Distribution and abundance of the fisher (*Pekania pennanti*) on the Makah Reservation and surrounding lands in northwestern Washington State

Report to Makah Tribe

In fulfillment of Collaborative Agreement 18WNYB010477

Rebecca M. McCaffery[1], Shannon Murphie[2], Patricia J. Happe[3], and Kurt J. Jenkins[1]

[1] U.S. Geological Survey Forest and Rangeland Ecosystem Science Center, Olympic Field Station, 600 E. Park Avenue., Port Angeles, WA 98362

[2] Makah Tribe, PO Box 116, Neah Bay, WA 98357

[3] National Park Service, Olympic National Park, 600 E. Park Avenue, Port Angeles, WA 98362

February 2020

Cover photo credit: Shannon Murphie, Makah Tribe.

Any use of trade, product, or firm names is for descriptive purposes only, and does not imply endorsement by the U.S. Government.

Contents

Conversion Factors

Multiply	By	To obtain
	Length	
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)

Multiply	By	To obtain
	Length	
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
kilometer (km)	0.5400	mile, nautical (nmi)
	Area	
square kilometer (km^2)	247.1	acre
square kilometer (km^2)	0.3861	square mile (mi^2)

Abstract

Fishers were extirpated from Washington State in the middle-to-late 1900s and were listed as a State endangered species in 1998. To restore the species in Washington and enhance its status throughout the West Coast of the U. S., the Washington Department of Fish and Wildlife and the National Park Service translocated 90 fishers from British Columbia to Washington's Olympic Peninsula from 2008 to 2010. Peninsula-wide camera-based monitoring conducted from 2013 to 2016 revealed that the reintroduced population had become established throughout much of the Peninsula. The goal of the project reported here was to enhance our understanding of the current status and distribution of fishers on the Makah Reservation and surrounding areas nearly a decade after their reintroduction, and to provide information to Makah biologists that will help inform future forest management planning. Our first objective was to determine habitat and landownership associations of fishers in the study area using occupancy models. Our second objective was to estimate the population size and density of fishers on and near Makah lands using capture-recapture methods based on genetic identifications of individual fishers. The 450 km^2 study area included all Makah lands (trust land and fee status land), as well as lands in private ownership and in Olympic National Park. We sampled fisher occurrence at 58 sites, each of which included two stations, with 1 remote-camera and 1 hair-snaring device at each station. Each site was sampled for 6 consecutive weeks each year, with weekly visits to collect photographic images and hairs for DNA analyses. We detected fishers at 13 sites in 2017 and at 9 sites in 2018. We identified 7 individual fishers from DNA, of which 2 were male and 5 were female. The best supported model of detection probability indicated that the previous detection of a fisher increased detection probability in subsequent sampling sessions. Model selection results for occupancy based on the top detection model indicated that occupancy increased as distance to the late-seral protected forest (i.e., wilderness) boundary decreased. We found no evidence that the proportion of medium and large trees, dense canopy cover, tribal trust and fee lands ("tribal non-wilderness"), or late-seral protected forest (hereafter, "wilderness") in the buffer had a strong influence on occupancy. Estimated occupancy in 2017 was 0.308 (0.187 – 0.461 95% CI), and in 2018 was 0.240 (0.124 – 0.412). Results from closed population models based on genetic capture-recapture data indicated that approximately 6-8 fishers inhabited the study area in 2017, but data collected in 2018 were inadequate to estimate abundance. Collectively, these results indicate that the mosaic of wilderness and actively managed forest lands has sustained a small population segment of fishers on Makah Tribal lands. Similar to the peninsula-wide study, we found that proximity to wilderness was the best predictor of fisher occupancy. This indicates that fishers, even at small spatial scales (i.e., much smaller than their home range), respond to features that occur in both wilderness and lands managed for timber production. We hypothesize that wilderness provides the requisite denning and resting sites that are key to fisher survival, whereas managed forest lands contain more abundant prey, especially snowshoe hares and mountain beavers. Our findings provide an important baseline for monitoring the status of fishers on the Makah Reservation in future years. Additional telemetry-based research would help to elucidate how fishers are using the mosaic of habitat conditions on Makah lands to meet their denning, resting, and foraging requirements.

Acknowledgments

This study was funded by a grant from the BIA Endangered Species Program, and The Makah Tribe, with significant in-kind contributions from the USGS and the NPS including staff time and equipment. We would like to thank R. McCoy and M. Pope for their logistical and administrative support throughout this study. Special thanks go out to our dedicated field staff, especially K. McLean, J. Johnson, N. Beatty, Z. Bruns, Z. Greene, I. Olson, E. James, R. Moss, and B. Ward for their efforts collecting data and assisting with data management. In addition, K. McLean helped prepare the map presented in this report. We thank Makah Forestry Enterprises, Green Crow, and Merrill & Ring for access to the study area and support of our work. We thank K. Pilgrim and the National Genomics Center for Wildlife and Fish Conservation at the Rocky Mountain Research Station for the analysis of fisher DNA samples. P. Haggerty was instrumental in helping to prepare the habitat covariates for the analysis. Finally, we thank Keith Aubry and David Wiens for their review of a previous draft of this report.

Introduction

The fisher (*Pekania pennanti*) once occupied coniferous forests at low to mid-elevations throughout most of the western United States (Powell et al. 2003). During the 20th century, fishers were extirpated from Washington State; they were listed as a State "Endangered" species in 1998. Remnant fisher populations in other coastal states are low or declining, prompting the U.S. Fish and Wildlife Service to list the West Coast Distinct Population Segment (DPS), which ranges from British Columbia, Canada, south through the Cascade, Klamath, and Sierra Nevada Ranges in Washington, Oregon, and California, USA, as federally threatened (U.S. Fish and Wildlife Service 2019). A final decision will not be made until April 2020, but the proposed listing does not include fishers that were reintroduced to Washington State.

