Forest fires have a short-term negative effect on the forest-dependent carnivoran the fisher (*Pekania pennanti*)

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Abstract

The combination of many years of fire suppression and global climate change is predicted to increase the extent and intensity of wildfires in certain parts of the world, particularly in western North America. Understanding the effects of forest fires on forest obligate species is an important question in ecology and is also a conservation and management concern. We used data collected from a long-term monitoring program to investigate the effects of 2 naturally-occurring, mixed-severity forest fires on forest obligate mammalian carnivorans, fishers (*Pekania pennanti*). We monitored fishers in a portion of their northern California and southern Oregon population, which is a population stronghold for the species in the western USA but is also increasingly vulnerable to high-severity forest fires. We developed a spatial capture-recapture model to estimate the short-term effects of these 2 fires on fisher abundance and distribution using genetic data collected non-invasively with hair snares 1 year prior to the fires and for 3 years following them. Fisher abundance decreased by approximately 40% following these fires, particularly in the areas of the fire footprints and irrespective of the burn severity. We also estimated that 4% of the northern California and southern Oregon population of fishers has been exposed to forest fires in the last 3 years. Our results suggest that the changing fire dynamics in the western USA will have strong, negative effects on fisher populations. Populations of other forest-obligate species may also fare poorly under these altered fire regimes.

Introduction

Wildfires influence natural ecological communities (Agee 1993) and many populations of animals and plants occupying fire-prone landscapes have evolved in response to the impacts of wildfire within the historic range of fire variability (Noss *et al.* 2006; Rockweit, Franklin &

Carlson 2017; Spies *et al.* 2018). The frequencies, distribution, and intensities of wildfires in some parts of the world have increased over time (Stephens 2005; Miller *et al.* 2009; Lutz *et al.* 2009; Miller *et al.* 2012) due to decades of fire suppression (Stephens *et al.* 2009; Calkin, Thompson & Finney 2015), past and current timber harvest practices (Odion *et al.* 2004) and climate change (Liu, Goodrick & Stanturf 2013; Westerling 2016; Abatzoglou & Williams 2016). Wildfires now lead to large scale losses of natural resources and to ecological changes (Adams 2013; Stephens *et al.* 2014; Spies *et al.* 2018), and they are becoming increasingly expensive to manage (Calkin *et al.* 2015). Departures from historical wildfire norms have prompted calls for restoration to natural fire regimes via forest restoration and fuels management (e.g., using tree thinning; Calkin *et al.* 2015; North *et al.* 2015), but short- and long-term consequences for wildlife living in these fire-prone landscapes are unclear.

A controversial dichotomy of perspectives has emerged regarding whether forest fires have negative impacts on populations of species that depend on mature forests. The most extensively studied of these species is the spotted owl (*Strix occidentalis caurina*), yet no consensus exists regarding how spotted owls respond to forest fires (Rockweit *et al.* 2017; Ganey *et al.* 2017). In general, wildfires negatively affect spotted owl survival (Clark, Anthony & Andrews 2011), reproduction (Jenness, Beier & Ganey 2004), occupancy and colonization rates (Clark, Anthony & Andrews 2013), but other studies found negative effects were primarily associated with increasing fire severities (Jones *et al.* 2016; Rockweit *et al.* 2017; Eyes, Roberts & Johnson 2017). Interpretations are further complicated by studies that report limited to no discernible effects of wildfires on owl survival and reproduction (Bond *et al.* 2002), occupancy (Bond *et al.* 2002; Roberts *et al.* 2011; Lee & Bond 2015), colonization and extinction rates (Lee, Bond & Siegel 2012), and no negative effects associated with increasing fire severities (Bond *et al.* 2009; Bond, Bradley & Lee 2016). Additional research is clearly needed to understand the effects of forest fires on mature-forest-dependent species.

The fisher (*Pekania pennanti*), a large member of the weasel family, depends on mature forests and is the focus of conservation concern in the Pacific states of the United States (USDI Fish and Wildlife Service 2016). Fisher populations suffered significant decreases and range contractions attributed to over-trappings for their fur, loss of forested habitats, and predator and pest control campaigns (Douglas & Strickland 1987; Powell 1993; Powell & Zielinski 1994). The Distinct Population Segment of fishers in the Pacific United States is currently proposed for protection under the federal Endangered Species Act (U.S. District Court for the Northern

District of California 2018) and increasing forest fire frequencies and intensities are identified as potential threats to fisher persistence (USDI Fish and Wildlife Service 2016). Limited evidence suggests that forest fires may decrease fisher occupancy (Thompson, Zielinski & Purcell 2011; Scheller *et al.* 2011; Garner 2013; Sweitzer *et al.* 2016), but no study has estimated empirically the effects of forest fires on fisher population numbers. Understanding how fishers are affected by wildfires of varying intensities will both aid in their conservation and also assist in understanding how forest-obligate species respond to wildfires (Tempel *et al.* 2015; Sweitzer *et al.* 2016).

We evaluated the effects of mixed-severity forest fires had on fishers in a portion of the Klamath-Siskiyou Ecoregion in northern California and southern Oregon, USA. The Klamath-Siskiyou Ecoregion is a global hotspot for biodiversity (Whittaker 1960; Myers *et al.* 2000) and it also supports the largest extant population and greatest density of fishers in the western United States (Matthews *et al.* 2011; USDI Fish and Wildlife Service 2016; Furnas *et al.* 2017). We capitalized on 2 forest fires that burned a combined total of 132 km^2 (~28%) of a 465 km^2 study area of a long-term fisher monitoring program (Green *et al.* 2018). Having data on the fisher population before the fires occurred offers an unprecedented ability to separate the effects of wildfires on fishers from other naturally-occurring variation. We hypothesized that the effects of forest fire on fisher density would be contingent on fire severity. Specifically, we tested the predictions that, 1) low-severity forest fire has little to no effect on fisher density, but that 2) high-severity forest fire cause decreases in fisher density. We also investigated the scale and distribution of all forest fires that have recently occurred throughout the fisher range in northern California and southern Oregon to estimate the threat that forest fires pose to fisher conservation and other species that depend on mature forests.

