



SIERRA NEVADA ADAPTIVE MANAGEMENT PROJECT (SNAMP)

Appendix D: Fisher Team Final Report

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i. Executive Summary

Fishers (*Pekania pennanti*) are a medium-sized mammalian carnivore with a pre-European distribution encompassing the boreal forest zone of Canada, the Great Lakes region and northeastern United States, a relatively limited portion of the Rocky Mountains in the United States, and mountainous areas of Washington, Oregon, and California, USA (Powell 1993). Ecologically, fishers are a mature or old forest-obligate species (Zielinski et al. 2005), and in central to eastern Canada and the northeastern United States their numbers were reduced historically by the combination of intensive trapping and loss of forest habitats (Powell and Zielinski 1994). The species is uncommon to rare in the western United States, and is a candidate for listing under the US Endangered Species Act. The California Department of Fish and Wildlife (CDFW) is reviewing the status of fishers in the state, with recommendations concerning listing to the Commission expected in 2014. In advance of listing decisions, conservation planning has been underway since 2013 to develop an approach to maintaining viable populations of fishers in both northwestern California and in the southern Sierra Nevada. Information from the SNAMP Fisher Project (published manuscripts, submitted manuscripts, and unpublished data) described herein has been included in a Southern Sierra Nevada Fisher Conservation Assessment developed by the Conservation Biology Institute, with input from a team of 13 fisher researchers and scientists.



Illustration 1: Image of an adult female fisher on a den tree in spring 2009.

The SNAMP Fisher Project was initiated by the UC Berkeley Fisher Science Team in Fall 2007 in association with multiple other SNAMP science programs designed to provide an independent evaluation of how vegetation management, prescribed by the 2004 Sierra Nevada Forest Plan Amendment, affects fire risk, wildlife, forest health and water. Fuel reduction management includes a mix of activities including mechanical mastication of shrubs and small trees, hand thinning/precommercial thinning, commercial thinning, and controlled burning.



Illustration 2: Fuel reduction management treatments observed in the SNAMP Fisher Study area; *mastication/mowing, control burning, commercial thinning*

The range and number of fishers in the Sierra Nevada declined by over 50% after the early 1900s (Spencer et al. 2014). A major goal of the SNAMP Fisher Project was to determine whether or not current rates of survival and reproduction will allow fishers to persist in the Sierra Nevada in the context of active forest management to reduce fuels and the risk of catastrophic wildfire. Toward this end, in October 2007 the SNAMP Fisher Team initiated fieldwork by placement of survey cameras (camera traps; O’Connel et al. 2011) in the focal study area referred to as the four “Key Watersheds” (Nelder Creek, Sugar Pine, White Chief, Rainier Creek) where the Before-After-Control-Impact (BACI) design for the larger SNAMP science program would be centered. Based on information provided by the Bass Lake Ranger District of the Sierra National Forest, three different fuel reduction projects were planned during the course of the study: Sugar Pine (main focus of SNAMP), Cedar Valley (started in fall/winter 2007-08), and Fish Camp (started in 2010-11). Our approach for assessing how fishers would respond to Strategically Placed Landscape Area Treatments (SPLATs) was designed to be multifaceted including (1) life history responses to fuels reduction (changes in survival, reproduction/fecundity, lifespan), (2) changes in local scale habitat use within individual home ranges, and (3) shifts or changes in habitat use at the home range scale of animal resource use/resource selection. The study required capturing, radiocollaring, and monitoring multiple individual fishers, which were monitored using high intensity aerial radiotelemetry to identify deaths and quickly recover carcasses for necropsy, and repeated camera surveys within and around the Key Watersheds. Specific objectives included:

1. *Determination of all key demographic parameters including age- and sex-specific survival, reproductive rates, and fecundity, and metrics on dispersal and movements*
2. *Identify population limiting factors based on cause-specific mortality due to*

predation, disease, and human-linked factors such as roadkill on local highways

3. *Evaluate the effects of SPLATs on occupancy, survival, and fecundity*

The SNAMP Fisher Project study area is at the northern edge of the southern distribution of fishers in California, encompassing the area bounded by the Merced River in the north and the San Joaquin River in the south. Administratively, the study area was within the Bass Lake Ranger District in the Sierra National Forest, but early in the study a radio-collared fisher dispersed north into Yosemite National Park, which effectively expanded the study to encompass the southern area of Yosemite National Park. The overall study area encompassed approximately 1300 km² of a topographically complex landscape with elevations ranging from 758 m to 2652 m. The smaller focal study area (Key Watersheds) was located in the approximate center of the study area, and the Key Watersheds entirely encompassed the firesheds designated for the SNAMP BACI design.

A range of standard methods were used in the study to live-trap, radiocollar and monitor survival status of individual fishers. Monitoring was accomplished almost entirely by fixed-wing aerial radiotelemetry, supported by an “in house” aviation program developed specifically for SNAMP Fisher and administered by the USDA Forest Service. Ground-based radiotelemetry was used to monitor female fishers during denning seasons, and to recover carcasses of deceased fishers. Camera traps were systematically placed in the Key Watersheds and elsewhere in the study near the center points of 1-km² grids. Camera traps in the Key Watersheds were surveyed for fisher activity in each year of the study, whereas those placed elsewhere were not (some of the “external Key Watershed” grids were surveyed in multiple years when other forest management projects had occurred). Three “Management Indicators” were developed and assessed annually to provide stakeholders and managers with information on the status of the fisher population in the study area. The indicators were designed to link to local, home range, and landscape scale responses by fishers to forest management activities occurring in the Bass Lake Ranger District. We also developed five different focused science efforts related to fisher ecology considered relevant for fisher management and conservation. These “Focused Science Topics” related to the use of camera traps for identifying gender of fishers, evaluating fisher activity patterns, and evaluating the distribution of several forest carnivores in relation to roads or other species. One of the focused science topics addressed roadkill mortalities of fishers on a major highway (Highway 41) that bisected the study area.

Surveys with camera traps were completed in 905 unique 1-km² grids throughout the overall study area, including 56 grids within the southern region of Yosemite National Park from a companion

study funded by the California Department of Fish and Wildlife. Fishers were detected in 448 of the unique grids surveyed, which helped to identify that fishers in this part of the southern Sierra Nevada were most common between 4500 and 6500 feet elevation (1372 and 1981 m elevation). Occupancy estimates for multi-year surveyed grids corrected for imperfect detection < 1.0 ranged from 0.62 to 0.80. Detection rates for fishers at camera trap stations were much higher in the fall, winter, and spring seasons compared to summer, likely due to availability of a more abundant and diverse prey base in summer compared to winter especially.

A total 110 individual fishers were captured and radiocollared Project from Dec 2007 to Dec 2013 (62 females, 48 males). Sixty-six (60%) of the 110 individual fishers radiocollared during the study were known to have died, including 32 females and 34 males. On average 10.5 radiocollared fishers died in each population year over the course of the study, and the most common cause of death was predation by felid carnivores (bobcats, *Lynx rufus*, and mountain lions, *Puma concolor*). Four deaths were caused by, or associated with, canine distemper virus in 2009 when a relatively small scale epizootic event occurred in the study area. Other disease deaths included Toxoplasmosis and septicemia. Septicemia-linked deaths were caused by injuries the animals suffered weeks or months before death that led to infection that sometimes contributed to starvation/emaciation. Two radiocollared fisher deaths were roadkills on Highway 41. Four others were directly linked to anticoagulant rodenticides being used in association with illegal marijuana grow sites in the Sierra National Forest, and a fifth mortality is suspected yet currently unconfirmed.

The SNAMP Fisher study generated information on all key vital rates needed to evaluate the population growth rate (λ) and for understanding whether the population has the potential to persist. We developed an age-structured matrix model to estimate a series of five deterministic population growth rates (λ) for the SNAMP Fisher study population using the observed, “empirical” data on denning rates and litter sizes (fecundity), and survival. Estimates of survival and fecundity were produced for five 2-year groups/pairs of years starting with population years 2008-09 and 2009-10 and ending with 2012-13 and 2013-14. The Leslie-matrix population model was used to integrate data on fisher survival for three age classes, and fecundity for four female age classes: juveniles and subadults (non-reproductive), and young and mature adults (reproductive). Estimates for λ for the SNAMP Fisher study area were below 1.0 in two 2-year groups (population decline), equal to 1.0 in one 2-year group (stable), and slightly positive in two 2-year groups (increasing population). Lambda across all years was 0.90, which was suggestive of general population decline, however, the annual and cumulative 95% confidence intervals all overlapped with 1.0.

Prior to the SNAMP Fisher study there was limited information on the distribution and abundance of fishers at the north margin of their extant southern range. Many years of survey-based research with cameras and track plates conducted by the US Forest Service Region 5 and the Pacific Southwest Research Station suggested that the population in the SNAMP Fisher study area was likely sparse (low density), and there had been no indication that “surplus animals” were dispersing northward into suitable, unoccupied habitat north of the Merced River in Yosemite Valley. We used resightings of individual radiocollared fishers in a Robust Design capture-mark-resight framework (CMR) to estimate the fisher population size and density in the overall SNAMP Fisher study area. The SNAMP Fisher study area corresponds to the “Fisher Habitat and Core Connectivity area 5” being used for conservation planning. Fisher population size ranged from 48.2 in 2010 to 61.8 in 2012, whereas mean population density ranged between 0.072 fishers/km² in 2010 and 0.093 fishers/km² in 2012. We considered data from other studies in California and elsewhere that used CMR methods similar to ours, and determined that the population density for fishers in the SNAMP study area was the lowest reported for the continental United States. Also, and in support of conservation planning, we used the mean density from the study to estimate that there were 93 fishers (range 80-107) in the Southern Sierra Nevada Habitat Core and Connectivity area 5.

Den cameras used in association with ground-based monitoring provided detailed information on the activities of 32 different individual adult female fishers during six spring denning seasons. Denning and reproduction in the SNAMP Fisher study area typically began in the last week of March, and adult female fishers ceased regular use of den trees in the first week of June. The earliest and latest known regular use of den trees was March 22, and June 20, respectively. Seventy-six (85%) breeding-age female fishers either exhibited denning behavior ($n = 63$) or were determined to have denned and weaned at least 1 kit. Among the 76 breeding-age females that initiated denning, 64 (84%) were identified as weaning kits. Overall, 72% of adult female fishers for which reproductive status was known produced at least 1 weaned kit. Eleven (17.5 %) of 63 cases of denning for females that were monitored during spring periods failed prior to kits being weaned. Eight den failures were due to death of the denning female; 7 denning females were killed by predators and 1 died after exposure to rodenticides. We were able to determine litter size for 48 of 59 denning females. A total of 73 kits were known produced, with an average litter size of 1.5. After accounting for known mortalities, we estimated that 64 of the 73 kits produced were weaned from den trees, whereas seven kits died or would have died had they not been rescued.

The availability of suitable den structures is a critical, and potentially limiting, feature of fisher

habitat The mean number of den trees used per female per denning season was 2.4 (range 1 to 5). We identified 125 unique structures used as natal or maternal dens, including 54 black oak trees, 41 incense cedar trees, 19 white fir trees, 10 sugar pine or ponderosa pine trees, and one canyon oak (*Quercus chrysolepsis*). We discovered that repeat use of den trees was not uncommon. Sixteen individual den trees were used more than once; 15 trees were used in two years, and one tree was used in four different den seasons. In most cases of repeat den tree use the same individual reused one or several den trees between successive years, but in two cases a female used a den that had been used by a different female in a previous year. Fifty-six percent of the unique individual trees used for denning in the SNAMP area were live trees ($n = 70$), whereas 44% ($n = 55$) were snags. Black oak was the most common tree species used for denning (live or snag; 43%), but a high percent of incense cedar were also used for denning (33%). Among snags used as denning structures, black oak and incense cedar were both commonly used (18% each), whereas white fir and pines (sugar pine or ponderosa pine) were less common as snag-type den trees (4% each). Overall mean DBH of black oak denning structures was 74.4 cm, 115.6 cm for incense cedar, 108.3 cm for white fir, and 111.2 cm for the two large pine species (sugar pine and ponderosa pine; Table 18). Mean heights of live trees were taller for live trees compared to snags of the same species, reflecting that many of the snags used for denning were at advanced stages of decay. The majority of denning structures used in the SNAMP Fisher study area (83%) were in the elevation range 4500 feet (1371 m) to 6000 feet (1829 m), and denning structures were typically embedded in areas of high canopy cover (mean = 72%). Shrub cover and aspect near den trees was variable, and most den trees had multiple large down trees/logs nearby, whereas concealment cover to the base of den trees averaged more than 45%. Also, information on denning habitats near den trees from high resolution Lidar (Zhao et al. 2012) identified that fishers selected den sites with tall trees and steep slopes within a 10-m radius of the den tree, and denning areas were associated with high levels of forest structural complexity and clusters of multiple large trees within 30-50 m.

Dispersal behavior by fishers is of high management interest in California because the southern Sierra Nevada population does not appear to be expanding geographically despite changes in management promoting restoration of suitable fisher habitat. Also, dispersal movements by fishers are potentially inhibited by exposure to multiple restrictive habitat elements (burned forests) and landscape features (steep river canyons). The SNAMP Fisher Project used a combination of data on juvenile and adult home ranges, and maternity assignments based on microsatellite DNA analyses to assess natal dispersal by young fishers based on Euclidean distance between natal areas and subadult or adult home

ranges. We also applied an “expert” cost surface to the landscape, and used Least Cost modeling approaches to predict more realistic dispersal paths/distances based on presence of restrictive habitat or landscape elements considered aversive to fisher movement and life history. The combination of field (juvenile home ranges) and genetic data (maternal assignments) allowed us to assess dispersal for 43 (74%) of 58 juvenile or young subadult fishers in the study population. The average Euclidean distance natal dispersal for female fishers was 5.8 km, compared to 9.8 km for male fishers. The longest Euclidean distance dispersal for a female fisher was 24.5 km, compared to 36.2 km for a male fisher. Although male fishers tended to disperse longer distances than females, the difference was not significant. One male fisher from the Kings River Fisher Project on the High Sierra District of the Sierra National Forest moved across the San Joaquin River canyon and immigrated into the SNAMP Fisher Study area. The Euclidean distance for this dispersal movement was over 36 km, but the more likely Least Cost dispersal path was predicted in the range of 67-69 km. In general, we found very limited evidence for male-biased natal dispersal according to any of the typical metrics reported in the literature for this life history phenomenon. Dispersal distances were not longer for males compared to females based on either Euclidean distances or more realistic Least Cost movement paths, and there was no difference in the proportion of each gender that dispersed, or that remained philopatric. Timing of dispersal was focused during mid-February into July when 80% of dispersal events were initiated and subsequently completed.

Management indicator 1 (occupancy/presence of fisher detections in 1-km² grids within the Key Watersheds), ranged from a low of 53% in 2012-13 to a high of 76% in 2011-12 (mean: $62 \pm 9.2\%$). An index of fisher activity developed for Management Indicator 1 indicated that the estimated detection rate (detections/100 camera survey days) ranged from 10.5 in 2010-11 to 18.6 in 2012-13 (mean: 13.9 ± 2.9). Camera year 2012-13 was atypical in that many grids in the Key Watershed were surveyed during summer when detection rates are significantly lower compared to fall and winter, and it was therefore possible that the low detection rate for 2012-13 was related to timing of surveys.

Management Indicator 2 (number of resident subadult and adult fishers using the Key Watersheds) identified an overall average of 5.0 subadult or adult females and 2.0 subadult or adult males using the Key Watershed focal study area. For both sexes, the number of resident fishers using the focal study area ranged from 6.2 to 7.7, and the variation among years was minor.

Management Indicator 3 (adult female survival in the study population): For this report we expanded the original Management Indicator 3 to estimate survival for adult female fishers for a sequence of 2-year groups of demographic data and included results for juvenile and subadult females,

and estimated population growth rates. Adult female survival ranged from a low of 0.69 in Year group 3 to a high of 0.86 in Year group 4. Relatively low levels of survival and reproduction suggested the population was in decline ($\lambda < 1.0$) between 2008 and 2010, stable between 2010 and 2012 ($\lambda \approx 1.0$), and increasing by 3-4%/year during 2012 to 2014 ($\lambda = 1.04$ and 1.03 , respectively).

Occupancy modelling indicated that fishers reduced their use of forest patches exposed to higher levels of restorative fuel reduction; i.e. persistence of occupancy declined with additional acreage treated for fire resiliency. However neither restorative nor extractive (i.e. commercial thinning) fuel reduction was related to either initial probability of occupancy or local extinction. This pattern is likely due to interaction of several factors. First, the overall spatial scale of treatments, both restorative and extractive, is relatively small compared to a fisher's home range. Second, evidence indicates that fishers simply shift their space use patterns to avoid small treated areas. And third, evidence indicates that the reduction of surface and ladder fuels may change the small mammal community, therefore limiting fisher prey availability.

We found that SPLATs caused an immediate 6% reduction in potential fisher habitat. However they also moderated the impact of fire, resulting in greater available fisher habitat within 30 years. In the absence of simulated fire, the amount of habitat steadily increased over time due to forest succession, and was actually slightly greater on the treated landscape in year 30 than in year 0. The net benefits of SPLATs for the Pacific fisher will depend upon the true, but unknown, probability that high-severity fire effects will occur on a given portion of the landscape. However, future probabilities for specific fire behaviors (e.g., crown-fire initiation) are difficult to estimate, and it is therefore difficult to quantify trade-offs associated with SPLATs in absolute terms (Finney 2005). We further note that the SPLATs which were implemented at Sugar Pine appeared to have relatively modest impacts on forest structure and simulated fire behavior, and that it may be necessary to evaluate additional SPLATs of different intensities over a larger scale to fully assess the effects of SPLATs on fisher habitat.

Fishers have been the focus of systematic monitoring in the southern Sierra Nevada since the mid-1990s. Recent analyses of baited track plate detection histories from 2002 to 2009 found no evidence that the population trajectory for fishers in the area has been significantly positive or negative, based on constant and positive persistent values (Zielinski et al. 2013). In contrast, recent genetics research suggests that the fisher population in the SNAMP Fisher study area was produced by a significant post-1990s population expansion involving dispersal of animals from south of the Kings River (Fisher Core Habitat Area 4) (Tucker et al. 2014). However genetic data are not typically used to

make inferences about population processes over extremely short periods in evolutionary time, and an expansion of such magnitude (approx. 30% range increase) would require a significantly positive population growth rate.

The combination of an overall negative population growth rate and the relatively small estimated number of fishers in Fisher Core Habitat area 5 ($n = 93$, range 80-107), warrants concern for the long term viability of the fishers in the region. Any small population will be at high risk to stochastic events such as disease and large perturbations to critical habitats (e.g. forest fires or drought; Noss et al. 2006), and genetic limitation resulting from genetic drift after founder events (Tucker et al. 2014) will hinder population recovery and expansion (Reed et al. 2003). Minimum viable population size has been under debate (Shoemaker et al. 2013, Reed and McCoy 2014), but at <500 individuals (Spencer et al. 2014), the current southern Sierra Nevada fisher population will likely require active management and conservation measures to maintain a positive growth rate across its entire range. The estimated population growth rate in the SNAMP Fisher study area reaffirms the vulnerability of the small, isolated population to external threats (Spencer et al. 2014), especially wildfires that are likely to increase in frequency and intensity with climate change. Moreover, the SNAMP Fisher study spanned a limited period of six years during which multiple novel threats to fisher survival within the study area were identified, and when three large wildfires significantly reduced availability of suitable habitat for fishers immediately to the south and north of the study site. We recommend continuous monitoring of the status of fisher populations in the southern Sierra Nevada region. Development of ways to mitigate for major threats to fisher survival and fisher habitats and population viability analyses are necessary for evaluating the long-term prospects for fishers in the southern Sierra Nevada. Data from the SNAMP Fisher study have provided important new insights on the status of a fisher population at the north margin of their current distribution in the southern Sierra Nevada Range, which will be useful towards developing a comprehensive conservation strategy for fishers in California.

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Introduction

Fishers are a medium-sized mammal carnivore with a pre-European distribution encompassing the boreal forest zone of Canada, the Great Lakes region and northeastern United States, a relatively limited portion of the Rocky Mountains in the United States, and mountainous areas of Washington, Oregon, and California, USA (Powell 1993). Ecologically, fishers are a mature or old forest-obligate species (Zielinski et al. 2005), and in central to eastern Canada and the northeastern United States their numbers were reduced historically by the combination of intensive trapping and loss of forest habitats. Although fishers have recovered portions of their former range in this region aided by reintroductions and more sustainable levels of timber harvest (Lewis et al. 2012, Powell et al. 2003), they remain uncommon to rare in the western states and provinces of the USA and Canada (Lofroth et al. 2010).

The fisher of the US Pacific states, or the West Coast Population Segment, is a candidate for listing under the US Endangered Species Act and is the target of recovery and conservation efforts (Lewis et al. 2012). The US Fish and Wildlife Service (USFWS) is under court order to make a proposal to list (or not) the West Coast population by September 2014 and to make a final decision to list, if proposed, by September 2015 (Center for Biological Diversity 2013). The fisher is also a candidate for listing under the California Endangered Species Act pursuant to a 2012 court order that compelled the California Fish and Game Commission to set aside its 2010 finding that listing was not warranted (Center for Biological Diversity v. California Fish and Game Commission et al. 2012). The California Department of Fish and Wildlife (CDFW) is reviewing the status of fishers in the state pursuant to the court order, with recommendations concerning listing to the Commission expected in 2014 (Spencer et al. 2014).

In the west coast states of the USA fishers currently exist in three remnant populations in southern Oregon, northern California, and the southern Sierra Nevada, California (Zielinski et al. 2005). In California the fisher occupies less than half of its historical range as described by Grinnell in the early 1900s (Grinnell et al. 1937). The decline in range extent and abundance of fishers in California into 2 remnant populations geographically separated by around 400 km had been considered due to widespread timber harvest and fur trapping during the early to mid-1900s (Zielinski et al. 2005), but recent genetic research suggests that the northern California and southern Sierra Nevada populations may have been genetically isolated prior to European settlement (Tucker et al. 2012). Notwithstanding uncertainty regarding the timing or cause of the range retraction, there may be fewer than 500 total fishers in the southern Sierra Nevada population (Spencer et al. 2011), where the species currently occupies approximately 4,400 km² of mid-elevation, mixed-coniferous forest (Spencer et al.

2014). While fishers in the western US are considered at risk of extirpation from disease and other factors (Lofroth et al. 2010; Spencer et al. 2014), the recent reintroduction of fishers at one site in Washington state and another in northern California is promising for maintaining the species.

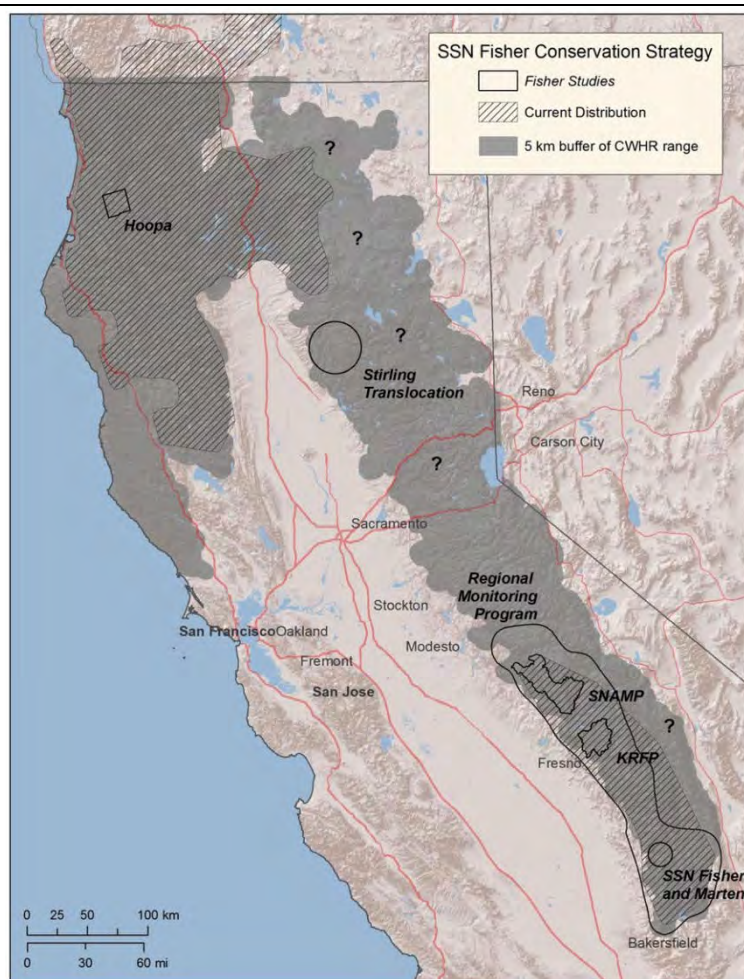
Fisher Range Size and Trends

Grinnell et al. (1937) described the original range of the fisher in California as including the entire western slope of the Sierra Nevada, the southern Cascades, Klamath Mountains, and northern Coast Range, a total area of ~100,000-110,000 km² (Spencer et al. 2014). Lofroth et al. (2010) estimated that the current range of the fisher in California represents <50 percent of the historical range, and fishers are currently absent from most of the northern and central Sierra Nevada, leaving a ~400-km gap separating the two populations in the state (Zielinski et al. 1995) (Fig. 1), one in the northern Coast Range and one in the southern Sierra Nevada (Spencer et al. 2014). Recent analysis suggests that these two population regions may have been genetically isolated prior to European

Figure 1. Current estimated distribution of fisher habitat in California, and location of major fisher field studies (Table 1 provides details on the field studies). Current distribution based on minimum convex polygon enclosing recent (>1970) fisher locality points from a comprehensive USFWS fisher locality database; map and methods used to produce the distribution boundaries are described in the Southern Sierra Nevada Fisher Conservation Assessment (Spencer et al. 2014). Question marks illustrate uncertainty on the degree to which eastern and northern portions of historical range were actually occupied.



Illustration 3: Camera trap image of a female fisher from SNAMP.



Map source: *Spencer et al. 2014*

settlement of California (Knaus et al. 2011, Tucker et al. 2012). Spencer and Zielinski (in review) used an updated fisher locality database to estimate their current geographic range in California at 55,000-60,000 km², with ~45,000-50,000 km² in northern California and 10,000-12,000 km² in the southern Sierra Nevada. Although the range areas estimated by Spencer and Zielinski (In review) included a mix of suitable and unsuitable habitats, the analysis suggested a 30-50 percent reduction compared to the historical range of the species. Caveats included that there is uncertainty about how wide “the gap” was historically, and how much of the mid elevation forest areas in the northern and central Sierra Nevada were actually occupied (Spencer et al. 2014).

Table 1. Major recent and ongoing field studies focused on the distribution, population biology and habitat use/requirements for fisher populations in California.

Study name	Location	Period	Brief description of research focus
Sierra Nevada Adaptive Management Project (SNAMP Fisher)	Bass Lake Ranger District, Sierra National Forest	2007-2013	Comprehensive study; population biology, space use, responses to vegetation management
USFS PSW Kings River Fisher Project (KRFP)	High Sierra Ranger District, Sierra National Forest	2007-2018	Comprehensive study; population biology, space use, responses to vegetation management
USFS PSW Sugar Pine Fisher Project (SPFP)	Bass Lake Ranger District, Sierra National Forest	2014-2016	Continuation of the SNAMP fisher study effort to document post-treatment population status.
USFS PSW Fisher Regional Monitoring Program (Regional monitoring)	National Forests in the southern Sierra Nevada: Stanislaus, Sierra, Sequoia, Inyo	2002-present	Landscape-level occupancy monitoring
Southern Sierra Fisher and Marten Study (SSN fisher and Marten)	Tule River Ranger District, Sequoia National Forest, Tule River Indian Reservation	1994-1996	Comparative study; marten and fisher home range, habitats, diets, and interspecific competition
California Department of Fish and Wildlife Fisher Translocation (Stirling translocation)	Stirling Management Area of Sierra Pacific Industries, Butte, Tehama counties	2009-present	Monitoring of introduced fishers; population biology, habitat and space use
Hoopa Valley Fisher Study (Hoopa Study)	Hoopa Valley Indian Reservation, Humboldt County	2004-present	Comprehensive study; population biology, space use, responses to vegetation management

Population Size and Trends

The southern Sierra Nevada fisher population is small (≈ 500 total individuals and < 300 adult fishers; Spencer et al. 2011), but appears to be stable over about the past decade (Zielinski et al. 2013). Following substantial population contractions in the past (Knaus et al., 2012), fishers in this part of California may have expanded in the late 20th century (Tucker et al. 2014). The overall distribution of fisher in the southern Sierra Nevada has been monitored by a mix of track plates and motion detecting cameras since the mid-1990s (Truex et al. 1998, Zielinski et al. 2005, Jordan 2007). Zielinski et al. (2013) analyzed occupancy records from this effort for the period 2002 to 2012, when a systematic survey design was in place, and found no detectable change in occupancy for the entire area or for any of the three subareas examined (Zielinski et al. 2013). The Zielinski et al. (2013) analyses suggest that despite fishers being protected from fur harvest for over 60 years during a time when large scale clearing of forest habitat was diminished (Collins et al. 2010), this population isolate is not showing significant evidence of numeric or spatial recovery. However, genetic patterns and survey data suggest that the population north of the Kings River may have expanded during the 1990s, before the regional monitoring program was established (Tucker et al. 2014).

Insight from prior research in the High Sierra District, Sierra National Forest (KRFP study, ≈ 60 Km southeast of the Key Watersheds) suggests that fisher population densities range from 0.07 to 0.28 fishers/km² (Jordan et al. 2011, Thompson et al. 2012). Records from research in northern California (Hoopa Study) indicate the potential for fisher densities to change rapidly. In the Hoopa Valley area of Northern California, fisher densities were estimated at 0.52 fishers/km² in 1998, but fell to 0.14 fishers/km² in 2005 (Matthews et al. 2013). Due to the apparent variability in density estimates, developing precise density estimates for different subpopulations and in different habitat types is critical for effective management.

Management and Conservation Planning

Federal and state resource agencies are currently developing strategies to aid in the maintenance of viable populations of fishers in both northwestern California and in the southern Sierra Nevada. As part of a cooperative agreement between the Conservation Biology Institute and USDA Forest Service Region 5, and with input from a team of 13 fisher researchers and scientists, a conservation strategy for fishers in the southern Sierra Nevada has been developed. The “SSN Fisher Conservation Strategy” is based on the findings from a conservation assessment that was previously completed by the same team of scientists. The SSN Fisher Conservation Assessment (Spencer et al.

2014) included a review of all previous published and credibly collected unpublished data on fisher ecology in the southern Sierra Nevada Region. Information from the SNAMP Fisher Project (published manuscripts, submitted/draft manuscripts, and unpublished data) were included in the SSN Fisher Conservation Assessment. In the Conservation Assessment and in association with conservation planning, the SNAMP Fisher overall study area is essentially Fisher Habitat Core Area 5 (Fig. 2).

Insights from SNAMP Fisher appear simultaneously encouraging and discouraging for management and conservation of the species. Causes of mortality were more diverse than was previously known, including evidence for periodic outbreaks of disease, and significantly higher levels of predation than previously documented for any other intensively studied population in North America. Fishers are challenged by the need to cross busy roads passing through foraging and

Table 2. Details and characteristics of Southern Sierra Nevada Fisher Conservation Assessment area “Fisher Core Habitat areas.”		
Core, Status ^a	Total area (suitable habitat) ^b	Primary (secondary) jurisdiction
1. Occupied	430 km ² (50.4%)	Sequoia NF
2. Occupied	936 km ² (62.2%)	Sequoia NF
3. Occupied	985 km ² (56.4%)	Sequoia NP (Sequoia NF)
4. Occupied	751 km ² (55.1%)	Sierra NF
5. Occupied	1096 km ² (57.4%)	Sierra NF (Yosemite NP)
6. and 7. No fishers	1678 km ² (55.8%)	Yosemite NP (Stanislaus NF)
^a Indicates whether known occupied by a breeding fisher population.		
^b Habitat suitability based on updated modeling for the Southern Sierra Nevada Fisher Conservation Assessment.		

Table 2 and Figure 2 Adapted from Southern Sierra Nevada Fisher Conservation Assessment (Spencer et al. 2014)

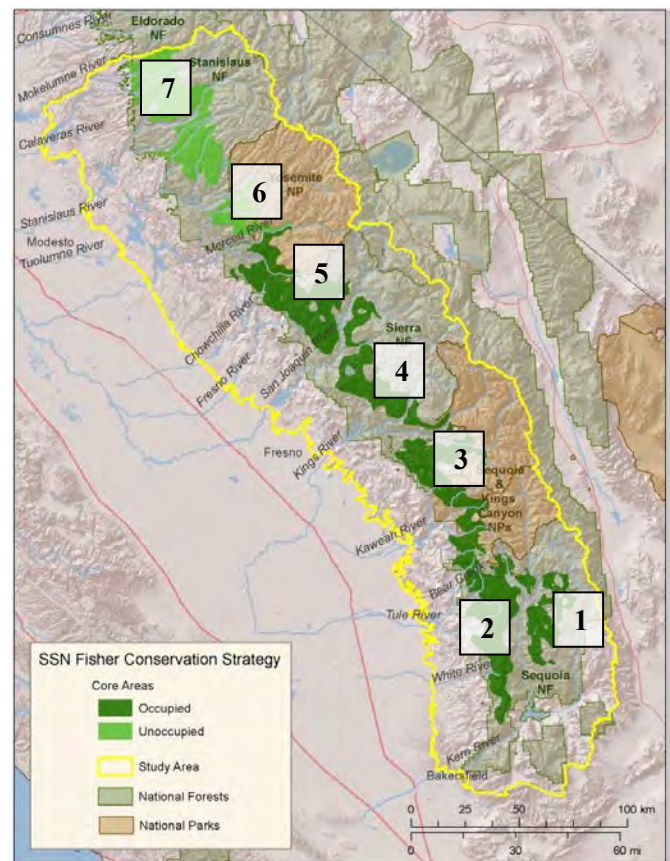


Figure 2. Distribution of occupied and unoccupied fisher habitat core areas in Southern Sierra Nevada Fisher Conservation Assessment area. Habitat cores were mapped as contiguous polygons having a predicted probability of fisher occupancy exceeding 0.41, and large enough to support ≥ 5 adult female fishers (see Spencer et al. 2014 for details).

denning habitats, they must avoid accidental entrapment in human structures, and coexist with illegal marijuana farmers spreading poisons on the landscape that kill or sicken them and their prey. Encouraging results include that, even with these challenges, fisher survival and demography at our southern Sierra Nevada study sites was comparable to other closely monitored populations in northern California and southern Oregon not considered at imminent risk of extirpation. Management should continue to maintain suitable denning, foraging, and resting habitats, as detailed in current planning documents (North et al. 2009). Moreover, the SNAMP Fisher study spanned a limited period of six years when multiple threats to fisher survival within Fisher Habitat Core Area 5 were identified, and when three large wildfires either significantly reduced availability of suitable habitat for fishers to the south and north of Core Area 5. We believe that continuous monitoring of the status of this population and mitigation of the major threats to persistence, along with population viability analyses, are necessary for evaluating the long-term viability of fishers in the southern Sierra Nevada.

Forest Management and Fisher Populations

Fishers were formerly widespread in mixed conifer forests across mountainous areas of northwestern California and in the Sierra Nevada of eastern California. Populations in the Sierra Nevada appear 30-50% reduced (Spencer et al. 2014) and it is possible that the isolated population of fishers in the southern Sierra Nevada will be impacted as the USDA Forest Service implements fuel reduction measures (Strategically Placed Land Allocation Treatments; SPLATs) to mitigate risk of catastrophic wildfire (Scheller et al. 2011). Fuel reduction treatments are becoming the dominant forest management activity in western forests in response to increases in the frequency of intense, stand-replacing forest fires over the past several decades (Mallek et al. 2013, Safford 2013). Advances in fire modeling have greatly improved managers' ability to plan and evaluate various landscape fuel treatment scenarios intended to reduce fire risks (Collins et al. 2010, Scheller et al. 2011). However, there remains a considerable gap between modeling landscape-scale fuel treatments and implementing them due to concern over the status of rare and uncommon species associated with multi-storied, late-seral stage forests, such as the fisher and spotted owl (*Strix occidentalis occidentalis*) (Naney et al. 2012, Truex and Zielinski 2013). Presence of fishers has strongly influenced managers' ability to delineate landscape-scale fuel treatments in this fire-prone region (Collins et al. 2010, Weir and Corbould 2010). The amended Sierra Nevada Forest Plan represents the most recent attempt to reconcile the need to reduce fuel loadings in Sierra Nevada mixed-conifer forests and retain characteristics of late-successional forests that are important for these species. The strategy involves a

network of “Strategically Placed Land Area Treatments” (SPLATs) that allow up to a 60% reduction in basal area and a 30% reduction in canopy cover in habitats used by fishers and spotted owls. In the long-term, this strategy may increase availability of important habitats for both organisms by reducing wildfire-induced losses (Spencer et al. 2011), but treatments may impact habitat quality for fishers in the near-term (Thompson et al. 2011).

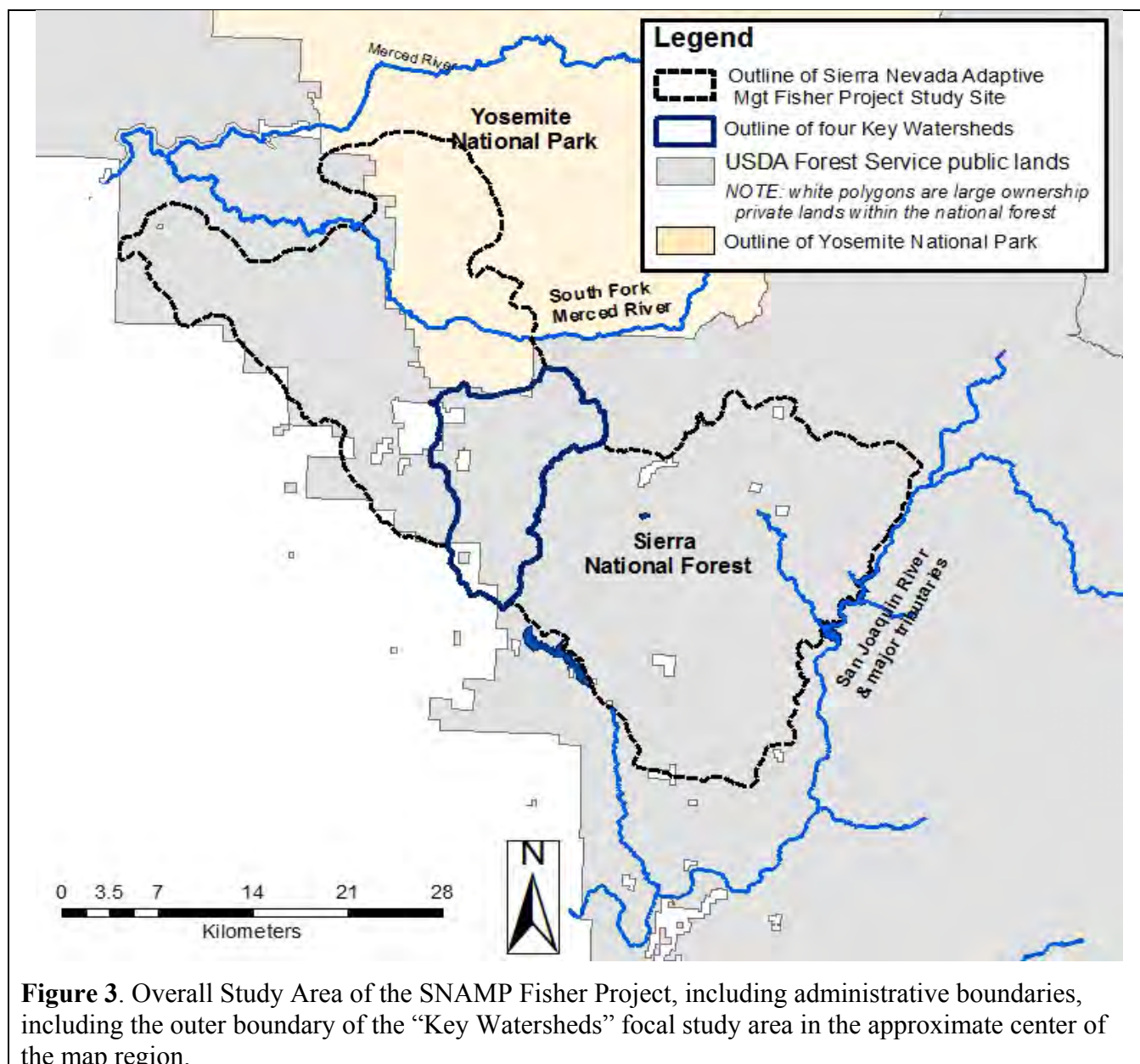
To provide a framework for balancing the habitat needs of fishers with fuel treatments intended to reduce fire risks, SNAMP initiated a coordinated effort to assess the effects of fuel treatments on many environmental features including the fisher, spotted owl, forest health, and water quality as quantity in the central Sierra Nevada. SNAMP began in 2007 and was designed to evaluate the effects and effectiveness of fuel treatments implemented according to the revised Sierra Framework (USFS 2004) under a design of stakeholder participation. SNAMP is a landscape-scale, ecosystem-level experiment in natural resource management and involves a Before-After-Control (BACI) design developed specifically to assess the impacts of SPLATs on the overall forest ecosystem (Popescu et al. 2012).