To enhance fisher populations in Washington and throughout the West Coast DPS, the Washington Department of Fish and Wildlife and the National Park Service translocated 90 fishers from British Columbia to Washington's Olympic Peninsula from 2008 to 2010. Initial monitoring efforts involved radio-tracking reintroduced individuals to provide data on survival, movements, home-range establishment, and reproduction (Lewis 2014, Lewis et al. 2016). From 2013 to 2016, a non-invasive, peninsula-wide monitoring program was implemented to determine the status of the fisher population 5-8 years after reintroduction to provide an initial evaluation of the success of the reintroduction (Happe et al. 2019). That study also provided an opportunity to evaluate previous assumptions made during fisher habitat suitability modeling, and improve our understanding of fisher habitat associations in coniferous forest ecosystems of the coastal Pacific Northwest. Occupancy models based on data obtained from camera-trapping were used to identify environmental correlates of fisher occupancy and to describe patterns of fisher distribution 5-8 years after the reintroductions. The researchers determined that the distribution of fishers had shifted from the drier northeastern parts of the Peninsula to the wetter southern and western portions of the Peninsula during the 4 years of the study (Happe et al. 2019). They also found that fisher occupancy was best explained by proximity to the boundary between wilderness and managed forest lands, suggesting that fishers were attracted to the diversity of landscapes and resources found near the boundary of protected and managed forests (Happe et al. 2019).

Since the reintroduction of fishers to Olympic National Park from 2008 to 2010, the Makah Tribe has been an active partner in monitoring their populations. During the Peninsula-wide fisher monitoring effort, Makah biologists documented the presence of 1 male and 1 female on the Makah Reservation in 2013, and observed fishers using Reservation lands during the study (2013-2016). However, neither the population status of fishers on the Reservation, nor their habitat and land-ownership associations were well understood.

The Makah Reservation is managed primarily for commercial timber production—the average age of harvested stands is 60-70 years. Intensive timber management has converted what were historically extensive old-growth forests to second growth. These stands never attain old-growth characteristics, unless they are set aside for preservation. Some old-growth habitat occurs in the Waatch Valley along the coast of Cape Flattery in the southwest corner of the Makah Reservation, and along the coastal strip of Olympic National Park adjacent to Makah lands. Historically, fishers inhabited mature and naturally regenerated forest landscapes throughout many forested regions of western North America (Lofroth et al. 2010). Recently, radio-tracking and remote-camera surveys have demonstrated that fishers also use a variety of managed second-growth forests and seral stages on the Olympic Peninsula (Lewis et al. 2016; Happe et al. 2019). The extent to which they use such forests is unknown, but available information suggests that reintroduced fishers may be using a relatively broad range of habitat conditions. Long harvest rotations on the Makah Reservation, conservation of remnant mature forest, and proximity to Olympic National Park may have facilitated the initial establishment of fishers on Makah Tribal lands following their reintroduction to the Peninsula.

The goals of our study were to enhance our understanding of the current status and distribution of fishers on the Makah Reservation and surrounding areas nearly a decade after their reintroduction on the Peninsula, and to provide reliable information to Makah biologists for designing forest management strategies that will benefit fishers. This study stemmed from the larger previous survey, which documented that reintroduced fishers had expanded distribution from release sites within Olympic National Park to the entire peninsula (Happe et al. 2019), including the Makah lands. Our study differed from the peninsula-wide study by using a denser sampling grid, which was designed to provide local information on the distribution and abundance of fishers on lands managed by the Makah Tribe. We used remote-camera stations with hair-snares to assess fisher distribution and population size in the study area during the summer months. Our first research objective was to investigate habitat and landownership associations for fishers in the study area using occupancy models. Occupancy models do not rely on individual identifications; thus, camera-traps could be used to relate fisher occupancy to habitat covariates of interest. Our second research objective was to use individual identifications from DNA obtained on hair-snares to estimate the population size and density of fishers in the study area using capture-recapture methods.

Methods

Study area and field methods

The 450 km^2 study area included all Makah lands (trust land and fee status land), as well as surrounding private forest land, and National Park Service land within Olympic National Park on the west coast and around the north end of Lake Ozette (Figure 1). Sampling locations (hereafter,

"sites") were determined using an evenly spaced grid with a 2-km interval generated in the Geospatial Modeling Environment (Beyer 2012). We used a clustered design that included 2 camera stations at each site (Sun et al. 2014a,b) located within 500 m of each other. The starting point for the grid was randomly selected based on the grid used in the Pensinsula-wide sampling project (Happe et al. 2019). After some modifications due to logistical constraints on sampling, we sampled 58 sites (i.e., 116 camera stations) at a density of 12 sites per 24 km^2. This area approximated the median size of a fisher home-range core area (males and females combined; Lewis et al. 2016). These 58 sites were divided into 4 smaller grids containing 14 or 15 sites, each of which was surveyed during a 6-week period. Grids 1, 2, and 3 were surveyed in 2017, while grids 2, 3, and 4 were surveyed in 2018 (Figure 1). Survey dates varied by grid and year. In 2017, grid 1 was surveyed between 22 May and 7 July, grid 2 was surveyed between 11 July and 25 August, and grid 3 was surveyed between 28 August and 13 October. In 2018, grid 2 was surveyed between 21 May and 6 July, grid 3 was surveyed between 9 July and 24 August, and grid 4 was surveyed between 4 September and 19 October.

At each site, we deployed 2 sampling stations within 500 meters of each other, based on our knowledge of fisher habitat relations and constraints imposed by local conditions (Royle et al. 2011). Each sampling station consisted of a baited hair-snare cubby, a camera mounted on a tree, and bait attached to the target tree, following Jenkins and Happe (2013). Hair-snare cubbies were constructed of corrugated plastic in a triangular shape with 3 copper wire gun brushes placed at each entrance to snag hair; we suspended a chicken leg from the center of the apex (see Happe et al. 2019). Remotely triggered cameras (Bushnell Trophy Cam HD, Overland Park, KS, USA) were mounted on a tree approximately 3-5 meters from a second tree (the bait tree) between azimuths of 340^o-20^o. The north-facing orientation of cameras reduced false triggering of the cameras caused by changes in direct solar radiation, while also reducing glare in the photographic images. Bait consisted of chicken meat enclosed in chicken wire and attached to the tree facing the camera and, occasionally, a can of fish-scented cat food nailed to the tree, also facing the camera. We applied a long-distance olfactory lure (Caven's Gusto, Minnesota Trapline Products, Pennock MN, USA) to the bait tree, and included a placard indicating the sample site, station, and date. Both the bait and hair-snare cubby were within the camera's field of view. Sampling stations were checked every 7-10 days for 50 days (Sweitzer et al. 2016), resulting in 6 sampling periods per station. During each visit, field crews collected photographic images from the cameras and gun brushes that contained animal hairs. When hair was collected on a gun brushes, we removed them wearing latex gloves, and immediately placed them in a tube with desiccant and a uniquely identifiable bar-code label. Hair samples were submitted to the United States Forest Service-Rocky Mountain Research Station's Conservation Genetics Lab in Missoula, Montana at the completion of each 50-day sampling session. Hair samples were tested to identify species. When DNA from hair samples was of sufficient quality and quantity, genetic analyses were used to identify individual fishers and determine their gender, based on protocols described by Happe et al. (2019).

