UNDERSTANDING A FISHER REINTRODUCTION

IN NORTHERN CALIFORNIA FROM 2 PERSPECTIVES

Annual Report for 2015

For the period October 2009 to December 2015

By

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We have written this report in fulfillment of our obligations to our collaborators and as part of our Memorandum of Understanding. It is primarily intended to inform our cooperators and other interested parties about the data we have collected through 2015, and about the application of those data to our objectives and to research hypotheses on fishers generally. The information contained herein should be considered preliminary and has not yet been reviewed by objective, third-party scientists. This report cannot be considered of the same quality or rigor as a peer-reviewed, scientific publication. Our intention is to present current and accurate information, but we cannot guarantee that information in this report is complete, free from error, or will not change in the future. Before citing this report, contact Roger Powell to learn whether pertinent publications are now available and, if not, that the information in this report has not be superseded or updated.

Summary

From late 2009 through late 2011, we released fishers (*Pekania pennanti*) (24F, 16M) onto the Stirling Management Area owned by Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California. We have monitored all translocated fishers and their progeny as closely as possible to document their survival, reproduction, dispersal, and home range development through 2015 (year-6). Released fishers experienced high survival during both the initial post-release period (4 months) and for up to 2 years after release. We have documented 26 fisher mortalities since 2009, including 5 in 2015. We have documented reproduction in all years of the study and from each of the 3 translocated cohorts. Of the 40 fishers in the 3 release cohorts, we tracked 32 (80%) long enough to document the establishment of home ranges. Males had larger home ranges and travelled further than females. Fishers from some source populations were infected with eye worms (*Thelazia californiensis*) and some fishers from Humboldt and western Trinity counties were infected with a previously undescribed trematode**.** During our annual trapping effort in October–November 2015, we captured 46 individual fishers (33 F, 13 M) 84 times, including 25 new fishers (16 F, 9 M), 22 juveniles (14 F, 8 M), 2 females and a male released in year-2, and 2 males released in year-3. Our best estimates of survival and reproduction are consistent with a stable or growing population on Stirling. Our population modelling, however, shows that short-term population stability can not be confirmed before year-10 of the project, or 2020.

Contents

Introduction

The human-assisted movement of animals goes back thousands of years in Europe (Alcover, 1980; Masseti, 1995) and more than a century in North America (reviewed by Bolen and Robinson, 2003) but, until recently, feasibility planning and research design have not been incorporated into translocations (Biggins et al., 2011; Breitenmoser et al., 2001; International Union for the Conservation of Nature, 2013; Lewis et al., 2012; Miller et al., 1990a, 1990b; Powell et al., 2012). Unfortunately, reintroductions of endangered species in recent decades have experienced frequent failures (Armstrong and Seddon 2008). Efforts to counteract failures have led to better planning and to introducing experimental design into reintroductions (e.g., Miller et al. 1990a,b; Lewis and Hayes 2004; Callas and Figura 2008; Biggins et al.2011). In addition, a critical factor that has received little attention is the effect on a source population of removing prime, reproductive, adult animals, animals with high reproductive value (v_x), for release elsewhere (Armstrong and Seddon, 2008; Powell et al., 2012). The effects on a source population of removing prime, reproductive animals are potentially greater than those of trapping similar numbers of animals for fur (Buskirk et al., 2012), which can include large numbers of non-reproductive juveniles.

Because of concern for the status of fishers in California, to understand how fishers in particular, and mammalian predators in general, respond to intensive forest management, and to understand better why some fisher reintroductions have succeeded while others have failed, the California Department of Fish & Wildlife (DFW; formerly Fish & Game), the US Fish & Wildlife Service (FWS), Sierra Pacific Industries (SPI) and North Carolina State University (NCSU) started collaborating in 2007 to re-establish a fisher population in the northern Sierra Nevada and southern Cascade Mountains of California. In 2009, the California Department of Fish & Wildlife gave final approval for 40 fishers to be reintroduced over 3 years (autumn 2009 – autumn 2011) onto SPI's 648 km^2 Stirling Management Area (hereafter "Stirling"), which is managed intensively for timber production (Figure 1). The Memorandum of Understanding initiating the research states that released fishers and their progeny are to be studied intensively for the 7 years following the year-1 reintroductions.

In a related effort to understand the fisher population in the far northeastern extent of the fisher's range in California, we began independently in 2006 to use non-invasive methods to estimate population parameters for the fishers living on the managed, forested landscape centered on the Klamath River in northern California and southern Oregon (Figure 1). Combining the non-invasive, genetic surveys conducted in this study area with the research on reintroduced fishers on Stirling provided the opportunity to broaden conservation benefits for fishers (e.g., Seddon et al. 2007, Sarrazin and Barbault 1996), to understand better the dynamics of fisher populations on managed landscapes, and to study a source population for a reintroduction. We moved fishers from this study area to Stirling in the winters of 2009-2010 and 2010-2011. These removals were targeted to lands owned by Michigan-California Timber Company (formerly Timber Products Company), meaning that fishers were removed from managed, industrial timberlands and released on a different but also managed landscape. The removed animals were targeted to be adult members of the population with high reproductive potential.

The objectives of this research are:

To estimate annual survival and reproduction of fishers on Stirling between 2010 and 2017.

To evaluate habitat selection by reintroduced fishers and their offspring to test the predictions of available landscape-scale models of habitat quality and suitability for fishers.

To evaluate fisher diet composition and prey distributions and abundances as a metric of fisher habitat quality.

To quantify energy expenditure, energy balance, and overall body condition of fishers and relate these metrics to habitat quality and fisher conservation.

To genotype genetic samples collected from reintroduced fishers and their offspring

To identify aspects of habitat associated with, and to test functional models for, natal dens, maternal dens and rest sites for fishers.

To quantify disease prevalence and exposure in translocated fishers and their offspring to determine the influence of disease on short and long-term persistence on the landscape.

To predict the placement, sizes, and shapes of home ranges of reintroduced fishers and their offspring using models of optimal home range choice and to test those predictions using data on actual use of space by those fishers.

To predict patterns of breeding by Stirling males from home range placement and familiarity with landscapes and to test those predictions using data on paternity of fishers born in the study area.

To evaluate the accuracy, precision and efficacy of a long-term fisher monitoring protocol during fall survey efforts 2013-2016.

