating the optimized cluster/station layout. The black dots depict two stations per cluster spaced 350 m apart with 3,500 m between each cluster.

Based on our simulations, two camera/hair snag stations were clustered within each grid cell 350 m apart with 3,500 m between each cluster (Figure A.1).

Genetic Sampling

The use of non-invasive genetic analyses in a capture mark–recapture framework is becoming more prevalent, and can be particularly beneficial when surveying for rare carnivores (McKelvey & Schwartz, 2004). To obtain genetic data, hair samples were collected by deploying hair snags across the study area as detailed above. Hair snag design was based on the protocol outlined by Schwartz et al. (2006) and consisted of gun brushes attached to the inside of triangular corrugated plastic tubing. The protocol was modified by closing off one end of the tubing, enabling the camera to focus on a

single opening to better ensure a clear camera detection should a visit occur. Each tube was 24 in. (~61 cm) long with beaver bait suspended at the inside back. Three 0.30 caliber gun brushes were attached 6 in. (2.54 cm) from the opening of the tube (Figure A.2).



Figure A.2. Photo of the hair snag method, consisting of 3 gun brushes attached to a 24 in. (~61 cm) long corrugated plastic tube with beaver bait suspended inside.

The study region has high rainfall levels, even in winter, and this high moisture was expected to accelerate the degradation rate of genetic samples in the field (Brinkman et al., 2010; Shanley et al., 2015). Stations were checked every 14 days to limit sample exposure to moisture (Brinkman et al., 2010; Stetz et al., 2015). Bait and gun brushes containing hair samples were replaced at each station check. When a hair sample was present, the gun brush was removed in the field and placed in a coin envelope within a plastic bag for transport. The plastic bag was labeled with the station name and date. Once out of the field, sterile latex gloves and tweezers were used to remove the hair from each gun brush. The hair sample was once again placed in a coin envelope with silica desiccant and marked with the station name and date collected and monitored for changes in moisture level. Samples from each individual gun brush were collected and stored separately. Genetic samples were stored at room temperature and shipped to the Laboratory for Ecological, Evolutionary, and Conservation genetics (LEECG) at the University of Idaho.

DNA was extracted from the hair samples and species identification was obtained by conducting a mitochondrial DNA (mtDNA) fragment analysis test following the protocols outlined in De Barba et al. (2014) and Davidson et al. (2014). These protocols have been demonstrated to distinguish the following species present in the study area: fisher, American marten, red fox (*Vulpes vulpes*), bobcat (*Lynx rufus*), and mountain lion (*Puma concolor*). Gray wolf (*Canis lupus*), wolverine (*Gulo gulo*), and dog (*Canis lupus familiaris*) samples produce the same fragments and can therefore be distinguished from other species but not from each other (Davidson et al., 2014). Similarly, coyotes (*Canis latrans*) and brown bears (*Ursus arctos*) can be distinguished from other species but

not from each other (Davidson et al., 2014). This protocol has not been tested on ermine (*Mustela erminea*), American mink (*Neovison vison*), red squirrel (*Sciurus vulgaris*), or North American beaver (*Castor canadensis*), all species that occur in the study area. Samples that failed to amplify under the initial PCR conditions were re-run at higher concentrations using twice the amount of DNA and with more PCR cycles (increasing from 35 cycles to 50). Samples exhibiting unexpected amplification peaks or questionable species calls were sequenced from the cytochrome *b* mtDNA region using primers adapted from Kocher et al. (1989), designed to target a specific region of 146 base pairs (bp) (5'-AAACTGCAGCCCCTCAGAATGATATTTGTCCTCA-3', 5'-TATTCTTTATCTGCCTATAC-ATRCACG-3') (Ferrell, 2000), and from the mtDNA control region using primers CTRL-L, TDKD, Gulo1F, H3R, SID L, and H16145 (Bidlack & Cook, 2001; Dalen et al., 2004; De Barba et al., 2014). Samples identified as fisher and marten by mtDNA fragment analysis or sequencing were sent to Dr. Michael Schwartz's lab at the U.S. Forest Service Rocky Mountain Research Station and National Genomics Center for Wildlife and Fish Conservation (NGCWFC) to obtain individual identification and sex-ratios. Due to concerns about distinguishing between marten and ermine samples, all marten samples were sequenced at NGCWFC by amplifying 360 bp using 16S rRNA universal primers (Hoezel & Green, 1992). The quality and quantity of template DNA were determined by 1.6% agarose gel electrophoresis. PCR products were purified using ExoSap-IT (Affymetrix-USB Corporation, OH, USA) according to manufacturer's instructions. Reactions were sequenced at Eurofins Genomics (Louisville, KY) using standard Sanger sequencing protocols. DNA sequence data were viewed and aligned with Sequencher (Gene Codes Corp., MI) and sequences were compared to reference sequences from known species. Fisher samples were analyzed at 16 microsatellite loci used in previous mustelid studies: MpP0059, Mp0144, Mp0175, Mp0197, Mp0200, Mp0247 (Jordan et al., 2007); Ma1 (Davis & Strobeck, 1998); Mer022, Mvis020, Mvis072 (Fleming et al., 1999); Ggu101, Ggu216 (Duffy et al., 1998); Lut604, Lut733 (Dallas & Piertney, 1998); Gg25 (Walker et al., 2001); Mf1.18 (Basto et al., 2010); and Mvi1321 (Vincent et al., 2003). Marten samples were analyzed at 13 microsatellite loci used in previous mustelid studies: Ma1, Ma2, Ma3, Ma8, Ma18, Ma19, Gg3, Gg7 (Davis & Strobeck, 1998); Ggu216, Ggu234 (Duffy et al., 1998); Mer041 (Flemming et al., 1999); MP0197 (Jordan et al. 2007); and Lut604 (Dallas & Piertney, 1998). DNA from hair samples was amplified using the multi-tube approach (Eggert et al., 2003; Schwartz et al., 2004). The resultant products were visualized on a LI-COR DNA analyzer (LI-COR Biotechnology). Data was error checked using program Dropout (McKelvey & Schwartz, 2005), GenAlEx (Peakall & Smouse 2006, 2012) and Microchecker (Van Oosterhout et al., 2004).