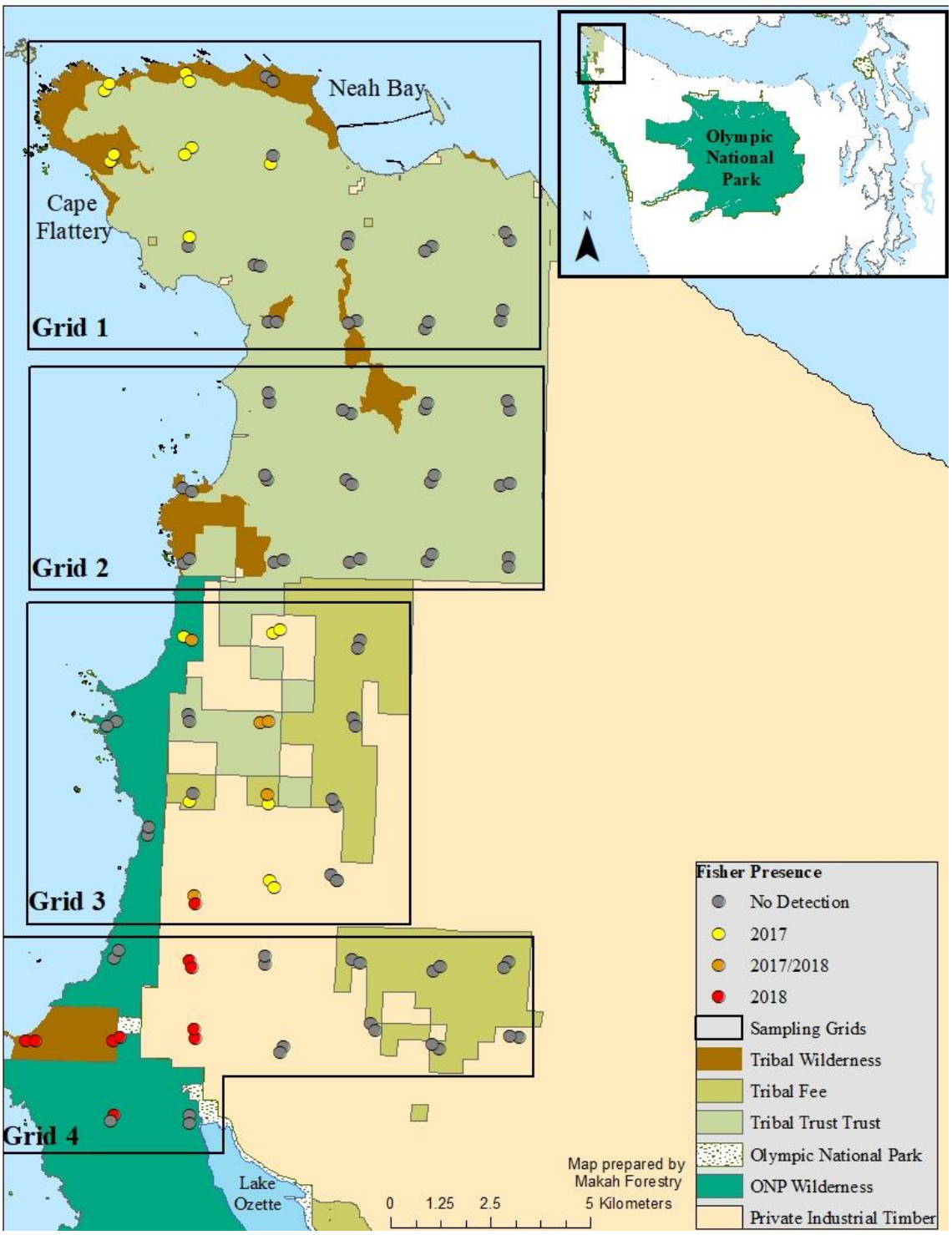

Figure 1. Study area for the study of fisher on the Makah Reservation and surrounding lands in 2017 and 2018. Black rectangles show the four study grids encapsulating the 58 paired camera stations (i.e., sampling sites). Dots indicate the paired camera stations at each site, with yellow dots showing 2017 detections of fisher, red dots showing 2018 detections of fisher, and orange dots showing stations where fishers were detected in both years. Grey dots indicate stations where fishers were not detected.

Modeling fisher detection and occupancy

We hypothesized that the percent cover of certain land-management designations and forest structural classes, as well as the distance to a wilderness boundary, would influence fisher occupancy in our study area. To describe habitat conditions around each sampling site, we established 400-m-radius buffers with the midpoint between the 2 camera stations as its center. We evaluated 3 land-management designations: wilderness (combining all of Olympic National Park wilderness with Makah reserves and protected lands), Makah tribe trust and fee lands, and other designations (private or state timberlands). Because the percent cover of these designations summed to 1 and, consequently, were strongly correlated, we only included wilderness and Makah trust and fee lands in our models. In addition, we calculated the mean distance of the midpoint between sampling stations at each site from the nearest boundary separating wilderness lands from other designations. We included this measurement because it was shown by Happe et al. (2019) to influence fisher distribution on the Peninsula (Happe et al. 2019), and because previous telemetry studies revealed that many translocated fishers established home ranges in proximity to both mature and managed forests (Lewis et al. 2016).

Within the buffers, we also estimated the percent cover of forest structural classes used to delineate potential fisher reintroduction areas in Washington during the state's feasibility study (Lewis and Hayes 2004): medium and large trees (>40% canopy cover and trees >25 cm QMD) and dense forest overstories (>70% canopy closure). We characterized forest structural characteristics based on 2017 data from the Landscape Ecology, Modeling, Mapping and Analysis (LEMMA) project of the USFS and Oregon State University (*https://lemma.forestry.oregonstate.edu/data*, accessed 8 May 2019).

