UNDERSTANDING A FISHER REINTRODUCTION

IN NORTHERN CALIFORNIA FROM 2 PERSPECTIVES

Annual Report for 2016

For the period October 2009 through December 2016

By

Roger A Powell*[1], Deana Clifford[2], Aaron N Facka[3], David Green[3], Sean Matthews[3] and Kevin P Smith[1,3]

[1]Department of Applied Ecology, North Carolina State University, Raleigh, North Carolina 27695

[2]Wildlife Investigations Lab, California Department of Fish and Wildlife, Rancho Cordova, California 95670

[3] Institute for Natural Resources, Oregon State University, Corvallis, Oregon

*Authors in alphabetical order after the 1st author

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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to learn whether pertinent publications are now available and, if not, that the information in this report has not be superseded or updated.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 2016 (year-7). 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 35 fisher mortalities since 2009, including 9 in 2016. 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 established 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 2016, we totaled 93 captures of 53 individual fishers (34 F, 19 M), including 26 new fishers (17 F, 9 M), 23 apparent juveniles (15 F, 8 M), 1 female that had been released in year-1 and 1 male 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

Summary....2

Contents....3

Introduction....4

Terms and Definitions
....5

General Condition and Ectoparasites....5

Population Monitoring on Stirling....5

Locations, movements and home ranges....6

Survival....8

Reproduction.... 11

Population Viability Analysis.... 12

Habitat Relationships.... 12

Food Habits and Prey Population Dynamicss.... 12

Non-invasive Sampling of Klamath Fishers.... 12

Population monitoring on Stirling.... 12

Literature Cited.... 13

Publications Related To Project.... 16

Papers Presented At Conferences.... 16

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 8 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-6, 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 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 246 captures of fishers on Stirling through December 2016 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 26 (11%) as showing below average condition, 120 (49%) as average, and 100 (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-7 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, only 20% of the trapped fishers carried 2 different taxa of ectoparasites and fewer than 5% 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.

Population monitoring on Stirling

From 17 October through 15 November 2016, 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 (17 October - 30 October), then moved to the west side (02 November - 15 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 2868 trap days (1432 east, 1436 west). We totaled 93 captures of 53 individual fishers (34F, 19M), yielding 3.2% trap success (number captures per 100 trap days). This capture rate was higher than in all other years of trapping (Table 1). As we experienced in previous years, capture success was greater on the East side (4.1 captures/100 trap days, or 4.1%) than on the West (2.4%). 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 26 new fishers (17F, 9M), 3 of which were adults (2F, 1M). We saw an increase in the number of juveniles captured in 2016 (15F, 8M).

Of fishers translocated to Stirling, we recaptured 1 female released in year-1 and 1 male released in year-3. Of fishers born on Stirling, we recaptured 1 born in 2011 (1F, 0M), 2 born in 2012 (2F, 0M), 6 born in 2013 (4F, 2M), 5 born in 2014 (1F, 1M), and 14 born in 2015 (7F, 7M).

Of the 34 female fishers captured, 23 were given new collars (Telonics MOD-125 or Lotek Litetrack). Six adult males received new collars (Sirtrack Kiwisat 303, Lotek Minitrack, or Telonics MOD-125). 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 11 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 4 females 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 91 captures of non-target carnivores for a capture rate of 3.2% (Table 2), lower than capture rates for non-target carnivores in 2015 (5.0%) and 2014 (3.6%), but higher than in 2013 (2.6%) and 2012 (1.2%). The capture rate for non-target carnivores was higher on the east side (4.4%) than the west side (2.0%). Ringtail (*Bassariscus astutus*) and Spotted skunks (*Spilogale gracilis*) were the most commonly captured non-target carnivores, accounting for 58.2% of the total.

At the conclusion of trapping in 2016, the age structure of the known fishers on Stirling emphasized young fishers (Figure 4). Fishers < 2 years old comprised 63% of all fishers known to be alive. 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 4 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.

Also at the conclusion of trapping in, 2016, the minimum known population size for the fishers on Stirling was 59 (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 5). 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 5). 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).