Methods

Study site and long-term data collection

We monitored fishers in a 465 km^2 portion (henceforth, "Klamath") of the Klamath-Siskiyou Ecoregion in northern California and southern Oregon to investigate the effects of forest fires on fisher populations (Figure 1a). Klamath was predominantly conifer and mixed conifer-broadleaf forest. Elevation ranged from 472 to 2269 m.

We have surveyed non-invasively for meso-carnivorans in Klamath annually between mid-September and early-December since 2006 (Green *et al.* 2018). In the summer of 2014, 2

forest fires burned areas on and adjacent to Klamath (Figure 1b): the Beaver Fire and Happy Camp Complex. The Beaver Fire burned 132 km^2 from July 30th to September 2nd and the Happy Camp Complex burned 543 km^2 from August 12th to October 31st. Both of these fires were ignited by lightning and burned at mixed-severity (Fig. 1b), with 47.5% and 27.7% of the Beaver Fire and Happy Camp Complex burning at high severity (i.e., $\geq$ 50% basal area mortality), respectively (Table 1). Sampling efforts in 2014 were unaffected by the fires and continued through 2016. Thus, we have 8 years of data before the fires occurred and 3 years of data following the fires. Previous research indicates that this population of fishers had been relatively stable up to 2013, despite approximately 20% of the population being translocated elsewhere between 2009 and 2011 (Green *et al.* 2018). Here, we used data from 2013 to 2016 to investigate the short-term effects of these fires on this fisher population. To disentangle the effects of the fires from any other naturally occurring variation, we used 2013 to serve as a baseline to determine changes to this fisher population that may have resulted from the fires. Previous results have indicated that the fisher population in 2013 was average in terms of the number of fishers present between 2006 and 2013 (Green *et al.* 2018).

Non-invasive sampling techniques

We deployed 100 survey stations in Klamath at the same locations each year (Fig. 1b). Each survey station consisted of a corrugated plastic box with one side closed with ½ inch (1.3 cm) hardware cloth, and the other side partially obstructed with 3 wooden slats (Figure 1c; Zielinski *et al.* 2006). We affixed a strip of non-poisonous glue board (Catchmaster 72MB, USA) to the underside of the bottom wooden slat so that mammals entering the plastic box were likely to leave a hair sample on the glue strip to be used for genetic analyses. Survey stations remained open for 4 to 6 consecutive weeks each year, were baited each week with a raw chicken drumstick and a can of wet cat food (Figure 1c), and were checked weekly for hair samples. Every glue strip with hair attached to it was put in a desiccant-filled vial in Klamath and analyzed to species. Samples identified to be from fishers were also analyzed to individual and sex (see Green *et al.* 2018 for additional details).

Spatial capture-recapture model

We developed a spatial capture-recapture model (Efford 2004; Royle & Young 2008) to determine the effects of the 2 wildfires on the population of fishers in Klamath. In our

hierarchical formulation, the spatially-explicit detection histories for individual i at survey station j in week w of year t (y_{ijwt}) were modeled as:

$$y_{ijwt} \sim \text{Bernoulli}(\lambda_{ijwt}K_{jwt}z_{it}),$$

where λ_{ijwt} is the average encounter rate, K_{jwt} is a binary variable indicating whether or not survey station j was open in week w of year t, and z_{it} is the partially-observed latent variable indicating whether or not individual i was present in the population in year t. We modeled the average encounter rate as a function of the probability of detection $p0_{ijwt}$ and a detection function g_{ijt} based on the distance to an individual's latent activity center s_{it}, such that $\lambda_{ijwt} = p0_{ijwt}g_{ijt}$. Previous research in Klamath and elsewhere have identified sex-specific detection probabilities of fishers (Popescu, Valpine & Sweitzer 2014; Linden *et al.* 2017; Green *et al.* 2018), and an increased likelihood of visitation following an initial detection (Sweitzer *et al.* 2016; Linden *et al.* 2017; Green *et al.* 2018); we modeled the logit-linear mean probability of detection ($p0_{ijwt}$) as:

$$\text{Logit}(p0_{ijwt}) = \beta0_t + \beta1*sex_i + \beta2*previousdetection_{ijwt},$$

where the probability of detection is a function of a year-specific intercept ($\beta0_t$), an effect of sex ($\beta1$), and the effect of a binary variable indicating whether or not the individual had visited this survey station in a previous week in this year ($\beta2$). We modeled the year-specific intercepts, $\beta0_t$, by assuming they were random effects, drawn from a uniform distribution: $\beta0_t \sim \text{Dunif}(-5, 5)$.

The detection function g_{ijt} described how the encounter rate of an individual decreases as a function of the distance between their activity center in that year (s_{it}) and the location of the survey stations, and was modeled with a Gaussian encounter probability such that:

$$g_{ijt} = e^{(-d_{ijt}^2 \,/\, 2\sigma_{kt}^2)},$$

where d_{ijt} is the Euclidean distance between the survey stations where an individual was located and its latent activity center, and σ_{kt} is the standard deviation of a bivariate normal distribution reflecting space-use modeled to vary by sex k and by year t. We modeled σ_{kt} as the random variable with an uninformative prior: $\sigma_{kt} \sim \text{Dunif}(0, 3000)$.