We used single-season occupancy models to estimate the probability of occurrence of fishers within stations while accounting for imperfect detection and environmental covariate effects (MacKenzie et al. 2002). We combined data from 2017 and 2018 due to low sample sizes, allowing us to evaluate spatial patterns of fisher occupancy throughout our study area. Two grids were sampled in both 2017 and 2018, but fishers were never detected on grid 2, and were detected at that same stations on grid 3 in both years. This allowed us to combine all 4 grids in a single analysis without omitting or altering data. These models assume that the occupancy status of each site is independent and conditional on the covariates included in the model. We used fisher detections from cameras in $i = 1, \ldots, 58$ sites during $j = 1, \ldots, 6$ visits. If a fisher was detected at either of the 2 cameras at each site during a check, we coded it as "present" for that site-visit combination.

We modeled detections of fishers at site i in visit j as $y_{ij} \sim Bernoulli(z_i \times p_{ij})$. Here, z_i was the latent (true) occupancy state, which indicated if site i was occupied ($z_i = 1$) or not ($z_i = 0$), and p_{ij} was the site and visit-specific detection probability, $Pr(y_{ij} = 1 | z_i = 1)$. We then modeled the latent occupancy state as $z_i \sim Bernoulli(\Psi_i)$, where Ψ_i was the site-specific probability of

occurrence, $Pr(z_i = 1)$. We used logit-linear models to examine the effects of covariates on detection and occupancy probabilities, where $logit(p_{ij}) = \boldsymbol{x}'_{ij}\boldsymbol{\alpha}$ and $logit(\Psi_{ij}) = \boldsymbol{x}'_i\boldsymbol{\beta}$. Here, $\boldsymbol{x}'_{ij}$ represented a vector of covariates that varied by site and/or sampling period, and $\boldsymbol{\alpha}$ and $\boldsymbol{\beta}$ were regression coefficients. We investigated the influence of the following covariates on fisher detection probabilities: ordinal date of the station visit (including both linear and quadratic terms, and an interaction with year) and whether or not a fisher was detected previously at the station. We looked at additive effects of these variables, singly and in combination, for a total of 6 detection models.

We modeled the probability of occupancy as functions of the land management designations and habitat variables described above. We examined pairwise correlations among all variables to avoid including highly correlated variables within a model ($r > 0.5$). We examined each variable singly, and developed a small set of additive models that included distance to wilderness edge in combination with a land-management covariate, a habitat covariate, or both. The dataset was too small to develop more complex models. Althogether, we evaluated 10 models of fisher occupancy in our study area (Appendix A). Separately, we examined null models of occupancy in 2017 and 2018 to generate an occupancy estimate for each year.

We first evaluated models of detection probability using a null occupancy model. We used model selection to choose the best model of detection probability, and used this detection model for subsequent modeling of occupancy. We used a maximum likelihood approach to model fisher detection and occupancy, using the extension package "unmarked" within the statistical software R (*https://cran.r-project.org/web/packages/unmarked/index.html*, accessed 10 May 2019).

Modeling fisher abundance

Due to the small number of individually identified fishers and the large number of hair samples that could not be identified to individual, we were unable to estimate fisher density using spatial capture-recapture models as originally proposed. To derive an initial estimate of fisher population size in the study area, we used non-spatial closed capture-recapture models. Closed capture-recapture models rely on the repeated sampling of individuals over time; thus, only hair samples that could be assigned genetically to individual fishers were included. We were able to estimate fisher population size in 2017, but captures were inadequate to estimate population size in 2018.

We examined 4 closed population models to estimate fisher population size in our study area. Each model differs in how detection probability is modeled, which can affect the resulting abundance estimate:

1. M0 – Null model. For this model, detection is constant across all time periods

2. Mb – Behavior model. For this model, detection upon first capture (p) is modeled separately from detection in subsequent captures (c), testing for trap-happiness (greater likelihood to capture individuals subsequent to first capture) or trap-avoidance (lower likelihood to capture individuals subsequent to first capture)
3. Mt – Time model. For this model detection varies over time
4. Mtb – Time + behavior model. This model combines models (2) and (3).

We used Program MARK to implement all models, and compared models using Akaike's Information Criterion adjusted for small sample sizes (AIC_c, Burnham and Anderson 2004). We present and discuss results from the four models.

Results

Fisher sampling and detections

We surveyed 58 sampling sites including 116 camera stations for the presence of fishers. We surveyed 43 sites in 2017 (grids 1, 2, and 3) and 43 sites in 2018 (grids 2, 3, and 4). Thus, sites on grids 1 and 4 were only surveyed in 1 year (Table 1).

Detections differed by both grid and survey year (Table 1, Figure 1). Fishers were detected at 13 different sites on grids 1 and 3 in 2017, and at 9 different sites on grids 3 and 4 in 2018. No fishers were detected in any of the sites on grid 2 in either year of the study.

Table 1. Summary of sampling effort from camera stations and resulting fisher detections.

	Sampling effort		Fisher detections			
Year	Grid #	# sites	Total detections (including both cameras per site)	Total # of site-visits with detections	# sites with detections	Occupancy estimate (95% CI)
2017	1	15	29	20	6	0.31 (0.19 – 0.46)
	2	14	0	0	0	
	3	14	29	19	7	
2018	2	14	0	0	0	0.24 (0.12 – 0.41)
	3	14	8	6	4	
	4	15	17	12	5	

We identified 7 individual fishers based on DNA from hair samples, of which 2 were male and 5 were female (Table 2). Six individuals were identified in 2017 but only 3 were identified in 2018. Two of the 3 individuals identified in 2018 were also detected in 2017. Frequency of capture and number of unique sites with detections of a given individual were higher in 2017 than 2018. 30 hair samples were identified to individual fisher in 2017, whereas 7 hair samples were identified to individual in 2018. There were 4 hair samples in 2017 and 2 in 2018 that were identified as fisher, but where the DNA was not of sufficient quality to determine individual. In each year, there were 6 instances where a fisher was caught on camera but no hair was collected. In each year, there was one instance where fisher hair was collected but no fishers were detected on camera due to camera malfunctions. In 2017, the DNA in the hair was not of sufficient quality to determine individual, and in 2018, the DNA in the hair was identified as individual 1351F. There were 15 hair samples in 2017 and 3 in 2018 where DNA was too degraded to determine species. Finally, in 2017 there were 3 hair samples identified as bear, and in 2018, 5 samples were identified as bear, 1 as mink, and 1 as chipmunk.

Table 2. Summary of genetic results from hair snare cubbies and resulting fisher capture histories.

