

The effects of mixed-severity wildfires on fisher (*Pekania pennanti*) population dynamics
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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 not be straightforward and may require long-term monitoring to fully understand. For example,

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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.

Methods

The Klamath study site

We monitored fishers in a 465 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.

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 Jolly-Seber open population model

We evaluated the effects of removing fishers on population dynamics in Klamath with a spatial Jolly-Seber open population model (Jolly 1965; Seber 1965). Traditional sampling for these models results in data from secondary sampling periods (e.g., weeks) across > 1 primary sampling periods (e.g., years), often referred to as the “robust design” (Pollock 1982). Recently developed Jolly-Seber models fit with spatial capture-recapture integrate the movements of individuals directly into their formulation to provide spatially-explicit estimates of population density (Gardner *et al.* 2010; Royle *et al.* 2014). Spatial Jolly-Seber population models have the capacity to improve the precision of parameter estimates (e.g., Ergon & Gardner 2013; Whittington & Sawaya 2015).

In these models, the spatially-explicit sampled locations of each individual i are assumed to be a function of the distance to its latent activity center s_i (Efford 2004; Royle & Young 2008). Locations of each s_i are then calculated by estimating individual-specific encounter probabilities and the distribution of activity centers in the defined state-space (S). The number of estimated s_i of individuals observed or unobserved in the state-space equates to the population size. We defined S as a continuous state-space bounded by the outside of a rectangular 9-km buffer around our sampling units, greater than 3 times the size of the estimated movement parameter (σ ; Royle *et al.* 2014). Initial models were run using a discrete state-space to investigate if elevation or canopy cover affected the distribution of fishers in Klamath, but these covariates were not significant (95% credible intervals overlapped 0); we reparametrized the model using a continuous state-space.

We fit our models using data augmentation methods (Royle & Dorazio 2008; Royle & Young 2008). We introduced 550 all-zero encounter histories (nz) to our population of observed individuals (n) to prevent any truncation of the number of fishers with activity centers in S . We also introduced the partially latent variable z_{it} indicating population membership for observed or unobserved individual i in year t . Following Gardner *et al.* (2010), $z_{i1} \sim \text{Bernoulli}(\Psi_{pop})$ for year $t = 1$ and $z_{it} \sim \text{Bernoulli}(\mu_{it})$ for all years $t > 1$, where:

$$\mu_{it} = (z_{it-1} \times \Phi_{t-1}) + (A_{it-1} \times \gamma_t)$$

Here the probability that an individual is estimated to be in the population in years $t > 1$ is a function of survival (Φ_{t-1}) from the previous year if already present (z_{it-1}) or of the probability that an individual is recruited (γ_t) into the population if not already present (A_{it-1}). In the latter part of this equation we used the term A_{it-1} to ensure that individuals can only be recruited into the population once such that:

$$A_{it-1} = 1 - \text{step}(a_{it} - 1)$$

$$a_{it} = \text{sum}(z_{i1:t})$$

We defined a positive detection y of individual i in year t at sampling site j in week w as the Bernoulli-distributed random variable: $y_{itjw} \sim \text{Bernoulli}(p_{itjw}g_{itj}z_{it}m_{tjw})$. In this equation, p_{itjw} was the probability of detecting individual i in year t at site j in week w . The detection function g_{itj} described how the encounter rate of individual i decreased as a function of the distance between its activity center in that year (s_{it}) and the location of sampling site j . The previously described latent variable z_{it} indicated population membership and m_{tjw} was a binary variable indicating whether or not in year t sampling box j was deployed in week w .

We hypothesized that there would be sex-specific detection probabilities and an increased probability of visitation following an initial detection as has been shown in previous studies

(Popescu, Valpine & Sweitzer 2014; Sweitzer *et al.* 2016; Linden *et al.* 2017). We also hypothesized that time of year may influence detection, and predicted that individuals would be more likely to visit sampling locations closer to winter due to decreased food availability. We modeled the logit-linear mean encounter rate (p_{ijw}) as:

$$\text{logit}(p_{ijw}) = \alpha_0 + \alpha_1 \times \text{sex}_i + \alpha_2 \times \text{previousdetection}_{ijw} + \alpha_3 \times \text{week}_w$$

where the mean encounter rate was a function of an intercept (α_0), a sex effect (α_1), an effect of a binary variable indicating whether or not the individual had visited this site in a previous week of that year (α_2), and a week-specific effect to estimate the effect of seasonality (α_3). Initial models varied α_0 by year, but there were no significant differences among α_0 in this parametrization (95% credible intervals overlapped each other) and this parameterization was ultimately dropped.