In September 2007 SNAMP Fisher was launched as 6-7 year study of fishers in the Bass Lake Ranger District, Sierra National Forest to determine population limiting factors, and to evaluate the effects of SPLATs on resource use, survival, and persistence of fishers in the southern Sierra Nevada. We used repeated surveys of small 1 km² blocks of forest habitat with automatic cameras, mark-recapture, and intensive monitoring of individual collared fisher for evaluating how SPLATs contribute to changes in habitat use, dispersal, survival, and reproduction by fisher.

Science Goals and Primary Objectives

1. Estimate population parameters including age and sex-specific survival, and fecundity
 - a. What are the vital rates (reproduction, survival, population growth rates)?
 - b. What is the population size and density in the study area?
 - c. What are the patterns of dispersal movements?
2. Identify population limiting factors in the region encompassed by the study area
 - a. What are the causes of mortality? Are predators, parasites or diseases important?
 - b. What are the patterns of home range and habitat use?
3. Evaluate effects of SPLATs on occupancy, survival and fecundity
 - a. Characterize resource use by fishers; do SPLATs influence habitat use
 - b. What are the patterns of fisher occupancy in relation to forest management?
 - c. Do patterns of fisher occupancy change before and after by SPLATs?

Project planning for the study was initiated in early 2007, and in September 2007 a research station near Bass Lake, California was established and staffed with a team of field biologists and research assistants. Field work was initiated in late October 2007. All permits were in place when we initiated live trapping on December 21, 2007 and captured the first two fishers on December 23, 2007. In late December 2007 we initiated the SNAMP Aviation program in cooperation with Forest Service Supervisory Pilot John Litton (R5 Regional Aviation Group, Lancaster, CA). One of the principal goals of the study was to maintain a minimum of 20 actively monitored fishers, a milestone that was achieved on July 23, 2008.



Site Description and Study Area

The SNAMP Fisher Project study area is at the northern end of the southern Sierra Nevada fisher population in California, encompassing the area bounded by the Merced River in the north and the San Joaquin River in the south (Fig. 3). Administratively, the focal study area for the study is the Bass Lake Ranger District in the Sierra National Forest, but early in the study a radio-collared fisher dispersed north into Yosemite National Park, which effectively expanded the study to encompass a part of Yosemite National Park (Fig. 3). Permission was granted by Yosemite National Park to monitor SNAMP radio-collared fishers for survival by fixed-wing aircraft overflights, but the study agreement did not extend to any significant ground-based activities by project staff.

The overall study area was the non-wilderness region of the Bass Lake Ranger District in the Sierra National Forest, near Oakhurst, California, and covered approximately 1300 km². This area encompasses a mix of public and private land and is topographically complex with elevations ranging from 758 m to 2652 m. Field work was carried out between 1,000 m and 2,400 m in elevation, corresponding to fisher occurrence in the region. Primary tree species in approximate order of abundance for conifers and then hardwoods in the overall study area are incense cedar (*Calocedrus decurrens*), white fir (*Abies concolor*), ponderosa pine (*Pinus ponderosa*), sugar pine (*Pinus lambertiana*), California black oak (*Quercus kelloggii*), canyon live oak (*Quercus chrysolepis*), mountain dogwood (*Cornus nuttallii*), and white alder (*Alnus rhombifolia*). Common shrubs and tree-like shrubs include whiteleaf manzanita (*Arctostaphylos viscida*), greenleaf manzanita (*Arctostaphylos patula*), mountain misery (*Chamaebatia foliolosa*), bush chinquapin (*Chrysolepis sempervirens*), mountain whitethorn (*Ceanothus cordulatus*), and snowberry (*Symphoricarpos mollis*).

Key Watersheds

We already noted that the distribution and overall abundance of fishers in the Sierra Nevada, California declined by 30-50% in association with fur trapping, and timber harvest that removed large expanses of mature and old growth forest habitats in the early to mid-1900s. One of our overarching organizing hypotheses was that fuel reduction management (SPLATs; commercial thinning, understory brush and small tree removal by hand thinning and machine mastication) might exacerbate this contraction by preventing maturation of second-growth forests in the southern Sierra Nevada to where they are less capable of supporting a long term viable fisher population. Therefore within the overall study area, we initiated more intensive monitoring within the Southern SNAMP study area: four “Key

Watersheds” that encompassed three Forest Service projects expected to occur in the study period near the communities of Fish Camp (Fish Camp Project), Sugar Pine (Sugar Pine Project), and Cedar Valley (Cedar Valley Project). The four Key Watersheds are the Sugar Pine, Nelder Creek, Rainier Creek and White Chief Branch watersheds (Fig. 4). A 1 x 1 km grid (1-km²) was overlaid for the Key Watersheds, and used to organize the sampling effort; National Forest land within each 1 km² cell were surveyed annually for fisher occupancy by automatic camera traps (O’Connell et al. 2011).

Elevation within the Key Watersheds increases from 900-1100 m in the south/southwest to >2200 m

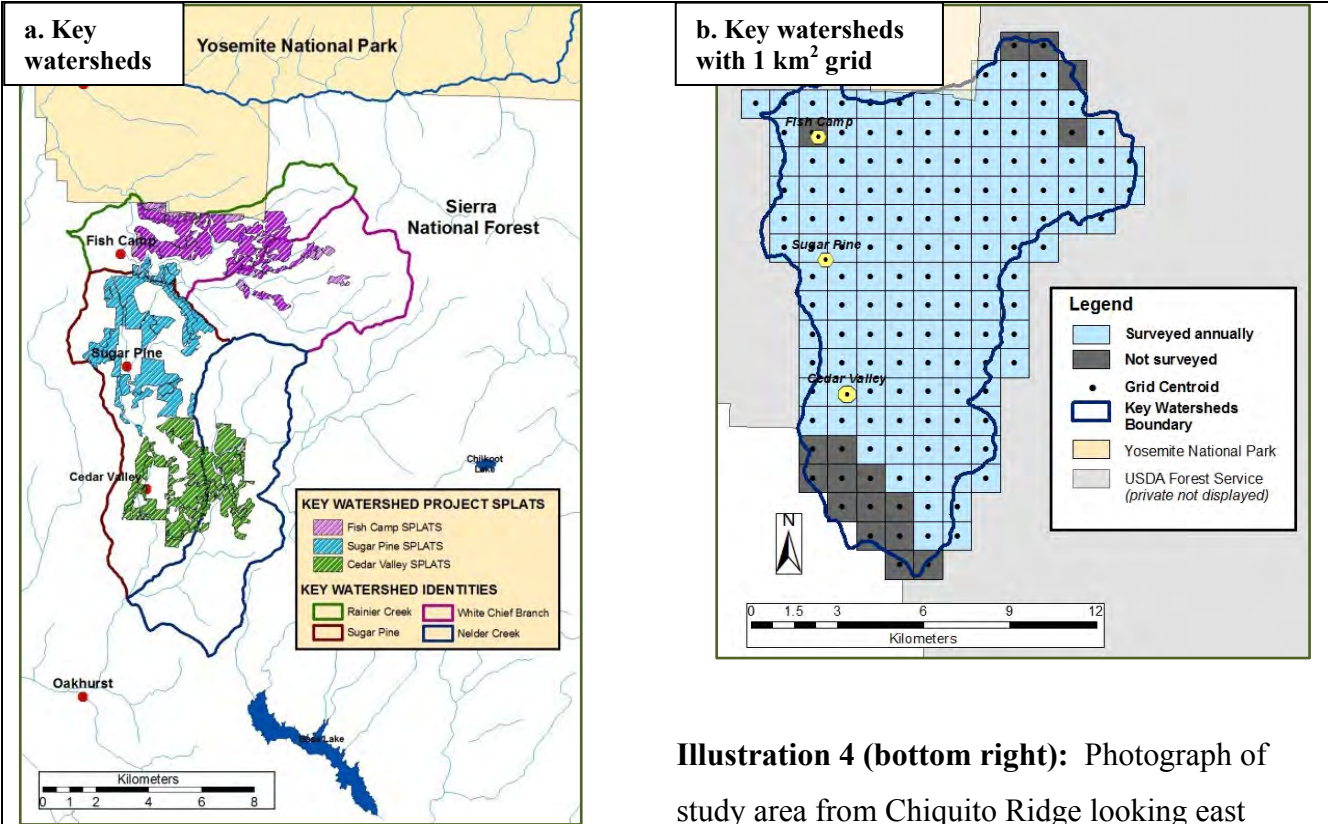


Illustration 4 (bottom right): Photograph of study area from Chiquito Ridge looking east

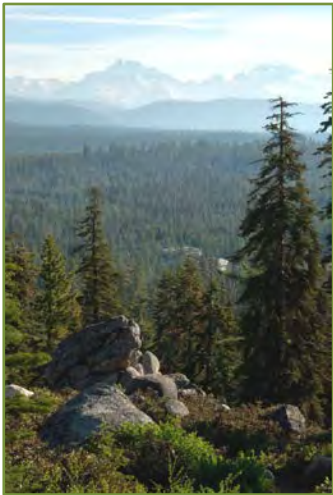


Figure 4. (a) Map views of Key Watersheds focal study area, including the SPLAT polygons originally produced for the Sugar Pine, Cedar Valley, and Fish Camp forest management projects. (b) Key Watersheds overlain with 1km² grid used to organize science effort for camera trap surveys. Yellow circles indicate the three communities located within the watersheds. NOTE: camera traps were placed at or very near the grid center points as plotted.

in the northeast portion of the study area. This elevation gradient corresponds with a mix of hardwoods (California bay, *Umbellularia californica*, Canyon live oak, black oak, and several conifer species at lower elevations (ponderosa pine, incense cedar; California Wildlife Habitat Relationship system MHW, PPN, and MHC habitat types), a mix of multiple conifers (Jeffrey pine, *Pinus jeffreyi*, white fir, incense cedar), and hardwoods (black oak, white alder, *Alnus* (ponderosa *rhombifolia*, mountain dogwood, *Cornus nuttallii*) between 1300 m and 1850 m (CWHR Habitat types SMC, MHC, PPN), and grading into red fir (*Abies magnifica*) and lodgepole pine (*P. contorta*) (CWHR Type RFR) above 1900 m. Giant sequoia (*Sequoiadendron giganteum*) are present, but primarily restricted to the Nelder Grove Historic Area within the Nelder Creek watershed. Permanent streams in the Key Watersheds are important for fishers and other wildlife and include Big Creek and Rainier Creek in the Rainier Creek watershed, Lewis Creek in the Sugar Pine watershed, and California Creek and Nelder Creek in the Nelder Creek watershed.

Methods

Vital Rates and Basic Population Parameters

Introduction and Background

Information on population size and demographic parameters is fundamental for managing wildlife populations, especially when declines in abundance or range size have occurred and the species is the focus of conservation management. The fisher is one such species, and it is at the center of intense conservation efforts as a candidate for listing under the USA Federal, Oregon, and California endangered species acts (USFWS 2004).

The southern Sierra Nevada fisher population is small (<500 individuals), appears to be stable over about the last 15 years (Zielinski et al. 2013a), but may have expanded from an even smaller population during the late 20th century (Tucker et al. 2014). Spencer et al. (2011) used a spatially explicit population model to estimate the potential fisher carrying capacity south of the Merced River and concluded there are probably <300 adult fishers. Fisher density estimates from the Sierra National Forest based on mark-recapture camera station data (Jordan 2007) or scat-detector dog data (Thompson et al. 2012), suggest that the total population (including adults, subadults, and some juveniles) could number up to ~450 total fishers in overall southern Sierra Nevada region of California.

The ecology and habitat use of fishers in the southern Sierra Nevada has been the focus of research since the mid-1990s (Jordan 2007, Mazzoni 2002, Truex et al. 1998). Insight from prior

research in this region suggests population densities may vary from 0.07 to 0.28 fishers/km² (Jordan et al. 2011, Thompson et al. 2012). Information on current density is needed for other areas of the southern Sierra Nevada as well, because as the area of suitable habitat available to fishers in the southern Sierra Nevada is refined by improved modeling (Spencer et al. 2014), density values can be used to develop more accurate estimates of fisher abundance for conservation planning. Although regional occupancy trends (Zielinski et al. 2013a) suggest that the southern Sierra Nevada fisher population is relatively stable, records from elsewhere indicate fisher densities can change rapidly. On the Hoopa Fisher Project study area in northern California, density was estimated at 0.52 fishers/km² in 1998, but fell to 0.14 fishers/km² in 2005 (Higley et al. 2013).

Resource agencies are currently developing strategies to aid in maintaining viable populations of fishers in the southern Sierra Nevada. Data from SNAMP Fisher have aided this process by providing current data on population size and demographic parameters for the species in an area at the northern margin of their current range in the southern Sierra Nevada. Using data on detections of fishers from camera surveys and live trapping to estimate population size and density and estimates of reproductive rates and fecundity from close monitoring of denning behavior, we integrated data on survival and demographic rates into a matrix model to estimate the population growth rate.

Field Methods

Live Trapping.--Although noninvasive methods can be used to generate important data on wildlife populations (Long 2008), estimating vital rates (survival, reproduction, dispersal) almost always requires trapping to radiocollar and then closely monitor the study animal. We followed standard live-trapping procedures previously developed for fishers in California (Jordan 2007, Matthews et al. 2013a), with only a few minor changes. Individual fishers were live-captured in steel mesh traps (Tomahawk Live Trap Company, Tomahawk, WI) modified to include a plywood cubby box to provide the animals with a secure refuge where they were less likely to injure themselves (Wilbert 1992). Trapping to mark animals with radiocollars was focused during the fall and winter seasons between December 2007 and March 2012. Also, with the exception of the first year of the study when we needed to capture fishers to initiate the study, we did not trap during the spring denning period (late March to mid-June) to minimize disturbance to reproduction. Live traps were baited with venison, and checked daily by late morning. Captured animals were restrained in a handling cone, and sedated using a mixture of Ketamine hydrochloride and Diazepam (1 mg Diazepam/200 mg Ketamine) injected intramuscularly. Sedated fishers were weighed, classified by age and gender based on

examination of teeth and genitalia, and measured for standard morphological features. Small samples of ear tissue were collected for microsatellite DNA analysis using a sterile dermal biopsy punch. Several strands of hair were removed from the nape and rump region, also for DNA analysis. Hair samples were stored in a dry paper envelope, whereas tissue samples were stored in 95% ethanol until analysis at the USDA Forest Service Wildlife Genetics Lab (Rocky Mountain Research Station, Missoula MT). Teats on females were measured for base diameter and height using digital calipers (± 1 mm), and those data were used to identify females that weaned at least 1 kit when they had not been monitored during the denning period (Matthews et al. 2013b). Each animal was permanently identified by subcutaneous insertion of passive integrated transponder (PIT) tags (Biomark, Boise, ID), and fitted with a radio collar (Holohil Systems Model MI-2M, Ontario, Canada) modified by attaching small bands (0.5-1.0 cm) of infrared reflective tape (3M[®] Scotchlite[™]) along the lengths of the antennas. Custom breakaway devices were inserted into radiocollars fitted to juvenile fishers to reduce the risk of injury or strangulation between recaptures (Thompson et al. 2012). Bands of infrared reflective tape and breakaways were modifications, not used in previous studies. After handling, we returned animals to the cubby box and released them at the point of capture after recovery from anesthesia. Capture and handling procedures followed American Society of Mammalogist guidelines (Sikes and Gannon 2011), and were approved by the Institutional Animal Care and Use Committee of the University of California, Berkeley (protocol R139).

Live-trapping is labor intensive, and the effort was designed to gain advantage from detections of non-collared fishers at camera traps. Live traps were most frequently placed in the same area of camera trap stations after cameras had been removed (to prevent interference with camera surveys). Data from camera detections were used to design linear traplines of 5-10 traps bracketing positive detection stations. Distance of separation between traps was typically ≥ 500 m, and traplines were usually successful at capturing targeted animals within five nights of trapping. Live-trap success was further enhanced in later years of the study by placing traps in locations where fishers had been captured in the past. Trap success was also enhanced by cleaning and sanitizing traps after captures. In winter, snow falling from tree branches can ice up the treadle mechanism inside live traps. We used lightweight, rectangular canvas tarps (1.12 width, 1.36 length) to protect the inside of the live traps from falling snow, and debris used to camouflage the traps. Traplines were generally removed the day after targeted fishers were captured, and always after 10 nights of trapping when no fishers were captured.



Illustration 5: Camouflaged live trap near the base of a white fir tree, and a radiocollared fisher being released after processing

Aerial Telemetry and Radiotelemetry Monitoring.--Tracking radiocollared animals from an aircraft is an alternative to locating them from the ground by homing or triangulation (Thompson et al. 2012). Researchers have been using fixed-wing aircraft to locate wildlife since the early 1970s (Mech 1974). The unique ability of observers in aircraft to rapidly search and locate radiocollared animals over large and inaccessible areas while allowing for nearly line of sight reception between transmitter and receiver makes aerial radio telemetry an attractive research technique in general (Gilmer et al. 1981), and specifically for studying fishers, which often occur in remote mountainous areas where access can be difficult (Weir and Corbould 2008). Partly for these reasons, we used fixed-wing airplanes to monitor and relocate radiocollared fishers for the entirety of the SNAMP Fisher Project. Beginning in December 2007, we worked with USDA Forest Service Supervisory Pilot John Litton to develop an aviation program in support of SNAMP Fisher, which was fully established in August 2008 when a full time pilot was hired and the first of two dedicated aircraft were based at the Mariposa-Yosemite Airport in Mariposa, CA.

The two USDA Forest Service-owned aircraft acquired for supporting the project were a Cessna (Cessna Aircraft Co., Wichita, KS) and a Piper PA-18 Super cub (Piper Aircraft Inc., Vero Beach FL). Two aircraft were considered necessary to maintain continuous monitoring of radiocollared fishers when routine maintenance or engine repair was necessary (John Litton, personal communication).



Illustration 6. Forest Service-owned Piper Supercub (left) and Cessna 185 (right) on the tarmac at the Mariposa Airport, California.

The optimal search procedure used when locating animals from light aircraft varies depending on the number of animals tracked, and the antenna configuration supported and approved for the airplane being used (Gilmer et al. 1981). Additional details are provided elsewhere (Thompson et al. 2012), but we used two, 2-element H antennas (Telonics Inc., Mesa, AZ) mounted in a sideways configuration on each wing strut, and a single 3-element Yagi antenna (Advanced Telemetry Systems, Isanti, MN) mounted forward-facing on the right wing strut. This antenna configuration was effective in allowing the pilot and biologist to search for radiocollared fishers using the Yagi antenna (detection range 5-20 km), and then switching to the H-antennas to localize to a relatively precise location above the animal using a circling technique (Seddon and Maloney 2004).

Fixed-wing flights (aerial telemetry missions) to locate radiocollared fishers in the study area were scheduled in advance for 4-6 missions/week, depending on weather conditions considered safe for departure and return to the Mariposa-Yosemite Airport. Flights typically occurred in the morning hours, and lasted 2-3 hours. Afternoon telemetry flights were relatively infrequent, and the large majority of aerial radiotelemetry locations were acquired in the AM period of the day. As part of each aerial-telemetry mission, we systematically searched for all active radiocollars deployed on fishers in the study area. Biologists in the airplane recorded (1) active/inactive status for each fisher, (2) time of location, (3) an estimated UTM location for each fisher (typically logged into a handheld GPS unit; Garmin 60 CSx, Olathe, KS), (4) a qualitative ranking for each location (poor, fair, good, excellent), and (5) a record of any radiocollared fishers that were not located. Additional descriptive details were often recorded related to the nature of weather conditions influencing the aircraft at the time of the location (turbulence, “bumpy”, clouds occluding visibility to the ground, etc.), or if the animal had moved an unusual distance or to an atypical area. At the end of each aerial telemetry mission, the



Illustration 7: Forest Service Cessna 185 airplane, and antenna configuration on the right side wing strut.

biologists summarized details on departure and return times and weather and flight conditions during the flight.

Aerial radiotelemetry can be efficient for locating animals that range over large areas in difficult terrain (Gilmer et al. 1981), but the accuracy, or precision of aerial telemetry locations is generally less than for ground-based radiotelemetry (e.g. triangulation; Koen 2005, Gantz et al. 2006). Location error from fixed-wing airplanes varies with flight speed, elevation above ground level, pilot and biologist experience, and signal reflection in rugged topography (Thompson et al. 2012). We assessed error for aerial radiotelemetry locations on the SNAMP Fisher project by calculating the Euclidean distance between GPS locations logged by biologists in the airplane and positions of test collars placed at known locations on the ground. Test collar locations were generally radiocollars that were placed in locations unknown to the biologist in the airplane; biologists were required to regularly estimate positions for test collars during aerial telemetry missions. Other aerial radiotelemetry locations used to quantify accuracy included dropped radiocollars, carcass locations, fishers in live traps, and female fishers in a cavity in a den tree whose locations were also unknown to the biologist in the airplane.

Fisher Reproduction

Background.--Den sites, where female fishers bear and raise their kits, are probably the most limiting habitat element for fisher populations in California. Females typically use more than one den during the denning season (late March to mid-June). Natal dens are where adult female fishers give birth and initially care for young, and they may then move kits to one or more maternal dens from early April to June until they are weaned (Powell et al. 2003, Matthews et al. 2013a). Reproductive dens, both natal and maternal, are nearly always cavities in large trees, live or dead, and are found in forest stands with dense canopy cover and complex multi-layered structure (Zhao et al. 2012). Suitable denning sites are likely a subset of suitable resting sites because the requirements are more stringent: (1) den cavities must be large enough to shelter both mother and kits for weeks rather than days; (2) the female needs to provision her young while they are restricted to the den, so dens must be located close to high-value foraging areas; and (3) denning begins in late March-early April, when temperatures are colder and slope position may be more critical in assisting with kit thermoregulation.

Identifying den trees and evidence of reproduction.—Female fishers exhibiting behavior consistent with denning were identified during late March-mid April and then monitored. Denning

behavior was characterized by an abrupt change from a pattern of successive aerial radiotelemetry locations being dispersed within a female's home range, to a pattern where locations were spatially clustered (3-5 locations within 500 m over a 7-day period; Zhao et al. 2012). When clustering of locations occurred, a biologist navigated to the area with a handheld Global Positioning System device to investigate. Standard ground-based radiotelemetry techniques with a handheld receiver (model R1000; Communication Specialists, Inc., Orange, California) and an H-type antenna were then used to home towards telemetry signals of radiocollared females. Once a collared female was isolated in an area, the biologist circled the fisher until the individual tree or snag was identified (Matthews et al. 2013a). When female fishers were localized to a possible den structure, 2-4 automatic "den cameras" that had been cleaned and de-scented were attached to nearby trees and focused on the bole of the den structure (scent and bait lures were not applied around den trees to avoid attracting other predators). We returned to these structures the day after initial placement of den cameras, and then every 3-5 days to confirm use for denning based on regular occupancy and images indicating up and down movements on the tree or snag. Trees and snags used ≥ 3 times in succession and with camera-based evidence of up-down movements were considered denning structures (Zhao et al. 2012). We defined "denning opportunities" as the total number of individual, breeding-age female fishers (≥ 24 months) either directly monitored in mid-March to June (Matthews et al. 2013a), or measured for teat size during July to January to assess weaning status (Matthews et al. 2013b). We considered kits weaned when denning behavior continued until 31 May or later (Matthews et al. 2013a), unless the female was known to have died before June 30.

Activities of known-denning female fishers were chronicled for the duration of each denning season by continuous monitoring with cameras and ground checks of den trees. Female fishers typically transfer kits from natal den trees in which they were born to 1-6 other maternal dens during April to June (Matthews et al. 2013a). Each time we had evidence that a denning fisher moved kits to a new maternal den (images of females transporting kits away from den trees, cessation of occupancy over multiple checks), we searched for the female using ground telemetry and repositioned cameras around the next den structure (Zhao et al. 2012). Den cameras were removed in mid-June when females ceased localizing to den structures.

Information on litter size was determined from images from den cameras, or, less frequently, by climbing den trees and using a video camera (Peep-A-Roo Video Probe System, Sandpiper Technologies, Manteca, CA) to count kits inside den cavities (Matthews et al. 2013a). We minimized disturbance to denning females by (1) restricting visits to den structures to service cameras to once every 3-5 days, (2) using deployments of multiple den cameras for obtaining the majority of kit counts, and (3) by not approaching den trees for climbing until ground-based telemetry indicated the female was well away from the den structure (Zhao et al. 2012).



Illustration 8: Female fisher F46 moving a 1-month old fisher kit from her natal den tree.

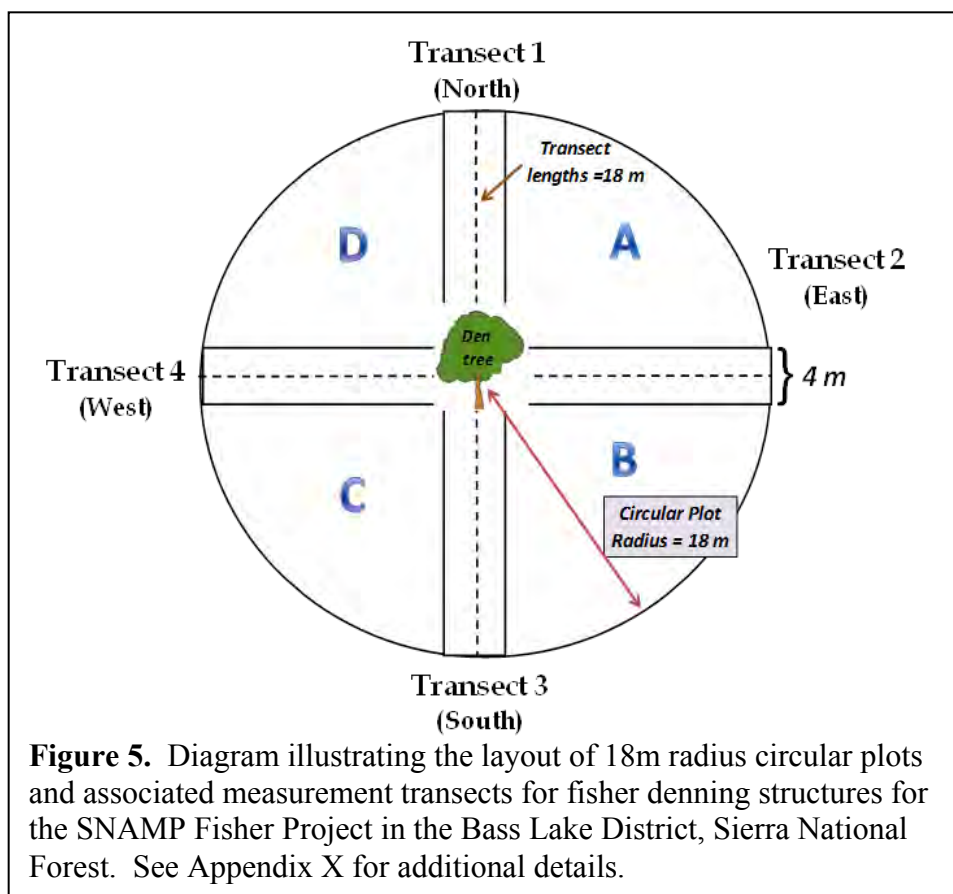
Maximum reproductive rate was estimated as the sum of the number of adult-age female fishers in the population that localized to den trees during the den season plus the number of adult females with enlarged teats that were not monitored but captured and measured before January, divided by the number of adult-age female fishers in the population during mid-March to late January. Weaning rate was estimated as the number of adult-age females known to have survived and localized to den trees through May plus those with enlarged teats that were captured after the den season, divided by the number of adult-age female fishers in the population during mid-March to late January. We note that measurements of teat size were shown to correctly identify over 90% of current year reproducing females that weaned at least 1 kit, and all but 3 adult females for which teat measurements were used to determine reproductive status were part of the Matthews et al. (2013b) dataset. Annual estimates of fecundity were calculated as the total number of weaned kits/number of females with known kit counts.



Illustration 9: Female fisher ascending a den tree. Note that the image from one camera also shows two other cameras positioned on adjacent trees that were monitoring the same den tree.

Habitat Characteristics within Fisher Denning Areas.--Denning structures are considered a limiting habitat element for fishers in California and elsewhere (Weir et al. 2012), but site characteristics immediately surrounding the denning structures are also important (Zhao et al. 2012). Current forest vegetation management to reduce hazardous fuel levels, improve the vigor of selected trees (pines and oaks), increase spatial heterogeneity, and provide forest products for society may impinge on denning habitats in a variety of ways (Naney et al. 2012, Powell and Zielinski 1994). These management actions can negatively affect fisher habitat (Weir and Corbould 2008), at least in the short term (Thompson et al. 2011), while others may have little impact on fisher habitat suitability (Spencer et al. 2014). Without detailed “local scale” information on habitat characteristics directly associated with fisher denning structures, it will not be possible to adequately manage Sierra Nevada forests in ways that will maintain viable fisher populations.

Habitat characteristics for denning structures.—We developed a protocol for determining the combination of biotic and abiotic characteristics female fishers are likely selecting/using for denning habitats. The protocol was designed to collect similar types of data as those being recorded by the Forest Health Team on the Core Plots in the Sugar



Pine area, while also capturing the same types of data being recorded by the USDA Forest Service PSW Kings River Fisher Project at den trees used by fishers in the High Sierra District, Sierra National Forest. Full details on how different habitat data were assessed are provided as an Appendix. Briefly, we used an 18m radius circular plot centered on the denning structure (Fig. 5) to organize collection of data on (1) canopy cover, (2) litter, duff, and coarse woody debris (associated abundance of fuels), (3) cover and height of herbaceous plants and understory woody shrubs (concealment cover), (4) slope and aspect, and (5) size, number, and height of trees and snags (3 size classes, 4 crown classes). Data on habitat characteristics for den trees were typically collected during late spring or summer, and always when the den trees were no longer in use for denning.

Fisher Survival

Background.--Understanding survival and the details of cause-specific mortality is fundamental for insight into the population biology of any species, and crucial for understanding the limits to population growth and recovery for rare or endangered species of wildlife. Historical loss and fragmentation of important habitats, combined with overexploitation by hunting and trapping are the

most common drivers of endangerment of wildlife (Lande 1993). Although changes in management may sometimes succeed in reversing problems associated with loss of critical habitats, emergence of new threats that impinge on survival or reproduction can counteract improvements that might otherwise reverse population declines. Emerging threats to survival and population persistence may be obvious such as exposure to novel pathogens and increased occurrence of road-kill deaths (Gaskill 2013, Litvaitis and Tash 2008), or less discernible and linked to changes in community structure or composition that produces an imbalance in predator-prey relations (Roemer et al. 2001).

Factors that contribute to limited growth and expansion of fisher populations in the southern Sierra Nevada are likely linked to a combination of resource and population phenomena. Fishers may be challenged by limited access to suitable resting and denning habitats (Purcell et al. 2009) or insufficient numbers of prey (Zielinski and Duncan 2004), whereas survival may be reduced by high rates of predation (Wengert 2013), wildlife-vehicle collisions (Chow 2009), and exposure to anticoagulant rodenticides (Gabriel et al. 2012a, Thompson et al. 2013). Although habitat requirements of fishers and their responses to forest management are increasingly well known (Aubry et al. 2013, Garner 2013, Zielinski et al. 2013), it has only recently been documented that high numbers of otherwise healthy fishers were succumbing to attacks by other forest carnivores (Wengert et al. 2014) and that illegal use of anticoagulant rodenticides and other toxicants associated with illegal marijuana grow sites on national forests and parks in the southern Sierra Nevada was contributing to both direct mortality and reduced survival of fishers in this region (Gabriel et al. 2012a, 2013, Thompson et al. 2013). Because of heightened concern over the stability of the small population of fishers in the southern Sierra Nevada, our primary objective was to evaluate factors contributing to variation in survival among fishers in the region. Young mammals (particularly males) often experience higher mortality early in life associated with dispersal and establishing independent home ranges (Chepko-Sade and Halpin 1987), and general naiveté with predators and other environmental risks (Farias et al. 2005, Murdoch et al. 2010). Fishers typically disperse before they reach maturity at ≈ 24 months (Arthur et al. 1993), when lower survival related to this life history event might be evident. Fishers may experience lower survival during fall and winter due to the combined effects of higher energetic costs associated with deep snow cover (Powell 1979) and prey limitation when several species of rodent prey (Zielinski and Duncan 2004) enter into torpor. Therefore, we hypothesized that (1) survival would be lower for juvenile and yearling fishers compared to adults, (2) males would experience lower survival than females related to higher rates of movement and potential longer dispersal distances, and (3) survival would be lower during fall and winter than in spring and summer.

Determination of survival rates.--We monitored the status (alive, dead, or missing) of radio-collared fishers from time of first capture until death, censorship (due to dropped or failed collars on animals that were not quickly recaptured), or the end of the study. Breakaway devices in the radio-collars occasionally resulted in premature detachment, requiring efforts to re-collar animals that were short-term missing (1-2 months). Because of the relatively common incidences when animals were missing for less than one month, we evaluated survival on monthly intervals rather than weekly or bi-weekly. Overall patterns of survival were determined using the Kaplan-Meier (KM) staggered entry method (Koen et al. 2007, Pollock et al. 1989, Price et al. 2010). KM models were used to produce estimates for annual survival and combined year survival (data pooled by month across all years) by population year. The population year was defined as April 1 to March 31 based on the timing of reproduction for female fishers in California with most offspring produced during March 21 to early April (Matthews et al. 2013a). Annual survival can be moderately to highly variable and may result in a negative population growth trajectory that may not be appropriate for a long-lived species with a generation time of two or more years. We therefore combined monthly data on survival status for individual fishers for five 2-year periods (population years 2 and 3, population years 3 and 4, population years 4 and 5, population years 5 and 6, and population years 6 and 7). Live-trapping to capture young-of-the year juveniles was focused during mid-October to February (a few juvenile fishers were occasionally captured before October or in early March). Kaplan-Meier models to estimate “annual” survival for juveniles were typically initiated in December, thereby producing survival estimates for juveniles for a 3-4 month period from December or January to March. When data for juveniles were pooled across population years, however, the dataset allowed for evaluating juvenile survival for the all years combined model for the 6 month period from October to March. Z-tests were used to compare estimates for combined year survival for all possible age and sex combinations. Significance levels (α) for multiple comparisons were adjusted for Type I error rates using the Bonferroni procedure (McCann et al. 2010).

Causes of Mortality

Background— Understanding the details of cause-specific mortality provides insight necessary to understand overall survival in relation to factors that are most likely to limit population growth and recovery for rare or endangered species of wildlife. Factors that contribute to limited growth and expansion of fisher populations in the southern Sierra Nevada, California are likely linked to a combination of resource and population-level phenomena. In addition to the challenges described

previously, it has been hypothesized that the recent change to more open canopy forest conditions with an understory of small trees and more shrubs is contributing to higher rates of predation by bobcats (*Lynx rufous*) and coyotes (*Canis latrans*) (Wengert 2013). Although the habitat requirements for fishers are generally known (Lofroth et al. 2010), details of cause-specific mortality in the southern Sierra Nevada had not been rigorously evaluated until the SNAMP and KRFP studies were initiated in 2007.

Monitoring to detect mortality.-- All radio-collars fitted to fishers on the SNAMP study were equipped with mortality or activity sensors, allowing us to detect inactive signals and investigate fisher mortalities and recover carcasses soon after death in most cases. When a mortality signal was detected, immediate attempts were made to investigate the area of the signal to recover shed radiocollars or recover the carcass when an animal was confirmed dead. Carcasses were generally recovered within 24 hrs of death, aided by the near daily aerial radiotelemetry missions.

Radiocollared fishers have been monitored effectively and relocated by ground-based radiotelemetry as part of KRFP study centered approx. 60 km southeast of the SNAMP Fisher Study Area (Garner 2013). However, most prior studies have been unable to identify causes of death for many deceased study animals because carcasses were not retrieved within 12-48 hours after death (Truex et al. 1998, Aubry and Raley 2006, Jordan 2007). Decomposition begins immediately after death, which can prevent identification of underlying disease processes directly linked to death (Gabriel 2013, Keller et al. 2012). Because of this, our primary rationale for monitoring radiocollared fishers by fixed-wing aircraft up to 6 days/week was to recover carcasses of animals as soon after death as possible. The protocol that was in place from the start of the study until approx. June 2012 was for the biologist in the airplane to use the audio system in the airplane to (1) transmit the estimated location coordinates for any radiocollared fishers detected on inactive pulse to the fisher study office, whereupon (2) a staff biologist in the vicinity would immediately investigate the location and recover the carcasses following an approved forensic protocol, (3) transport the carcass to the fisher study office where they were placed in -20⁰ C freezer for storage until (4) a necropsy could be scheduled at the UC Davis School of Veterinary Medicine.

Once a radiocollar is activated and deployed, it will typically function (emit a radio signal) for 18-24 months until the battery is expended. In 2011 the SNAMP Fisher project began using radiocollars from Advanced Telemetry Systems (ATS), but discovered that many of the electronic “mortality switches” built into the ATS radiocollars became defective within 8-10 months of being deployed on study animals. Electronic mortality switches are designed to emit pulses at twice the

normal pulse rate when the radiocollar has been stationary for at least 8 hours. When the mortality switches in the ATS brand radiocollars became defective they began to emit intermittent and then consistent false inactive signals. SNAMP Fisher was forced to modify the inactive signal protocol by first plotting locations determined for inactive signals in ArcGIS, whereupon a decision on whether to investigate the location was based on a judgment of the distance of separation between successive aerial radiotelemetry locations (investigation triggered when successive locations were <1000 m apart). The revised protocol was situation specific: the first time an ATS collar was detected emitting an inactive signal, efforts were made to investigate the location immediately. Subsequent inactive signals from that collar were examined carefully prior to on-the-ground investigation. This process may have delayed the recovery of a limited number of carcasses. Efforts were made to correct for the problem of false inactive signals by replacing defective collars with collars of a different manufacture as soon as possible.

When fisher carcasses were discovered we followed a standardized forensic protocol for collecting samples and documenting circumstances at mortality sites using photographs and diagrams of mortality sites (Wengert et al. 2013). When predation was suspected as the cause of death (e.g. obvious punctures, partial consumption), we recorded information on the characteristics of the predation event including patterns of consumption and evidence of caching or burying. Samples included swabs of visible bite wounds, clipped fur from near the bite wounds (clipped to avoid fisher DNA in root bulbs), swabs of the claws and teeth, and any non-fisher hairs left on or near the carcass (Wengert et al. 2013). Carcasses were double-bagged in plastic bags, labeled, and transported back to the field offices where they were frozen in a -20°C freezer until being shipped to University of California, Davis for necropsy.

Pathology.--We submitted all carcasses for necropsy and disease and DNA assessment to cooperating pathologists at the University of California Davis, Veterinary Medical Teaching Hospital, and California Animal Health and Food Safety Laboratory in Davis, CA. When possible, the team of pathologists determined cause of death for each fisher using all available information, including necropsy examination, disease and toxicological results, DNA forensics, evidence recovered or identified as important from the mortality site, and habitat characteristics around the carcass. During necropsy, liver samples were collected and subsequently tested for the presence of anticoagulant rodenticide residues using liquid chromatography-tandem mass spectrometry to screen for presence of anticoagulant rodenticides and high-performance liquid chromatography to quantify positive samples. When predation was determined to be the cause of death, all lesions attributed to predation were

described in detail. To distinguish between ante and post-mortem wounds (i.e. between predation and scavenging), we noted whether the lesions had associated hemorrhage and edema. In 14 cases, too few remains were present to identify hemorrhaging at wound sites, so only molecular analyses were conducted in these cases. Age-class at time of death was estimated as adult (≥ 24 months), subadult (12-23 months), and juvenile (< 12 months) based either on tooth wear or cementum annuli counts.