Year	Fisher ID		Fisher detections			
	Fisher number	Sex	Grid where detected	Total detections (incl. both stations)	Total number of site-visits	Number of sites with detections
2017	354	M	1	12	9	4
	358	F	1	4	3	1
	1290	M	3	5	4	2
	1298	F	3	2	1	1
	1292	F	3	2	2	2
	1281	F	3	5	4	2
2018	1281	F	3	1	1	1
	1290	M	4	2	2	1
	1351	F	4	4	3	2

In addition to the fisher detections summarized above, 24 other species or species groups were detected on remote cameras (Table 3). Black bears and spotted skunks had the highest number of detections. Although spotted skunk detections were similar in both years, black bear detections were more common in 2018 compared to 2017. Snowshoe hare and mountain beaver, which are potentially important prey items for fishers in our study area (National Park Service, unpublished data) were rarely detected during both survey years (Table 3).

Table 3. Number of times that a species or species group was detected on cameras in 2017 and 2018 on the Makah Reservation, by total number station-visits.

Taxonomic group	Species	2017	2018
		Total number of station-visits	Total number of station-visits
Carnivores	Black Bear	125	214
	Spotted Skunk	154	158
	Bobcat	25	22
	Coyote	18	10
	Cougar	2	3
	Weasels (*Mustela* spp.)	36	22
	Fisher	39	18
	Raccoon	29	9
	Domestic Dog	2	1
	Mink	4	6
Ungulates	Black-tailed Deer	32	32
	Elk	0	1
Small and medium mammals	Douglas Squirrel	44	69
	Mice and Voles	41	39
	Northern Flying Squirrel	8	10
	Snowshoe Hare	6	13
	Chipmunks	8	13
	Mountain Beaver	3	1
Birds	Passerines	37	46
	Jays and Crows	30	22
	Raven	63	50
	Blue and Ruffed Grouse	5	4
	Turkey Vulture	3	0
	Owl species	1	1
	Bald Eagle	1	0

Fisher detection probability and occupancy

The best supported model for the probability of detecting a fisher indicated that a previous detection of a fisher strongly influenced detection on the current visit. Specifically, fisher detection probability was higher when a fisher had previously been detected at the site (0.647 [0.067 SE]) than when a fisher had never been detected at the site (0.109 [0.125 SE]; Figure 2). There was no effect of sampling date on fisher detection probability (Appendix A). Therefore, we only included “previous fisher detection” in our models of occupancy.

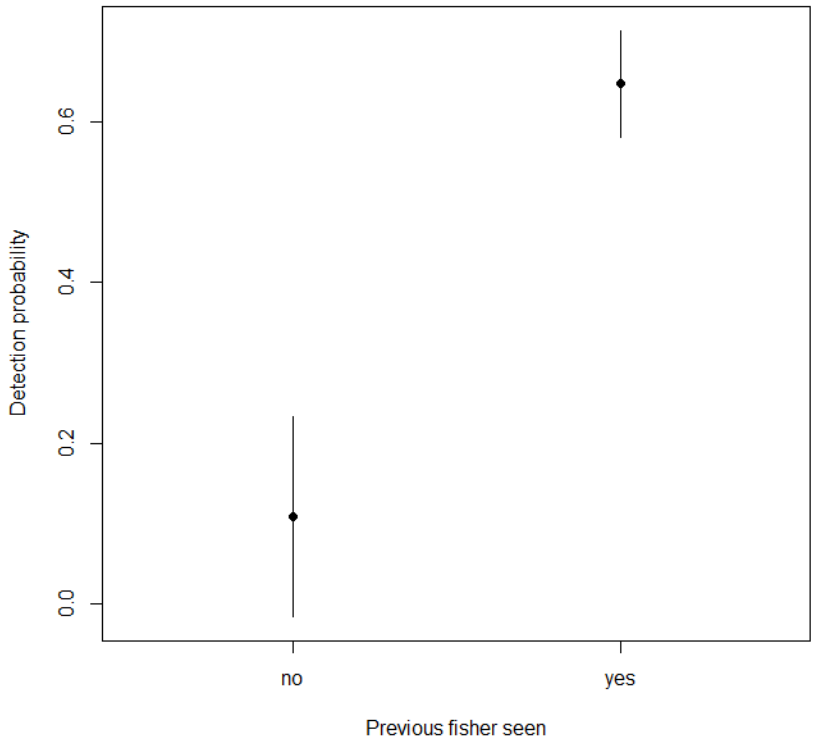

Figure 2. Detection probability (mean +/- standard error) as a function of whether or not a fisher was previously seen at the camera, with "no" indicating no fisher previously seen, and "yes" indicating that a fisher was previously seen.

Model selection results for occupancy based on the best-supported detection model indicated that distance to a wilderness boundary was the best predictor of fisher occupancy, along with the proportion of medium and large trees in the buffer (Table 4). The distance variable was present in all models up to 86% cumulative AIC weight. The model-averaged coefficient for distance to a wilderness boundary indicated that the probability of occupancy decreased as distance from a wilderness boundary increased ($\beta = -0.0024 \pm 0.0015$[SE]; Figure 3; Appendix A). The best-supported model also included the proportion of medium and large trees in the buffer, though this variable was not included in any other models containing up to 95% of the model weight, and the model-averaged coefficient shows negligible support for this habitat coefficient ($\beta = 2.36 \pm 4.68$; Figure 3; Appendix A). There was no model support for dense canopy cover, the proportion of tribal trust and fee lands in the buffer ("non-wilderness"), or the proportion of wilderness in the buffer (Appendix A).

Mean occupancy was slightly higher in 2017 than 2018. We estimated that site occupancy was 0.308 (0.187 – 0.461 95% CI) in 2017, with a mean detection probability of 0.491 (0.378 – 0.606). Estimated occupancy in 2018 was 0.240 (0.124 – 0.412), with a mean detection probability of 0.291 (0.171 – 0.449; Table 1).

Table 4. Model selection results for predicted occupancy of fishers in and surrounding Makah lands on the northwestern Olympic peninsula in 2017-2018, with detection probability (p_{ij}) modeled as p_{ij} (previous fisher). Occupancy covariates examined include distance to late seral forest boundary (distance to late seral), proportion tribal trust and fee lands (tribal non-wilderness), proportion late seral protected forest (all late seral protected), proportion medium and large trees (med/lg trees), and percent closed canopy (closed canopy cover).