We modeled the detection function (g_{ij}) from a Gaussian encounter probability such that:

$$g_{ij} = e^{(-d_{ij}^2 / 2\sigma_k^2)}$$

where d_{ij} is the Euclidean distance between the sampling units where an individual was located and the location of its activity center that year (s_{it}), and σ_k is the standard deviation of a bivariate normal distribution reflecting space-use, also called the “movement parameter”, and was calculated independently for each sex (k). Sex was estimated as the Bernoulli-distributed random variable: $\text{Sex}_i \sim \text{Bernoulli}(\Psi_{\text{sex}})$. Data were collected during the fisher dispersal season (Arthur, Paragi & Krohn 1993; Matthews *et al.* 2013), but the movement parameter is robust to violations of stationary movement around the latent activity centers (Royle, Fuller & Sutherland 2016). We modeled activity centers independently each year for every individual as a homogeneous Poisson point process in S (Royle *et al.* 2014). Population size and density were calculated as a function of the number of estimated activity centers located within the Klamath study area.

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 uninformative prior distributions for all parameters. Spatial capture-recapture parameter estimates were calculated from 4,000 MCMC samples, taken from four chains run for 3,000 iterations, thinned by three, following an adaptation and burn-in period of 1,000 iterations. 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 2015, our sampling units were open and functioning for 2704.1 ± 388.5 sampling nights. During this time we collected 241.6 ± 46 samples that were submitted for genetic analyses. Of these samples submitted for analyses, 212.6 ± 47.6 (87.6 ± 6.4 %) were of high enough quality to identify the species of the visitor from 2006 to 2015. For samples identified as fisher, 83.9 ± 8.9 % were of sufficient quality to determine sex and genotype. We identified a total of 178 unique individuals from 2006 to 2015, with 28.1 ± 3.8 individual fishers detected each year. Fishers were detected at multiple sampling units each year (1.7 ± 0.2). Inter-annual re-capture rates were also fairly stable over time; 15.8 ± 2.9 individuals sampled each year had been identified to be present in previous years.

The spatial Jolly-Seber 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). The spatial capture-recapture model indicated the fisher population remained stable between 2013 (year prior to the fire) and 2014 (year of the fire with mop-up operations underway during survey period; Figure 5). The fisher population declined after wildfires between 2014 and 2015 (Figure 5), with notable decreases within the burn area in both 2014 and 2015 (Figure 6). Fisher density was affected most by >50% change in canopy within the burn area between 2013 and 2015 (Figure 7).

Current conclusions and future directions

Our results 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.

The relative stability of the overall population between 2013 and 2014 and a decline in density within the burn area suggests fishers moved outside of the burn area immediately following the fire but remained within our larger study area. The significant decline in the overall

population between 2014 and 2015 appears largely driven by declines within the burned area. We suspect fishers began redistributing themselves on the landscape immediately following the fire and the significant decline observed between 2014 and 2015 was a function of a lag effect as fishers redistributed themselves on the post-fire landscape. Similarly, Sweitzer et al. (2016) found fisher occupancy in the southern Sierra Nevada Mountains was negatively correlated with higher levels of managed burning and forest fires. They also found local persistence of fishers declined in areas with more fuels management reduction, but did not decline in areas of prior forest fires or extractive activities; they hypothesized declines in local persistence were likely temporary. Continued, post-fire monitoring of fishers in the Klamath is a unique opportunity to empirically test the short- and long-term response of fishers to wildfire.

We also observed a greater negative effect on male fisher density. This could be due, in part, to their larger home ranges (Lofroth et al. 2010) and ability to re-establish themselves in areas of their pre-fire home ranges that were not burned. We also found fisher densities declined across all levels of fire severity and densities declined the most in regions with more than 50% change in canopy cover. Thus, wildfires where canopy declines by more than 50% will have the largest, negative effect on fishers. Wildfires have burned 25% of the Klamath region in the last three decades, with 33% of the area having burned at high severity (Tepley et al. 2016). The Klamath is also subject to an increasing frequency of large fires (Barbero et al. 2015; Rogers et al. 2011; Westerling and Bryant 2008), and it is projected to experience minimal conifer regeneration in even moderate-sized high-severity-burn patches (Tepley et al. 2016). All of these ecological changes are likely to exacerbate the potential threat to fisher population persistence (Naney et al. 2012).