To estimate abundance, survival and recruitment, population growth rate and occupancy for the source population of fishers through 2016.

To estimate the effects on abundance and population growth rate, if any, caused by removing 9 adult fishers from a source population (an estimated 17% of the population) in 2009-2010 for release on Stirling.

To evaluate the original non-invasive study design, redesign the monitoring protocol as necessary, and test the redesigned protocol for use as a monitoring tool for the reintroduced fisher population on Stirling.

To investigate the effects of intensive forest management for timber production and fire and associated salvage operations on fisher population dynamics.

Here we report on research activities that address these goals directly for year-5, January–December 2015 of the project. We review non-invasive research in the Klamath Region and the reintroduction activities to date.

Terms and Definitions

See the Annual Report for 2014 for our definitions of *success* of the re-introduction and success of the project, definitions of *establishment* and *viability*, and definitions of the 6 years of the project.

General condition, disease, and ectoparasites

We assessed the health of all fishers that we captured on Stirling by conducting detailed physical examinations at the time of capture. We collected blood, mucosa and fecal samples to determine disease exposure to pathogens that could affect population health through either direct mortality of adults or kits, or through impaired reproduction. We sent samples to the Integral Ecology Research Center, McKinleyville, California, where they will be tested for exposure to canine distemper virus and to *Toxoplasma gondii* (the causative agent of toxoplasmosis), and to infection with canine parvovirus at a later date. Since the inception of this project, fishers captured on Stirling have been assessed, generally, as being in good health. We have seen no systemic physical abnormalities in either adult or, more importantly, young fishers born on Stirling that would cause us to believe the population is currently suffering from inbreeding effects or other issues that cause us concern. Nevertheless, we collect genetic information on all animals translocated and born on Stirling for later evaluation, specifically if problems hypothesized to be related to inbreeding should arise. During physical examinations, at least 2 biologists (usually a field biologist and a wildlife veterinarian) have graded fishers for general condition based on the condition of their teeth, skin and fur, musculature, obvious wounds or injuries, ectoparasite load, weight, and amounts of fat over the hips and ribs. We defined poor condition as having obvious, serious injuries or disease (including high ectoparasite load) and very low levels of body fat relative to other fishers. We defined excellent condition as having no signs of serious injury, having all carnassial and canine teeth and little wear on incisors and premolars, and having high levels of fat over hips and ribs. We defined average condition as being not obviously in poor or excellent condition. Fishers in average condition may have minor injuries and may have missing or highly worn teeth, but have no conditions that are obviously negatively affecting the fisher. When we have encountered animals that for some reason did not fit into our 3 categories, we graded them as below or above average at our discretion.

Of 197 captures of fishers on Stirling through December 2015 where we evaluated the fishers, including recaptures of reintroduced fishers and captures of fishers born-on-site, we have graded none as being in poor condition. We have graded 23 (12%) as showing below average condition, 94 (48%) as average, and 80 (40%) as above average or excellent. The average body condition may change through time on Stirling and, though the condition of animals to date gives us no cause for concern, we advocate continued monitoring of overall health and condition for as long as feasible.

Through year-5 of our research, we have collected ectoparasites of 4 taxa from fishers. Fleas and ticks have been relatively common (Figure 2). The data show variation, likely due to environmental conditions, but no distinct patterns. We do not know why the occurrence of eye worms (*Thezalea*) varies so much. The percent of fishers that are infected with these 3 parasites on Stirling are similar to infestations elsewhere in California. In each year at least 50% of fishers trapped on Stirling had at least one ectoparasite (Figure 2b). Yet, fewer than 20% of the trapped fishers carried 2 different taxa of ectoparasites and fewer than 10% of the fishers carried all 3 taxa. Generally, when parasites do occur on fishers, the infestation is light to moderate in severity. If occurrence of parasites on fishers increases through time, it could indicate decrease in habitat quality, decrease in prey availability, or some other change in the abilities of specific fishers on Stirling to deal with ectoparasites. At present, our best evidence suggests that the processes driving ectoparasite occurrence on fishers are similar on Stirling and elsewhere in California. We shall continue to examine all fishers captured on Stirling for infection and other health-related issues that may affect the population.

In previous years we reported the occurrence of a new trematode species living in the perianal tissue of fishers. This parasite is still known only from a restricted geographic range in the coastal areas of California (Clifford et al. 2012). To date we have captured no fishers infected with these trematodes on Stirling, but we remain vigilant in examining all fishers for indications they may be infected and we remain optimistic that we did not transfer the parasite to Stirling.

Locations, movements and home ranges

The responses of fishers to being released onto Stirling, specifically their site fidelity after release, is an important measure of how those fishers perceived their environment and its habitat quality upon release (Berger-Tal and Saltz 2014). We have noted in previous reports that some released fishers did wander, or explore, and at times did settle into areas off the district (Powell et al. 2012). As of 2015, the majority of locations of fishers have occurred within the boundaries of Stirling or very near to it (Figure Locs). Similarly, most den locations have occurred on Stirling. Because the majority of our research effort occurs on Stirling (Figure 3), these data are not representative of all fishers in the reintroduced population. We know that some fishers live on adjacent lands. Nonetheless, a minimum of 40 fishers remain on Stirling annually, representing a core population. Consequently, our data show that some fishers have found enough habitat of sufficient quality for them to stay on the study area.

In year-1, we implanted female fishers with IMP-310 very high frequency (VHF) transmitters made by Telonics (Mesa, Arizona) and 4 (of 9, 44%) failed prematurely (< 8 months). In year-2, we used Telonics MOD-125 collars for females. None failed prematurely. In year-3 we outfitted females with MI-2i collars made by Holohil Systems Ltd (Carp, Ontario, Canada) because they weighed less than the Telonics MOD-125 collars and their external design was less bulky. Two of the 14 (14%) Holohil transmitters failed prior to their estimated battery expectancy of 24-30 months. In year-4, we outfitted females (12) only with Telonics MOD-125 collars. In year-5, we outfitted 12 females with refurbished Telonics MOD-125 collars and 5 females with refurbished Holohil MI-2i collars. In year-6 we outfitted 19 females with Telonics MOD-125, and 5 with Holohil MI-2. In years-2 – 6, we outfitted young fishers born on Stirling with radio collars only if the fishers had necks that were unlikely to grow substantially (>2 cm) in the future.

