

Southern Sierra Nevada Fisher Conservation Assessment

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Abbreviations, Acronyms, and Definitions

Canopy closure	Proportion of the sky hemisphere obscured by vegetation when viewed from a single point on the ground
Canopy cover	Percentage of ground covered by a vertical projection of the tree canopy
CBI	Conservation Biology Institute
CDFW	California Department of Fish and Wildlife
Core (habitat core)	Large ($>15 \text{ mi}^2$; 40 km^2) contiguous areas of fisher habitat within which fishers can establish home ranges and comeingle as a population, generally separated by unsuitable habitat areas.
CWHR	California Wildlife Habitat Relationships system
DBH	Diameter at breast height
FIALT	Fisher Interagency Leadership Team
Fire severity	Measure of the amount of tree biomass lost to a fire
FTT	Fisher Technical Team
Hoopa	Hoopa Valley Indian Reservation in northwestern California
KRFP	Kings River Fisher Project
Maxent	A maximum-entropy algorithm used for species distribution modeling
NPS	National Park Service
PSW	Pacific Southwest Research Station of the US Forest Service
SNAMP	Sierra Nevada Adaptive Management Project
SSN	Southern Sierra Nevada
USFS	USDA Forest Service
USFWS	US Fish and Wildlife Service

Preface

The Southern Sierra Nevada Fisher Conservation Strategy (SSNFCS) is a multi-agency effort to develop a scientifically sound approach for sustaining and recovering an isolated population of Pacific fisher (*Pekania pennanti*) in the southern Sierra Nevada. It tiers to a broader interagency effort to develop a conservation assessment and strategy for fishers throughout the Pacific states and southern British Columbia (the Fisher West Coast Assessment: Lofroth et al. 2010, 2011; Naney et al. 2012). The West Coast Assessment and Strategy were developed in response to a 2004 finding by the US Fish and Wildlife Service (USFWS) that listing the west coast fisher distinct population segment¹ under the Endangered Species Act was “warranted but precluded by higher priority actions . . .” While the 3 volumes of the West Coast Assessment are published and available, the draft West Coast Strategy was never finalized nor adopted by the participating agencies, and the development of more localized strategies was encouraged. The fisher is currently (January 2015) proposed for listing under the US Endangered Species Act and is a candidate for listing under the California Endangered Species Act.

The small size and isolation of the southern Sierra Nevada fisher population, its association with forests experiencing increasingly severe fire conditions, and uncertainties about how fires, other disturbances, climate change, and forest management may impact fishers and their habitat, precipitated the need for a collaborative, interagency conservation planning effort in the southern Sierra Nevada. In late 2012, a Fisher Inter-Agency Leadership Team (FIALT) was formed to initiate and support development of the SSNFCS, with the intent to build on accomplishments of the West Coast Strategy. The FIALT—comprising executive representatives of the Sierra Nevada Conservancy, USDA Forest Service, USFWS, National Park Service, and California Department of Fish and Wildlife (CDFW)—ensures that the effort has sufficient resources and that the outcomes are tailored to meet agency needs.

The FIALT selected Dr. Wayne Spencer of Conservation Biology Institute (CBI) to assemble and direct a Fisher Technical Team (FTT) comprising leading local experts in fisher biology, forest ecology, and related topics in the southern Sierra Nevada. The FTT is responsible for analyzing, synthesizing, and documenting the available science and developing a conservation strategy in consultation with other experts and agency representatives. In particular, the FTT consults with an extended group of Special Topic Advisors, who provide additional expertise on such related topics as silviculture, fire ecology, land management, and fisher mortality factors. The FTT is developing 3 products: (1) this Fisher Conservation Assessment document, (2) a Fisher Conservation Strategy document, and (3) a Decision Support System to aid implementation of the Strategy. Participating agencies and landowners may choose to adopt all or portions of these products for their use, but are not obligated to do so.

The FIALT designated a Core Support Team to serve as an interface between the FIALT, FTT, Special Topic Advisors, and other interested entities. The Core Support Team provides logistical and administrative support and facilitates stakeholder involvement and public outreach through various communications channels, including meetings, webinars, emails, and a website. These avenues

¹ Under the federal Endangered Species Act, a Distinct Population Segment is a population of a species or subspecies that is both discrete (geographically or biologically) from others and significant relative to the species as a whole.

provide opportunities for stakeholders and the public to provide feedback on the process and products, and to pose or answer relevant questions.

This Conservation Assessment document summarizes the current state of knowledge about fishers and fisher habitat in the southern Sierra Nevada, building on the copious information already summarized for the West Coast Assessment (Lofroth et al. 2010, 2011; Naney et al. 2012), but with specific focus on the southern Sierra Nevada. In addition to synthesizing published literature and agency reports, the Assessment summarizes abundant new scientific information from recent fisher studies and habitat modeling efforts in California. As of this writing (January 2015), much of this new content has not yet been published in the peer-reviewed literature; consequently, this Assessment was subjected to independent scientific peer review by 5 experts on fishers and forest ecology, and revised accordingly.

