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David Green, Ph.D.
Institute for Natural Resources
Oregon State University
170 SW Waldo Place
Corvallis, OR 97331-8680
602/481-4524
david.green@oregonstate.edu

Title: Estimates of population growth, reproduction, and survival of a reintroduced population of fishers (*Pekania pennanti*) on a landscape managed for timber production

Running Head: Green et al. · Fisher reintroduction on commercial timberland

DAVID S. GREEN[1], *Institute for Natural Resources, Oregon State University, 170 SW Waldo Place, Corvallis, OR 97331-8680 USA*

AARON N. FACKA, *Institute for Natural Resources, Oregon State University, 170 SW Waldo Place, Corvallis, OR 97331-8680 USA*

KEVIN P. SMITH, *Department of Applied Ecology, North Carolina State University, Campus Box 7617, Raleigh, NC 27695-7617 USA*

SEAN M. MATTHEWS, *Institute for Natural Resources, Oregon State University, 170 SW Waldo Place, Corvallis, OR 97331-8680 USA*

ROGER A. POWELL, *Department of Applied Ecology, North Carolina State University, Campus Box 7617, Raleigh, NC 27695-7617 USA*

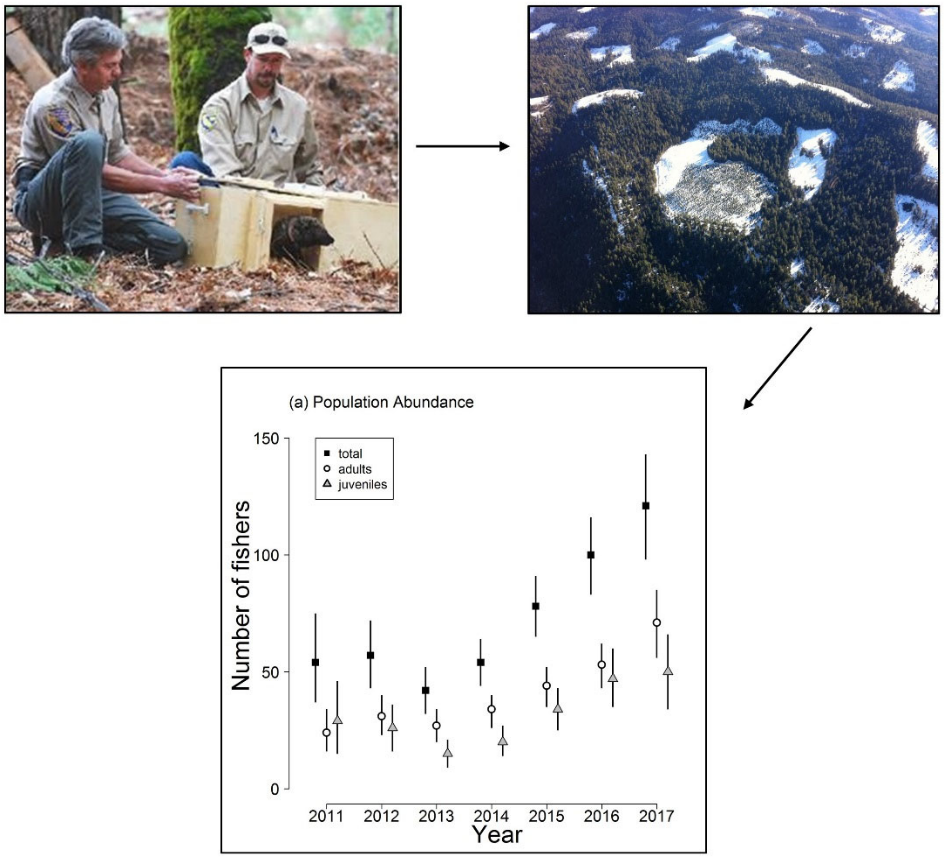
Highlights

- Fishers currently occupy 43% of their historic range
- Reintroduced fishers on commercial timber land yielded a growing population
- Density of the reintroduced population was equivalent to other populations

[1] Corresponding Author: david.green@oregonstate.edu

- Landscapes managed for timber will be important for future fisher conservation

Graphical abstract



Abstract

Understanding the role of landscapes managed for commercial timber production in the conservation of forest-dependent species is a priority for preserving ecological integrity and fostering socioeconomic wellbeing. The forest characteristics that are generally associated with the survival, reproduction, and persistence of forest-dependent species (e.g., large diameter trees, multi-dimensional structure, downed woody debris) are often believed to be at odds with forest management and timber production. The fisher (*Pekania pennanti*) is a mesocarnivoran associated with mature forests whose range has decreased substantially since the mid-1800s. Fishers represent the perceived conflict between forest-dependent species and forest management because they require areas of complex forest structure that provide sufficient prey, escape cover, and suitable structures for reproduction. Understanding if fishers can coexist with timber management will provide a critical test of the compatibility among forest-dependent species and landscapes managed for timber production. We reintroduced 40 fishers (24 females, 16 males) between November 2009 and December 2011 onto a landscape managed for timber production to establish a new fisher population and to evaluate the viability of fisher populations on forests commercially managed for timber production. We studied this reintroduced population of fishers for 8 years following the reintroduction using annual live-capture and year-round tracking with radio telemetry. Using population modeling with spatial capture-recapture, we estimated this population of fishers to be growing during the time of study. The density of the reintroduced fisher population in 2017 (10.8 fishers/100 km^2) was within the range fisher density across the western United States. The reintroduction of fishers to previously occupied portions of their distribution is a critical component of fisher conservation and will play a large role in the recovery of the species in western portions of its range. Our results highlight that forest conditions with a legacy of uneven-aged management and structural retention in forests managed for commercial timber production may be important for future fisher reintroductions.