Results

There were 204 genetic samples extracted. Of those, 36 samples amplified under the original standard protocol, and an additional 81 samples amplified after being run with twice as much DNA and more cycles. In total, 117 samples amplified and 16 of those samples underwent sequencing to clarify species identification. Analysis at LEECG determined that 64 samples were from marten and 16 were from fisher. Twenty-seven samples were identified as beaver (the bait). When marten and fisher samples were analyzed at NGCWFC, 7 of the samples previously identified as marten were found to belong to ermine, one was identified as beaver, and 7 samples that had previously been identified as marten had DNA that was too poor for species identification at the Schwartz lab. One female fisher and two male marten were individually identifiable. NGCWFC noted that amplification success for marten was much lower than previous projects they have worked with whereas fisher amplification success was similar.

At LEECG, the DNA fragments from the 7 ermine samples that were previously identified as marten during initial screening were analyzed again using Genemapper 3.7 software (Applied Biosystems) to identify whether a distinct banding pattern was present that would distinguish ermine from marten samples. The examination revealed that no distinct banding pattern existed, indicating that this particular test cannot distinguish ermine from marten samples. Based on these results, combined with results from NGCWFC, 50 of the samples collected were definitively marten and 7 were definitively ermine. Because this test cannot differentiate between marten and ermine samples, the 7 samples that had DNA too poor for species identification at NGCWFC remained inconclusive and may belong to either marten or ermine. Overall, 6 species were identified from the hair samples collected, with 7 samples remaining unknown after sequencing. Eighteen DNA samples identified two species within one sample for a total of 135 separate species calls. The total species identification amplification rate, including unknown species, was 66.2%, and the total successful species identification rate was 62.7%. Six percent of marten and 43.7% of fisher samples were successfully identified to individual. A full list of species identified through genetic sampling is summarized below (Table A.1).

Table A.1. Final species identification results from 204 DNA samples from hair collected from January–April 2018 in Juneau, Alaska. 18 samples amplified two species from one sample for a total of 135 separate species calls.

Species	Number of samples	Frequency in samples
Fisher (<i>Pekania pennanti</i>)	16	7.8
Marten (<i>Martes americana</i>)	50	24.5
Ermine (<i>Mustela erminea</i>)	7	3.4
Marten or Ermine	7	3.4
Beaver (<i>Erethizon dorsatum</i>)	28	13.7
Wolverine (<i>Gulo gulo</i>)	5	2.5
Gray Wolf/Dog (<i>Canis lupus</i> spp.)	15	7.4
Unknown	7	3.4
Samples that Failed to Amplify	69	33.8

Discussion

Even though it was unfeasible to conduct a density analysis with the limited amount of genetic data collected, this portion of research was helpful in illuminating the challenges associated with obtaining non-invasive genetic information from fisher in SEAK.

The study design and field collection of genetic samples for this study faced several challenges. Even though the station spacing for this study was thoroughly thought-out, the home range size of fisher is unknown for this region. If fisher home range in SEAK is significantly different than in other areas, a different station design that coincided more accurately with fisher home range may have yielded more genetic data.