Acknowledgments

Funding for this project was provided to the Conservation Biology Institute (CBI) by Sierra Nevada Conservancy, Region 5 of the USDA Forest Service, Resources Legacy Fund, and the USFWS. In addition to acknowledging contributions of members of the FIALT and FTT, we provide special thanks for the following contributions: University of California-Berkeley, University of California-Davis, and Pacific Southwest Research Station (PSW) allowed use of unpublished data and analyses; Heather Romsos of CBI performed the spatial mapping and modeling as well as many statistical analyses; Jerre Ann Stallcup (CBI) and Joe Drennan (Garcia and Associates) assisted with technical editing and meeting facilitation; Greg Schroer (Sierra National Forest) also assisted with meeting arrangements and provided input on the process. This document was greatly improved based on comments provided by 5 peer reviewers—Reginald Barrett, Jeffrey Lewis, Kevin McKelvey, Robert Naney, and Mark Schwartz—as well as by numerous stakeholders and agency representatives, including Greta Wengert, Keith Slauson, Joe Sherlock, Marc Meyer, Chad Hanson, Rebecca Green, Mourad Gabriel, Steven Brink, and Justin Augustine.

1 Introduction

1.1 Context

This Conservation Assessment summarizes information on fishers (*Pekania [Martes] pennanti*²) in the southern Sierra Nevada pertinent to developing a Fisher Conservation Strategy for the region. It builds on information already summarized in documents prepared by an Interagency Fisher Biology Team for the West Coast Fisher Conservation Assessment and Strategy (Lofroth et al. 2010, 2011; Naney et al. 2012), which covers fishers and fisher habitat in the Pacific states (California, Oregon, and Washington) and southern British Columbia. This document repeats some key information from the West Coast Assessment, but focuses primarily on new information and additional details concerning fishers and fisher habitat from recent and ongoing research in California, especially the southern Sierra Nevada.

Fishers are medium-sized carnivores that inhabit dense, mixed-coniferous temperate and boreal forests throughout North America (Powell 1993). The fisher of the Pacific states, or the West Coast Distinct Population Segment³, is proposed for listing under the federal Endangered Species Act and has been the target of recovery and conservation efforts (Lewis et al. 2012). The fisher is also a candidate for listing under the California Endangered Species Act pursuant to a 2013 court order (*Center for Biological Diversity v California Fish and Game Commission*; Cal Sup Ct CGC-10-50520) that forced the California Fish and Game Commission to set aside its 2010 findings that listing was not warranted. The CDFW is reviewing the fisher's status in the state pursuant to the court order.

In California, fishers occur in 2 separate populations, 1 in the north coastal and Klamath region (including a recently translocated population in the southern Cascades and northern Sierra Nevada) and 1 in the southern Sierra Nevada (Figure 1). Genetic results suggest that these populations, separated by about 400 kilometers (km) or 250 miles (mi), have been disjunct for thousands of years and that the southern population is genetically distinct (Knaus et al. 2011; Tucker et al. 2012). While genetic data cannot be used to estimate the size of the historical distribution gap, the magnitude of genetic differentiation suggests that it was probably quite wide (J. Tucker, US Forest Service [USFS], personal communication). However, it is clear from historical records that the gap was once narrower than it is now: reliable observations and museum specimens documented fishers north of the Merced River to the central Sierra Nevada during the 19th and early 20th centuries (Price 1894; Grinnell et al. 1937; Chow 2009).

The southern Sierra Nevada population is well-studied by a variety of monitoring and research studies (Table 1, Figure 1), which together reveal that the population is small (at most a few hundred individuals), stable (neither expanding nor contracting in recent years, following range contraction during the 19th and 20th centuries), and at risk of further reduction or extirpation from an array of mortality agents (Section 3.1.3), habitat changes, and demographic stochasticity.

² The fisher, formerly included in the genus *Martes* (*Martes pennanti*), is now recognized, on the basis of new genetic analyses, as comprising a new genus (*Pekania pennanti*; Koepfli et al. 2008; Sato et al. 2012).

³ Under the federal Endangered Species Act, a Distinct Population Segment is a population of a species or subspecies that is both discrete (geographically or biologically) from others and significant relative to the species as a whole.