Keywords: fisher, Jolly-Seber, *Pekania pennanti*, population monitoring, reintroduction, spatial capture-recapture, timber harvest

1. Introduction

Human activities related to forest management have substantially contributed to population declines of forest-obligate carnivores (Estes et al., 2011; Laliberte and Ripple, 2004; Ripple et al., 2014). Carnivore declines can lead to decreases in forest productivity and an overall loss of ecosystem integrity (Berger et al., 2001; Terborgh, 2001). Maintaining forests with late-successional characteristics (e.g., large diameter trees, multiple canopy layers, large volumes of dead wood) during timber harvest, fuels management, fire suppression, uncharacteristically-severe fire, and forest pests and pathogen outbreaks is a focal concern for conserving forest carnivores (Davis et al., 2015; Ruggiero et al., 1994). We are just beginning to understand the interactions and tolerances of forest-obligate predators in human-altered landscapes, and recovery of these species will require critically evaluating conservation measures available across forests managed for multiple uses and at varying intensities.

The fisher (*Pekania pennanti*) is a forest-obligate mustelid associated with late-successional, low- to mid-elevation coniferous and mixed conifer-hardwood forests in North America (Lofroth et al., 2010; Powell, 1993; USDI Fish and Wildlife Service, 2016). Fishers suffered significant range contractions and population declines before 1950 that were attributed to unregulated trapping for fur and habitat losses related to logging (Aubry and Lewis, 2003; Krohn, 2012; Powell, 1993; Powell et al., 2017). Fishers are now estimated to occupy approximately 43% of their historic range (Lewis et al., 2012). Conservation concern for fisher persistence resulted in closures of fisher trapping seasons (Aubry and Lewis, 2003; Lewis and Stinson, 1998; Powell, 1993), and fishers now appear to show signs of recovery in the eastern United States (LaPoint et al., 2015; Lewis et al., 2012). Recovery of fishers in the western United States, however, has not been as successful, leading the U.S. Fish and Wildlife Service to propose listing the distinct population segment of fisher in California and Oregon as threatened (U.S. Fish and Wildlife Service 2019). The mechanisms for the lack of fisher recolonization in the western United States remain unknown but hypotheses include suppression by larger carnivores (LaPoint et al., 2015; Gabriel et al., 2015; Wengert et al., 2014), competition with sympatric mesopredators (Green et al., 2018), and current timber management practices being incompatible with fishers (Aubry and Houston, 1992; Lewis and Stinson, 1998; Powell, 1993; Powell and Zielinski, 1994).

Understanding if fishers can persist and thrive in forests that are managed for commercial timber production remains an important area of research. Critical evaluations of fisher population viability on landscapes of varying management intensities have received surprisingly little attention (Facka, 2017; Raley et al., 2012; USDI Fish and Wildlife Service, 2016). Managed forests will play an important role in fisher recovery; forested habitat in the western United States available for fisher recolonization include landscapes managed for timber harvest (Callas and Figura, 2008). Fishers occupy and reproduce in these forests (Facka et al., 2016; Matthews et al., 2013; J. L. Thompson, 2008) but the history, magnitude, and intensity of harvest of late-successional forests were also one of the primary causes for fisher declines across North America (Douglas and Strickland, 1987; Powell, 1993; Powell and Zielinski, 1994). Thus, it is important to understand if fisher populations can coexist with silvicultural systems used in commercial timber production.

Reintroductions have been pivotal in the recovery of several fisher populations (Lewis et al., 2012) and they are likely to continue in their scope and distribution over time. Fishers have been reintroduced more than 30 times throughout North America, with 3 occurring in Washington state since 2008 (Happe et al., 2016; Lewis, et al., 2013; Powell et al., 2017). Reintroduction projects, however, often fail to develop, to evaluate empirically, or to implement long-term monitoring programs identifying and measuring defensible, *a priori* metrics of reintroduction success (Fischer and Lindenmayer, 2000). For example, Lewis et al. (2012) reported that 26 of 30 (86%) fisher reintroductions they evaluated had known outcomes, but outcomes were often based on qualitative, post-hoc determinations made by the resource agency responsible. Discrepancies in the successes of fisher reintroductions have also varied by geography; nearly twice as many reintroductions have been successful in eastern North America compared to western North America (Lewis et al., 2012). Differences in forest composition and timber practices have been proposed as contributing to the observed disparity in reintroduction success across North America (Buskirk and Powell, 1994; Lewis et al., 2012), but evidence to support this hypothesis is lacking. The act of reintroduction itself offers a rare opportunity to test hypotheses involving fishers and managed forests, and performing a reintroduction of fishers with adequate post-release monitoring on a managed forest could help shine light on this important concept (Facka 2017).

We used a reintroduction of fishers to a privately-owned forest managed for commercial timber production in northern California to evaluate the viability of fishers on landscapes managed for timber production. Intensive post-release monitoring of this population took place using annual live-capture and year-round monitoring with radio telemetry. In the current study, we developed a spatial capture-recapture population model using the data from the capture effort and radio telemetry to estimate population growth, reproduction, and survival of the reintroduced population. We then compared the estimated demography of this reintroduced population with other fisher populations to evaluate the success of this reintroduction and to determine if fishers can persist on managed landscapes in California.

2. Methods

2.1 Study Site and Fisher Reintroduction

We reintroduced fishers to an unoccupied portion of the species' historic range in the northern Sierra Nevada of California (Figure 1), the Stirling Management Area of Sierra Pacific Industries (henceforth, "Stirling"; Callas and Figura 2008; Facka et al. 2016) in portions of Plumas, Butte, and Tehama counties. Sierra Pacific Industries logged approximately 600 ha annually using even-age management. Two main goals of this reintroduction were to 1) establish a new population of fishers in California, and 2) evaluate the viability of fisher populations on forests commercially managed for timber production. We released 15 fishers (9 females, 6 males) in December 2009 and January 2010, 13 fishers (7 females, 6 males) in November 2010 through February 2011, and 12 fishers (8 females, 4 males) in November and December 2011 (Facka 2016, Facka et al. 2016). Further information regarding the specifics of the reintroduction on Stirling and Stirling's ecology are reported in Facka et al. (2016).