To estimate the effects of wildfires on fisher density and distribution in Klamath, we modeled individual activity centers each year as an inhomogeneous Poisson point process in the state-space S (Royle *et al.* 2014). We divided S into a 1.5-km x 1.5-km grid of R grid cells and we estimated the probability of s_{it} being inside grid cell r in year t ($prob_{rt}$) using the intensity function:

$$s_{it} \sim \text{Categorical}(prob_{1:R,t})$$

$$prob_{rt} = \frac{\mu_{rt}}{EN_t},$$

where μ_{rt} is the predicted number of fishers being in grid cell r in year t and EN_t is the expected population size in year t (Royle *et al.* 2014); $EN_t =$ sum($\mu_{1:R,t}$). For the year before the fires occurred (i.e., 2013), we modeled the predicted number of fishers in each grid cell to vary solely by an intercept ($\alpha0_{,2013}$): $\mu_{r2013} = area_r * e^{(\alpha0,2013)}$; $area_r$ is the area of the grid cell. In the years following the fires and through the end of the study (i.e., 2014 to 2016), we modeled μ_{rt} as:

$$\mu_{rt} = area_r * e^{(\alpha0t + \alpha1*2013density_r + \alpha2t*low_r + \alpha3t*high_r)},$$

where the predicted number of fishers in each grid cell was modeled to vary as a function of a year-specific intercept ($\alpha0_t$), an effect of the estimated number of fishers in that grid cell in 2013 ($\alpha1$; standardized to have a mean of 0 and a standard deviation of 1), and the effects of the proportion of the grid cell that was burned at less and greater than 50% severity ($\alpha2$ and $\alpha3$, respectively). Fire severity data were acquired from the Rapid Assessment of Vegetation Condition after Wildfire (RAVG). Fishers are a mature forest dependent species, which are strongly associated with forests that have $\geq$ 60% canopy cover (Lofroth *et al.* 2010). We defined anything <50% and anything $\geq$50% basal area mortality as low and high severity, respectively, to best identify how these wildfires affected the fisher population in Klamath. We included the estimated density of fishers in that grid cell from 2013 to account explicitly for spatial variation in the density and distribution of fishers prior to the wildfires, and to allow us to determine the effects of the mixed-severity wildfires on the fishers that were present in the grid cells that were burned. We modeled $\alpha2$ and $\alpha3$ as time series variables to account for any changes to spatially explicit densities that occurred over time. We modeled the year-specific intercepts, $\alpha0_t$, by assuming they were random effects drawn from a uniform distribution: $\alpha0_t \sim$ Dunif(-5, 5).

We used a Bayesian approach and fit our models using data augmentation (Royle, Dorazio & Link 2007). We introduced a sufficiently large number of all-zero encounter histories to our population of observed individuals to prevent any truncation of the number of individual fishers with activity centers located in S. The partially-latent variable indicating population membership (z_{it}) equaled 1 with certainty for individuals positively identified in a given year, and we estimated this parameter for all individuals in all other years as $z_{it} \sim$ Bernoulli(Ψ_t), where $\Psi_t = \frac{EN_t}{M}$ and M is the total number of observed and unobserved individuals being monitored. The number of individuals N alive in year t, N_t, was thus:

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

We fit our model using the Markov chain Monte Carlo (MCMC) methods of JAGS (Plummer 2003). We used uninformative prior distributions for all estimated parameters. Parameter estimates were calculated from 3,600 MCMC samples, taken from 3 chains run for 6,000 iterations, thinned by 5, and following a burn-in of 3,000. We assessed model convergence by examining trace plots and $\hat{R}$ values for convergence (Gelman & Hill 2006; Gelman *et al.* 2013). We used posterior distributions to calculate percent probabilities and to determine significance of parameter estimates, which was calculated as the percent of posterior draws greater or less than zero, depending on the sign of the median value.

Understanding the threat of forest fires for the fishers in the Klamath-Siskiyou Ecoregion

The fishers in our Klamath study site constitute a sample of fishers from the population of fishers in the Klamath-Siskiyou Ecoregion (Fig. 1a). Previous research has estimated that the ecoregion and adjacent areas have between 2,507 and 4,184 fishers spread across 47,907 km^2 (Furnas *et al.* 2017). The effects of the changing fire regimes in western North America on the ecoregion, however, are unknown (USDI Fish and Wildlife Service 2016). To understand the threat of wildfires to this population of fishers, we estimated the proportion of the ecoregion and adjacent areas that have burned from 2015 through 2017, the years for which data on the distribution and severity of forest fires are publicly available, and which are the years following the population estimate by Furnas *et al.* (2017).

Results

We deployed our survey stations for 496 ± 45 (yearly mean $\pm$ SD) sampling weeks during 2013 through 2016. We collected 292 ± 60 samples per year that were submitted for genetic analyses, of which, 272 ± 59 ($93 \pm 5\%$) were of sufficient quality to identify the species of the visitor. We collected 120 ± 39 samples from fishers, and $83 \pm 13\%$ were of sufficient quality to determine sex and genotype. We identified a total of 92 unique individuals from 2013 through 2016, with 32 ± 1 individual fishers detected each year. Each year, we detected 2 ± 2 fishers at multiple sampling units.

Previous research indicated that the population of fishers in Klamath was relatively stable before the fires occurred (Green *et al.* 2018; Fig. 2). Our spatial capture-recapture model showed a 40% decrease in the total number of fishers in Klamath, apparent the first full year after the fires (2015; Fig. 2). This decrease was most striking within the areas that actually burned, but