This study also had to navigate the balance between deploying an adequate number of genetic stations with the logistical feasibility of checking those stations in the field within an acceptable timeframe that would minimize sample degradation due to moisture. Within the logistical capabilities of the study, the ultimate number of stations deployed enabled hair snags to be checked every 14 days on a constant rotation. Although the addition of more stations would have potentially yielded more samples, the time between station checks would have been extended, increasing the likelihood of DNA degradation. On the other hand, enacting shorter time frames between station checks may have yielded more viable hair samples that were less affected by moisture, but would have limited the number of stations that were able to be checked. One suggestion to implement in the future is to

divide the study area in half and increase the number of sampling stations within the smaller area; this could potentially decrease the time between station checks. Each half of the study area could be sampled on alternate years so that the complete 200 km² area is ultimately sampled across two seasons.

Additionally, ermine proved to be quicker at commandeering bait than was anticipated, and often immediately entered a hair snag to remove the bait after it had been freshly added at a station check, based on camera trap evidence. This lack of bait decreased the chances of fisher and other animals entering the hair snag after the bait was gone. It would be advantageous to devise a way for bait to be more difficult to remove from the hair snags in the future to ensure that opportunities to attract fisher into the tubes to snag hair are not missed. This could be accomplished by placing the bait in suspended chicken wire bags that would provide a barrier between the animal and bait, thereby preventing the bait's immediate removal. Another option would be to add lure into the center of the cubby box, rather than only applying it in a bait box outside of the snag. This would entice animals into the hair snag, and be less available to be stolen than solid bait. It would also remain relatively pungent between station checks to continually draw animals into the hair snag.

Previous studies have had success with the cubby/gun brush method of collecting hair samples that was employed in our research. However, even though the gun brush size used in this study was consistent with the Schwartz et al. (2006) protocol, the brushes in our particular study were relatively ineffective in capturing follicles from the hair samples collected. Another fisher study has had success using larger caliber (0.50) gun brushes, although this larger sized gun brush was used in conjunction with a different corrugated plastic cubby design; it is unclear if using a larger gun brush with our specific design would have increased the effectiveness of collecting samples (B. Barry, personal communication).

The Schwartz et al. (2006) protocol for this cubby design was also modified in our research to close off one end of the cubby, whereas the original protocol calls for the cubby to be open at both ends, with three gun brushes placed at each opening. This modification was done to allow for clear animal identification from remote cameras by forcing the animal to enter the snag from only one direction. Due to the modification of our cubby boxes, there were only three brushes available to collect hairs, rather than the six in the initial protocol. The presence of fewer gun brushes may have decreased the number of hair samples collected. Furthermore, it is possible that closing off the cubby may have reduced air flow inside when compared to a cubby that remained open at both ends.

Closing off one end may have caused more condensation to build up around the samples and increased sample degradation rate.

Studies in Alberta, Canada have had success collecting fisher samples using barbed wire wrapped around trees (Stewart et al., 2017). This method was ultimately not used in this research due to the increased risk of sample exposure to moisture, mixed samples, and the logistical difficulties associated with transporting large amounts of barbed wire in the field. However, if the study area was divided into smaller portions as suggested above, this could allow for shorter sampling intervals and travel distances to check stations, which could make this method more feasible to employ in SEAK.

In the future, I would suggest leaving both ends of the cubby open so that more gun brushes can be included to increase the chances of capturing hair samples. This would also increase air flow to the samples and potentially decrease their degradation rate. Additionally, using lure coupled with well-secured bait in the center of the hair snag would increase the potency of attractant within the snag and potentially increase the likelihood of a fisher entering and leaving hair. Testing larger gun brushes may also be a logical next step towards developing an effective fisher genetic survey protocol in SEAK. Additionally, and although it was not an intended focal point of this research, the genetic analysis indicated that further measures need to be incorporated into the De Barba et al. (2014) protocol to make marten and ermine samples distinguishable.

Overall, this study highlighted the difficulties in collecting adequate genetic data from fisher in SEAK, and was unexpectedly helpful in identifying an area for improvement in genetic species identification between marten and ermine. By adjusting the sampling methods, an additional season of genetic sampling may yield sufficient hair samples to determine fisher abundance, and contribute further knowledge to the population status of fisher in SEAK.

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Appendix B – Photos of Camera/Hair Snag Stations



Figure B.1. Photo of a baited camera/hair snag station deployed north of Juneau, Alaska from January–April 2018.



Figure B.2. A fisher detected on remote camera visiting a baited hair snag station deployed north of Juneau, Alaska from January–April 2018.



Figure B.3. A fisher detected on remote camera visiting a baited hair snag station deployed north of Juneau, Alaska from January–April 2018.



Figure B.4. A fisher detected on remote camera entering a baited hair snag cubby deployed north of Juneau, Alaska from January–April 2018.