<u>2.2 Monitoring fishers using annual live-capture and telemetry</u>

We monitored the reintroduced population of fishers on Stirling through 2017. During 2011 through 2017, we placed 204 ± 54 [83, 239] (annual mean ± SD; [min, max]) traps that were spaced 972 ± 422 [106, 2982] m apart, and left them open for 28 days; one 14-day interval on the east side and one 14-day interval on the west side of Stirling (Figure 1). We used Tomahawk live traps (model 207, Tomahawk Live Trap Company, Tomahawk, WI) and other similar wire mesh box traps, each modified to include a wooden cubby box to reduce stress during capture (Seglund, 1995; Wilbert, 1992). We baited traps with a raw chicken drumstick and an olfactory lure (Gusto, Minnesota Trapline Products, Pennock, Minnesota, USA), and checked them daily. We released non-target animals when found in traps.

Upon their first capture, we immobilized fishers with Tiletamine HCL and Zolazepam HCL (7mg/kg; Telazol®, Fort Dodge Animal Health, Fort Dodge, Iowa, USA) using methods that were approved by the Institutional Animal Care and Use Committee from North Carolina State University (09-007-O). Under anesthesia, each fisher received a Passive Integrated Transponder tag (134.2 kHz Super Tag; Sterile, Biomark, Inc., Boise, Idaho) for future identification, and we collected data on sex, reproductive status, physiological condition, disease exposure, weight, and morphology. We collected feces for diet analyses, blood to check for diseases and for DNA identification, and an upper premolar to estimate age using cementum annuli (Arthur et al., 1992). Henceforth, we refer to all animals born the year of the survey (i.e.,

~7 months old at the time of capture) to be juveniles, and all others to be adults. We outfitted a subset of fishers with Very High Frequency telemetry collars (VHF; Telonics IMP-325 or MOD-125; Mesa, Arizona or Holohil MI-2i, Carp, Ontario, Canada), Global Positioning System collars (Minitrack GPS Lotek Wireless, Newmarket, Ontario, Canada) or Argos collars (Kiwisat 202 or 303 Platform Terminal Transmitters, Sirtrack, Havelock North, New Zealand) to monitor their movements on Stirling. We monitored all fishers daily and located fishers with VHF transmitters by triangulation, using the program 'Location of a Signal' (LOAS 4.0, Ecological Software Solutions LLC, Hegymagas, Hungary). We collected most locations of fishers with VHF collars using ground techniques, although we occasionally used fixed-wing and helicopter surveys.

2.3 Spatial capture recapture model with live-capture and telemetry data

We used Spatial Capture-Recapture (Efford, 2004; Royle and Young, 2008) to estimate fisher density on Stirling annually. Most applications of spatial capture-recapture arise from sampling that is conducted using the Robust Design (Pollock, 1982), for which data are collected over a series of secondary periods (e.g., weeks) spanning >1 primary periods (e.g., years). Individuals can be detected at >1 monitoring device during a single secondary period (e.g., hair-snare, infrared trail camera). These individual encounter histories are then modeled as latent random variables arising from independent Bernoulli, Poisson, or Binomial trials (Royle et al., 2014). In the current study, we monitored fishers using live-traps and, thus, individuals could only be captured at ≤1 site during each secondary period. We initially fit our spatial capture-recapture model using a multinomial distribution to accommodate this structural design (Distiller and Borchers, 2015; Romairone et al., 2018), but this was computationally intensive to run and we were unable to proceed with it. Thus, we ultimately developed a spatial capture-recapture model that fit the capture histories for fishers using a Bernoulli distribution and an open population model (e.g., Green et al. 2018; Gardner et al. 2010).

We modeled whether or not individual i was ever captured in year t at trap j (y_{itj}) as random variables that were a function of the location of an individuals' latent activity center in that year (s_{it}), such that $\Pr(y_{itj} = 1|s_{it})$, in addition to other variables described below. The capture efforts occurred annually in October; we estimated activity centers for the period of October to September as a homogeneous Poisson point process in the state-space S (Royle et al., 2014). We modeled the capture history as:

$$y_{itj} \sim \text{Bernoulli}(\lambda_{itj} g_{itj} z_{it} m_{jt}),$$

where the detection or non-detection of individual i in year t at trap j was a function of the average probability of capture (λ_{itj}), the distance between trap locations and its activity center that year (g_{itj}), and the partially-observed latent variable of population membership in that year (z_{it}). Not all traps were open each year, so we also included a binary variable indicating whether or not trap j was open in year t to account for this (m_{jt}). Male fishers often have a higher capture probability than female fishers (e.g., Green et al., 2018; Linden et al., 2017; Popescu et al., 2014), and not every trap was open each year for the full 14 days. Thus, we modeled λ_{itj} as:

$$\text{Logit}(\lambda_{itj}) = \beta 0 + \beta 1 * \text{sex}_i + \beta 2 * \text{daysopened}_{tj}$$

where the average capture probability is a function of a intercept (β0), and the linear effects of sex (β1) and the number of days that trap j was opened in year t (β2). We modeled g_{itj} as:

$$g_{itj} = \exp(-d_{ijt}^2 / 2\sigma_x^2)$$

where an individual's capture probability is a function of the Euclidean distance between the trap locations and the location of its activity center that year (d_{ijt}^2) and the *SD* of a bivariate normal distribution reflecting space-use that we calculated independently for each sex x ("movement parameter"; σ_x). We incorporated the location data collected through radio telemetry to increase the precision of our estimates for the movement parameters and the location of activity centers (Linnell et al., 2018; Royle et al., 2013; Sollmann et al., 2013). All x and y coordinates from the telemetry locations of fisher i in year t ($L_{it,xy}$) were modeled as the Bivariate normal random variable:

$$L_{it,xy} \sim \text{Normal}(s_{it}, 1/\sigma_x^2),$$

where telemetry locations were a function of an individual's activity center in that year and the sex-specific estimated movement parameter σ. To ensure independence of telemetry observations (Sollmann et al., 2013), and for computational efficiency we only used locations that were separated by >2 weeks. We also only used locations collected from November through September to ensure independence of the telemetry locations from any activities associated with the capture efforts.