also occurred throughout all of Klamath (Figs. 3 & 4). Baseline detection probabilities for fishers were consistent over time ($\beta0_{2013}$ median and 95% Credible Interval on the logit scale = -1.8 [-2.4, -1.2], $\beta0_{2014}$ = -1.7 [-2.3, -1.0], $\beta0_{2015}$ = -1.6 [-2.1, -1.1], $\beta0_{2016}$ = -1.6 [-2.1, -1.1]; Table 2). Sex had a 90% probability of affecting detection, with males being more likely to be detected than females ($\beta1$ median and 95% Credible Interval on the logit scale = -0.5 [-1.3, 0.2]), and animals had a 100% probability to be more likely to be detected after an initial visit to the survey station ($\beta2$ median and 95% Credible Interval on the logit scale = 2.3 [1.8, 2.8]). The movement parameters of males were consistently larger than those of females (Table 2). Compared to other years, the movement parameters for both male and female fishers were also the largest in 2015, indicating that fishers traveled greater distances the first full year after the fires (Table 2). The intercept for the intensity function was similar among years ($\alpha0_{2013}$ median and 95% Credible Interval on the logit scale = -2.5 [-2.9, -2.1], $\alpha0_{2014}$ = -2.7 [-4.1, -2.2], $\alpha0_{2015}$ = -3.1 [-4.3, -2.6], $\alpha0_{2016}$ = -2.9 [-4.3, -2.4]). The predicted density of fishers in each grid cell from 2014 to 2016 had a 100% probability of being positively correlated with the density of fishers in 2013 ($\alpha1$ median and 95% Credible Interval on the logit scale = 0.8 [0.4, 1.6]), indicating that barring changes in habitat, fishers exhibit some degree of site fidelity. The proportion that each grid was burned in the fires affected the density of fishers. The probability that low and medium severity fire had negative effects on fisher density was 77% ($\alpha2$ median and 95% Credible Interval on the logit scale = -1.4 [-6.4, 2.01), and was 95% for high severity fire ($\alpha3$ median and 95% Credible Interval on the logit scale = -2.2 [-7.4, 0.3]).

Fisher density varies considerably across the Klamath-Siskiyou Ecoregion (Furnas *et al.* 2017) and so does the occurrence of forest fires. Approximately 4% of the fisher population in the ecoregion and surrounding areas has been exposed to forest fire at varying severity levels from 2015 through 2017 (Table 3, Figure 5).

Discussion

The 2 forest fires that occurred in 2014 had strong, negative effects on the fisher population in Klamath. We estimated a 40% reduction in the number of fishers due to the fires, a decrease that became apparent the first full year following the fires and persisted for at least 2 more years (Fig. 3). The number of fishers decreased throughout our entire study area and the decreases were not limited solely to within the fire perimeter (Figs. 3 & 4). The greatest declines in the estimated number of fishers, however, were primarily located in the fire footprints (Fig. 4).

Thus, our results indicate that the increasing fire extent and intensity that is currently occurring in western North America are likely to affect the fisher population of the Klamath-Siskiyou Ecoregion negatively. In the years since the last estimate of the fisher population in the ecoregion and adjacent areas, approximately 4% of the fisher-occupied range has been burned by forest fires (Fig. 5), and the extent and intensity of wildfires is expected to increase under future climate scenarios (Stephens 2005; Miller *et al.* 2009; Lutz *et al.* 2009; Miller *et al.* 2012). The fisher population of the Klamath-Siskiyou ecoregion and adjacent areas appears likely not to fare well given the current fire landscape, unless actions are taken to improve conditions for fisher.

Evaluating empirically the effects of forest fires on animals that depend on mature forests is particularly challenging and we were fortunate to have already been studying fishers in an area that burned. Many studies are unable to disentangle the effects of fires themselves from other naturally occurring variations due to a lack of pre-fire data, to small sample sizes, or to examining only short term effects (see Rockweit *et al.* 2017). Our results are limited in that they reflect the effects of only 2 fires in a vast landscape and consider only the short-term effects of these fires. It is also possible that the negative effects of fires on fishers may not persist for many years into the future or are a function of some other ecological phenomenon occurring on the landscape. Fisher populations that experience forest fires in other regions may not decrease, as we found, and we expect that consensus on the effects of mixed-severity fires on forest species will require studying a wide range of taxa across varying fire characteristics and severities and for long periods of time following the fires. Despite these caveats with the current study, previous research that includes pre-fire data align with our results; Swetizer *et al.* (2016) found fisher occupancy to be low in areas with high levels of managed burning and fires, and Rockweit *et al.* (2017) reported that medium- and high-severity fires decreased owl survival and had varying effects on recruitment.

The fire regime in the Klamath-Siskiyou Ecoregion and neighboring western landscapes is changing following decades of fire suppression, timber harvest and climate change (Odion *et al.* 2004; Calkin *et al.* 2015; Abatzoglou & Williams 2016). The long-term effects of the changing fire regime on forest obligate species are uncertain and require future research. The short-term effects, however, are likely to be negative as we have demonstrated here. For species that are limited in their distributions and sensitive from a conservation perspective, like the fisher, population decreases in the short-term could evoke irreparable damage to their long-term persistence. The Klamath-Siskiyou Ecoregion, where we studied fishers, is the geographic center

of the largest population of fishers in the western United States (USDI Fish and Wildlife Service 2016; Furnas *et al.* 2017). In the time since Furnas *et al.* (2017) estimated an average density of 6.6 fishers per 100 km² (3,196 fishers) in the ecoregion and adjacent areas, nearly 2000 km² (4%) of their 48,760 km² estimated distribution has burned (Table 3, Fig. 5). Although fishers are native to landscapes that have experienced forest fire historically, changing fire regimes in recent and future decades may have lasting negative effects (USDI Fish and Wildlife Service 2016).

Some mammalian carnivoran populations are decreasing (Ripple *et al.* 2014), and the effects of wildfire on their populations are not well understood. Fires may have positive effects on populations of some mammalian carnivorans by increasing prey populations, by increasing habitat suitability, or decreasing populations of competitors. Fires may also have negative effects by making mammalian carnivorans more susceptible to predation, by altering habitats permanently to be unsuitable, or by killing the carnivorans directly. Unfortunately, understanding the long-term effects of wildfires on mammalian carnivorans without long-term monitoring is challenging because many carnivorans have relatively slow life histories, making it difficult to differentiate between short- and long-term effects. Thus, a paucity of studies to date have been able to disentangle the effects of fire on mammalian carnivorans and, those that have, mostly measured occupancy or space-use (e.g., Dees, Clark & Van Manen 2001; Green *et al.* 2015). Our results are some of the first conclusive evidence to demonstrate that wildfires have negative effects on the population of a mammalian carnivoran at least in the short-term. Information on long-term effects of fire on mammalian carnivorans is severely needed, especially those that can indicate the underlying mechanisms that may be driving species population declines (i.e., changes in survival or reproduction).