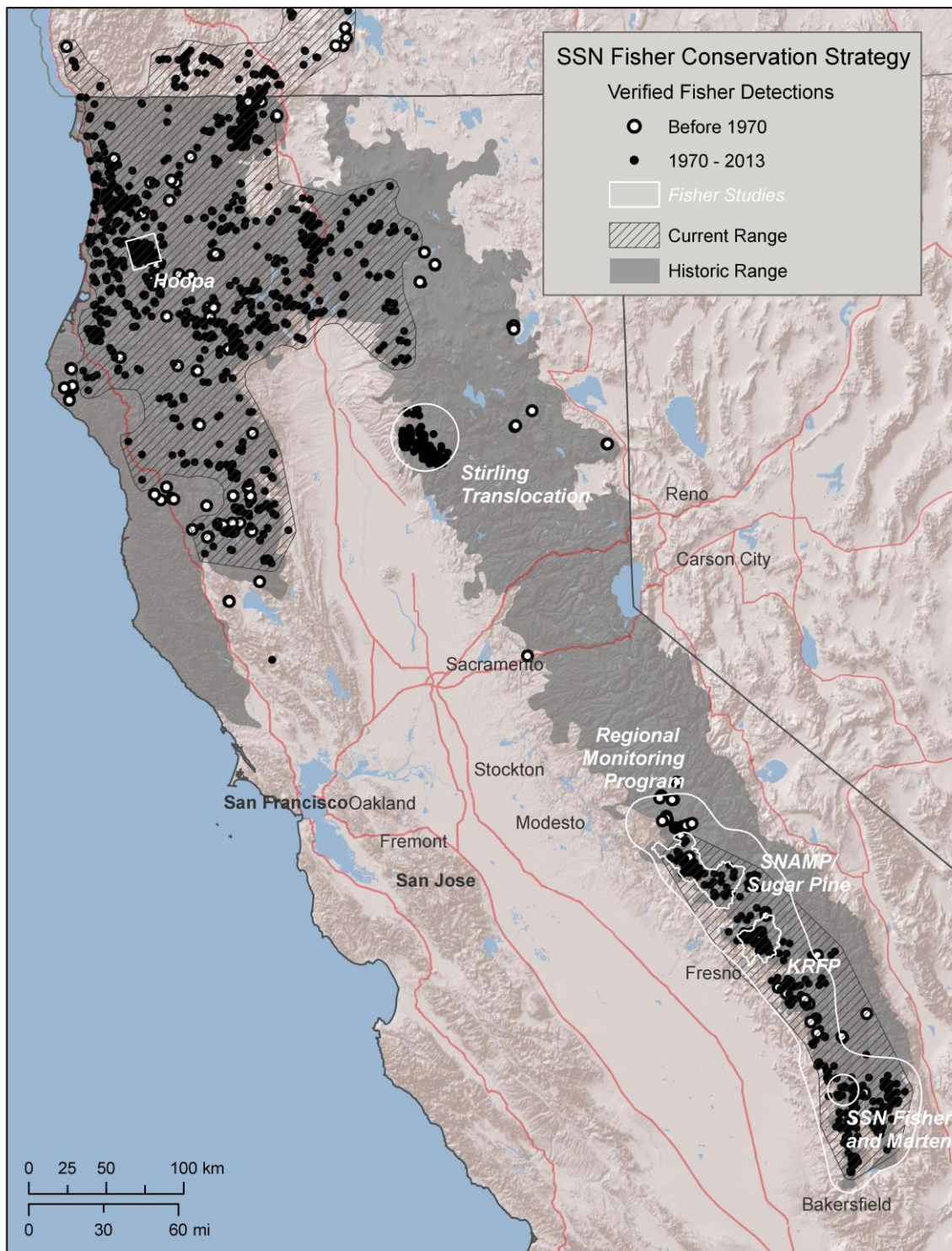


Figure 1. Fisher detections, current and presumed historical range, and location of major fisher field studies in California. Fisher detections include only records considered highly reliable by USFWS (unpublished data). Current distribution is a minimum convex polygon enclosing post-1970 fisher observations, buffered by 5 km (3 mi); historical distribution is the California Wildlife Habitat Relationships (CWHR) range for fisher based on Grinnell et al. (1937). See Table 1 for names and descriptions of fisher studies.

Table 1. Recent and ongoing fisher studies in California and abbreviations used in text.