We fit our models using data augmentation (Royle and Dorazio, 2008; Royle and Young, 2008) by introducing a sufficient number of non-captured individuals to our capture dataset. The partially-latent variable z_{it} indicated population membership for individual i in year t. We had known birth years for all individuals that were reintroduced to Stirling, and for all animals that were captured in live-traps. The oldest animal reintroduced to Stirling was born in 2004; we coded z_{it} to run from 2004 (t = 1) and continue through 2017. We set $z_{i1} \sim$ Bernoulli(Ψ), where Ψ is equal to the number of reintroduced animals that were born in 2004 divided by the total number of individuals monitored (captured or augmented). For years t > 1, z_{it} was modeled as the Bernoulli random variable:

$$z_{it} \sim \text{Bernoulli}(\mu_{it})$$

$$\mu_{it} = (z_{it-1} * \Phi_t) + (A_{it-1} * \Gamma_t),$$

where the probability that an individual is estimated to be in the population is a function of survival from the previous year if they have already been born ($z_{it-1} * \Phi_t$) or the probability that an individual is born into the population if they have not already been born ($A_{it-1} * \Gamma_t$). The A_{it-1} term ensures that animals can only be born into the population once during the study:

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

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

We coded known birth years from captures and known death years from observed mortalities into z_{it}. We modeled sex of all individuals as the Bernoulli distributed partially latent variable, $\text{Sex}_i \sim \text{Bernoulli}(\Psi_{sex})$, where sex was estimated as coming from the population level sex-ratio (Ψ_{sex}).

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 30,000 MCMC samples, taken from 3 chains run for 100,000 iterations, thinned by 10, and following a burn-in of 10,000. We assessed model convergence by examining trace plots and $\hat{R}$ values for convergence (Gelman et al., 2013; 1996). We used posterior distributions to calculate percent probabilities, defined as the proportion of draws greater than or less than 0. All statistical code to perform the spatial capture-recapture analysis is included in Supplement 1. Population size and density each year were calculated as a function of the estimated number of activity centers in the Stirling study area, which was defined as the minimum concave polygon encompassing the locations of our live-traps buffered by the upper 95% Credible Interval for the movement parameter σ for males (Stirling study area = 1102 km^2).

2.4 Review of fisher density, population numbers, and demographic rates

We performed a literature review to determine how the density and survival estimates of the reintroduced fisher population on Stirling compared to other populations of fishers in the western United States. Using Web of Science, we searched for all published articles that included the keywords “fisher” or “*pennanti*” and “density”.

3. Results

Our live traps were open for 16,942 sampling nights between 2011 and 2017 (yearly mean ± SD = 2,420 ± 748). A total of 165 individual fishers (36.7 ± 17.1) were captured and identified from 390 captures (55.7 ± 28.4). The number of individual fishers that we captured increased over time (total number of unique captures each year: 2011 = 14, 2012 = 29, 2013 = 22, 2014 = 32, 2015 = 46, 2016 = 53, and 2017 = 61). Fishers were captured in 1.5 ± 0.2 different traps each year. We radio-collared 67 fishers (21 males and 46 females), and collected 421 total locations that were used in the current analysis (mean number of locations ± SD per fisher separated by 2 weeks: 6.3 ± 4.3). We also regularly captured other mesocarnivorans in our live-traps; in descending order of frequency we captured spotted skunk (*Spilogale gracilis*) on 221 occasions (yearly mean ± SD: 31.6 ± 17.4), ringtail (*Bassariscus astutus*) on 158 occasions (22.6 ± 9.6), gray fox (*Urocyon cinereoargenteus*) on 154 occasions (25.7 ± 20.4), opossum (*Didelphis virginiana*) on 62 occasions (8.9 ± 7.5), and striped skunk (*Mephitis mephitis*) on 10 occasions (2 ± 1).

Our spatial capture-recapture model estimated that the population of fishers on Stirling increased from 2012 to 2017, despite a decrease in the total number of fishers from 2012 to 2013 (Figure 2). Males were 99% less likely than females to be captured in live-traps and, unsurprisingly, the number of days that a trap was opened each year had a 100% probability of increasing trap success (Table 1). The movement parameters for fishers were estimated to vary significantly by sex; female sigma values were equivalent to a 11 km^2 home range (mean ± SD: 1872.11 ± 53.94 m), and male sigma values were equivalent to a 25 km^2 home range (mean ± SD: 2797.65 ± 99.68 m). The distribution of fishers on Stirling varied among years (Figure 3). By modeling the year that each fisher was born, we were able to construct an estimated age structure on Stirling (Figure 4). We estimated a mean density of 10.8 fishers per 100 km^2 on Stirling in 2017. This density is similar to the estimated densities of fishers in the literature

(Table 2). The probability of an individual being male was estimated to be (mean ± SD [95% CI]) 0.35 ± 0.04 [0.28, 0.43], suggesting a female-biased sex ratio.

4. Discussion

The reintroduction of fishers to Stirling yielded a growing population by the end of the current study. The number of fishers estimated to reside on Stirling increased every year from 2013 to 2017. By the end of our study, estimated recruitment and the survival probability of fishers had stabilized and were quite high, indicating that this new population of fishers may be self-sustaining. The establishment of a growing population of fishers following reintroduction is an important first step to understanding how fishers and timber management need not be mutually exclusive.