The future conservation of fishers and other forest-obligate species will require an understanding of the tradeoffs between short- and long-term effects of fuels management alternatives and varying wildfire severities to habitat and population densities (Tempel *et al.* 2015; Sweitzer *et al.* 2016). Although the short-term effects of fuel management and restoration on forest-dependent species are variable, they can be negative to species associated with closed-canopy forests and dense understory (Pilliod *et al.* 2006; Scheller *et al.* 2011; Tempel *et al.* 2014; Ganey *et al.* 2017; Spies *et al.* 2018). Limited evidence suggests that fuels treatments have negative effects on fisher occupancy and population size (e.g., Thompson *et al.* 2011; Scheller *et al.* 2011; Garner 2013; Sweitzer *et al.* 2016) and spotted owl space-use and reproduction (e.g.,

Tempel *et al.* 2014). Whether these effects are smaller than the indirect, positive effects of fuels treatments to reducing the extent and severity of wildfires is unclear, but the negative consequences of proactive fuels management may be more desirable if they outweigh the negative effects of high-severity fires (Sweitzer *et al.* 2016; Rockweit *et al.* 2017; Ganey *et al.* 2017). Conservation of forest-obligate species in changing fire regimes will require the balance between the short-term impacts of fuel management and restoration with the negative effects of high-severity fires under changing climate scenarios.

Acknowledgements

Table and Figure Legends

Table 1. Descriptive information about the burn severities of the two forest fires that occurred in our study area during the summer of 2014. A burn severity of 0 indicates an area within the fire perimeter that showed no difference in % basal area mortality between 2013 and 2014. All burn severity data were acquired from Rapid Assessment of Vegetation Condition after Wildfire (RAVG).

Table 2. Derived posterior parameter estimates of annual population abundance (N), sex and year specific movement parameters in m (σ), the intercept of the intensity function describing the distribution of activity centers in Klamath by year ($\alpha0$), the effect of the density of fishers in 2013 on later fisher density ($\alpha1$), the effect of the proportion of a grid cell burned at low to medium severity ($\alpha2$) and high to very high severity ($\alpha3$) on fisher density, the intercept for the probability of detection by year ($\beta0$), and the effects of sex ($\beta1$) and previous visit ($\beta2$) on detection probability. The number of fishers (N) refers to the total number of fishers estimated to have activity centers within Klamath, which is defined as the minimum convex polygon encompassing all of our survey sites.

Table 3. Area of occupied fisher habitat in northern California and southern Oregon burned by wildfires since 2015 when data were collected for a regional population estimate (Furnas et al. 2017). Burn severity was categorized as low (<50% basal area mortality) and high (≥50% basal area mortality) using Rapid Assessment of Vegetation Condition after Wildfire (RAVG).

Figure 1. Study site and methods for monitoring the effects of two forest fires on fishers. (a) The location of the Klamath study site (red star) within the population of fishers in the Klamath-Siskiyou Ecoregion (blue shading). (b) The distribution and severity of the two mixed-severity forest fires that occurred in the summer of 2014 in Klamath around the 100 survey sites (purple squares). The Beaver Fire is the northern fire and the Happy Camp Complex Fire is the southern fire. Areas that burned at low and high severity are indicated in orange and red, respectively. This satellite image is courtesy of Google Earth. (c) The non-invasive sampling box used to collect genetic samples from fishers in Klamath. The metal track extending out the front of the box would be slid into the box before deployment.

Figure 2. Fisher population abundance over time in Klamath. We present the median and 95% Credible Intervals for (a) the total number of fishers and (b) the breakdown of these numbers by sex. The red vertical line in both figures indicates the timing of the two forest fires. Modeling for Green *et al.* (2018) was performed using a Jolly-Seber open population spatial capture-recapture model, whereas our model did not formally link years over time. Estimates from 2006 to 2013 were reproduced with permission from Green *et al.* (2018).

Figure 3. Predicted posterior density and distribution of fishers in each 1.5x1.5 km grid cell in Klamath from 2013 to 2016 determined with spatial capture-recapture. The dashed lines indicate the convex hull encompassing our survey sites and what we defined as the Klamath study area. The solid lines indicate the perimeters of the two wildfires. Cooler and warmer colors indicate relatively fewer and more estimated activity centers in each grid cell each year, respectively.

Figure 4. Predicted change in fisher density and distribution from 2013 to 2016. This value was calculated as the difference between the number of estimated fisher activity centers in each 1.5x1.5 km grid cell in 2013 subtracted from the number of activity centers in 2016. The resulting values were then standardized to have a mean of 0 and a standard deviation of 1. Cooler and warmer colors indicate relatively fewer and more estimated activity centers in 2016 compared to the number that were there in 2013, respectively.

Figure 5. Predicted density and distribution of fishers in the Klamath-Siskiyou Ecoregion in northern California and southern Oregon (reproduced with permission from Furnas *et al.* (2017)). The distribution of wildfires that have occurred after this estimate are outlined in red. The location of the current study is indicated with a black star (★).

Table 1.

Burn severity (% basal area loss)	Beaver Fire	Happy Camp Complex Fire
0	21.44 (15.95%)	194.68 (35.81%)
> 0 - 10	23.75 (17.67%)	108.06 (19.87%)
> 10 - 25	12.31 (9.16%)	46.27 (8.51%)
> 25 - 50	13.07 (9.72%)	44.32 (8.15%)
> 50 - 75	10.95 (8.15%)	33.12 (6.09%)
> 75 - 90	7.57 (5.63%)	20.28 (3.73%)
≥ 90	45.34 (33.73%)	96.98 (17.84%)
Total area burned (km^2)	134.43	543.71

Table 2.