Molecular Analyses-- Forensic samples were processed and analyzed for predator (either felid or canid) DNA according to the methods of Wengert et al. (2013). Because multiple polymerase chain reaction (PCR) products were occasionally obtained when the products were visualized on an agarose gel, we gel-excised the appropriately sized fragment (200–300bp for felids and 400 for canids) and extracted DNA using Qiagen Qiaquick Gel Extraction kit according to the manufacturer's instructions. The PCR products were sequenced, then aligned using RidomTraceEdit (Ridom GmbH, Würzburg, Germany). Sequences were cross-referenced on GenBank using Basic Local Alignment Search Tool (BLAST) to match them to the most closely aligned sequence to identify species of predator DNA.

Population Growth Rates

Background.—All wildlife will respond to changes in habitat, food availability, and recent weather conditions by fluctuations in animal numbers. Fishers are no exception to this general pattern (Jensen et al. 2012). As a rare species, however, variation in population size or abundance of fishers is of conservation concern for maintaining a minimum viable number of animals on the landscape (Spencer et al. 2011, Reed and McCoy 2014).

Information on the growth trajectory for fishers in the SNAMP Fisher study area is unknown, but there are several competing hypotheses regarding population status in the broader region encompassing the SNAMP Fisher study area. Research conducted between 2002 and 2012 suggested no evidence for population increase (Zielinski et al. 2013), which may indicate that despite protection from fur trapping and development of policies designed to better sustain sensitive birds and mammals (North et al. 2009), fishers in the southern Sierra Nevada may not be in numeric or spatial recovery. In agreement with the “no increase” hypothesis, a body of research suggests that fishers in the Sierra Nevada experienced a range retraction of 30-50% over the last 75-100 years (Zielinski et al. 2005, 2013, Spencer et al. 2014). An alternative hypothesis is that fishers were very uncommon in our study area prior to 1990, and the current population resulted from a northward expansion from south of the Kings River (Fisher Habitat Core Area 3; Fig. 2; Tucker et al. 2014), equivalent to an approx. 30% increase in distribution in the overall southern Sierra Nevada region based on analyses of fisher habitat

by Spencer et al. (2014) (Fig. 2). This alternative view is based primarily on genetic evidence of population subdivision (Tucker et al. 2012, 2014), and potentially supported by increasing fisher detections from track plate and camera trap monitoring after the mid-1990s (Zielinski et al. 1995, 2005, 2013). Ecology and habitat use of fishers in the southern Sierra Nevada has been the focus of research since the mid-1990s, and although much insight on fisher ecology has been gained with regards to home range size, population density, and habitat use for resting and other activities (Jordan et al. 2011, Thompson et al. 2012, Purcell et al. 2009), no prior study has reported on the growth rate of any fisher population in this area.

Population growth rates and Leslie-matrix modeling.— Intensive scientific assessment as part of the SNAMP Fisher study has generated information on all key vital rates needed to evaluate the population growth rate (λ) in the area, critical for understanding whether the population has the potential to persist, or if it is in decline. We developed an age-structured matrix model to estimate a deterministic population growth rate (λ) for the SNAMP Fisher study population using observed data on denning, fecundity, and survival. We defined 2 “adult” age classes (24 months, ≥ 36 months) for developing and including estimates of fertility in the matrix model for the population. Fertilities (F_i) were calculated for adult-age female fishers as:

$$F_i = b(i)P_i \quad \text{Equation 1}$$

where fecundity, $b(i)$, was the mean number of female kits weaned per reproductive female (sex ratio at birth assumed = 0.5), and P_i was the age-specific survival rate (Gotelli 2001). Age-specific survival rates were estimated for radiocollared juvenile (6-11 months), subadult (12-23 months) and adult-age (≥ 24 months) female fishers in the study area using monthly encounter histories in Kaplan-Meier staggered entry model analyses (KM survival). Survival estimates were produced for combined data on numbers of radiocollared fishers in each age class during the six year study period (All-year), as well as for a series of five 2-year periods. Data on numbers of fishers in each age class for each pair of years were combined for KM survival estimates, starting with population years 2008-09 and 2009-10, and ending with population years 2012-13 and 2013-14.

Data from radiocollared animals from the study area indicated that female fishers commonly die by 6-8 years of age. We therefore included 8 age classes in our Leslie Matrix (\mathbf{A}) formulation, where the numbers of fishers in each age class n_1 to n_8 at time $t+1 = \mathbf{A} \times \mathbf{n}$ vector at t_0 according to equation 2:

$$\begin{bmatrix} n1(t+1) \\ n2(t+1) \\ n3(t+1) \\ n4(t+1) \\ n5(t+1) \\ n6(t+1) \\ n7(t+1) \\ n8(t+1) \end{bmatrix} = \begin{bmatrix} F1 & F2 & F3 & F4 & F5 & F6 & F7 & F8 \\ P1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & P2 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & P3 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & P4 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & P5 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & P6 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & P7 & 0 \end{bmatrix} \times \begin{bmatrix} n1(t0) \\ n2(t0) \\ n3(t0) \\ n4(t0) \\ n5(t0) \\ n6(t0) \\ n7(t0) \\ n8(t0) \end{bmatrix} \quad \text{Equation 2}$$

Estimates for fertility were from data on weaning rates, fecundity, and survival for known-age juvenile (F_1 ; 1 month old), subadult (F_2 ; 12 months), young adult (F_3 ; 24 months), and adult fishers ($F_{4,5,6,7,8} \geq 36$ months).

Numbers of fishers in age classes n_2 , n_3 , and n_4 for the \mathbf{n} vector at time t_0 were based on the number of radiocollared female fishers present at the start of population year 3 on April 1, 2009 ($n_2 = 5$, $n_3 = 6$, $n_4 = 5$, $n_5 = 5$), whereas n_1 was the number of juvenile females in the radiocollared population on Dec 31, 2009 (4 animals). We multiplied the Leslie matrix by the new vector of abundances for N_{t+1} for 20 successive years, and summed the number of individuals in each age class each year to obtain a total N , and the population growth rate (λ) for year $t+1$ was calculated as N_{t+1}/N_t . After several years a stable age distribution was achieved and λ converged to a constant value, which was the estimate of λ for the set of demographic parameters evaluated. We calculated a lower and upper range for λ based on the 95% lower and 95% upper C.I.s for age-specific survival and age-specific fertility. Finally, due to uncertainty in estimates for several demographic parameters related to methodology [small body size prevents radiotelemetry-based monitoring of survival for juveniles until 6 months of age (Facka et al. 2013); teat measures used to identify weaning for a small subset of adult females were less than 100% accurate in assigning reproductive status (Matthews et al. 2013a)], We evaluated the sensitivity of the matrix model to 20% reductions in rates of survival and fertility for each age class. Fertility is linked to age-specific survival according to Equation 1, and changes to fertility associated with reductions in survival were carried into the model when evaluating sensitivities.

Population Size and Density

Background

Information on population size and demographic parameters are fundamental for managing wildlife populations, especially when declines in abundance or range size have occurred and the species is the focus of conservation management. As previously noted, the overall southern Sierra Nevada fisher population is small (<350 adult fishers; Spencer et al. 2014), but appears to be stable over about the past decade (Zielinski et al. 2013). Research focused on the ecology and habitat use of fishers in the southern Sierra Nevada has been ongoing since the mid-1990s (Jordan 2007, Mazzoni 2002, Truex et al. 1998), but primarily for the area encompassed by the Kings River Fisher Project in the High Sierra District, Sierra National Forest (Fisher Habitat Core area 4; Fig. 2). For that area, Jordan et al. (2011) used a capture-mark-resight/recapture design (CMR) to estimate a density of 0.063-0.109 fishers/km² in 2002-2004, whereas Thompson et al. (2012) used scat detector dogs and genetic detections in a spatially explicit CMR framework to estimate a fisher density of 0.065-0.28 fishers/km². Information on density is needed for other areas of the southern Sierra Nevada as well, because as the area of suitable habitat available to fishers in the southern Sierra Nevada is refined by improved modeling (Fig. 2; Spencer et al. 2014), density values can be used to develop more accurate estimates of fisher abundance for conservation planning. Information from the SNAMP Fisher study has aided the process by providing current data on population size and density for the Fisher Core Habitat area 5 (Table 2). Here, we estimated fisher population size and density in the middle four years of the study using mark-resight techniques (McClintock and White 2009) from camera trap surveys and live trapping.

General Methods for Camera Surveys and Camera Traps

Motion sensing camera traps (Silent Image Professional, Rapidfire PC85; RECONYX Inc., Holmen, WI) were systematically deployed near the center of 1-km² grids in the study area beginning at the start of each of five “fall-winter” camera survey years (October 15-October 14 the following year). Placement of camera traps within 1-km² grid cells was determined based on the presence of habitat elements important for fishers (e.g., presence of mature or large diameter trees, moderate to steep slopes, canopy cover $\geq 60\%$, proximity to permanent streams; Purcell et al. 2009, Zielinski et al. 2004). Cameras were focused on bait trees upon which we attached baits and applied scent lures as attractants. Baits were small pieces of venison (140-250 grams) in a dark colored sock (reduced consumption by insects), and 8-10 hard-shell pecans strung onto a wire (initial purpose was to index

squirrel abundance, but were also consumed by fishers). Scent lures were Hawbaker's Fisher Scent Lure (Fort Loudon, PA) dabbed on the bait sock, Caven's "Gusto" scent lure (Minnesota Trapline Products, Pennock, MN) applied near the base of the bait trees and on several nearby trees, and ~4 grams of peanut butter smeared on the nut ring (Popescu et al. 2014). Camera survey stations were typically visited (checked) every 8-10 days over 32-40 days to refresh scent lures and bait, and to maintain camera units, but the protocol varied depending on whether the survey station was within the Key Watershed part of the study area, or outside that area. Survey cameras within the Key Watersheds were left in place a minimum of 32 days (four 8-10 day checks), whereas cameras outside this area were deployed for a minimum two 8-10 day checks but removed on check two or three if fishers had been detected. We removed survey cameras after four checks unless the unit had been disturbed (most frequently by black bears, *Ursus americanus*) to where the bait tree was out of view or if the unit had been inoperative due to expended batteries or malfunction for more than five days during a check period. In those cases the survey was extended by one or more 8-10 day periods to assure adequate survey effort (Slauson et al. 2009).

Camera surveys, live trapping, and radiocollar data.—Camera surveys were done during all months of each camera survey year, but the time frame of interest for this part of the study was October 15 to March 15, related to assumptions for mark-resight analyses of a closed population scenario. There are 145 1-km² grid cells within or overlapping the Key Watersheds boundary; 128 of them are at least 50% USDA Forest Service ownership, and were surveyed in all four survey years. A total 319 1-km² grid cells external to the Key Watersheds and within the study area boundary (Fig. 4) were surveyed in at least one fall-winter camera survey year, and 221 (69%) of those were surveyed in two or more years.

Full details on live-trapping to radiocollar and mark individual fishers was provided above. However, for the purposes of mark-resight analyses, data on captures and recaptures of known fishers were included in the mark-resight dataset. Also, fishers sometimes shed their radiocollars, or collars separated at the breakaways as designed. Dropped radiocollars were retrieved from the field, and the locations of shed radiocollars were included in the resight dataset.

Monitoring and home ranges.—Radio collared fishers were monitored for activity status and relocated 4-6 days/week throughout the year by fixed-wing airplane. Standard methods were used to obtain locations from the airplane as previously detailed, and mean error associated with aerial telemetry locations was estimated at 339 m.

Location records were used to develop home range models for individual fishers using the fixed kernel density method in Home Range Tools for ArcGIS 9.3 (Rodgers et al. 2007). Ninety-five percent fixed kernel home ranges were produced for individual animals for four fall-winter (October 15 to March 16) periods from 2008 to 2012 when ≥ 25 locations were available for an individual fisher. Home range area estimates from fixed kernel utilization distributions are sensitive to the choice of bandwidth as a smoothing parameter (Gitzen and Millspaugh 2003). We used the Ad Hoc method to identify the most appropriate reference bandwidth for smoothing fisher home ranges and minimizing formation of multiple polygons (Kie et al. 2010, Kie 2013).

Resighting and Mark-resight Analyses.—Radiocollared fishers detected by cameras were identified by the antennal pattern of bands of infrared reflective tape (Popescu et al. 2014). Detections of known fishers were counted once per camera station per calendar day. We were not able to unambiguously identify all radiocollared fishers detected at camera traps due to occasional loss of bands and breakage of antennas; these detections were scored as collared unknown. Non-collared animals were counted as unmarked seen.

We considered the population as approximating closure during Oct 15 to Mar 16 because (1) most mortalities in the study site occurred between mid-March and September, (2) natal dispersal by juvenile-age fishers in the population was focused during March to August, and (3) fisher reproduction in California begins the third week in March (Matthews et al. 2013a, this study). Data on individual fisher resightings at camera stations or live traps were scored based on presence within 1-km² grid cells. Individual animal detection histories were developed identifying whether fishers were available for resighting based on the presence of survey cameras or live traps within the boundaries of their 95% fixed kernel fall-winter home ranges. Data were also compiled on the numbers of survey cameras and live traps deployed, survey camera nights, and live trap nights for the fall-winter resight period.

The resighting data were analyzed using robust design mark-resight, log-normal Poisson models (McClintock and White 2009). The mark-resight robust design is analogous to the mark-recapture robust design of Kendall et al. (1995) and Kendall et al. (1997), in that it allows for individual covariates in modeling resighting



Illustration 10. Example of infrared tape on collar antenna used to identify individual fishers for mark-resight analyses.

probabilities, and an open population between primary sampling occasions. Along with data on marked animals, mark-resight models incorporate sightings of unmarked animals, while the robust design allows for estimating abundance (N), apparent survival between primary intervals (ϕ), mean (α) and overall resighting probabilities (λ), random individual heterogeneity (σ^2), and transition probabilities between observable and unobservable states (γ'' and γ') (McClintock and White 2009). The parameter of interest, abundance (N), is a derived parameter, as Poisson log-normal models estimate the number of unmarked individuals in the population, U (McClintock and White 2009).

The Poisson log-normal mark-resight model takes the following form:

$$[\alpha(.) \sigma(.) U(.) \phi(.) \gamma''(.) \gamma'(.)]$$

in which ϕ and γ'' (and γ') were modeled using a *sin* link, while α , σ , and U were modeled using a *log* link.

The model assumptions are: (1) geographic closure, (2) population closure within primary intervals, (3) no loss of marks, (4) no error in identifying marked and unmarked animals, (5) equal resighting probability for both marked and unmarked individuals, and (6) sampling is with replacement within secondary periods (McClintock and White 2009). We used camera survey years as the primary sampling intervals and the number of resights and live trap recaptures for marked fishers within each primary occasion as the resighting histories (Appendix I). Along with capture histories, robust-design Poisson log-normal models require three other quantities: (1) marked superpopulation, the number of marked individuals known to be in the population during primary interval j , (2) number of times marked individuals were sighted, but individual marks could not be identified, and (3) total unmarked individual sightings during primary interval j .

Because camera and live trapping was unbalanced across the study region among years, we added a grouping variable for subregion, with three subregions defined by the spatial segregation of camera trap efforts (Fig. 6) Each fisher was assigned to a particular mark-resight subregion based on the position of its 60% fall-winter home range isopleth. In addition, we included *area* and *time* (primary sampling interval) covariates, *cams* (camera effort for each subregion during each primary sampling interval in hours), and *live* (number of days live trapping was conducted) to account for variation in resighting probabilities, individual covariates *weight* and *sex* to account for individual and gender-based resighting probabilities. In the model parameterization, state transition probabilities remained constant [$\gamma'(.)$ and $\gamma''(.)$], apparent survival was modeled as function of region [$\phi(area)$], and different combinations (additive and interactions) of the individual and time and region-based

covariates were allowed.

We considered 19 candidate models and used AICc [Akaike Information Criterion adjusted for small sample size; (Burnham and Anderson 2002)] to rank models. We used model averaging for the top ranked models with a cumulative Akaike weight >0.95 to compute parameters and unconditional variances. The *Area* grouping parameter allowed for estimating population size and density for each subregion separately. We conducted analyses in program RMark v2.1.7 (Laake 2013) for R 3.0.2 (R_Core_Team

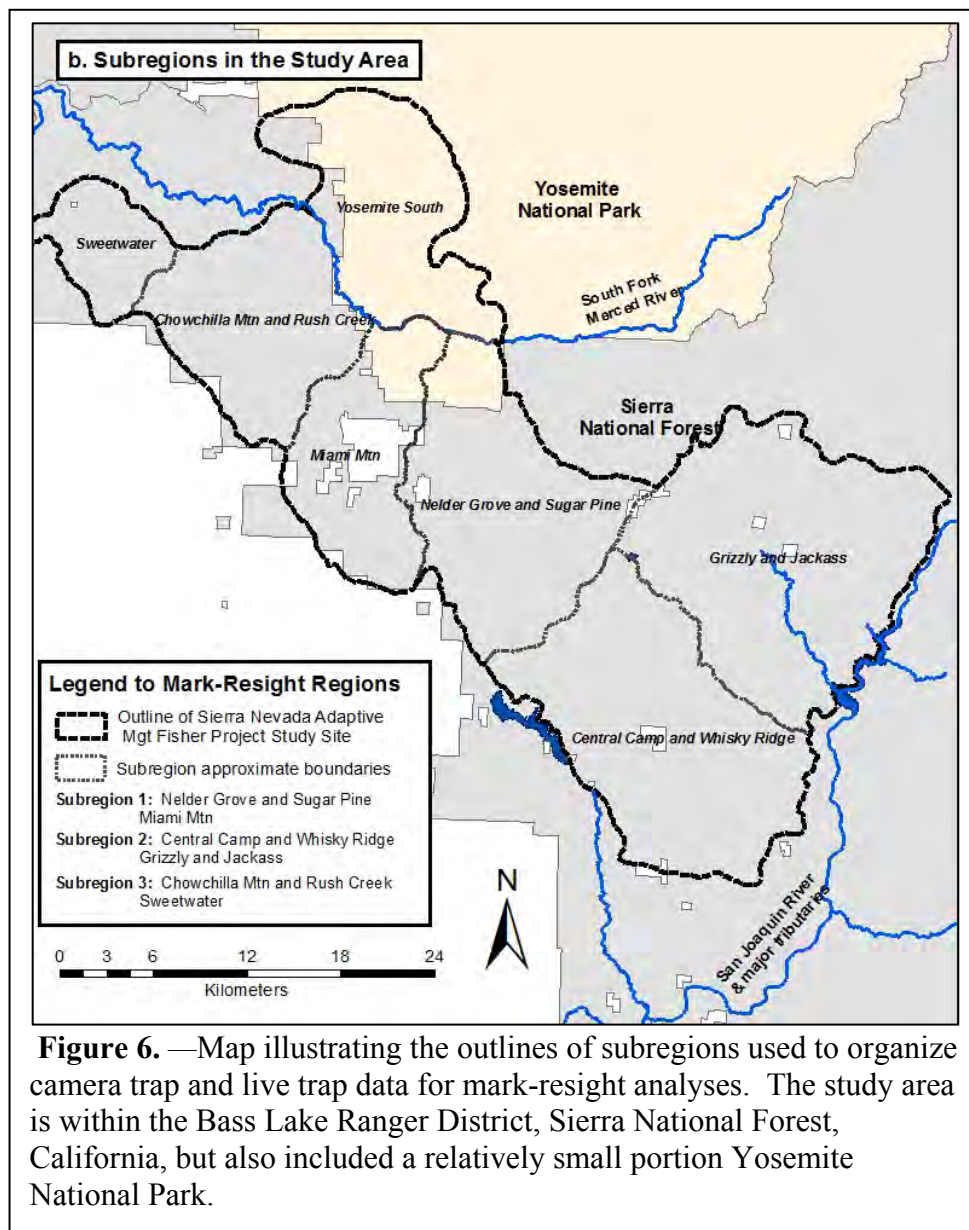


Figure 6. —Map illustrating the outlines of subregions used to organize camera trap and live trap data for mark-resight analyses. The study area is within the Bass Lake Ranger District, Sierra National Forest, California, but also included a relatively small portion Yosemite National Park.

2013), which is an interface for program MARK (White and Burnham 1999). Lastly, the subregion and year-specific abundances were converted to densities by dividing population estimates by the area sampled by cameras and traps for each subregion and year. Areas sampled were estimated from subregion- and year-specific polygons created in ArcGIS 10.2 that encompassed the centroids of all 1-km² grid cells with a survey camera or a live trap with a fisher capture during October 15 to March 16. We then plotted the fall-winter home ranges with the sampling polygons, and, based on visual assessment of spatial intersection of the 95% home range isopleths, applied a 1300 m buffer for each polygon. The width of the buffer for the polygons was the radius of the mean 95% fixed kernel fall-

winter home range for subadult and adult female fishers in the population (mean = $20.8 \text{ km}^2 \pm \text{SE } 0.89$, $n = 70$; Jordan 2007), which encompassed most areas used by radiocollared fishers resident in each subregion and excluded areas below or above the typical elevation range of fisher camera detections in the study area.

Dispersal Movements

Background

By simply moving from one habitat patch to another, dispersal of individuals has consequences not only for fitness, but also for population dynamics, population genetics, and species distribution at the landscape scale (Chepko-Sade and Halpin 1987, Lambin 1994, Clobert et al. 2001). For these reasons, processes that foment dispersal behavior have been the focus of research interest in relation to inbreeding avoidance, intraspecific competition for mates and resources (Estes-Zumpf and Rachlow 2009, Wolff et al. 1988), and costs and benefits of dispersal, particularly in relation to gender (Pusey 1987).

Natal dispersal, permanent movement from the natal area to the location where individuals reproduce or would have reproduced depending on survival (Howard 1960), is the most common type of dispersal. Gender bias in which one sex, typically males, disperses more frequently or farther than the other, has been documented in many mammals (Greenwood 1980; Pusey 1987, Sweitzer and Berger 1998). Proximate mechanisms triggering natal dispersal and potentially influencing dispersal distance include population density (Gaines and McClenaghan 1980), habitat quality (Lidicker 1975), and body condition (Dufty and Belthoff 2001, Nunes and Holekamp 1996). Information on dispersal provides insights on how far, and over what sorts of terrain, individuals may move and therefore how populations may be demographically and genetically interconnected or isolated. Barriers or impediments to dispersal reduce gene flow and may prevent populations from colonizing or recolonizing suitable habitat areas.

Dispersal behavior by fishers is of high management interest in California where the species currently occupies less than half of its historical range as described in the early 1900s (Grinnell et al. 1937). In the southern Sierra Nevada conservation planning area, fishers occupy approx. $4,400 \text{ km}^2$ of mid-elevation, mixed-coniferous forest between the Merced River in Yosemite National Park in the north to the Greenhorn Mountains in the Sequoia National Forest in the south (Fig. 2, Table 2). The southern Sierra Nevada population does not appear to be expanding geographically (Zielinski et al. 2013), despite changes in management promoting redevelopment of suitable fisher habitat in the Sierra

Nevada (North et al. 2009). Dispersal movements by fishers are potentially inhibited by exposure to multiple restrictive habitat and landscape features (Spencer et al. 2014, Tucker 2013). Moreover, Matthews et al. (2013a) suggested that because of their relatively limited vagility, conservation-directed management to promote fisher recovery in formerly occupied portions of their range in California may require translocations, unless population growth rates significantly exceed 1.0.

We used information on juvenile home ranges, likely maternal home ranges (determined by genetic analyses), and adult home ranges to evaluate patterns in natal dispersal for fishers in the SNAMP Fisher study area. We hypothesized that (1) young male fishers would disperse at a higher rate than females, (2) dispersal distances for males would be longer than for females, and (3) long distance movements would be more frequent for males compared to females. Because of their large home ranges, dispersal movements by fishers may be mediated by restrictive landscape features (Carroway et al. 2011). Therefore, in addition to estimating Euclidean distance between juvenile or maternal home ranges and adult home ranges, we also used a least-cost corridor analyses with an expert opinion-based cost surface to estimate both short and longer distance movement paths associated with natal dispersal.

Assessing dispersal using home range models

Location records were used to develop home range models for individual fishers using the fixed-kernel density method in Home Range Tools for ArcGIS 9.3 (Rodgers et al. 2007; ESRI, Redlands, CA). We developed 95% fixed-kernel home range models for juvenile, subadult, and adult-age fishers when ≥ 25 locations were available for the pre-dispersal or post-dispersal period of interest. Approximate center positions (centroids) were estimated for each home range using the XTools extension in ArcGIS (Data East LLC, Novosibirsk, Russia). Because both area estimates and shapes of fixed kernel home ranges are sensitive to the choice of bandwidth as a smoothing parameter (Gitzen and Millsaugh 2003), we used the Ad Hoc method to identify the most appropriate reference bandwidth for smoothing fisher home ranges and minimizing formation of multiple polygons (Kie 2013). Finally, in some cases radiocollars were shed by juveniles within a few weeks of initial capture, before ≥ 25 locations records had been acquired. In these cases we used centroids from 100% Minimum Convex Polygons for natal area centroids (Aubry and Raley 2006).

Dispersal distance by Euclidean geometry

Minimum distances moved between natal or maternal home ranges, and subadult or adult home

ranges were estimated as the Euclidean distance between each pair of centroids. For juvenile fishers without maternity assignments, we used fall and winter location records to identify a centroid for natal areas, but excluded locations that were associated with initiation of dispersal. Fall and winter location records for juvenile fishers were visually screened in ArcGIS to identify calendar dates associated with initiation of the exploratory, or transitional period of the dispersal process (Vangen et al. 2001). Location datasets used to develop home ranges for juvenile fishers (natal area home ranges) were truncated by date to exclude transitional movements. Transitional movements were not apparent for all juvenile fishers, however, and in these cases we used the pool of location records from capture to approx. March 31 for the natal area home range.

Microsatellite genetic analyses for identifying maternity

We used microsatellite genotypes to assign maternity for juvenile and subadult fishers, which allowed for estimating natal dispersal from the centroids of denning season home ranges for their mothers. Whole genomic DNA was extracted from fisher tissues and hair using the QIAGEN Dneasy Tissue Kit (Qiagen, Valencia, CA, USA) according to manufacturer's instructions. We analyzed 111 samples at the following thirteen microsatellite loci: *Ma1*, *MP059*, *MP144*, *MP175*, *MP197*, *MP200*, *MP247*, *Ggu101*, *Ggu216*, *Lut604*, *Lut733*, *Mer022* and *Mvis002* (Davis and Strobeck 1998; Flemming et al. 1999, Dallas and Piertney 1998; Jordan et al. 2007). These loci were previously found to be variable in fishers in the Southern Sierra (Jordan et al. 2007; Tucker et al. 2014).

Maternity of kits was evaluated using two approaches; the first was by evaluating allele sharing. Fishers in the southern Sierra Nevada were previously known to be genetically limited (Wisely et al. 2004, Knaus et al. 2011), and prior to step 1 we used insight from field associations (capture positions, home range patterns, denning season data to identify small subsets of 3-6 adult females considered possible mothers for each juvenile/subadult. These subsets of possible mothers were further narrowed to a smaller “candidate set” by excluding those that did not share alleles with the juveniles. We applied the maximum likelihood approach in program CERVUS v3.03 (www.fieldgenetics.com) to evaluate the candidate set of females for maternity assignment. CERVUS is a Windows-based software package for inferring parentage in natural populations wherein laboratory typing error is considered along with data on population allelic frequencies, the number of candidate mothers, and the proportion of potential mothers sampled in Monte Carlo simulations, which produce confidence levels for the candidate set of putative parents (Slate et al. 2000). The confidence measure of CERVUS is based on delta, which is the difference between the likelihood ratio for the most likely

candidate and the second most likely candidate (Marshall et al. 1998). We assigned maternal-offspring pairs based on likelihood ratio LOD scores (natural log of the likelihood ratio) using both strict (99%), and relaxed (95%) confidence. In the last step we considered the maternity assignments with field data to verify, or select the next most likely female from the LOD scores based on the known biological status of putative mothers (reproductive or non-reproductive, age as juvenile, subadult or adult in the season juveniles were born). In several cases, the overall analysis was unable to link juveniles to a known, radiocollared female in the population. Developers of CERVUS previously determined that the analytical procedure correctly assigned parentage for ~92% of known fathers in red deer (*Cervus elaphus*) (Slate et al. 2000). In our study we evaluated the performance of CERVUS to correctly identify mothers using five known mother-offspring pairs.

Dispersal distances for juvenile or subadult-age fishers with maternity assignments were estimated as the Euclidean distance between the centroid of the denning season home range of the mother, and the centroid for the subadult or adult home range where the juvenile settled. In some cases the mother had not been monitored during the denning season when a juvenile was produced. In these cases we used the centroid for the female's "annual" home range. Annual home ranges were calculated when we had at least 75 location records, with a minimum of five locations in at least three of the four seasons of the year. Seasons were spring (Mar 21 to Jun 20), summer (Jun 21 to Sep 20), fall (Sep 21 to Dec 20), and winter (Dec 21 to Mar 20).

Least-cost paths for natal dispersal

Dispersal is most often reported as the Euclidean, or straight-line distance between the natal area and the subadult or adult home range (Matthews et al. 2013). Fishers in the southern Sierra Nevada inhabit mountainous areas within a limited elevation range and with a mix of forested areas with mild topography, and ridges and deep river canyons with extreme topographic relief. In these types of landscape and habitat conditions opportunities for straight-line movement traversing multiple kilometers will be constrained.

Least-cost modeling is an approach for assessing potential animal routes across the landscape based on the assumed cost of movement between locations or termini (Beir et al. 2008). Least-cost models have previously been used to predict dispersal paths for mammals from empirical data (Driezen et al. 2007), and we took a similar approach in this study. Connectivity analysis was performed between centroids of natal and established juvenile home ranges for 24 female and 20 male fishers with Linkage Mapper (McRae and Kavanagh 2011). Linkage Mapper uses a resistance to movement

(cost) surface layer to delineate least cost paths between focal point pairs. A cost surface layer was developed that assigned a resistance score (inverse permeability value) representing the cost to fishers of moving through each land cover type, including potential risk and averse responses to roads and steep topography in river canyons (Fig. 7).

Expert opinion resistance scores were modified from those developed previously for Sierra Nevada fisher least-cost corridor models (Spencer and Rustigian-Romsos 2012) by (1) simplifying the land cover divisions, (2) expanding the overall cost range, and (3) incorporating recent published and unpublished data on fisher ecology summarized in a conservation assessment developed for fishers in the southern Sierra Nevada by a group of 13 research scientists (Spencer et al. 2014). We used the Polynomial Approximation with Exponential Kernel (PAEK) algorithm in ArcMap 9.3.1 (ESRI 2009) to smooth the movement paths for purpose of display. Length of least cost paths between juvenile or maternal home range centers and subadult or adult home range centers were calculated in ArcGIS 10.2. Basic metrics on least cost paths (means, range, standard error of the mean) were produced and summarized for comparison with mean dispersal distances from Euclidean geometry.

Analysis

Mean dispersal distances were compared between female and male fishers using two-sample *t*-tests. We also assessed potential gender differences in dispersal behavior using Pearson χ^2 or log-linear model analyses. We used dispersal distances to classify each fisher as being either philopatric (dispersal distance ≤ 5.4 km) or a disperser (dispersal distance > 5.4 km), where 5.4 km was the diameter of average 95% fixed kernel home range of adult females fishers in the study population (22.93 km², $n = 56$; Table 31). We also assessed the overall pattern in dispersal behavior by assessing the proportion of each gender that was very philopatric (dispersal distance < 2.7 km; one-half the diameter of the mean 95% fixed-kernel home range for adult female fishers), short distance philopatric ($2.7 \text{ km} \leq \text{dispersal distance} < 5.4 \text{ km}$), a mid-distance disperser ($5.4 \text{ km} \leq \text{dispersal distance} < 10.8 \text{ km}$), or a long distance disperser (dispersal distance greater than 10.8 km; 2x the diameter of the average adult female home range).

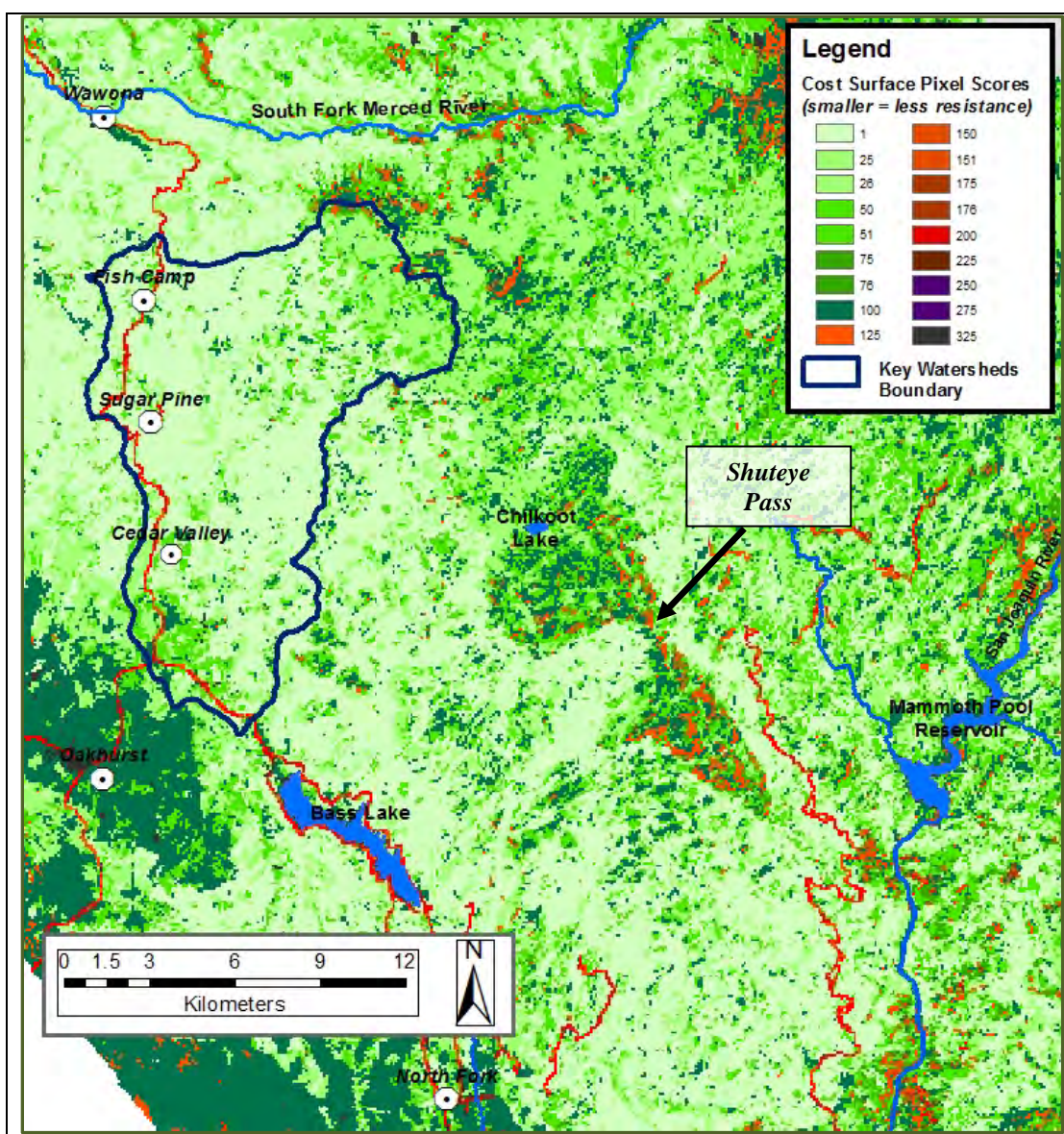


Figure 7. Illustration of the Expert opinion cost surface used to develop Least Cost movement path for dispersing fishers. The map encompasses the portion of the SNAMP Fisher Study Area including the Key Watersheds and the area to the southeast including Chilkoot Lake and Mammoth Pool Reservoir. NOTE: Chilkoot Lake is at the northwestern edge of the Chiquito Ridge, a high elevation region including Little Shuteye Peak, Shuteye Peak, and Shuteye Pass; notice the narrowness of restrictive habitat at the Shuteye Pass topographic feature.

Home Range Dynamics Methods

Background

Among terrestrial vertebrates, mammalian carnivores have the largest home ranges for their body size of any organism. The fisher, like other mammalian carnivores, occupies a relatively large

amount of space for its body mass, with average annual home range areas of 38 km² for adult males and 15 km² for adult females across North America (Powell 1994). Studies of the two remnant populations in California have produced home range area estimates generally consistent with North American averages: 22 to 58 km² for adult males and 5 to 15 km² for adult females (Boroski et al. 2002, Zielinski et al. 2004). Zielinski et al. (2004) also reported intraspecific variation in home range size between adult females of the northern coastal and southern Sierra Nevada populations.

Intraspecific variation in home range size has been linked to ecological factors such as population density, prey availability, body mass, and latitude (Buskirk and McDonald 1989, Gompper and Gittleman 1991), and to methodological factors such as sampling interval and duration (Buskirk and McDonald 1989, Swihart and Slade 1985). Our review of the literature suggests that little attention has been paid to potential relationships between home range size and field techniques used to obtain animal locations. Further, the choice of an appropriate bandwidth, or smoothing parameter when creating utilization distributions is a critical step during kernel-based home range estimation in need of standardization (Kie et al. 2013).

We present and discuss home range dynamics for fisher in the Sierra National Forest, while also describing seasonal variation in home range movements for female and male fishers. We hypothesized that aerial telemetry would be more likely than ground telemetry to detect animals outside of their core use areas and during dispersal events and sallies, and would therefore produce larger home range estimates. Our primary objective was to compare our fisher home range sizes with those generated by other studies in the southern Sierra Nevada, where ecological factors would be held relatively constant. Additionally, we wished to characterize variation in space use between genders, among age classes, and across seasons for our study population.

Locations and analyses

Fisher locations from live-trap captures, dropped or shed radiocollars, carcasses, dens and rest trees, camera trap detections, a small number of GPS radiocollars (Telemetry Solutions, Livermore, CA) and position estimates from fixed-wing aerial radiotelemetry (Table 3) were used to delineate home ranges. Accuracy of locations obtained by homing to den tree or rest tree locations by ground telemetry, camera trap detections, capture positions, and carcass and dropped collar positions were generally ± 15 m from a handheld global positioning system device. We addressed and minimized autocorrelation by discarding locations in excess of two per animal per day, or less than 8 hours apart per individual. Location records were used to develop home range models for individual fishers using

the fixed kernel density method in Home Range Tools for ArcGIS 9.3 (Rodgers et al. 2007). Ninety-five percent fixed kernel home ranges were produced for individual animals for four seasons, and for “annual” population years. Seasons were defined as follows: fall - September 21 to December 20; winter - December 21 to March 20; spring - March 21 to June 20; summer - June 21 to September 20. Season-specific home range models were produced when ≥ 25 locations were available for a fisher. Annual home range models were developed when we had location records in fall, winter, and least one other season, and sample size was ≥ 75 for all annual home range models. Kernels were smoothed using the minimum proportion of reference bandwidth that produced a contiguous home range polygon (Kie 2013). Areas (km^2) of home ranges were calculated during kernel processing. We tested for differences in home range areas between males and females, stratified by age and season, using two sample t-tests ($P < 0.05$). Potential differences in home range size among seasons was assessed using repeated measures analysis of variance (ANOVA) ($P < 0.05$).

Table 3. Types of locations determined for fishers during the study period from December 2007 to December 2013. Data are for study in the Bass Lake District, Sierra National Forest, CA.			
Location type	No. of locations	UTM accuracy	Description of methods and details
Aerial telemetry ^a	31,367	± 339 m	Standard methods by fixed-wing airplane (Sweitzer 2013)
Camera trap detections	2,454	± 10 m	Position of camera trap; individuals fishers identified by infrared tape on radiocollar antennas (Popescu et al. 2014)
GPS radiocollar ^b	633	± 15 m	Used on limited number of animals ($N = 8$) in 2009 and 2010
Den or rest tree	526	± 10 m	Homing to trees by ground radiotelemetry during spring denning seasons; did not identify rest trees in other seasons
Live trap capture	277	± 10 m	Trap positions for known ID captures; most live-trapping was in October to March
Shed radiocollars, fisher carcass	97	± 10 m	Homing to inactive signals by ground radiotelemetry
^a Accuracy determined as the mean Euclidean distance between aerial telemetry location and fixed position of test collars ($n = 501$) on the ground. Test locations also included locations to dropped radiocollars, carcass locations, and fishers in live traps (Technicians were "blind" to locations of test collars, or other test locations). ^b Used for limited duration and primarily on male fishers (596 locations for 6 different males; 37 locations for 2 females). SNAMP Fisher ceased using GPS collars due to poor reliability and bias in fix rates; fix rates were high when GPS collars were left in open areas with mild topography, and low when GPS collars were placed at locations with dense overhead canopy and steep topography.			