Although our annual estimates of fisher density indicated an upward trend, we did detect what appeared to be a small decline in fisher numbers from 2012 to 2013 (Figure 2a). We suspect this decline was related to low fecundity or juvenile survival in 2013. The overall probability of survival was lower in 2013 compared to prior and subsequent years (Figure 2a), and there were fewer juveniles estimated to be on Stirling in 2013, compared to 2011 and 2012 (Figure 4). It is also possible that this may have been due to a decrease in the number of adult male fishers (Figure 2b), or that this was the first year when we were not augmenting the population with translocated fishers. High post-release mortality, among other characteristics of the founding individuals and their progeny, have direct implications on population growth, the probability of establishment, and the risk of inbreeding and loss of genetic variation (Armstrong and Seddon, 2008; Seddon et al., 2014). The small population on Stirling appeared resilient to the relatively high level of mortality in 2013 and continued to grow to establishment in 2017. Continued monitoring of this fisher population will provide the best opportunity to investigate population persistence and to identify any potential drivers of population change in the future (e.g., losses in genetic variation; Ewen et al., 2012).

Our implementation of a spatial capture-recapture population model allowed us to incorporate many unique aspects of the work completed on Stirling that have implications for other monitoring efforts. Fishers on Stirling were monitored using annual live-capture efforts and year-round telemetry efforts. By formally linking capture histories across years, we were able to use a population model to monitor survival and recruitment over time. Our application of a spatial capture-recapture model offers a robust framework for future reintroduction efforts to determine outcomes and inform management strategies if reintroduced animals were unable to establish a self-sustaining population (Royle et al., 2017). An important caveat is that it can be computationally intensive to use population models in a spatial capture-recapture framework (i.e., our models took 30 days to run), and this may limit future implementations.

The annual trapping effort here on Stirling was not designed to be modeled using a spatial capture-recapture framework, and this presented many challenges during model development. Many of these problems were because the main goal of our trapping efforts were to deploy collars to monitor fishers on Stirling year-round using telemetry. We had low capture probabilities for fishers in our live-traps and very few individuals were captured at more than one different trap (mean number of traps individuals were captured at = 1.5). These two pieces of data are critical for monitoring populations with spatial capture-recapture (Royle et al., 2014), and research should be designed to maximize capture probabilities and distribution of captures for studies using spatial capture-recapture. To overcome these limitations, we incorporated the telemetry data to inform our estimates of the movement parameter. In simulations without the telemetry data, we could not calculate reliable estimates of the fisher population on Stirling over time. Furthermore, the effort to monitor fishers on Stirling using live-capture and telemetry was expensive; it cost $1,415,010 to monitor this population of fishers on Stirling for the 6 years we have presented here. This sum of money is unlikely to be an option for future monitoring efforts on Stirling, and recent developments in non-invasive monitoring with genetics are a cost-effective and viable alternative to use in the future (Green et al., 2018; Linden et al., 2017). Some insights into population characteristics and threats (e.g., age structure over time, disease monitoring), however, are simply not available using non-invasive monitoring and current genetic tools for fishers.

The reintroduction of fishers to the commercially-managed forests of Stirling offered an opportunity to evaluate the compatibility of fishers and timber management and to understand the role that these types of forests might play in future conservation actions for fisher. Although forest management was hypothesized to prevent fishers from reoccupying portions of their former range in the western United States (Aubry and Houston, 1992; Lewis and Stinson, 1998; Powell, 1993; Powell and Zielinski, 1994; Powell et al., 2017), many of the forests that fishers now occupy are a mosaic of plant communities and seral stages that include the mid- to late-seral forests that are representative of legacy timber production (Raley et al., 2012). Our estimates of fisher density and survival on Stirling were consistent with those reported by other studies representing a range of management histories, logging intensities, and seral stages throughout California (Table 2). Thus, current forest conditions on Stirling appear to have the capacity to support a population of fishers. The recolonization of fishers to unoccupied portions of their former range are unlikely to occur as a function of dispersal (Aubry and Lewis, 2003; Lewis and Stinson, 1998), indicating that a series of reintroductions and post-release monitoring efforts may be needed in the western United States. Our results indicate that commercially-managed ownerships with forest conditions similar to Stirling have the capacity to support fishers and that public-private partnerships for reintroduction efforts will play an important role in their recovery in the western United States.

5. Conclusions

Here we monitored the outcome of a reintroduction of 40 fishers to a landscape managed for timber production in northern California. The main results from our study are as follows:

(1) Fishers were reintroduced to a managed landscape.

(2) Forest conditions on this managed landscape were suitable for the establishment of a fisher population.

(3) The reintroduced population appeared to decline 3 years after the reintroduction, but increased annually in years 4 through 7.

(4) This is one study and should not be generalized to all managed landscapes. More research is needed to ensure the results from Stirling are comparable to other forests managed for commercial timber production with different management histories and contemporary forest conditions.

Previous research has identified timber harvest to be an important factor limiting fisher recolonization in the western United States, and although this is likely an important factor, our results indicate that other limiting factors require additional attention.

Acknowledgments

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Tables and Figures

Table 1. Derived posterior parameter estimates of annual population abundance (*N*) and density of fishers (D) in the Stirling study area, sex specific movement parameters in m (σ), the intercept for the capture probability (β0), the effects of sex (β1) and number of trapping days (β2) on capture probability, year specific recruitment rates (Γ), and year specific survival probabilities (Φ). We separate *N* by the total population (*T*) and the number of adults (i.e., animals > 1 year of age; *A*).