We radio-tracked 29 females during the calendar year 2015, 9 for only a few weeks after being trapped in October or November, 12 all year, and 8 for part of the year. The females wearing transmitter collars maintained home ranges spread widely across Stirling and onto adjacent land. Consequently, we targeted females who lived centrally to locate daily and attempted to locate peripheral females less often. Given incidents of bad weather, the mountainous terrain, limited personnel, and myriad other conditions that affect VHF telemetry, we rarely achieved this goal. For all females, we averaged 2.5 + 1.0 (±SD) estimated locations per female per week for weeks each female was tracked. We averaged 96 triangulations per female per year. For each estimated location, however, almost as many attempted locations did not meet the selective criteria we used when triangulating locations. Sometimes we did not locate females frequently because they moved beyond the perimeter of the area we searched regularly and sometimes females used parts of the study area that blocked their transmitters' signals, leading to an unknown bias in our estimates of their movements. Female fishers do not travel as widely as do males, however, limiting the effects of bias, if it existed (Powell 1994). At the end of 2015, we were actively tracking 23 female fishers.

We outfitted adult male fishers with Platform Terminal Transmitter (PTT) collars that work with the Argos satellite system and were made by SirTrack (KiwiSat 303; Havelock North, New Zealand). The satellites tracked these collars even when conditions did not permit ground tracking and, thereby, obtained more location estimates per male fisher than we obtained per female using VHF telemetry. Young males are not good candidates for wearing collars because their necks may grow rapidly. During the course of 2015 we followed a total of 7 males, starting the year with 4, some of whom died and lost collars, and ending the year with 4, which included new males captured in autumn. We outfitted two fishers with Global Positioning System (GPS) collars in 2015.

Although the batteries in the Argos collars should last over a year, some collars have failed before their projected lifetime. Many failures whose causes we documented were caused by fishers chewing and, thereby, shortening the transmitter antennas. In other cases, the main transmitter body was damaged or lost and therefore did not function. A few collars dropped from fishers early in the research due to failed attachment bolts, a problem that we have resolved. Despite premature failures, the Argos collars have provided location data that we simply would not have obtained using traditional VHF technology. Several males have made sojourns to places (e.g., Central Valley or north of California Highway 44) that we did not expect or have searched but they ultimately returned to the general area of the releases. We would never have tracked those long-distance temporary forays using traditional technology. On the whole, the Argos collars on male fishers have functioned for long periods and have provided more location data with less bias than possible with VHF transmitters.

We averaged 262 ± 456 locations/male/year across all study years and 310 ± 402 locations/male in 2015 (Table 1). All Argos location estimates are classified into 1 of 6 error classes, some of which will be suitable for some analyses but not others. Individual males averaged 48 ± 21 locations/male/year from the 2 categories with smallest error and 24 ± 61 locations in 2015 (Table 1).

Triangulations constitute the majority of estimated locations of females and young males. For fishers tracked with VHF telemetry, approximately 85% of all estimated locations were triangulations. Another 5% of VHF locations were estimated from fixed-wing aircraft or a helicopter and 10% were "walk-ins". Walk-ins included visual observations of fishers and locations of identifiable den or rest trees. Walk-ins also included trapping locations, mortality locations, and locations where fishers dropped collars. Additionally, we have located >200 individual rest locations; >90% of these were in trees, though some fishers rested under rocks, in stumps or in debris piles. Locating rest sites is biased towards finding sites in trees because fishers in trees broadcast strong telemetry signals. Location information from cameras at dens and baited stations will also be incorporated into final analyses, but those data have yet to be incorporated into our locations database.

Understanding and estimating error for our triangulations is a critical component of future analyses. We will evaluate triangulation error in two ways: 1) calculating triangulation error for test collars in known locations (both moving and stationary; n .50) and 2) comparing triangulations to "walk-in" locations for fishers that were located on the same day (usually within the same hour) in den and rest trees. A preliminary analysis of triangulations *vs* walk-in locations yielded a mean error (±SD) of 102 ± 132 m. These are preliminary results since we are finalizing protocols and software for estimating locations using triangulation. As part of our final analyses, we shall test

for relationships between triangulation error and other variables, such as azimuth angle, weather, etc. As with triangulations, we estimate error of aerial locations by having personnel who do not know the known locations of transmitters locate those transmitters. All walk-ins provide fine-scale (<20 m) information about fishers' locations.

We are able to assess true error rates for Argos locations of each error class by comparing satellite locations to known locations of males held in captivity, of collars that have been dropped (the day they are dropped is known from activity data), or of dead fishers. The mean error for Argos locations estimated across all error classes is 767 ± 1241 m. Our calculated mean error for locations in each error class are consistent with expected error predicted by the Argos service (Sauder et al. 2012; Table 2). Locations in error classes 3 and 2, predicted to have the least error, have mean error of 195 ± 247 m and 458 ± 460 m (± SD). Location estimates from the error class 3 had a maximum error of 2400 m but 91% of locations were within 350 m of the true location. Future analyses will attempt to understand better how environmental factors influence error and how we can implement other metrics provided by Argos (e.g., error radius and geographic dilution of precision [GDOP]) to eliminate locations that are highly inaccurate.

We have attempted to monitor fishers during all times of day and night to ensure that our information is not biased to one time period. VHF transmitters are more difficult to locate at night, particularly in the winter when temperature, weather and road conditions hinder access to the study area. Thus, the vast majority of VHF telemetry locations have been collected during daylight hours (8 am to 4 pm; Figure 4). We have programmed Argos collars to be located during different times of day, leading the distribution of locations of fishers wearing those collars to be relatively even across all times. We programmed GPS collars to locate themselves across all times of day, leading to a very even distributions of locations.

We are collecting enough location data to estimate annual home ranges for most fishers. Thirty locations represent a reasonable minimum sample size for estimating annual home ranges with fixed-kernel methods, though having more locations is preferable (Fieberg and Börger 2012, Seaman et al. 1999, Noel 1993, Seaman and Powell 1996). We have more than100 estimated locations per year for many fishers.