SNAMP Fisher Management Indicators

Background

In 2008 there was great interest in new information developing from SNAMP Fisher that might be important for management and management planning. We therefore developed three Indicators for fisher management that would provide insight on the status of the study population of fishers in the Bass Lake District, Sierra National Forest. These management indicators were chosen based on information that could be summarized annually, and that linked to the likely responses of fishers to management and potential habitat change at the local (sub home range scale), home range, and population level (larger landscape scale relevant to District-level forest management; Table 4).

Table 4. Overview of potential negative effects of fuel reduction treatments and other forest management activities on the biology and natural history of fishers, organized according to three scales in the SNAMP Fisher study area, Bass Lake District, Sierra National Forest.

<i>Scale of effect, Description</i>	<i>Likely response</i>	<i>Data requirements</i>	<i>Management Indicator</i>
LOCAL: SPLATs may cause habitat patches to become less suitable for current use; foraging, refuge/escape cover	Use of treated areas declines or ceases	Before/After and ongoing use of areas altered by management activities	Use of 1-km ² grids within Key Watersheds (or in other areas), estimated annually using camera trap surveys
HOME RANGE: SPLATs may reduce availability of key resources such as den sites, rest sites, availability of prey	Individual fishers cease use of treated areas	Monitor individual fishers, acquire locations, develop home range models, track dispersal movements	Estimate of the number of individual fishers using the Key Watershed focal study area during population years
POPULATION/REGION: Multiple projects implemented every few years may degrade suitable habitat for fishers; population source areas become sink areas	Survival and reproduction decline; population size and density decline over time	Information on survival and reproduction of individual fishers in the overall study area (min 20 fishers monitored by radiotelemetry at all times)	Survival and reproduction of fishers in the overall study area. Estimate population growth rate, evaluate population viability in the Sierra National Forest

Management Indicator 1: Occupancy/Activity of fishers within Key Watersheds.

Beginning October 2007 we implemented regular camera trap surveys of all 1-km² grids that are partly or entirely encompassed by the boundary of the SNAMP Fisher Key Watersheds (Fig. 4). Several grids that were predominantly private lands (e.g., the Fish Camp area), or that were below the typical elevation range of fishers in the region (< 914 m) were not surveyed unless we had permission of access from the landowner. The annual resurvey of the Key Watersheds was a research priority in all years, and camera trap surveys were initiated at the start of each camera survey year (mid-October), continuing until most grids had been surveyed by late winter or early spring. High elevation areas (northeast portion of Key Watersheds) were generally surveyed first, due to more difficult access during mid and late winter. Deep snow conditions in most winters required use of snowmobiles or an all-terrain-vehicle modified with tracks to access high elevation grid centers. Results of the Key Watersheds camera surveys were summarized according to (1) fisher presence or absence, and (2) level of fisher activity based on the number of days fishers were detected in each grid.

Management Indicator 2: Number of resident female and male fishers using the Key Watersheds area.

Juvenile fishers exhibit exploratory movements, and sometimes dispersed away from their natal areas where we first captured and monitored them. Dispersal by juvenile fishers often extends into summer when they are 13-15 months old and considered subadults. We considered subadult fishers (12 to 23 months old) to be “settled” after natal dispersal in late August/September. Ninety-five percent fixed kernel home range models were developed from location records during September 1 to March 15 for all radiocollared fishers (Sept-March home range). Analyses were completed in ArcGIS 9.3.1 to estimate the proportion of each Sept-March home range for subadult and adult fishers that were included or “intersected” within the boundary of the Key Watersheds. Management Indicator 2 was calculated as the sum of the proportions of individual subadult and adult Sept-March home ranges within the Key Watersheds focal study area. Management Indicator 2 was calculated for female and male fishers for each of six September to March 15 periods beginning September 2008 and ending March 2013.

Management Indicator 3: Survival of adult-age female fishers in the SNAMP Fisher Study Area

Survival, and survival of adult females in particular, is an important demographic parameter necessary for understanding the population growth trajectory for most vertebrate wildlife species (Murdoch et al. 2010). All radiocollared fishers were monitored 4-6 days/week by fixed-wing aerial

telemetry to assess live/dead status. Information on survival status for radiocollared fishers was organized by month of each population year (Apr 1 to March 31), and analyzed using the Kaplan-Meier (KM) staggered entry method (Koen et al. 2007, Pollock et al. 1989, Price et al. 2010). KM models were used to produce estimates of annual survival and combined year survival for the study population. Annual survival can be moderately to highly variable, thereby suggesting a negative population growth trajectory that may not be appropriate for a long-lived species with a generation time of two or more years. We therefore combined monthly data on survival status for individual fishers for five 2-year periods (population years 2 and 3, population years 3 and 4, population years 4 and 5, population years 5 and 6, population years 6 and 7) beginning April 2008 and ending March 2014. We further extended the inference for this management indicator by estimating survival for juvenile and subadult females, and by compiling data on weaning reproductive rates and weaning litter sizes from data collected on fisher reproduction during April 2008 to June 2013. These data were used to estimate fertility rates, which, along with data on juvenile, subadult, and adult female survival, were used to estimate deterministic population growth rates using a four age class Leslie Matrix population model.

Fisher response to fuel management

Occupancy modeling

For the purpose of occupancy surveys, we deployed cameras in the 128 1-km² grids that were $\geq 50\%$ public lands and within the 4 focal watersheds. We also deployed cameras in areas with recent histories of extractive or restorative fuel reduction, between 2002 and October 2008, or because forest management projects were planned to occur in the areas before December 2011. Most of these grids were repeat surveyed in 7 different camera years, as part of our initial plan of using a BACI framework for the occupancy analyses (Popescu et al., 2012). However because we were not aware of all planned or prior Forest Projects within the study area when the project was initiated, some of the multi-season grids were added to the group that were repeat surveyed several years after the first camera year (2007-08).

The distribution of fishers in the southern Sierra Nevada, CA is constrained by elevation, and closely associated with mixed-conifer forest habitats with relatively large trees, and high canopy cover (Davis et al., 2007). We therefore developed local, patch-specific biophysical covariates for use in analytical models of occupancy. We calculated the mean elevation (*elev*) for each surveyed grid, which was always included in occupancy analyses with its quadratic term (*elev*²). These covariates

were standardized. Habitat covariates included the proportion forest (i.e., total tree) and hardwood cover (*denMD*) based on land-cover data derived from satellite imagery (CWHR CalVeg; USDA Forest Service 2012). We did not include covariates representing average tree size and slope because of their colinearity with forest cover and elevation.

There were a diversity of forest management activities that occurred on the Sierra NF from 2002 (5 years before the start of our study) until the last camera survey year starting in October 2013 (period 2002 to 2013). Most of the management activities we used for covariates were developed from the USDA Forest Service FACTs database. FACTs (Forest Service Activity Tracking System; <http://www.fs.usda.gov/main/r5/landmanagement/gis>), is a tracking system including a geospatial database of forest management activities that occur on national forest service lands in California and elsewhere (FACTs User Guide 2013). Polygon layers included in the FACTS database are associated with attributes detailing management activity codes, and dates for when activities were initiated and completed. There are known uncertainties in FACTS with regards spatial precision, area of treatment polygons, and lack of details on whether a treatment activity was completed for an entire polygon (Garner, 2013). We also know that some entries represent perimeters encompassing smaller subunits treated at the same time as well as some areas unaffected by the management activity (Zielinski et al., 2013). Nevertheless, FACTS data constitute the best available and consistent record of the annual management activities that occurred on national forest lands in our study area.

Two recent studies used FACTS information to assess how fishers respond to disturbances from Forest Projects elsewhere in the southern Sierra Nevada (Garner 2013; Zielinski et al., 2013). We considered FACTS activities that were previously used in those studies, but also reviewed full descriptions of each management activity included in the FACTs User Guide (2013) when identifying a subset of 24 that were considered as potentially influencing local scale habitat use by fishers related to how each altered forest habitat structure or if they represented significant ground-disturbing activities (Zielinski et al., 2013; FACTs User Guide 2013). For example, we included forms of harvest (e.g., code 4152 Group Selection Cut) and vegetation management (e.g., code 4220 Commercial Thinning, code 4580 Mastication/Mowing) that would have direct effects on the basis of their disturbance and alteration of forest structure (Zielinski et al., 2013). We excluded activities that did not meet this criterion, and several that rarely occurred, or that silviculturalist Dave Smith with the Sierra NF recommended against using (e.g., code 4290 Administrative Changes; code 4314 Pretreatment Exam for Reforestation; code 4530 Prune; code 4511 Tree Release and Weed; code 4552 Area Fertilizing; code 4980 Other Tree Improvement; code 4540 Control of Understory Vegetation).

There were 4 other activities or events that were not systematically tracked by the FACTS system; hazard tree removals (e.g. hazard tree logging), private timber harvests (THPs), and historical or recent wildfires. Hazard tree logging was the removal of medium and large trees (no DBH restriction) within 91 m of forest roads when they were considered likely to fall during storms, or if they were decadent or diseased (SNFP 2004). Information on hazard tree logging in the Bass Lake Ranger District was available for 2009, 2010, and 2011, and we were provided with GIS shapefiles identifying road segments along which hazard tree logging occurred. Private timber harvest occasionally occurred on large parcels of private land within or adjacent to the Sierra NF in Madera County and Mariposa County. Harvesting of timber on private lands in California requires preparation of Timber Harvest Plans (THPs) that are reviewed and approved by state agency Calfire, which was our source for geospatial data on private THP activities in Madera County and Mariposa County (<ftp://ftp.fire.ca.gov/forest>). Basic records on the estimated spatial extent of wildfires that occurred in the national forest portion of study area were maintained by the Sierra NF, and included polygon shapes and ignition dates of wildfires that occurred from 1911 to 2013. We also acquired geospatial data on natural ignition and management fires for Yosemite NP for 1930 to 2008, which was sufficient for our analyses because there were no camera surveys completed in southern Yosemite NP after May 2009. Attribute information included with the various geospatial data were used to assign activities and wildfires to individual camera survey years. For example, if a management activity was identified as completed before October 15, 2009, the disturbance was assigned to camera year 2008-09.

We used ArcGIS 10.2 (ESRI, Redlands, CA) to estimate the area of each 1-km² surveyed grid with hazard tree logging, private timber harvest, and wildfires, which were merged with the FACTs information for 2002 to 2013. After merging, we reviewed the entries, and removed polygons that were duplicated in several years (e.g. those with the same FACTS code with identical shapes and areas but with different years of completion). We also removed any duplicate wildfire records that were included in both the FACTs data and in the local Sierra NF wildfire database. We then used the detailed descriptions of each FACTs activity type to create 3 composite variables for use as covariates for occupancy analyses. Covariates for extractive fuel reduction (*log.5*) and restorative fuel reduction (*hazfuel.5*) included the cumulative areas of these activities in each grid in the 5 years immediately preceding each camera trap survey. For example, the *hazfuel.5* covariate for any grids that were surveyed in camera year 2012-13, was calculated as the sum of the areas (m²) of all restorative fuel reduction activities that occurred in those grids during fiscal years 2007-08, 2008-09, 2009-10, 2010-11, and 2011-12, from which we calculated the proportion of the 1-km² grids disturbed by the

treatment. Because of the coordinated series of extractive and restorative fuel treatments associated with SPLATs, multiple different treatments could be applied on the same forest stand within a 5 year period (Zielinski et al., 2013). It was therefore possible that the cumulative area of a grid that was treated during a 5-year period could exceed 1-km². In the few cases where this occurred (*hazfuels.5* only), the proportion of the grid treated was truncated at 1.0 (100%).

The third composite variable that was related to fisher presence in model analyses was for managed burning and wildfires within each 1-km² grid. When we reviewed the FACTS and Sierra NF and Yosemite NP databases, it was apparent that managed burning was uncommon in the study area during 2002 to 2013. Although managed burns were commonly planned in the Sierra NF portion of the study area as part of SPLAT-based fuel reduction, many managed burns were cancelled and not rescheduled because weather conditions were not suitable, or because burning was prohibited by the San Joaquin Valley Air District (D. Martin, personal communication). Also, a late summer managed burn in Yosemite NP in 2009 escaped containment and burned 7,425 ha (Big Meadow Fire), which discouraged other managed burns in the region for several years thereafter. We therefore combined information on managed burning and the longer time-series of wildfires in the study area into a single composite variable representing managed burn+wildfires within 50 years of a survey (*burn.1.50*).

We used multi-season occupancy models to evaluate the importance of forest management covariates to explain the persistence of fishers at occupied grids and colonization of unoccupied grids (Zielinski et al., 2013). We defined colonization (γ) as the probability that a grid unoccupied in year t would be occupied in year $t + 1$, and modeled it as: $\text{logit}(\gamma) = \beta_{\gamma 0} + \beta_{\gamma 1}X_1 + \beta_{\gamma 2}X_2 + \dots$. We defined persistence as 1 - extinction where extinction (ϕ) was the probability that a grid occupied in year t would be unoccupied in year $t + 1$, and modeled it as: $\text{logit}(\phi) = \beta_{\phi 0} + \beta_{\phi 1}X_1 + \beta_{\phi 2}X_2 + \dots$. The multi-season models also included a component for occupancy in the initial year a site was surveyed: $\text{logit}(\psi_{\text{initial}}) = \beta_{\psi_{\text{initial}0}} + \beta_{\psi_{\text{initial}1}}X_1 + \beta_{\psi_{\text{initial}2}}X_2 + \dots$.

We created a detection history of whether a fisher was observed by a camera trap within each grid during each consecutive survey period after set-up or re-baiting for up to 5 8-10 day periods during a survey year. This was repeated for up to 6 consecutive years (e.g., 00101 00000 01110 00010 01101 00000) for every grid. If surveys did not occur during any of the 5 periods and 6 seasons at any of the grids these data were treated as missing data. Models were solved by maximum likelihood estimation (MLE) via R statistical software (Version 3.0.1, www.r-project.org) using the *unmarked* package and the *colext* function. (Fiske and Chandler 2011). We followed an information-theoretic approach for evaluating the relative importance of different candidate models, and for assessing the relative

importance of individual covariates [sum of AIC weights (AIC_{wi}) for candidate models including each covariate; Burnham and Anderson 2002].

Covariates for potentially explaining detection probability included a categorical, first order Markov process reflecting whether a fisher was detected in the previous survey period in a season (*auto.y*; Hines et al., 2010; Slauson et al., 2012), the number of functional camera days in a survey period divided by 10 (*camdays*), *denMD*, and a categorical variable representing whether the survey was conducted in summer (*summer*) instead of in fall to spring.

Due to the smaller sample size of sites available for fitting multi-season models ($n = 361$), we only evaluated the role of forest management covariates (*log.5*, *hazfuels.5*, and *burn.1.50*) in explaining annual transitions in occupancy state (colonization and extinction). For the initial occupancy component of the multi-season models, we restricted potential explanatory covariates to *denMD* and *elev + elev²*. For multi-model evaluations of multi-season models, we first fit models including all 8 combinations of the forest management covariates on the colonization component and an intercept-only extinction component. We considered any covariate with a relative importance value > 0.65 to be predictive and important for colonization. Next, we fit models including all 8 combinations of the forest management covariates on the extinction component multiplied by all combinations of colonization covariates identified as important. We deemed any covariate in the extinction models with a relative importance value > 0.65 as predictive for explaining local extinction. Finally, we computed model-averaged parameter estimates for the colonization and extinction covariates identified as important. Model averaging was based on only those models summing to the top 0.95 of model weights.

Integration

Development of vegetation map

We refer the reader to Appendix C for more complete details because we used the same mapping procedure at Sugar Pine as we did at Last Chance. In summary, we developed a pre-treatment vegetation map using a combination of LiDAR, high-resolution digital color-infrared (CIR) aerial imagery, and an intensive network of field plots. First, we used LiDAR and CIR data to create an initial polygon-based map where the polygons represented areas of homogeneous vegetation in terms of species, vertical structure, basal area, and canopy cover. We collected the LiDAR and CIR data before the SPLAT implementation, and we sampled vegetation at the field plots before and after treatment. We then used the field-plot data to impute detailed attributes (e.g., tree lists and fuels

models) for each polygon. Thus, we derived two different maps (with and without treatment), which we used in fire and forest-growth modeling.

Modeling fire and forest dynamics

We again refer the reader to Appendix C for more complete details of the fire and forest-growth simulations because we followed the same general procedure at Sugar Pine as we did at Last Chance. We used FARSITE (Finney 1998) to simulate a likely wildfire scenario based on the weather conditions during the 2014 French Fire, which burned 13,837 ac (5,602 ha) 12.5 mi (20 km) southeast of the study area. We obtained weather information from the Batterson Remote Automatic Weather Station, limited to the active burning period of the French Fire (August-September 2014), which served as the basis of our fire modeling. Moisture content for live and dead woody fuels and live herbaceous fuels used in the model were equivalent to 97th percentile weather conditions. Our ignition location was established using fire-origin point data supplied by the Bass Lake Ranger District of the Sierra National Forest. Based on the mapped data, we identified an area with the highest ignition frequency, which was located on the west ridge of the Cedar Valley watershed (see Figure 1). The simulation duration was set to allow the fire perimeter to expand through the entire study area.

For all four scenarios (treated/fire, untreated/fire, treated/no fire, untreated/no fire), we then simulated 30 years of forest growth on the study area in 10-year time steps using the Forest Vegetation Simulator (FVS; Dixon 2002) with the Fire and Fuels Extension (FFE; Reinhardt and Crookston 2003). The simulations were performed using the integrated platform ArcFuels (Ager et al. 2006, Vaillant et al. 2011), which runs FVS-FFE to produce the forest structure inputs needed for FARSITE.

Assessing the effects of fire and SPLATs on fisher habitat

We identified canopy cover and large trees as the most important elements of forest structure for fisher habitat because fisher den and resting locations in the southern Sierra Nevada were associated with high canopy cover and large trees (Zielinski et al. 2004, Purcell et al. 2009, Thompson et al. 2011). We defined fisher habitat as forest stands where the canopy cover was $\geq 60\%$ and the density of large trees (≥ 24 in [61.0 cm] dbh) was ≥ 15.4 trees/ac (38 trees/ha).

We defined the canopy cover threshold for fisher habitat as $\geq 60\%$ because 95% fixed kernel home ranges for 16 adult female fishers in the Kings River Project area in the Sierra National Forest

averaged 63% (Thompson et al. 2011). Furthermore, fisher resting habitat sites are characterized by high canopy cover that is typically >60% (Purcell et al. 2009, Thompson et al. 2011), and the California Wildlife Habitat Relationships database uses a 60% canopy cover threshold as one of the criteria in its definition of high-quality fisher reproductive habitat (California Department of Fish and Game 2008).

We defined a large tree as ≥ 24 in (61.0 cm) diameter at breast height (dbh) because resting trees at the lower end of the size distribution (i.e., mean minus the standard error) in two different studies were of a similar size (Zielinski et al. 2004, Purcell et al. 2009). Thus, any tree ≥ 24 in dbh was potentially suitable as a fisher resting site. Next, we determined the threshold density of large trees (i.e., 24 in dbh) by examining stand-level tree lists surrounding den locations of 28 female fishers in the Kings River study area from 2008-2013 (Rebecca Green, unpublished data). When there were multiple dens per female, we randomly chose a single den for that individual. Data for natal dens were used preferentially; natal dens were where the young were born. We used data at maternal den locations for 7 females for which natal den locations were not available; fisher young were moved to maternal dens when conditions were no longer suitable at the natal dens. We defined the threshold density for large trees as ≥ 15.4 trees/ac (38 trees/ha) because this was the median density of large trees surrounding the 28 den locations.

Results

Basic Fisher Population

A total of 110 individual fishers were captured in live traps as part of the SNAMP Fisher Project from Dec 2007 to Dec 2013 (62 females, 48 males). In the first 3.5 months of trapping in population year 1 we captured 3.6 noncollared (“new”) fishers/100 trap nights, and 6.8 total fishers/100 traps nights. During population years 2008-09 to 2011-12 a mean of 0.94 previously unknown individual fishers were captured per 100 trap nights, and there were an average 2.43 total captures per 100 trap nights. Data on traps nights were not available for 2012-13 or for March to December 2013, when a total of 9 new fishers and 19 total captures occurred (Table 8).

No fishers died during capture and handling in the study. However, one adult female fisher captured in October 2009 did not fully recover. The female fisher was held in the trap cubby overnight for additional time to recover, but died the next morning while in transit to the Fresno Chaffee Zoo for treatment by a wildlife zoo veterinarian. A necropsy completed for the fisher identified her cause of

death as septicemia from a previously fractured jaw, which led to emaciation and starvation.

An overarching goal of the study was to monitor a minimum of 20 radiocollared fishers at all times, which was considered a requirement for producing reliable estimates of survival and reproduction for the population. The study achieved that milestone in mid July 2008, about six months after live trapping was initiated in late December 2007 (Fig. 10, Table 9). There were brief

periods in several years when the radiocollared population declined below 20 individuals (Fig. 10). The annual oscillation in numbers of radiocollared fishers was related to the combination of dropped or shed radiocollars (breakaway units built into radiocollars parted as designed), and mortality which was focused during spring and summer in all years of the study. After the end of our annual pause in live trapping during the spring denning season, the number of radiocollared fishers gradually or rapidly increased when trapping resumed, and as young-of-the-year juvenile fishers were recruiting into the study population in the fall and winter (Fig. 10). With the exception to the first and last year of the study, we were able to monitor survival for at least 40 different fishers in each population year. Notably, in 2011-12 we were monitoring more than 40 individual fishers for several successive months (Fig. 10, Table 9).

Table 8. Summary data on numbers of trap nights, new fishers captured, and recaptures for the SNAMP Fisher Project from December 2007 to December 2013. Population years start April 1 and end March 31

Population Year	Trap nights ^a	New individual s ^b	Recapture s	Total captures
2007-08	280	10	9	19
2008-09	2793	34	40	74
2009-10	2898	20	52	73
2010-11	2173	15	30	44
2011-12	1914	22	27	48
2012-13	<i>No data^c</i>	8	11	19
2013 ^d	<i>No data</i>	1	8	9

^a Number of traps set for capture during an overnight period.
^b Includes one orphan fisher kit captured in a live trap in 2010-11, and one orphan fisher kit captured in a live trap in 2011-12.
^c PSW Forest Service trapping, no data on trap nights.
^d Apr 1 to Dec 31; end of SNAMP Fisher.

Table 9. Number of radiocollared fishers being monitored for the SNAMP Fisher Project at the start and end of six different population years.

Population Year	Start N	End N	Individual fishers N ^a
2007-08		7	11
2008-09	6	30	41
2009-10	30	32	51
2010-11	32	32	55
2011-12	32	44	59
2012-13	44	33	50
2013 ^b	33	14	33

^a Number of individual fishers radiocollared and monitored for ≥ 1 day
^b Apr 1 to Dec 31; end of SNAMP Fisher

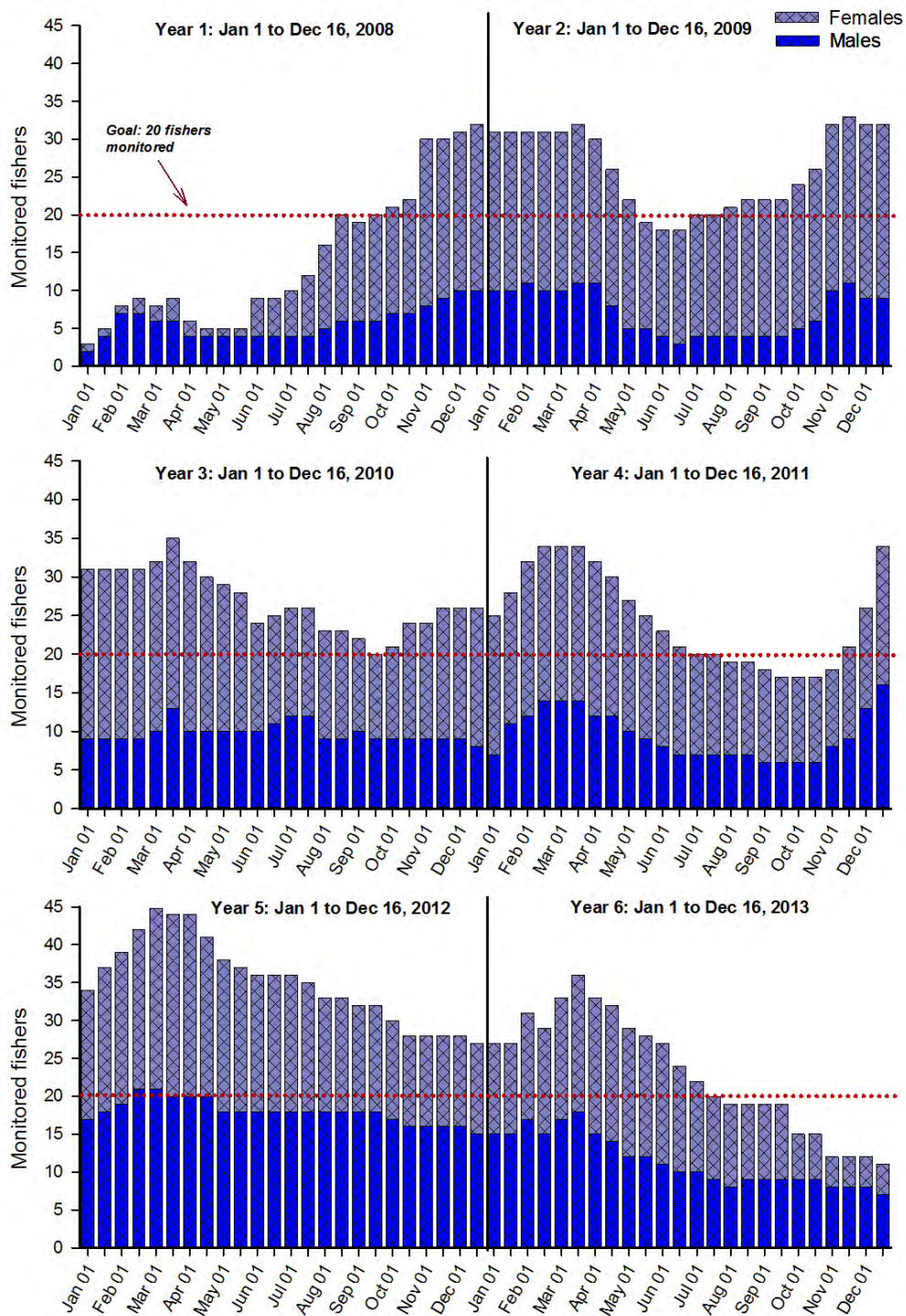


Figure 10. Number of radiocollared fishers that were monitored for survival and reproduction (females) during the period of SNAMP Fisher from December 2007 to December 2013.

Basic Camera Survey Results

Camera trap surveys were a major aspect of SNAMP Fisher in all years. In the overall period of the study we surveyed for fisher presence in 905 unique 1-km² grids. The distribution of camera surveys extended from Yosemite Valley in the north, to the slopes above the San Joaquin River canyon to the south and southeast (Fig. 11). Surveys occurred within Yosemite National Park in winter 2009 only, research that was part of a companion study organized by Reginald Barrett and funded by the California Department of Fish Wildlife. We also obtained data from camera trap surveys in 24 grids located north of the Merced River in Yosemite Valley (not displayed) that were completed by cooperating biologists from Yosemite National Park or the Central Sierra Nevada Environmental Research Center (CSERC). Fishers were not detected in any of the 24 grids, reinforcing that the Merced River is the northern edge of the range of fishers in the southern Sierra Nevada.

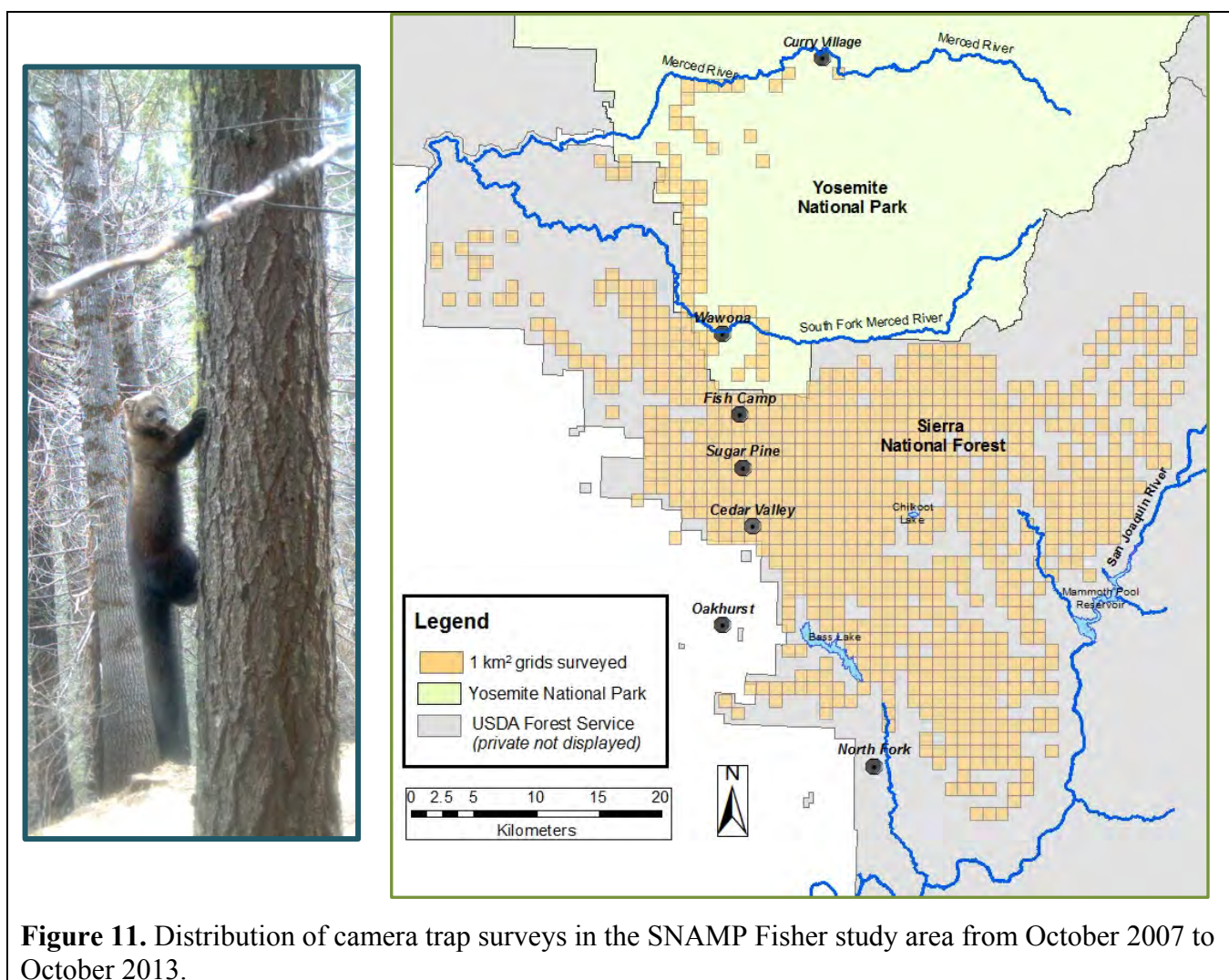
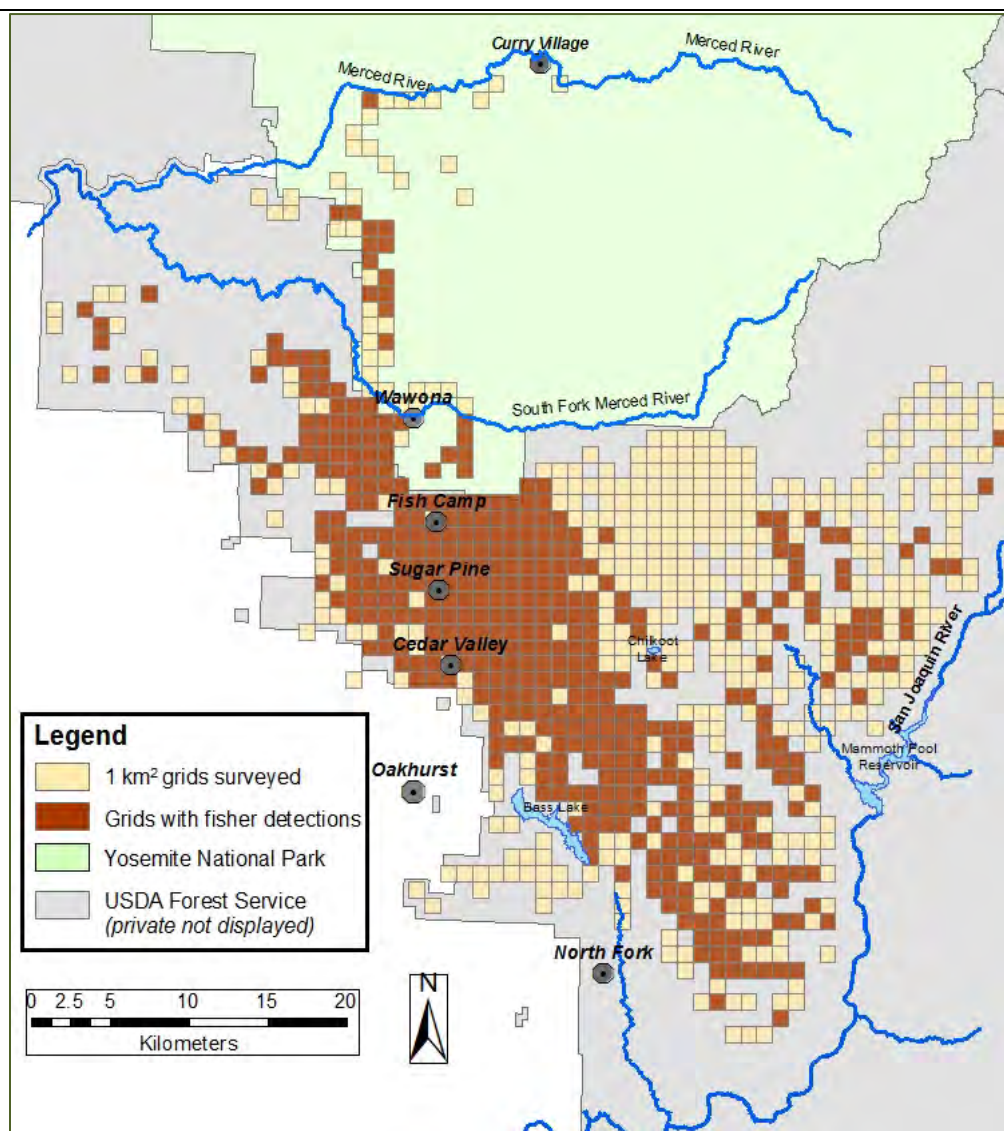


Figure 12. Distribution of fisher detections from camera trap surveys in the overall SNAMP Fisher study area in the period from October 2007 to October 2013.



Fisher activity was identified in 448 of the 905 unique grids surveyed (Fig. 12). We used information on the proportion of surveyed grids surveyed with fisher detections to estimate the elevation range for fishers in the overall study area (Fig. 13). Fishers were most common between 4500 and 6500 feet elevation (1372 and 1981 m elevation). Fisher detections were uncommon above 7500 feet (2286 m) elevation, but the pattern suggested fishers occasionally use private lands outside of the Sierra National Forest as low as 3000 feet (914 m) elevation (Fig. 13).

Camera trap effort was focused in the Key Watershed focal study area. The number of 1-km² grids surveyed ranged from 122 in 2007-08 and 133 in 2012-13 (Table 10). Across the larger overall SNAMP Fisher study area we surveyed 204 1-km² grids in 2012-13 and 409 grids in camera year

2010-11 (Table 10).

Naïve occupancy for all grids surveyed varied from ≈ 0.60 in 2008-09 to ≈ 0.40 in 2009-10 (Table 10). Occupancy for multi-year surveyed grids (corrected for probability of detection < 1.0) oscillated from ≈ 0.80 in 2007-08 to 0.62 in 2009-10 and then increased back to ≈ 0.80 in 2011-12 (Fig. 14).

In addition to basic naïve occupancy (presence/absence), we assessed fisher activity based on the number of occasions that fishers visited camera trap stations. Visit occasions were defined as distinct event periods when fishers activated the

motion sensors with at least a 15 minute break between successive visits. Review of images suggested this was an appropriate period of time separating distinct visit periods. We scored a total 4727 fisher visits to camera trap stations during the study (range 583 to 951; Table 11). Fisher visits ranged from 11.6 (2010-11) to 20.4 (2012-13) per 100 trap nights (Table 11). However, and in accordance with our finding of lower probability of detection for fishers during summer season compared to fall and winter

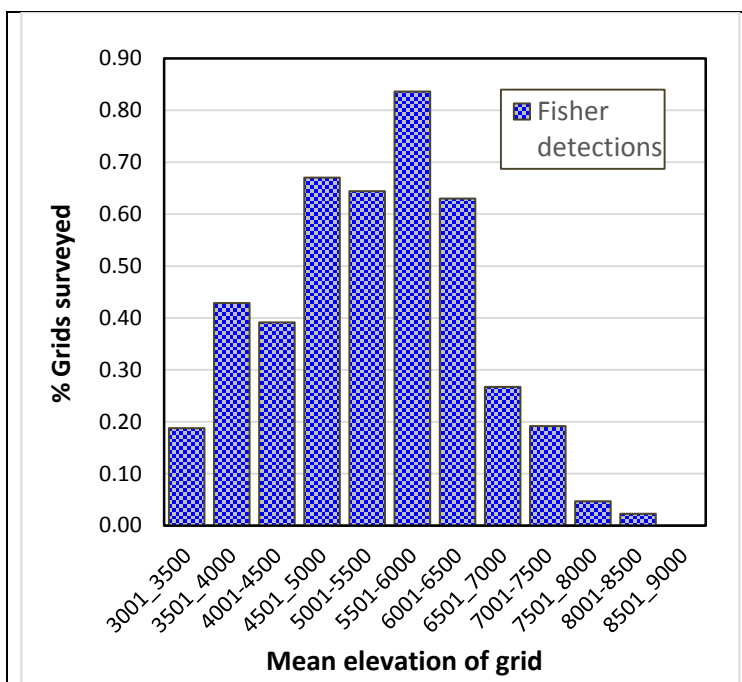


Figure 13. Elevation range of fishers in the SNAMP Fisher study area based on the proportion of grids surveyed with fisher detections in 500 foot (152 m) elevation bins.

Table 10. Number of 1km² grids surveyed with camera traps by camera survey year (Oct 15 to Oct 14).

Camera Year	Key Watersheds			Outside Key Watersheds			Entire study area		
	Grids	Fisher detected	Naïve occupancy	Grids	Fisher detected	Naïve occupancy	Grids	Fisher detected	Naïve occupancy
2007-08	122	71	0.582	98	41	0.418	220	112	0.509
2008-09	129	75	0.581	212	128	0.604	341	203	0.595
2009-10	127	75	0.591	275	100	0.364	402	175	0.435
2010-11	125	82	0.656	284	80	0.282	409	162	0.396
2011-12	128	98	0.766	226	104	0.460	354	202	0.571
2012-13	133	70	0.526	71	41	0.577	204	111	0.544

All years unique grids surveyed: $N = 905$

^a Some grids were surveyed twice during a camera year; those grids were counted once for this summary.

seasons (Popescu et al. 2014), fisher visits/100 camera survey days was very low during summer (3.6), and highest during winter (33.3; Table 12). Higher probability of detection during winter is due to presence of fewer species of squirrels and other prey during winter compared to in summer. For example, California ground squirrels (*Otospermophilus beecheyi*) and long-eared chipmunks (*Tamias quadrimaculatus*) enter into torpor (hibernation) during winter, and data on alligator lizards (*Elgaria multicarinata*) and other summer season prey are not available.

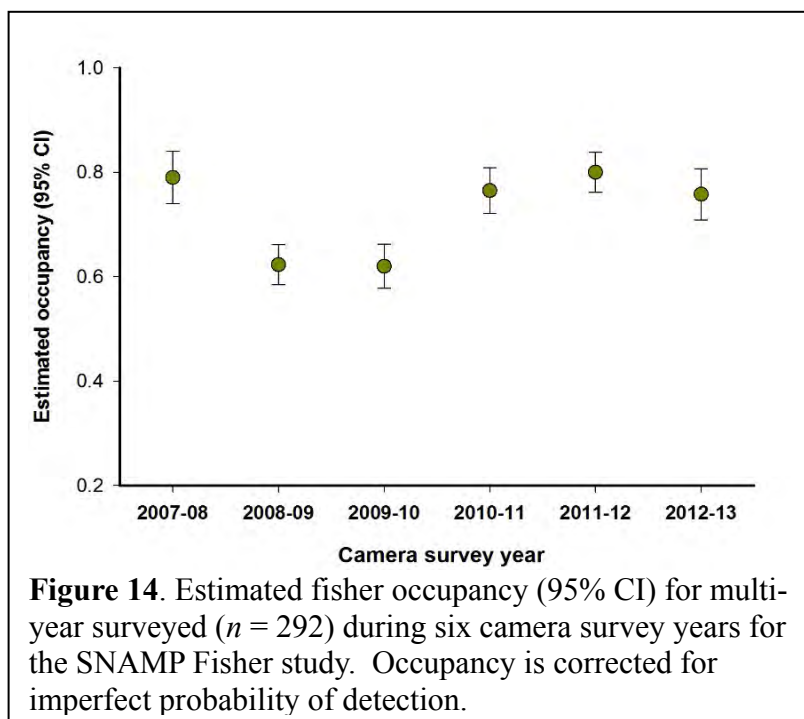


Table 11. Summary data on the number of camera days for all camera traps used to survey for fishers (effort), and the number of fisher visits during each camera survey year (~Oct 15 to Oct 14).