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, most locations of fishers have occurred within the boundaries of Stirling or very near to it (Figure 6). Similarly, most den locations have occurred on Stirling. Because our research effort concentrates 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 50 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 Telonics IMP-310 very high frequency (VHF) transmitters (Mesa, Arizona) and 4 (of 9, 44%) failed prematurely (< 8 months). In year-2 and beyond, we used a mix of Telonics MOD-125 and Holohil Mi-2 collars. In years 2-7, 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 35 females during the calendar year 2016, 11 for only a few weeks after being trapped in October or November, 10 all year, and 14 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 the mountainous terrain, limited personnel, weather that limits travel and myriad other conditions that affect VHF telemetry, we rarely achieved this goal. For all females in 2016, we averaged 2.1 ± 0.8 (±SD) estimated locations per female per week for weeks each female was tracked. We averaged 58 triangulations per female per year, although for animals tracked throughout the entirety of 2016 (N=10), the average was 118 triangulations. 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 2016, we were actively tracking 24 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 2016, we followed 9 males, starting the year with 5, some of whom died or lost collars, and ending the year with 6, which included new males captured in autumn. We outfitted two male fishers with Global Positioning System (GPS) collars in 2016.

Although the batteries in the Argos collars should last over a year, many 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 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 and would not have searched. Many of these males ultimately returned to the general area of their releases. We would never have tracked those long-distance temporary forays VHF 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 167 ± 464 locations/male/year across all study years and 201 ± 392 locations/male in 2016 (Table 3). 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 44 ± 59 locations/male/year from the 2 categories with smallest error and 14 ± 15 locations in 2016 (Table 3).

Triangulations constitute most of our estimated locations of females and young males. For fishers tracked with VHF telemetry, approximately 85% of all estimated locations were triangulations. Another 4% of VHF locations were estimated from fixed-wing aircraft or a helicopter and 11% 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 >250 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 predominantly in all directions. Fisher resting under rocks, in stumps or in debris piles send signals that are truncated in both strength and direction, making them more difficult to identify and locate. 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 of 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 4). 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 1 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 7). 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 and 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 5 shows mean estimates for areas of 95% utilization distributions for 2010-2014 using h = 750, 1000 and 1500.

Table 5 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 the 2015 Annual Report, we presented examples of home ranges built using energy as the currency.

Survival

Through December 2016, we confirmed the deaths of 35 fishers (24 F, 10 M, 1 sex unknown). One female slated for release died in captivity in December 2009. During 2010, transmitter failure limited our ability to document mortality 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: 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; 2016 - 8F, 1M. Trapping in autumn 2016 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 been 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 2016.

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 (California Animal Health and Food Safety Lab at the University of California Davis,) with the assistance of Deana Clifford or Mourad Gabriel (Integral Ecology Research Center). Dr Woods 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 (Wildlife Investigations lab of the California Department of Fish and Wildlife), 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) around apparent punctures caused by possible predator canines, and polyester swabs within all apparent puncture wounds caused by possible predators. When only partial remains existed, 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, California, 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 with scant remains only, DNA from other species could have been associated with predation or scavenging.

Causes of death include drowning in a water tank, killed by bobcat, road-kill and systemic disease (vasculitis, hepatitis, hypertension and pneumonia) of unknown origin (Woods and Wengert, unpublished). Gross and histologic findings suggestive of hypoxia, hyperthermia and suffocation were documented for the translocate-candidate fisher that died in captivity but the cause of death could not be definitively confirmed (Munson, unpublished). Some fisher carcasses in poor condition yielded bobcat DNA, suggesting that these fishers had been killed or scavenged by bobcats. Predation forensics on other fishers suggest predation (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).