We define an animal's home range to be that part of the landscape in which it lives that it maintains updated within its cognitive map of the landscape (Powell and Mitchell 2012). For this report, for logistic reasons, we assume that 95% utilization distributions for fishers' use of space provide reasonable estimates of home ranges. We have estimated utilization distributions using a fixed kernel smoothing program. Such programs smooth data using a kernel and a smoothing parameter, "h", whose values are, ideally, related to aspects of the biology or management goals for the animals being studied. Silverman's (1990) kernel "k2" is a bell-shaped kernel with finite bounds, is leptokurtotic and, therefore, resembles the distribution of telemetry error for experienced researchers; we use "k2". Many researchers choose "h" to minimize internal error within a distribution of location estimates, and we have advocated this approach in the past (Seaman and Powell 1996, Powell 2000). Such choice of "h", however, ignores the biology of the animals studied, chooses different values for "h" for different animals, and even for different random samples from a single data set, making comparisons between studies nearly impossible. For fishers, different values of "h" provide insight into different aspects of their biology. For our fishers, h=750 m appears to estimate reasonably well the probability of where a researcher will be able to find a given fisher using telemetry. Average daily movements of fishers suggest that 1500 m should estimate where a fisher can travel over the coming day. Average distances across distributions of location estimates suggest that 1000 m will estimate the overall range of space a fisher uses but not its small scale preferences. Values of "h" tailored to match the estimated error for each location estimate should provide the best estimates of fishers' habitat preferences. Table 3 shows mean estimates for 95% utilization distributions for 2010-2014 using h = 750, 1000 and 1500.

Table 3 shows that males have larger areas of use than do females and that larger values for "h" lead to larger utilization distributions. Daily tracking of fishers suggests that females established home ranges primarily within Stirling. Some females have travelled to adjacent Forest Service or private lands and one traveled north .22 km onto the Lassen Management Area of Sierra Pacific Industries; she died, however, within 3 months of release. Additionally, female fishers have denned in trees on both the Lassen and Plumas National Forests, but usually within 2 km of the Stirling border. One female born on site and initially captured in early 2012 established a home range primarily off Stirling in the Rock Creek area which borders both the Lassen and Plumas National Forests.

Male fishers have also established home ranges over most of Stirling. Since males have larger home ranges than females and disperse more widely, they have been located on adjacent lands more often than females. Several males have established home ranges off Stirling and up to 40 km from where they were released and we no longer track these fishers because their home ranges are outside the area we trap each year. If those males that we no longer track have movements and survival similar to those we do track, untracked males may have a substantial presence on Forest Service lands, private timber lands and SPI holdings adjacent to or near Stirling. We have not tracked most juvenile males born on Stirling that have, or will, disperse long distances and, consequently, we do not know how far away males that originated on Stirling may establish home ranges.

The utilization distributions we have presented above weight all location estimates equally and, therefore, give insights into where fishers spend time. One can calculate utilization distributions based on currencies other than time. In last year's Annual Report, we presented examples of home ranges built using energy as the currency.

Population monitoring on Stirling

From 12 October through 9 November 2015 we conducted a large-scale trapping effort on Stirling to capture as

many fishers as possible and to outfit or re-outfit these fishers with functional transmitters. We spread our trapping effort across Stirling and adjacent lands focusing on areas where fishers were known to live, had been previously detected, or areas we considered likely to have fishers (Figure 3). To maximize efficiency, we split the study area into east and west of Butte Creek. We trapped the east side for 14 trap days (12 October - 25 October), then moved to the west side (26 October - 09 November) for 14 days. Logistical constraints precluded or curtailed trapping in some areas we thought may have resident fishers.

We deployed approximately 100 traps each night, totaling 2865 trap days (1439 east, 1426 west). We totaled 84 captures of 46 individual fishers (33F, 13M), yielding 2.9% trap success (% captures per 100 trap days). This capture rate was considerably higher than in all other years of trapping (Table 4). As we experienced in previous years, capture success was greater on the East side (3.8 captures/100 trap days, or 3.8%) than on the West (2.0%). Whether this difference is related to our releasing more fishers on the East side (30 *vs* 10), we do not know. Fishers have certainly moved across the study area since the initial releases. We captured 25 new fishers (16F, 9M), 3 of which were adults (2F, 1M). We saw an increase in the number of juveniles captured in 2015 (14F, 8M), 10 of which were captured more than once.

Of fishers translocated to Stirling, we recaptured 1 female released in year-1, 1 female released in year-2, and 1 male released in year-3. Of fishers born on Stirling, we recaptured 3 born in 2011 (3F, 0M), 6 born in 2012 (5F, 1M), 7 born in 2013 (6F, 1M), and 4 born in 2014 (3F, 1M).

Of the 33 female fishers captured, 20 were given new collars (Telonics MOD-125 or Holohil MI-2), and 4 kept their old collars (Holohil MI-2). All 5 adult males received new collars (Sirtrack Kiwisat 303 or Lotek Minitrack). Fishers have dispersed widely across Stirling now and limited personnel and other resources prevent us from tracking them all consistently. Therefore, although the majority of the 9 females that we did not collar could have carried them, we were restricted by the number of collars we had and by our ability to track them all consistently. We failed to capture 1 female and 2 males whose transmitters were still functioning, demonstrating that even when we placed traps within the known home ranges of fishers we do not always capture them.

We had 142 captures of non-target carnivores for a capture rate of 5.0% (Table 5), higher than capture rates for non-target carnivores in 2014 (3.6%), 2013 (2.6%) and 2012 (1.2%). The capture rate for non-target carnivores was slightly lower on the east side (4.3%) than the west side (5.6%). Spotted skunks (*Spilogale gracilis*) and grey foxes (*Urocyon cinereoargenteus*) were the most commonly captured non-target carnivores, accounting for 73.2% of the total.

At the conclusion of trapping in 2015, the age structure of the known fishers on Stirling emphasized young fishers (Figure 5). Fishers < 2 years old comprised 50% of all fishers known to be alive at the end of 2015. Many fishers older than 2 years of age are still in the population but the young age structure suggests healthy reproduction and recruitment. The age distribution in Figure 5 is our best estimate of the true age distribution of the Stirling fisher population but is accurate only to the extent that our trapping results were representative for the population. We do not know why so few 1-year old fishers appear in Figure 5. Survival for that age group could have been low in 2015 but our survival analyses do not show low survival for that age class (Figure 6). Alternately, and more likely, we simply captured few 1-year old fishers during trapping in 2015.