Study name	Abbreviation in text	Location	Dates	Focus	Notes
Sugar Pine Fisher Project	Sugar Pine	Bass Lake Ranger District, Sierra National Forest (NF)	2014-2016	Post-treatment monitoring of the SNAMP fuel reduction projects	Continues some studies initiated by SNAMP
CDFW Fisher Translocation Project	Stirling translocation	Stirling Mgmt Area of Sierra Pacific Industries, Butte/Tehama counties (northern Sierra Nevada/southern Cascades)	2009-present	Monitoring study of introduced fisher population: reproduction, habitat use, space use, translocation methods	Outside assessment area
Kings River Fisher Project	KRFP	High Sierra Ranger District, Sierra NF	2007-present	Comprehensive field study of fisher population biology: life history, space use, habitat, responses to vegetation management	
Sierra Nevada Adaptive Management Project	SNAMP	Bass Lake Ranger District, Sierra NF	2007-2013	Comprehensive field study of fisher population biology: life history, space use, habitat, responses to vegetation management	Completed as of Dec. 31, 2013; some studies continuing as Sugar Pine
Hoopa Valley Fisher Study	Hoopa	Hoopa Valley Indian Reservation, Humboldt County	2004-present	Comprehensive field study of fisher population biology: life history, space use, habitat, responses to vegetation management	Outside assessment area
USFS Pacific Southwest Region (5) Fisher Regional Monitoring Program	Regional monitoring program	National Forests in the southern Sierra Nevada: Stanislaus, Sierra, Sequoia, Inyo	2002-present	Landscape-level occupancy monitoring using non-invasive techniques	
Southern Sierra Fisher and Marten Study	SSN fisher and marten study	Tule River Ranger District of Sequoia NF, Mountain Home Demonstration State Forest, and Tule River Indian Reservation	1994-1996	Comparative study of marten and fisher home range and habitat characteristics, diet, and interspecific competition	

1.2 Objectives and Scope

This Conservation Assessment summarizes information about fishers in the southern Sierra Nevada, including population size, distribution, and trends; ecology; habitat requirements across multiple spatial scales (from the population or landscape scale to the scale of individual denning or resting structures); and threats to fishers and their habitat. It is a focused review of information most relevant to crafting a strategy to conserve the population in the southern Sierra Nevada.

The geographic scope of this assessment (the assessment area) is the Sierra Nevada south of the Mokelumne River and mainly west of the Sierra Nevada crest (Figure 2). Although the breeding fisher population is currently restricted to forested areas south of the Merced River (Yosemite Valley), the assessment also applies to lands north of the Merced that were once inhabited by fishers and could be again (e.g., on the Stanislaus National Forest). Because the Conservation Assessment and Conservation Strategy are intended to cover all lands that might contribute to fisher conservation, both public and private, the geographic scope was delineated primarily using watersheds (excluding elevations below 152 m [500 ft]) rather than administrative boundaries. However, the northern boundary was adjusted to coincide with the northern boundary of the Stanislaus National Forest to facilitate forest planning efforts.

This assessment provides the biological and ecological foundations for a Southern Sierra Nevada Fisher Conservation Strategy and a Decision Support System to help implement that Strategy. Management implications and recommendations derived from this document will be described in the Conservation Strategy. To address threats to the southern Sierra Nevada fisher population, the Strategy must balance competing and uncertain risks of various actions (or lack of actions) and be informed by the outcomes using an adaptive management approach.