Anticoagulant rodenticide compounds have been documented in liver tissue of fishers. Translocated fishers could have been exposed either prior to or post-release. The finding of multiple compounds in 1 fisher indicates possible exposure from multiple source points or uses. The overall significance or potential impacts of sublethal exposure to anticoagulants for fishers and other wildlife is 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). In June, 2014, the California Department of Pesticide Regulation restricted the use of second-generation anticoagulant rodenticide 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). This regulation change has the potential to reduce non-target wildlife exposure risk from household use, especially in urban/suburban areas, but may have any impact at reducing use of anticoagulant rodenticides at illegal marijuana cultivation sites, thought to be the most likely source of exposure for fishers (Gabriel 2012, Thompson 2013). To determine definitively if anticoagulant exposure is occurring for fishers on Stirling, we shall test liver samples from recovered fishers that were born on Stirling.

We analyzed monthly survival for fishers 1 year old or older 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. In 2016, we collared 55% (29) fishers that we captured, but relatively high numbers were not collared and, therefore, not a part of these analyses. The number of fishers we did not collar has increased throughout the study as the total numbers of animal captured and population size have increased. We used Akaike's Information Criterion corrected for small sample size (AICc) to rank hypotheses that could explain the pattern of mortalities and survival that we documented (Table 6).

Because patterns in survival have not changed throughout the study, we have focused this year on evaluating potential changes in survival through time. As in previous reports, we analyzed survival first as constant through time, with only affects due to sex and to sex × age class, where age classes were defined as juveniles (<1 year old), yearlings 1≤ age <2), and adults (≥ 2 years old). Additionally, we modeled survival as changing by month × year, treating year first as a class variable, then as a continuous variable, and then with male and female survival potentially differing across study years, again with year as a class variable. In addition, because 2016 was the year with the highest number of mortalities, we modeled survival as different in 2016 compared to previous years, and then as 2016 different from other years and with male and female survival potentially differing. In previous reports, survival was best described by models that incorporated within year affects associated with the seasons of reproduction. Hence, this year we constructed 4 "reproduction" models: 2 that considered the reproductive season to be the same for males and females (April-August ; Powell & Leonard 1983], and 2 with the male reproductive season occurring March-June. For each seasonal model, we then tested whether males and females had similar rates of survival during their respective seasons or different rates of survival.

The highest ranked hypothesis suggested that males and females have different reproductive seasons but that rates of survival are similar between sexes in their respective reproductive seasons and similar in the non-reproductive season (Survive_results). Both males and females have highest rates of monthly survival during the non-reproductive season (0.988, credible interval 0.980-0.993) whereas both have reduced rates when they are attempting to mate (for males) and when females are rearing kits (0.962, CI 0.940-0.976). If fishers did not have reduced survival during the reproductive season, the annual survival rate would be 0.86. Notably, during 2016 we observed mortalities dispersed throughout the year (4 occurring after August and 1 in February), in contrast to previous years when 44% of mortalities occurred during the reproductive season. Despite documenting high numbers of mortalities in 2016, our analysis did not suggest any changes in survival across years. Our estimate of annual survival is 0.79 (CI 0.72-0.84), regardless of whether we estimate annual survival based on the reproductive season or from a simple constant model.

A critical aspect of our research on Stirling is to understand how habitat and management practices affect fisher survival and reproduction. Thus, we evaluated how well our understanding of habitat affected fisher survival through 2016. We hypothesized that fishers would have differential morality rates based on the habitats where they lived. To test this hypothesis, we first estimated the utilization distributions (UDs) of fishers in all years. We defined a year as beginning in October (ostensibly when kits are weaned and independent from their mothers) and ending on the final day in September. We estimated all UDs using all locations taken from telemetry, walk-ins, captures, and Argos collars where the error class was 1,2 or 3 (Table 4). We used a Kernel Density Estimator (KDE) with a fixed kernel, Silverman's k2, and a smoothing parameter (bandwidth) of 500 m. for most locations. For Argos locations, we set bandwidth at 500 m or to the error radius associated with the error class (1 = 1500 m, 2 = 250-1500m, and 3 = 250m), which ever was larger. From our KDE estimates, we generated isopleths that bounded 25, 50, 75, 90, and 95% of the total probability. We then estimated the mean habitat quality and the standard deviation of the habitat quality occurring in those isopleths for each animal and year. We used the Thomasma habitat suitability index (Thomasma et al. 1996) to measure habitat quality because in previous studies we found this model best explained fisher selection and avoidance of space (Facka et al 2016, Appendix). Fisher home ranges included areas not on Stirlig and; therefore, we used the Gradient Nearest Neighbor (GNN) data set (https://lemma.forestry.oregonstate.edu/about) to generate a version of the Thomasma HSI that extended beyond Stirling (hereafter "GNN HQ"). Habitat quality on Stirling also changed by year and the GNN dataset was static. Consequently, we also generated 6 Thomasma HSI models that covered Stirling onlyusing forest inventory data provided by SPI for each year from 2010 to 2016 (hereafter "Stirling HQ"). We estimated the mean and standard deviations of habitat quality from the single GNN-based model for each isopleth and across years for fishers. In contrast, we used the Thomasma habitat model from the most appropriate year to estimate mean and standard deviation of UD isopleths. Because we had 5 isopleths, 2 habitat models and 4 metrics for each model (40 potential covariates to survival) we limited our analysis of habitat quality on survival to only the 50% isopleth for each individual fisher.