Camera year	Camera days (all cameras) ^a	Camera days (Fisher grids) ^b	Fisher visits ^c	Visits per 100 camera days ^c
2007-08	7914	4328	583	13.5
2008-09	10605	5550	794	14.3
2009-10	14955	5990	951	15.9
2010-11	16457	5614	649	11.6
2011-12	14059	7149	949	13.3
2012-13	7584	3926	801	20.4

^a Estimated days that camera traps were functioning and focused on the bait tree

^b Functional camera days for grids with fisher detections

^c Based on images sequences with fishers (fisher detections) that were separated by a minimum of 15 minutes.

^d Fisher visits divided by functional camera days for grids with fisher detections x 100

Table 12. Fisher visit to survey cameras and camera effort by season from October 25, 2007 to October 15, 2013. Seasons were based on the solar cycle: Fall - Sep 21 to Dec 20; Winter - Dec 21 to Mar 20; Spring - Mar 21 to Jun 20; Summer - Jun 21 to Sep 20.

Season	Fisher visits	Estimated camera days	Camera days (925 to 2135 m elev) ^a	Visits per 100 camera days	Visits per 100 camera days (fisher elevation)
Fall	1103	19027	7010	5.8	15.7
Winter	2543	22113	7629	11.5	33.3
Spring	733	12069	5032	6.1	14.6
Summer	348	18365	9597	1.9	3.6

^aCamera days within the typical elevation range for fishers in the SNAMP Study area [925 m (3000 ft) to 2135 m (7000 ft); see Fig. 9.



Illustration 17: Winter period and summer period fisher detections at camera trap survey stations

Fisher Denning and Reproduction

Denning period

Den cameras provided detailed information on the activities of 32 adult female fishers during six spring denning seasons. Based on information from the spatial clustering of aerial telemetry locations, ground-based telemetry, and den cameras, denning was initiated in the last week of March in most years (earliest date was March 22), and females typically ceased regular use of den trees in the first week of June (Table 13). The latest known regular use of a den tree was June 20 in spring 2012. It is likely that females continued to use trees as short term den/rest structures during summer when their dependent kits were trailing them, but we did not attempt to systematically identify those types of short duration use structures.

Table 13. Estimated dates for the initiation and end of denning by female fishers in the Sierra National Forest, California. Data are from March 2008 to June 2013.

Season	Mean start of denning ^a		Mean end of denning	
	N	Estimate	N	Estimate
2008	1	27-Mar-08	1	4-Jun-08
2009	12	27-Mar-09	9	6-Jun-09
2010	13	25-Mar-10	8	4-Jun-10
2011	8	1-Apr-11	5	5-Jun-11
2012	11	31-Mar-12	7	2-Jun-12
2013	12	27-Mar-13	7	1-Jun-13

^aEstimated from spatial clustering of sequential aerial radiotelemetry locations (Zhao et al 2012).

Table 14. Summary data on denning activities by female fishers determined from monitoring den trees with remote cameras. Data are from the Sierra National Forest, California from April 2008 to June 2012.

Spring	Female fishers	Den trees monitored	Monitor days ^a	Avg monitor days/female	Total detections ^b	Daily rates	
						Detection rate ^c	Up/Down movements ^d
2008	1	3	37	37.0	30	0.568	1.1
2009	13	45	449	34.5	366	0.601	1.2
2010	14	47	479	34.2	353	0.568	1.1
2011	9	28	260	28.9	264	0.725	1.1
2012	11	37	406	36.9	439	0.733	1.3
2013	12	40					

^a Excludes periods when females moved kits and the next maternal den tree had not yet been located.

^b All detections identified as a departure or return to the den tree, as well as events when fishers were at the base of den trees but not identified as departing or returning.

^c Proportion of days den trees were being monitored for which at least one detection was made by den cameras

^d Mean number of return and departures movements for days when fishers were detected by cameras.

In the spring 2008 den season SNAMP Fisher monitored a single female fisher, but in all other years we monitored at least nine individual females (Table 14). The mean number of dens used per female per season was 2.4 (range 1 to 5), and the mean number of cameras used to monitor each den structure was 3.1. On average each denning female was monitored with den cameras 34.3 days/season (range 28.9 to 37; Table 14), excluding days or periods when successive use maternal den trees were yet to be identified. Fifteen female fishers were monitored with den cameras in one den season, 11 were monitored in two seasons, three were monitored in three seasons, two were monitored in four seasons, and one was monitored in five denning seasons

Denning activity, litter size and weaning rates

Denning status was determined for 89 of 93 total denning opportunities for breeding-age (≥ 24 months) females in 6 denning seasons from 2008 to 2013 (Table 15). We were unable to adequately monitor 4 breeding-age females for determining denning status when radiocollars were shed ($n = 3$) or ceased functioning ($n = 1$) within the first 31 days of a denning season. The average date that females initiated denning behavior was March 28 (range March 22 to April 9). The average date that females ceased localizing to den trees was June 9 (range May 30 to June 22).

Seventy-six (85%) breeding-age female fishers either exhibited denning behavior ($n = 63$) or were determined to have denned and weaned at least 1 kit based on size of teats ($n = 13$; Table 15). Among 76 breeding-age females that initiated denning, 64 (84%) were identified as weaning kits. Overall, 72% of 89 known status, adequately monitored denning opportunities for breeding-age females produced at least one weaned kit (Table 15).

Eleven (17.5 %) of 63 cases of denning for females that were monitored during spring periods failed prior to kits being weaned (Table 16). Three of the 11 denning failures were females that initiated denning but ceased localizing to natal den trees 17, 35, and 41 days later, potentially related to the death of kits. Eight den failures were due to death of the denning female; seven deaths were by attacks by predators, and one was the result of a denning female either dying of internal bleeding induced by exposure to rodenticides, or from the combination of trauma from being struck by a vehicle on a highway and internal bleeding related to exposure to rodenticides. One of the seven females that died from predator attack was infected with canine distemper virus, which may have contributed to her vulnerability (Keller et al. 2012).

Table 15.--Summary of female fisher (≥ 2 years old) denning and weaning rates by age class and year on the Bass Lake Ranger District in the Sierra National Forest, California, 2008-2013.

Pop Year	No. Adult Females ^a	Monitored mid-Mar to May 31 ^b	Teats measured (Jul to Jan) ^c	Denning ^d	Proportion denned ^e	Unknown status ^f	Failed ^g	Died while denning	Weaned ^h	Proportion weaned ⁱ
Age class (Years)										
2	30	26	1	21	0.78	3	1	2	18	0.67
3-5	63	48	13	55	0.87		3	5	46	0.75
≥ 6	12	10	2	9	0.75			1	8	0.67
Year										
2008	11	2	9	9	0.82				9	0.82
2009	17	14	3	15	0.88		1	1	13	0.76
2010	17	15	1	14	0.88	1		3	11	0.69
2011	16	11	3	12	0.86	2	1	1	10	0.71
2012	17	17		14	0.82			3	11	0.65
2013	15	15		12	0.86	1	1		10	0.79

^a All females ≥ 24 months of age that were known in the population during the year. Includes females that were captured after the end of the denning season in mid June.

^b Number of females monitored by radio telemetry during all of part of the period before they died or had a dropped/failed collar after denning status had been determined.

^c Number of females the were not monitoring during the denning period, but were captured during July to January when teat measurements were taken and used to determined weaning status as described by Matthews et al. (2013).

^d Number of females that exhibited denning behavior, or that were determined to have weaned at least one kit based on teat measurements.

^e Number of denning females divided by the number of adult females minus the number of females of unknown status.

^f Number of females (≥ 2 years old) for which denning was unknown or suspected, but dropped or failed radiocollars prevented determination of denning status.

^g Number of females (≥ 2 years old) that exhibited denning behavior and ceased denning behavior prior to weaning.

^h Number of denning females that were known alive and exhibited denning behavior until after May 31.

ⁱ Number of females that denning to weaning divided by the number of adult females minus the number of females with unknown status.

Table 16.—Information on female fisher kit production for six spring denning seasons (March 21 to June 31) in the Sierra National Forest, California, October 2008 to June 2013.

Age class (Years)	Denning females ^a	Denning Females with kit counts ^b	Kits ^c	Litter size ^d	Denning female deaths ^e	Known kit deaths ^f	Denning Denned to Weaning ^g	Kits weaned ^h	Kits weaned per litter (fecundity)
2	19	15	21	1.40	1		14	19	1.27
3-5	32	26	40	1.60	3	7	22	34	1.32
≥6	8	7	12	1.71			7	12	1.71
Year									
2008	2	1	1				1	1	
2009	12	9	15	1.5	1		10	15	1.5
2010	13	11	20	1.8	3	7	7	13	1.9
2011	8	7	11	1.6	1		7	11	1.6
2012	14	11	16	1.5	3	2	10	14	1.4
2013	10	8	10	1.3			8	10	1.3

^a Number of females (≥2 years old) that exhibited denning behavior and were monitored by radiotelemetry, den cameras, or both. Excludes females whose reproductive status was not known and those that initiated denning behavior but ceased denning before May.

^b Number of denning females for which kit counts were determined by images from den cameras, den cavity video camera, or both.

^c Total number of kits counted.

^d Number of denning females with kit counts divided by the number of kits counted.

^e Number of denning females known to have died during the denning season while provisioning kits in den trees. Numbers of kits in litters were not known for all of the denning females that died.

^f Kits that were known present in den trees when the mother died, or those that were found dead inside den cavities. This estimate assumes that 5 orphan kits that were removed from den cavities would have perished if they had not been rescued.

^h Number of monitored denning female fishers exhibiting denning behavior that continued to weaning.

Six of eight deaths of denning females occurred when the locations of den trees were known and were being monitored. In one case den camera images included a bobcat with a dead kit in its mouth, and the partial carcass of the denning female was recovered nearby. In a second case the den structure was a large, unstable snag, and we did not attempt to climb the tree to determine litter size due to safety considerations. In each of the other four cases we climbed the den trees to assess litter size, and recover kits in accordance with California Department of Fish and Wildlife policy. A total five live kits were recovered from two of the den trees (litter size 2, 3), two deceased kits were found in a den cavity of the third tree, and we failed to find kits in the fourth tree. For this fourth tree, the lack of images of the female from den cameras suggested she had moved the litter to a different, unidentified den tree several days prior to her death.

The five orphan kits that were rescued were raised in captivity by a local wildlife rehabilitation organization licensed by the California Department of Fish and Wildlife, and under the care and supervision of a professional zoo veterinarian. One of the orphan kits died in captivity by urinary tract blockage attributed to a parasitic nematode, whereas the other 4 survived captive rearing. Two kits from one litter were released within their mother's home range, and the two kits from the second litter were released into an area with suitable fisher habitat abutting the south margin of the study site.

We used a combination of images from den cameras ($n = 43$) and den cavity investigations with a video camera ($n = 4$) to determine litter size for 48 of 59 denning females that were monitored (Table 16). A total 73 kits were known produced, and average litter size was 1.5 (Table 16). After accounting for known mortalities of denning females, we estimated that 64 of the 73 kits produced were weaned from den trees, whereas seven kits died or would have died had they not been rescued (Table 16).

Denning structures

We identified 125 unique structures used as natal or maternal dens, including 54 black oak, 41 incense cedar, 19 white fir, 10 sugar pine or ponderosa pine, and one canyon oak (*Quercus chrysolepis*) (Table 17).

Repeat use of den trees was not uncommon. Sixteen individual den trees were used more than once; 15 trees were used in two years, and one tree was used in four different den seasons. In all but two cases of repeat den tree use the same individual reused one or several den trees between successive years. In two cases a female used a den that had been used by a different female in a previous year. Successive dens of females that used more than 1 den structure were located an average of 413 m apart ($n = 52$, range 75-1398),

and the average total distance moved between successive dens was 693 m ($n = 31$, range 75-1687). Also, the distance between the natal den tree and the first maternal den tree averaged 419 m, whereas successive use maternal den trees were in closer proximity (mean=287 m, $t_{69} = 1.75$, $P=0.04$), potentially because older kits are larger in size and mass and therefore more difficult for the female to carry.

Fifty-six percent of the unique individual trees used for denning in the SNAMP area were live trees ($n = 70$), whereas 44% ($n = 55$) were snags (Table 17). Black oak was the most common live tree used for denning, but a high percent of incense cedar were also selected by female fishers (Table 17). Among snags used as denning structures, black oak and incense cedar were both commonly used, whereas white fir and pines (sugar pine or ponderosa pine) were less common as snag-type den trees (Table 17). Overall, black oaks and incense cedar were the two most common tree species used for denning (Table 17).

Table 17. Information on the number of times (denning events) different species of trees were used for denning by female fishers in the SNAMP Fisher study. Includes counts for live trees, snags, and both types of denning structures.

Tree type, Species	Denning events ^a	Percent within group	Unique structures ^b	Repeat use structures ^c
<i>Live trees</i>				
Black oak	34	43	31	3
Incense cedar	25	32	19	4
White fir	14	18	14	
Sugar pine	3	4	3	
Ponderosa pine	2	3	2	
Canyon oak	1	1	1	
<i>Snags</i>				
Black oak	27	42	23	
Incense cedar	27	42	22	4
White fir	5	8	5	5
Pine species ^d	5	8	5	
<i>Live tree or snag</i>				
Black oak	61	43	54	3
Incense cedar	52	36	41	8
White fir	19	13	19	5
Pine species	10	7	10	
Canyon oak	1	1	1	
Total den structures	143		125	16

^a Count of all known denning events for each species of tree.

^b Count of individual trees; those used in multiple seasons counted once.

^c Number of individual trees used \geq two times for denning; one live cedar tree was used by the same female in four successive denning seasons, but all other repeat use trees were known used in two den seasons only.

^d Pine snags could not always be identified as sugar pine or ponderosa

Habitat characteristics of den structures

Mean diameter at breast height (DBH) of black oak denning structures was smaller than that for other tree species used (Table 18). Mean heights of live trees were taller than snags of the same species (Fig. 15), reflecting that many of the snags used for denning were at advanced stages of decay.

Table 18. Basic information on the size (DBH) and height of trees (live or snag) used as denning structures by female fishers in the SNAMP Fisher study from March 2008 to June 2013.

Tree species	Live trees			Snags or dead trees		
	<i>n</i>	Mean DBH (cm)	Mean height (m) ^a	<i>n</i>	Mean DBH (cm)	Mean height (m)
Black oak	30	74.2	21.7	5	69.5	8.8
Incense cedar	18	127.2	32.5	22	105.1	16.4
White fir	14	110.8	33.9	22	103.7	27.4
Pines	5	112.8	37.4	5	109.6	27.6

^aData on mean tree height are for the subset of den trees for which detailed data on habitat measurements were completed (*n* = 84).

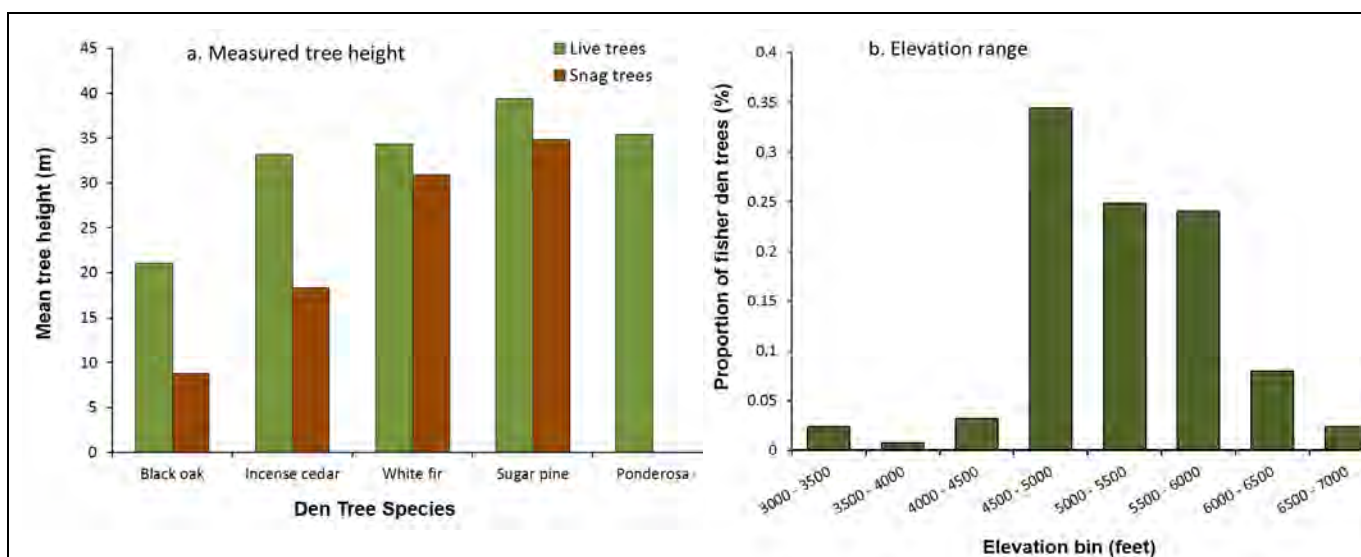


Figure 15. Summary information on the (a) mean height of denning structures (unique trees), and the (b) elevation range for fisher den trees for the SNAMP Fisher Study during 2008 to 2013 (6 denning seasons).

The majority of denning structures used by fishers in the SNAMP Fisher study area were in the elevation range from 4500 (1371 m) to 6000 feet (1829 m; 83%, *n* = 104; Fig. 15b). Additional information obtained from circular habitat plots assessments included indications of high canopy cover, limited herbaceous cover, and relatively low shrub cover near most den trees (Table 19).

Concealment cover was 64% low ground cover, 46% high ground cover, and 38% and 36% low shrub and high shrub cover, respectively. On average, belt transects within the circular habitat plots around den trees included an average of 6.5 down logs (coarse woody debris, CWD; logs/branches with a minimum large end diameter of 15 cm, ≥ 1 m total length). Many denning structures were on steep slopes (Table 19) but there was no obvious preference for aspect (Fig. 16).

Table 19. Basic habitat attributes around fisher den trees for the SNAMP Fisher Study area in spring 2008 to spring 2012.

Attribute ^a	Mean	Range
Canopy cover	72%	30-94%
Shrub cover	19%	0-82.5%
Herbaceous cover	6%	0-29%
Prevailing slope	37%	3-75%

^a Habitat attributes are from circular plots (18 m radius) centered on fisher den trees ($n = 82$). Habitat data were not available for other confirmed den trees.

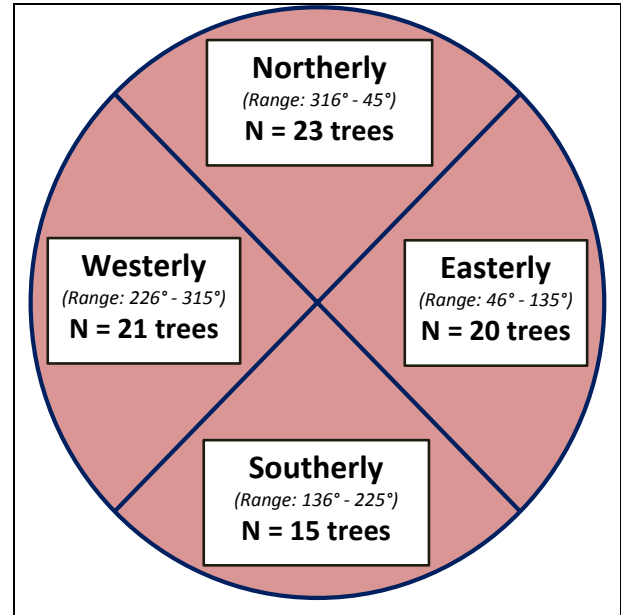


Figure 16. Aspect orientation of fisher den structures on the SNAMP Fisher Study. Data are for 79 den trees.



Illustration 10: Black oak used for denning in spring 2011. Image of tree from a distance (left), and view of the forked branching higher up the bole of the denning structure (right).

Activity patterns of denning females

Additional insight on denning activities by adult female fishers was provided by analyses of den camera images. Adult females were detected by den cameras at known active dens an average of 0.64 times/day (range 0.57 to 0.73) (Table 14) and the mean number of detections of up and down movements ranged from 1.1 to 1.3 per day (Table 14), indicating that fishers do not typically leave and return to den trees multiple times a day. In addition to information on male visits to den trees (we obtained image sequences of eight mating, or copulation events at the base of den trees), den cameras identified three occasions when a female fisher briefly returned to a den tree at least one day after she had already moved kits to another tree nearby (Table 20). On eight occasions den cameras detected other female fishers (non-collared or different collared fisher) at den trees of female fishers (Table 20).

Table 20. Summary data on denning activities by female fishers determined from monitoring den trees with remote cameras. Data are from the Sierra National Forest, California from April 2008 to June 2012.

Spr g	Departin g	Returnin g	Base tree ^a	Bringing food to tree ^b	Kit move	At tree after kits moved	Other female at tree
2008	15	9	6		1		
2009	118	198	35		12	1	
2010	133	163	37	1	20		2
2011	120	127	10		14	1	6
2012	178	234	11	8	25	1	
2013 ^c					<i>min 11</i>		
^a Detections at base of tree, or on the tree for which directionality or activity was uncertain							
^b Detections when the female was carrying objects as they returned and ascended the den tree							
^c General information only available for 2013.							

Information from the 83 occasions when females were detected moving kits was used to estimate fecundity. A total of 1295 detections were identified as female fishers departing from, or returning to the den tree, whereas there were 99 detections of females at or near the base of den trees that could not be unequivocally classified except as active outside the den cavity (Table 20). We were able to identify 316 image sequences consistent with either continuous den attendance, or continuous time away from the den when denning females were likely foraging. Den attendance bouts were shortest late in the den season and longest in the middle of the den season (Table 21). Forage bouts away from den trees were shortest early in the den season and approximately similar in duration thereafter (Table 21).

Table 21. Information on den attendance and away from den tree foraging excursions, developed from analyses of data from cameras used to monitor fisher den trees during five denning seasons. Data are from the Sierra National Forest, California from April 2008 to June 2012.

Den attendance bouts (minutes)					Forage away bouts (minutes)			
Season ^a	Cases	Mean	Min ^b	Max ^c	Cases	Mean	Min	Max
Early	38	371.0	4.4	1072.5	64	235.1	34	746.1
Middle	43	535.6	2.3	996.2	53	431.8	29.3	811.4
Late	68	323.3	6.0	1049.4	50	405.5	22.8	807.8
Overall	149	396.7	2.3	1072.5	167	348.6	22.8	811.4

^a Seasons were Early (March 26 to April 20), Middle (April 21 to May 15), and Late, (May 16 to June 11), identified by dividing the overall den season into three 25 day periods from late March to mid-June.
^b shortest duration bout.
^c longest duration bout.

Fisher Survival

Sixty-six (60%) of the 110 individual fishers radiocollared during the study were known to have died, including 32 females and 34 males (Table 22). Excluding population year 2007-08, an average of 10.5 radiocollared fishers perished each population year (Fig. 17). The mean number of deaths by gender for population year 2008-09 through population year 2013-14 was 5.3 for females and 5.2 for males.

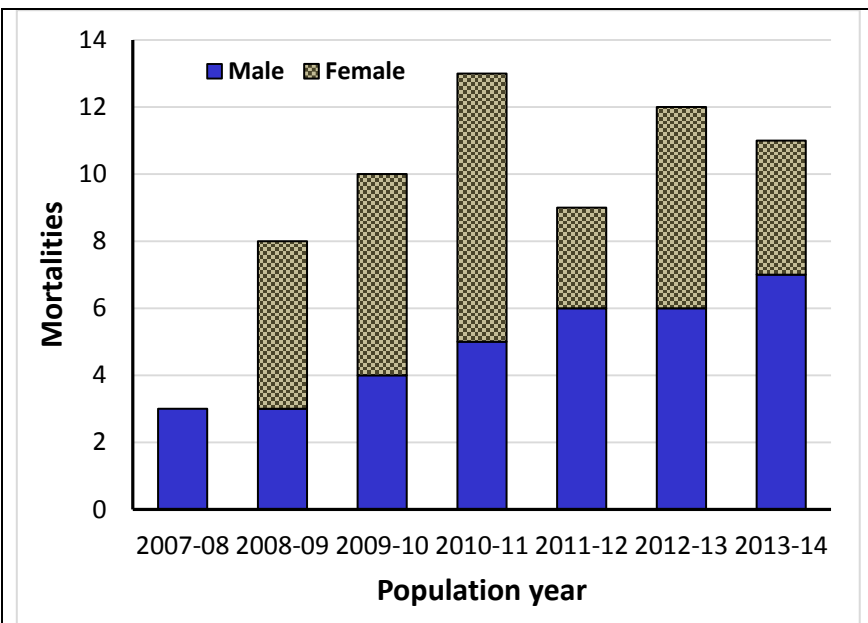


Figure 17. Number of radiocollared female and male fishers known to have died during the SNAMP Fisher Study.

Fisher survival with population data combined into 2-year periods was generally higher for adult and juvenile fishers than for subadults (Table 23). Ninety-five percent confidence intervals overlapped for females and males in all two-year period with the possible exception of subadults in year group 3. Two-year survival rates among females ranged from a low of 47% to a high of 89% for

Table 22. Review of all known deaths of radiocollared fishers in seven population years (Apr 1 to Mar 31), summarized by gender, and cause-specific mortality from necropsy examinations by pathologists at the UC Davis School of Veterinary Medicine (Davis, CA). Necropsy reports were not available for 16 fishers.

Year, Gender	Predation ^a	Disease ^b	Starvation- related injury, septicemia ^c	Road kill	Rodenticide toxicosis	Indeterminate, unknown ^d	Pending necropsy
2007-08							
Female							
Male	1	1		1			
2008-09							
Female	4		1				
Male	1	1		1			
2009-10							
Female	5		1				
Male		3			1		
2010-11							
Female	5		1		1	1	
Male	4					1	
2011-12							
Female	3						
Male	5		1				
2012-13							
Female	3						3
Male	1			1	1	1	2
2013 ^e							
Female							4
Male							7
All years							
Female	20		3		1	1	7
Male	12	5	1	3	2	2	9

^a One female death by predation in 2009-10 may have been related to the animal being weakened/sick from CDV when it encountered a coyote (*Canis latrans*); further discussed by Keller et al. (2012).

^b Three disease deaths were associated with canine-distemper virus, one was considered by Toxoplasmosis, and one was due to pleruritus+pneumonia.

^c Most deaths in this category were associated with prior injury that contributed to starvation and septicemia.

^d Necropsies were completed, but cause of death could not be determined.

^e Includes deaths of two male fishers that died January 1, 2014 and March 31, 2014. Although this was after the end of SNAMP Fisher, both of the animals were radiocollared as part of SNAMP.

Table 23. Estimates of survival ($s(t)$), for radiocollared fishers using population data combined for analysis into a series of five 2-year groups beginning in population year 2 (2008-09) and ending in population year 7 (2013-14), and for all years of data combined. Survival was assessed using Kaplan-Meier staggered entry analyses. Population years were from April 1 to March 31, and ages were defined as juvenile, < 12 months, subadults, 12 to 23 months, and adults, ≥ 24 months.

Year group, Gender	Juveniles		Subadults		Adults	
	$s(t)$	95% CI	$s(t)$	95% CI	$s(t)$	95% CI
<i>2008-09, 2009-10</i>						
Female	0.80	0.58-1.02	0.47	0.28-0.66	0.81	0.66-0.96
Male	0.83	0.50-1.17	0.40	0.10-0.70	0.73	0.52-0.95
<i>2009-10, 2010-11</i>						
Female	0.80	0.59-1.01	0.67	0.42-0.92	0.70	0.54-0.86
Male	0.60	0.30-0.90	0.43	0.06-0.79	0.71	0.52-0.91
<i>2010-11, 2011-12</i>						
Female	0.74	0.54-0.94	0.89	0.71-1.07	0.69	0.53-0.86
Male	0.67	0.42-0.92	0.50	0.29-0.71	0.56	0.37-0.75
<i>2011-12, 2012-13</i>						
Female	0.80	0.55-1.05	0.73	0.52-0.94	0.86	0.71-1.00
Male	0.83	0.56-1.11	0.92	0.77-1.06	0.61	0.44-0.77
<i>2012-13, 2013-14^a</i>						
Female			0.73	0.48-0.98	0.74	0.56-0.93
Male			0.75	0.33-1.17	0.66	0.49-0.83
<i>All Years; Dec 07-Mar-14</i>						
Female	0.75	0.60-0.89	0.71	0.57-0.84	0.74	0.64-0.83
Male	0.60	0.42-0.78	0.57	0.40-0.74	0.64	0.54-0.75

^a Insufficient data for estimating survival for juveniles in this Year group.

subadult females (Table 23). Two year survival for juvenile females was always $\geq 74\%$, whereas among adult females it ranged from a low of 0.69% to a high of 0.86 (Table 23). Fisher survival for all years combined was highest for juvenile females and lowest for subadult males (Table 23). Also, although not significantly different, survival was consistently higher for females compared to males (all age classes; Table 23). In general, survival among females is more important for understanding the status of this fisher population than male survival, particularly because there were a good number of males in the population in all years of the study (Fig. 10).

Causes of Mortality

Necropsy reports have been completed for 50 of the 66 radiocollared fishers that died during

the SNAMP Fisher study. Assignment of cause-specific mortality was possible for 47 of the 50 animals with necropsy reports (94%). Three necropsies reports were indeterminate with regards cause of death for the fisher (Table 22). To date, a known cause of death has been determined for 71% of the 66 mortalities.

Among known-cause mortalities predation was the primary cause of death, accounting for 68% of 47 known-cause deaths (Fig. 18). Deaths by disease, injury-related starvation or septicemia, and human-linked factors combined to account for 32% of known-cause mortalities (Fig. 18).

Predation accounted for nearly twice as many known cause deaths for females (43%) than for males (26%), whereas all of the disease and roadkill deaths were males (Fig. 18). Serological testing of blood samples collected at captures revealed low levels of exposure to canine distemper virus in the study population (Gabriel 2013). However, in spring 2009 a relatively small scale epizootic of CDV occurred in the study population, which contributed to the deaths of four fishers; three by direct infection, and one that was killed by a coyote attack, but was likely in a weakened state due to presence of CDV infection (Table 22, Fig. 18; Keller et al. 2012).

In Spring 2009 the SNAMP Fisher team recovered the first fisher known to have died by toxicosis after exposure to rodenticides. In total, three fishers were known to have died after exposure to rodenticides as of June 2014, including two males and one female. The discovery of death associated with rodenticides led to two peer-reviewed

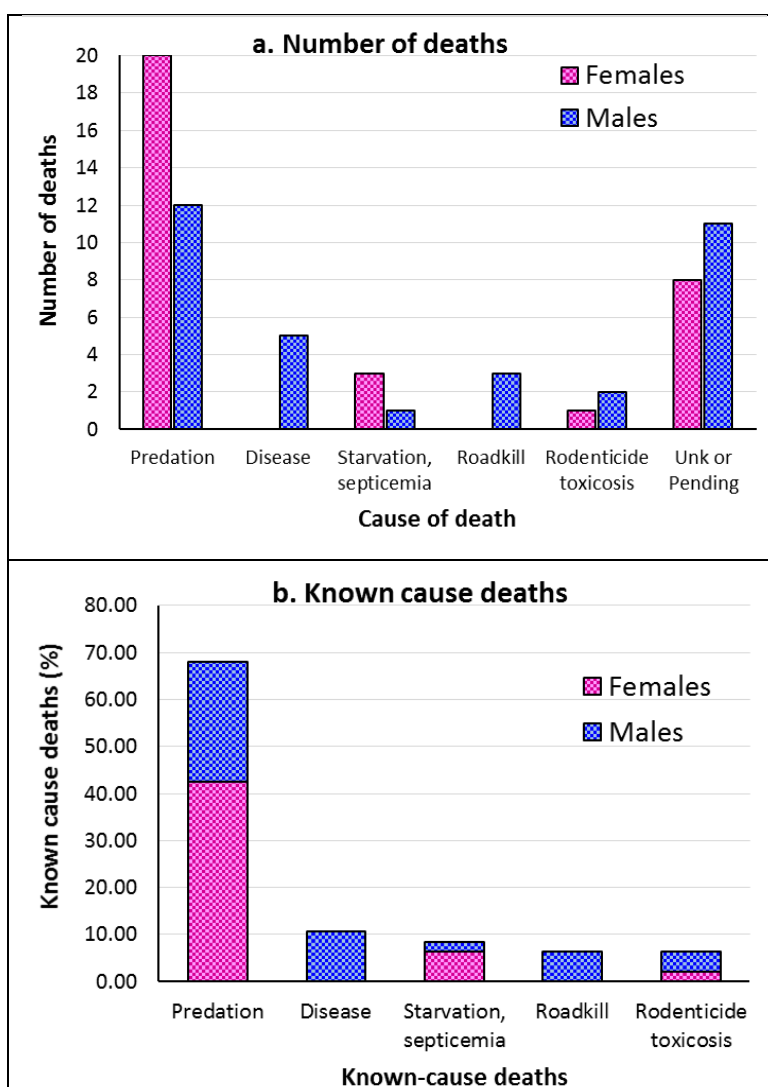


Figure 18. Summary of necropsy-determined causes of death for radiocollared female and male fishers (a), and percent of known cause mortalities for female and male fishers on the SNAMP Fisher Project from Dec 2007 to Dec 2013.

papers. One detailed issues with anticoagulant rodenticides on public lands (Gabriel et al. 2012) and a second paper revealed that female fishers with larger numbers of marijuana grow sites within their home ranges experience reduced survival (Thompson et al. 2013).



Illustration 11: Remains of a female fisher killed by a predator (left), and a male fisher that was determined to have died by infectious disease (right).

Population Growth Rates

Empirically developed estimates of key demographic parameters needed to estimate a deterministic growth rate for the population (λ) were developed during the study (Tables 22, 23). Estimates for λ were below 1.0 (population decline) in two 2-year groups, equal to 1.0 in one 2-year group (stable), and slightly positive in two 2-year groups (increasing population) (Table 24). The All Years λ was 0.90, which was suggestive of population decline, however, the range for all results overlapped 1.0.

Table 24. Demographic parameters and deterministic population growth rates (range) for five two-year groups^a, and for population data for all years of the study combined (All years).

Parameter, Age class	Year group 1	Year group 2	Year group 3	Year group 4	Year group 5	All Years
<i>Weaning reproduction</i>						
Young adult	0.67	0.89	1.00	0.83	0.70	0.68
Adult	0.75	0.67	0.83	0.82	0.87	0.74
<i>Weaning litter size</i>						
Young adult	1.27	1.57	1.50	1.20	1.20	1.19
Adult	1.31	1.31	1.60	1.55	1.40	1.45
<i>Weaning fecundity (b_i)^b</i>						
Young adult	0.42	0.70	0.75	0.50	0.42	0.41
Adult	0.49	0.44	0.67	0.64	0.61	0.53
<i>Survival (P_i)</i>						
Juvenile	0.76	0.80	0.74	0.80	1.00	0.75
Subadult	0.47	0.67	0.89	0.73	0.73	0.71
Adult	0.81	0.70	0.69	0.86	0.74	0.73
<i>Fertility (b_i)P_i</i>						
Young adult	0.20	0.47	0.67	0.36	0.31	0.29
Adult	0.40	0.30	0.46	0.55	0.45	0.39
	0.87	0.88	1.00	1.04	1.03	0.90
<i>Leslie Matrix λ^c</i>	(0.65-1.08)	(0.63-1.12)	(0.77-1.22)	(0.81-1.26)	(0.77-1.22)	(0.71-1.12)

^a Two-year groups were 2008-09 and 2009-10 (1), 2009-10 and 2010-11 (2), 2010-11 and 2011-12 (3), 2011-12 and 2012-13 (4), and 2012-13 and 2013-14 (5).

^b Fecundity is the number of female offspring produced, calculated as weaning reproduction*weaning litter size*0.5 (assumes equal sex ratio at birth)

^c The range for λ was based on the 95% confidence intervals for the survival rates for the five two-year groups (Table 23). The range for λ for the All years data was based on the 95% CIs for the means for weaning reproductive rate and litter size, and for the 95% CIs for age-specific survival (Table 23).

Population Size and Density

Population size was estimated for the middle four population years of the six year study. In population year 1 (2007-08) we had only a small number of fishers radiocollared during the last few months of that period (Tables 9, Fig. 10) and camera trap images for the entire population year 2012-2013 were not available due to the conclusion of SNAMP

Table 25.—Summary data on camera and live trap activities within 4 fall-winter camera survey years (October 16 to March 15) in the Bass Lake District, Sierra National Forest Study area, October 2008 to March 2012.

Subregion, Year	Camera surveys		Live traps		Estimated area surveyed (km ²) ^a
	Grids	Nights	Grids	Nights	
<i>Subregion 1. Nelder Grove, Sugar Pine, Miami Mountain</i>					
2008-09	147	4462	121	1027	223.2
2009-10	160	5817	161	875	307.2
2010-11	132	4995	72	411	214.3
2011-12	141	5245	147	1016	224.6
<i>Subregion 2. Central Camp, Whisky, Grizzly, Jackass</i>					
2008-09	48	1289	17	158	267.6
2009-10	12	349	56	272	248.0
2010-11	20	1048	47	237	244.2
2011-12	65	2522	80	316	305.5
<i>Subregion 3. Chowchilla Mountain, Rush Creek, Sweetwater</i>					
2008-09	16	400	25	144	128.8
2009-10	2	79	39	252	111.8
2010-11	1	33	22	124	136.2
2011-12	14	513	32	149	132.8
^a Based on a 1300 m buffer applied to polygons encompassing grids surveyed by cameras and grids with live trap captures.					

Fisher field work. During the central four year period we captured and radiocollared 101 individual fishers (57 females and 44 males) on 258 occasions during 9732 trap-nights between December 2007 and March 2012. Resighting efforts, both by camera and live traps, varied by subregion and, to a lesser extent, year (Table 25). Camera traps accounted for 86% of 1421 total radio-marked fisher detections, with live trap recaptures providing 201 sightings.

Mean overall abundance across all subregions ranged from 48.2 individuals in Year 2 to 61.8 individuals in Year 4. Variation was at least partly related to differences in area surveyed among years (Table 26). Estimates of areas sampled were generally consistent within subregions among years (Table 26). The increase in area surveyed in Subregion 1 in fall-winter 2009-10 was due to a program that extended camera surveys north into the Yosemite South region of Yosemite National Park (Fig. 3) in winter 2010. In fall-winter 2011-12 study effort was expanded in the Grizzly and Jackass subregion when non-collared fishers were detected in areas that had not been surveyed previously. Mean annual population density for the three subregions ranged from 0.072 to 0.097 fishers/km² (Fig. 19).

Subregion 1 had consistently high average densities (0.073-0.125 individuals/km²), with an increasing trend across the last three years of the period (Table 26, Fig. 19). Subregion 3 had initial low density (0.056 ± 0.005 individuals/km²), but gradually increased by the end of the period (0.106 ± 0.005 individuals/km²). Subregion 2 showed no particular trend, and average densities varied across seasons between 0.066 (fall-winter 2009-10) and 0.092 individuals/km² (fall-winter 2010-11). Temporally, mean population density was lowest in fall-winter 2009-10 at 0.075 ± SE 0.006 individuals/km²,

and increased thereafter to a high of 0.097 ± SE 0.008 in fall-winter 2011-12 (Fig. 19). Mean

Table 26.—Mark-resight estimates of population size for three subregions in 4 Fall-Winter survey years (October 16 to March 15) in the Bass Lake District, Sierra National Forest, October 2008 to March 2012.

