

The effects of mixed-severity wildfires on fisher (*Pekania pennanti*) population dynamics
Baseline report of population dynamics pre-wildfires
In partial fulfillment of Cooperative Agreement Award F15AC00857

David S. Green^{1*}, Sean M. Matthews¹, Robert C. Swiers², & Roger A. Powell²

¹ Institute for Natural Resources, Oregon State University, Corvallis, Oregon, USA

² Department of Applied Ecology, North Carolina State University, Raleigh, North Carolina, USA

* contact information: greendav@oregonstate.edu or (602) 481-4524

Introduction

Fire is an integral ecological process in many parts of the world. Many ecosystems have adapted to, and are reliant on, fires for maintaining structural dynamics, for affecting nutrient cycling processes, for maintaining plant reproduction, and for creating landscape heterogeneity (Noss et al. 2006; Keeley et al. 2011). The combination of many years of fire suppression and global climate change is predicted to increase the frequency and intensity of wildfires in certain parts of the world (Westerling 2006; Flannigan et al. 2009). The long-term effects of this changing fire regime are a major source of conservation and management concern.

While much research has been dedicated to understanding the effects of wildfires on vegetation, birds, and small mammals, research on the effects of wildfires on carnivores is much more limited. A lack of baseline information about carnivore populations before fires occur has hindered our abilities to understand the short- and long-term effects of these wildfires and, thus, much of the previous research has focused on use of burned landscapes by carnivores (e.g., Dees et al. 2001; Jennings et al. 2016). The effects of wildfires on carnivore population numbers may

Please note the data, analyses, and results reported herein, though accurate, should be considered preliminary. More comprehensive analyses and results will be available in future reports and publications. Until such time, please consider citing the following as: Green, D.S., S.M. Matthews, R.C. Swiers, and R.A. Powell. 2016. The effects of mixed-severity wildfires on fisher (*Pekania pennanti*) population dynamics, baseline report of population dynamics pre-wildfires, in partial fulfillment of Cooperative Agreement Award F15AC00857. Institute for Natural Resources, Oregon State University, Corvallis, Oregon.

not be straightforward and may require long-term monitoring to fully understand. For example, carnivores may be affected by altered prey populations following a fire, or indirectly affected by decreased cover required for hunting, denning, and concealment following a fire. Thus, an important area of research is to understand the effects these ecological perturbations have on carnivore populations that occur in fire-prone environments, like populations of fishers (*Pekania pennanti*) in western and northcentral North America.

Fishers are a medium-sized forest carnivore that depend on structurally diverse forests with relatively dense canopies (Lofroth et al. 2010). Fishers require tree cavities for denning and reproduction (Raley et al. 2012; Zhao et al. 2012), select regions with complex structure as preferred resting sites in late-seral forests (Zielinski et al. 2004; Purcell et al. 2009; Aubry et al. 2013), and may experience increased predation pressure in regions with limited above-ground biomass (Weir & Corbould 2010; Raley et al. 2012). Extant, native fishers in the western United States currently reside in two spatially isolated populations (Zielinski et al. 2005; Tucker et al. 2012). Some of the habitats where fishers still remain are also prone to frequent wildfires (Safford & Van de Water 2014). Large-scale wildfires have the capacity to reduce, fragment, or eliminate habitat for fishers, so much so in fact, that the increasing frequency and intensity of wildfires are identified as major threats to their long-term conservation (Naney et al. 2012). Thus, fishers are well-suited to serve as a model for studying the effects of wildfires on forest-obligate carnivores.

To date, much of the previous research on the effects of wildfires on fishers has been limited to monitoring their presence in, and occupancy of, burned landscapes (e.g., Hanson 2015; Sweitzer et al. 2016). In the current research we use data collected from a long-term monitoring program to investigate the effects of two naturally-occurring mixed-severity wildfires on a

population of fishers. To our knowledge, no long-term, before and after research exists examining the effects of wildfire on an obligate, late seral forest carnivore. Data from before the fires and after allow us to disentangle the effects of the fire from naturally occurring variation in population dynamics. We developed a spatial capture-recapture model to investigate the effects of these fires on fisher population dynamics. In the current report, we present results from initial population models of the 8 years of data prior to the occurrence of wildfires.

Methods

The Klamath study site

We monitored fishers in a 587 km² portion of the Klamath-Siskiyou ecoregion in northern California and southern Oregon (henceforth, “the Klamath”) to investigate the effects of wildfires on fisher populations (Figure 1). The Klamath was predominantly comprised of conifer and mixed conifer/broadleaf forest. Elevation ranged from 472 to 2269 m.