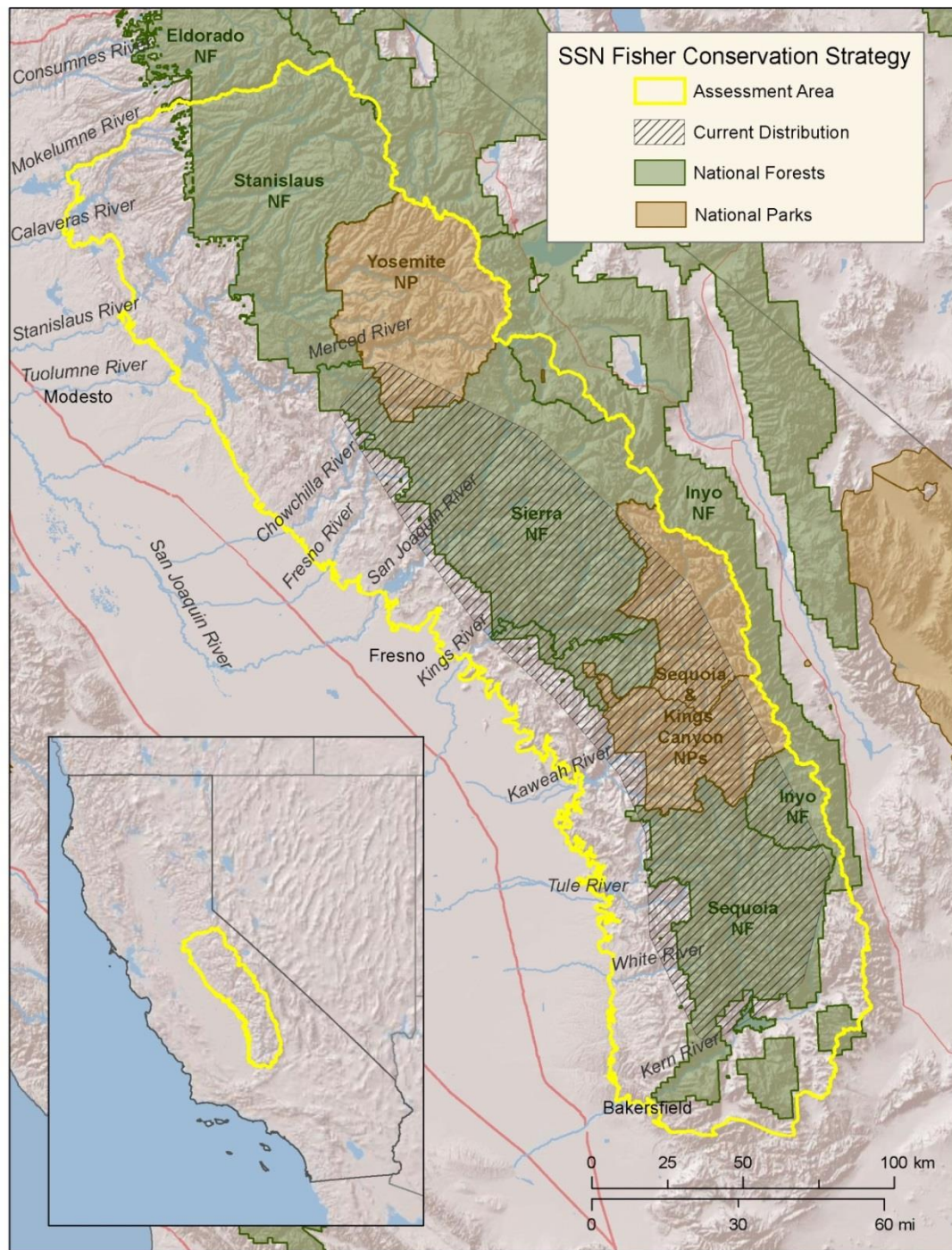


Figure 2. Southern Sierra Nevada Fisher Conservation Assessment Area. Delineated using watershed sub-basins clipped at the 152-m (500-ft) elevation contour (California Interagency Watershed Mapping Committee 1999) and extended north to the Mokelumne River (border of Stanislaus National Forest). Current fisher distribution approximated as on Figure 1.

2 Population Distribution and Trends

2.1 Range Size and Trends

Lofroth et al. (2010) estimated that the current range of the fisher in California represents <50% of the historical range as described by Grinnell et al. (1937), who considered the range in the late 19th and early 20th centuries to include 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 (~38,600-42,500 mi²). Currently, fishers are absent from most of the northern and central Sierra Nevada, leaving a ~400-km (250-mi) gap separating 2 remaining populations, 1 in the northern Coast Range and Klamath Province (including a translocated population in the northern Sierra Nevada/southern Cascades) and 1 in the southern Sierra Nevada (Zielinski et al. 1995). Recent genetic analysis suggests that these populations were separated prior to European settlement in California (Knaus et al. 2011; Tucker et al. 2012), but historical records are not clear about the width of the gap and extent of occupation in the northern and central Sierra Nevada. Grinnell et al. (1930) reported that fishers were sometimes trapped near Eagle Lake in the Mount Lassen region, but localities there were excluded without explanation by Grinnell et al. (1937), who mapped localities in the Sierra Nevada as far north as Sierra County based on trapper reports from 1919 through 1924. Exact locations of some of those records are also uncertain, as Grinnell et al. (1937) noted that some localities might indicate the “residence or post office of trapper” rather than actual capture location. Price (1894) reported that a fisher was observed on Mount Tallac, southwest of Lake Tahoe, noting that they prefer “the high wooded ridges of the west slope of the Sierra >4,000 ft (1,219 m).”

This assessment used an updated fisher locality database compiled by USFWS to estimate the fisher’s historical (prior to 1970) and recent (1970-2013) range in California (Figure 1) and to evaluate the likely extent of the historical range gap in the Sierra Nevada. Only fisher observations considered highly reliable⁴ were used, thus removing some questionable accounts mentioned above. We used the California Wildlife Habitat Relationships (CWHR) program range map, which is based on Grinnell et al. (1937), to estimate a potential historical range size of 102,431 km² (39,549 mi²). We used a minimum convex polygon around reliable fisher records (buffered by 5 km [3 mi]) from 1970 to 2013 to estimate a recent range size of 52,465 km² (20,257 mi²). Of this, 12,865 km² (4,967 mi²) are in the southern Sierra Nevada, and 39,600 km² (15,290 mi²) are in northern California. This represents a ~50% reduction from the presumed historical range—similar to the Lofroth et al (2010) estimate. However, we suspect that the historical range depicted in Figure 1 overestimates how much of the northern and eastern portions of California’s inland mountains were occupied historically by fishers.