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 for 2016 and 2017, allowing us look at longer-term effects of the wildfires. We are also working with the National Genomics Center for Wildlife and Fish Conservation to individually identify gray fox and ringtail genetic samples to investigate effects the wildfires had on the broader meso-carnivore community.

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Figures and Table

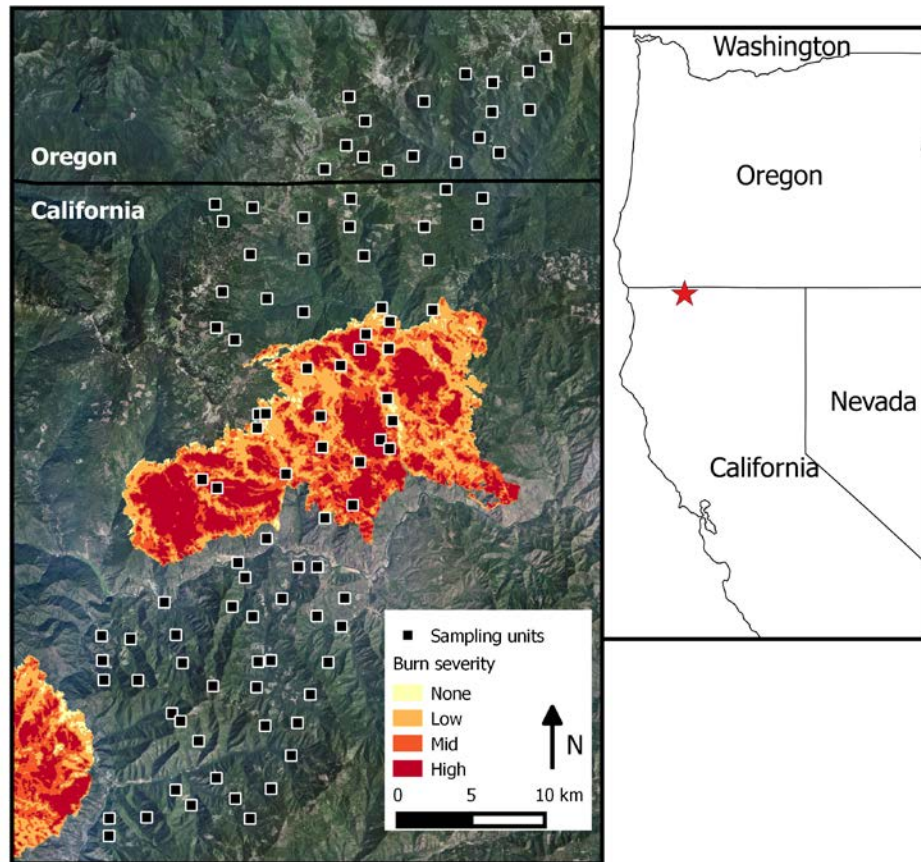


Figure 1. Study site for monitoring fisher population dynamics in response to two wildfires in Northern California. Both wildfires occurred in the summer of 2014. The Beaver Fire is the northern fire and the Happy Camp Complex Fire is the southern fire depicted in this figure. The black squares show 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.

