

## AN ABSTRACT OF THE THESIS OF

Brent R. Barry for the degree of Master of Science in Wildlife Science presented on June 1, 2018.

Title: Distribution, Habitat Associations, and Conservation Status of Pacific Fisher (*Pekania pennanti*) in Oregon

Abstract approved: \_\_\_\_\_

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Fishers (*Pekania pennanti*) are medium sized mustelids endemic to North America. Two fisher populations persist in Oregon: an indigenous population in southwestern Oregon, and a reintroduced population in the southern Cascade Mountains. Despite candidacy for listing under the Endangered Species Act, current information on fisher populations in Oregon is scarce. We conducted surveys using motion-activated cameras and scent detecting dog teams to assess the distribution of fishers. We quantified the potential for both populations to expand through time using a spatially explicit reaction-diffusion equation under varied initial conditions and growth scenarios. We deployed 1,855 camera survey stations equating to 591 sample units collecting 4,779,178 photographs. Detection dog teams surveyed 196 sample units. We detected fishers at 96 unique sample units and at 148 individual camera stations and 39 detection dog units, confirming the presence of the indigenous and reintroduced populations. The southern Cascade Mountains reintroduced population appears to have shifted or possibly occupy 67% less area than previously believed and the population extent was less than expected, except under our lowest growth model. We confirmed a larger indigenous population, but with a spatial extent 26% less than previously believed potentially as a result of recent wildfires or time-limited expansion. The indigenous population extent matched expectations under several growth scenarios suggesting multiple pathways to reach the contemporary distribution. There was no evidence of either population expanding into historically occupied forests. Understanding which factors limit fisher population growth could provide guidance for additional population supplementation efforts, categorize suitable habitat, and identify significant species interactions.

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Distribution, Habitat Associations, and Conservation Status of Pacific  
Fisher (*Pekania pennanti*) in Oregon.

by  
Brent R. Barry

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

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Brent R. Barry, Author

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# **CHAPTER 1**

## General Introduction

Brent R. Barry

### *Distribution*

Fishers (*Pekania pennanti*) are a medium sized carnivore widely distributed throughout the forested areas of North America (Powell and Zielinski 1994). Since the mid-1800s fishers have experienced a dramatic range contraction largely due to anthropogenic influences such as, habitat loss (Powell and Zielinski 1994), historical over-trapping, predator-control crusades and incidental trapping (Zielinski and Lewis 1996). The core of the fisher's geographic range lies within the boreal forests of Canada where they were never extirpated (Gibilisco 1994). The central, eastern, and western portions of the fishers distribution experienced historical contractions and recovery has progressed in most areas but in the west (Gibilisco 1994, Lewis et al. 2012).

Fishers along the Pacific coast reside in a peninsular distribution from southern British Columbia to the southern Sierra Nevada of California (Gibilisco 1994). Currently this population occupies a small fraction of their historic range (Lewis et al. 2012) despite the availability of habitat considered to be suitable, limited forest fragmentation (Lofroth et al. 2010) and an extensive post trapping recovery period (Bailey 1936, Zielinski and Lewis 1996, Lewis and Stinson 1998). Fishers within this area have a unique haplotype separate from fishers elsewhere in their range (Drew et al. 2003, Wisely et al. 2004) and exhibit some of the highest genetic structuring reported for a mammalian carnivore (Wisely et al. 2004). Due to these factors fishers in the Pacific were proposed for Federal listing under the Endangered Species Act in 2004 as a West Coast distinct population segment (USFWS 2004) but the proposed ruling was later withdrawn (USFWS 2016). Conservation efforts such as reintroductions, translocations and habitat restorations are ongoing in these states with additional restorative actions planned in the near future (Lewis 2014, Halsey et al. 2015, Hiller 2015).

## *Habitat*

In western North America fishers are most regularly associated at large spatial scales with conifer, mixed-conifer or mixed conifer-hardwood forests with moderate to dense canopy cover and at low to mid elevations (Jones and Garton 1994, Weir and Harestad 1997, Aubry and Raley 2006, Davis et al. 2007, Zielinski et al. 2010, Raley et al. 2012). High elevation areas are thought to be unsuitable for fishers due to increased energetic costs while travelling through powdery snow (Raine 1983, Krohn et al. 1997). Historical fisher distributions in the Pacific are consistent with areas that receive comparatively little snow and extant remnant populations within the Pacific appear to fit this paradigm (Aubry and Houston 1992, Krohn et al. 1997, Davis et al. 2007, Lofroth et al. 2010, Sweitzer et al. 2016). In the Pacific Northwest, low to mid elevation forests historically contained the most merchantable and accessible timber (Harris 1984). These forests were subsequently degraded through resource extraction activities like clear-cut logging (Harris et al. 1982, Harris 1984, Morrison 1988, Bolsinger and Waddell 1993, Kennedy and Spies 2004).

Fisher habitat often contains different forest types and successional stages, while maintaining a high proportion of late successional forest or forest with structural features associated with mature stands, and a low amount of open or non-forested environments (e.g. grassland, wetlands and areas of low canopy cover) (Powell and Zielinski 1994, Weir and Corbould 2010, Sauder and Rachlow 2014). Landscapes containing high floristic and successional diversity are thought to provide a wide variety of prey species (Powell and Zielinski 1994, Raley et al. 2012). Hagar (2007) suggested that a diverse understory assemblage of herbs, broad-leaved trees and shrubs is central to increasing food web complexity across multiple trophic levels in Pacific Northwest conifer forests. The decreasing availability of late

successional forests may limit fisher distributions (Powell and Zielinski 1994), specifically reduction in features associated with complex vertical (e.g. snags, trees) and horizontal (closed canopies, large logs) structures commonly found within, but not exclusive to, late successional forests (Raley et al. 2012). Fishers' reliance on habitat and structural complexity maybe related to denning and resting locations, prey availability, and escape cover from predators.

Throughout their range, fishers are arboreal cavity obligates for reproduction (Coulter 1966, Aubry and Raley 2006, Matthews et al. 2013, Green et al. 2018) and require resting structures for thermoregulatory benefits and protection from predators (Raley et al. 2012). Fishers select rest sites and den sites with characteristics commonly found in late successional forests; large diameter trees, coarse downed woody material, and snags (Powell and Zielinski 1994, Zielinski et al. 2004b, Aubry and Raley 2006, Aubry et al. 2013, Matthews et al. 2013, Green et al. 2018). Nevertheless, fishers have been shown to occupy and reproduce in mixed managed forest landscapes not considered to be mature or late-successional (Matthews et al. 2013, Facka et al. 2016, Lewis et al. 2016). Managed landscapes where fisher persist maintain a juxtaposition of fisher habitat elements, typically via the retention of substantial mature residual components in harvested stands such as trees with cavities, large logs, and snags (Klug 1997, Weir and Corbould 2010, Matthews et al. 2013).

Hardwoods have been hypothesized to contribute an important role to fisher habitat at multiple scales. At fine spatial scales, some hardwood species have been shown to provide critical structures for resting and denning due to their tendency to develop cavities (Zielinski et al. 2004b, Higley and Mathews 2009, Green et al. 2018). Fishers select hardwoods more often than conifers for rest and den sites in regions where both hardwood and conifers occur, even if they are only a minor component of the area (Lofroth et al. 2010, Green et al. 2018). At larger

spatial scales, forests that contain a hardwood component, especially mast-producing hardwoods, have been hypothesized to provide fishers with more diverse and abundant prey (Carroll et al. 1999, Zielinski et al. 2004a, Yaeger 2005). Nonetheless, selection for mixed conifer-hardwood forests has not been demonstrated and whether forests containing hardwood elements provide additional prey resources remains unproven (Raley et al. 2012), but see Jensen et al. (2012).

### *Fishers in Oregon*

Fishers were once widespread in Oregon, described as occupying the forested areas of the Cascade Range, west into the Coast Range, and the Willowa Mountains in the northeast (Bailey 1936) (Fig. 1). By the mid-1900s they were considered extremely rare or extirpated (Mace 1970); whether fishers were fully extirpated at one point remains unknown (Olterman and Verts 1972, Yocom and McCollum 1973). Currently, Oregon contains two lineages of fishers: an indigenous population spanning parts of northern California and southwestern Oregon and a reintroduced southern Oregon Cascade Mountains population.

The indigenous fisher population is part of the largest remnant population in the Pacific states, and persists in parts of northwestern California and southern Oregon (USFWS 2016). The introduced population was stocked with fishers from British Columbia and Minnesota in 1961 and again in 1977-1981, representing the earliest population supplementation efforts in the Pacific states (Kebbe 1961, Aubry and Lewis 2003, Drew et al. 2003). The Oregon Department of Fish and Wildlife, USDA Forest Service, and private timber companies aimed to reintroduce fishers as a natural predator of porcupine (*Erethizon dorsatum*), which were considered a nuisance at the time (Kebbe 1961). The reintroduction in 1961 was considered to have failed but the 1977-1981 effort is considered to be successful (Aubry and Lewis 2003). Prior to 2008, verifiable occurrence records suggested this population had not expanded despite ~30 years post

reintroduction and remained genetically isolated from the indigenous population (Aubry and Lewis 2003, Drew et al. 2003, Wisely et al. 2004). In 2008, fishers with genetic profiles consistent with the reintroduced population were detected south of the known reintroduced population area for the first time (J. Stevens, Bureau of Land Management, personal communication; S. Hayner, Bureau of Land Management, personal communication). In 2014, genetic data confirmed a female, likely from the indigenous population, had crossed Interstate 5 and produced hybrid offspring with the reintroduced population in this area of suspected recent colonization (Pilgrim and Schwartz 2012;2014;2015). These detections and a lack of contemporary landscape level surveys exemplify that our current understanding of fisher distributions in Oregon is incomplete.

## **CHAPTER 2**

Distribution, habitat associations, and conservation status of Pacific  
fisher (*Pekania pennanti*) in Oregon

Brent R. Barry, Katie Moriarty, and Taal Levi



## Introduction

Information regarding the geographic range of species is critical to conservation planning and management (Ferrier 2002, Funk and Richardson 2002). In the United States range contraction is a key qualification to receive regulatory protections afforded by the Endangered Species Act (USFWS 2016). Detecting shifts in species distributions are often challenging due to a lack of systematically collected monitoring data, leaving decision makers with vague or qualitative information plagued with substantial uncertainty (Hijmans et al. 2000, Loiselle et al. 2008, Aubry et al. 2017). This uncertainty has legal, regulatory, and policy implications when used to evaluate the effects of management activities or to decide whether listing under the Endangered Species Act (ESA) is warranted.

Paramount to recovering species suffering from declines and to restore connectivity between populations is the availability of habitat. Habitat is usually defined as the resources and conditions present (abiotic or biotic) in an area that facilitate occupancy (Morrison et al. 2012). The quantification of habitat quality is a key component of applied ecology to meet management objectives (Van Horne 1983, Franklin et al. 2000, Johnson 2007) or designate critical habitat (Hagen and Hodges 2006, Camaclang et al. 2015) and to inform ecological theory (Morrison et al. 2012). Habitat quality is often described as a function of fitness or per-capita population growth, where conditions favoring increased survival and reproduction are considered higher values of habitat quality (Van Horne 1983, Johnson 2007). We can, therefore, infer habitat quality as a reflection of population growth when measuring the efficacy of species recovery objectives.

Populations that suffer from reduced, contracted, or isolated ranges are typically of conservation interest due to increased vulnerability to genetic, demographic, or stochastic

processes (Shaffer 1987, Stacey and Taper 1992, Lande 1993, Keller and Waller 2002). Species suffering from reduced ranges inherently have a reduced capacity to buffer against habitat loss from environmental stochastic processes (Glynn and De Weerd 1991, Lande 1993), or catastrophes such as floods, wildfires (Brown et al. 2001), tsunamis (Linnell et al. 2018) or other events. Consequently, management aimed at improving conditions for imperiled populations frequently emphasize reducing habitat loss. Specifically, in western North America, managers often focus on methods to reduce impacts of fire regimes that have been pushed outside of their historic norms (McKenzie et al. 2004, Westerling et al. 2006) and can further reduce habitat availability (Courtney et al. 2004, Spies et al. 2006).