At the end of our trapping effort in autumn, 2015, the minimum known population size for the fishers on Stirling was 50 (total fishers captured + non-captured fishers still wearing functional transmitter collars). We have retrospectively adjusted the minimum population sizes for previous years, accounting for fishers that were not know to be alive in those years but that since been captured, showing that they must have been alive in those past years (Figure 7). The minimum number alive values suggest a slight decrease in the population size in 2013, though less severe than estimated during that year, with a probable rebound in 2014 and into 2015. Calculations of the minimum number of females alive indicate that the female population size has been relatively stable or growing slightly since the final releases of fishers in 2011 (Figure 7). Thus, the observed decrease in minimum number alive size during 2013 appears to be related to changes in the number of adult males. Consistent with these numbers, we observed a relatively high number of male mortalities during 2013 and early 2014 (n= 5).

Survival

Through December 2015, we confirmed the deaths of 26 fishers (16 F, 8 M, 2 sex unknown). One female slated for release died in captivity in late 2009. During 2010, premature transmitter failure limited our ability to document survival yet we still documented the deaths of 3 females. Since 2011, however, we have tracked almost all females continuously for the year or until death (if they died): 2011 - 2 F, 1 M; 2012 - 4 F, 1 M; 2013 – 1 F, 3 M; 2014 - 2 F, 2 M, 2 unknown; 2015 – 3 F, 1 M. Trapping in autumn 2015 allowed us to capture fishers whose collars had failed in previous years as well as fishers that were captured in previous years but had not collared. We used data from telemetry and trapping to examine patterns and rates of survival for reintroduced and Stirling-born fishers for December 2009 through December 2015.

We analyzed monthly survival using "known fates" analyses within program MARK (White and Burnham 1999). Known fates analyses account for each time period when fishers were known to be alive or were found dead. Fishers that we could not document as either alive or dead within any month were censored and, therefore, these fishers were not used to estimate survival for that time period. We used Akaike's Information Criterion corrected for small sample size (AICc) to rank 14 hypotheses that could explain the pattern of mortalities and survival that we documented (Table 6).

We developed 14 hypotheses from 9 variables hypothesized to affect survival of reintroduced fishers and included a null hypothesis of constant mortality over time (Table 6). The variables were 1) "Sex" (due to differences in size, movements, etc. between the sexes) 2) "Reproductive Season" (females have high activity levels, which leads to high mortality, in April - August [the time of lactation and highest energy output by females, Powell & Leonard 1983], males have high activity levels, and greatest risk of mortality, in March – May to find females and compete with other males), 3) Age of animals estimated at the time of their capture for animals less than 1 year old, fishers between 1 and 2 years old (1-year) and fishers older than 2-years old (adults), 4) Time (survival changes through time). Note that time was generically tested where all months and years were hypothesized to have different rates of survival, but we also considered monthly (5] Month) and yearly (6] Year) changes to survival in addition to interactive and additive combinations of those characterizations of time. We also tested 8 hypotheses with combined variables: 1) "Age + Reproduction where there sexes were modeled differently during the reproductive season" where we assumed differences in survival among the age groups and during the reproductive and non-reproductive times of year (because juveniles do not reproduce, avoiding the costs of the reproductive season), 2) Age + Reproduction where there sexes were modeled the same during the reproductive season", 3) "Age with annual variation" where we modeled juvenile and 1-year old survival differently in each year, 4) "Sex + Age" where we modeled differences in age that were different between the sexes, 5) "Sex × Month × Year" (because sex-specific mortalities could differ among months), 6) "Sex × Year" (sex-specific mortality that is similar in pattern, but different in magnitude, through years, 7) "Month × Year" (patterns of monthly survival are similar in pattern, but different in magnitude, across all study years, and 8) "Sex + Month" (sex-specific patterns in monthly survival across years). We have not modeled survival based on if an animal was reintroduced vs born on Stirling or based on release or capture cohort. Such models were tested in past reports but no new animal have been reintroduced and differences in age and time are largely redundant to cohort.

The highest ranked hypothesis included the Age and Reproductive season with the sexes modelled as not equal (Table 6). The second highest ranked hypothesis included differences among ages and between sexes. Monthly survival estimates for females and males during reproduction 0.96 (95% CI = 0.93-0.98). These differences in monthly survival translate to relatively large declines in annual survival (Figure 8). During all other periods, estimated monthly survival rates were higher for both sexes (0.99; 0.97-0.99). We estimate that the annual survival rate for adult fishers, including breeding and non-breeding times of years, is 0.78 (0.53-0.81). This estimate of annual survival is marginally smaller than last years' high estimate of 0.80. The pattern of survival has been consistent since we first released fishers and indicates that the time of year, and potentially the reproductive status of females and males, explains fisher survival better than other models we have explored. Additionally, in previous years we did not model age explicitly but age was an important variable in this year's analysis. Strikingly, survival of adults is lower than that of both juveniles and 1-year old fishers (Figure 6). Perhaps, this occurs because juvenile survival for this analysis can be estimated only from the time of capture in the fall through to their first birthday and 1-year olds only remain in that age class for a single year. Old adults eventually die and this likely accounts for the relatively lower rate of estimated survival. Changes in the estimates of survival are not large through time but do change because of sample size and simple variation rather than an indication of a true change in survival. Annual survival does not show strong changes through time (year to year; Figure 8). Additional survival analyses with varied data sets where we used individual covariates such as weight were not strongly supported (Facka et al. In prep). Further, age and reproductive status were relatively well supported and indicate these variables are consistent across different analyses. Future analyses of survival will include covariates based on home range composition and estimates of habitat quality.

In general, survival is high for fishers throughout the year but reaches its nadir during the reproductive period. Nine of the 13 (81%) females documented as having died did so in April - August. One of those females was found dead in October but had obviously not died recently. That female had not been located since August and, therefore, we dated her death to August. Additionally, 90% (9 of 10) females that died during the period of kit dependency (April-August) were clearly lactating or were known to have had kits in the months prior to their deaths. Five of 7 males (71%) died in March - May, coinciding with the peak of their breeding behavior. In 2015, we documented only 1 male dying. That male was a year-2 translocated fisher that was 9 years old at the time of death in late April. We also documented the death of another year-2 translocated female in October of 2015. Thus,

at the end of 2015 3 originally reintroduced fishers (1 F; 2 M) were known alive and on Stirling.