Occ Model	*w*	K	AIC_c	ΔAIC_c	Cum *w*
Med/lg trees + Distance to wilderness	0.3524	5	199.87	0.00	0.35
Distance to wilderness	0.2885	4	200.27	0.40	0.64
Tribal non-wilderness + Distance to wilderness	0.2212	5	200.8	0.93	0.86
Tribal non-wilderness + Dense canopy cover	0.0551	5	203.58	3.71	0.92
Tribal non-wilderness	0.0472	4	203.89	4.02	0.96
Tribal non-wilderness + Med/lg trees	0.0174	5	205.88	6.02	0.98
Dense canopy cover	0.0066	4	207.81	7.95	0.99
Wilderness	0.0054	4	208.21	8.35	0.99
Wilderness + Dense canopy cover	0.0037	5	208.97	9.11	1.00
Med/lg trees	0.0025	4	209.79	9.92	1.00

Fisher abundance

Because detection probabilities were high, our estimates of fisher abundance generally approximated the raw count of individual fishers detected during the study in 2017. The null model and behavior model were equivalent and accounted for 99% of the model weight (Table 5). Estimated fisher abundance was highest and most variable (8 [6 – 31, 95% CI]) with the behavior model, and approximately equivalent for the remaining 3 models, each producing mean estimates of about 6 fishers, and confidence intervals ranging from 6 – 12 fishers (Table 5). Our results provide support for a trap-happy response, whereby the initial probability of detection p (0.21 [0.025 – 0.73]) was lower than the probability of subsequent detection (0.53 [0.31 – 0.73]); these findings are consistent with results from our occupancy analysis.

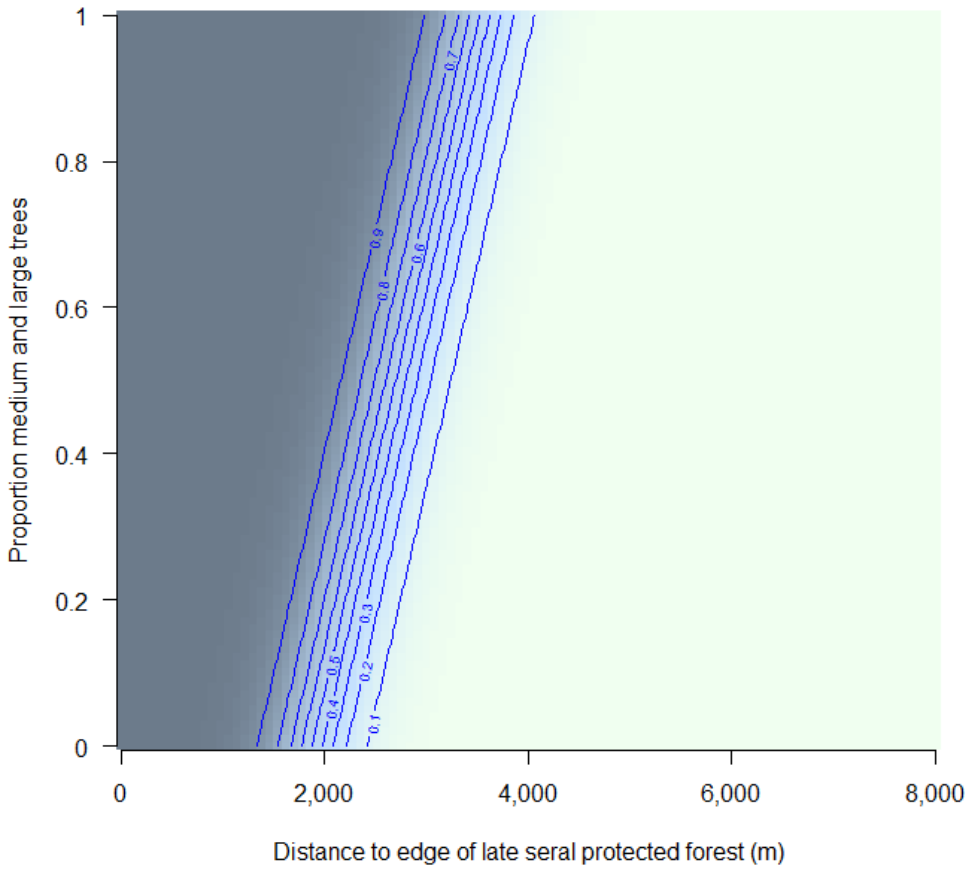

Figure 3. Fisher occupancy estimated from the top model as a function of distance to late seral forest boundary (m) and the proportion of medium and large trees in a 400-m buffer around the cameras, Northwestern Olympic Peninsula, 2017-2018. Blue lines indicate mean expected occupancy, where occupancy decreases from dark blue to light blue.

Table 5. Model selection results and estimated abundance of fishers in and surrounding Makah lands on the northwestern Olympic peninsula in 2017.

Model	Description	*w*	K	AICc	ΔAICc	Cum. *w*	Abundance estimate (95% CI)
M0	Null	0.4976	1	51.19	0.00	0.50	6.2 (6.0 – 9.2)
Mb	Behavior	0.4890	2	51.22	0.03	0.99	7.9 (6.1 – 31.3)
Mt	Time	0.0102	6	58.96	7.77	1.00	6.1 (6.0 – 8.5)
Mtb	Time + Behavior	0.0023	7	61.89	10.70	1.00	6.3 (6.0 – 12.3)

Discussion

In this study, we (1) examined habitat and land-management associations with fisher occupancy, and (2) estimated fisher population size on the Makah Reservation and surrounding lands on the northwestern Olympic Peninsula, Washington, in 2017 and 2018. This study paralleled the larger study conducted across the entire Olympic Peninsula from 2013 to 2016 (Happe et al. 2019), but with a denser sampling grid. Similar to the Peninsula-wide study, we found that fisher occupancy declined as distance to a wilderness boundary increased. We found no association between fisher occupancy and either Tribal fee, Trust, or wilderness lands. This result indicates that fishers, even at small spatial scales (i.e., much smaller than their home range), are responding to features found near the interface between protected forest lands and lands that are more intensively managed for timber production. Our findings support Happe et al.'s (2019) hypotheses that wilderness may provide the requisite denning and resting sites that are key to fisher survival, while managed forest lands likely contain more abundant prey, especially snowshoe hares and mountain beavers. Targeted studies of prey abundance in these areas, combined with telemetry work to identify the extent to which fishers move between these 2 land designations, would help clarify this association.