Parameter	Mean	SD	Credible Interval		
			2.50	**50.00**	**97.50**
NT_{2011}	55.23	10.64	37.00	54.00	77.00
NT_{2012}	57.92	7.55	42.00	57.00	71.00
NT_{2013}	42.71	5.01	33.00	42.00	52.00
NT_{2014}	54.10	5.37	44.00	54.00	64.00
NT_{2015}	78.30	6.79	66.00	78.00	92.00
NT_{2016}	99.31	8.45	82.00	99.00	115.00
NT_{2017}	119.65	11.51	96.00	119.00	141.00
NA_{2011}	25.01	4.91	16.00	24.00	34.00
NA_{2012}	31.63	4.64	22.00	31.00	40.00
NA_{2013}	27.65	3.55	21.00	27.00	34.00
NA_{2014}	33.87	3.71	26.00	34.00	40.00
NA_{2015}	43.79	4.49	35.00	44.00	52.00
NA_{2016}	52.48	5.02	43.00	52.00	62.00
NA_{2017}	70.18	7.51	55.00	70.00	84.00
D_{2011}	4.98	0.96	3.34	4.87	6.94
D_{2012}	5.22	0.68	3.79	5.14	6.40
D_{2013}	3.85	0.45	2.98	3.79	4.69
D_{2014}	4.88	0.48	3.97	4.87	5.77
D_{2015}	7.06	0.61	6.04	7.03	8.38
D_{2016}	8.95	0.76	7.39	8.93	10.37
D_{2017}	10.79	1.04	8.66	10.73	12.71
σ_f	1872.11	53.94	1769.38	1872.02	1982.00
σ_m	2797.65	99.68	2598.40	2802.69	2978.81
$\beta 0$	-3.32	0.32	-3.97	-3.31	-2.74
$\beta 1$	-0.41	0.17	-0.74	-0.41	-0.07

$\beta 2$	0.11	0.02	0.06	0.11	0.16
Γ_{2011}	0.11	0.04	0.05	0.11	0.21
Γ_{2012}	0.11	0.03	0.06	0.11	0.18
Γ_{2013}	0.08	0.02	0.04	0.08	0.13
Γ_{2014}	0.10	0.03	0.06	0.10	0.16
Γ_{2015}	0.18	0.04	0.11	0.18	0.27
Γ_{2016}	0.34	0.07	0.22	0.34	0.50
Γ_{2017}	0.59	0.16	0.33	0.58	0.93
Φ_{2011}	0.67	0.10	0.46	0.67	0.86
Φ_{2012}	0.55	0.11	0.33	0.55	0.77
Φ_{2013}	0.46	0.10	0.28	0.46	0.67
Φ_{2014}	0.69	0.10	0.49	0.69	0.87
Φ_{2015}	0.80	0.09	0.61	0.81	0.95
Φ_{2016}	0.66	0.07	0.51	0.66	0.81
Φ_{2017}	0.71	0.08	0.55	0.72	0.86

Table 2. Mean (Standard Deviation) fisher density and survival reported in the literature from portions of California and Oregon. “Population” indicates the portion of the west coast distinct population segment of fishers in California and Oregon where the study was performed (e.g., Northern California and Southern Oregon [NCSO], Southern Sierra).

Density (fishers/100 km^2)	Survival	Population	Study
7.9	-	NCSO	(Buck, 1982)
44.0	-	NCSO	(Buck et al., 1983)
6.6	-	NCSO	(Furnas et al., 2017)
6.69 (0.59)	0.8 (0.05)	NCSO	(Green et al., 2018)
33 (26.87)	-	NCSO	(Matthews et al., 2011)
8.0	-	NCSO	(Mullis, 1985)
10.97 (2.12)	0.88	Southern Sierra	(Jordan, 2007)
7.97 (2.55)	0.94 (0.01)	Southern Sierra	(Jordan et al., 2011)
8.5 (1.94)	0.75 (0.11)	Southern Sierra	(Sweitzer et al., 2015)
9.5 (3.83)	0.94 (0.09)	Southern Sierra	(J. L. Thompson, 2008)
12.4	-	Southern Sierra	(C. M. Thompson et al., 2012)
10.79 (1.04)	0.71 (0.08)	NCSO	This study

Figure 1. Sites where fishers (*Pekania pennanti*) were reintroduced between 2009 and 2011 to the Stirling Management Area, owned and managed for commercial timber production by Sierra Pacific Industries. Portions of Stirling with ≤40% canopy cover, depicted in orange, were usually the result of even-aged clearcut harvesting, the rehabilitation of brush fields to conifer trees, or fire salvage harvesting of substantially damaged timberlands. These younger forests grow into forests with >40% canopy cover, represented in green, between 15 and 30 years following harvest and may involve pre-commercial thinning at 8 to 15 years following harvest. The locations where fishers were released are indicated with purple triangles, and the locations of live-trapping locations are noted with blue circles.