We characterized the sites where we found fisher carcasses or partial remains and took photographs. Fisher carcasses with sufficient remains and little to moderate autolysis were necropsied by Leslie Woods, an experienced wildlife pathologist at the California Animal Health and Food Safety Lab at the University of California Davis, with the assistance of Deana Clifford. She examined all major tissues to identify lesions, and performed immunohistochemical, toxicological, bacteriological, parasite, and virological diagnostics as needed. Carcasses that were moderately to severely decomposed or did not contain adequate viscera (partial remains) were necropsied by Deana Clifford and Jaime Rudd at the CDFW WIL, with select tissues (when present) examined microscopically by Dr. Woods. For any fisher carcass with evidence of predation, Greta Wengert (Integral Ecology Research Center) conducted molecular forensics to determine the species of predators that contacted the carcass and could have been responsible for killing the fisher (Wengert et al. 2014). Samples collected for predation analyses included hairs observed on the carcass that were thought to be from a predator (not fisher), matted fur (presumably matted with predator saliva) and tissues around apparent punctures caused by possible predator canines, and polyester swabs within all apparent puncture wounds caused by possible predators. When only partial remains were found, bones and the remaining transmitter (implant or collar) were sampled for genetic material from predators or scavengers. DNA was extracted from samples using DNeasy Blood and Tissue extraction kits (Qiagen, Valencia, CA, USA). Polymerase Chain Reaction (PCR) was run on each sample using primers specific to the families Felidae and Canidae; resultant PCR products were sequenced, and sequences were cross-referenced on GenBank to determine species identity. These methods have been used successfully on carcasses of 57 fishers (from multiple studies) killed by other predators to determine predator species (Wengert 2014; G.M. Wengert, unpublished data). In cases where only scant remains were recovered, DNA from other species could have been associated with predation or scavenging.

To date, a total of 26 carcasses (16 Translocates, 9 Stirling-born, and 1 translocate candidate that died while in captivity prior to release in year-1) have been submitted for examination. Four of these carcasses were submitted in 2015 (2 translocated fishers and 2 Stirling-born fishers). All but 2 carcasses (partial remains only submitted in 2015) have been examined. A complete set of formalin-fixed tissues from 4 fishers (2 translocated fishers, 1 Stirling-born fisher, 1 candidate for translocation) that had partial remains. The remaining 14 carcasses examined were either too decayed or lacked tissues for histologic examination. In addition, liver samples from 5 and a muscle sample from 1 of the 24 necropsied carcasses were tested for the presence of 7 different anticoagulant rodenticide compounds (ARs). The remaining 13 carcasses were not tested for ARs due to lack of suitable tissue for testing. Predation forensic analysis has been attempted on 6 carcasses; 5 partial carcasses are awaiting forensic testing pending fund availability.

Necropsy, predation forensics and toxicology findings for all fisher carcasses collected on the project to date are summarized in Appendix 1. Cause of death was definitively determined for 4 fishers: 1) a female fisher found in a water tank in July 2010, 8 months post-release, was confirmed to have drowned but also had severe fascial and cellular inflammation with necrosis of the hind leg muscle that could have affected her ability to ambulate normally; 2) a female fisher found dead in December 2011, a few days post-release, had systemic disease (vasculitis, hepatitis, hypertension and pneumonia) of unknown origin; 3) a female fisher found dead in May 2012, 5 months post-release, was killed by a bobcat, and 4) a male fisher found dead on Highway 32 in March 2011, 3 months post-release, was confirmed to have died from vehicular trauma (Woods and Wengert, unpublished). Two additional dead, Stirling-born fishers, a lactating female and a kit recovered at the same time in September 2014, appear to have died from drowning in a water tank. Autolysis precluded histological examination to rule out other diseases in one carcass and histological results are pending on the second carcass. Gross and histologic findings suggestive of hypoxia, hyperthermia and suffocation were documented for the translocate-candidate fisher that died while in captivity prior to be release in 2009, but the cause of death could not be definitively confirmed (Munson, unpublished). Bobcat DNA was amplified from the carcasses of 2 female fishers found dead in June 2010, 4 and 6 months post-release, but carcass condition for these fishers was not adequate to determine if the fishers had been killed or scavenged by bobcats. Predation forensics conducted on a female fisher found in July 2012, 18 months post-release resulted in weak amplification of felid DNA, but repeat testing was inconclusive. Samples tested from 2 additional carcasses found post release in 2012 suspected to be predation cases did not amplify any predator DNA (Wengert, unpublished). In general, the causes of mortality observed are consistent those found by other studies in California (M. Gabriel and G.M. Wengert, unpublished data) but our inferences from the data are limited by the lack of carcasses recovered in suitable condition for us to determine the cause of death.

Anticoagulant rodenticide compounds were present in the liver tissue of 4 of 5 the fishers tested. Two fishers (the year-1 female that died in captivity prior to release and the year-3 female that died shortly after release due to systemic disease) were exposed to brodifacoum. A third fisher (the year-2 male that died due to vehicular trauma) was exposed to both

brodifacoum and bromadialone. Results for the fourth fisher (a year-2 female found in a river with unclear cause of death in 2015) showed a trace amount of brodifacoum. For the 3 reintroduced fishers, anticoagulant exposure could have occurred pre-capture or post-release, as the half-life of these 2 second generation anticoagulants in liver tissues is >150 days (Vandenbroucke et al, 2008). In contrast, for the fisher that died in 2015, positive result for exposure to anticoagulant rodenticides indicates exposure on the Stirling study area because she lived her entire life on the study area. Anticoagulant rodenticides were not detected in the a Stirling-born fisher that died in April 2013 and had only muscle available for testing. Since anticoagulant rodenticides are stored long-term primarily in the liver, exposure cannot be definitively ruled out for this fisher.

The finding of multiple compounds in a single animal may indicate exposure from multiple source points or uses. The overall significance or potential impacts of sublethal exposure to anticoagulants in fishers and other wildlife are largely unknown, but widespread exposure and cases of direct mortality due to anticoagulant toxicity in fishers and other wildlife species has raised significant conservation concerns (Gabriel et. al 2012, 2015). In June, 2014, the California Department of Pesticide Regulation restricted the use of second-generation AR products containing brodifacoum, bromadiolone, difenacoum, and difethialone to certified pesticide applicators, thus these compounds are no longer available over the counter (California Department of Pesticide Regulation 2014). It is anticipated that this regulation change will reduce non-target wildlife exposure risk from household use, especially in urban/suburban areas, but it is unclear if it will have any impact at reducing AR use at illegal marijuana cultivation sites, thought to be the most likely source of exposure for fishers (Gabriel 2012, Thompson 2013). Analyses of fisher mortalities since 2012 indicate that exposure to and toxicity of anticoagulant rodenticides continues. We shall continue to test liver samples from any recovered fishers that were born on Stirling.