Regardless of these uncertainties in mapping, the fisher was once more widely distributed north of the Merced River (Price 1894; Grinnell et al. 1937; Zielinski et al. 2005; Chow 2009) and has experienced a substantial reduction in total range size within California since European settlement. Genetic analyses and survey data indicate that the range was even smaller during the 20th century than today and that the range expanded northward from south of the Kings River to the Merced River in recent decades (Tucker et al. 2014). Based on reliable records prior to 1970, the southern edge of the

⁴ Based on recommendations of Aubry and Jagger (2006), USFWS considered records highly reliable if supported by physical evidence (e.g., museum specimens, captures, photographs) or, in the absence of physical evidence, observations from reliable sources of fishers “trapped or treed by dogs and released.”

range gap was probably at least as far north as Placerville in El Dorado County, ~140 km (87 mi) north of its current location⁵.

From 2009 to 2011, 40 fishers were translocated from the Klamath region onto the “Stirling Tract” owned by Sierra Pacific Industries in Butte and Tehama counties, in the northern portion of the gap in fisher distribution (Facka and Powell 2010; Figure 1). Home range establishment and breeding have been documented in this translocated population. As this newly established population is well outside the southern Sierra Nevada assessment area, it is not part of the Conservation Assessment and Conservation Strategy.

2.2 Elevation Distribution

Throughout their range, fishers are associated with low to mid-elevation forests (Lofroth et al. 2010). In the assessment area, fishers have been detected from ~1,000 m (3,280 ft) to 3,134 m (10,280 ft, Laymon et al. 1991; Boroski et al. 2002; Mazzoni 2002; Jordan et al. 2005, 2007; Green 2007; Purcell et al. 2012). However, the population is concentrated within a narrower elevation band of mixed-conifer forest, which varies in elevation and width with latitude and climate. In the Sierra Nevada Adaptive Management Project (SNAMP) study area (Sierra National Forest), at the northern end of occupied fisher habitat, fisher detections are concentrated between ~1,220 and 2,140 m (4,000-7,000 ft) (Figure 3, R. Sweitzer, Great Basin Institute, unpublished data). On the Sequoia National Forest farther south, fishers have been detected over a broader and generally higher elevation range (~1,220-2,740 m [4,000-9,000 ft], with a peak around 1,830-2,140 m [6,000-7,000 ft], Figure 4). Most of the highest elevation detections (>2,440 m [8,000 ft]) were on the Kern Plateau (southeastern portion of Sequoia National Forest), which accumulates less snow than other portions of the assessment area at similar elevations. The regional monitoring program has not detected martens (*Martes caurina*) on the Kern Plateau, where fishers were detected at elevations more typical of marten occupancy (J. Tucker, Forest Service [FS] Region 5, unpublished data). Reduced snow accumulation on the Kern Plateau may allow fishers to exploit higher elevation areas there than elsewhere and may not favor occupancy by martens, which are highly snow-adapted (Krohn et al. 1995, 1997).

2.3 Population Size and Trends

Although there has not been a definitive census, the current southern Sierra Nevada fisher population almost certainly numbers <500 total individuals (Spencer et al. 2011) and probably <300 adult fishers and has been stable over the past decade based on occupancy estimates from the regional monitoring program (Zielinski et al. 2013a). Regardless of the precise size, populations of a few hundred individuals, with only a small proportion of breeding-age females, are at elevated risk of extirpation due to stochastic events.

Occupancy was estimated separately for 3 zones: the northwestern (west slope of Sierra National Forest), the southwestern (west slope of Giant Sequoia National Monument and Sequoia National Forest), and the southeastern (Kern Plateau). The overall probability of occupancy is 0.367, with the lowest estimates on the Kern Plateau (0.261) and the highest in the southwestern zone (0.583, Zielinski et al. 2013a). There was no detectable change in occupancy from 2002 to 2009 for the entire assessment area or for any individual zone (Zielinski et al. 2013a). However, genetic patterns and

⁵ See historical locality east of Sacramento on Figure 1, which represents 5 fishers killed for their pelts near Placerville during July 1916 (Anonymous 1917).

survey data strongly suggest that the population expanded into areas north of the Kings River during the 1990s, before the regional monitoring program was established (Tucker et al. 2014).