In the Pacific Northwest region of North America, the range contraction of fishers (*Pekania pennanti*), a medium-sized member of the mustelid family, has prompted three petitions to list the fisher as Threatened under the ESA since 1990 (Beckwitt 1990, Carlton 1994, Greenwald et al. 2000). In 2004, the US Fish and Wildlife Service proposed a ruling to list a distinct population segment for the Pacific States fishers under the ESA as “warranted but precluded” (USFWS 2004). Species found to be “warranted but precluded” are such that merit listing, but remain on the candidate species list due to other higher priority actions. The proposed “warranted but precluded” finding was later withdrawn by the US Fish and Wildlife Service, which concluded that the stressors previously identified were not existential threats to the range-wide fisher distinct population segment (USFWS 2016). This decision is currently under litigation (Geis and Loaire 2016).

Although fishers have been extensively studied in California (Zielinski et al. 1995, Carroll et al. 1999, Gabriel et al. 2015, Sweitzer et al. 2016, Furnas et al. 2017), little is known about the status of fishers in Oregon, leaving a critical information gap relevant to the regulatory

status of fishers and ongoing litigation. Fishers were once widespread in the state, described as occupying the forested areas of the Cascade Range, west into the Coast Range, and the Wallowa Mountains in the northeast (Bailey 1936) (Fig. 1). By the mid-1900s they were considered extremely rare or extirpated (Mace 1970); whether fishers were fully extirpated at one point remains unknown (Olterman and Verts 1972, Yocom and McCollum 1973). Currently, Oregon contains two lineages of fishers: an indigenous population spanning parts of northern California and southwestern Oregon (hereafter “indigenous population”) and a reintroduced southern Oregon Cascade Mountains population (hereafter “reintroduced population”; Fig. 1). The range estimates for both populations were developed by the US Fish and Wildlife Service from previous range mapping (Lewis et al. 2012) and additional verified documented fisher occurrences since 2003 (USFWS 2016). Nonetheless, whether fisher occupied the designated region was unknown.

Range-wide surveys have not been completed for either the indigenous or reintroduced population, and limited survey effort has been expended elsewhere. The absence of systematic surveys in many areas reduces the ability to infer the spatial extent or connectivity among known populations and to detect potential remnant fisher populations or populations that may have been established over the preceding decades. Further, habitat modeling conducted during the ESA listing process predicts large tracks of suitable habitat within the Coast Range and the central and northern Cascade Range of Oregon where coordinated survey efforts have been low or absent (Fitzgerald et al. 2014) (Fig.1).

The current distribution of fishers is likely to be influenced by the availability of suitable habitat and the ability of fisher populations to expand into that habitat. Comparing the expected spatial extent of fishers to the actual extent can help determine whether unoccupied areas are

unsuitable or too distant to expect fisher occurrence. Thus, modelling the spatial growth of fishers informs whether the absence of fishers in some areas was the result of time-limited expansion (i.e., fishers haven't had an opportunity to recolonize) and should also be correlated with habitat quality. We predicted suitable habitat could be available and unoccupied simply because fishers were excluded due to time-limited expansion constraints. Alternatively, if predicted habitat was accessible but unoccupied, we inferred habitat quality, as measured by per-capita population growth, was locally poor. We initiated a regional effort spanning much of the contemporary and historic range of fishers in western Oregon. Our objectives were to (1) describe the current distribution of fishers by surveying within and outside of range projections, (2) simulate population growth with multiple growth values, and (3) evaluate the similarity between simulations and the observed fisher distribution.

## **Methods**

### *Study Area*

We conducted the largest carnivore survey in Oregon - 64,280 km<sup>2</sup> of the former fisher range in western Oregon from the Cascade Crest to the Pacific Ocean, excluding the northern portion of the Coast Range, urban areas, and non-forested areas in the Willamette, Umpqua, and Rogue River valleys. A four lane highway (Interstate 5) bisects the study area, and the majority of urban development is centered on this interstate. Our study area predominantly included federal lands (e.g., US Forest Service, Bureau of Land Management), but also included private industrial timberlands (e.g., Hancock Timber, Weyerhaeuser), and state owned lands (e.g., Oregon Department of Forestry). Elevation ranges from 0-2,286m (0-7,500ft) with the highest elevations in the Cascade Mountains and Klamath Mountains. Precipitation is highly variable across the study area ranging from 50cm to 300cm (PRISM Climate Group Oregon State

University 2015) but primarily occurs between October and April as rain at lower elevations and the coast, and as snow at high elevations.

### *Study Design*

To describe the current range of fishers we combined data from multiple surveys that used two methods, motion-activated cameras, (hereafter “camera traps”), and scent detection dog teams. To systematically survey western Oregon for fisher, we created a state-wide survey grid with 3-km spacing, then used a stratified random sample of grid locations with a minimum of 6-km spacing between locations (Beyer 2014). The spacing between sample units in our study design was intended to minimize the likelihood of detecting fishers across multiple sample units as this was approximately a female home range size. Stations were within 1000 m of a road or highway or 250 m of a trail for accessibility, and in forested cover types (e.g., we avoided water, grassland). We did not survey if stations occurred in non-permissioned private property, extreme terrain, or there were local safety concerns. Sample units consisted of 4 camera traps and/or a constrained time and area search by a detection dog team.

### *Camera trap Surveys*

We predominantly used two camera trap models (Browning, Model# BTC-6HD, Morgan, UT and Bushnell, Model# 119776, Overland Park, MI) with standardized camera modes (3 picture burst per trigger), time stamps, high or normal sensor sensitivity if excessive vegetation shots were taken, and a 5 second delay between shots. We used only black LED flash cameras to decrease the probability an animal being deterred by infrared flash. To obtain a high predicted probability of detection as recommended by Sweitzer et al. (2016), we surveyed for a minimum of 60 and 35 days ( $58.34 \pm 24.19$ ,  $43.53 \pm 33.28$ , mean  $\pm$  standard deviation), during summer (June 1-September 30) and winter (October 1- May 31), respectively. Cameras were checked at

variable intervals depending on the project, season, and locality (Table 1). At each sample unit, we placed three baited camera traps with a randomized bait treatment in an equilateral triangle spaced 1000m apart centered on the sample unit location, similar to the current Sierra Nevada Forest Carnivore protocol (Truex et al. 2013, Zielinski et al. 2013). An additional unbaited camera was set on a game trail, footpath, old logging road, or drivable road found within 50-150m of the northern most baited station to better detect animals not attracted to bait or lure. We baited with either chicken (250g), fish-flavored cat food (5.5oz can) with holes to disperse the scent, or “the kitchen sink” (chicken (250g), fish-based cat food (5.5oz can), an apple, and a biodegradable wood or cardboard plate (23cm) containing peanut butter-oat mixture (500 ml) and sliced apple) (Fig. 2). Each bait type was paired with an olfactory lure (Gusto, Minnesota Trapline Company, Pennock, MN). Multiple bait types were used for the purpose of evaluating efficiency in fisher detectability (see *Supplemental Information*).

Baited cameras were offset 50 m from maintained roads to minimize edge effects and potential theft, set 0.5–1.0 m above the ground, 2-4 m from the bait tree, and facing north to reduce direct sunlight and poor exposures. The baited tree also included a 100 cm long measuring strip marked in 10 cm increments to help distinguish animal size, and station-specific signs for photo record keeping were secured to the bait tree.

Photos were initially viewed in the field to determine if a target species was present and a hair snare should be deployed. Photos were later processed using photo-editing software *Picasa* (Google 2015). Each photo was viewed and the metadata edited to reflect species present in the photo, treatments assigned to the camera station (e.g., bait type, olfactory lure), and other station variables. The metadata was extracted using exiftool (Harvey 2016), and analyzed using program R (Team 2013)

### *Scent Detection Dog Team Surveys*

Scent detection dog teams consisted of a handler and dog with a minimum of 480 hours of training using lab and simulated field trials (e.g., hidden scats on training boards and in the duff). Maintaining dog focus for rare species can be increased by having multiple targets, allowing a reward for any of the target species. Although we only report fisher results, the teams were instructed to collect scat from fisher, Pacific marten (*Martes caurina*), bobcat (*Lynx rufus*), mountain lion (*Puma concolor*), and porcupine so we could gain information on potential competitors, predators, and other species of interest. Detection dog teams searched for a minimum of 4 hours within a 3x3 km<sup>2</sup> area centered on the random location and could then expand within 5x5 km<sup>2</sup> for a total of 6 hours. Scats were photographed, placed in paper bags, labeled, and dried.

Scats were collected from May-September 2016, and March-September 2017. We used DNA metabarcoding (Ji et al. 2013) to identify the species of the defecator. We extracted DNA for species identification in a room dedicated to processing degraded DNA using the DNeasy Blood and Tissue kit (Qiagen, USA). We performed the extractions in batches of ~15 and included an extraction blank as a negative control to monitor for cross-contamination. We used slightly modified vertebrate primers 12SV5F (TTAGATACCCCACTATGC) and 12SV5R (YAGAACAGGCTCCTCTAG) to amplify the ribosomal mitochondrial 12S gene region (adapted from (Riaz et al. 2011)). We performed polymerase chain reactions (PCR) in replicates of three per scat with Qiagen Multiplex PCR Kit (Qiagen, USA). Details on PCR cycling conditions and Illumina library preparations are provided in the Supplemental methods. The libraries were sent to the Center for Genome Research and Biocomputing at Oregon State University for 150-bp paired-end sequencing on the Illumina HiSeq 3000 platform (Illumina Inc,

San Diego, California, USA), demultiplexed and clustered with a custom shell script, and taxonomically assigned using BLAST (<https://blast.ncbi.nlm.nih.gov/Blast.cgi/>) against all 12S sequences in GenBank (<https://www.ncbi.nlm.nih.gov/genbank/>) and against a custom 12S library created for vertebrates not present in GenBank.

### *Estimating Range*

We chose to delineate the distribution of fishers using a bivariate normal kernel density estimator with a fixed bandwidth and a 95% isopleth (Worton 1989). Kernel density estimators create a two-dimensional probability density surface to locate a point at a given place, typically applied to home range studies (Worton 1989, Seaman and Powell 1996). Here we assumed that a kernel density surface would be analogous to a detection density surface. The smoothing parameter bandwidth is often chosen using biological criteria, by removal of outliers, or through least squares minimization (Silverman 1986, Seaman and Powell 1996). The spacing between sample units in our study design was intended to minimize the likelihood of detecting fishers across multiple sample units, thus, we assumed the smoothing parameter was more informative when the bandwidth was set to the minimum spacing between the center of sample units (6 km) and thereby overlapping multiple individuals. Sample units were filtered to ensure the minimum 6 km buffer between sample unit centers, which were occasionally violated due to field logistics or where intensive sampling was used to answer questions about survey methodologies outside the scope of this paper. The distribution of fishers was broken into two discrete population areas; the indigenous population west of Interstate 5 and the Southern Cascade Mountains population east of Interstate 5.

### *Spatial Extent Model*



We quantified the approximate expectations for the current distribution of fishers in the Oregon Cascade Mountains and Klamath-Siskiyou Mountains by using a spatially explicit reaction diffusion equation. We chose to model females exclusively because adult female survival has been correlated with fisher population stability (Spencer et al. 2011), and females tend not to disperse as far as males (thus limiting the growth of the population across space) (Aubry et al. 2005, Aubry and Raley 2006, Matthews et al. 2013). Our model used demographic parameters reported in the literature and assumed homogenous suitable habitat.

Whether fishers were fully extirpated from Oregon remains unknown. Some have suggested that few if any fishers persisted by the 1940's (Mace 1970, Harris et al. 1982, Aubry and Lewis 2003). Others concluded that fishers were present but rare based on a small number of incidental trapping records and unverified sightings in counties adjacent to a remnant population in California, or observed near release locations from reintroductions in 1961 (Olterman and Verts 1972, Yocom and McCollum 1973). We were unable to determine whether fishers were previously extirpated such that contemporary observations represent a colonizing wave from California or whether small isolated populations persisted in Oregon largely undetected. Thus, due to a lack of comprehensive verifiable historic records we modeled two scenarios for the indigenous population in the Klamath-Siskiyou Mountains; one where we conservatively assumed that fishers were completely absent from the state and recolonized from a remnant population in northwestern California (hereafter "extirpation and recolonization scenario") or that fishers remained at low densities in isolated refugia near the California border (hereafter "wilderness refugia scenario"). In the extirpation and recolonization scenario we assumed that a remnant population persisted in northwestern California within areas with >2 observations made from 1960-1974, as described by Yocom and McCollum (1973) and Schempf and White (1977).