We compared models of survival that incorporated habitat quality to a model of with constant survival across space. Though we know that within-year models best describe survival generally, we considered a null model sufficient to test how habitat covariates related to survival. We found that describing survival by mean habitat quality (using either the GNN-based data or the year-specific Stirling data) was less informative than the simple constant survival model (ΔAICc range 1.1-2.9). In contrast, we found that a model incorporating the standard deviation of habitat quality from the GNN data set was slightly better at describing survival than the null model (ΔAICc = 0.46). The relationship between survival and standard deviation of habitat quality was negative, suggesting that as the variation of habitat quality within the 50% isopleth of the utilization distribution increased the probability of survival decreased. Nevertheless, this estimate had low precision (β = -1.39 ± -3.02 - 0.22). Additionally, the models incorporating mean habitat quality suggested that survival increased concordant with habitat quality. As with standard deviation; however, the actual relationship was weak (β = 1.23 ± -0.82 - 3.29). Importantly, we note that many of the mortalities we documented were unlikely to be related to habitat quality. For example, the 3 females that died in water tanks most likely experienced simple accidents unrelated to habitat quality. Indeed, we note that many females that died in 2016 were found in areas with high habitat quality and relatively low levels of disturbance (specifically, logging disturbance; Figure 8). We observed this same pattern across years. Some fishers living in areas with low habitat quality died but nothing indicated clearly that habitat quality, as we have described it, was an important factor contributing to the fishers' deaths. In 2016 specifically, we observed that 3 females died in roughly the same area along Butte Creek in the center of Stirling and in relatively good habitat. These deaths may reflect the presence of high predator populations or, perhaps, one predator that was skilled at killing fishers or that specifically targeted fishers. Additionally, our habitat models may simply be insufficient to describe habitat as it relates to survival.

Because we estimated the mean and standard deviation of habitat quality occurring in the same isopleths using two data sets we evaluated how well correlated these estimates were to one another. We used linear regression to evaluate how estimates taken from GNN related to those from Stirling forest inventory data for the same isopleths. We found that mean habitat quality of the same 50% isopleth was correlated ($y = 0.10$, $\beta = 1.06 \pm 0.252$, $r^2 =$ $F_{1,108} = 0.26, 17.8$, $p < 0.001$; Figure 9). In contrast, the standard deviations of habitat quality for the same isopleth of the 2 models were not related to one another ($y = 0.003$, $\beta = -0.11 \pm 0.349$, $r^2 = 0.01$, $F_{1,108} = 0.100$, $p = 0.75$). Though r^2 for the relationship between the GNN and Stirling data was not high, the significance of the relationship indicated that, on average, the data sets identified similar habitat quality for this model. Such information may be useful for using the GNN data for areas that do not have data with high resolution. The lack of a relationship between the standards deviations suggested that differences in resolution and scale for the 2 data sets is important. The scale for the GNN data was 30x30 m pixels whereas the scale for SPI forest inventory data was forest stands.