Non-invasive surveys of meso-carnivores have taken place in the Klamath annually since 2006 between mid-September and early December. In the summer of 2014, two wildfires burned areas on and adjacent to the Klamath (Figure 1): the Happy Camp Complex and the Beaver Fires. We have 8 years of data before the fires occurred, 2 years of data following the fires and, thus, a unique situation to investigate the effects of wildfires on fishers in the Klamath. In the winters of 2009 and 2010, the fisher population in the Klamath also served as a source for a reintroduction of fishers to the northern Sierra Nevada and northern Cascade Mountains (Callas & Figura 2008). A total of 9 fishers (4F, 1M in 2009 and 3F, 1M in 2010) were removed, and of these, 8 were translocated to an area in the Stirling Management Area east of Chico, CA. One female in 2009 died in captivity. Relatively little research is conducted on source population

dynamics following translocation efforts (Armstrong & Seddon 2007; Lewis et al. 2012). Thus, an additional objective in our research was to understand the effects of these translocations on source population dynamics.

Non-invasive sampling techniques and individual identifications with genetics

We attempted to deploy 100 survey stations in the Klamath at the same locations near streams and on ridge tops each year (Figure 1). At each survey station we placed a non-invasive sampling box made of corrugated plastic (25x25x75 cm; Figure 2). Each box was baited with a raw chicken drumstick and a can of wet cat food. The back of the sampling box was closed with hardware cloth, and the front was partially obstructed with 3 wooden slats (Figure 2). We fixed a glue strip to the underside of the bottom wooden slat so that a mammal coming into the box to obtain the bait was likely to leave a hair sample attached to the glue strip.

Survey stations remained open for a maximum of 6 consecutive weeks each year, but were occasionally deemed to be “nonfunctional” if damage to the unit would have prevented an animal from leaving a viable genetic sample (e.g., if the box was flattened by a black bear; *Ursus americanus*). Stations were checked weekly, and each hair sample attached to a glue strip was immediately put in a desiccant-filled vial and sent to the USDA Forest Service Rocky Mountain Research Station for genetic analyses.

At the USDA Forest Service Rocky Mountain Research Station, DNA was extracted from hairs with follicles attached to each glue strip to optimize amplification, and then genotyped using the multi-tube approach recommended for non-invasive samples (Taberlet et al. 1996). All samples were amplified twice at each locus, and some were amplified a third time if the initial amplification resulted in a lack of consensus scores (Schwartz & Monfort 2008). If

these three scores did not prove to be consistent, then samples were discarded as being of insufficient quality for genetic analyses. For the samples with sufficient DNA, the program DROPOUT was used to screen for any potential errors in genotyping (McKelvey & Schwartz 2004). Any samples identified to contain putative errors were re-amplified an additional three times. Following the multi-tube test and the DROPOUT screens, field information in GIS was used to evaluate the likelihood of observing a recaptured genotype in a given location (Marucco et al. 2010).

Spatial capture-recapture model

We developed a spatial capture-recapture model to determine the effects of the two perturbations (i.e., wildfire, translocation) on the population demography of fishers in the Klamath. In spatial capture-recapture models, the spatially-explicit locations of each individual i are assumed to be a function of the distance to their latent activity center s_i (Efford 2004; Royle & Young 2008). Locations of s_i are then calculated by estimating individual encounter probabilities and the distribution of activity centers in the defined state-space (S).

It is important for S to be delineated as an area large enough to identify s_i for all individuals, even those where s_i is located outside of the vicinity of the study area. Home ranges of fishers in the Klamath have been documented to be (mean \pm SE) 9.6 ± 2.8 and 30.6 ± 8.6 km² for females and males, respectively (summarized from Lofroth et al. 2010). In the current research, S was defined as a 10-km buffer encompassing our sampling units (1836 km² in total) to accurately identify the locations of all activity centers of fishers. The locations of activity centers in the current research were modeled as an inhomogeneous Poisson point process in S (Royle et al. 2014). We divided S into a 1-km x 1-km grid and the location of each sampling unit

was then assigned to the grid cell where it was located, and the probability of s_i in year t (s_{it}) being at the center of grid cell j (prob_j) was modeled as an intensity function.