This spatial extent was then initialized to 3/4 of carrying capacity in 1974. In the wilderness refugia scenario, the remnant population from the extirpation and recolonization scenario remains unchanged, but we assumed wilderness areas near the Oregon-California border (Kalmiopsis Wilderness, Red Butte Wilderness, and Siskiyou Wilderness) acted as refugia, containing a low density of fishers (we assumed 1/4 of carrying capacity). We used currently recognized wilderness areas harboring low density fisher populations in these simulations due to the absence of roads that would facilitate trapping, a predominant driver of fisher declines (Zielinski and Lewis 1996, Aubry and Lewis 2003), and protection of late successional habitat (Harris et al. 1982, Harris 1984). In both scenarios, the Cascades were populated by the known number of individuals ( $N=12$ ) at specified release locations and dates from reintroductions in 1977-1981, as the 1961 effort appeared unsuccessful (Aubry and Lewis 2003).

First, we constructed a modeling landscape ( $425 \times 425 \text{ km}^2$ ,  $180,625 \text{ km}^2$ ) comprised as an array of  $25 \text{ km}^2$  pixels. Following Levi et al. (2009), and Levi et al. (2011), we used a spatially explicit, reaction–diffusion equation to model the density of fishers at each location ( $x, y$ ) at time  $t$ , displayed as the number of individuals at each time step ( $N_{x,y,t}$ ). The density of fishers in the next time step,  $N_{x,y,t+1}$ , was a function of the simulated population size ( $N_{x,y,t}$ ), growth  $R(N_{x,y,t})$ , and migration  $M(N)$ :

$$N_{x,y,t+t} = N_{x,y,t} + R(N_{x,y,t}) + M(N)$$

We modeled density-dependent population growth,  $R(N_{x,y,t})$ , with the discrete theta-logistic equation:

$$R(N_{x,y,t}) = rN_{x,y,t} \left( 1 - \left( \frac{N_{x,y,t}}{K} \right)^\theta \right).$$

We used a value of  $\theta = 2$  for the strength of density-dependent declines in per-capita population growth. We used a carrying capacity,  $K$ , of 1 female per  $25 \text{ km}^2$ , which is similar to an

average indigenous population density estimate, 1.65 fishers per 25km<sup>2</sup>, which for our purposes is an overestimate due to the inclusion of males (Furnas et al. 2017). In addition, estimates derived from female home ranges reported in Aubry and Raley (2006) for the introduced population result in a similar value of ~1 female per 25km<sup>2</sup> with an assumption of non-overlapping home ranges (Table 2). We varied values of  $K$  to determine model sensitivity to this parameter and found differences population size estimates but not spatial extent (see *Supplemental Information*).

The maximum intrinsic growth rate,  $r$ , is the growth rate that we would expect a fisher population to exhibit in the absence of density dependence. This parameter was estimated using a modified Euler-Lotka equation (Skalski et al. 2008) and demographic parameters reported in the literature (Table 2). We estimated the maximum intrinsic growth rate,  $r$ , to be 0.24 for fishers. We further varied demographic inputs using suboptimal reported values to compare different growth scenarios below our predicted maximum. We defined moderate growth as  $r = 0.14$  and low growth as  $r = 0.04$ . We expect, *a priori*, the highest level  $r$  to be the most plausible because the demographic information used to parameterize  $r$  came from populations where density dependence was likely occurring. Our lowest  $r$  value is an unreasonable expectation for fisher based on estimates of  $r$  derived from body size and temperature (Hennemann 1983). Due to the linkage between habitat quality, fitness and population growth these values can also be considered indicators of habitat quality.

The migration term,  $M(N)$  was modeled as a diffusion process, where fishers move down a gradient from densely populated cells into less densely populated or unoccupied cells. Diffusion is faster when the difference between cell densities is greater (when the gradient is steeper). The migration term was represented as:

$$M(N) = D \nabla^2 N$$

where  $D$  is the diffusivity constant (distance<sup>2</sup>/time). The diffusivity constant was estimated using fisher dispersal metrics (see *Parameter Estimation* section). The Laplace operator,  $\nabla^2$ , often used to model heat flow or wave propagation, allows simulated fishers to expand from a known point. In two dimensions the Laplace operator was defined as:

$$\nabla^2 N = \frac{\partial^2 N}{\partial x^2} + \frac{\partial^2 N}{\partial y^2}$$

We used the “five-point stencil” technique (up, down, left, right) to apply the Laplace operator within our model. The relative amount of change per location is approximated from the values of the neighbors in four directions. For cells 5-km across, and for a one-year time step, the stencil approximates the Laplace operator as:

$$D \nabla^2 N \approx D (N_{x+1,y,t} + N_{x-1,y,t} + N_{x,y+1,t} + N_{x,y-1,t} - 4N_{x,y,t}).$$

In this exploration, we approximate the spatially-explicit simulation to infer potential population growth over the past 41 years for the indigenous population and the 36 years since release for the introduced population.

## Results

### *Distribution*

We deployed 1,855 camera stations for 92,059 trap nights and obtained 4,779,178 million photographs. All cameras stations were operational for an average of 42.24 (SD =40.03) and 88.8 (SD = 89.26) days during summer and winter respectively. Fishers were detected at 148 camera stations within 70 sample units. While no probability of detection estimates are provided here, we have confidence that our camera surveys detected populations of fisher when present. At baited stations, median latency to first detection ranged between 23.7- 49.8 days during summer and 17.8-20.0 days during winter (see *Supplemental Information*) which is less than our mean

survey duration for each season.

Detection dog teams surveyed 196 3x3 km<sup>2</sup> units over 185 survey days, averaging 17.3km traveled over an average of 7.0 hours within each unit. Dog teams collected 863 scats for all targeted species. Fisher were detected at 39 units and 148 genetically confirmed scats were collected. Some sample units were sampled both by detection dogs and camera surveys, resulting in 96 unique sample units that detected fishers.

### *Range Delineation*

We estimated the distribution of fishers to include 8,625 km<sup>2</sup> in Oregon (Fig. 3). The indigenous population was projected to occur across 5,566 km<sup>2</sup>, which was 74% of the range depicted by the US Fish and Wildlife Service. The largest discrepancies between our range estimates and previous range assessments occurred in the coastal segment of the Klamath Mountains, predominantly from within the perimeter of the Biscuit fire that burned in 2002. Two detection dog units contained fisher scats within the Biscuit Fire burn area, however, detections within the burn perimeter appear rare. Several scats were found in close proximity but outside the burn perimeter (Fig. 4).

The southern Cascade Mountains fishers were previously estimated to range across 7,063.5 km<sup>2</sup> (including 204 km of the indigenous east of Interstate 5) but we estimate a considerably smaller range of 3,059 km<sup>2</sup>, 43% of prior estimates. Areas in the far southern Cascades show a comparative increase but this is likely overestimated due to highly developed urban areas and non-forested areas contained within the kernel density estimate. The largest divergences occurred in the north and western portion of the prior estimated range (Fig. 3).

### *Spatial Extent Model*

In modelling the low growth extirpation and recolonization scenario, the simulated

introduced population expanded little and was unable to maintain high densities or reach estimated carrying capacity ( $K$ ) indicating that dispersal movements highly impacted our resulting local density (Fig. 5c,f). The remnant indigenous population was unable to colonize beyond the California border into Oregon and the two fisher populations remained isolated (Fig. 5c, f).

In our moderate growth extirpation and recolonization scenario, both populations expanded substantially. The indigenous population colonized most of the area south of the Rogue River and the introduced population reached the Interstate 5 corridor to the west. Connectivity was predicted between these populations near where Interstate 5 crosses the Oregon-California border (Fig. 5d, g).

The high growth extirpation and recolonization scenario simulated the indigenous population exceeding the Rogue River, the introduced population expanded beyond Interstate 5 and the populations completely merged (Fig. 5e). In modelling low growth wilderness refugia scenario, where fishers remained in Oregon but in low density wilderness refugia, fisher persisted at moderate densities within the remnant populations and at very low densities all the way to the Rogue River but were absent from areas near the Interstate 5 corridor (Fig. 5h). At both moderate and high growth rates in the wilderness refugia scenario the populations are highly integrated, maintain high densities, and expand into substantially more northerly areas than other simulations (Fig. 5d,g,e,h). Based on our field observations, it appears the reintroduced population is most similar to the lowest modeled growth (e.g., 5c, f) and the indigenous population is most similar to the extirpation and recolonization moderate growth scenario (e.g., Fig. 5d,g) but could plausibly be explained by several scenarios.

In the years after release population sizes varied under different growth and initial value

scenarios (Table 3). The modelled introduced population size was 48.7 females when  $r$  was low ( $r = 0.04$ ), 598.5 females under moderate growth ( $r = 0.14$ ), and 1378.9 females ( $r = 0.24$ ) at the highest expected growth rate estimate (Table 3). In the extirpation and recolonization scenario, the initial indigenous population size was 205.5 females and grew to 511.1 females with low growth ( $r = 0.04$ ), 1298.0 females with moderate growth ( $r = 0.14$ ), and 1932.5 females under high growth ( $r = 0.24$ , Table 3). In the wilderness refugia scenario, the initial indigenous population size was 214.0 females in 1974 and grew to 562.11 females with low growth ( $r = 0.04$ ), 1646.3 females with moderate growth ( $r = 0.14$ ), and 2404.1 females under high growth ( $r = 0.24$ , Table 3).

## Discussion

Fishers occur in fewer places than were previously believed and neither the indigenous nor the reintroduced fisher populations appear to have expanded or recovered portions of their range in Oregon beyond what was previously estimated by the US Fish and Wildlife Service (Fig. 3). In contrast, our estimates indicate range reductions for fishers of 26% for the indigenous population and 67% for the introduced population. Given the paucity of systematic range-wide monitoring for fishers in Oregon, these prior range depictions and subsequent discrepancies between range estimations should be treated with caution. Nonetheless, results shown here reduce the uncertainty surrounding fisher distributions and provide systematic, temporally succinct, comprehensive baseline information to assess the status of fisher distributions in the future.

The indigenous population could be considered relatively common where they occur but were largely absent from the coastal segment of the Klamath Mountains, specifically from within the perimeter of the Biscuit Fire (2002, Fig. 4c). In addition, no animals were detected north of

the Rogue River or northeast of Interstate 5 where suitable habitat is predicted to occur (Fig. 4c). The absence of detections north of these features coupled with predicted suitable habitat suggests they may be barriers to colonization and expansion of fisher populations. Pacific marten, a related species with a similar niche and behavior to fisher, were found to be unlikely to cross forest openings of 100m or greater (Moriarty et al. 2015). It is plausible that the combination of a large river alongside a four-lane highway creates a barrier that fishers are not willing to cross although capable of doing so.

The introduced population appears to have contracted, shifted south, or the previous population extent was incorrectly estimated. Fishers persist near some of the 1977 release sites, but appear to be absent from most of the Cascade Mountains (Fig. 4b). Our results suggest that fishers have had time to colonize well beyond the reintroduction area even under modest growth scenarios and have failed to do so. Given the number and spacing of detections in the Cascade Mountains, the population appears small and relatively isolated.

Fishers are commonly reintroduced animals and reintroductions of fishers have been generally considered successful in Eastern North America (77%) but not in Western North America (43%)(Lewis et al. 2012). This may be a vast overestimate as many reintroductions thought to be successful were, in fact, recolonization of natives without significant contribution of the reintroduced genes, or ambiguous contribution of the reintroduced genes (Stewart et al. 2017). We verified that the reintroduced fisher population in the Oregon Cascade Mountains persists but is lesser in extent than previously believed, and thus raises questions whether this should be considered a “successful reintroduction”. If the goal of reintroduction is beyond simply establishing a self-sustaining population, and is instead to recover historically occupied areas, then reintroduced individuals need to both propagate and expand.



The reaction-diffusion models suggest the introduced population has had adequate time to greatly expand their distribution but has generally failed to do so. Expansion within the models was uniform due to the underlying assumption of homogenous suitable habitat, however, the reintroduced population appears to have expanded or potentially just shifted south. Previous fisher habitat models suggest large tracts of suitable forest exist north of the reintroduction area (Fitzgerald et al. 2014). Under moderate or high growth scenarios the reaction-diffusion models show fishers expanding into these areas but our distributional surveys were unable to verify such expansions. The absence of detections within “suitable habitat” raises questions about what constitutes fisher habitat in the Central Cascade Mountains, and what the limiting factors are to this population. Population growth is mediated through habitat quality (Van Horne 1983, Johnson 2007), thus, the introduced population’s similarity to low growth is not a reflection of the maximum intrinsic growth rate but is instead suggestive of extremely low habitat quality, which may be suppressing population expansion.