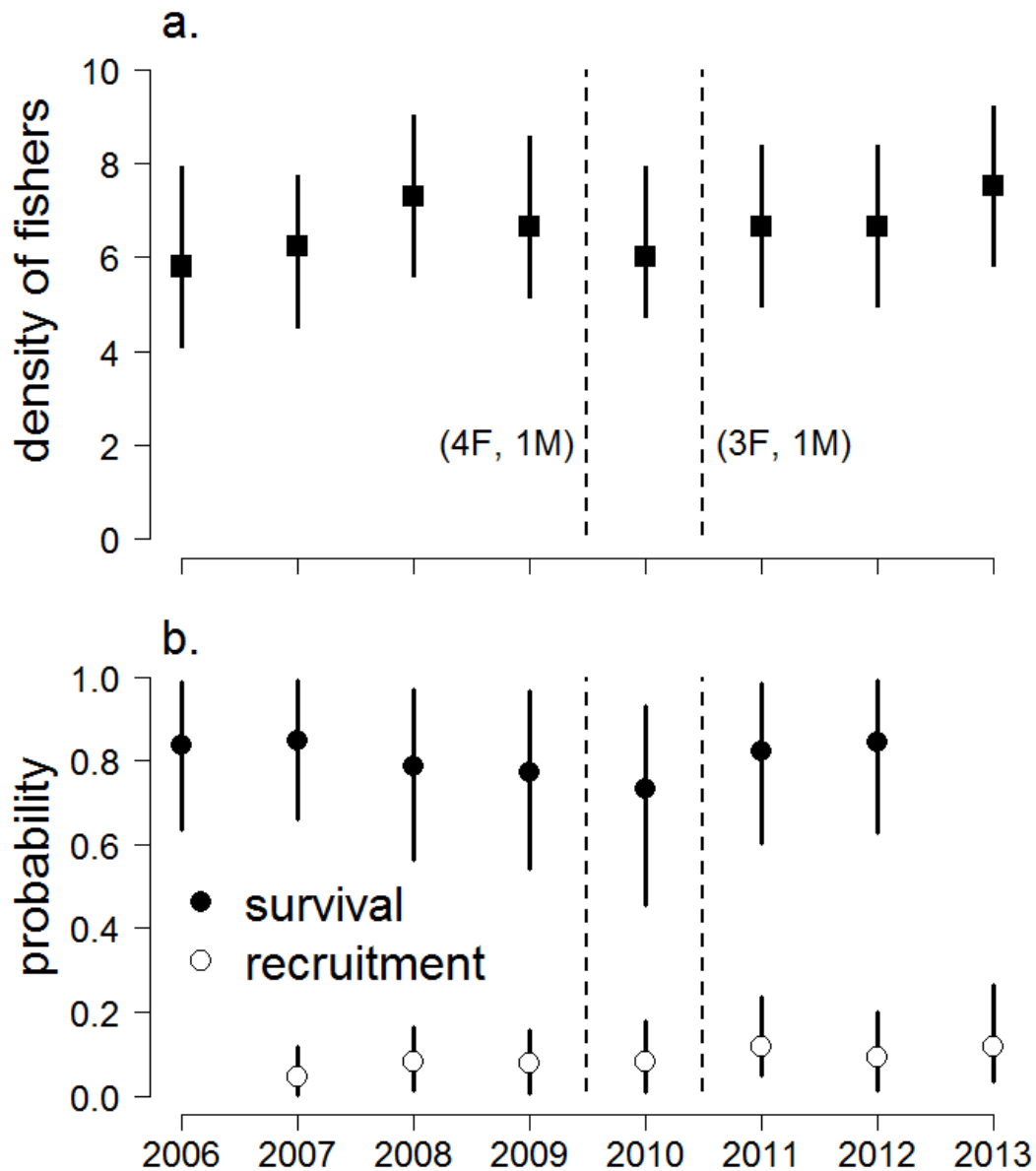


Figure 3. The modeled median and 95% credible intervals of (a) fisher density, and (b) apparent survival and recruitment in Klamath, estimated by a spatial Jolly-Seber open population model. Density is reported as the number of fishers per 100 km² within the convex hull encompassing our sampling sites. Apparent survival indicates the probability of an individual surviving to the following year and apparent recruitment indicates the probability of an individual being recruited into the population in that year. The dashed lines indicate the timing of removals along with the quantity and sexes of fishers removed. Density, survival, and recruitment estimates were not significantly different among years (95% credible intervals overlapped).