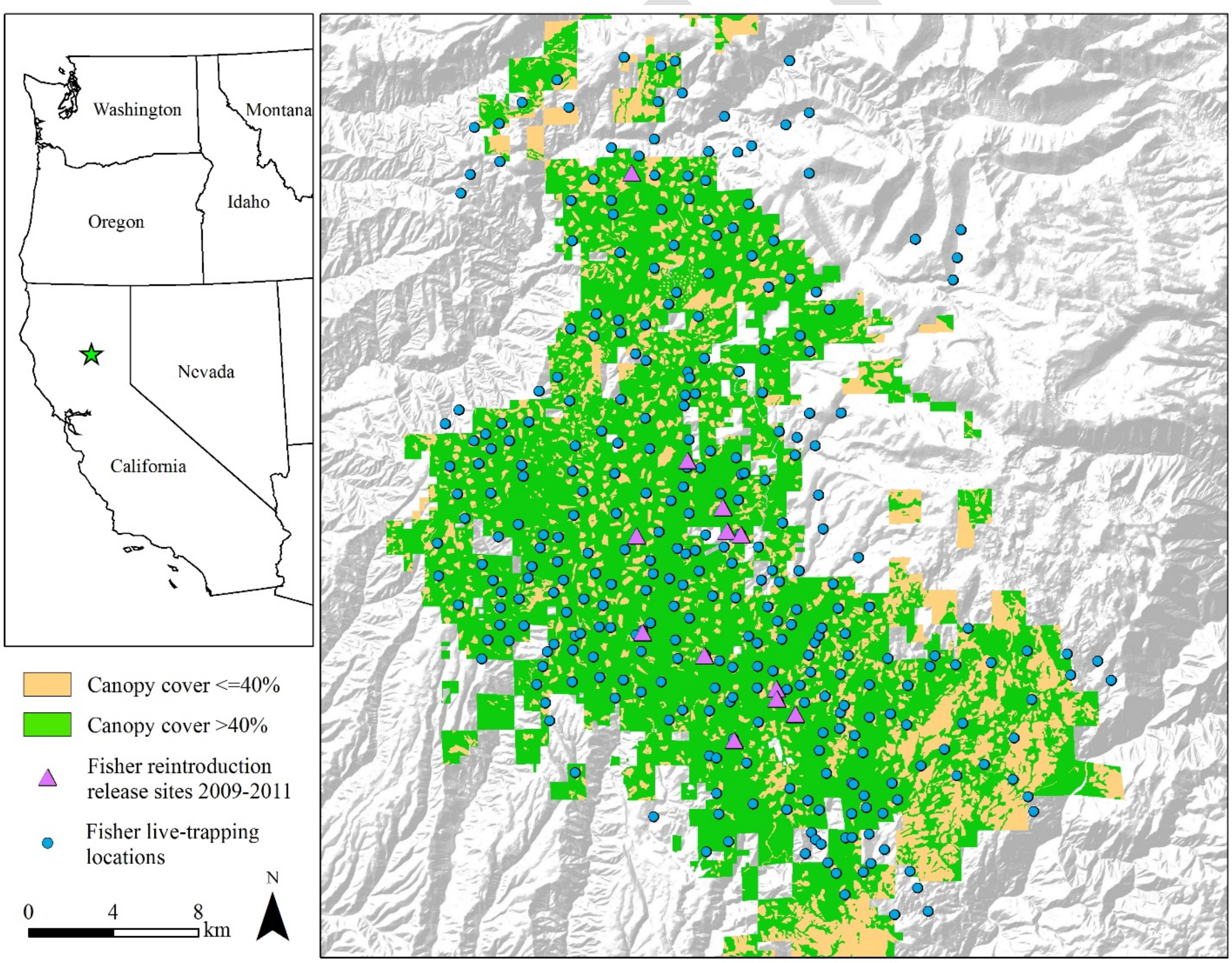


Figure 2. The estimated median and 95% credible intervals of (a) population abundance, (b) the number of adult fishers separated by sex, and (c) the survival and recruitment probabilities over time on Stirling estimated by a spatial capture-recapture population model.

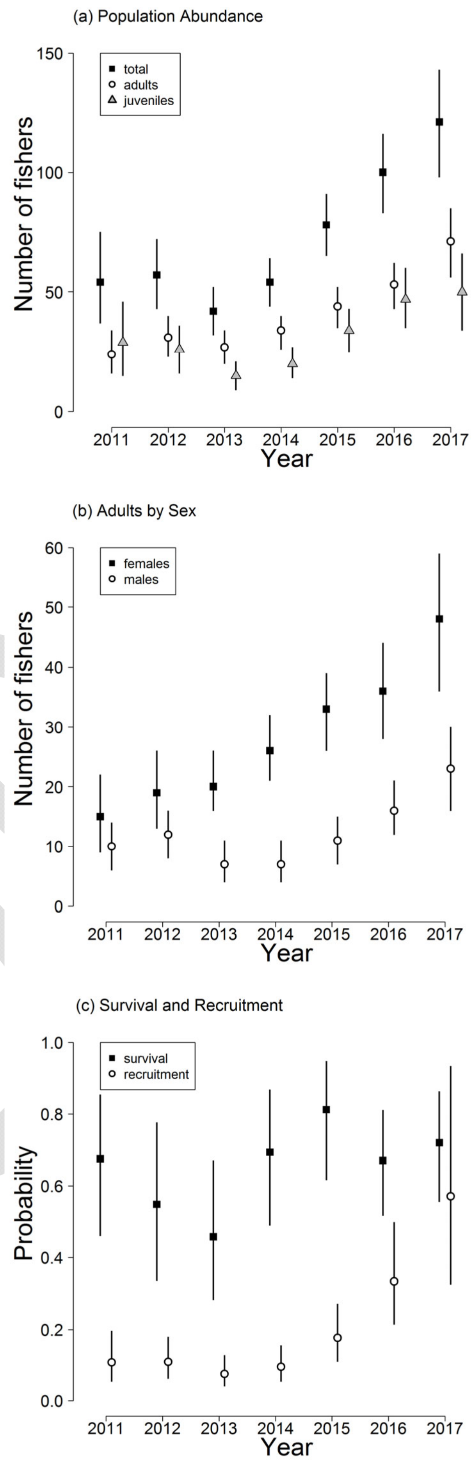


Figure 3. The mean estimated distribution of fishers on Stirling over time. Dark green and dark purple indicates comparatively fewer and more fishers in the 1 km^2 grid cell in that year, respectively.