The two reaction-diffusion scenarios for the indigenous population, extirpation and low density refugia, coupled with different growth rates show strong differences between the expected spatial extent for each projection. While it is challenging to determine historical conditions for fisher in Oregon due to limited documentation, the scenarios outlined here provide support for plausible descriptions relative to the contemporary and historic distribution of fishers. For instance, the absence of fishers north of the Rogue River where suitable conditions are modelled to occur could be due to time-limited expansion and not because this feature is a barrier to the movement of fisher populations. These modeling scenarios suggest that we should see a significant amount of fishers north of the Rogue under high growth conditions if fisher were extirpated, or with moderate and high growth levels if fisher persisted in low density refugia. The

observed contemporary extent of the indigenous population appears to be most similar to extirpation and recolonization under moderate growth (Fig. 5 D). The absence of detections north of the Rogue River, predicted absence of fishers south of the Rogue near the Oregon coast, and predicted early stages of connectivity between populations approximately where Interstate 5 crosses the California-Oregon border are all consistent with current field observations. If true this would suggest that fishers within the indigenous population were extirpated and have been expanding their population in areas of moderate to high habitat quality.

The absence of fisher detections from within the area of the Biscuit fire is of concern because species suffering from reduced ranges inherently have less capacity to buffer against habitat loss from environmental stochastic processes (Glynn and De Weerd 1991, Lande 1993). Wildfire has previously been identified as a threat to fisher habitat and conservation (USFWS 2014) but the relationship between fishers and fire is poorly understood. Nevertheless, habitat is likely rendered unsuitable for fishers when stand-replacing fire removes canopy cover at large spatial scales and reduces the prevalence of structural elements required for rest and den sites (Weir and Harestad 1997, Weir and Corbould 2010, Aubry et al. 2013). Characterizing the typical dynamics of the Klamath-Siskiyou mixed-severity fire regime has been difficult due to complex interactions between weather, fuels, and topography that produce intricate spatial and temporal variation in patterns of burn severity within and between fires (Agee 2005, Halofsky et al. 2011). The Biscuit Fire, however, appears to have been unusually large and severe for the Klamath-Siskiyou region based on estimates of crown damage (Odion et al. 2004) and area affected by surface fire (Campbell et al. 2007, Thompson and Spies 2009). In total, the Biscuit fire burned over 2,020 km<sup>2</sup>, of which 1,861 km<sup>2</sup> were within Oregon (Azuma et al. 2004), representing ~25% of the indigenous fisher population range in Oregon.

Further exemplifying the potential scale of this stressor, in 2017 the Chetco Bar fire burned another 773 km<sup>2</sup> within the range of indigenous fisher, although large portions were within the Biscuit scar. According to our estimated range, the 2017 wildfire season in southern Oregon burned 10 % of the indigenous range and 3 % of the introduced range (Fig. 4). We detected fishers at 7 sample units during the spring and early summer that were burned by wildfire later that year (Fig. 4). The absence of fisher detections from within the Biscuit Fire and the burning of a substantial additional component of the range of fisher in Oregon during the 2017 fire season indicates that fire poses a potential danger to the stability and recovery of fisher populations in Oregon. A fire of similar size and severity to the Biscuit could affect much of the remaining habitat available to fishers in either population (Fig. 4c).

There are several limitations to our surveys and analyses. The study design deployed for these surveys and the analysis presented herein was intended to detect populations across large spatial scales, and not designed to detect every individual within a population. Thus, we anticipate there are likely individuals outside the distribution depicted, but it is unlikely that an additional population exists. While the insights provided by the reaction-diffusion models illuminate the potential history and patterns of fisher expansion in this area, they should be treated primarily as a thought experiment. The underlying assumptions used to parameterize the model and those used to inform the initial conditions of the indigenous population contain large amounts of uncertainty. Additionally, the assumption of static homogenous suitable habitat is unrealistic. Oregon has a long complicated history with natural disturbance and anthropogenic factors that have influenced forest quality and quantity through time (Harris et al. 1982, Ripple et al. 2000, Aubry and Lewis 2003, Kennedy and Spies 2004;2005). Events like the Biscuit fire would likely have slowed expansion and reduced carrying capacity locally which is unaccounted

for within our models. The reaction-diffusion models are best suited to bracket the uncertainty surrounding fisher distributions and narrow the scope of possible areas we would expect to find fishers. They can also be used to generate hypotheses regarding inconsistencies between model results and observed distributions to be addressed in future studies. Exemplary of this potential application is the absence of fishers north of the reintroduction areas. Reaction-diffusion methods suggest even with modest growth scenarios and suitable habitat that fishers would occupy this area, but the absence of fishers here stimulates subsequent diagnostic hypotheses.

### *Conclusion*

The indigenous population of fishers appears to occupy substantially less area than previously believed and remains south of the Rogue River. This could be due to time-limited expansion if fishers were historically extirpated, or possibly due to fire disturbance. This population remains susceptible to high severity fires, which is highlighted by fires that occurred during our study. The introduced population has not expanded and appears to be highly reduced compared to previous estimates. Other Pacific fisher populations suffer from high rates of predation, especially predation by bobcats (Wengert et al. 2014), and have been subject to direct or secondary poisoning via the use of rodenticides and toxicants (Gabriel et al. 2015). The influence of the rodenticide use and predation rates on fishers in Oregon is unknown, but one indigenous female fisher has been poisoned (D. Clayton, personal communication). This study was not intended to address potential limiting factors, but allows a platform for future studies to be strategic – for instance, demographic data can be collected on fishers in areas with predicted low and high densities or stratified by conspecific species occurrence (e.g., bobcats). Based on our simulations, we predict higher growth capacity (high survival and fecundity) in the indigenous population and extremely low growth capacity (low survival, fecundity, or both) in

the introduced population.

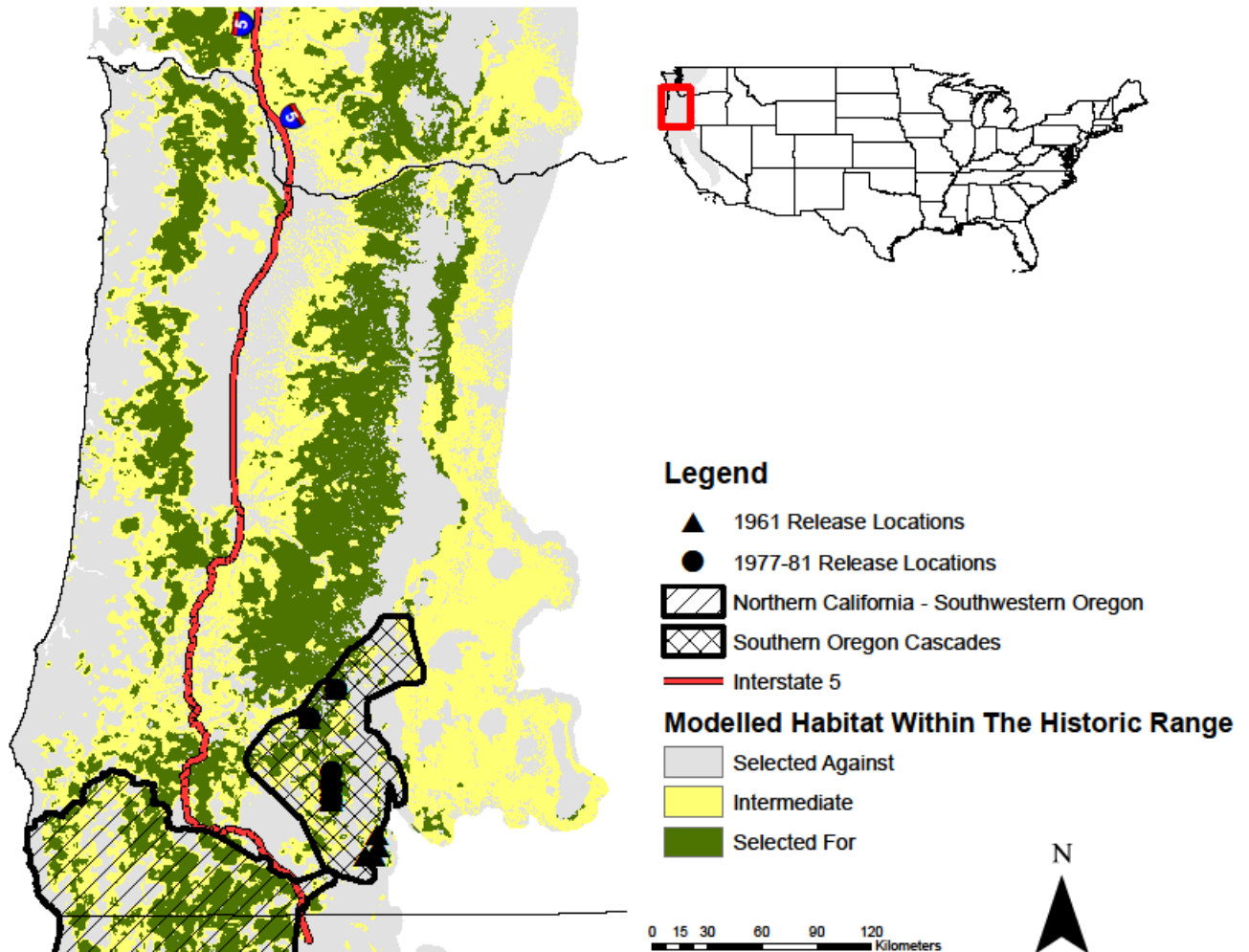


Figure 1. Historic range of fisher and contemporary habitat model from Fitzgerald et al. (2014). Areas in green indicate habitat selected by fisher, areas in yellow indicate intermediate selection, and areas in gray indicate areas within the historic range but suspected to be selected against. The estimated distribution of indigenous fisher in Oregon prior to this work are single hatches, and the estimated range of the introduced population is double hatched.



Figure 2. We used olfactory lure (Gusto, Minnesota Trapline©) paired with one of three randomly selected bait types (see inset). Our baits included chicken (A), fish-flavored cat food (B), or the kitchen sink - chicken, cat food, an apple, peanut butter and oats (C). We placed a measuring strip (1-m) marked in 10cm increments to measure animal and a signboard depicting the station identity. Unbaited trail cameras (D) were offset from a baited station along drivable roads, old roads, or game trails.

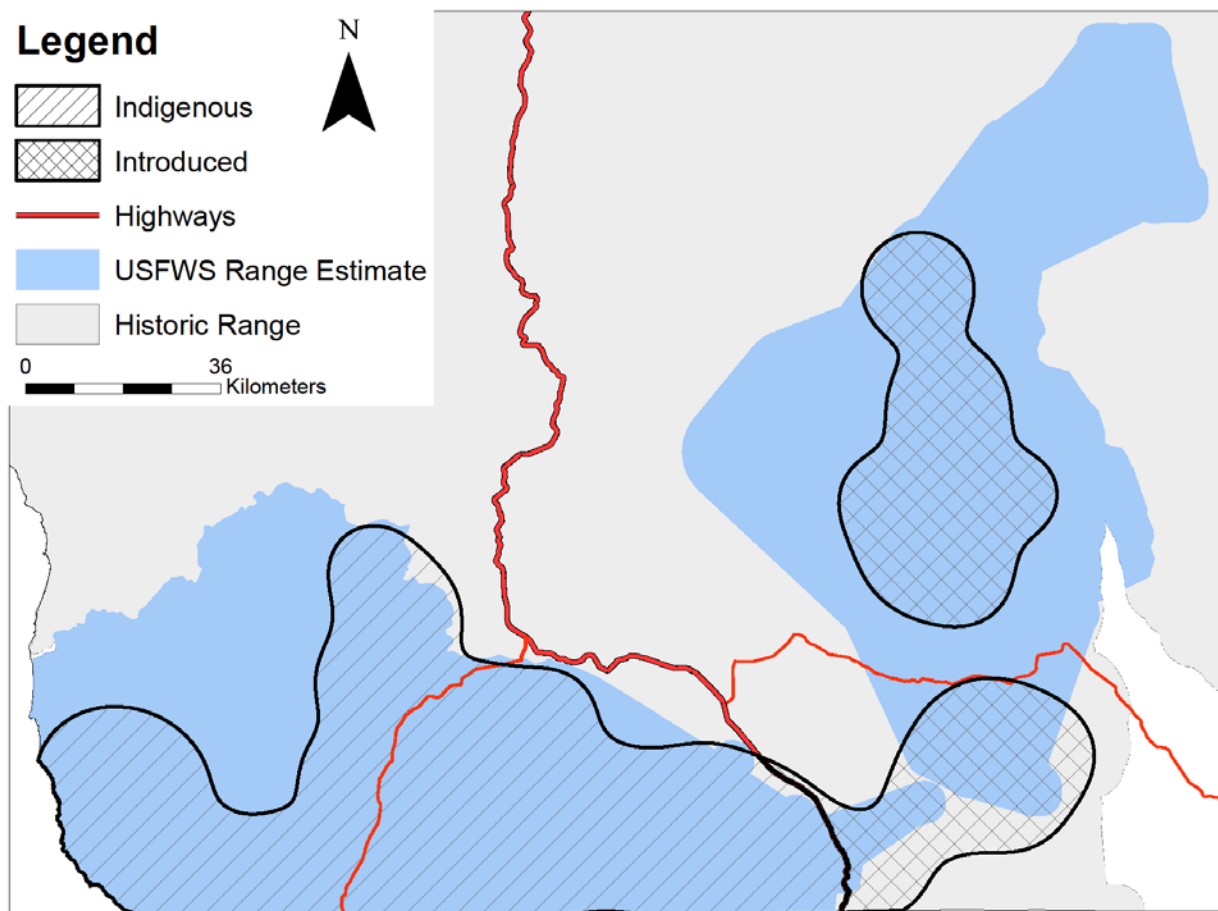


Figure 3. Range estimates of fisher in Oregon estimated using a kernel density analysis of detections using a 6km bandwidth and 95% isopleth. The estimated range by the US Fish and Wildlife Service (2016) is in blue.