We used a Bayesian approach and fit our models using data augmentation (Royle et al. 2007). We introduced a sufficiently large number of all-zero encounter histories (nz) to our population of observed individuals (n) to prevent any truncation of the number of individual fishers with activity centers located in S . We also introduced the partially latent variable $z(i, t)$ indicating population membership for observed or unobserved individual i in year t equal to M ($M = nz + n$). We set $z(i, t) = 1$ with certainty for individuals sampled in a given year, and estimated this parameter for all remaining years. For all years $z(i, t) \sim \text{Bernoulli}(\Psi)$, and the number of individuals N alive in year t was thus:

$$N_t = \sum_{i=1}^M z(i, t)$$

We defined the number of detections y of individual i in grid cell j in year t as the Poisson-distributed random variable: $y_{ijt} \sim \text{Poisson}(p_{ijt}g_{ijt}z_{it}m_{jt})$. In this equation, p_{ijt} was the probability of detecting an individual in a grid cell if their activity center was the centroid of the grid cell, g_{ijt} was a detection function describing how the encounter rate of an individual decreases as a function of the distance between their activity center and the center of a grid cell, and m_{jt} was the number of weeks a sampling unit was functioning in grid cell j during year t . Previous research has identified sex-specific detection probabilities of fishers (Popescu et al. 2014), and an increased likelihood of visitation following an initial detection (Sweitzer et al. 2016); we modeled the log-linear mean encounter rate for individual i in grid cell j in year t (p_{ijt}) as:

$$\log(p_{ijt}) = \beta_0 + \beta_1 \bullet \text{sex}_i + \beta_2 \bullet \text{previousdetection}_{it}$$

where the mean encounter rate is a function of a population level intercept (β_0), a sex-specific effect (β_1), and the effect of a binary variable (0 or 1) indicating whether or not the individual had visited a sampling unit in previous years (β_2). We modeled the detection function (g_{ijt}) with a Gaussian encounter probability such that:

$$g_{ijt} = e^{(-d_{ijt}^2 / 2\sigma^2)}$$

where d_{ijt} is the Euclidean distance between the sampling units where an individual was located and the center of the grid cell of its activity center, and σ is the standard deviation of a bivariate normal distribution reflecting space-use also called the “movement parameter.” To restrict the capture probability to 0 for years when no sampling units in a grid cell were functional, we introduced the variable m_{jt} indicating the number of weeks that a unit in grid cell j was open in year t . Thus, the expected number of detections for individual i in a grid cell without a functioning sampling unit for the entire season was set to 0.

Model fitting and assessment

We fit our model using the Markov chain Monte Carlo (MCMC) methods of JAGS (Plummer 2003) in R v. 3.2.3 (R Core Team 2016) with the jagsUI package (Kellner 2014). We used vague prior distributions for all estimated parameters, specifically, Uniform(-10, 10) for β_0 , Normal(0, .01) for β_1 and β_2 , Uniform(0, 30) for σ , and Uniform(0, 1) for Ψ . Parameter estimates were calculated from 4,500 MCMC samples, taken from 3 chains run for 5,000 iterations, thinned by 3, and following a burn-in of 500. We assessed model convergence by examining trace plots and \hat{R} values for all parameter estimates (Gelman & Hill 2007; Gelman et al. 2013). All descriptive statistics are presented as yearly mean \pm standard deviation and all parameter estimates are presented as a median and 95% credible intervals.

Results

From 2006 to 2013, our sampling units were open and functioning for 2708.1 ± 439.4 sampling nights. During this time we collected 225.9 ± 28.4 samples that were submitted for genetic analyses. Of these samples submitted for analyses, 212.1 ± 19.5 (90.9 ± 6.1 %) were of high enough quality to identify the species of the visitor from 2007 to 2013 (genetic analyses were more limited in 2006). For samples identified as fisher, 84.4 ± 8.2 % were of sufficient quality to determine sex and genotype. The number of unique individuals sampled each year showed little variation. We identified a total of 139 unique individuals from 2006 to 2013, with 27.0 ± 3.4 individual fishers detected each year. Fishers were detected at multiple sampling units each year (1.7 ± 1.2). Inter-annual re-capture rates were also fairly stable over time; 16.6 ± 2.8 individuals sampled each year had been identified to be present in previous years.

The spatial capture-recapture model indicated the population of fishers in the Klamath was relatively stable before the fires occurred and for the three years immediately following the removal of fishers for translocations (Figure 3; Table 1). Estimated locations of fisher activity centers varied each year on the landscape (Figure 4).

Current conclusions and future directions

The current modeling efforts indicate the population of fishers in the Klamath was relatively stable from 2006 to 2013. The abundance estimates are unchanged among years, with no statistically significant differences (95% credible intervals overlap; Figure 3, Table 1). The removals of 5 fishers in 2009 and 4 fishers in 2010 appear to go undetected; our results show that removing this quantity of fishers for translocation did not alter the abundance or density of fishers in the Klamath in the short-term.