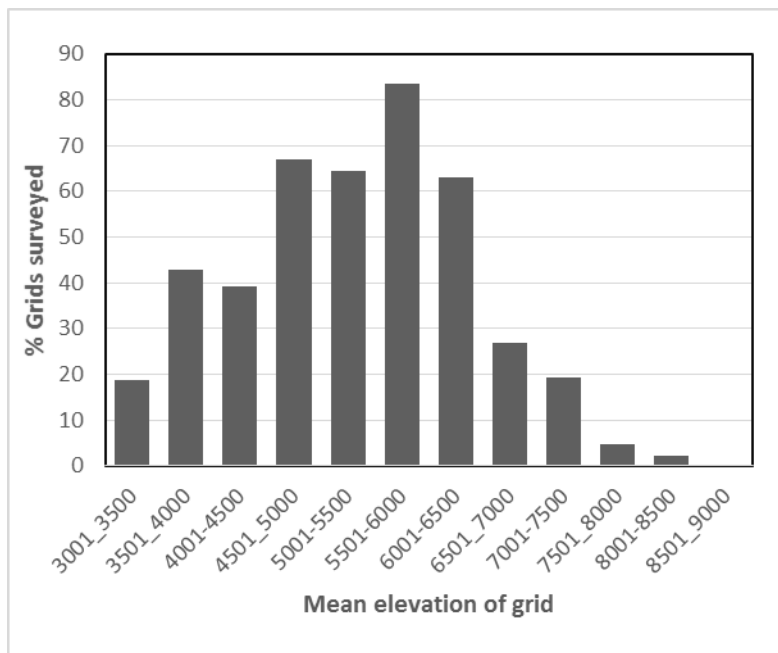


Figure 3. Percent fisher detection rate at survey grids by elevation in the SNAMP area. Source: R. Sweitzer, unpublished data from the SNAMP fisher study, Oct. 2007—Oct. 2011.

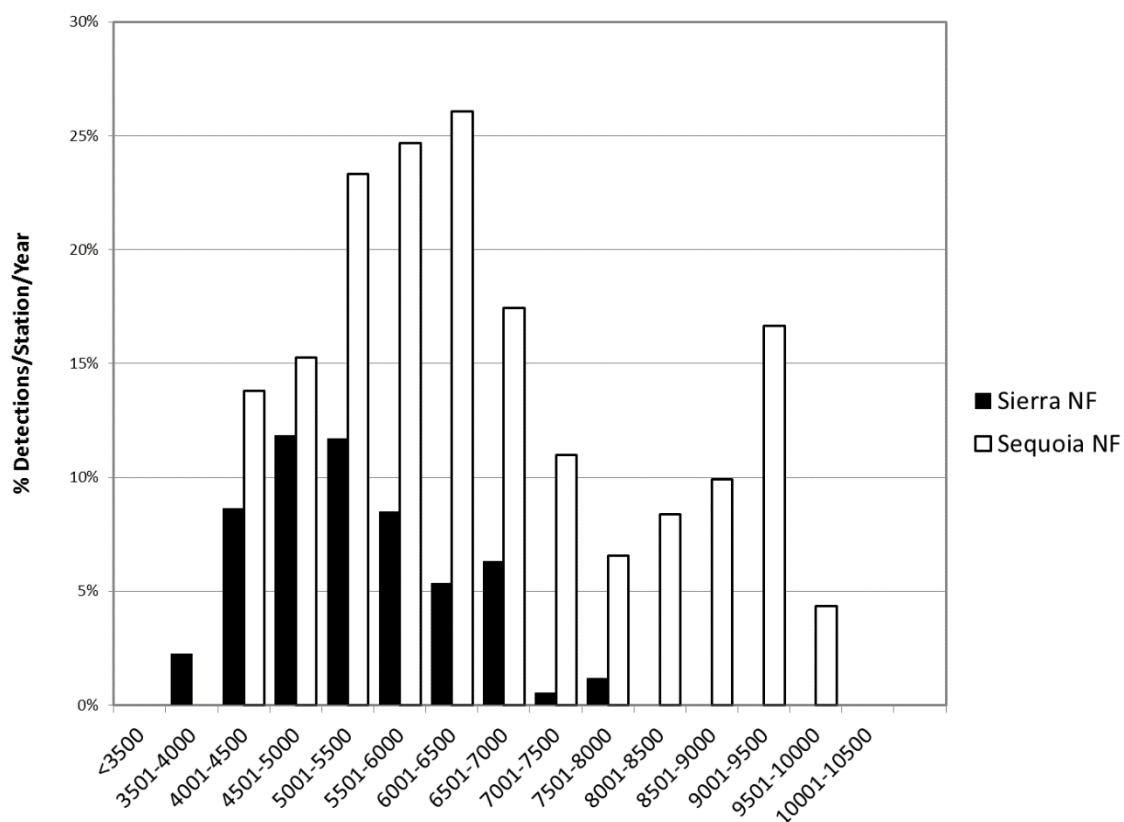


Figure 4. Percent fisher detection rate at survey stations by elevation, 2002-2012 on Sierra and Sequoia National Forests. Source: J. Tucker, unpublished data from the regional monitoring program. Most detections >2,440 m (8,000 ft) are on the Kern Plateau.

Using scat-detector dog data and simulation methods, Thompson et al. (2012) estimated there are 80 ± 33 (SD) fishers in the 650-km^2 (250-mi^2) Kings River Fisher Project (KRFP) study area. The density estimates from this approach were considered biased, however, due to skew in the posterior distribution from Markov chain Monte Carlo simulations. Consequently, the authors recommended using the mode of $0.104\text{ fishers/km}^2$ ($0.269\text{ fishers/mi}^2$) as an unbiased point estimate of fisher density. This corresponds with Jordan's (2007) previous density estimate using more traditional capture-mark-recapture methods: $0.095\text{-}0.134\text{ fishers/km}^2$ ($\sim 0.246\text{-}0.347\text{ fishers/mi}^2$). Applying the Thompson et al. (2012) modal density estimate of $0.104\text{ fishers/km}^2$ provides a population estimate of ~ 68 fishers in the KRFP study area.