Parameter	Mean	SD	Credible Interval		
			2.5	50	97.5
N_{2013}	39.28	6.4	27	38.5	51
N_{2014}	40.18	6.49	27	40	52
N_{2015}	24.54	3.75	17	24	31
N_{2016}	26.56	4.99	18	26	36
$\sigma_{2013,m}$	1696.25	179.79	1379.59	1683.42	2098.21
$\sigma_{2014,m}$	1637.15	173.87	1339.36	1625.29	2021.47
$\sigma_{2015,m}$	2165.82	170.72	1841.05	2166.08	2468.47
$\sigma_{2016,m}$	1859.35	151.63	1597.79	1845.41	2185.85
$\sigma_{2013,f}$	1155.3	248.65	783.6	1109.99	1791.55
$\sigma_{2014,f}$	1090.78	299.42	664.14	1040.91	1772.49
$\sigma_{2015,f}$	1395.98	258.36	1003.66	1351.35	1995.26
$\sigma_{2016,f}$	1111.76	235.9	757.08	1084.81	1683.52
$\alpha 0_{2013}$	-2.52	0.2	-2.93	-2.52	-2.11
$\alpha 0_{2014}$	-2.81	0.44	-4.11	-2.73	-2.19
$\alpha 0_{2015}$	-3.17	0.43	-4.32	-3.08	-2.57
$\alpha 0_{2016}$	-3.02	0.45	-4.27	-2.93	-2.39
$\alpha 1$	0.82	0.29	0.38	0.77	1.6
$\alpha 2$	-1.61	2.14	-6.37	-1.39	2.01
$\alpha 3$	-2.55	1.99	-7.41	-2.2	0.28
$\beta 0_{2013}$	-1.81	0.32	-2.41	-1.81	-1.16
$\beta 0_{2014}$	-1.71	0.34	-2.34	-1.72	-1
$\beta 0_{2015}$	-1.63	0.25	-2.1	-1.64	-1.12
$\beta 0_{2016}$	-1.61	0.26	-2.11	-1.62	-1.08
$\beta 1$	-0.52	0.4	-1.29	-0.53	0.24
$\beta 2$	2.27	0.24	1.81	2.26	2.77

Table 3.