Our estimates of fisher abundance in the Klamath are slightly smaller than, but do not differ significantly from, previous estimates (Swiers 2013). Both our current results and previous work indicate a stable population of fishers in the Klamath before the wildfires occurred in 2014, and for up to 3 years following the translocation efforts. Two major benefits to the current modeling approach over previous analytical techniques include more precise estimates of population sizes and spatially explicit densities.

Managing forests for fishers is a challenging endeavor for conservation officials in the western United States. Fuels treatments and other wildfire prevention activities (e.g., prescribed burning, pre-commercial thinning, understory removal) can result in short-term negative consequences for fishers (Scheller et al. 2011; Sweitzer et al. 2016), but may be less severe in the long-term than the habitat change from an uncontrolled wildfire (Scheller et al. 2011; Truex & Zielinski 2013). The current research is imperative for assisting carnivore management and conservation efforts in the western United States, and for managing fire-prone forests for long-term fisher conservation (Naney et al. 2012).

The results presented here are preliminary. We intend to fit the spatial capture recapture model for fishers as a spatial Jolly-Seber open population model (Jolly 1965; Seber 1965; Gardner et al. 2010; Royle et al. 2014), allowing us to estimate apparent rates of sex-specific survival and recruitment. By fitting the model concurrently with data from all years through 2015 (available May 2016), we shall be able to investigate effects the wildfires had on fisher population dynamics and on habitat heterogeneity. Finally, if funding becomes available, we shall conduct multi-species analyses of data for additional medium-sized carnivores collected from sooted track plates set at our sampling units.

Acknowledgements

This multiyear collaboration would not be possible without the support by many people and organizations. We would like to thank the many technicians, industry collaborators (Sierra Pacific Industries, Fruit Growers Incorporated, California Michigan Timber Company), and state (California Department of Fish and Wildlife) and federal (U.S. Fish and Wildlife Service, USDA Forest Service) wildlife agencies and personnel for their dedication to this research. We also thank North Carolina State University and the National Council for Air and Stream Improvement for their assistance in funding this research.

Figures and Table

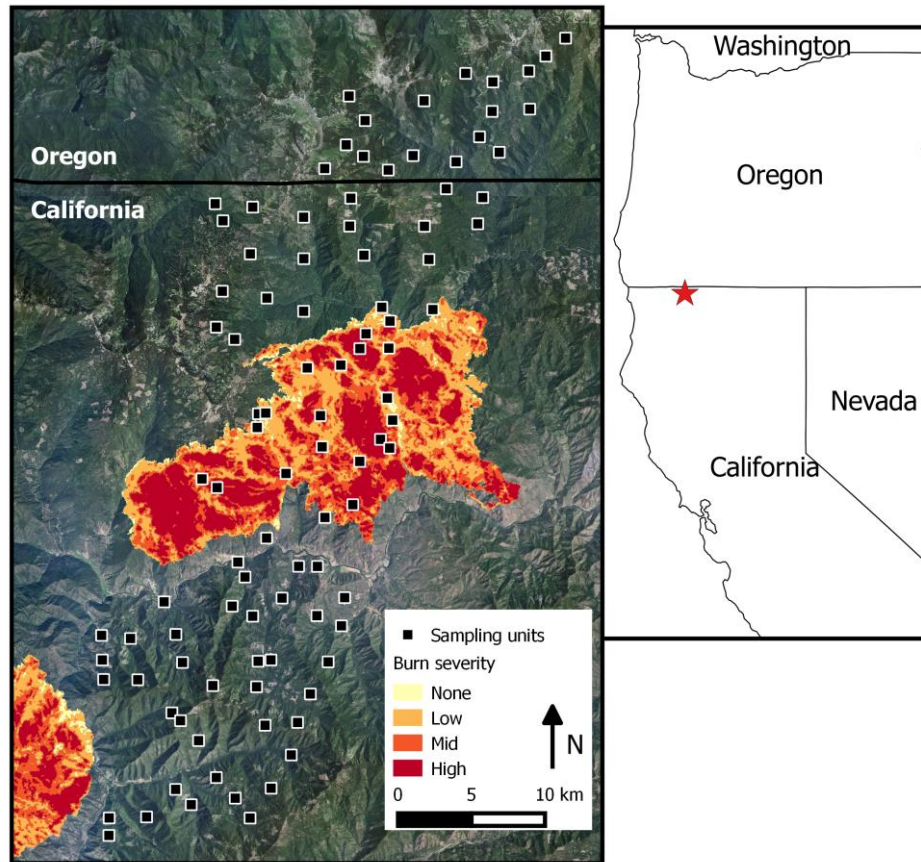


Figure 1. Study site for monitoring fisher population dynamics in response to two wildfires in Northern California. Both wildfires occurred in the summer of 2014. The Beaver Fire is the northern fire and the Happy Camp Complex Fire is the southern fire depicted in this figure. The black squares show the locations of our survey sites. The light-colored area of low elevation bisecting the study site in approximately the middle is the valley containing the Klamath River. This satellite image is courtesy of Microsoft Bing basemaps.