Subregion, Year	<i>n</i>	95% C.I.	Density ^a	Density range ^b
<i>Subregion 1: Nelder Grove, Sugar Pine, Miami Mtn</i>				
2008-09	27.9	23.6-32.2	0.125	0.106-0.144
2009-10	22.3	19.0-25.6	0.073	0.062-0.083
2010-11	19.1	16.3-22.0	0.089	0.076-0.103
2011-12	23.2	20.2-26.2	0.103	0.090-0.117
<i>Subregion 2: Central Camp, Whisky Ridge, Grizzly, Jackass</i>				
2008-09	18.8	10.5-21.2	0.070	0.044-0.097
2009-10	16.3	10.4-21.4	0.066	0.037-0.094
2010-11	22.5	15.4-24.5	0.092	0.066-0.118
2011-12	24.6	17.8-26.5	0.080	0.062-0.099
<i>Subregion 3: Chowchilla Mtn, Rush Creek, Sweetwater</i>				
2008-09	7.2	5.8-8.6	0.056	0.045-0.067
2009-10	9.7	8.7-10.6	0.086	0.078-0.095
2010-11	10.0	8.8-11.3	0.074	0.065-0.083
2011-12	14.0	12.8-15.3	0.106	0.096-0.115

^a Population size (*n*) divided by the estimated sample area for the subregion in the Fall-Winter camera year, included in Table 1.
^b Calculated based on the lower and upper values of the 95% C.I., divided by the estimate of the sampled area provided in Table 1.
^c Sum of the year- and subregion-specific estimates of population size from the mark-resight analyses.

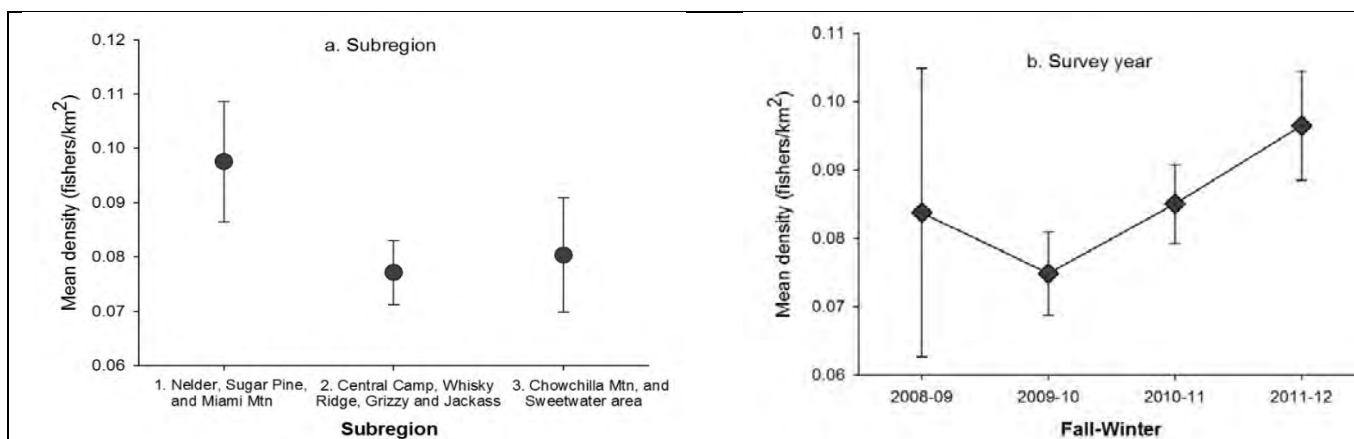


Figure 19. Mean density (± SE) of fishers in three subregions during 4 fall-winter camera survey years (a), and change in mean density (± SE) of fishers during 4 fall-winter camera survey years (b).

population density was consistently high in the last 2 years of the study across all subregions (0.089 – 0.106 individuals/km²).

Dispersal Behavior and Movements

The combination of field data and genetic data allowed for the *possibility* of assessing dispersal for 33 female and 25 male fishers that were captured as juveniles ($n = 53$), or young subadults ($n = 8$; ≤ 18 months old) (Table 27). Fifteen of those fishers (25.8%) either died, disappeared, or were caught too late in the year to define a juvenile home range (Table 27). Dispersal was assessed for 43 (74%) of the 58 animals, based on identification of likely natal areas from either field data or genetic-based maternity assignments (Table 27).

Considering data for dispersal using either field or genetic-based natal area determination and based on Euclidean distances, male fishers tended to disperse longer distances than females, but the difference was not significant (Table 28). The longest Euclidean distance dispersal for a female fisher was 24.53 km, compared to 36.17 km for a male fisher, however the large range of dispersal distances for both sexes precludes precise statistical comparison.

Euclidean dispersal movements often originated from within the Key Watershed focal study area, but other Subregions of the study area produced dispersing animals as well (Fig. 20). There was no clear patterning with regards directionality of dispersal, except perhaps the general northwest to southeasterly orientation associated with the Sierra Nevada range (Fig. 20).

One male fisher immigrated into the SNAMP Fisher Study area in the Bass Lake Ranger District from south of Shaver Lake within the High Sierra District. This fisher, KRFP ID M38 (SNAMP ID M47), was originally captured and marked with a PIT tag on the Kings River Fisher Project in December 2010. M38 was recaptured by the KRFP researchers in the KRFP study area in February 2012, when he was released without a radiocollar due to an abrasion from his original radiocollar. M38 was captured 13 months later in March 2013 within the SNAMP study area. Although his Euclidean distance-based dispersal track was estimated at ≈ 36 km, it is more likely that his dispersal track was more circuitous, and in the range of 67-69 km (Fig. 21).

Dispersal movements predicted by Least Cost Movement (LCP) analyses over landscape features considered restrictive to fishers produced longer mean dispersal distances than Euclidean paths (Table 29, Fig. 22). Nevertheless, and in accordance with data from Euclidean distances, there was no evidence for a significant gender-bias in LCP predicted dispersal tracks (Table 29).

Table 27. Review of information on juvenile or subadult fishers captured on the SNAMP Fisher study for which dispersal assessments were possible from field data, maternal assignments from genetic analyses, or from either source.

Maternal year, Gender	<i>n</i>	Dispersal not assessed ^a			Dispersal assessed			
		Died	Missing, disappear	Late capture	Field ^b	Genetics ^c	Both ^d	Total ^e
<i>2007</i>								
Female	3	1			2	2	2	2
Male	3	1			2	1	1	2
<i>2008</i>								
Female	8	2			4	5	3	6
Male	4			1	3	3	3	3
<i>2009</i>								
Female	7	1	1		5	3	3	5
Male	4	1			3	3	3	3
<i>2010</i>								
Female	6	1			5	4	4	5
Male	8	1			5	6	4	7
<i>2011</i>								
Female	7	1			4	6	4	6
Male	4			1	3	3	3	3
<i>2012</i>								
Female	2	1		1				
Male	2			1	1			1
<i>All years</i>								
Female	33	7	1	1	20	20	16	24
Male	25	3		3	17	16	14	19

^a Dispersal was not assessed if the animals died before <18 months old, when they were missing and not recaptured, or if they were captured after mid-January (<10 months old).

^b Animals for which home ranges allowed identification of likely natal areas (juvenile home ranges), as well as post dispersal home ranges as subadults or adults.

^c Animals for which maternal assignments were made using DNA analyses; natal areas were based on maternal home ranges.

^d Animals for which dispersal could be assessed using both field data (juvenile home ranges) and maternal assignments from genetic analyses.

^e Number of juveniles/young subadults for which dispersal could be assessed using either field-based home range data, or genetic-based maternal assignments.

Table 28. Estimates of mean Euclidean distances moved by dispersing fishers ≤ 18 months old on the SNAMP Fisher study. Dispersal was estimated by (1) distance between centroids for juvenile home ranges and subadult or adult home ranges, (2) distance between centroids for maternal home ranges (based on genetic-based maternity assignments) and adult or last known home ranges, or (3) distance between either juvenile home range centroids (fishers without maternity assignments) or maternal home range centroids and adult or last known home ranges.

Dispersal, Gender	<i>n</i>	Mean distance (SE)	Range	<i>t</i> -test contrasts ^a
<i>1. Juvenile to adult home range (field data)</i>				
Female	20	4.89 (1.36)	0.24-22.26	<i>t</i> ₃₅ = 1.35, <i>P</i> = 0.19
Male	17	8.48 (2.39)	0.94-36.17	
<i>2. Maternal to adult home range (genetics)</i>				
Female	20	5.00 (1.21)	0.46-24.53	<i>t</i> ₃₄ = 1.32, <i>P</i> = 0.20
Male	16	7.44 (1.41)	1.82-21.20	
<i>3. Juvenile or Maternal to adult home range (combined field and genetics)</i>				
Female	24	5.76 (1.26)	0.52-24.53	<i>t</i> ₄₁ = 1.67, <i>P</i> = 0.10
Male	19	9.81 (2.22)	0.94-36.17	

^a Unequal variance *t*-tests.

^a Unequal variance *t*-tests.

Table 29. Mean Least Cost Movement paths (LCP) developed to evaluate dispersal by fishers ≤ 18 months old in the SNAMP Fisher Study area. LCP tracks were estimated for (1) dispersal between centroids for juvenile home ranges and subadult or adult home ranges, (2) for dispersal between centroids for maternal home ranges (based on genetic-based maternity assignments) and adult or last known home ranges, and for (3) dispersal between either juvenile home range centroids (fishers without maternity assignments) or maternal home range centroids and adult or last known home ranges.

Dispersal, Gender	<i>N</i>	Mean Least Cost path	Range	<i>t</i> -test contrasts ^a
<i>1. Juvenile to adult home range</i>				
Female	20	7.53 (2.39)	0.47-44.09	<i>t</i> ₃₅ = 0.90, <i>P</i> = 0.38
Male	17	11.63 (4.11)	1.03-69.82	
<i>2. Maternal to Adult home range</i>				
Female	20	6.95 (1.62)	0.47-34.06	<i>t</i> ₃₄ = 1.07, <i>P</i> = 0.29
Male	16	9.52 (1.77)	1.85-26.15	
<i>3. Juvenile or Maternal to Adult home range</i>				
Female	24	8.76 (2.11)	0.47-44.09	<i>t</i> ₄₁ = 1.16, <i>P</i> = 0.25
Male	19	13.48 (3.71)	1.03-69.82	

^a Unequal variance *t*-tests.

^a Unequal variance *t*-tests.

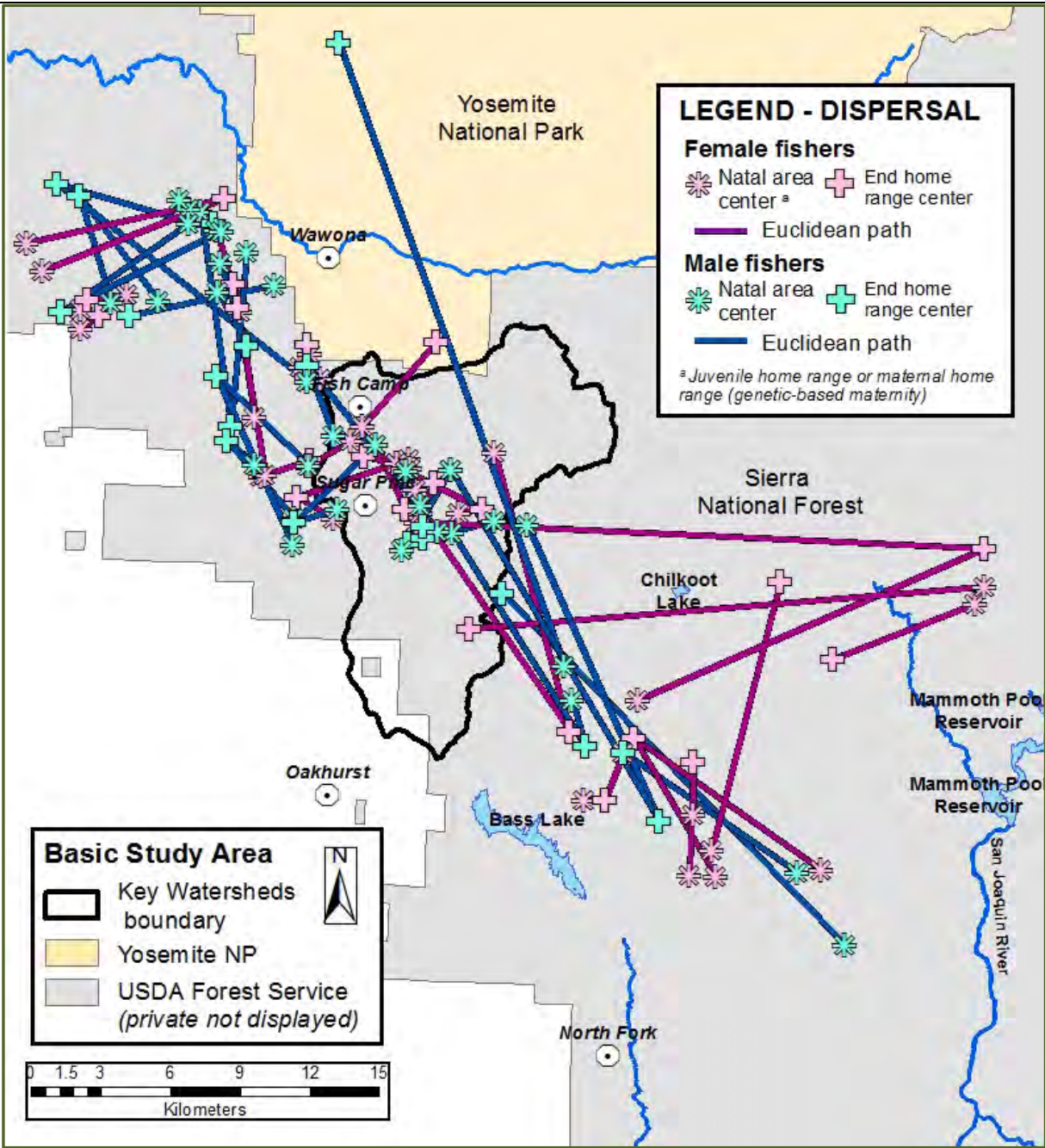


Figure 20. Plot of Euclidean distance dispersal movements for juvenile and young subadult female and males fishers within the SNAMP Fisher Project study area. NOTE: plot excludes the dispersal track for fisher M47 (KRFPP fisher that dispersed north from south of Shaver Lake (High Sierra District) to near “Central Camp” in the Bass Lake District).

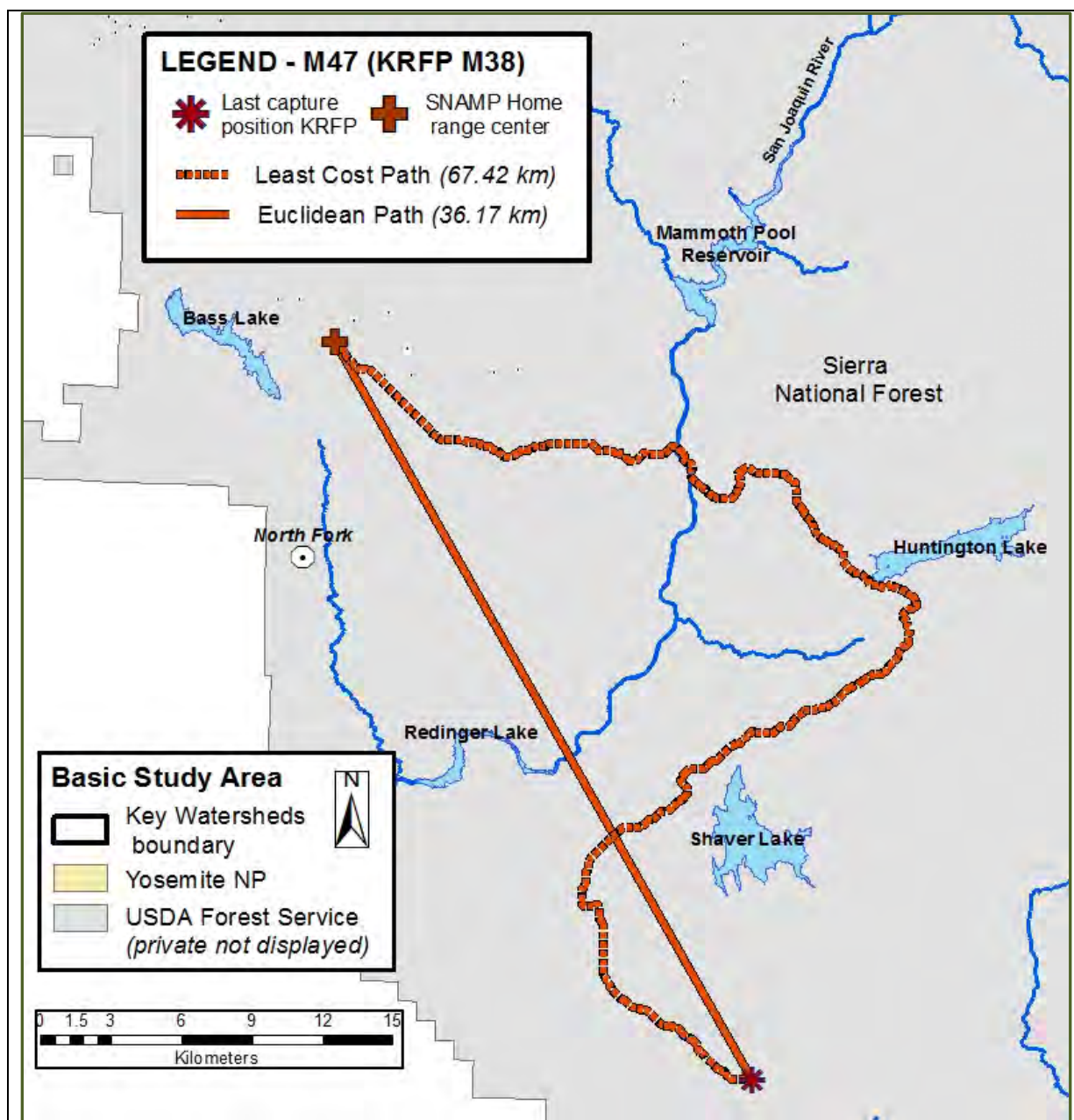


Figure 21. Plot of the Euclidean dispersal track for KRFP Fisher M38 from his last live-trap position in February 2013 to his postdispersal home range centroid near Central Camp within the SNAMP Fisher Study area. The plot also includes the estimated Least Cost Movement path for the M38 dispersal event, which we consider more realistic given the very steep, and vertical cliffs typical of the San Joaquin River canyon east of Redinger Lake.

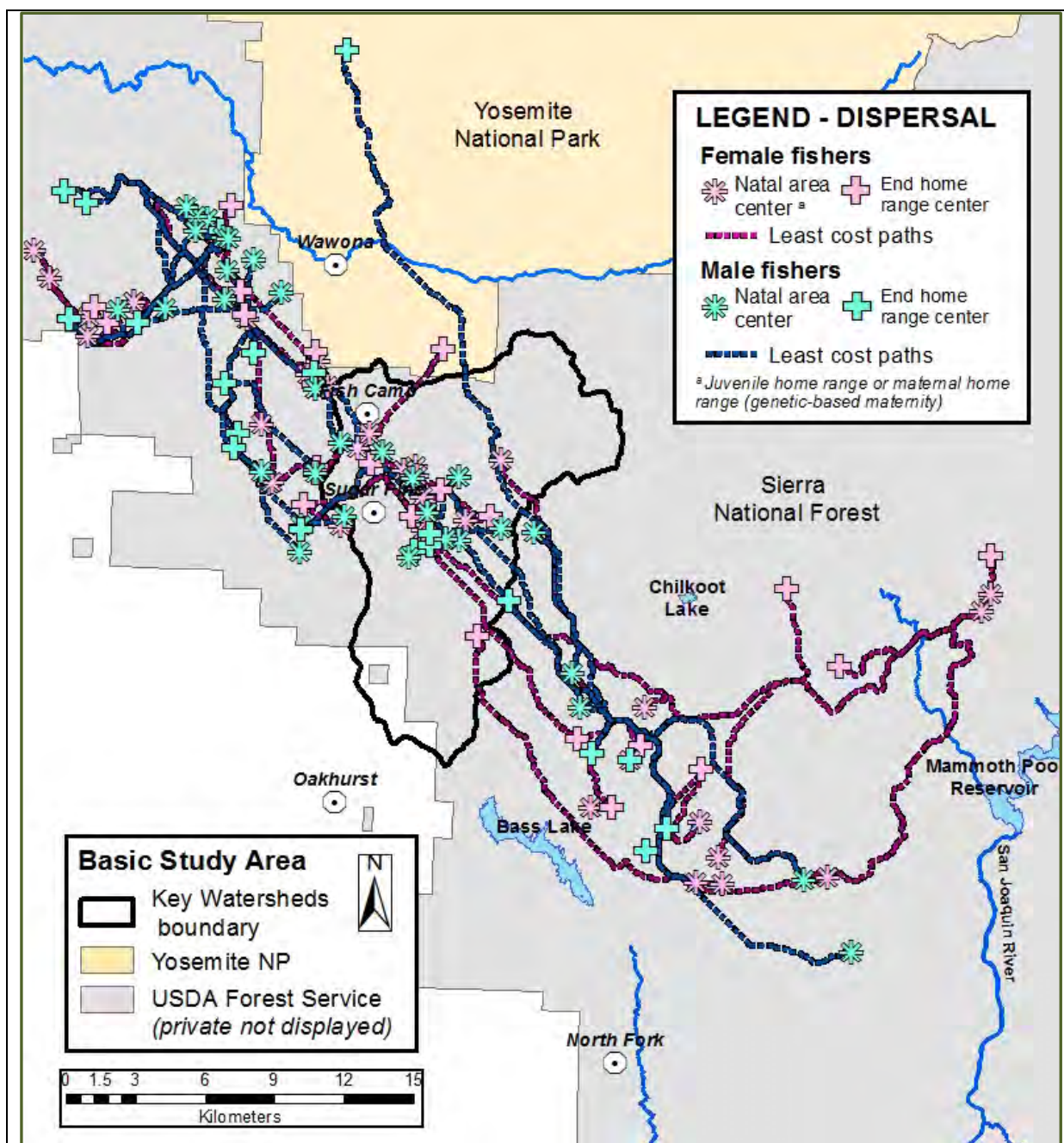


Figure 22. Estimated Least Cost Movement paths for young fishers (≤ 18 months) that were assessed for dispersal in the SNAMP Fisher study area from 2008 to 2013. Least cost movement path were developed as a more realistic way to assess fisher movements given that a number of landscape and habitat features are known avoided or restrictive to fishers as part of their overall natural history.

Young female fishers appeared somewhat more philopatric than male fishers, based on the proportion that moved less than the mean diameter of the annual home range for adult females in the study population) (Fig. 23). The pattern was not significantly different however (Pearson $\chi^2 = 1.12$, $P = 0.29$). Also, there was no statistical evidence that male fishers dispersed farther than female fishers when dispersal distances were scored based on two levels of philopatry and two levels of dispersal (Euclidean distance Likelihood ratio $\chi^2 = 3.89$, $P = 0.27$; Fig. 24). The same analysis using LCP distances visualized in Fig. 22 was also nonsignificant (LCP Likelihood ratio $\chi^2 = 1.87$, $P = 0.60$; Fig. 22). However, it was noteworthy from a genetic perspective that 67% of females were philopatric, compared to about 45% of young males (Fig. 24; Table 30).

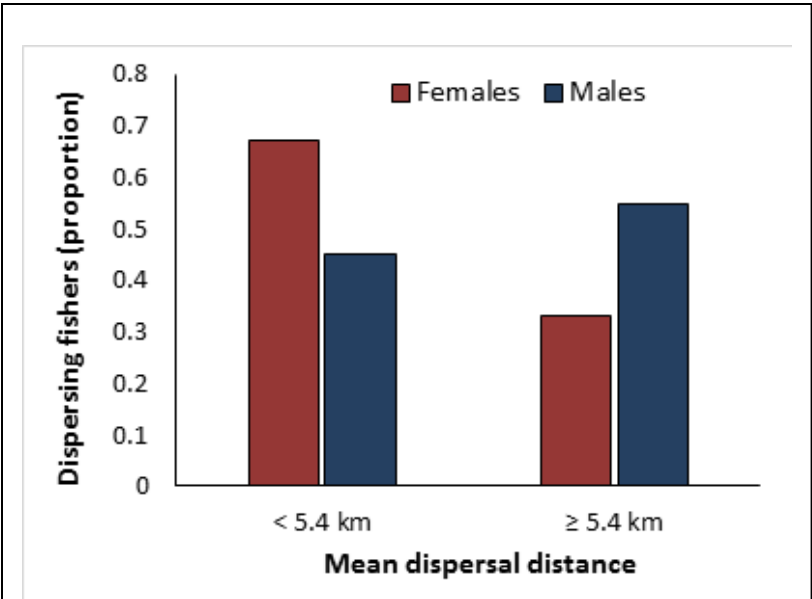


Figure 23. Proportion of female and male fishers that dispersed less than the diameter of the mean annual home range for adult female fishers (22.99 km; diameter = 5.401 km) in the SNAMP Fisher study area. Fishers that dispersed <5.4 km were considered as exhibiting philopatry, whereas those that moved >5.4 km were considered dispersers.

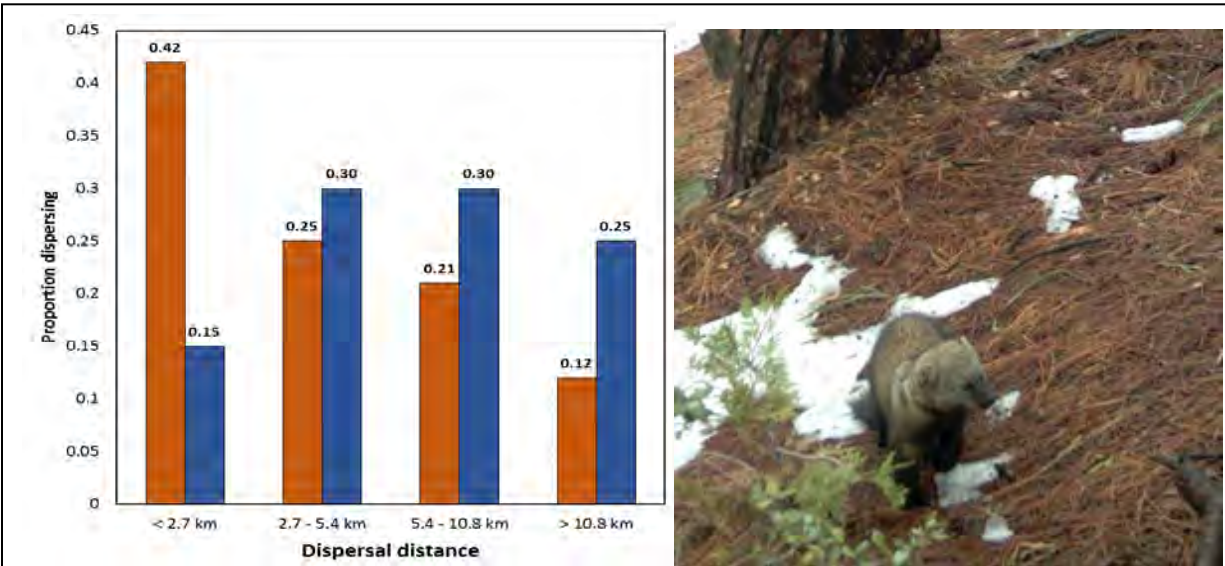


Figure 24. Plot of four different categories of dispersal distances for male and female fishers. Categories are based on dispersal distances of less than 0.5, 1.0, 1.5 and >2 times the mean diameter of the annual adult female home range in the study area. **Illustration 12 (right):** female fisher on the move.

Information on timing of dispersal is important for understanding whether juveniles captured in fall and winter were resident (born near the area of capture and initial locations), or if they originated elsewhere. Five dispersal events (20.8%) were initiated by juvenile fishers during fall to mid-winter (Table 30). Fourteen (58.3%) were initiated during the late winter to mid-spring time frame, and five started in late spring or summer (Table 30). Thus, nearly 80% of natal dispersal events occurred after February 5 when fishers were 11-13 months old.

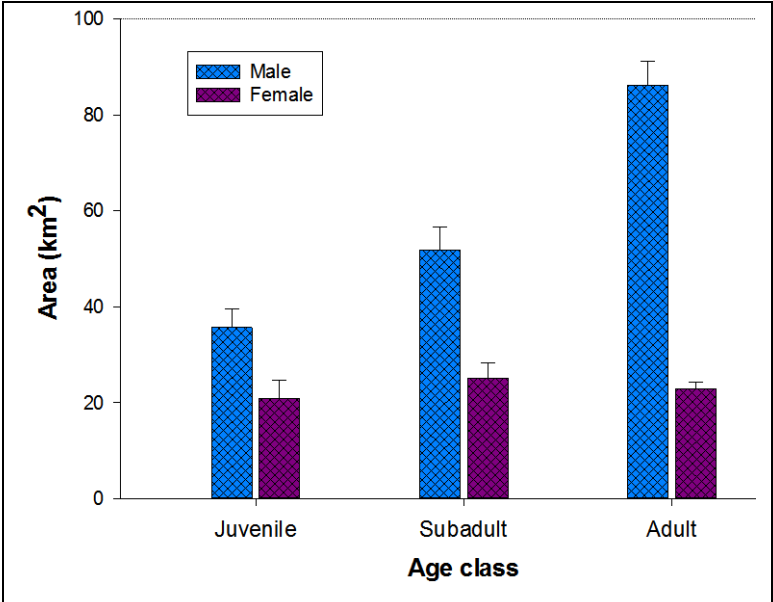
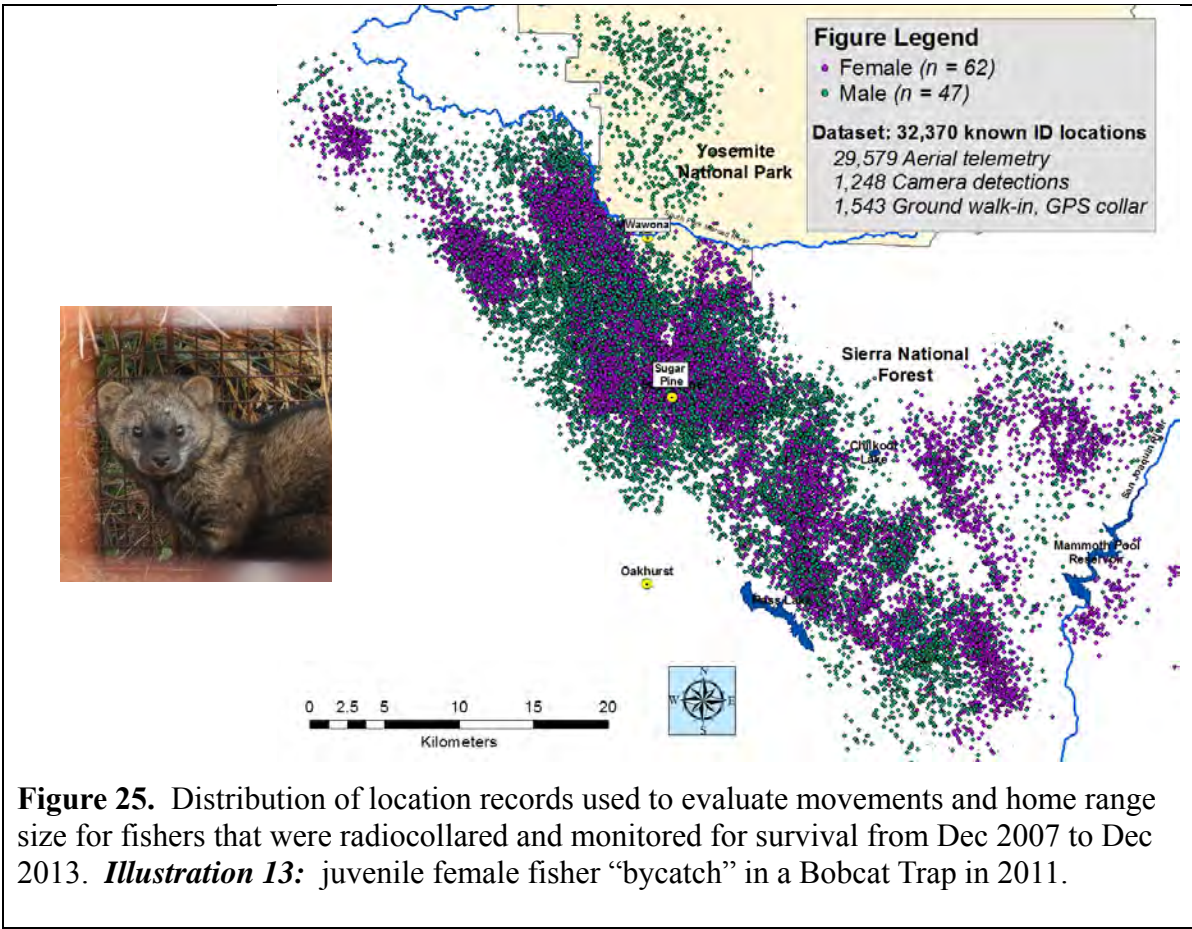
Table 30. Information on periods of the year when juvenile fishers initiated transitional movements as part of natal dispersal^a, and numbers of young fishers (<18 months old) that were philopatric, or that dispersed more than 1 diameter of the mean adult female home range (22.99 km; diameter = 5.41).

Dispersal parameter	Female	Male	Total
<i>Timing of dispersal initiation</i>			
Fall to mid-winter (Oct 15 - Feb 4)	2	3	5
Late winter to mid-spring (Feb 5 - May 5)	7	7	14
Late spring or summer (May 6 - Sep 20)	2	3	5
<i>Dispersal distance</i>			
Short distance philopatric (< 2.7 km) ^b	10	3	13
Philopatric (2.7 km to 5.4 km) ^c	6	6	12
Medium distance dispersal (5.4-10.8 km) ^d	5	6	11
Long distance dispersal (>10.8 km) ^e	3	5	8
^a Data on initiation of dispersal were for a smaller subset of juveniles ($n = 22$) that made transitional movement that were apparent based on aerial telemetry locations and home range models			
^b <0.5X diameter of mean adult female home range			
^c 0.5-1X diameter of mean adult female home range			
^d 1-2X diameter of mean adult female home range			
^e >2X diameter of mean adult female home range			

Home Range Dynamics

We obtained and processed $\approx 35,365$ location records from all sources (Table 3; Fig. 25) from October 2007 to December 2013. The location dataset was screened for errors and duplicates (same day, same animal, <8 hrs apart in time), after which approx. 32,370 of the locations were retained for detailed analyses of movements (home ranges) for 109 different fishers. Most of the location records were from aerial radiotelemetry (88%), which were less accurate than other types of locations in the database (Table 3).

Annual 95% fixed kernel home range areas differed by gender for all age classes (Fig. 26), with mean values ranging from 20.98 km² for juvenile females to 86.18 km² for adult males (Table 31). Male fishers are larger in body mass and morphological size than females (Powell 1993), and size dimorphism was already evident between genders when juvenile fishers were captured and



measured in October and November (7-8 months old) (Focused Science Topic 1; Table 36). Body size is closely related to home range size in mammals (Swihart et al. 1988), which helps explain the larger size of annual home ranges for all age classes of male fishers in this study (Table 31, Figure 21).

Although fishers have previously been described as exhibiting intrasexual territoriality (Powell et al. 2003), we noted considerable overlap between the annual home ranges of adults of the same sex (Fig. 27). Annual home ranges

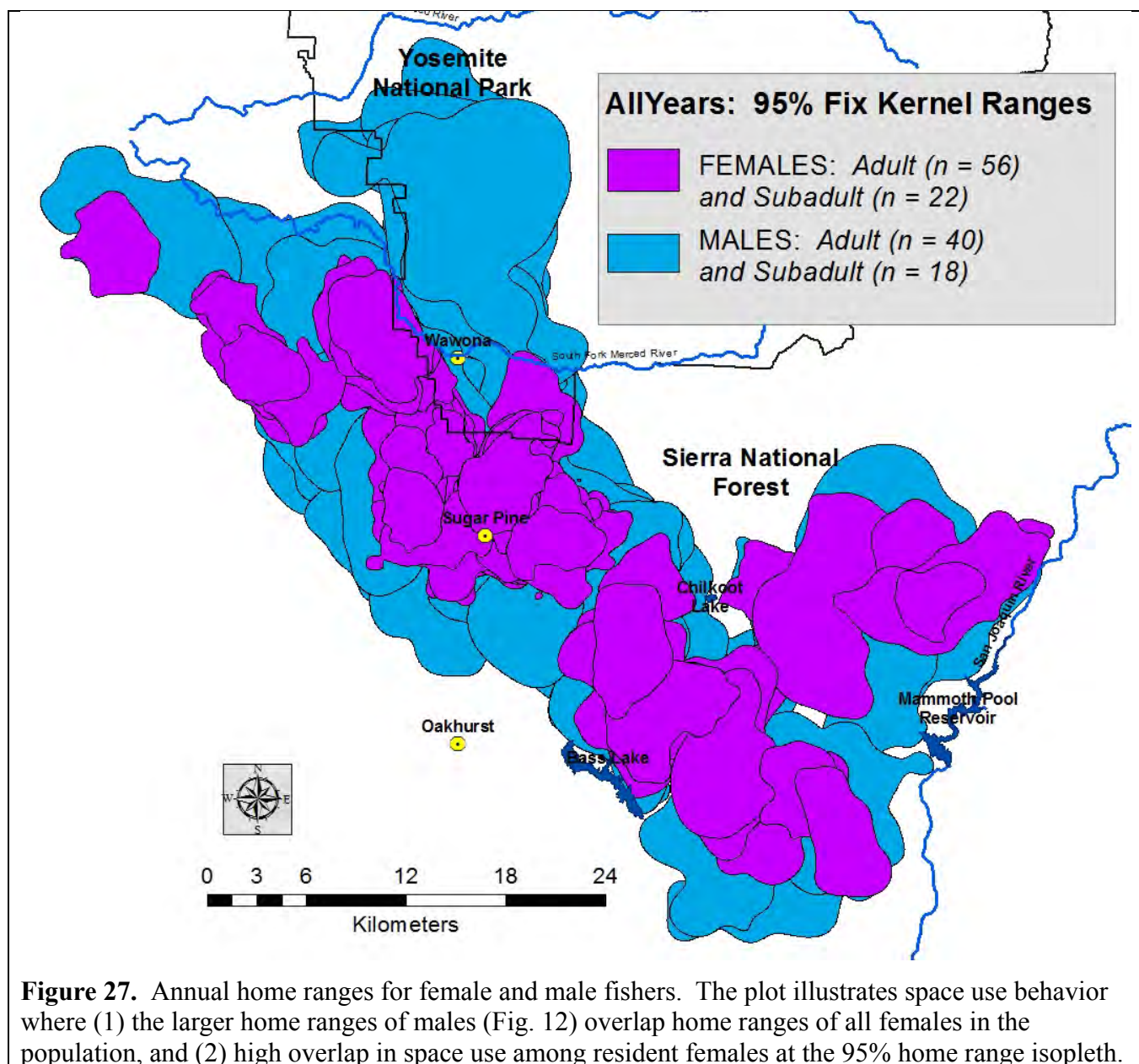


Figure 27. Annual home ranges for female and male fishers. The plot illustrates space use behavior where (1) the larger home ranges of males (Fig. 12) overlap home ranges of all females in the population, and (2) high overlap in space use among resident females at the 95% home range isopleth.

overlapped extensively among neighboring females, but overlap declined at the 70 or 60 percent fixed-kernel isopleths. These results suggest that female fishers maintain exclusive intra-sexual territories in their core use areas. Adult males move widely during the breeding season, resulting in widely overlapping use areas during spring (Popescu et al. 2014).

Home range sizes for fishers varied seasonally (Table 32; Fig. 28). Adult female home ranges were smallest during the spring, and reproducing females have smaller home ranges than non-reproducing females during this time when mothers are constrained to the den area and provisioning kits at den structures. Home ranges of denning females were smaller than non-reproductive female home ranges through the summer, before offspring become independent. Size of seasonal home ranges among adult

male fishers was smallest during the summer and largest during the spring, reflecting wide movement associated with mating during March and April (Table 32). In contrast, seasonal home ranges of subadult males (likely non-reproductive) were largest during winter and relatively stable during spring, summer, and fall (Table 32).