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.

Reproduction

Fishers on Stirling have produced kits in all 7 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. Figure 10 shows the locations of dens for all years of this study through 2016.

The mean value for initial denning rate for the entire study is 0.82 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 unequivocal 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 12 females that died while they were denning (or a mean of 1.7/yr). At a minimum, we estimate that these deaths of mothers represent the deaths of 19 kits (2.7 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.

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 11A), 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 11B that we have generally found dens earlier after 2011. 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 11A). Some females never move their kits, which is not shown in Figure 11A. 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 19 adult females tracked in 2016, 17 exhibited behavior consistent with denning. Throughout the spring and summer, we documented 30 kits from 15 denning females (2.0 kits per female). Through 2016 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 2 non-collared females that appeared to have raised kits in 2016. Based on these metrics, we estimate that a minimum of 16 project females gave birth in spring 2016 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 2016.

For all adult females captured in autumn 2016, 92% had nipple sizes indicative of having lactated earlier in the year, and nipple sizes of 67% of adult females not tracked with telemetry indicated lactation (2 of 3).

We have documented females denning across Stirling, on other private lands and on national forest lands (Figure 10). Of 205 natal and maternal den trees that we found in 2010-2016 (Table 8), black oaks (*Quercus kelloggii*) were most common for both natal and maternal dens (49%; Table 8). Female fishers used incense-cedars (*Calocedrus decurrens*) second most commonly (15%), followed by Douglas-firs (*Psuedotsuga menziesii*) at 8%. Female fishers used live trees (46 of 64 dens) most often as natal dens but, later in the denning season, as kits began to travel with their mothers, females used snags, hollow logs and piles of debris slightly more often than live trees for maternal dens (70 live trees and 73 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 2 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 the Annual Report for 2015 as Appendix 3 and we refer readers to that report.

Non-invasive Sampling of Klamath Fishers

We continued to monitor the fisher population on a 587 km^2 portion of the Klamath-Siskiyou ecoregion in northern California and southern Oregon (Figure 12). In our Annual Report for 2015, we detailed analyses of the effects of removing fishers from that population and we refer readers to that section of that Annual Report. Analyses of data through 2016 emphasize interactions among the fisher, grey fox (*Urocyon cinereoagenteus*) and ringtail (*Basariscus astutus*) populations and emphasize the effects of the Beaver Creek and Happy Camp fires.

Population monitoring on Stirling

We conducted non-invasive surveys for fishers on Stirling in 2016 at the same locations as those from previous field seasons. The number of fisher genetic samples we collected in 2014 and 2015 were very few, too few in fact, to estimate the abundance of fishers using current analytical techniques. Thus, we increased our sampling effort and adjusted our protocol this year to maximize the probability of collecting a sufficient number of genetic samples from fishers for analyses. Specifically, we increased the frequency at which we visited our sampling boxes to two times per week, we stored all samples on site in a climate controlled office, and we shipped samples to the genetics laboratory every week.

Non-invasive sampling units were open from September 12th through October 26th, 2016. Each of the 96 sites were run for 42 days, for a total of 4032 trap nights. We collected a total of 610 samples that were submitted for genetic analysis to the Rocky Mountain Research Station. A sufficient amount of quality DNA was found in 560 of the 592 samples with hair (94.6%). Of these, fisher were detected from 86 samples (15.4%). We anticipate this quantity of fisher samples will prove sufficient for our intended statistical analyses.

Other species detected, in descending order of frequency, were gray fox (35.9%), black bear (23.9%), spotted skunk (10%), opossum (3.8%), mouse (2.3%), dusky-footed woodrat (2%), domestic dog (1.3%), golden-mantled ground squirrel (1.1%), striped skunk (0.89%), raccoon (0.71%), Douglas squirrel (0.36%), deer (0.36%), chipmunk (0.18%), domestic cat (0.18%), and flying squirrel (0.18%). For nine samples, DNA was obtained from the bait (chicken, 1.6%).

We will analyze the individual identifications of fishers using spatial capture-recapture techniques when those data become available.

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