Figure 2. The non-invasive sampling device used to collect genetic samples from carnivores in the Klamath.

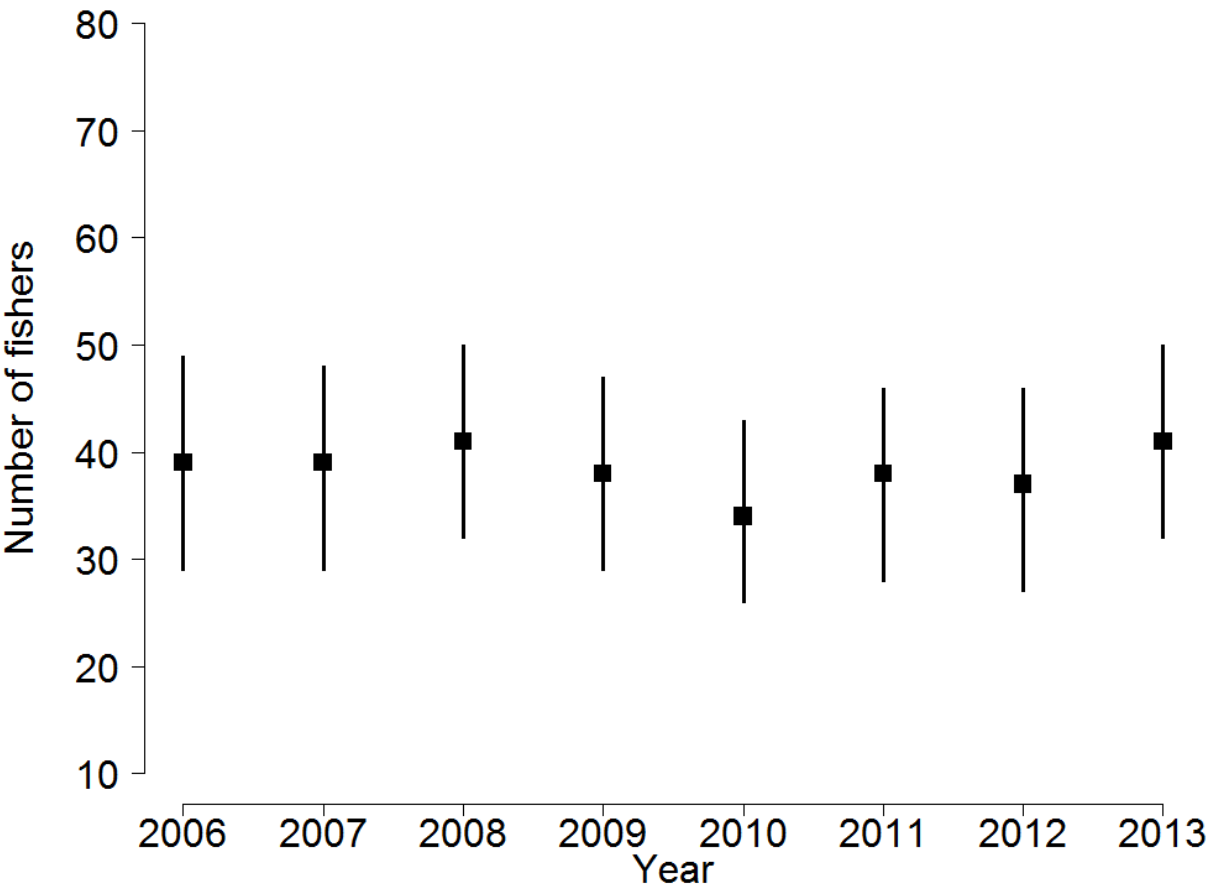


Figure 3. Estimated annual abundance (■) and 95% credible intervals of fishers in the Klamath.