Excluding the spring season home range for adult males, home range size was largest for all age and sex classes of fishers during winter, likely due to scarcity of prey

Table 31. Mean annual and core use home range sizes ($\text{km}^2 \pm \text{SE}$) for radio-tracked fishers at the SNAMP site, December 2007 to March 2013

Age/Sex	N	Annual ^a	Core use ^b
<i>Juvenile (<12 months)^c</i>			
Female	10	20.98 \pm 3.76	6.59 \pm 1.18
Male	4	35.68 \pm 3.83	11.86 \pm 1.02
<i>Subadult (12 to 23 months)</i>			
Female	22	25.15 \pm 3.20	8.59 \pm 1.09
Male	18	51.85 \pm 4.76	18.15 \pm 1.66
<i>Adult (≥ 24 months)</i>			
Female	56	22.93 \pm 1.36	7.78 \pm 0.59
Male	40	86.18 \pm 4.87	30.23 \pm 1.78

^aAnnual home ranges were estimated for fishers for which locations were available for ≥ 6 months between Apr 1 and Mar 31; number of location records used for annual home models ranged from 77 to 326.

^bCore use home range estimated using methods described in Seaman and Powell (1990) and Bingham and Noon (1997); $\sim 2/3$ of the core use areas were the 60% isopleth, the remainder were the 70% isopleth.

^cHome ranges for juvenile fishers monitored ≥ 5 months in Oct to Mar period; excludes fishers that exhibited dispersal movement behavior.

Table 32. Mean home range sizes (95% Fix Kernel; $\text{km}^2 \pm \text{SE}$) for fishers during four seasons^a of the year. Data for animals radio-collared on the Sierra National Forest, CA from December 2007 to March 2013.

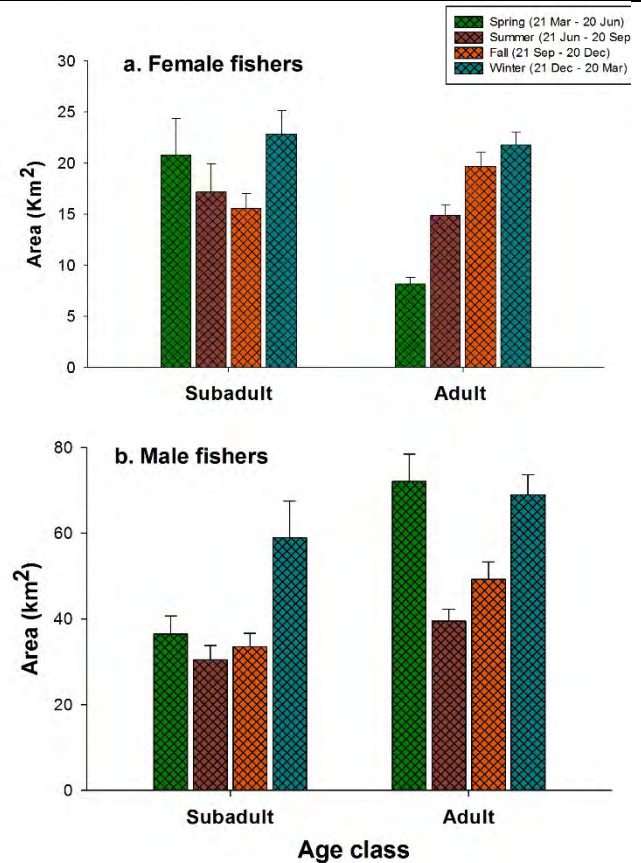
Age	n	Spring	n	Summer	n	Fall	n	Winter
<i>Juvenile^b</i>								
Female				11	16.24 \pm 2.52	17	18.72 \pm 2.46	
Male				4	20.10 \pm 3.51	9	48.92 \pm 12.73	
<i>Subadult^b</i>								
Female	17	20.78 \pm 3.59	21	17.19 \pm 2.73	21	15.61 \pm 1.41	22	22.87 \pm 2.28
Male	12	36.48 \pm 4.26	13	30.49 \pm 3.31	14	33.51 \pm 3.15	19	58.90 \pm 8.61
<i>Adult</i>								
Female ^c	59	8.18 \pm 0.64	43	14.92 \pm 1.02	50	19.70 \pm 1.37	50	21.77 \pm 1.28
Male	35	72.07 \pm 6.39	34	39.49 \pm 2.75	32	49.25 \pm 4.04	37	68.91 \pm 4.73

^aSeasons were Spring: 21 Mar to 20 Jun; Summer: 21 Jun to 20 Sep; Fall 21: 21 Sep to 20 Dec; Winter: 21 Dec to 20 Mar.

^bExcludes home ranges for fishers that exhibited movements associated with dispersal

^cIncludes home ranges for adult females that denned during the spring period of each year and excludes nondenning adults.

Figure 28. Plot illustrating size of the mean seasonal home range size (SE bars) for female (a) and male (b) fishers from the SNAMP Fisher Project. NOTE: the scale is different for the two plots, which helps to illustrate similarities in habitat use patterns for the different age and sex groups. More details are provided in Table 32. (*Illustration 14*: adult female fisher departing a black oak den tree in spring 2011)



SNAMP Fisher Management Indicators

Management indicator 1 (occupancy/presence of fisher detections in 1-km² grids within the Key Watersheds) ranged from a low of 53% in 2012-13 to a high of 76% in 2011-12 (Table 33). The index of fisher activity developed for Management Indicator 1 indicated that the estimated detection rate (detections/100 camera survey days) was highest in 2012-13 and lowest in 2010-11. It was unusual that the detection rate was highest in the same year that naïve occupancy was lowest (Table 33). Camera year 2012-13 was atypical in that many grids in the Key Watershed were surveyed during summer when detection rates are significantly lower (Popescu et al. 2014). It was therefore possible that the low occupancy for 2012-13 compared to most other years was related to timing of surveys.

Spatially, the distribution of fisher active grids changed among years (Fig. 29). Visually, there was the appearance that fisher detections were somewhat reduced in the Cedar Valley Project region of the Key Watersheds (center-south; Figs. 4, 29) immediately after project implementation. There were also changes in fisher detections in the northeast region of the Key Watersheds, which may have been associated with mastication and other activities associated with the Fish Camp Project (Figs. 4, 29).

Visual comparisons of presence/absence are not appropriate for detecting patterns or trend in occupancy (persistence, extinction, recolonization) related to forest management projects, however. Detailed, multi-year occupancy modeling analyses are underway, which include the proportion of each grid treated in each of six years by different forest management activities. Models also include other covariates potentially important for understanding detection histories and habitat use (e.g., season, elevation).

Table 33. Management indicator for fisher activity in the Key Watershed focal study area, based on the number of 1 km² grids in which fishers were detected during annual surveys with camera traps.^a Camera trap surveys were completed in each of six "camera years" (\approx Oct 15 to Oct 14) using our standard protocol.

Camera year	Grids surveyed	Grids with fisher detections	Naïve occupancy	Fisher detections per 100 survey days ^c
2007-08	122	71	0.582	11.4
2008-09	129	75	0.581	13.2
2009-10	127	75	0.591	15.2
2010-11	125	82	0.656	10.5
2011-12	128	98	0.766	14.2
2012-13	133	70	0.526	18.6

^a Camera trap surveys were completed in each of six "camera years" (\approx Oct 15 to Oct 14) using a standard protocol (see report)

^b Number grids with fisher detections divided by the total number of grids surveyed; occupancy rate is not corrected for a survey-specific probability of detection < 1.0 .

^c Estimated as the number of functional camera survey days with fisher detections, but excluded camera days for grids with no fisher detections.

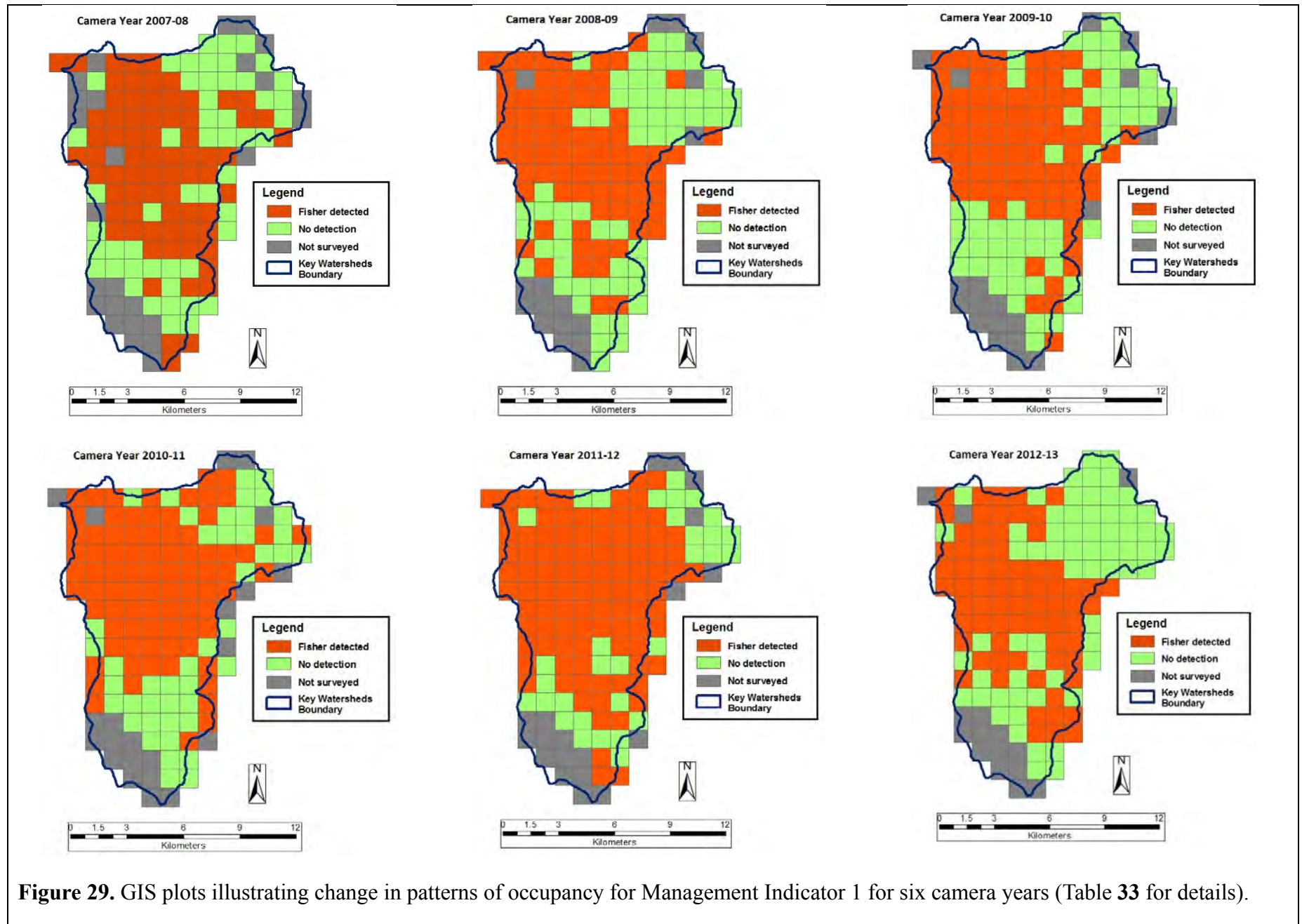


Figure 29. GIS plots illustrating change in patterns of occupancy for Management Indicator 1 for six camera years (Table 33 for details).

As an extension to Management Indicator 1, We also created an overall index of fisher activity for each grid based on mean number days in each camera survey year with fisher detections and the proportion of survey years with fisher detections (Fig. 30). The index illustrates that fisher activity was consistently high in the center and northwest region of the Key Watersheds, and lowest from Cedar Valley southward (Fig. 30).

Management Indicator 2 identified an average of 5.0 subadult or adult females and 2.0 subadult or adult males using the Key Watershed focal study area across all years (Table 34). For both sexes combined, the number of resident fishers using the focal study area ranged from 6.2 to 7.7, and the variation among years was small (Table 34, Fig. 31).

Table 34. Management indicator for the number of resident subadult and adult fishers using the Key Watershed focal study area for their various home range activities during Sep 1 to Mar 15 of each year.			
Year	Females ^a		
	Males	Both genders	
2007-08 ^b			
2008-09	5.6	2.1	7.7
2009-10	6.1	1.4	7.5
2010-11	4.1	2.1	6.2
2011-12	4.0	2.9	6.9
2012-13	5.0	1.7	6.7

^a Numbers are based on the sum of the proportion of each individual fishers' 95% fixed kernel home range included within the Key Watershed region.

^b Because of the limited number of fishers radiocollared during the first project year ($n = 7$, before March 31, 2008) it was not informative to calculate this Management Indicator in that year.

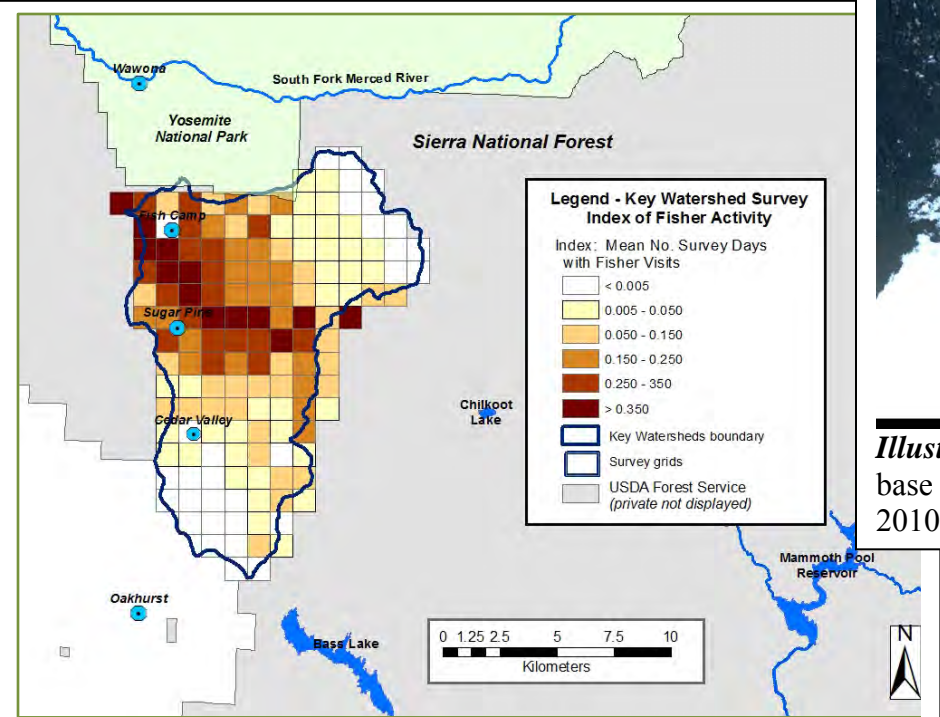


Figure 30. Index of fisher activity from repeat camera surveys completed in the Key Watersheds focal study area. The Index was calculated as the mean no. of days with fisher activity for years that the grid was surveyed/1 + proportion of surveyed years with fishers detections in the grid.

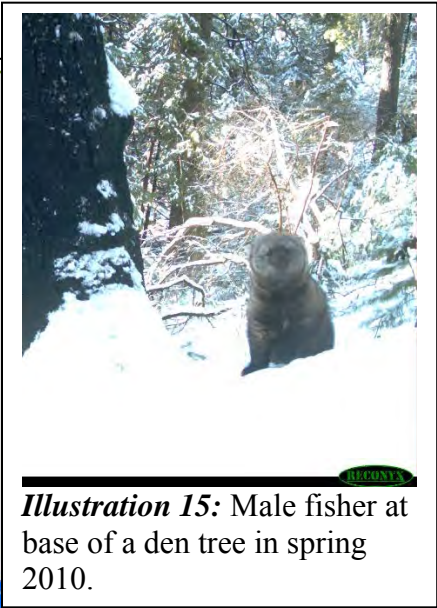
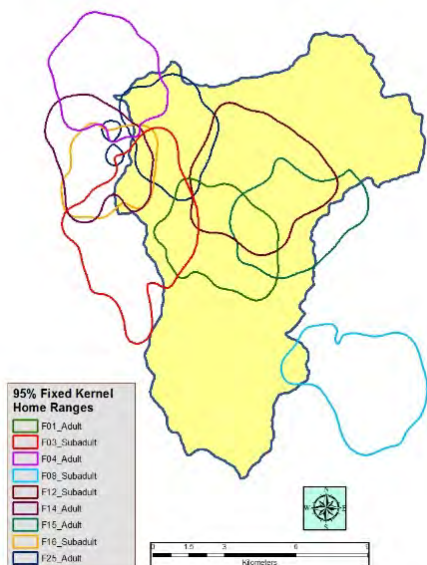
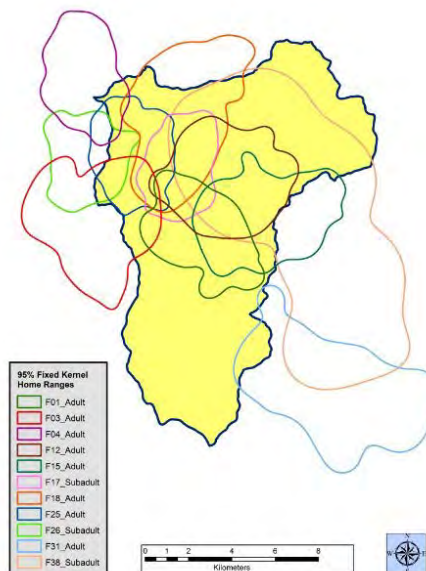


Illustration 15: Male fisher at base of a den tree in spring 2010.

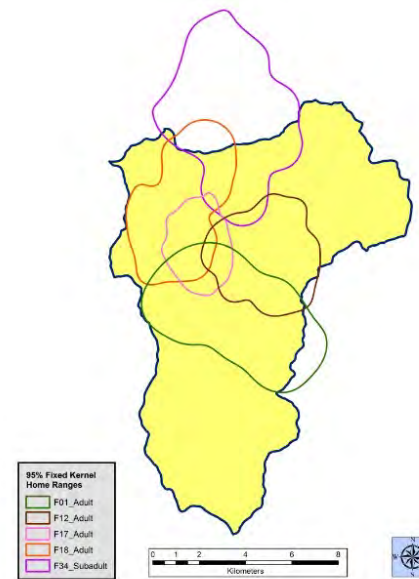
Population Year 2: 5.6 resident females



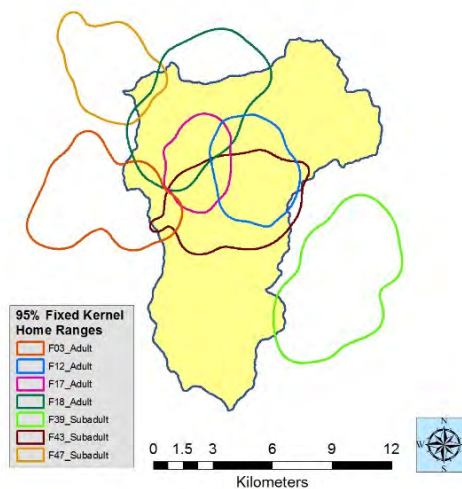
Population Year 3: 6.1 resident females



Population Year 4: 4.1 resident females



Population Year 5: 4.0 resident females



Population Year 6: 5.0 resident females

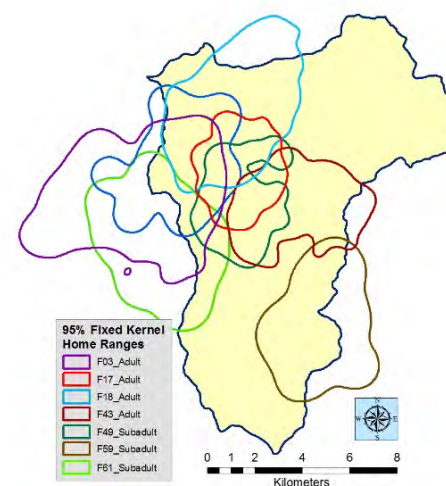


Figure 31. Estimated number of subadult and adult female fishers (resident females) with home ranges including portions of the Key Watersheds focal study area. Polylines are individual 95% fixed kernel home ranges based on location records during the Sep 1 to Mar 15 period during each population year (Apr 1 to Mar 31).

The original Management Indicator 3 was recast to estimate survival for adult female fishers for a sequence of 2-year groups of demographic data (data combined for Kaplan-Meier models of survival). For the first 2-year group, we included data for the small number of fishers ($n=7$) that were captured and radiocollared from mid December 2007 to March 31, 2008. We further summarized data on Juvenile and subadult female survival, and calculated point estimates of weaning reproduction and weaning litter size for each of the five 2-year groups (Table 24). Expanded Management Indicator 3 identified that adult female survival ranged from a low of 0.69 in Year group 3 to a high of 0.86 in Year group 4 (Table 35). Corresponding data on survival for juvenile and subadult females and data on reproduction identified that relatively low levels of survival and reproduction suggested the population was in decline ($\lambda < 1.0$) population between 2008 and 2010, stable between 2010 and 2012, and increasing by 3-4%/year during 2012 to 2014 (Table 35). However the fact that 95% CI for λ overlapped 1.0 in all years indicates that these values should be interpreted carefully.

Table 35. Expanded Management Indicator 3 for adult female survival, including Leslie Matrix population growth rates for two year running average starting in 2008 and ending in spring 2014. Population years start April 1 and end March 31. Numbers in parentheses for survival are the 95% CIs based on Kaplan-Meier staggered entry survival analyses for the group of years identified.

Year group, Demographic rate	Juvenile	Subadult	Adult	λ^a
1. 2007-08, 2008-09, 2009-10 <i>b</i>				
Survival, $s(t)$	0.76 (0.53-0.99)	0.47 (0.28-0.67)	0.81 (0.66-0.96)	0.87 (0.65-1.08)
2. 2009-10, 2010-11				
Survival, $s(t)$	0.8 (0.59-1.01)	0.67 (0.42-0.92)	0.70 (0.54-0.86)	0.88 (0.63-1.12)
3. 2010-11, 2011-12				
Survival, $s(t)$	0.74 (0.54-0.94)	0.89 (0.71-1.07)	0.69 (0.53-0.86)	1.00 (0.77-1.22)
4. 2011-12, 2012-13				
Survival, $s(t)$	0.80 (0.55-1.05)	0.73 (0.52-0.94)	0.86 (0.71-1.00)	1.04 (0.81-1.26)
5. 2012-13, 2013-14				
Survival, $s(t)$	1.0	0.73 (0.48-0.98)	0.74 (0.56-0.93)	1.03 (0.77-1.22)

^a Population growth rate was estimated using the demographic parameters developed for each Year group in a Leslie-Matrix model formulation described previously. The range in values for λ was based on the 95% CIs for survival for each age class when producing fertility (F_i) rates using the equation $F_i = b_i P_i$, where $b(i)$ was fecundity, and P_i was the age-specific survival rate (see Table 23).

^b Year group 1 includes information for a small number fishers monitored for survival from late December to March 2008. All other year groups include two population years of data.

Fisher response to fuel management

Management disturbances and wildfire

Our analyses of FACTS and other extractive and restorative management activities revealed that the estimated area of forest disturbing activities that occurred in the study area was highest for restorative fuel reduction, moderate for logging, and lowest for managed burning and natural or human caused wildfires (Table 36). We estimated that there was an annual average 1.9% (SD 0.70) of the study area treated for restorative fuel reduction each year from 2002-03 to 2012-13, and 20.6% of the study area was disturbed by these activities in all 11 years. We estimated that there was an annual average of 1.1% (SD 0.70) of the study area with extractive fuel reduction each year from 2002-03 to 2012-13, and an estimated 12.1% of the study area was disturbed by logging in all 11 years. We estimated that there was an annual average of 0.25% (SD 0.28) of the study area with managed burning each year from 2002-03 to 2012-13, and an estimated 2.8% of the study area was disturbed by managed burns in all 11 years. Also, the combined area disturbed by all 3 management activities averaged 36.3 km²/year from 2002-03 to 2012-13, which represented an annual disturbance of 3.2%/year from SPLATs in the overall study area. Our fire variables included managed burns+forest fires, and we estimated that the annual average portion of the study area with managed burns+wildfires was 0.56%/year (SD, 0.83) from 2002-03 to 2012-13, and 6.2% of the overall study area was exposed to those disturbances in the 11 years. Also, in the 44 years from 1957 to 2001, we estimated that 130.2 km² (11.6%) of the overall study area was burned by wildfires.

Multi-season occupancy

The mean detection probability for fishers per 8-10 day survey period in the 361 multi-season survey grids was 0.31 (95% CI: 0.28, 0.37). Naïve initial occupancy among the multi-season grids was 0.66, whereas our modeled estimate for initial occupancy averaged across survey sites was 0.75 (95% CI: 0.59, 0.87). Mean annual persistence (1-extinction) was 0.87 (95% CI: 0.82, 0.91), whereas the annual colonization rate was 0.34 (95% CI 0.28, 0.42).

Our multi-season occupancy modeling identified a single best model for local colonization that included the intercept only (Table 37). Covariates *hazfuels.5*, *log.5*, and *burn.1.50* were included in 3 lower ranking colonization models with support, but the relative importance for each individual variable was ≤ 0.35 . We therefore fit an intercept-only colonization component in our subsequent evaluation of extinction covariates.

Table 36. Estimates of the areas (km²)^a disturbed by logging activities, fuel reduction treatments, and managed burns in the Bass Lake District, Sierra National Forest, and southwestern Yosemite National Park in 11 camera survey years (Oct 15 to Oct 14) from 2002 to 2013 as well as wildfire activity in 5-year periods from 1957 through 2001.

5 yr period or survey year	Restorative fuel reduction		Extractive fuel reduction		Managed burns + forest fire	
	Area	Study area (%)	Area	Study area (%)	Area	Study area (%)
1957 to 1961					36.40	7.28
1962 to 1966					5.30	1.06
1967 to 1971					6.05	1.21
1972 to 1976					3.43	0.69
1977 to 1981					4.65	0.93
1982 to 1986					11.46	2.29
1987 to 1991					41.91	8.38
1992 to 1996					0.99	0.20
1997 to 2001					20.05	4.01
2002-03	23.7	2.10	11.6	1.03	3.5	0.31
2003-04	13.7	1.21	2.9	0.26	3.7	0.33
2004-05	13	1.15	7	0.62	4.3	0.38
2005-06	26	2.31	6.8	0.60	2.4	0.21
2006-07	34.5	3.06	13.1	1.16	5.3	0.47
2007-08	15.8	1.40	2.1	0.19	34.0	3.02
2008-09	29.1	2.58	11.4	1.01	3.8	0.34
2009-10	27.4	2.43	27	2.39	1.0	0.09
2010-11	12.4	1.10	13.8	1.22	6.1	0.54
2011-12	12.6	1.12	24.3	2.16	0.1	0.01
2012-13	23.9	2.12	16.1	1.43	5.7	0.50
<i>Total area</i>	<i>232.1</i>		<i>136.1</i>		<i>69.87</i>	

^a Areas of disturbance were derived from FACTs data, private timber harvest data, and Sierra National Forest and Yosemite National Park databases (Table 1). The overall study area was 1125.6 km² (Fig. 1), which was used to estimate percent disturbance of each type within the study area.

^b Totals for 1957 to 2001, and 2002-03 to 2012-13, respectively.

^c Means for 1957 to 2001 (44 years), and 2002-03 to 2012-13 (11 years), respectively.

Our multi-season models evaluating local extinction identified a single top model including covariate *hazfuels.5* only (*hazfuel.5* relative importance = 0.98) (Table 37). There were 2 models with support that included the covariates *log.5* and *burn.1.50*, but the individual relative importance metrics for both were low. We found that fisher persistence (1 - extinction) was negatively associated with

hazfuels.5; probability of persistence decreased by 27% as the proportion of the grid treated for cumulative restorative fuel reduction increased from 0 (occupancy = 0.89, 95%CI 0.85, 0.92) to 1.0 (occupancy = 0.65, 95%CI 0.46, 0.81).

Table 37. Candidate models for multi-season occupancy evaluations of local patch extinction and colonization for camera trap surveys for fishers in the Bass Lake District, and southwestern Yosemite National Park, California from Oct 2007 to Oct 2014.

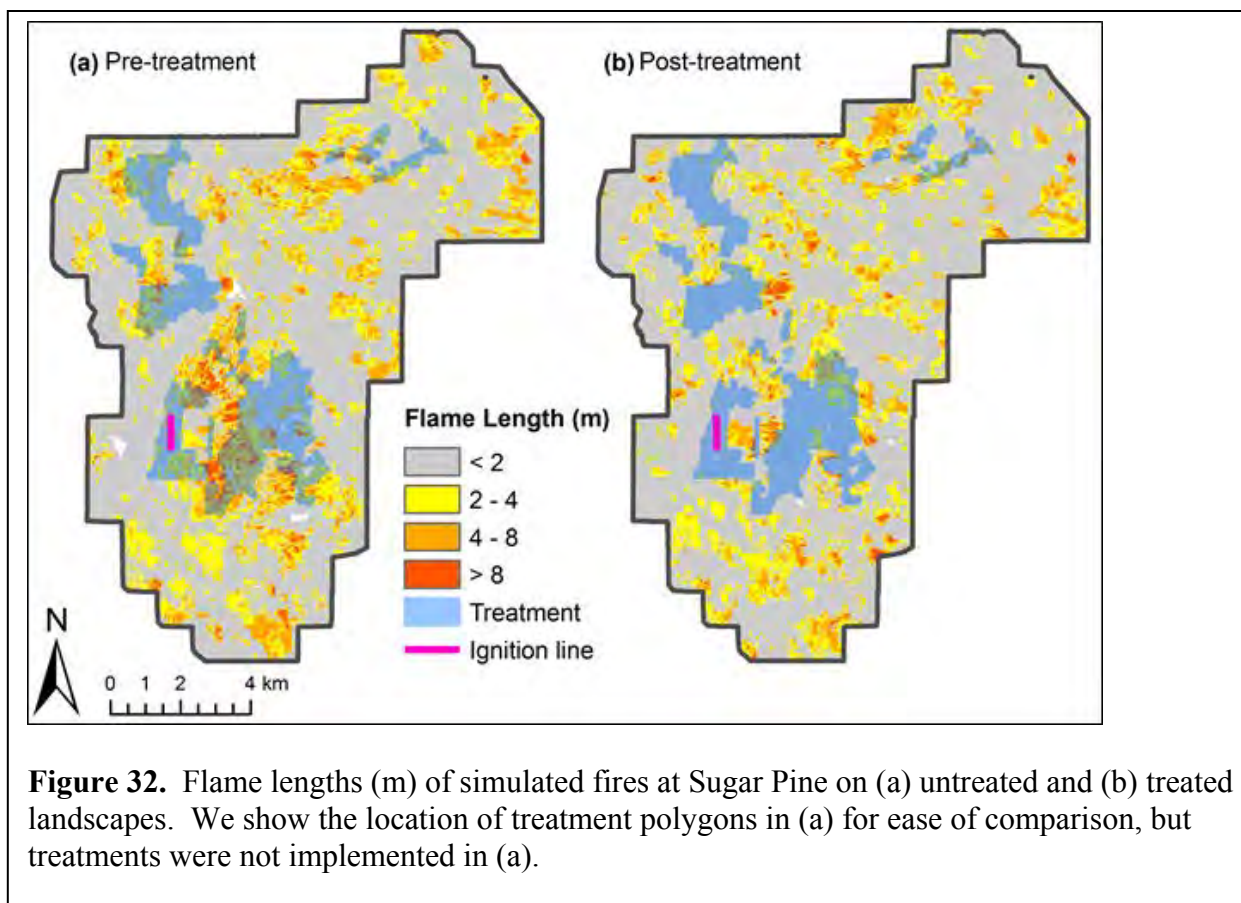
Model, covariate	AIC	Δ AIC	AIC _{wt}	Cumulative AIC _{wt}	Covariate importance
<i>Colonization</i>					
intercept only	4211.96	0.00	0.34	0.34	
<i>hazfuels.5</i>	4213.15	1.19	0.19	0.53	0.35
<i>log.5</i>	4213.87	1.91	0.13	0.66	0.27
<i>burn.1.50</i>	4213.95	2.00	0.13	0.79	0.27
<i>hazfuels.5 + log.5</i>	4215.14	3.19	0.07	0.86	
<i>hazfuels.5 + burn.1.50</i>	4215.15	3.19	0.07	0.93	
<i>burn.1.50 + log.5</i>	4215.87	3.91	0.05	0.97	
<i>hazfuels.5 + burn.1.50 + log.5</i>	4217.14	5.19	0.03	1.00	
<i>Extinction</i>					
<i>hazfuels.5</i>	4205.26	0.00	0.50	0.50	0.98
<i>hazfuels.5 + log5</i>	4207.08	1.82	0.20	0.70	
<i>hazfuels.5 + burn.1.50</i>	4207.11	1.85	0.20	0.90	
<i>hazfuels.5 + burn.1.50 + log.5</i>	4208.96	3.70	0.08	0.98	
intercept only	4212.67	7.42	0.01	0.99	
<i>log.5</i>	4214.16	8.90	0.01	0.99	0.29
<i>burn.1.50</i>	4214.61	9.36	0.00	1.00	0.28
<i>burn.1.50 + log.5</i>	4216.05	10.80	0.00	1.00	

Integration

Fire modeling

Fuels treatments reduced the intensity of the simulated fire, as evidenced by the predicted flame

lengths (Figure 1). On the untreated landscape, 68.6%, 18.4%, 11.2%, and 1.8% of the study area experienced flame lengths of <2, 2-4, 4-8, and >8 m, respectively. In contrast, on the treated landscape, 75.1%, 16.4%, 7.5%, and 1.0% of the study area burned at these flame lengths. Collins et al. (2011) noted that flame lengths >2 m often corresponded to areas with crown fire initiation (i.e., torching). Thus, a greater proportion of the untreated landscape was exposed to potential crown fire (31.4%) than for the untreated landscape (24.9%).



Assessing the effects of fire and SPLATs on fisher habitat

We found that SPLATs caused an immediate, slight reduction in potential fisher habitat. The entire area of the four watersheds was 35,103 ac (14,206 ha), and in year 0, there were 16,013 ac (6,480 ha) of potential fisher habitat on the untreated landscape compared to 13,938 ac (5,641 ha) on the treated landscape (Figure 2). In the absence of simulated fire, the amount of habitat steadily increased over time and was actually slightly greater on the treated landscape in years 10 and 30 (Figure 2). When fire was simulated, SPLATs had a slight, positive effect on the amount of potential fisher habitat up to 30 years later. In year 30, there were 14,653 ac (5,930 ha) of potential fisher habitat on the untreated landscape compared to 15,254 ac (6,173 ha) on the treated landscape (Figure

2).

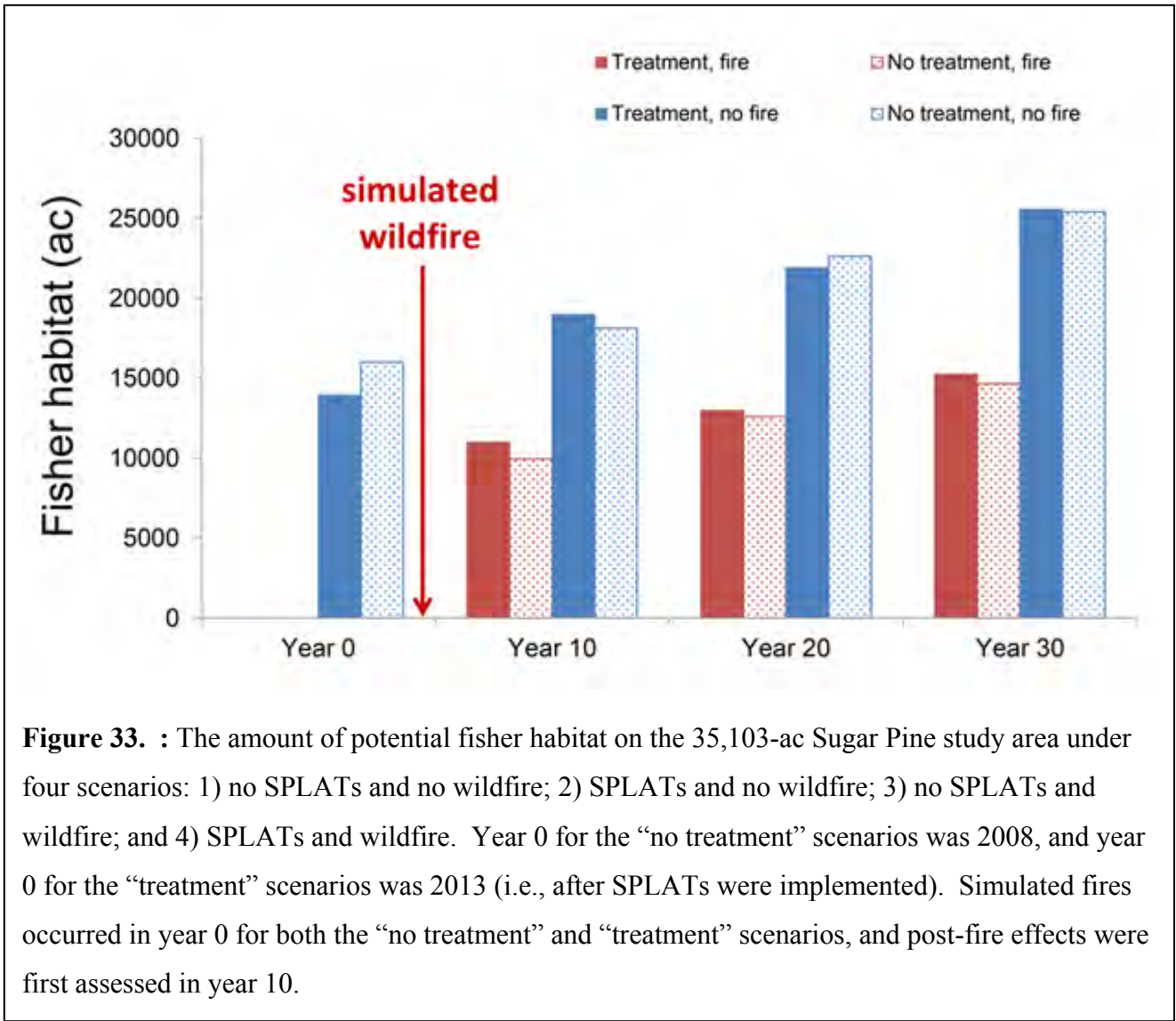


Figure 33. : The amount of potential fisher habitat on the 35,103-ac Sugar Pine study area under four scenarios: 1) no SPLATs and no wildfire; 2) SPLATs and no wildfire; 3) no SPLATs and wildfire; and 4) SPLATs and wildfire. Year 0 for the “no treatment” scenarios was 2008, and year 0 for the “treatment” scenarios was 2013 (i.e., after SPLATs were implemented). Simulated fires occurred in year 0 for both the “no treatment” and “treatment” scenarios, and post-fire effects were first assessed in year 10.

Discussion

Reproduction and Basic Demog”estimasteraphy

Empirical data on reproductive rates and litter sizes are important for understanding the ability of a population to withstand challenges to survival, and to produce realistic estimates of population size in landscape level population models being developed for conservation planning (Lofroth et al. 2010, Spencer et al. 2011). The basic life history of fishers with regards reproduction is generally well known. Fishers have a long gestation period due to the reproductive strategy of delayed implantation

(Powell 1993). Once the blastocyst implants in the uterine wall 10-11 months after fertilization of the egg, embryonic development resumes and ≈ 36 days later 1-4 kits are typically born.

Parturition for fishers in northern California typically occurs in mid to late March (Matthews et al. 2013a), and female fishers in the SNAMP Fisher population were no exception based on initiation of denning around March 22-31 (Table 13). Also, duration of denning for fishers in our study (≈ 70 -75 days; Table 13), and cessation of localization to den trees in early to mid-June was typical of elsewhere in the western United States (Matthews et al 2013a, Aubry and Raley 2006). The mean denning rate for female fishers in the SNAMP Fisher study was 0.85 (Table 15), which was slightly lower than for fishers in the Hoopa Fisher Study in northern California (0.88; Matthews et al. 2013a), similar to in the Kings River Fisher Project area (0.86; R. Green, unpublished), and higher than in southern Oregon (0.59; Aubry and Raley 2006). The average weaning rate from SNAMP females in our study area (0.74; Table 15) was higher compared to the Hoopa Fisher Study (0.65; Matthews et al. 2013a), and in southern Oregon (0.44; Aubry and Raley 2006). However, as Matthews et al. (2013a) noted, reports of low weaning rates from some studies may be due to issues with age assignment. On our study we closely tracked most animals in the population from when they were juveniles until death, and ages for nine female fishers that were captured early in the study were determined by cementum annuli (Mattson's Laboratory, Milltown, MT; Poole et al. 1994).

Average litter sizes are larger for female fishers in eastern North America (2-4 kits/litter; Paragi et al. 1994, York 1996) compared to in the western United States where litter sizes are most commonly 1 or 2 kits (Lofroth et al. 2010, Aubry and Raley 2006, Matthews et al. 2013). The mean litter size from SNAMP Fisher (1.5 kits/litter, Table 16) was similar to reports from the Kings River Fisher Project (1.6; R. Green, unpublished), but lower than 1.8 kits/litter reported for the Sequoia National Forest, California (Truex et al 1988), 1.9 kits/litter from the Hoopa Fisher Study in northern California (Matthews et al. 2013a), or 1.8 kits/litter in southern Oregon (Aubry and Raley 2006).