As with the Peninsula-wide study, we found no evidence that the proportion of medium and large trees or dense canopy cover has a strong influence on fisher occupancy. As noted by Happe et al. (2019), canopy cover regenerates quickly in our northwestern forests, and may not be a good predictor of fisher denning or resting habitat. Indeed, canopy cover was relatively high throughout the study area, with an average of 88% dense canopy cover in the buffer. The percent of medium and large trees in the buffer was more variable (mean = 58%, SD 24%), but still represented the range of values for this habitat variable where fishers have been detected in other studies (summarized by Happe et al. 2019).

Fisher detections and estimated fisher occupancy were lower in 2018 than 2017. Although a different set of grids was sampled each year (grids 1-3 in 2017 and 2-4 in 2018), we also noted that bear detections were much higher in 2018 than 2017 (Table 3). Bears and other species are attracted to the bait left for fishers and regularly ate bait and destroyed hair-snare cubbies when visiting the camera stations. In 2017, the bait and cubby were non-functional in 50% of all station checks, with half of those disruptions attributed to bears. In 2018, the bait and cubby were non-functional in 63% of all station checks, with 59% of those disruptions attributed to bears. This could have affected fisher detections on cameras as well as limited hair sampling. While fishers were detected on cameras on occasions where bait and cubbies were destroyed, we do not know if those fishers visited the stations before or after the bait and cubby were destroyed. Although bait persistence (i.e., the proportion of the sampling interval that bait was present) was not important to fisher detection in the Peninsula-wide study (Happe et al. 2019), it may be useful to investigate the effects of bait persistence on fisher detections in future studies or to check camera stations more frequently.

We were surprised to find that no fishers were detected in grid 2 during either year of the study. This grid was surveyed during different time periods in each year and appeared to contain habitat features that were similar to the other grids. In addition, fishers have been observed using this area during past monitoring efforts. Further work using telemetry or cameras could clarify the extent to which fisher may be using that part of the Reservation.

Our estimate of fisher population size approximated the number of fishers detected during our study. Seven individual fishers were detected during 2 years of surveying (Table 2), whereas our estimate of the number of fishers present in 2017 ranged from 6-8. However, our estimate did not include the many fisher detections that could not be identified to individual. One of the challenges of estimating population size was the large number of unknown individuals detected during the study. Although hair samples were collected weekly, many of the samples did not contain DNA of sufficient quality to identify species or individuals. Future researchers should look for ways to reduce DNA degradation in collected hair samples.

Finally, one of our original objectives was to use spatial capture-recapture methods to estimate the density of fishers in our study area. Due to the small number of individuals detected and their limited spatial extent, we were unable to use this class of models to estimate density. Spatial models can provide more accurate estimates of population size because they incorporate known space use of individuals. Further camera and hair-snare work, in combination with telemetry, may provide the opportunity to estimate density of fisher in this and other areas on the Peninsula in the future.

Collectively, these results indicate that the existing mosaic of wilderness and actively managed forest lands has sustained a small population segment of fishers using Makah Tribal lands. The data presented here provide a useful baseline for future monitoring of status of fishers on the Makah Reservation. Additional telemetry-based research would help us to better understand how fishers are using this landscape mosaic to meet their denning, resting, and foraging requirements.

Literature Cited

Beyer, H. L. 2012. Geospatial modelling environment (version 0.7.2.1). Software. URL: *http://www.spatialecology.com/gme*.

Happe, P.; K. J. Jenkins; R. McCaffery; J. Lewis; K. Pilgrim; and M. K. Schwartz. 2019. Occupancy patterns in a reestablishing fisher population following reintroduction. Journal of Wildlife Management. *https://doi.org/10.1002/jwmg.21788*

Jenkins, K. J., and P. J. Happe. 2013. Sampling design and field protocols for non-invasive fisher surveys on the Olympic Peninsula, Washington (Version 2013-1).

Lewis, J.C. 2014. Post-release movements, survival, and resource selection of fishers (pekania pennanti translocated to the Olympic Peninsula of Washington. Ph.D. Dissertation. University of Washington. Seattle.

Lewis, J. C., and G. E. Hayes. 2004. Feasibility assessment for reintroducing fishers to Washington. Washington Department of Fish and Wildlife, Olympia, USA.

Lewis, J. C., K. J. Jenkins, P. J. Happe, D. J. Manson, and M. McCalmon. 2016. Landscape-scale habitat selection by fishers translocated to the Olympic Peninsula of Washington. Forest Ecology and Management 369:170–183.

Lofroth, E. C., C. M. Raley, J. M. Higley, R. L. Truex, J. S. Yaeger, J. C. Lewis, P. J. Happe, L. L. Finley, R. H. Naney, L. J. Hale, A. L. Krause, S. A. Linvingston, A. M. Myers, and R. N. Brown. 2010. Conservation of fishers (*Martes pennanti*) in South-Central British Columbia, Western Washington, Western Oregon, and California—Volume I: Conservation Assessment. USDI Bureau of Land Management, Denver, Colorado, USA.

Powell, R. A., S. W. Buskirk, and W. J. Zielinski. 2003. Fisher and marten *in* G. A. Feldhamer, B. C. Thompson, and J. A. Chapmen editors, Wild mammals of North America: biology management, and conservation 2nd ed. Johns Hopkins University Press, Baltimore, Maryland. pp. 635-649.

Royle, J. A., A. J. Magoun, B. Gardner, P. Valkenburg, and R. E. Lowell. 2011. Density estimation in a wolverine population using spatial capture-recapture models. Journal of Wildlife Management 75:604-611.

Sun, C. C., A. K. Fuller, and J. A. Royle. 2014a. Trap configuration and spacing influences parameter estimates in spatial capture-recapture models. PLoS ONE 9(2):e88025 (1-9).

Sun, C. C., A. K. Fuller, and J. A. Royle. 2014b. Correction: trap configuration and spacing influences parameter estimates in spatial capture-recapture models. PLoS ONE 10(10):e0141634 (1-6).

Sweitzer, R. A., B. J. Furnas, R. H. Barrett, K. L. Purcell, and C. M. Thompson. 2016. Landscape fuel reduction, forest fire, and biophysical linkages to local habitat use and local persistence of fishers (*Pekania pennanti*) in Sierra Nevada mixed-conifer forests. Forest Ecology and Management 361:208-225.