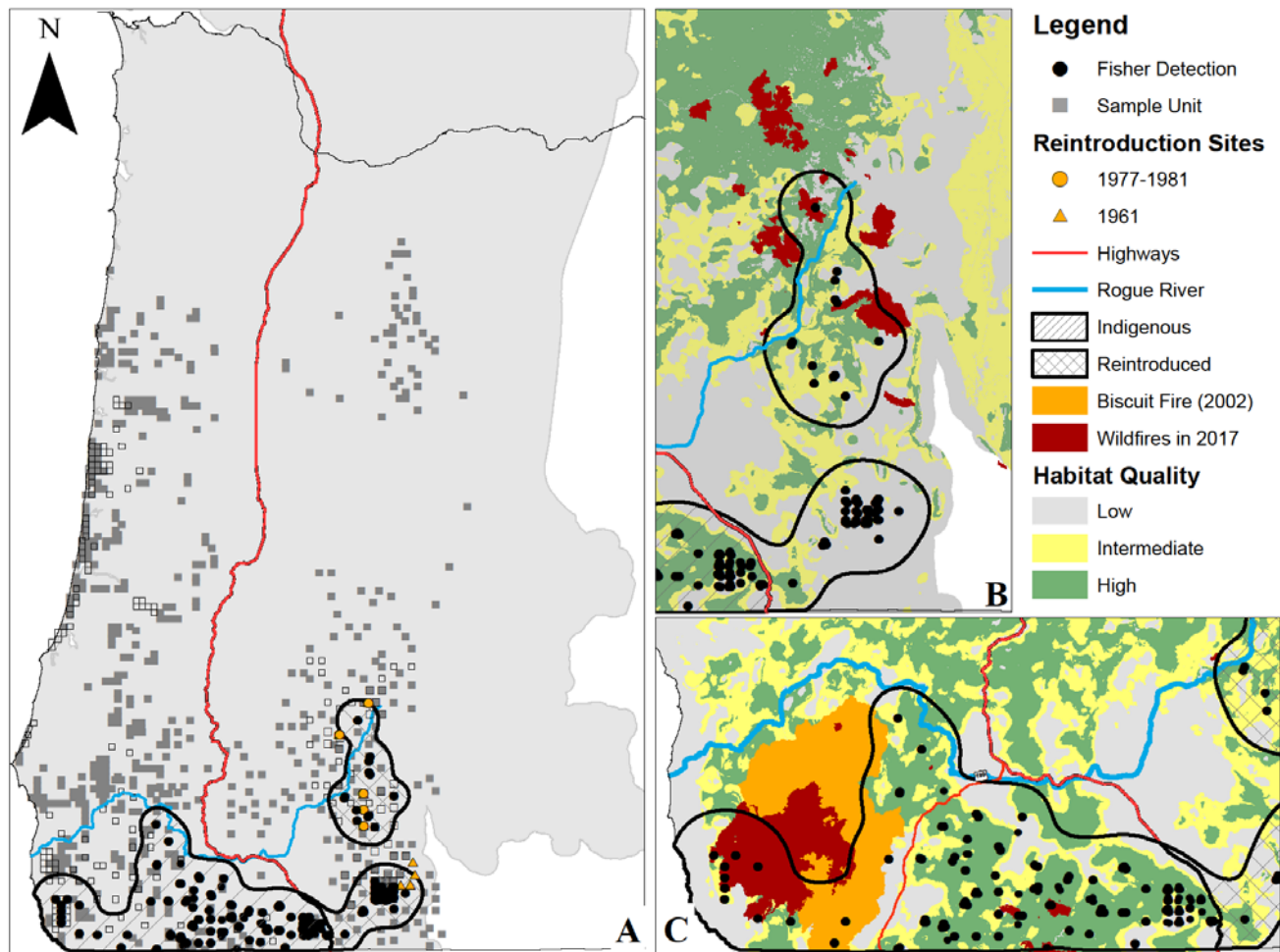


Figure 4. Results from combined survey efforts. Sample units are dark grey squares consisting of 4 camera traps, detection dog surveys are open boxes, and fisher detections are black dots. The shaded area depicts the Pacific fisher Distinct Population Segment analysis area. The crosshatching represents the estimated introduced population distribution and the hatch represents the estimated range of the indigenous fisher distribution. Tight clusters of detections represent areas of known fisher occupancy that were intensively sampled for methodological reasons outside the scope of this research (but see Supplemental Information). Inset (A) Fisher detections and survey effort within the Oregon portion of the distinct population segment. Fisher detections with modeled fisher habitat, the Biscuit Fire, and fires during the 2017 survey, for the introduced population (B), and indigenous population (C)

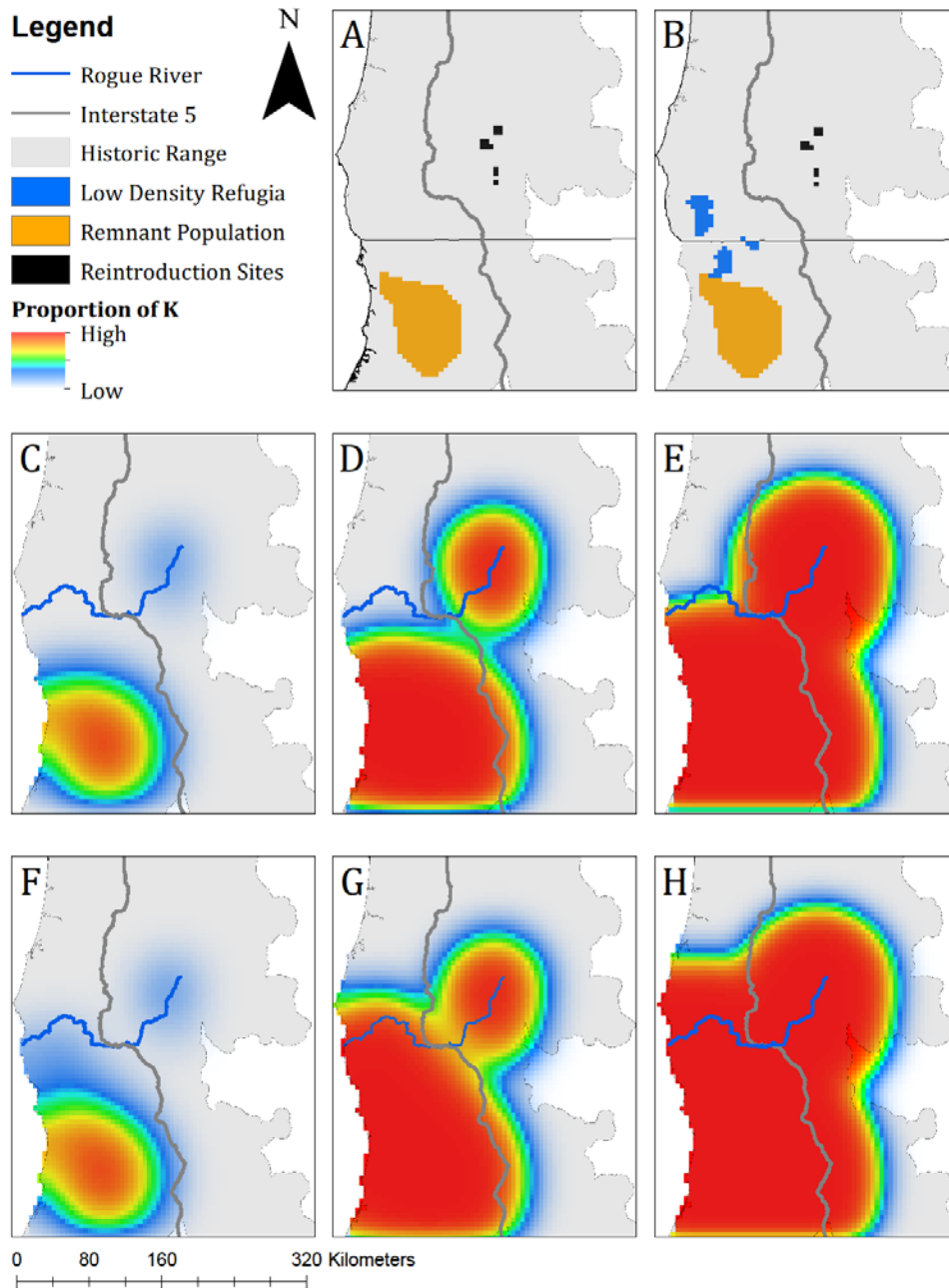


Figure 5. Estimated expansion of the Southern Oregon Cascade Mountains Population 36 years after reintroduction and the Indigenous Populations 41 years after extirpation using a reaction-diffusion model (females only). The indigenous population was modelled with two scenarios; complete extirpation from Oregon and recolonization from a remnant population in northern California (A), and low density refugia (B). The model used three levels of estimated  $r$  for fishers, low, moderate, and high values ( $r$ ) in the scenario with complete extirpation (C,D,E for  $r = 0.04, 0.14$ , and  $0.24$  respectively) and wilderness refugia (F,G,H, for  $r = 0.04, 0.14$ , and  $0.24$  respectively) based on different levels of adult female survival. The maximum expansion of each population in isolation for Indigenous (D) and Reintroduced Populations (E).

Table 1. We used camera traps and detection dog teams to document fisher distribution in Oregon. We used several baits: kitchen sink (K), cat food (F), chicken (C), chicken-cat food (CF), and lures: gusto (G), megamusk (M), and no lure used (N).

Project	Sample Units	Stations	Year	Start Date	Deployment Days <i>x</i> (SD)	Bait	Lure	Photographs	Fisher Photos
Mt. Hood	32	122	2015-2016	Oct.-June	221 (58)	K, F, C	G	996,528	0
SWOR	15	66	2015	Aug.-Sep.	124 (90)	K, F, C	M, G	179,640	5,374
SWOR	78	297	2016	Jan.-March	43 (45)	K, F, C	G	698,442	813
SWOR	105	406	2016	May-Oct.	61 (31)	K, F, C	G	2,243,619	3,715
SWOR	60	224	2017	May-Oct.	62 (15)	K	G, N	376,010	1,157
CMD	101	381	2015	May-Oct.	16 (6)	CF	G	216,843	803
OFIC	200	359	2015	Jan.-Oct	22 (4)	C,F	G	68,096	761

Table 2. Input values for estimation of  $r$  and spatial extent model with pertinent justifications.

Variable	Modelled value	Justification
Age of first parturition	2	Powell and Zielinski (1994), Powell et al. (2003)
Average number of female kits/year ( $m$ )	1.0	Aubry and Raley (2006), Mathews et al. (2013b)
Survivorship to age at first parturition ( $la$ )	0.42	
Kit survival (age 0-1)	0.6	Koen et al. (2007), York (1996), Sweitzer et al. (2015)
Yearling survival (age 1-2)	0.7	York (1996), Krohn et al. (1994), Sweitzer et al (2015)
Range of adult survival (age 2+)	0.64, 0.79, 0.94	Higely and Matthews (2009) Sweitzer et al. (2015)
Females/km <sup>2</sup> ( $K$ )	0.04	Aubry and Raley (2006), Furnas et al. 2017, Sweitzer et al. (2015)
Female Dispersal Distance	6.0km	Aubry and Raley (2006), Mathews et al. (2013b), Sweitzer et al. (2015)

Table 3. Estimated population size of females for the introduced population, and the indigenous population under two growth scenarios extirpation and recolonization, and wilderness refugia.