We continue to radio-track and retrieve dead fishers as quickly as possible, since understanding survival rate is critical for understanding population dynamics, which is major objective of our research. To meet all of our goals, we are committed to locating all fishers on Stirling as often as possible to determine causes of mortality in the reintroduced population. Limitations on manpower, relatively few aerial surveys (<2 per month), and widely spaced individual fishers often preclude detecting and recovering dead fishers quickly. Future survival analyses will incorporate biologically relevant covariates (e.g., body mass, age, home range components), if possible, to yield a mechanistic understanding of the factors that affect survival of fishers on Stirling.

Reproduction

Fishers on Stirling have produced kits in all 6 springs since the first releases. Our daily searches for female fishers provide a good knowledge base of their daily movements. We suspect that a female has denned and given birth to kits when we locate her in a very localized area, especially in the same tree, on successive tracking occasions in late March and early April. We then verify denning by monitoring via telemetry and remote cameras. The mean value for initial denning rate for the entire study is 0.79 with a low of 0.63 and high of 0.90 (Table 7). In other studies, females have sometimes aborted or lost litters after they started denning (Matthews et al. 2013a). To date, we have not documented the loss of entire litters except when females have died while denning; we assume that all offspring die when their mother has died. Since 2010, we have documented 9 females that died while they were denning (or a mean of 1.5/yr). At a minimum, we estimate that these deaths of mothers represent the deaths of 13 kits (2.2 per year; Table 7). We know that kits that are old enough can survive without their mothers but we do not know what age might be considered the threshold. If we assume that kit mortality comes only when their mothers die, then we can estimate that kit survival is 88% (68 of 77; Table 7). Nonetheless, we know that some kits die in dens even when their mothers live. We know that our estimate of litter size is an under-estimate because mothers move kits without being photographed. Our estimate of litter size is also an underestimate because kits that have died in dens are not documented. Thus, our estimate of litter size already incorporates some kit mortality. If we combine our estimate of kit survival through denning (78%) with autumn survival following capture (80%), we get an estimate of kit survival (from time of litter size counts to age 1 yr) of 62%. We urge caution in using this estimate of kit survival but, thus far, it provides our only information for this aspect of reproductive success and survival.

Fishers on Stirling have denned and given birth at times similar to denning by fishers elsewhere (Powell 1993). Natal dens (dens in which females give birth) are most often found during the final two weeks in March or first week in April (mean = week 13.3; Figure 9A), with the earliest den found on 17 March and the last found 19 April. Because a female must localize movements before we even look for a den tree, our identification of den trees comes several days, maybe even weeks, after a female has given birth and thus our denning dates are biased late. We note in Figure 9B that we have generally found dens earlier and earlier each year. This trend may reflect our inexperience in finding den trees under difficult conditions during those early years (e.g., high snow fall, downed trees). Alternately, all females in 2010 and most in 2011 had recently been released to our study site, potentially causing the females that had been moved to give birth late. One of our early findings from this project is that the time

when we released females influences their probabilities of giving birth (Facka et al. 2016). Translocation may have also caused females to delay births. Females move their kits to maternal dens throughout the spring and summer with highly variable timing and without a pattern (Figure 9B). Some females never move their kits, which is not shown in Figure 9B. Though we attempt to locate females and their kits throughout the summer we consider the denning season to be effectively concluded by the end of June (week 27) and most females move kits often to rest trees that they use while foraging.

Of 18 adult females tracked in 2015, 14 exhibited behavior consistent with denning. One of these females appeared to have early denning failure. She left a what appeared to have been den 7 days after we found it, suggesting litter failure. Another female dropped her collar approximately 10 days after we confirmed a natal den, and one female died in early May. We documented 21 kits from 11 females (1.9 kits per female). All 4 females whom we suspected not to have denned (from tracking via radio telemetry) were captured in the fall and examinations of their teat size corroborated the telemetry evidence that they had not den successfully.

Through 2015 and for Table 7, all reproduction metrics have been based only on females we tracked through telemetry. We collected additional data on birth rate each fall by examining the teats of females for signs of lactation (Matthews et al. 2013b). In addition to females confirmed to den and have kits, we captured 3 not-collared females that appeared to have raised kits in 2015. Based on these metrics, we estimate that a minimum of 14 project females gave birth in spring 2015 and subsequently survived until autumn. We cannot know how many total kits these females had or how many of them survived but, nonetheless, all metrics indicate that the majority of adult females gave birth on Stirling in 2015.

For all adult females captured in autumn 2015, 70% had nipple sizes indicative of having lactated earlier in the year, and nipple sizes of 100% of adult females not tracked with telemetry indicated lactation (3 of 3).

We have documented females denning across Stirling, on other private lands and on national forest lands (Figure 10). Of 148 natal and maternal den trees that we found in 2010-2015 (Table 8), black oaks (*Quercus kelloggii*) were most common for both natal and maternal dens (49%; Table 8). Female fishers used Douglas-firs (*Pseudotsuga menziesii*), incense-cedars (*Calocedrus decurrens*) and white firs (*Abies concolor*) in similar numbers (10%, 11% and 9%), and used other trees less commonly. Female fishers used live trees (35 of 46 dens) most often as natal dens but, later in the denning season, as kits began to travel with their mothers, females used snags slightly more often than live trees for maternal dens and even used hollow logs and piles of debris as dens or rest sites (48 live trees and 54 snags, logs, and debris). In 2010-2012, SPI committed resources to collect data on vegetative and topographic characteristics within 90 m of den sites. Future analyses will examine patterns of female denning and movements (locations and timing) relative to topography (temperature related movements), time of year, predators and other factors that might influence female decisions to establish and move dens.

Population Viability Analysis

We reported viability analyses in the Annual Report for 2014 and those analyses are still appropriate to the population of fishers on Stirling. We refer readers to last year's report.