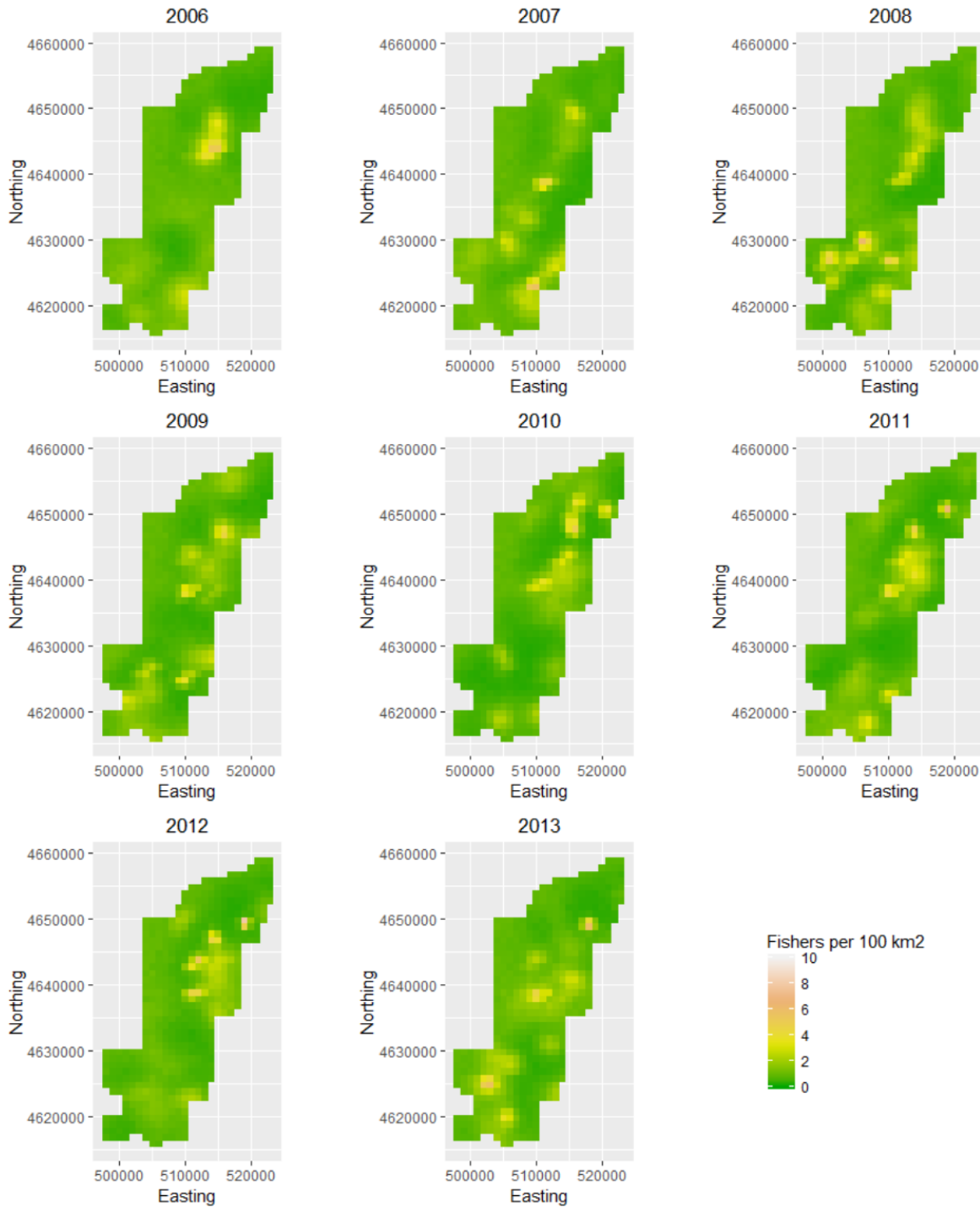


Figure 4. Density maps indicating the locations of fisher activity centers in the Klamath. The scale bar and color scheme indicate the number of estimated activity centers/100 km² encompassing the 1 km² grid cell.

Table 1. Derived posterior parameter estimates of annual population density, abundance, and population growth of fishers in the Klamath. Parameters are presented as median [95% credible interval].