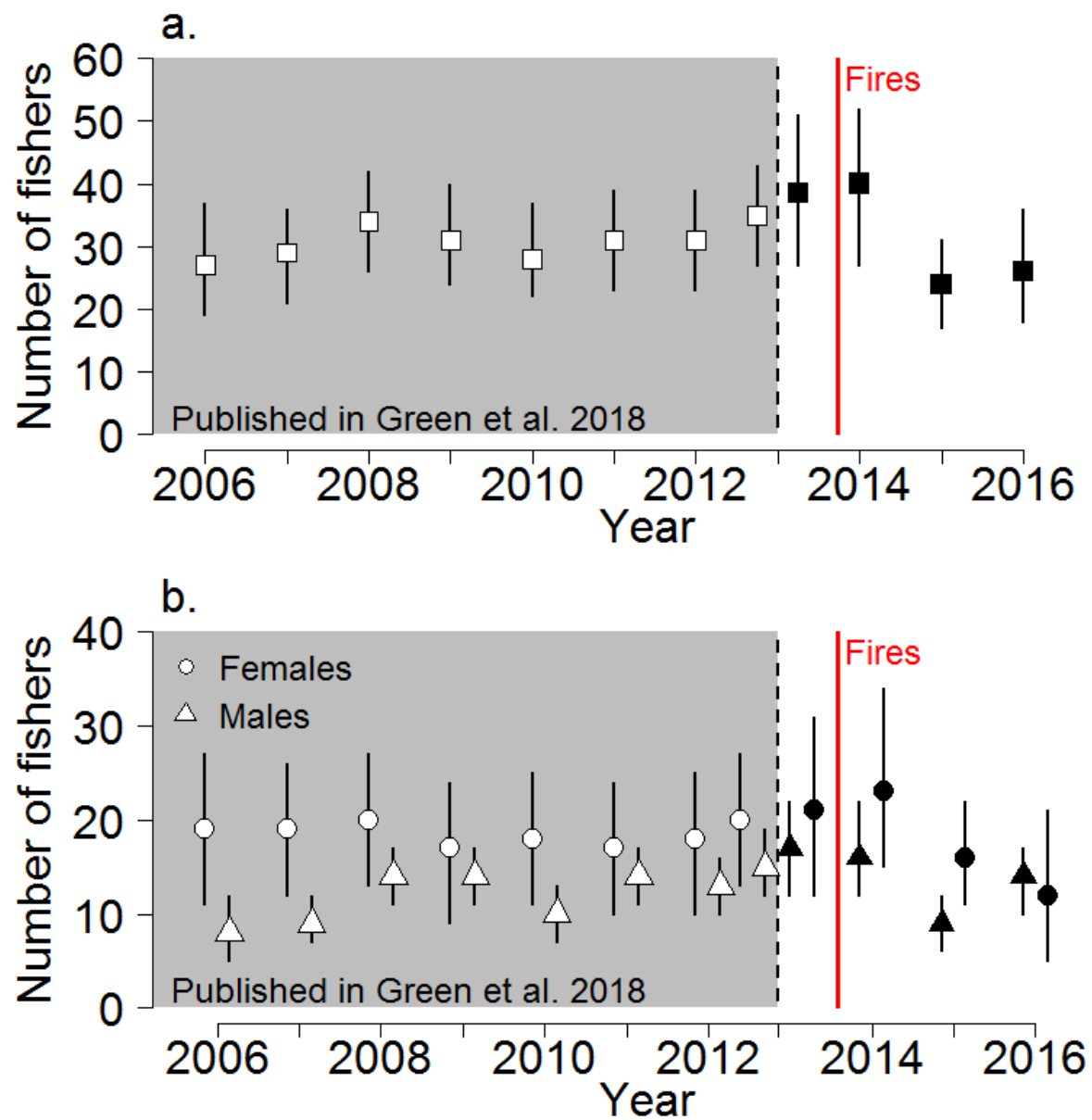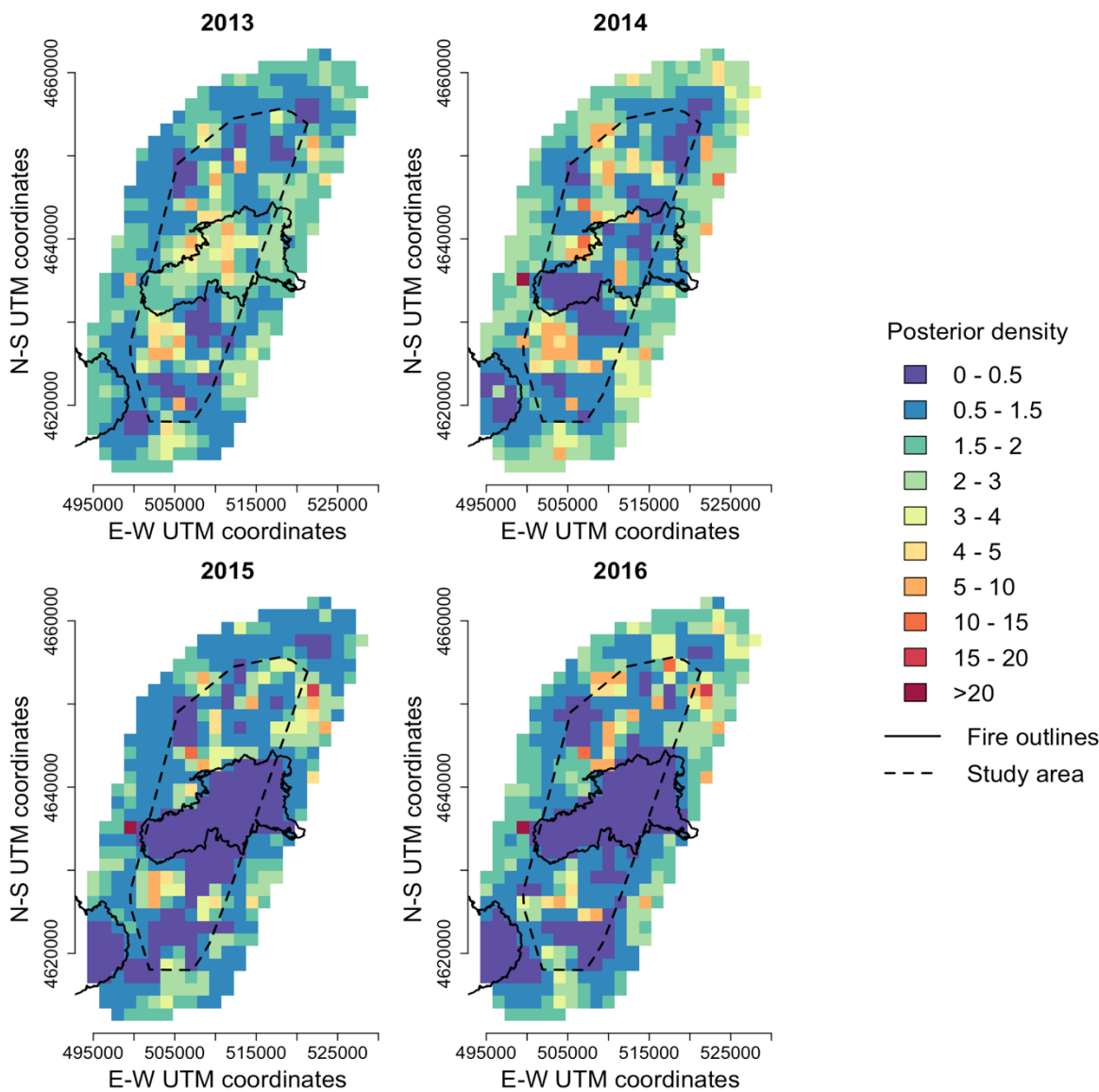| Year	Low severity	High severity
2015	930.8 km^2	200.3 km^2
2016	122.4 km^2	27.4 km^2
2017	1313.5 km^2	600.5 km^2
Total	1184.7 km^2	828.2 km^2

Figure 1.
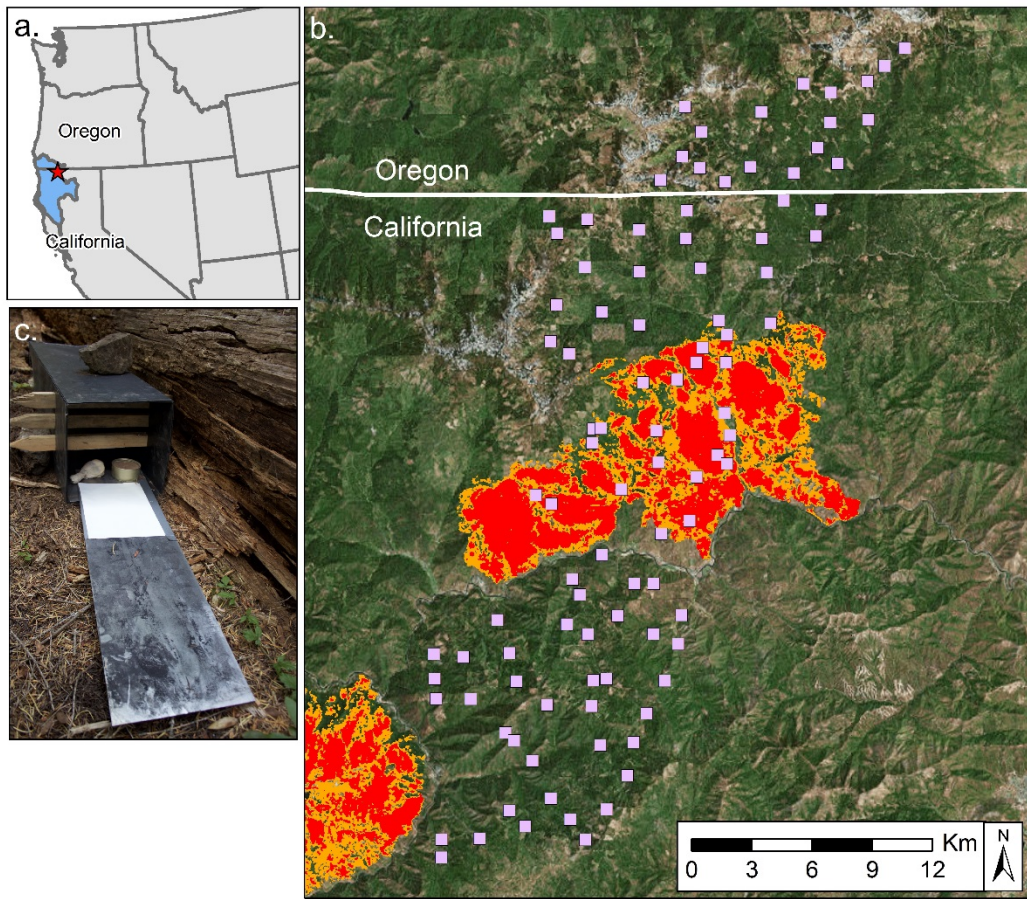
Figure 2.