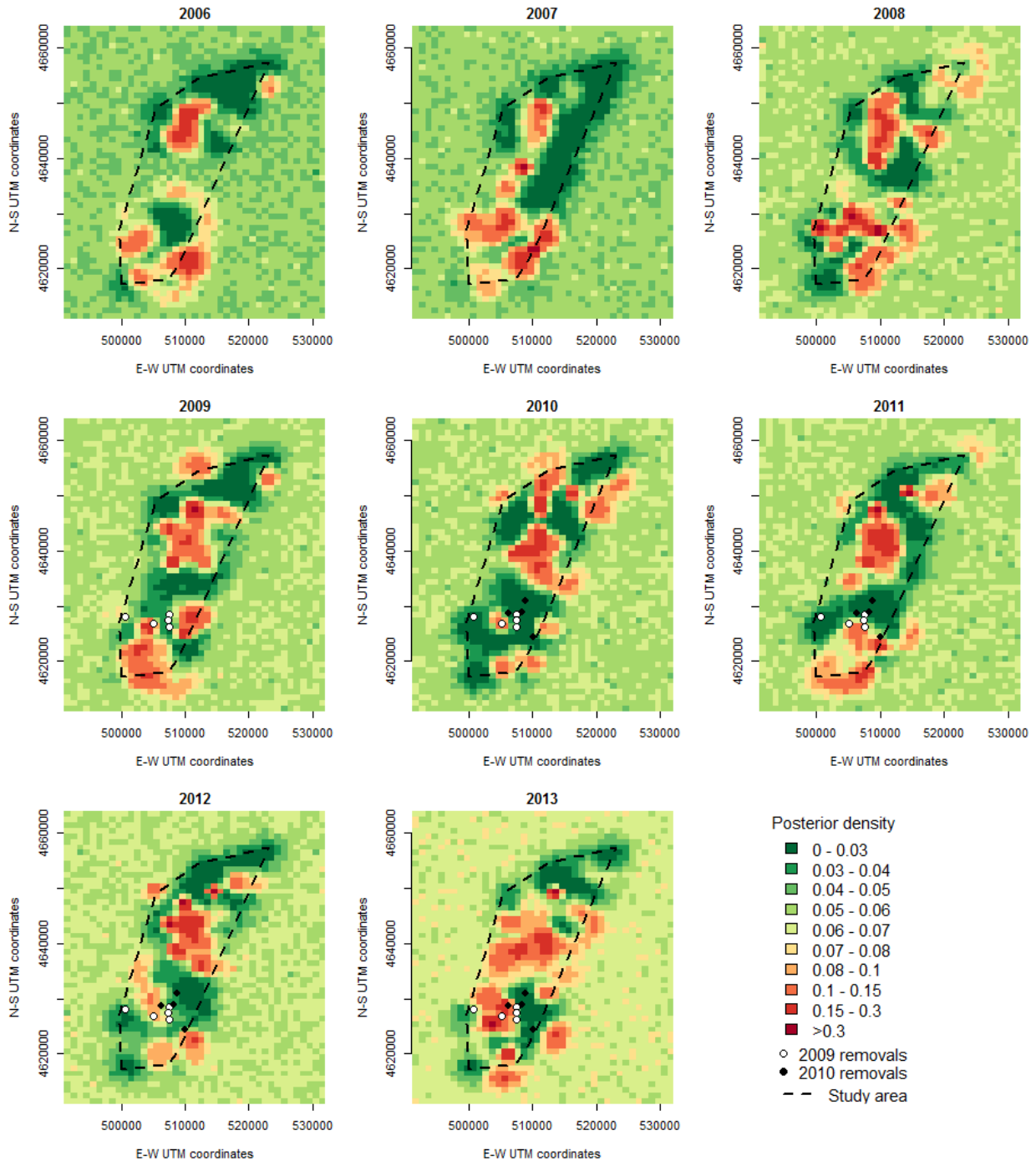


Figure 4. Modeled locations of estimated activity centers of fishers in Klamath between 2006 and 2013, as determined with a spatial Jolly-Seber open population model. Warmer colors indicate a larger quantity of fisher activity centers in the 1x1-km² grid cell in that year. The locations of live-capture sites used to remove fishers in the winters of 2009 and 2010 are noted with circles, color coded by year of removal. The convex hull encompassing our sampling sites is indicated with a dashed line.