<b>Scenario</b>	<b>Growth</b>	<b>r value</b>	<b>Population Size</b>
Introduced Population	Initial	-	12
	Low	0.04	48.7
	Moderate	0.14	598.5
	High	0.24	1378.9
Extirpation and Recolonization	Initial	-	205.5
	Low	0.04	511.11
	Moderate	0.14	1298.0
	High	0.24	1932.5
Wilderness Refugia	Initial	-	214.0
	Low	0.04	562.11
	Moderate	0.14	1646.3
	High	0.24	2404.1

\*See Supplemental Information for details on how population size varies with different  $D$ ,  $K$ , and  $\theta$ .

**CHAPTER 3**  
General Conclusion and  
Future Research

Brent R. Barry

The Endangered Species Act (ESA) requires the decision to list or delist a species be based on five broadly defined threat categories: habitat loss, overutilization, disease or predation, inadequate regulatory mechanisms, or any other reason (ESA sec. 4(a)(1)(A)-(E)). Paramount to achieve ESA success, therefore, is eliminating the threat(s) that led to a species' imperilment. Regional threats to Pacific fisher implicated in their decline (overharvest, habitat loss, and non-compensatory mortality) are viewed as non-operative at this time (USFWS 2016), yet, we have concluded from an extensive survey effort that the range of fishers in Oregon is similar to or potentially smaller than previously estimated and fishers have failed to colonize large portions of the Cascades. The contemporary distribution of the indigenous fisher population can plausibly be explained via time-limited expansion (i.e. unoccupied parts of the coastal segment of the Klamath Mountains and north of the Rogue River), but, the reintroduced population resides near some of the largest contiguous blocks of habitat on the west coast (Fitzgerald et al. 2014) and time-limited expansion does not inhibit access to these areas (Fig. 5). The lack of population expansion demonstrated by fishers may be indicative of one or several unidentified threats acting to suppress recovery, or suggests that aspects of forest conditions within these areas of the Cascades are insufficient to facilitate fisher occupancy.

Most habitat suitability models correlate structural elements (e.g. forest age, canopy structure) and abiotic factors with patterns of observed spatial occupancy to infer habitat quality. Reliance on structural elements alone does not account for community-level species interactions such as prey availability, predation risk, and intraguild competition. Such direct measures of top-down and bottom-up forces are important drivers of habitat quality but have been difficult to quantify at large spatial scales (Bean et al. 2014). The community-level interactions that directly influence habitat quality may be a key factor in determining the occupancy dynamics of rare and

endangered taxa in the Pacific Northwest.

It has become widely accepted that fisher populations in the western U.S. suffer from high rates of interspecific predation and this has been suggested as potentially limiting the range of fisher (Lofroth et al. 2010, Gabriel et al. 2015, Sweitzer et al. 2015). Results from a recent study in California found that 76% of female deaths were attributed to interspecific killing, most predated by bobcats (*Lynx rufus*) (Gabriel et al. 2015) and adult female survival has been shown to be the most important demographic parameter to the stability of fisher populations (Spencer et al. 2011, Sweitzer et al. 2015). These findings suggest top-down community effects are key factors in fisher conservation and population dynamics (Wengert 2013). Further support for the potential effect of predation on fisher population dynamics comes from a habitat suitability model, which demonstrated that using bobcat occupancy as a modelling covariate dramatically reduces the amount of suitable habitat (Halsey et al. 2015).

Research conducted on the reintroduced population has shown that adult female survival was relatively high, and that observed demographic parameters at that time indicated a relatively healthy population (Aubry and Raley 2006). The average annual survival rate of fishers >1 year of age was 82% (85% for males, 78% for females) and predation was not the leading cause of death of females (Aubry and Raley 2006). On average 59.4% of adult females gave birth to kits each year; however, the average annual reproductive success rate was only 44% (defined as raising at least 1 kit to be >2 months of age)(Aubry and Raley 2006). Survival rates of radio-collared juveniles appeared to be high and dispersal was male-biased with females more likely to establish home ranges in proximity to their natal areas (Aubry and Raley 2006). Nonetheless, despite these indications of a relatively healthy population, it appeared and continues to appear, that the population in the Cascades is relatively isolated to the reintroduction area. The observed



dissonance between reasonable demographic parameters and distributional surveys indicate that fisher in the Oregon Cascades might be dispersal limited due to habitat quality. Of paramount interest is kit/early juvenile survival (>2 months but < 7 months of age) and the fate of dispersing females. Researchers in the Cascades noted “It was not possible for us to reliably determine survival of kits beyond this time (>2 months) because the kits became more mobile and difficult to observe” (Aubry and Raley 2006). Therefore, it is possible that from the early stages of kit mobility (2 months) to the development of a larger body size and establishment of a home range (>7 months), fishers are heavily susceptible to predation or have difficulty accessing suitable habitat to disperse into.

Mustelids like fisher have a high basal metabolic rate, suggesting that prey availability may also be an important determinant of suitable habitat (Powell 1979). Several lines of evidence support this claim but it’s difficult to link prey abundance with fisher population growth. Fishers populations in the northern and eastern portion of their range appear to experience a population response consistent with changes in prey abundance (Bowman et al. 2006, Jensen et al. 2012). When fishers were reintroduced to the Olympic Peninsula, WA they predominantly established home ranges in areas suspected to contain higher prey availability (Lewis et al. 2016). Collectively, these findings indicate community-level interactions play a role in limiting fisher populations (both top-down and bottom-up) but there have been few attempts to quantitatively link these processes at the landscape scale.

The recent conservation strategy for fishers in the Pacific has emphasized translocation to establish new populations and expedite distributional recovery (Hayes and Lewis 2006, Lewis 2014, Halsey et al. 2015, Hiller 2015, Facka 2016). Translocation (capture, transport, and release of individuals) is a common approach for imperiled species (Seddon 1999, Hayward and Somers

2009) because it provides an option for immediate gains in distribution while avoiding the socioeconomic costs of restoring connectivity in landscapes plagued by current and residual legacies of human land use. Translocation also provides researchers with a unique arena to test ecological theories and the application of management strategies (Facka 2016). Nonetheless, as evidenced by the reintroduced population in the Cascades, a translocation based strategy may not create self-sustaining populations in the long-term unless “suitable habitat” is capable of being identified and present in sufficient quantities to maintain species viability in perpetuity. Further, the failure to monitor and research reintroduced populations over long time scales, such as the case with fishers in the Cascades, diminishes the capacity to identify demographic and ecological fulcra that support or impede population persistence. The uncertainty generated through deficient monitoring has cascading effects down legal, regulatory, and policy pathways when reintroduced populations of imperiled species fail and the species in question continues to decline. While the regional threats implicated in historic fisher declines are viewed as non-operative (USFWS 2016), insights from the oldest reintroduced population within the Pacific suggest underlying specters inhibiting the recovery of fishers warrant new examinations of habitat, predation, and prey availability.

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## **Supplemental Information**

### Methods

### *PCR*

We performed polymerase chain reactions (PCR) in replicates of three per scat with Qiagen Multiplex PCR Kit (Qiagen, USA). Each reaction was amplified with a unique 8 base pair tag on the 5' end of the forward and reverse primer to identify individual scats after pooling. PCR reactions were carried out in a total volume of 20  $\mu$ L using the following reagent mixtures: 10  $\mu$ L Qiagen Multiplex PCR Master Mix, 0.4  $\mu$ L of each primer for a final primer concentration of 200 nM, 0.2  $\mu$ L bovine serum albumin (BSA), 6.2  $\mu$ L of water, and 2  $\mu$ L final DNA extract elution (including extraction controls and a PCR blank). Following a 15 min, initial denaturation at 95°C, the cycling conditions were: 40 cycles of 94°C for 30 seconds, 58°C for 90 seconds, 72°C for 90 seconds, and a final extension at 72°C for 10 min. We normalized and pooled the PCR products and used NEBNext Ultra II Library Prep Kit (New England BioLabs) to adapt the library pools into Illumina sequencing libraries (Illumina Inc, San Diego, USA). Libraries were purified using the Silica Bead method from Aline Biosciences, USA. Finally, we sent the libraries for sequencing by the Center for Genome Research and Biocomputing at Oregon State University on the Illumina HiSeq 3000 platform. The resulting sequence reads were filtered, annotated, paired and demultiplexed by CGRB. The sequences were compared with published sequences using the NCBI Nucleotide BLAST.

### *Spatial Extent Model Parameters:*

*Parameter Estimation Maximum Intrinsic Growth Rate (r):* The maximum intrinsic growth rate,  $r$ , was estimated using a modified Euler-Lotka equation (Skalski et al. 2008),

$$e^{ra} - e^{-M}(e^r)^{a-1} - ml_a = 0$$

where  $r$  is the maximum intrinsic growth rate,  $a$  is the age at first birth,  $m$  is the fecundity constant (number of female offspring/female/year),  $e^{-M}$  is the probability of survival, and  $l_a$  is the probability of survival to maturity. We obtained a range of parameter estimates associated with the maximum reproductive output of fishers in wild populations from the literature (Table 2). Adult annual female survival has been shown to be the most important demographic parameter to the stability of fisher populations (Spencer et al. 2011), and estimates of  $r$  are subject to variability due to fluctuations in annual survival,  $e^{-M}$ ; thus, we estimated maximum intrinsic growth rate ( $r$ ) assuming low, average, and high survival rates reported for female fishers (Table 2).

*Parameter Estimation Diffusivity (D):* To estimate diffusivity  $D$ , we assumed dispersal was the primary mechanism for geographic expansion of a fisher population (Aubry and Raley 2006, Mathews et al. 2013). Female fishers tend to be philopatric, dispersing shorter distances, but long distance dispersal (>50 km) has been documented for both sexes (Arthur et al. 1993, Mathews et al. 2013, York 1996) and average dispersal distances vary by region, sex, and study. Thus, we used female dispersal estimates derived the reintroduced population to estimate  $D$ .

In estimating  $D$ , we assumed there were no native fishers in the landscape, and released  $N=12$  fishers at year  $t=0$  into a  $25\text{km}^2$  cell. During the next year ( $t=1$ ), the model would be reflect the number of juvenile females that dispersed distance  $X_i$ . We let the mean displacement

$m = \frac{1}{N} \sum_{i=1}^N X_i(t)$ , where  $X_i(t)$  was the position of the  $i$ -th fisher. Diffusion was depicted as the

mean square displacement, which is represented by  $msd = \frac{1}{N} \sum_{i=1}^N (X_i(t) - m)^2$ . Finally, the

diffusion coefficient expressed in two dimensions (x,y), in time step ( $t=1$ ) was,  $D = \frac{msd}{4}$ . Of

the four female fisher dispersal movements that were observed by Aubry and Raley (2006), two did not disperse from their natal areas, one dispersed 17km, and the fourth dispersed ~7km.

These dispersal estimates scaled to a 5x5 cell results in a  $D$  estimate of 0.485 cells/yr.

*Sensitivity Analysis:* We performed a sensitivity analysis to determine the effect of a parameter's value on the modelled spatial extent and population size. We held  $r$  constant at 0.24, our *a priori* hypothesized most likely value, and used the extirpation and recolonization scenario as initial conditions. Diffusivity ( $D$ ), carrying capacity ( $K$ ), and strength of the density dependence ( $\theta$ ) were varied by  $\pm 50\%$ . Spatial extent was effect most effected by the diffusivity parameter,  $D$ , and was negligibly effected by changes to  $K$  or  $\theta$  (Fig. 6). Population size was most influenced by  $K$  and least influenced by  $\theta$  (Table 4).