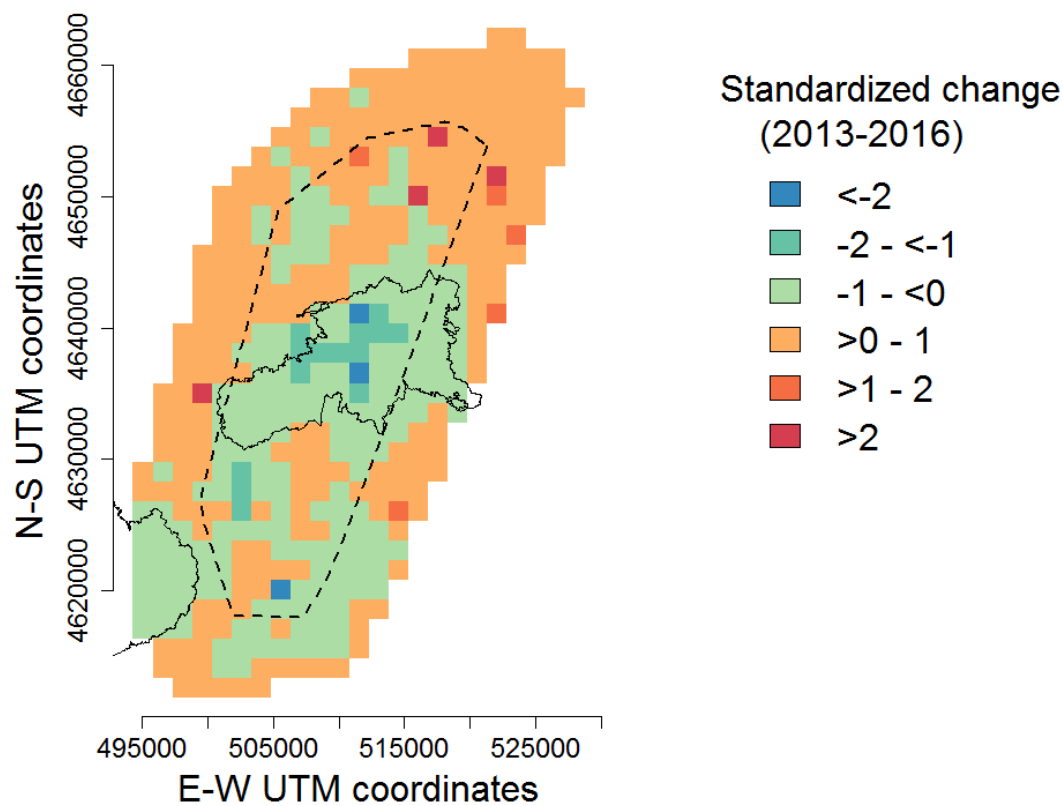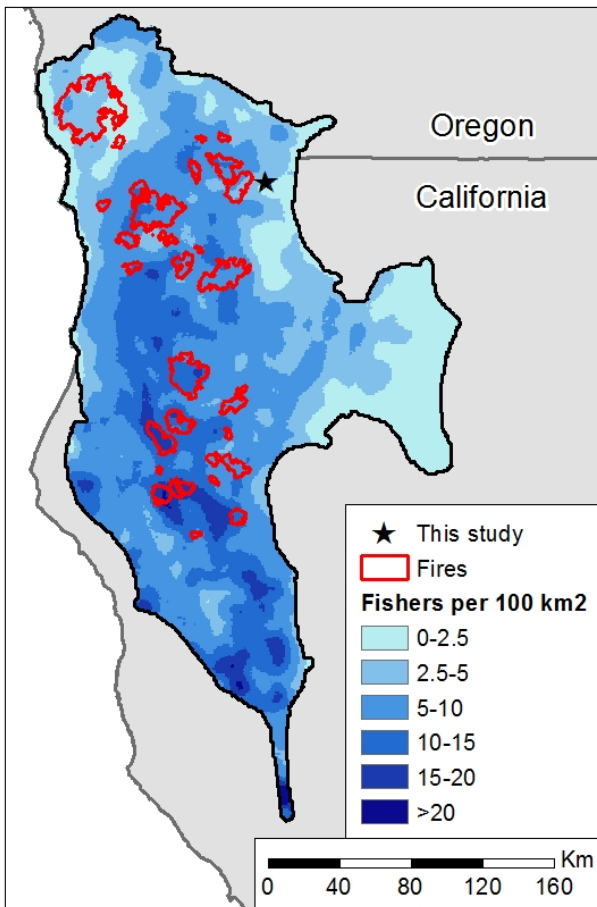**Figure 3.**

Figure 4.

Figure 5.

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