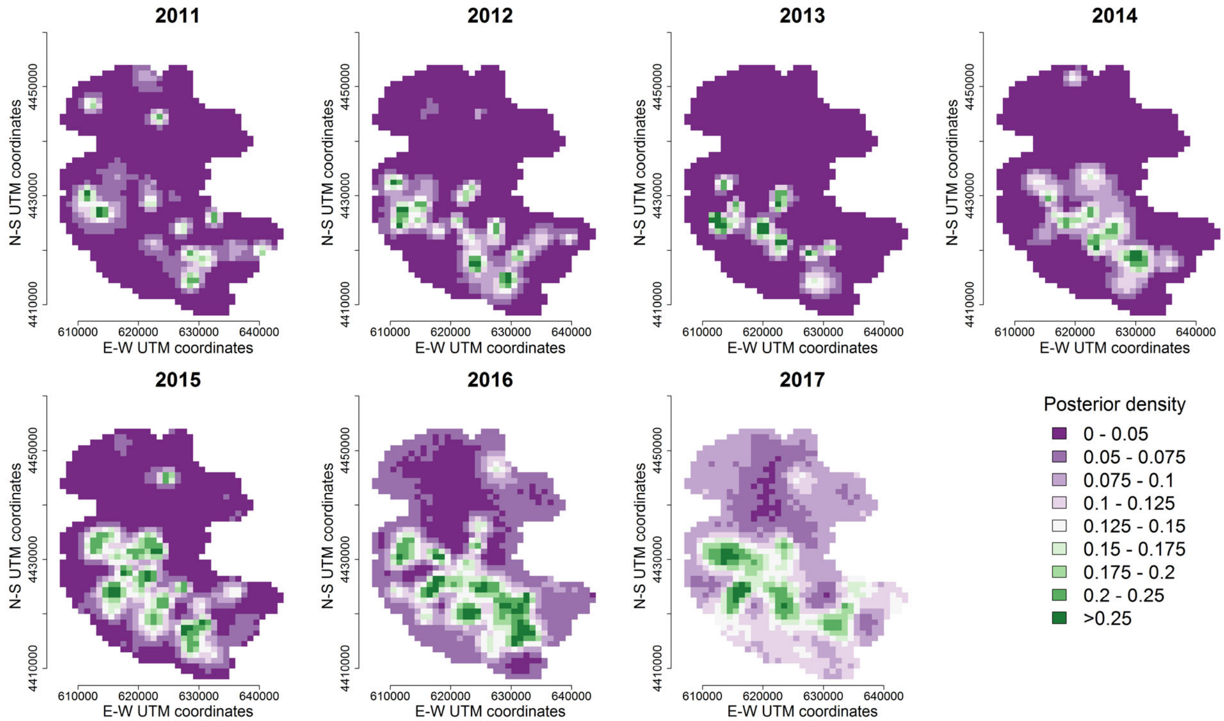
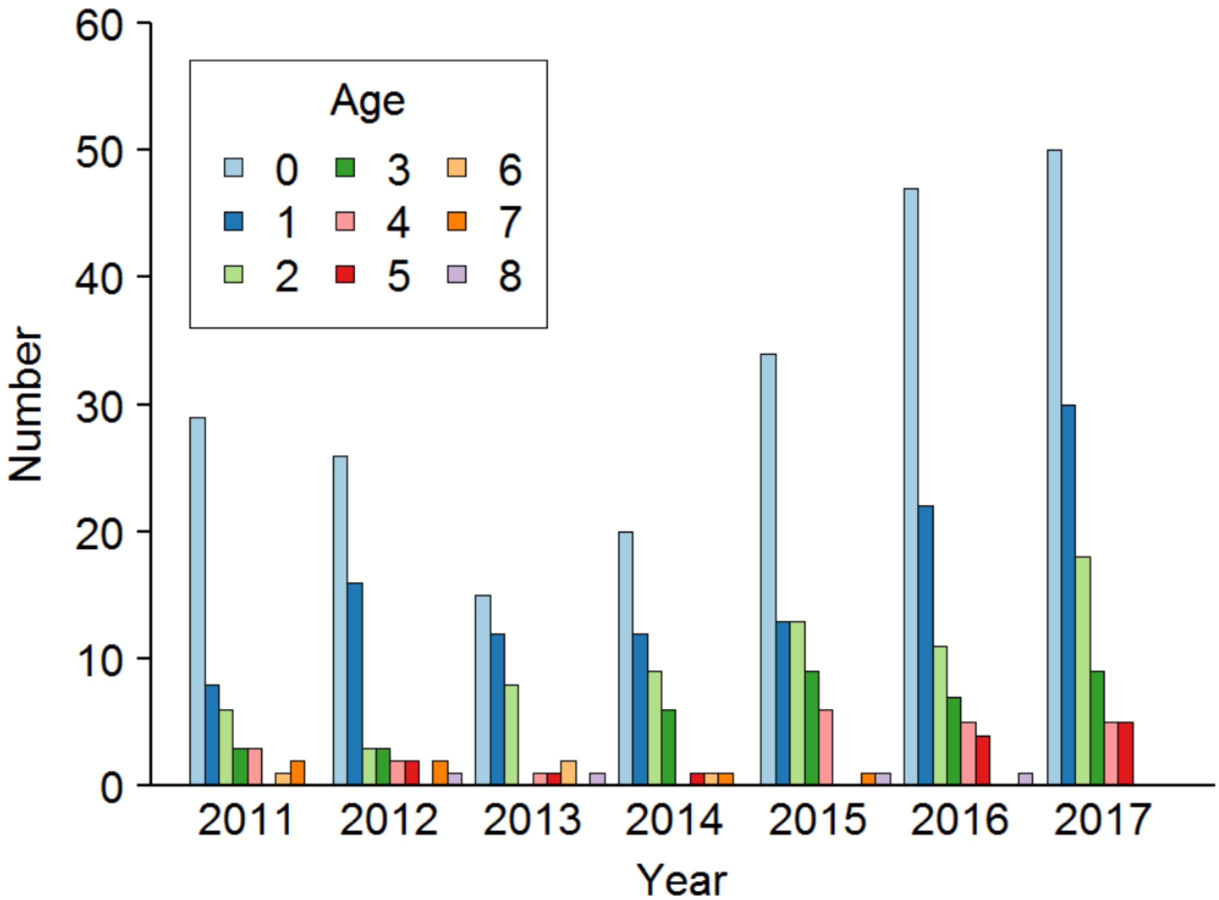


Figure 4. The estimated median age structure of fishers on Stirling over time as estimated by spatial capture-recapture population modeling. Here, age = 0 refers to juveniles that were born the spring before our capture efforts that fall (i.e., fishers that were ~7 months old at the time of capture).



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