Close monitoring of denning behavior by several current studies including SNAMP Fisher is providing insight on difficulties female fishers encounter while attempting to reproduce. In the course of five denning seasons, we documented eight cases when females died or were killed with dependent kits in den cavities (Tables 15, 16). Death of denning female fishers appears fairly common, based on reports from the Hoopa Fisher Study ($n = 5$; Matthews et al. 2013a), the USFS Kings River Study ($n = 3$; C. Thompson, unpublished data), the ongoing California Department of Fish and Wildlife, U.S. Fish and Wildlife, and Sierra Pacific Industries "Stirling Fisher Reintroduction Project" ($n = 5$; Powell et al. 2013), and the Olympic Fisher Reintroduction Project ($n = 2$; Lewis et al. 2012). Evidence that a

significant number of females exhibiting denning behavior may die before weaning is important because weaning rates may be biased high unless estimates are based on complete monitoring through the duration of the denning period (Facka et al. 2013).

Denning Structures and Denning Habitats

Across their range in North America female fishers give birth to kits in cavities in live trees or snags (Paragi et al. 1996, York 1996, Weir et al. 2012, Zhao et al. 2012, Matthews et al. 2013a). Den cavities and habitats in the immediate vicinity of denning structures provide protection from predators and inclement weather during the early spring to late spring when females are rearing their young (Weir et al. 2012). Most female fishers use more than one denning structure during a den season (range 1-6; Matthews et al. 2013a), and female fishers on the SNAMP Fisher study used an average of 2.4 different denning structures per den season (range 1-5), compared to 3.4 on the Kings River Fisher Study (R. Green, unpublished data), and 3.1 in northwestern California (Matthews et al. 2013). Female fishers may use more than one denning structure in a season for several reasons: to accommodate kit growth by moving to larger cavities, to reduce predation risk, as bobcats and mountain lions may discover a den location due to odors from the accumulation of urine and feces, to move closer to unexploited foraging areas, and to avoid exposure to feces and parasites that may accumulate in den cavities.

The lower mean number of den trees used by female fishers on the SNAMP Fisher study area compared to on the KRFP or the Hoopa Fisher Study may be related to disturbance by researchers. Biologists on both the KRFP and Hoopa Fisher studies climb den structures of most known denning female fishers to obtain kit counts, and they also attempt to extract kits from den cavities to measure body size, collect tissue samples, and to insert PIT tags for later identification (Thompson et al. 2011, Matthews et al. 2013a,b). This process requires presence of multiple biologists at the den tree for periods of 60 to 180 minutes. Although we occasionally ascended den trees in the SNAMP study area to obtain kit counts ($n = 9$ total den tree climbs during six denning seasons), most kit counts ($\approx 90\%$) were obtained using images from 2-4 motion-sensing cameras placed around denning structures to monitor and chronicle denning activities remotely. Moreover, we noticed that some individual female fishers were sensitive to presence of technicians setting up den cameras (the process requires 30 to 60 min), based on short duration use of their denning structures after the first visit. Our den camera protocol was adjusted to minimize time needed to setup and service den cameras by quickly switching out memory cards and reviewing images away from the denning structure. Also, whenever possible,

we did not approach den trees to service cameras when radiotelemetry identified presence of the denning female.

On the nearby Kings River Fisher Project where habitats are similar to the SNAMP Fisher study area, denning structures used by female fishers were most commonly black oaks (54%: 50% live, 4% snags). Overall, 91% of denning structures used on the Kings River Fisher Project were live trees. When repeat use den trees were counted just once, 43% of the unique denning structures used in the SNAMP Fisher study area were black oak trees (Table 17; 25% live, 18% snags), and the remaining unique den trees were primarily incense cedar (33%) or white fir (15%). Only 56% of the denning structures used by female fishers in the SNAMP study area were live trees.

Weir et al. (2012) noted that trees need to have two very specific features for female fishers to use them for denning; some form of physical damage to the tree bole to provide access for decay organisms, and the damage must be of particular dimensions to provide predator-secure access for the female to the interior of the tree via frost cracks, fire-scars, large anchored branches pulling out from the tree bole, or woodpecker holes (Lofroth et al. 2010). McDonald (1990) noted that live black oaks are susceptible to internal decay and probably last longer on the landscape than conifer snags. However, 26% of 125 unique den structures used by fishers on the SNAMP study site were in conifer snags (Table 17), suggesting they are not especially uncommon on the landscape in our area. Also, our observations of incense cedars and white fir suggested these two tree types were susceptible to the types of damage identified by Weir et al. (2012), particularly with regard to fire scars for cedar trees. Many of the cedar trees selected for use as den trees had basal fire scars, and the actual den cavities in both cedar trees and white fir were commonly associated with large branch break points.

Habitat and site characteristics immediately surrounding the denning structures are likely important for appropriate thermal conditions, availability of prey, and avoidance of predators (escape cover and concealment cover). Denning structures used in the SNAMP Fisher study area were generally larger than available trees and snags; mean DBH was relatively large (larger for conifers than the hardwoods) and mean tree heights were taller for live conifers compared to conifer snags or oaks in general (Table 18, Fig. 15). Canopy cover was greater than 80% in the vicinity of many den trees (Table 19). Shrub cover near den trees was variable, as was aspect (Table 19, Fig. 16). Most den trees had multiple large down trees/logs nearby, and concealment cover to the base of den trees averaged more than 45%. Although detailed analyses of data from fixed radius habitat plots have not been completed, habitat characteristics were developed from high resolution Lidar data for many den trees within the Key Watersheds. As part of collaborative work with the Spatial Team, Zhao et al. (2012)

identified that fishers selected den sites with tall trees and steep slopes within a 10-m radius of the den tree, high forest structural complexity within 20 m, large tree clusters within 30 m, and high canopy cover and larger mature trees within 50 m. Finally, at the larger landscape scale, the mean elevation for denning structures used in the SNAMP Fisher study was 1,591 m (Fig. 15).

Fisher Survival and Cause-specific Mortality

The SNAMP Fisher study uncovered a wider diversity of causes of mortality for fishers in the region than anticipated (Table 22). In the first five years of the study many newly deceased fishers were recovered before the pilot and biologist in the airplane landed back at the Mariposa Airport, and almost always within six hours of the first indication that an animal's radiocollar was pulsing inactive. Although the fixed-wing aerial telemetry effort was expensive, the study identified the first known death for the species caused by active infection with Canine Distemper Virus (CDV). We also recovered the fresh carcass of a fisher in spring 2009 that was subsequently determined to have died from exposure to anticoagulant rodenticides. This discovery prompted testing of archived tissue samples of dead fishers throughout California and in the western United States, leading to two peer-reviewed papers focused on the problem of rodenticides and other poisons used at clandestine marijuana grow sites on California public lands (Gabriel et al. 2012, Thompson et al. 2013). Moreover, an important and very real benefit from the investment in an aviation program in support of SNAMP Fisher has been the discovery that survival and reproduction of fishers in the Sierra National Forest is challenged by multiple factors external to, and not directly linked to current forest management activities.

Well over half of the individual fishers captured and radiocollared during the study had perished as of April 2014. Known sources of cause-specific mortality in the SNAMP Fisher study population include high numbers of attacks by predators (interspecific killing; Wengert et al. 2014), roadkill deaths on Highway 41, infection by canine distemper virus (CDV; Keller et al. 2012) and *Toxoplasma gondii*, injury-induced starvation or septicemia, entrapment in a water tank, and acute toxicosis and hemorrhaging caused by exposure to rodenticides (Gabriel et al. 2013, Thompson et al. 2013). Most deaths from infectious disease, roadkill, and rodenticide exposure were males. On other hand, proportionally more females than males were killed by predators (Fig. 11), and investigation with collaborators from UC Davis indicated that males are less susceptible to death by bobcat attack than females (Wengert et al. 2014).

The diverse threats to survival that impinge on population growth in our study population are not unique to the southern Sierra Nevada region. Fishers in the Hoopa Valley in northern California area also died as a result of predation (Wengert et al. 2014), disease (Gabriel 2013), and rodenticides and other toxicants (Gabriel et al. 2013). Also, fishers that were reintroduced in northern California as part of the Stirling Fisher Reintroduction Project site have succumbed to predation, disease, and trauma from collisions with vehicles (Powell et al. 2013).

Survival estimators generally assume that live-trapping and radiocollars do not influence survival of study animals. Based on necropsies and extensive pathological tests completed on carcass remains of 47 dead fishers, no mortalities on the SNAMP Fisher study site were *directly* attributable to capture-related injury or radiocollars (e.g., strangulation or infection from chafing on the neck). However, one adult female fisher failed to survive the capture process to recovery and release. We acknowledge that the stress of capture and anesthesia contributed to the death of this female, even though detailed pathological examination revealed that she was extremely emaciated and suffering from systemic infection from serious injury (laceration to the rostrum, fractured mandible, partially disarticulated lower jaw) prior to capture (Gabriel 2013).

Analyses of live/dead status of individual fishers from SNAMP Fisher indicated that survival rates for adult females were within the range observed for other areas in the western United States. Overall survival for adult female fishers was 0.74, compared to 0.77 at the KRFP site (Sweitzer et al., In revision), which was higher than the 0.61 rate reported for a smaller sample of radio-collared female fishers on the Sequoia National Forest south of our study region (Truex et al. 1998). Aubry and Raley (2006) reported an adult female survival rate of 0.78 from a study in southwestern Oregon, whereas Higley et al. (2012) estimated adult female survival at 0.77 to 0.79 in northwestern California based on two different analytical methods (Known-fate models and Capture-Mark-Recapture, respectively). Jordan et al. (2011) reported a combined male-female adult survival rate of 0.94 for research completed in the KRFP study area in 2002-2004, however, their survival estimate was based on camera detections rather than radio-collared individuals, and the values reported were considered to have low precision related to tag loss and other factors.

In general, survival of male fishers on the SNAMP Fisher study area was consistently lower among all age and sex classes compared to females (Table 22). All year survival for SNAMP Fisher males ranged from 0.57 to 0.64, which was lower than adult male survival for fishers in southwest Oregon (0.85; Aubry and Raley 2006), in the Sequoia National Forest, California (0.73; Truex et al. 1996) and in northwest California (0.75 to 0.72; J. M. Higley unpublished report).

At the outset of the study we anticipated that survival would be lower for juvenile and subadult fishers compared to adults, as is typical for several species of mesocarnivores (Farias et al. 2005, Murdoch et al. 2010). Although survival among subadults trended lower than for adults, juvenile survival by both male and female fishers was often very similar or trended higher than adult survival (Table 22). We believe this is unlikely and an artifact of our inability to monitor juvenile survival during their first six months of life. Juvenile fishers are small in size and body mass during summer, and in our study and for most prior studies attempting to ascertain survival, juveniles were not fitted with radiocollars until fall or winter when many individuals within the cohort may have already perished (Facka et al. 2013). Even less is known about survival of kits when they are being provisioned inside den cavities (Lofroth et al. 2010). This is important because modeling efforts using empirically derived demographic parameters identify that population size and likelihood of persistence are relatively insensitive to juvenile survival (Buskirk et al. 2012, Spencer et al. 2011), potentially because juvenile survival is biased high.

The all year estimate of female survival on the SNAMP Fisher site was higher for juveniles, similar for subadults, and lower for adults compared to parameter values used by Spencer et al. (2011) to simulate fisher population dynamics under different management scenarios for the southern Sierra Nevada region. Our combined year estimates of survival for juvenile females was 0.75 (value used by Spencer et al. = 0.50), 0.71 for subadult females (value used by Spencer et al. = 0.70), and 0.74 for adult females (value used by Spencer et al. = 0.90). Spencer et al. (2011) reported their model was relatively insensitive to juvenile and subadult survival (and other demographic parameters), but highly sensitive to adult female survival. Our empirically derived estimate for adult female survival (Table 23) was 15% lower than 0.90, which is important because Spencer et al. (2011) noted that a 5% decrease in female survival produced an approximate 18% reduction in the ending population size 40 years after model initiation. Similarly, 10% and 25% reductions in female survival resulted in 37% to 72% reductions in the ending population size. The significantly higher survival rate we estimated for juvenile females might ameliorate the reduced end year population size associated with 15% lower adult female survival. A new modeling effort is underway that will integrate new information on demographic rates from the SNAMP and KRFP study sites (Spencer et al. 2014).

Wildlife populations are exposed to a variety of mortality factors, which vary in importance towards limiting or impinging on population growth. Predation was clearly identified as the most important source of mortality on the SNAMP Fisher study site (Table 21, Fig. 11). Data on percent deviation in survival described by Sweitzer et al. (In revision) indicated that predation was more

important than disease processes and human-linked factors for limiting fisher survival at the site.

Disease in the form of canine distemper, toxoplasmosis, or pleuritis+pneumonia caused death for five radiocollared fishers during the SNAMP Fisher study, and an additional four fishers died of septicemia or starvation due to puncture wounds or other injury (Table 22; Gabriel 2013). The death of four fishers on our study by infection or starvation after suffering wounds or debilitating injury was not unusual or surprising for an animal as active as the fisher. Other long term studies of radio-collared fishers have reported similar circumstances (Aubry and Raley 2006, Weir and Corbould 2008).

Infectious disease has been a conservation concern for the two isolated populations of fishers in California since exposure to CDV and other pathogens was first documented in northern California in the early 2000s based on serological testing (Brown et al. 2008). Canine distemper is of special concern because widespread, near catastrophic population-wide mortality among multiple species of endangered and uncommon carnivores has been reported (Timm et al. 2009, Williams et al. 1998, Woodroffe 1999). An outbreak, or localized epizootic of CDV that likely originated on the SNAMP site in spring 2009, and then spread south into KRFP during summer 2009 resulted in death of four fishers (Keller et al. 2012, Table 2). This disease-related mortality event confirmed that exposure by fishers to CDV and other agents of disease is of conservation concern for fishers in the western United States in general (Gabriel et al. 2012b), but particularly for the small, isolated population of fishers in the southern Sierra Nevada (Gabriel 2013). One fisher on the SNAMP study site was also confirmed to have died by complications after parasitic infection by *Toxoplasma gondii* (Gabriel 2013). Although exposure of fishers to *Toxoplasma gondii* was previously documented for fishers in North America (Larkin et al. 2011), this was the first case where complications from toxoplasmosis resulted in death (Gabriel 2013).

Wildlife-vehicle collisions may be a locally-critical mortality factor. Highway 41 is a very busy road locally referred to as the Wawona Road once it enters Yosemite National Park near the small community of Fish Camp. During the study period six non-collared fishers were also known to have been killed by vehicle strikes on Highway 41. Nine fishers were known to have been killed by vehicles along a 42 km stretch of Highway 41 during January 2008 to March 2013 in Yosemite National Park. Chow (2009) previously reported 4 fisher roadkill deaths between 1992 and 2004 along the same section of Highway 41, identifying this roadway as problematic for fisher survival in the region. Roadkill deaths of fishers have been reported in northern California as well, including two near Trinity Lake, (in either Shasta County or Trinity County – not specified; Truex et al. 1998), eight

along paved highways in Humboldt and Siskiyou County (Gabriel 2013), and one in Butte County (Powell et al. 2012). In total, we are aware of 34 documented cases of fisher mortality by vehicle-strikes in California from 1992 to 2013 (Table 21). Moreover, seven fisher deaths were reported in western Washington state in association with the Olympic Fisher Reintroduction Project (Lewis et al. 2012 unpublished report), and fishers regularly die on highways in British Columbia (R. D. Weir, personal communication), and the northeastern United States (Douglas and Strickland 1987, York 1996).

Our original prediction was that survival would be lowest during winter compared to in spring or summer. Sweitzer et al (In revision) found that this prediction was not supported by the data, and that a disproportionate number of fisher deaths occurred during spring and summer. Increased mortality of fishers in this period is potentially related to exposure to second generation anticoagulant rodenticides, which is typically applied most heavily in the spring growing season. Expanded testing for anticoagulant rodenticides in archived tissues for fishers that died on our study before 2009, and for fishers that died on the Hoopa fisher study in northern California revealed that the majority of the animals had been exposed to anticoagulant rodenticides (>80%) and other toxicants being broadcast dispersed around illegal marijuana grow sites on California public and tribal lands (Gabriel et al. 2012a). Ongoing investigations focused on this issue indicate that use of anticoagulant rodenticides at illegal grow sites is focused during spring and early summer when the marijuana plants are small and vulnerable to herbivory by rodents and insects (Gabriel et al. 2012a, 2013, Thompson et al. 2013). A total of eight fishers (three from SNAMP Fisher, five from the Hoopa fisher study in northern California) have now been documented as dying from exposure to rodenticides or other toxicants associated with marijuana grow sites died during April to June (Gabriel 2013).

Another human-linked source of death for fishers in our study was entrapment or drowning in water tank. At the SNAMP site in spring 2008 we recovered the carcass of a non-collared fisher on the ground next to an open water tank (the cover had been ajar) where maintenance crews servicing the tank deposited the animal. Truex et al. (1998) and Powell et al. (2012) both reported deaths of single radio-collared fishers in abandoned water tanks at research sites in north central California, whereas Folliard (1997) recovered skeletal remains of eight fishers from an abandoned water tank on private timberlands in northwestern California. Finally, L. Davis (personal communication, Sept 7, 2013) reported the death of a radio-collared fisher that maneuvered into a relatively short section of an upright culvert during a study of fishers in the Cariboo-Chilcotin region of British Columbia, Canada (Davis 2008). It appears that death of fishers by entrapment in water tanks and other human structures

may not be uncommon. Folliard's (1997) 15 year old recommendation that abandoned water tanks on private and public forests in California be covered, or modified by inserting branches or poles so that fishers and other wildlife can self-rescue should be applied whenever possible.

Population Size and Density

Prior to this study there was limited information on the distribution and abundance of fishers at the north margin of their extant range in the southern Sierra Nevada. Despite many years of surveys with cameras and track plates, the lack of evidence of fishers north of Yosemite Valley suggested that the population in the SNAMP Fisher study area was likely sparse (low density). Also, there had been no indication that surplus animals were dispersing northward into suitable, but unoccupied habitat north of the Merced River (Spencer et al. 2011, Spencer et al. 2014). Moreover, reports of multiple roadkill fishers along Highway 41/Wawona Road between the south boundary of the park and the tunnel just north of Yosemite Valley suggested that dispersal and the overall population was being limited by deaths on that highway (Chow 2009).

Federal and state agencies are currently developing strategies to manage for long term viable populations of fishers in the southern Sierra Nevada, and six years of intensive study as part of the SNAMP Fisher study has recently produced the first estimates of abundance for the region. We estimated the size of the fisher population in the overall SNAMP study population at 48 to 62 individuals (Table 4). Narrow confidence intervals for the population estimates were likely due to the combination of a relatively high probability of detection (0.4 to 0.75) for our camera protocol when cameras were within the home ranges of radiocollared fishers (Popescu et al. 2014) (Table 4).

Mean annual population density for the three Subregions of the overall study area ranged from 0.072 to 0.097 fishers/km² (Fig. 2), which was consistent with data from two previous studies of fishers in the High Sierra District of the Sierra National Forest, located 50 km south of our study site. Jordan et al. (2011) used a similar CMR design to estimate a density of 0.063-0.109 fishers/km² for the Kings River study area in 2002-2004. Thompson et al. (2013) used scat detector dogs and genetic detections in a spatially explicit CMR framework modified for variable search intensity to estimate a fisher density of 0.065-0.28 fishers/km² for the Kings River Fisher Project area in fall 2007. Thompson et al. (2013) emphasized that a modal density of 0.104 fishers/km² was the most appropriate point estimate developed from their research. At a research site on the Hoopa Valley Indian Reservation (Hoopa Fisher Study) in northern California, Higley et al. (2013) used CMR methods to

determine that density of fishers was stable and increasing at 0.12-0.29 fishers/km² over a 9-year period from 2005-2013. In central Massachusetts, USA, Fuller et al. (2001) applied CMR models to camera sightings and determined fisher densities of 0.19-0.25 fishers/km². Considering the subset of studies that used CMR methods, the densities we estimated for the SNAMP Fisher study area are the lowest reported (Table 4).

As previously detailed, conservation planning is underway for fishers in the southern Sierra Nevada, including new modeling to estimate areas of suitable habitat for fishers in occupied “core” regions within the southern Sierra Nevada (Spencer et al. 2014). Our study area is within Habitat Core and Connectivity Area 5, for which the area of suitable habitat was estimated as 1,096 km² (Table 2, Spencer et al. 2014). We calculated the mean density and 95% C.I. for 12 area- and year-specific densities developed by our CMR modeling (Table 4; 0.085 fishers/km², 95% C.I. 0.073-0.097), and estimated that there were 93 (range 80-107) fishers in the Southern Sierra Nevada Habitat Core and Connectivity area 5.

In the context of similar data from other studies, the population of fishers in the Bass Lake Ranger District extending into southern Yosemite National Park is small, genetically limited (Tucker et al. 2014), and exists at a density that is lower than has been reported for any part of California or North America with the exception of boreal forest regions of northern British Columbia, Canada (Weir and Corbould 2006). Moreover, there are important challenges to the long term viability of fishers in the southern Sierra Nevada region as a whole, including periodic epizootics of canine distemper (Keller et al. 2012), exposure to poisons and other toxicants that directly and indirectly reduce survival (Thompson et al. 2013), and large, catastrophic wildfires capable of eliminating thousands of hectares of foraging and denning habitat in short periods of time (days or weeks; Final Update on 2013 Rim Fire: <http://inciweb.nwcg.gov/incident/article/3660/21586/>).

Dispersal and Home Range Movements

Information on dispersal provides important insight on how far individuals of a species may move on their own, which is valuable for understanding the potential that unoccupied but otherwise suitable habitat will be colonized or recolonized by the species without management intervention. For their body size, fishers appear to be relatively poor dispersers and large scale genetic substructure analysis supports this observation (Kyle et al. 2001). Fisher movement ecology varies by age, sex,

season, and habitat characteristics. Juvenile dispersal may vary widely, depending on habitat availability and landscape permeability.

Intensive monitoring of individual fishers by fixed-wing aircraft, in combination with an expansive trapping effort across the entire SNAMP Fisher study area provided insight on dispersal that would have been difficult to acquire otherwise. Microsatellite DNA analyses to identify maternity for many juveniles and some subadults further extended our inference to larger numbers of females ($n = 24$) and males ($n = 19$).

We found limited evidence that natal dispersal was male-biased according to any of the typical metrics reported in the literature for this life history phenomenon. Dispersal distances were not longer for males (mean = 8.46 km) compared to females (4.89 km) based on either Euclidean distances or for more realistic Least Cost movement paths (Table 29, Figs. 17). There was no difference in the proportion of each gender that dispersed, or that remained philopatric (Fig. 17, Table 29), and, similar numbers of males ($n = 5$) and females ($n = 3$) undertook long distance dispersal movements from their likely natal areas (Fig. 18). Timing of dispersal in the SNAMP Fisher study population was focused during mid-February to July, and the longest distance dispersal event a female fisher in the population undertook was 22.3 km (44.1 by the Least Cost Path), compared to 36.2 km for a male (69.8 by the Least Cost Path)(Tables 28, 29). We did document dispersal by several fishers across landscape features previously identified as restrictive based on population genetics (Tucker et al. 2012, Wisely et al. 2004). Four fishers regularly moved across the Chiquito Ridge (via Shuteye Pass), and two male fishers transitioned across the San Joaquin River canyon.

Our data on dispersal differed from reports from southern Oregon and northwestern California. Aubry and Raley (2006) reported that mean juvenile male dispersal distance was 29 km, while the mean dispersal for females was 6 km. Dispersal distance in the Hoopa area of Northern California averaged 4.0 km (range = 0.8-18.0 km) for 7 females, and was 1.3 km for one male (Matthews et al. 2013a), however the authors noted that their focus on capturing adult females limited their ability to estimate male dispersal.

The maximum known dispersal distance for fishers from the literature was 100 km (York 1996), while the maximum observed movement of a translocated individual in unoccupied habitat was 163 km (Lewis et al. 2012). The relatively limited number of long distance dispersal events noted during the six year SNAMP Fisher study suggests that long distance movements are uncommon and that the effective dispersal distance may be less than maximum dispersal capacity (Tucker et al. 2013).

Population Growth and Threats to Population Persistence

Estimates of λ for fishers derived from empirical data specific to the area of inference are rare for California, and absent for the southern Sierra Nevada. The All Year survival and empirically derived demographic rates produced a λ of 0.90 (range 0.77-1.22). While this point estimate suggests a negative growth rate, it was encouraging that the range for the all year population growth rate extended above 1.0 (Table 34). Elsewhere in California, Higley et al. (2013) integrated data on apparent survival from CMR models and data on reproduction in a series of random effects models to evaluate λ for fishers in the Hoopa Fisher Study. Two models produced λ estimates close to or greater than 1 (Both sexes, Females only; see Higley et al. 2013). Swiers (2013) used Robust Design models, software program POPAN, and Pradel models to develop information on demographic rates, population size, and population growth rates for assessing whether removal of adult fishers from a population in northern California/southern Oregon for translocation elsewhere negatively affected population growth. Swiers' (2013) top ranked Pradel model produced a population growth rate of 1.06 (95% CI = 0.97-1.15), suggesting a stable or slightly increasing population after nine 'prime breeding adult' fishers had been live-trapped and removed from the population for translocation.

We identified several sources of mortality in the study population, and indication of a possible overall negative growth rate for the population was in accordance with the fact that 60% of the 110 fishers that were radiocollared died (Table 22). The matrix model we developed was realistic and based on current knowledge of fisher life histories in California, but some demographic parameters were less well known than others. Survival of juvenile fishers during the three month period from mid-June to October is poorly known for our study, and for all other detailed studies of fishers in California (Facka et al. 2013). The estimate for juvenile female survival used in the matrix model was based on the 6-7 month period from October to March, which likely overestimated the number of juveniles recruited into the population. However, a basic sensitivity analysis indicated that the population growth rate was insensitive to variation in fertility for all age classes, and least sensitive to juvenile survival compared to subadult and adult survival.

SNAMP Fisher Management Indicators

Three management indicators we developed in 2008-09 as a mechanism for interim reporting on the status of fishers in the study area appeared useful when considered in relation to data on

population growth rates and population density. Naïve occupancy in the Key Watershed was lowest in Camera years 2007-08, 2008-09, and 2009-10 when population growth rates were negative, but then increased in the later years when the growth rate was stable or positive (Tables 33, 35). The number of resident female fishers using the Key Watersheds did not track changes in population growth rates as closely, but the proportion was lowest in Population years 4 and 5 when the growth rate was negative or at approximate stasis. Adult female survival tracked change in population growth rate closely, declining from 2-year group 1 to 3, and then increasing afterwards (Table 34). Also, population density was in decline from 2007 to 2009, but then increased during Camera years 3 (2010-11) and 4 (2011-12) (Fig. 13), coincident with improved survival among juvenile and adult female survival (Table 34). We recommend that future long term studies consider developing similar metrics as a monitoring tool, and for interim reporting to interested stakeholders.

Fisher response to fuel management

Concerns that initiation of focused management to reduce fuel levels in Sierra Nevada mixed-conifer forests to correct for 90 to 100 years of fire suppression might have negative effects on habitat use by fishers were only partly supported by results from our study. Fisher occupancy was not negatively associated with either extractive or restorative fuel reduction, though disturbances from restorative fuel reduction had a negative effect on local scale persistence. We believe that the lack of a relationship between extractive fuel reduction and occupancy by fishers was most likely due to the combination of related factors. First, the overall extent of logging in our study in the 11 years from 2002 to 2013 was likely much lower than historically, and was likely further diminished by poor market conditions for wood products when a severe recession began in 2008. Second, estimates of annual disturbance from extractive fuel reduction among occupancy survey grids was equivalent to levels known “tolerated” by fishers elsewhere in the Sierra NF (Zielinski et al. 2013). Among the 361 multi-season survey grids, 172 of them encompassed 51.9 km² of disturbance from extractive fuel reduction, representing disturbances of 2.7%/year to grids with disturbance, and 1.3%/year among all grids. Zielinski et al. (2013) investigated tolerance of fishers to forest management in the High Sierra District, Sierra NF, and reported that 14 km² patches of forest habitat with high use by fishers typically had 2.6% of the areas disturbed by forest management annually, whereas 14 km² patches of forest with low use by fishers averaged 3.5% disturbance/year. Thus, the areas of extractive fuel reduction in our study were comparable to the 2.6% disturbance in fisher high use forest patches in the High Sierra District, Sierra NF, and below some threshold of $\geq 3.5\%$ management disturbance/year that would likely cause fishers to forage elsewhere (Zielinski et al., 2013).

Our occupancy modeling supported the hypothesis that fishers would reduce their use of local patches of forest exposed to proportionally higher levels of cumulative restorative fuel reduction. Nevertheless, an important prediction from our multi-season model was that small patches of forest with 100% cumulative 5-year disturbance from mechanical mastication and reduction of understory trees and surface fuels would maintain an occupancy of 0.65. Thus, even at what would be considered a very high level of disturbance, fishers were not predicted to completely cease using those areas. For context, an occupancy rate of 0.65 for fishers elsewhere in the southern Sierra Nevada would be considered high, and a positive observation with regards long term continuation of occupancy (Zielinski et al., 2013).

Ladder fuels, surface fuels, and thick layers of duff targeted under SPLAT-based management provide important habitat for squirrels and rodents preyed on by fishers, owls, and other forest carnivores (Kelt et al., 2013). Therefore, if forest patches that were extensively treated for restorative fuel reduction harbored less abundant prey, fishers may have shifted to nearby less disturbed forest patches to forage. The possibility that thinning of trees and shrubs, and reduction in understory surface fuels (coarse woody debris) has a negative effect on rodent populations has been considered by several recent studies. Meyer et al. (2007) reported reduced captures of northern flying squirrels in forest stands that were thinned and underburned in the High Sierra District, Sierra NF. Treated stands had reduced canopy cover and relatively shallow litter depth, and Meyer et al. (2007) considered that reduced abundance of flying squirrels may have been due to reduced abundance of truffles (fruiting bodies of hypogeous fungi) when duff was removed or reduced in depth after fuel reduction. Amacher et al. (2008) reported a negative effect of fuel reduction treatments (without follow-on burning) on abundance of deer mice, a positive effect of managed burning for deer mice, but no detectable effects of thinning or burning treatments on long-eared chipmunks, California ground squirrel, or brush mouse (*Peromyscus boylei*) at a research site in the north-central Sierra Nevada. Amacher et al. (2008) suggested that scattered debris and wood shards from rotary mastication was associated with the negative treatment effect for deer mice, whereas follow-on burning removed residual woody debris and thinned the understory, thereby improving conditions for deer mice. Converse et al. (2006) reported lower density or a trend for lower density for gray-collared chipmunks (*Neotamias canipes*) and Mexican woodrats (*Neotoma mexicana*) in thinned+burned forest stands in Arizona, which was linked to reduced coarse woody debris and reduced density of shrubs. In that same study abundance of deer mice increased after thinning+burning, and there was no treatment-linked change in abundance for golden-mantled ground squirrel (*Spermophilus lateralis*) (Converse et al., 2006). In restoration-treated

ponderosa pine forests in Arizona, Lobeberger et al. (2011) found that winter season home ranges of tassel-eared squirrels (*Sciurus aberti*) disproportionately encompassed areas that had not been treated, whereas in other seasons their home ranges included a subset of the treated stands that retained relatively high canopy cover. Bull and Blumton (1999) indexed presence of small mammals from track surveys in lodgepole pine (*Pinus contorta*) and mixed-conifer forest stands treated for fuel reduction in northeastern Oregon. We were unable to identify studies that reported responses of Douglas squirrels or dusky footed woodrats (*Neotoma fuscipes*) to fuel reduction treatments, but, based on habitat associations for *Neotoma* (Innes et al., 2007; Kelt et al., 2013), understory thinning and removal of surface fuels and coarse woody debris may be problematic for woodrats (Lehmkuhl et al., 2006), whereas Douglas squirrels are a habitat generalist and less likely to be negatively impacted by fuel reduction (Coppeto et al., 2006, Herbers and Klenner, 2007; Kelt et al., 2013). Kelt et al. (2013) suggested that small mammal assemblages in the Sierra Nevada showed relatively limited responses to canopy thinning under current forest management. Abundance of small mammals in the Sierra Nevada has been linked to variation in production of cones or hard mast by pines and oaks (Coppeto et al., 2006; Wilson et al., 2008), which is important because a general pattern in many studies we reviewed was that interannual variation in abundance of small mammals was evident, and either masked or was much more important than the smaller effects introduced by fuel reduction-induced change to habitats (Converse et al., 2006; Coppeto et al., 2006; Amacher et al., 2008, Wilson et al., 2008; Kelt et al., 2013). We therefore conclude that reduced persistence of local scale habitat use by fishers in grids with larger areas treated for restorative fuel reduction was not likely to have been caused by changes in abundance of rodent prey from the associated disturbance to their habitats.

We consider it likely that the predicted 27% decline in persistence of local scale habitat use when cumulative restorative fuel reduction in a 1-km² grid approached 1.0 (100%) was associated with fishers shifting to forage in adjacent areas with less disturbance. A 27% decline in persistence of occupancy coupled with an annual colonization rate of 34%, suggests that fishers are flexible with regards local scale habitat use, and they might resume use of treated areas after several years of ecological recovery. Modeling analyses by Thompson et al. (2011) applied to a fisher occupied area of the High Sierra District, Sierra NF (Bear Fen) suggested that tree thinning (≤ 89 cm DHB) in mixed-conifer forest did not significantly reduce habitat suitability or “displace” habitat components from reference conditions in home ranges of resident female fishers. Based on these results from a nearby area in the Sierra NF, we believe it likely that fishers in our study area are likely to resume using forest patches treated for restorative fuel reduction within a few years of extensive disturbance. Also, fishers

are known to adjust space use to avoid disturbed areas within their home ranges. Garner (2013) reported that resident fishers included areas treated for extractive+restorative fuel reduction in their overall and core home ranges in proportion to availability on the overall landscape. At the finer scale of individual locations, Garner (2013) found that those same resident fishers avoided using areas within ≈ 200 m of fuel treatments. We interpret this result as consistent with ours; fishers were predicted to continue using 1-km² patches of forest with more extensive cumulative disturbance by fuel treatments, but at a reduced level compared to areas with less disturbance. Finally, our assessment of how fishers responded to forest management was at the scale of 1-km² patches of forest, which was small in relation to resident adult female (≈ 23 km²) and resident adult male home ranges in our study area (86 km²; Sweitzer, In review – SNAMP Report). If a 1-km² patch of habitat within the home range of a resident female fisher was 100% treated for fuel reduction of any type, 95.7% of that animal's home range could remain available for normal levels of foraging, contingent on SPLATs being dispersed on the landscape and not locally concentrated as appears typical (Modhaddas et al., 2010).

Integration

We found that the SPLATs at Sugar Pine slightly reduced simulated fire behavior and resulted in greater amounts of projected fisher habitat up to 30 years after the fire. In the absence of simulated fire, we found that the SPLATs had an immediate, negative effect on the amount of fisher habitat, but SPLATs did not generally have a negative effect on fisher habitat when we modeled future forest growth for 30 years. In all scenarios, the differences between the treated and untreated landscapes were small.

Our results were in general agreement with prior findings. Thompson et al. (2011) performed an analogous study to ours, in which they modeled fire and forest growth under treatment and no treatment scenarios and assessed fisher habitat suitability in the southern Sierra Nevada. They projected that fuels treatments had slight negative effects on fisher habitat in the absence of fire, but provided significant positive benefits up to 37 years after simulated fire. Truex et al. (2013) suggested that less fisher resting habitat was present immediately after mechanical fuels treatments were implemented in the Sierra Nevada. However, fishers consistently used areas in the southern Sierra Nevada where some timber harvest had occurred, so it may be possible to implement fuels-reduction treatments at an extent and rate that achieves fire-hazard-reduction goals (Zielinski et al., 2013).

As we noted in Appendix C for the California spotted owl, the net benefits of SPLATs for the Pacific fisher will depend upon the true, but unknown, probability that high-severity fire effects will occur on a given portion of the landscape. However, future probabilities for specific fire behaviors (e.g., crown-fire initiation) are difficult to estimate, and it is therefore difficult to quantify trade-offs associated with SPLATs in absolute terms (Finney 2005). We further note that the SPLATs which were implemented at Sugar Pine appeared to have relatively modest impacts on forest structure and simulated fire behavior, and that it may be necessary to evaluate additional SPLATs of different intensities over a larger scale to fully assess the effects of SPLATs on fisher habitat. Nonetheless, we have no reason to believe that Forest Service managers should alter their current policy of avoiding the placement of SPLATs near known fisher denning sites (U.S. Forest Service 2004) because these sites have significant biological importance for this species.

Management Implications of Findings from SNAMP Fisher

Fishers have been the focus of systematic monitoring in the southern Sierra Nevada by track plates, hair snares, and camera traps since the mid-1990s (Truex et al. 1998, Zielinski et al. 2005, Jordan 2007). Analyses of baited track plate detection histories from 2002 to 2009 for the entire southern Sierra Nevada fisher population found no evidence that the population trajectory for fishers in the area has been significantly positive or negative, based on constant and positive persistent values (Zielinski et al. 2013). In contrast, Tucker et al. (2014) suggested that the fisher population in the SNAMP Fisher study area was produced by a significant post-1900s northward population expansion involving dispersal of animals from south of the Kings River (Fisher Core Habitat Area 4; Fig. 2). Tucker et al. (2014) reported evidence of ‘strong genetic clustering’ to the north of Little Shuteye Peak (part of a high elevation ridge that forms the east boundary of Subregion 2 in our study area; Fig. 6), which, along with evidence for other small genetic clusters, was suggestive of multiple founder events associated with contemporary population expansion. Data from track-plate surveys in the Sierra National Forest in the early 1990s rarely detected fishers (Zielinski et al. 1995, 2005), which suggested a very sparse population in the SNAMP Fisher study area (Fisher Core Habitat area 5; Table 2, Fig. 2), compared to the more recent surveys in 2002-2009 (Tucker et al. 2014). Tucker et al (2014) postulated that very few fishers were present in the SNAMP Fisher study area prior to the 1990s, and that an expansion that occurred only during the last 20-25 years produced the population in this region.

Genetic data are not typically used to make inferences about population processes operating over extremely short periods in evolutionary time. The genetic analyses of Tucker et al. (2014), and the large increase in fisher detections in the region encompassing our entire study area between the early 1990s and 2002-2009 (Zielinski et al. 2013), suggest that a significantly positive population growth rate would be a requirement for understanding the current distribution and abundance of fishers in the SNAMP Fisher study area. During the period from 2007 to 2014, our results suggest that the fisher population in this region has not been experiencing consistently positive or significant population growth (Table 24).

The suggestion of an overall negative population growth rate, the low density, and the relatively small estimated number of fishers in Fisher Core Habitat area 5 ($n = 93$, range 80-107), warrants concern for the long term viability of fishers in the region. Any small population will be at high risk to stochastic events such as disease and large perturbations to critical habitats (e.g. forest fires or drought; Noss et al. 2006), and genetic limitation resulting from genetic drift after founder events (Tucker et al. 2014) will hinder population recovery and expansion (Reed et al. 2003). Minimum viable population size has been under debate (Shoemaker et al. 2013, Reed and McCoy 2014), but at <500 total individuals (Spencer et al. 2004), the current southern Sierra Nevada fisher population will likely require active management and conservation measures to maintain a positive growth rate across its entire range. The observed variation in fisher abundance and rates of population growth in the SNAMP Fisher study area (Table 4) reaffirms the vulnerability of the small, isolated population to external threats (Spencer et al. 2014), especially wildfires that are likely to increase in frequency and intensity with climate change (Bonan 2008, Safford et al. 2012). Moreover, our study spanned a limited period of six years when multiple threats to fisher survival within the study area were identified and during which three large wildfires further isolated the population by significantly reducing the availability of suitable habitat immediately to the south and north of the study site. We recommend continuous monitoring of the status of fisher populations in the southern Sierra Nevada region. It will be necessary to mitigate for major threats to fisher survival while maintaining contiguous expanses of suitable fisher habitats, and detailed analyses using realistic and empirically developed data on population parameters are necessary for evaluating the long-term viability of fishers in the southern Sierra Nevada. Data developed from the SNAMP Fisher study have provided important new insights on the status of a fisher population at the northern margin of their current distribution in the southern Sierra Nevada Range, which will be useful towards developing a comprehensive conservation strategy for fishers in California.

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