U.S. Fish and Wildlife Service. 2019. Endangered and threatened wildlife and plants; threatened species status for the West Coast Distinct Population Segment of fisher. Federal Register 84(21):644–645.

Appendix A.

Model coefficients and standard errors[1] for covariates predicted to affect fisher detection probability on the Makah Reservation and surrounding lands on the Olympic Peninsula, Washington, summers 2017-2018. Detection covariates examined include whether or not a fisher was previously detected at the site (previous fisher), Julian date (date), quadratic effect of Julian date (date^2). Occupancy was modeled as constant in all detection models.

Detection Model	K	AIC	ΔAIC	cum *w*	pInt	SEpInt	ppf	SEppf	pdate	SEpdate	pdate2	SEpdate2	psiInt	SEpsiInt
previous fisher	3	207.89	0	0.66	-2.10	1.28	2.71	1.32					0.50	2.29
date + previous fisher	4	209.88	2	0.91	-2.12	1.34	2.73	1.39	0.013	2.03			0.53	2.45
date + date^2 + previous fisher	5	211.86	3.97	1	-2.05	1.38	2.72	1.36	0.014	0.204	-0.036	0.23	0.47	2.29
null	2	224.39	16.51	1	-0.16	0.20							-0.76	0.29
date	3	224.57	16.68	1	-0.19	0.21			-0.255	0.191			-0.75	0.29
date + date^2	4	226.35	18.46	1	-0.32	0.35			-0.252	0.189	0.103	0.22	-0.75	0.29
Model-averaged coefficients and SEs					-2.10	1.30	2.71	1.34	0.004	1.004	-0.003	0.07	0.50	2.33

[1]pInt=coefficient for detection model (p) intercept
SEpInt=standard error of coefficient for dectection model intercept
ppf=cooefficient for effect of previous fisher on detection probability
SEppf=standard error of coefficient for effect of previous fisher
pdate=coefficient for effect of Julian date on detection probability
SEpdate=standard error of coefficient for effect of Julian date on detection probability
pdate2=coefficient for quadratic effect of Julian date on detection probability
SEpdate2=standard error of coefficient for quadratic effect of Julian date on detection probability
psiInt= coefficient for occupancy model (psi) intercept
SEpsiInt= standard error of coefficient for occupancy model intercept

Appendix A. Continued.

Model coefficients and standard errors[1] for covariates predicted to affect fisher occupancy probabilities on the Makah Reservation and surrounding lands on the Olympic Peninsula, Washington, summers 2017-2018. All occupancy models are based based on the top-ranked detection model, which included an effect of previous fisher detections on the probability of subsequent fisher detections. Occupancy covariates examined include distance to late seral forest boundary (distance to late seral), proportion tribal trust and fee lands (tribal non-wilderness), proportion late seral protected forest (all late seral protected), proportion medium and large trees (med/lg trees), and percent closed canopy (closed canopy cover) in a 400-m buffer around each site.

Occupancy Model	pInt	SEpInt	ppf	SEppf	psiInt	SEpsiInt	psitribal	SEpsitribal	psiAllW
Med/lg trees + Distance to late seral	-2.32	0.27	2.94	0.40	7.64	4.73			
Distance to late seral	-2.37	0.27	3.00	0.40	7.27	6.07			
Tribal non-wilderness + distance to late seral	-1.83	0.39	2.43	0.48	3.07	1.92	-2.12	1.47	
Tribal non-wilderness + closed canopy cover	-1.69	0.53	2.30	0.61	-3.77	3.44	-2.67	1.56	
Tribal non-wilderness	-1.92	0.80	2.53	0.85	2.13	3.94	-2.96	3.52	
Tribal non-wilderness + Med/lg trees	-1.91	0.82	2.52	0.87	2.20	4.21	-2.95	3.50	
closed canopy cover	-2.51	0.62	3.11	0.68	-9.31	20.20			
All late seral protected	-2.01	0.94	2.62	0.98	-0.01	1.27			2.03
All late seral protected + closed canopy cover	-2.02	0.87	2.63	0.92	-4.17	4.42			1.76
Med/lg trees	-2.08	1.22	2.69	1.26	0.11	2.11			
Model-averaged coefficients and SEs	-2.16	0.39	2.78	0.49	5.32	4.88	-0.80	1.22	0.02

(Continued)

Appendix A. Continued.

Occ Model	SEpsiAllW	psidistW	SEpsidistW	psiclosedCC	SEpsiclosedCC	psicomb	SEpsicomb
Med/lg trees + Distance to late seral		0.00	0.00			6.72	7.87
Distance to late seral		0.00	0.00				
Tribal non-wilderness + distance to late seral		0.00	0.00				
Tribal non-wilderness + closed canopy cover				6.06	4.51		
Tribal non-wilderness							
Tribal non-wilderness + Med/lg trees						-0.15	2.81
closed canopy cover				13.04	29.20		
All late seral protected	3.76						
All late seral protected + closed canopy cover	3.43			4.83	5.85		
Med/lg trees						0.63	2.13
Model-averaged coefficients and SEs	0.35	0.00	0.00	0.44	2.62	2.37	4.69

[1]pInt=coefficient for detection model (p) intercept
SEpInt=standard error of coefficient for dectection model intercept
ppf=cooefficient for effect of previous fisher on detection probability
SEppf=standard error of coefficient for effect of previous fisher
psiInt=coefficient for occupancy model (psi) intercept
SEpsiInt=standard error of coefficient for occupancy model intercept
psitribal=coefficient for effect of proportion tribal trust and fee lands in the buffer on fisher occupancy
SEpsitribal=standard error of coefficient for effect of proportion tribal trust and fee lands
psiAllW=coefficient for effect of proportion of all protected late-seral forests in the buffer on fisher occupancy
SEAllW=standard error of coefficient for effect of all protected late-seral forests
psidistW=coefficient for effect of distance to late-seral forest boundary on fisher occupancy
SEdistW=standard error of coefficient for effect of distance to late-seral forest boundary
psiclosedCC=coefficient for effect of proportion of closed canopy (>70%) in the buffer on fisher occupancy
SEpsiclosedCC=standard error of coefficient for effect of closed anopy cover
psicomb=coefficient for effect of proportion of medium and large trees (>25cm dimeter at breast height) in the buffer on fisher occupancy
SEpsicomb=standard error of coefficient for effect of proportion of medium and large trees