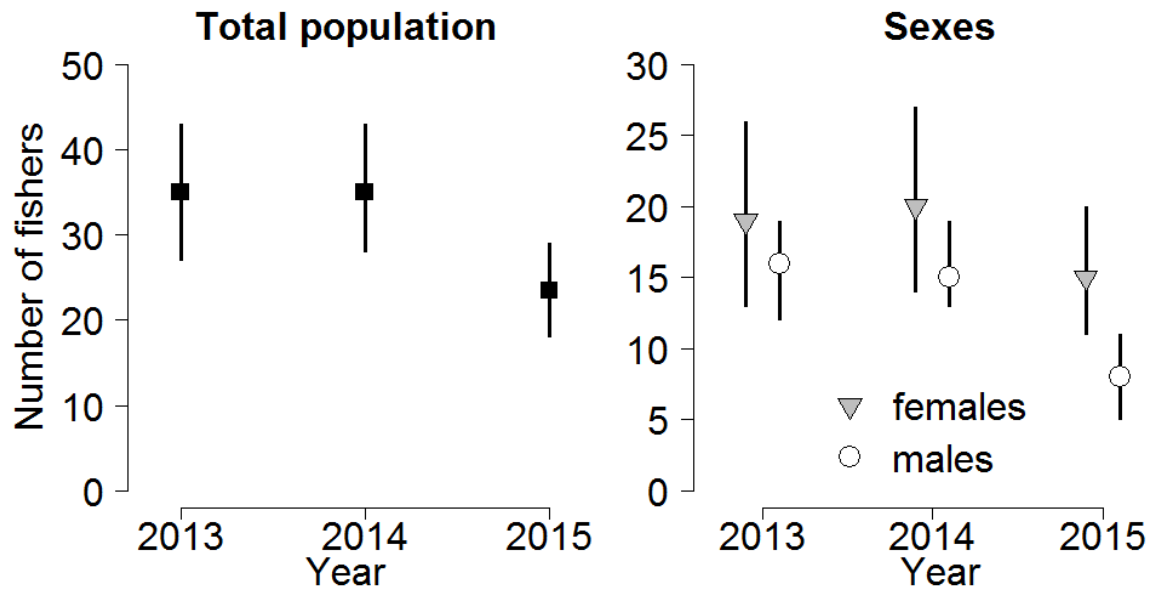


Figure 5. Estimated median and 95% Credible Intervals of the number of fishers and their sexes at our study site the year before the fires (2013), the year during the fires (2014), and the first year after the fires (2015).

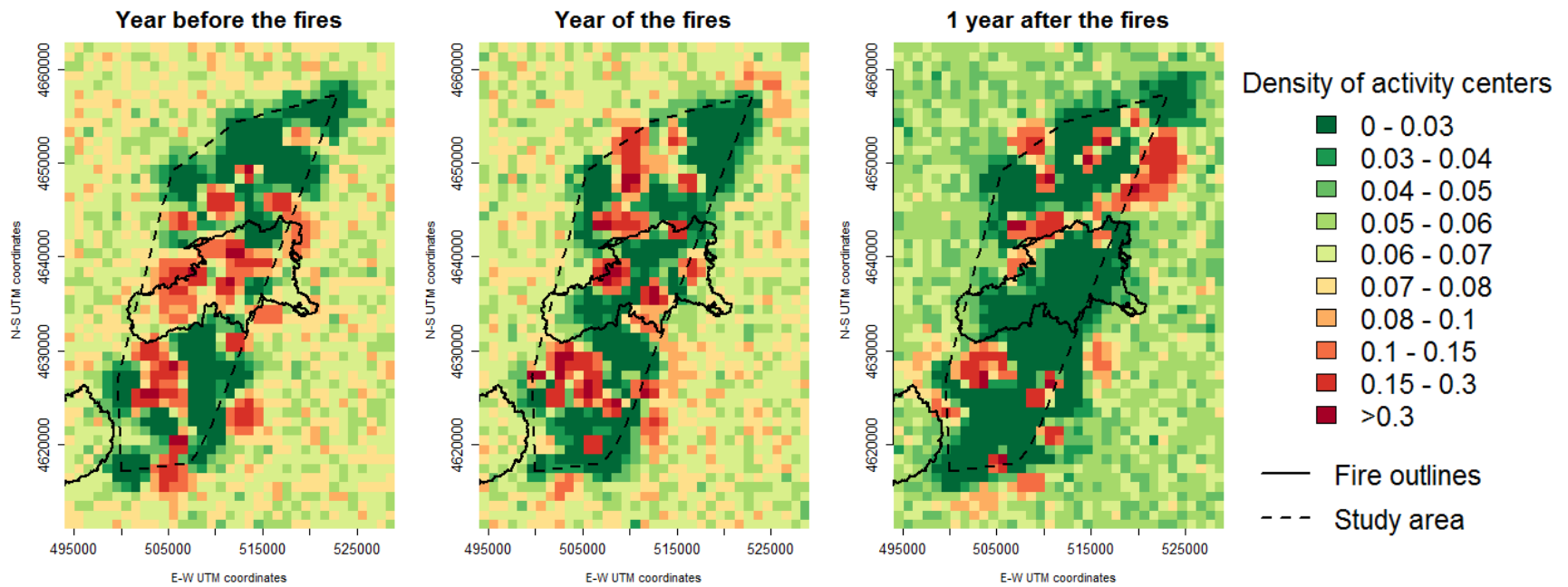


Figure 6. Estimated distribution of fisher activity centers on the landscape the year before the fires (2013), the year of the fires (2014), and 1 year after the fires (2015).