Year	Density (fishers/100 km²)	Abundance	Lambda
2006	6.64 [4.94, 8.35]	39 [29, 49]	-
2007	6.64 [4.94, 8.18]	39 [29, 48]	1 [0.71, 1.35]
2008	6.99 [5.62, 8.69]	41 [32, 50]	1.06 [0.78, 1.4]
2009	6.47 [5.11, 8.18]	38 [29, 47]	0.92 [0.67, 1.2]
2010	5.79 [4.43, 7.33]	34 [26, 43]	0.91 [0.64, 1.21]
2011	6.47 [5.11, 8.18]	38 [28, 46]	1.09 [0.78, 1.45]
2012	6.3 [4.94, 8.18]	37 [27, 46]	0.98 [0.72, 1.33]
2013	6.99 [5.62, 8.69]	41 [32, 50]	1.11 [0.81, 1.49]

Literature cited

- Armstrong, D. P., and P. J. Seddon. 2007. Directions in reintroduction biology. *Trends in Ecology and Evolution* **23**:20–25.
- Aubry, K. B., C. M. Raley, S. W. Buskirk, W. J. Zielinski, M. K. Schwartz, R. T. Golightly, K. L. Purcell, R. D. Weir, and J. S. Yaeger. 2013. Meta-analyses of habitat selection by fishers at resting sites in the pacific coastal region. *The Journal of Wildlife Management* **77**:965-974.
- Callas, R. L. and P. Figura. 2008. Translocation plan for the reintroduction of fishers (*Martes pennanti*) to lands owned by Sierra Pacific Industries in the northern Sierra Nevada of California. California Department of Fish and Game. 80 pp.
- Dees, C. S., J. D. Clark, and F. T. Van Manen. 2001. Florida panther habitat use in response to prescribed fire. *The Journal of Wildlife Management* **65**:141–147.
- Efford, M. 2004. Density estimation in live-trapping studies. *Oikos* **106**:598-610.
- Flannigan, M. D., M. A. Krawchuk, W. J. de Groot, B. M. Wotton, and L. M. Gowman. 2009. Implications of changing climate for global wildland fire. *International Journal of Wildland Fire* **18**:483-507.
- Gardner, B., J. Reppucci, M. Lucherini, and J. A. Royle. 2010. Spatially explicit inference for open populations: estimating demographic parameters from camera-trap studies. *Ecology* **91**:3376–3383.
- Gelman, A., J. B. Carlin, H. S. Stern, D. B. Dunson, A. Vehtari, and D. B. Rubin. 2013. Bayesian data analysis, 3rd edition. CRC Press, Boca Raton, FL.
- Gelman, A., and J. Hill. 2007. *Data Analysis Using Regression and Multilevel/Hierarchical Models*. Cambridge University Press, New York.
- Hanson, C. T. 2015. Use of higher severity fire areas by female Pacific fishers on the Kern Plateau, Sierra Nevada, California, USA. *Wildlife Society Bulletin* **39**:497–502.
- Jennings, M. K., R. L. Lewison, T. W. Vickers, and W. M. Boyce. 2016. Puma response to the effects of fire and urbanization. *The Journal of Wildlife Management* **80**:221-234.
- Jolly, G. M. 1965. Explicit estimates from capture-recapture data with both death and immigration-stochastic model. *Biometrika* **52**:225-247.
- Keeley, J. E., J. G. Pausas, P. W. Rundel, W. J. Bond, and R. A. Bradstock. 2011. Fire as an evolutionary pressure shaping plant traits. *Trends in Plant Science* **16**:406–411.
- Kellner, K. 2014. jagsUI: Run JAGS (specifically, libjags) from R; an alternative user interface for rjags.