Habitat Relationships

During 2015 we distributed to all cooperators a report entitled "Fisher Habitat Selection on Stirling Management District 2010-2014: A Critical Test of Our Understanding of Fisher Habitat Needs". That report is appended to this Annual Report for 2015 as Appendix HABITAT and we refer readers to that report.

Food Habits and Prey Population Dynamicss

During 2015 we distributed to all cooperators on this project a report entitled "Fisher (*Pekania pennanti*) prey availability and habitat use on managed timberlands in Northern Sierra Nevada". That report is appended to this Annual Report for 2015 as Appendix PREY and we refer readers to that report.

Non-invasive Sampling of Klamath Fishers

We monitored fishers in a 587 km^2 portion of the Klamath-Siskiyou ecoregion in northern California and southern Oregon (henceforth, "the Klamath"; Figure 11). 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 11): the Happy Camp Complex and the Beaver Fires. We have 8 years of data before the fires occurred, and at this point, 2 years of data following the fires. One objective of the current analysis is to determine the status of the population before these fires occurred. In the winters of 2009 and 2010, the fisher population in the Klamath also served as a source for a reintroduction of fishers to the northern Sierra Nevada and northern Cascade Mountains (Callas & Figura 2008). A total of 9 fishers (4F, 1M in 2009 and 3F, 1M in 2010) were removed, and of these, 8 were translocated to an area in the

Stirling Management Area east of Chico, CA. One female in 2009 died in captivity. Relatively little research is conducted on source population dynamics following translocation efforts (Armstrong & Seddon 2007; Lewis et al. 2012). Thus, an additional objective was to understand the effects of these translocations on source population dynamics.

Non-invasive sampling techniques and individual identifications with genetics

We attempted to deploy 100 survey stations in the Klamath at the same locations near streams and on ridge tops each year (Figure 11). At each survey station we placed a non-invasive sampling box made of corrugated plastic (25x25x75 cm). 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. We fixed a glue strip to the underside of the bottom wooden slat so that a mammal coming into the box to obtain the bait was likely to leave a hair sample attached to the glue strip.

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

At the USDA Forest Service Rocky Mountain Research Station, DNA was extracted from hairs with follicles attached to each glue strip to optimize amplification, and then genotyped using the multi-tube approach recommended for non-invasive samples (Taberlet et al. 1996). All samples were amplified twice at each locus, and some were amplified a third time if the initial amplification resulted in a lack of consensus scores (Schwartz & Monfort 2008). If these three scores did not prove to be consistent, then samples were discarded as being of insufficient quality for genetic analyses. For the samples with sufficient DNA, the program DROPOUT was used to screen for any potential errors in genotyping (McKelvey & Schwartz 2004). Any samples identified to contain putative errors were re-amplified an additional three times. Following the multi-tube test and the DROPOUT screens, field information in GIS was used to evaluate the likelihood of observing a recaptured genotype in a given location (Marucco et al. 2010).

Spatial capture-recapture model

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

It is important for S to be delineated as an area large enough to identify s_i for all individuals, even those where s_i is located outside of the vicinity of the study area. Home ranges of fishers in the Klamath have been documented to be (mean ± SE) 9.6 ± 2.8 and 30.6 ± 8.6 km^2 for females and males, respectively (summarized from Lofroth et al. 2010). In the current research, S was defined as a 10-km buffer encompassing our sampling units (1836 km^2 in total) to accurately identify the locations of all activity centers of fishers. The locations of activity centers in the current research were modeled as an inhomogeneous Poisson point process in S (Royle et al. 2014). We divided S into a 1-km x 1-km grid and the location of each sampling unit was then assigned to the grid cell where it was located, and the probability of s_i in year t (s_{it}) being at the center of grid cell j (prob$_j$) was modeled as an intensity function.

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

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

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

$$\log(p_{ijt}) = \beta_0 + \beta_1 \bullet sex_i + \beta_2 \bullet previousdetection_{it}$$

where the mean encounter rate is a function of a population level intercept (β_0), a sex-specific effect (β_1), and the

effect of a binary variable (0 or 1) indicating whether or not the individual had visited a sampling unit in previous years (β_2). We modeled the detection function (g_{ijt}) with a Gaussian encounter probability such that:

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

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

Model fitting and assessment

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

Results

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

The spatial capture-recapture model indicated the population of fishers in the Klamath was relatively stable before the fires occurred and for the three years immediately following the removal of fishers for translocations (Figure 12; Table 9). Estimated locations of fisher activity centers varied each year on the landscape (Figure 13).

Current conclusions and future directions

The current modeling efforts indicate the population of fishers in the Klamath was relatively stable from 2006 to 2013. The abundance estimates are unchanged among years, with no statistically significant differences (95% credible intervals overlap; Figure 12, Table 9). 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.

Non-Invasive Sampling on Stirling

In the current report we present preliminary results from the non-invasive dataset accumulated in the fall 2013 field season. We monitored fishers in a 229 km^2 portion of the Stirling Management Area in this initial year of non-invasive sampling (Figure 14). The non-invasive sampling protocol designed for Stirling consisted of 16 sampling units (10.4 km^2) with up to 3 station replicates placed within each sampling unit. Our sampling protocol was later expanded to a total of 27 sampling units and 78 survey stations in 2014 and 2015 (see previous annual report for further information).

Our sampling units were open and functioning for 1085 total sampling nights from 24 September to 31 October 2013. During this time, we collected 140 samples that were submitted for genetic analyses at the Rocky Mountain Research Station of the USDA Forest Service. Of these samples submitted for analyses, 28 were identified as fisher, of which, 25 were of sufficient quality to determine sex and genotype. Our genetic results identified a total of 12 unique individuals to have visited our stations in 2013 (9F, 3M). We modeled this non-invasive dataset using the same methods as the Klamath dataset as outlined above. All results presented herein are displayed as medians and [95% credible interval].

Preliminary analyses estimated 32 [19, 43] fishers with home ranges in the vicinity of the sampling devices in 2013. The modeled activity centers for these individuals varied across

the landscape, but their density was highest in the southwestern portion of the study area (Figure 14). The probability of detection non-significantly differed between the sexes; males and females had a 0.40 [0.12, 0.90] and 0.34 [0.17, 0.78] probability of being detected at a sampling unit at the center of their home range, respectively.

We look forward to incorporating future years of data into this non-invasive modeling framework to estimate the efficacy of non-invasive sampling compared to traditional mark-recapture techniques.

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