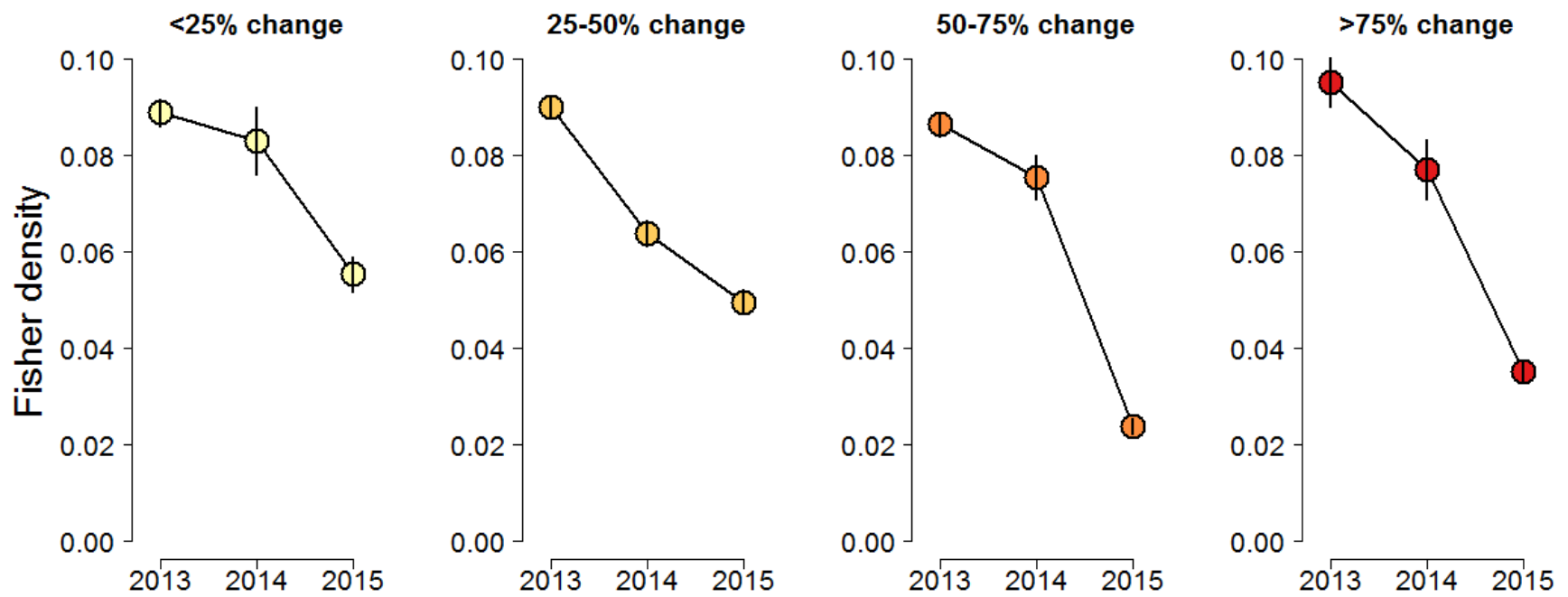


Figure 7. Mean \pm SE density of fishers per km² over time in grid cells located within the Beaver Creek Fire. The proportion of grid cells with >50% change in canopy cover had a significant negative effect on fisher density after the fire in 2014.

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

Year	Density (fishers/100 km²)	Abundance
2006	5.81 [4.09, 7.96]	27 [19, 37]
2007	6.24 [4.52, 7.74]	29 [21, 36]
2008	7.31 [5.59, 9.03]	34 [26, 42]
2009	6.67 [5.16, 8.6]	31 [24, 40]
2010	6.02 [4.73, 7.96]	28 [22, 37]
2011	6.67 [4.95, 8.39]	31 [23, 39]
2012	6.67 [4.95, 8.39]	31 [23, 39]
2013	7.53 [5.81, 9.25]	35 [27, 43]

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