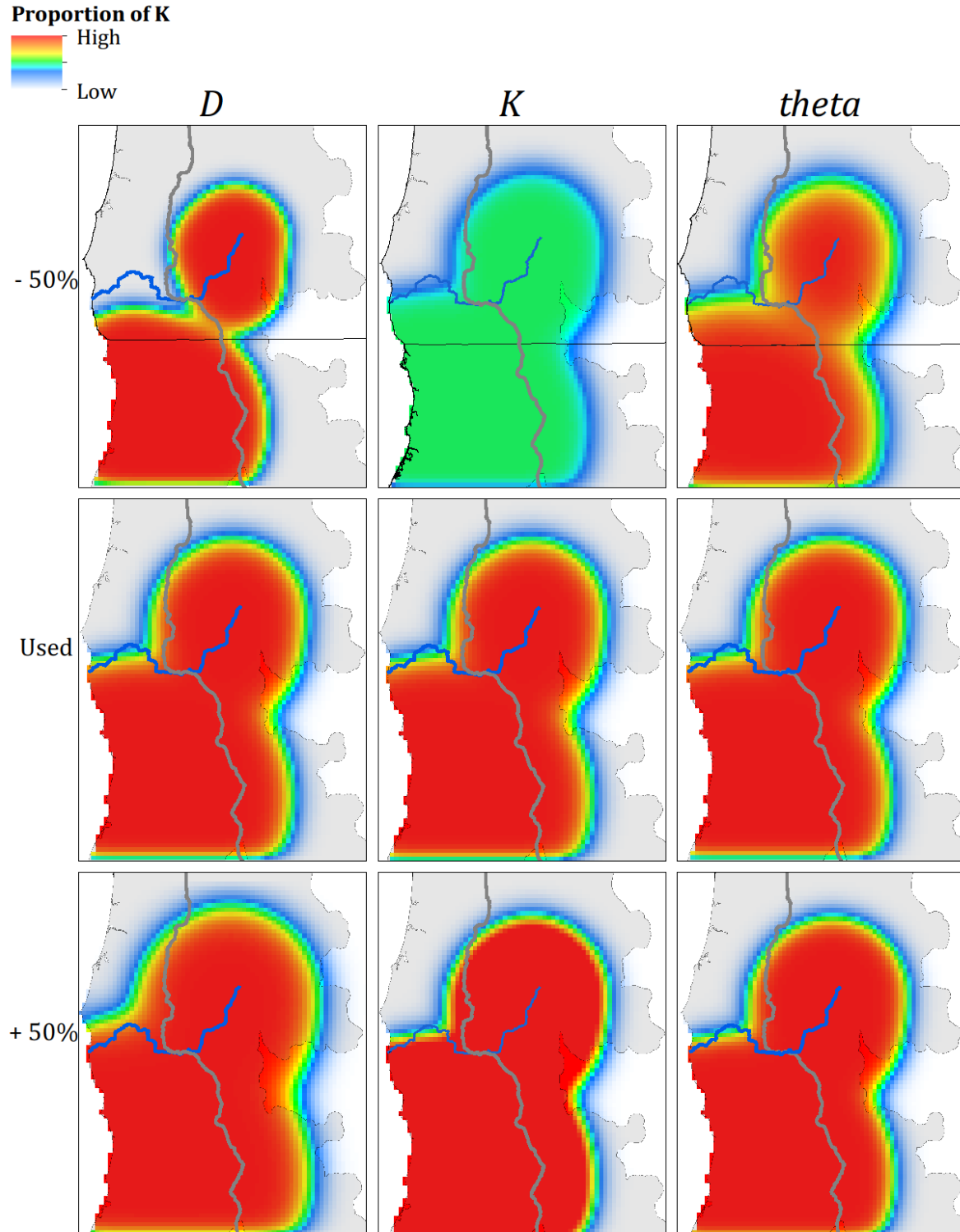


Figure 6. Sensitivity of input parameters for the spatial extent model. Inputs to the model diffusivity ( $D$ ), carrying capacity ( $K$ ), and strength of the density dependence ( $theta$  or  $\theta$ ) were varied by 50% from the used values.

Table 4. Sensitivity of population size to the parameter ( $\pm 50\%$ ) with corresponding percent change in population size.

<i>Parameter</i>	-50%	+50%
<i>D</i>	-25%	+18%
<i>K</i>	-48%	+46%
<i><math>\theta</math></i>	-14%	+4%



## **Supplemental Information**

### Genetic Sampling and Results

## Methods

If a camera station detected a fisher, a hair-snare device was immediately deployed in addition to the camera trap. Hair-snares were checked and re-baited at least once a week for a minimum of three weeks with the same bait type used at the camera station. Hair-snares devices consisted of three .50 caliber gun brushes inside an 81 x 25 cm triangular shaped cubby placed at the base of the bait tree or 20 x 46cm pvc tube attached to the bait tree, capped on one end and the open side facing down. The hair-snare device was placed such that the entrance was visible to the camera trap. If gun brushes contained hair samples they were placed in a labeled vial with desiccant and stored in a cool dry area.

Fisher hair samples were sent to the National Genomics Center at the United States Forest Service Rocky Mountain Research Station for genetic analysis. Samples were analyzed for population structure using a 300bp region of the mitochondrial DNA control region (Drew et al. 2003, Vinkey et al. 2006, Schwartz 2007) and genotyped using 16 mustelid loci: *Mp0059*, *Mp0144*, *Mp0175*, *Mp0197*, *Mp0200*, *Mp0247* (Jordan et al. 2007), *Ma1*, *Gg25* (Davis and Strobeck 1998), *Mer022*, *Mvis002*, *Mvis072* (Fleming et al. 1999), *Ggu101*, *Ggu216* (Duffy et al. 1998), *Lut604*, *Lut733* (Dallas and Piertney 1998), *Mvi1321* (Vincent et al. 2003) *Mfl.18* (Basto et al. 2010) and a mustelid sexing marker (Hedmark et al. 2004).

## Results

We obtained a total 41 hair samples believed to be from fishers, 16 samples from surveys in 2016 within the hybridization zone of the Cascade Mountains and 25 from surveys in 2017 in the Klamath-Siskiyou Mountains. We were unsuccessful in obtaining samples from all areas where fishers were detected, notably near the reintroduction areas, the coast, and from comparatively few sites within the indigenous population area. Haplotypes were obtained for 15

of the 2016 samples and 18 of the 2017 samples. Three haplotypes were observed (Drew-Hap1, Drew-Hap2 and Drew-Hap9). Drew-Hap1 and Drew-Hap2 are two haplotypes observed previously in fishers from the indigenous population (Drew et al. 2003). Drew-Hap1 was observed only in the Klamath-Siskiyou Mountains. Drew-Hap2 was observed in the Klamath-Siskiyou Mountains and from one individual in the Cascade Mountains. Drew-Hap9 has been detected in fisher populations introduced to southwestern Oregon (Drew et al. 2003) and was observed only in the Cascade Mountains.

We obtained genotypes and sex identification for 32 of the samples, 14 from 2016 and 18 from 2017. In 2016, six individuals were identified in the Cascade Mountains (four females and two males) and were all recaptures of fishers previously identified from the area. In 2017 we obtained genotypes and sex identification for 18 of the fisher samples. Six individuals were identified (four females and two males) from the Klamath-Siskiyou Mountains and represent new individuals to the DNA database of fisher in the region.

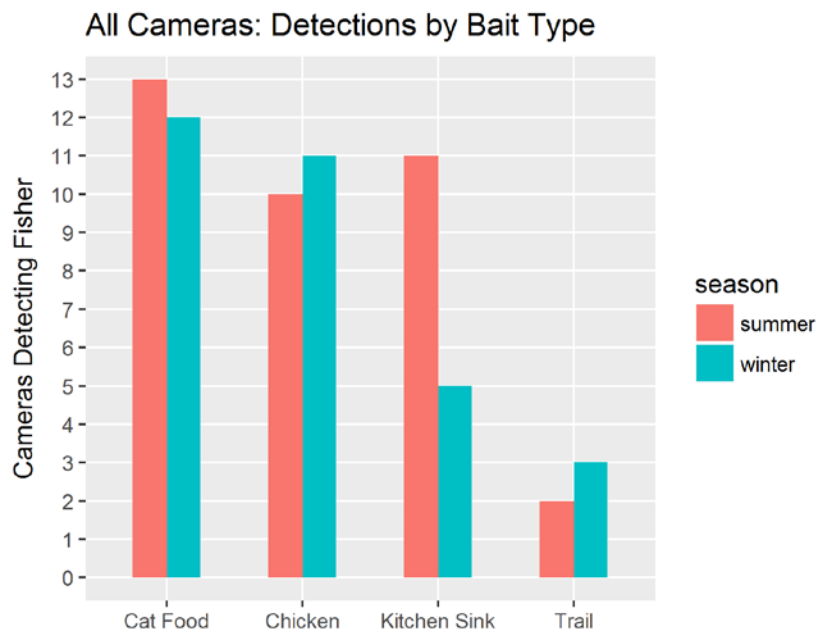
## **Supplemental Information**

### Fisher Detectability

To assess fisher detectability in regards to bait type and season, we surveyed some units both during winter (Oct.1-May 31) and summer (June 1-September 30), hence forth “paired” and compared these to all camera sets. These results were entered in a binomial and Poisson generalized linear model for number of cameras detecting fisher and latency to first detection, respectively. For all cameras, fishers were detected significantly less frequently at stations sampled in the summer than winter ( $p=0.019$ ) and less at trail cameras than baited stations ( $p<0.001$ , Figure 7). The frequency of cameras detecting fisher did not vary significantly between baited stations. Latency to first detection was significantly different for trail cameras ( $p<0.001$ ) and kitchen sink ( $p<0.001$ ) than chicken or cat food, and was shorter in winter ( $p=0.002$ ). At baited stations, median latency to first detection ranged between 23.7- 49.8 days during summer and 17.8-20.0 days during winter (Figure 7b). Of the paired stations, we detected fishers significantly less frequently at trail cameras than at baited stations ( $p=0.0066$ ) but there was no significant difference between bait type or season (Fig. 8). Latency to first detection was variable, but lowest with chicken and cat food during both summer and winter survey periods (Figure 8b). Baited camera stations were more effective than non-baited trail sets at detecting fishers (Figure 7a, 7b), and we aim to determine whether this is the case for all species (e.g., bobcat, lion, coyote). Fishers were detected more often and quicker during winter (Figure 8a, 7b), but such detections could include dispersing juveniles and not represent year-round populations. We will analyze the spatial distribution of detections between seasons. Meanwhile, our data suggest that most fishers will be detected at baited stations within 25 days during winter and 50 days during summer (Figure 7b). Currently, it appears that all bait types are similarly effective at detecting fishers, with fewer detections at kitchen sink types during winter.



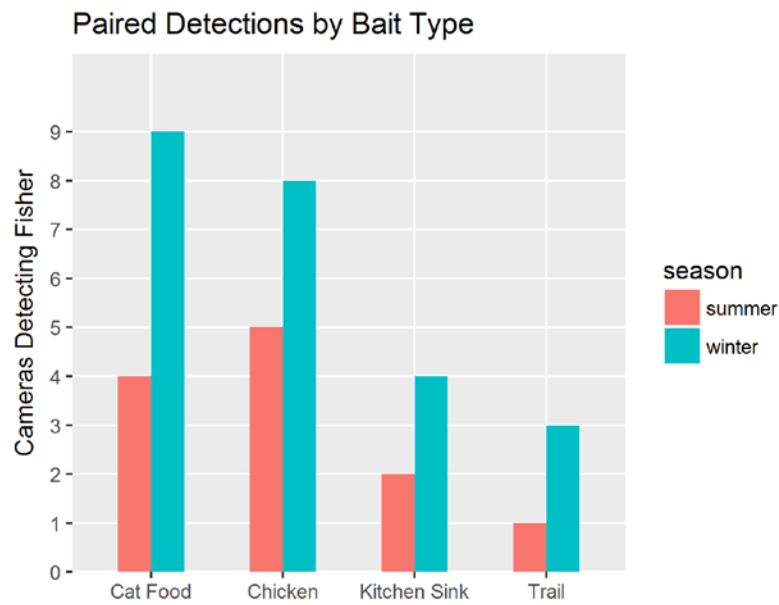
A



B

Figure 7. We detected fisher at 36 of 261, and 31 of 134 camera stations during summer and winter, respectively (A). We report the first detection to describe survey durations, latency to first detection, (B).

A



B



Figure 8. We compare frequency of camera stations and latency of first detection, or the number of days surveyed to the first detection (boxplots), at sample units surveyed in both winter and summer 2016 ( $n = 43$  camera stations). Fishers were detected at 8 and 20 camera stations in summer and winter, respectively

## **Supplemental Information**

Varying remote camera methodology to assess Pacific fisher detectability in a multispecies framework

Brent Barry, Alexa Myers, Katie Moriarty, and Taal Levi



### *Introduction:*

Camera surveys have become an increasingly popular non-invasive technique in wildlife monitoring, particularly to determine presence/absence and species distributions. Remote photography allows the research of biota that are otherwise difficult to detect. Nonetheless, camera survey methodologies have yet to be standardized. Variation in methods, such as camera distance to a focal location, may vary for optimally detecting species of different sizes.