- Lewis, J. C., R. A. Powell, and W. J. Zielinski. 2012. Carnivore translocations and conservation: insights from population models and field data for fishers (*Martes pennanti*). *Plos One* **7**:e32726.
- Lofroth, E. C. et al. 2010. Conservation of fishers (*Martes pennanti*) in south-central British Columbia, western Washington, western Oregon, and California-Volume 1: Conservation Assessment. USDI Bureau of Land Management, Denver, Colorado, USA.
- Marucco, F., L. Boitani, D. H. Pletscher, and M. K. Schwartz. 2010. Bridging the gaps between non-invasive genetic sampling and population parameter estimation. *European Journal of Wildlife Research* **57**:1-13.
- McKelvey, K. S., and M. K. Schwartz. 2004. Providing reliable and accurate genetic capture–mark–recapture estimates in a cost-effective way. *Journal of Wildlife Management* **68**:453-456.
- Naney, R. H. et al. 2012. Conservation of Fishers (*Martes pennanti*) in South-Central British Columbia, Western Washington, Western Oregon, and California–Volume III: Threat Assessment. USDI Bureau of Land Management, Denver, Colorado, USA.
- Noss, R. F., J. F. Franklin, W. L. Baker, T. Schoennagel, and P. B. Moyle. 2006. Managing fire-prone forests in the western United States. *Frontiers in Ecology and the Environment* **4**:481-487.
- Plummer, M. 2003. JAGS: A program for analysis of Bayesian graphical models using Gibbs sampling. Available from <http://www.ci.tuwien.ac.at/Conferences/DSC-2003/Drafts/Plummer.pdf>.
- Popescu, V. D., P. Valpine, and R. A. Sweitzer. 2014. Testing the consistency of wildlife data types before combining them: the case of camera traps and telemetry. *Ecology and Evolution* **4**:933–943.
- Purcell, K. L., A. K. Mazzoni, S. R. Mori, and B. B. Boroski. 2009. Resting structures and resting habitat of fishers in the southern Sierra Nevada, California. *Forest Ecology and Management* **258**:2696-2706.
- R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>.
- Raley, C. M., E. C. Lofroth, R. L. Truex, J. S. Yaeger, and J. M. Higley. 2012. Habitat ecology of fishers in western North America: a new synthesis. Pages 231–254 in K. B. Aubry, W. J. Zielinski, M. G. Raphael, G. Proulx, and S. W. Buskirk, editors. *Biology and conservation of martens, sables, and fishers: a new synthesis*. Cornell University Press, Ithaca, New York, USA.
- Royle, J. A., R. B. Chandler, R. Sollmann, and B. Gardner. 2014. *Spatial capture-recapture*. Academic Press.
- Royle, J. A., and K. V. Young. 2008. A hierarchical model for spatial capture-recapture data. *Ecology* **89**:2281–2289.

Royle, J., R. M. Dorazio, and W. A. Link. 2007. Analysis of multinomial models with unknown index using data augmentation. *Journal of Computational and Graphical Statistics* **16**:67–85.

Safford, H. D., and K. M. Van de Water. 2014. Using fire return interval departure (FRID) analysis to map spatial and temporal changes in fire frequency on national forest lands in California. Res. Pap. PSW-RP-266. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 59 p.

Scheller, R. M., W. D. Spencer, H. Rustigian-Romsos, A. D. Syphard, B. C. Ward, and J. R. Strittholt. 2011. Using stochastic simulation to evaluate competing risks of wildfires and fuels management on an isolated forest carnivore. *Landscape Ecology* **26**:1491–1504.

Schwartz, M. K., and S. L. Monfort. 2008. Genetic and endocrine tools for carnivore surveys. Pages 238–262 in R. A. Long, P. MacKay, W. J. Zielinski, and J. C. Ray, editors. *Noninvasive survey methods for carnivores*. Washington DC, USA.

Seber, G. A. F. 1965. A note on the multiple-recapture census. *Biometrika* **52**:249–259.

Swiers, R. C. 2013. Non-invasive genetic sampling and mark-recapture analysis of a fisher (*Martes pennanti*) population in Northern California used as a reintroduction source (master's thesis, North Carolina State University).

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.

Taberlet, P., S. Griffin, B. Goossens, S. Questiau, V. Manceau, N. Escaravage, L. P. Waits, and J. Bouvet. 1996. Reliable genotyping of samples with very low DNA quantities using PCR. *Nucleic acids research* **24**:3189–3194.

Truex, R. L., and W. J. Zielinski. 2013. Short-term effects of fuel treatments on fisher habitat in the Sierra Nevada, California. *Forest Ecology and Management* **293**:85–91.

Tucker, J. M., M. K. Schwartz, R. L. Truex, K. L. Pilgrim, and F. W. Allendorf. 2012. Historical and contemporary DNA indicate fisher decline and isolation occurred prior to the European settlement of California. *Plos One* **7**:e52803.

Weir, R. D., and F. B. Corbould. 2010. Factors Affecting Landscape Occupancy by Fishers in North-Central British Columbia. *Journal of Wildlife Management* **74**:405–410.

Westerling, A. L. 2006. Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. *Science* **313**:940–943.

Zhao, F., R. A. Sweitzer, Q. Guo, and M. Kelly. 2012. Characterizing habitats associated with fisher den structures in the Southern Sierra Nevada, California using discrete return lidar **280**:112–119.

Zielinski, W. J., R. L. Truex, F. V. Schlexer, L. A. Campbell, and C. Carroll. 2005. Historical and contemporary distributions of carnivores in forests of the Sierra Nevada, California, USA. *Journal of Biogeography* **32**:1385-1407.

Zielinski, W. J., R. L. Truex, G. A. Schmidt, F. V. Schlexer, K. N. Schmidt, and R. H. Barrett. 2004. Home range characteristics of fishers in California. *Journal of Mammalogy* **85**:649-657.