Fisher (*Pekania pennanti*) distributions have contracted (Lewis et al. 2012) and few populations remain on the West Coast (USFWS 2016). Detecting and monitoring low-density cryptic species, like fishers, is challenging (Linden et al. 2017). Most surveys do not test multiple methods. We uniquely surveyed for fisher in an area with over 8 years of telemetry monitoring, which confirmed the presence of more than 5 reproductive female fishers in our study area. This level of survey intensity builds confidence for optimizing survey efficiency for managers. Without confidence in monitoring, managers have limited ability to describe the landscape and evaluate potential effects of management.

Species interactions, such as predator and prey interactions, can influence species distributions (Sala and Graham 2002). A multispecies approach with a standardized protocol could allow for efficient monitoring of the vertebrate community, including a focal species of interest: fisher.

### *Objectives:*

The objects of this study were to (1) describe detectability of the mammalian community using different combinations of baits and lures, and (2) assess the distance in which camera sets

maximized fisher detectability, potential prey (small-bodied), and predators (large-bodied animals).

### *Study Design and Methods*

We assessed fisher detectability in the Ashland Watershed, Oregon. We deployed remote cameras in areas of known fisher occupancy (2015, n=66 cameras). 15 sample units were stratified by access (<1km) with a minimum spacing of 3-km. Sample units consisted of 4 remote camera stations: 3 baited & 1 unbaited trail camera. The baited camera stations had 1 of 3 treatments; a can of wet cat food, 2 chicken legs, or “kitchen sink” – a mixture of peanut butter, oats, cat food, chicken, and half of an apple. The baited stations had 1 of 2 types of olfactory lures along with the bait: Megamusk and Gusto (Minnesota Trapline Company ©). A visual lure, a spinning aluminum pie pan, was or was not present. Cameras were set at a distance of ~2m or ~4m from the camera to the bait, labeled “near” and “far” respectively. The distance from the bait tree to the camera was measured in centimeters. Cameras were checked to replace bait, batteries, and SD cards after 7-10 days and then left unchecked for 90 days depending on logistical constraints.

Species that were detected at  $\geq 5$  cameras were excluded from analysis due to small sample size. Bait and lure effectiveness were analyzed using logistic regression per species. Distance from bait to camera and species detection histories were analyzed using logistic regression.

### ***Results***

Cameras operated for a total of 11,489 trap nights. 5,374 photos of fisher were taken, with a total number of 179,640 photos.

- **Small-bodied animals** had detections at a higher proportion of cameras the closer they were to the camera. Mice and flying squirrel were significant (Figure 10).
- **Medium-bodied animals** had detections at a higher proportion of cameras the closer they were to the camera, though bobcat had detections at a higher proportion of cameras at farther distances (Figure 10) but not significantly.
- **Large-bodied animals** had seemingly no effect by distance from camera to bait (Figure 10). These results were not significant.
- **Fisher:** Compared to trail sets, fishers were detected more often at baited sets, Kitchen sink treatments significantly so regardless of lure.
- **Flashers:** Spotted skunk and long-tailed weasel were weakly correlated with more detections when using a flasher.
- **Chicken baited stations + Gusto:** Spotted skunk and woodrat detections were significantly positively correlated, significantly less deer and gray fox detections.
- **Kitchen sink baited stations:** Woodrat had significantly more detections with this bait and a Gusto combination, while gray fox had significantly less detections with a kitchen sink and Megamusk combination.
- **Cat Food + Gusto baited stations:** Woodrat had significantly more detections with this bait and lure combination, while gray fox had significantly less detections with this combination.
- **Near baited stations:** Mice, chipmunks, and flying squirrels had detections at a significantly high proportion of cameras the closer the camera was to the bait, while birds were weakly correlated.

*Discussion*

- Baited sets generally showed a positive correlation with detections compared to trails, suggesting the use of bait for future surveys. Detections might be less on trail sets because animals infrequently use the features where cameras are set, or populations exist at low densities.
- Fishers were generally detected at a high proportion of cameras with closer distances, and were significantly correlated with kitchen sink baited stations. For managers, to increase the likelihood of detecting fisher over trail sets, we suggest placing the camera closer to the bait and using a kitchen sink treatment. The type of scent lure seemingly makes no difference.
- Most species were not associated with stations treated with a flasher, however spotted skunks and long-tailed weasel show a weak association, though it is unclear why. Additional replicates are needed to assess whether the result was an anomaly.
- Species were more likely to be detected the closer the camera was placed to the bait. Therefore, to increase probability of species detection of small and most medium-bodied species, we suggest placing cameras ~2-4m from the bait.

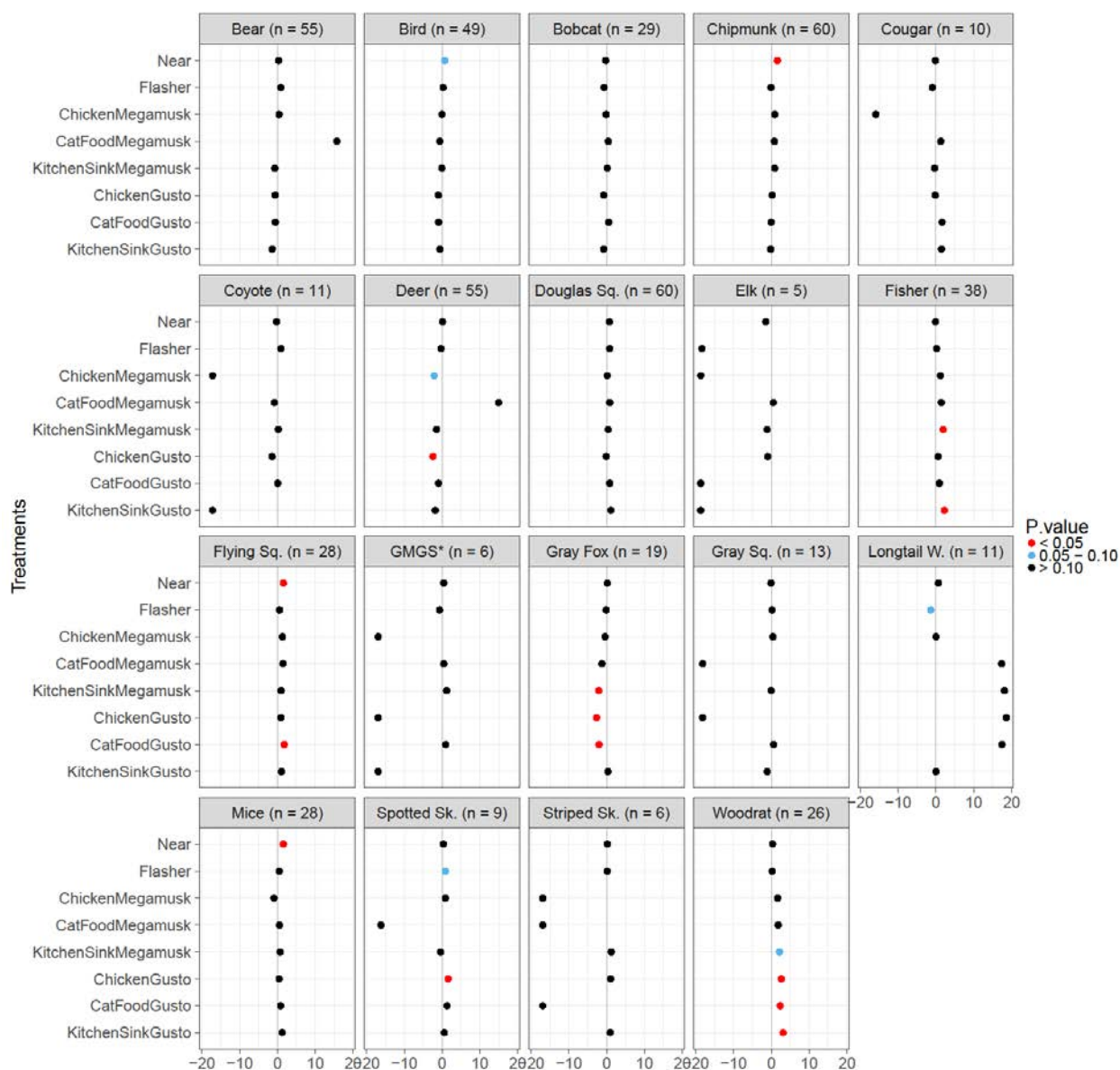


Figure 9. Effects of all possible bait and lure combinations, the presence of a flasher, and set distance represented as a categorical variable on detection probability of species that were detected on  $\geq 5$  cameras. Dots represent the estimate coefficients from logistic regression comparing detection history to attractant treatment. Right of the zero line is a positive correlation, while a dot to the left of the line is a negative correlation. \* = Golden-Mantled Ground Squirrel.

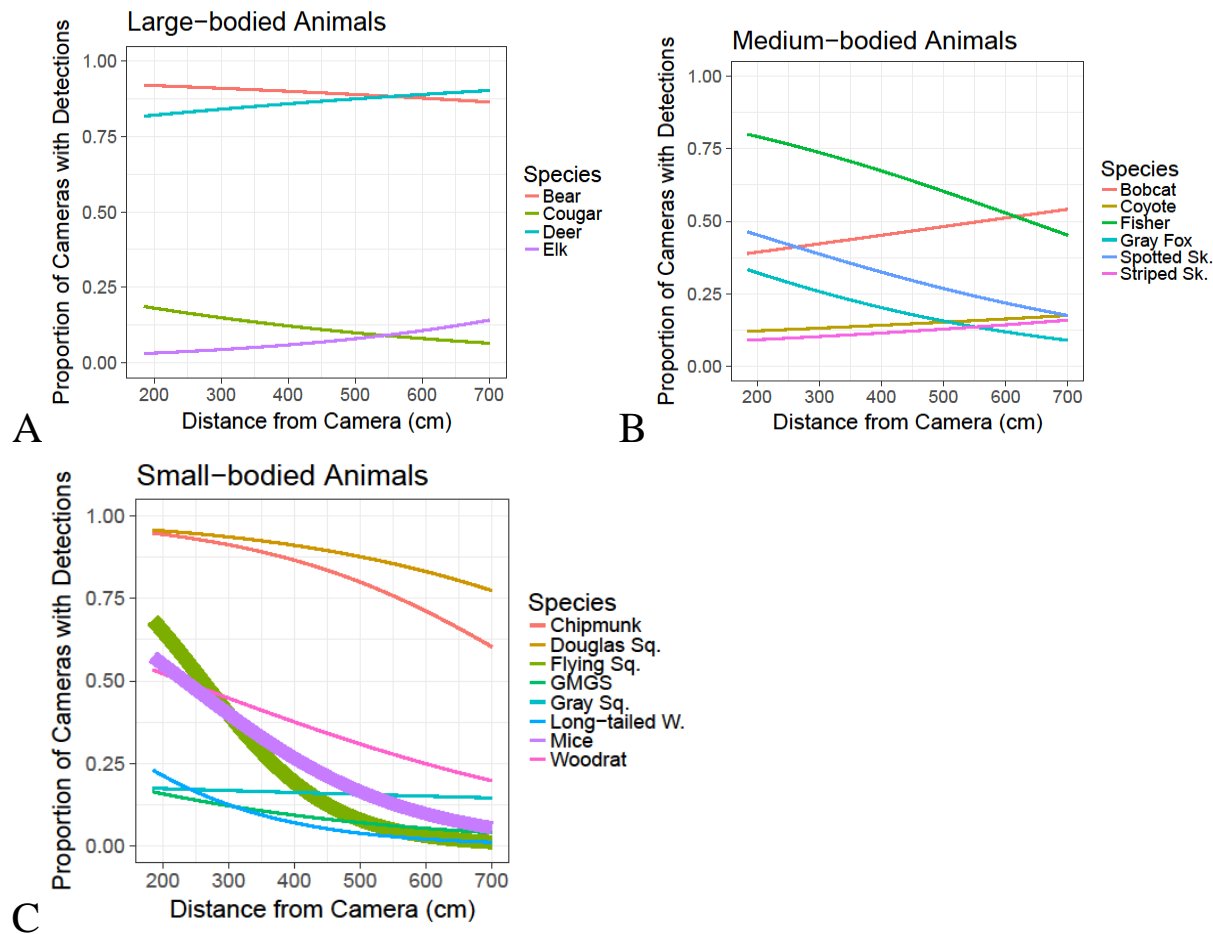


Figure 10. Logistic regression results comparing the proportion of cameras with detections (0-1) with distance from camera (180-700 cm) broken into 3 categories: (A) large-bodied animals, (B) medium-bodied animals, and (C) small-bodied animals. Trend significance is indicated by a thicker line on the plot. GMGS = Golden-Mantled Ground Squirrel.