In the nearby SNAMP study area, R. Sweitzer et al. (unpublished data) used camera and trap re-sightings to estimate a mean annual fisher density of $0.072\text{-}0.093\text{ fishers/km}^2$ ($0.187\text{-}0.241\text{ fishers/mi}^2$) and a mean annual population size of 48-62 fishers from 2008 to 2012.

Although regional occupancy trends (Zielinski et al. 2013a) suggest that the southern Sierra Nevada fisher population is relatively stable, records from elsewhere show that fisher densities can change rapidly. In the Hoopa Valley fisher study area in northern California, density was estimated at 0.52 fishers/km^2 ($\sim 1.35\text{ fishers/mi}^2$) in 1998, but fell to 0.14 fishers/km^2 in 2005 ($\sim 0.36\text{ fishers/mi}^2$, Matthews et al. 2013).

Spencer et al. (2011) and Zielinski et al. (2013a) noted that the southern Sierra Nevada population is not expanding geographically, despite the apparent existence of suitable, unoccupied habitat north of the Merced River. This could be due to physical impediments to movement (landscape features that inhibit dispersal or increase risks to dispersing fishers, such as roads, rivers, and open areas), low numbers of dispersers from occupied habitats, or a combination of these factors (Spencer et al. 2011; Tucker 2013; Thompson et al. 2014).

2.4 Genetic Diversity and Population Subdivision

The southern Sierra Nevada fisher population has low genetic diversity in both mitochondrial and nuclear DNA, is genetically distinct from other populations, and shows evidence of population subdivision by dispersal impediments (Knaus et al. 2011; Tucker et al. 2012, 2014). The genetic evidence suggests that the southern Sierra Nevada population became isolated from other populations and experienced a $\sim 90\%$ decline in effective population size thousands of years ago, such that the gap in California's fisher distribution may reflect Ice Age events (Knaus et al. 2011; Tucker et al. 2012; Thompson et al. 2014). Knaus et al. (2011) found the southern Sierra Nevada population to be fixed for a single mitochondrial DNA haplotype that differs from the next most closely related haplotype in northwestern California by 9 base-pair substitutions—a divergence that probably occurred thousands of years ago. A recent analysis added 209 new genetic samples from the 2 California fisher populations and reinforced the Knaus et al. (2011) findings: the southern Sierra Nevada fisher population is fixed for a unique haplotype not found in northwestern California (K. Pilgrim, USFS, unpublished data; J. Tucker, personal communication).

A more recent, though less pronounced population bottleneck also likely occurred in the northern and central portions of the Sierra Nevada in the late 19th and early 20th centuries (Tucker et al. 2012). Genetic patterns suggest that the population contracted southward to the southern tip of the Sierra Nevada following European settlement, but re-expanded northward to the Merced River in recent decades (Tucker et al. 2012, 2014). Genetic analyses cannot reveal how wide the distribution gap was in historical or prehistoric times, so the extent of fisher distribution in the northern and central Sierra Nevada prior to European settlement remains unknown. The southernmost tip of the Sierra Nevada

(e.g., south of the Tule River watershed) may have served as a refuge for fishers during the post-European contraction, possibly due to the rugged terrain and lesser impacts from logging, mining, and trapping that began with the gold rush (Beesley 1996, Tucker et al. 2012).

The current southern Sierra Nevada population shows evidence of within-population genetic differentiation, with 3 genetic subpopulations separated at the Kings River and Tule River watersheds, in or near the Mountain Home Demonstration State Forest (Tucker et al. 2012, 2014). The Kings River Canyon appears to be a moderately strong dispersal impediment (i.e., a filter but not a complete barrier to fisher movement), due to steep, largely unforested slopes. The genetic subdivision at the Tule River watershed does not yet have an obvious explanation and may be an artifact of population contraction and re-expansion across this area during the past century (J. Tucker, personal communication). There is also some evidence of weaker subdivisions within the northern subpopulation, suggesting there may be dispersal impediments associated with Little Shuteye Peak, San Joaquin River, and Kaiser Wilderness (Tucker et al. 2014). The genetic patterns north of the Kings River are consistent with the hypothesis that the fisher population expanded northward into this area in recent decades, as also suggested by survey data. During the 1990s, surveys routinely detected fishers south of the Kings River, but rarely north of it (Zielinski et al. 1995, 2005), whereas more recent monitoring data (2002 to present) show fishers well-established north of the Kings River, as far north as the Merced River (Zielinski et al. 2013a).

Genetic isolation findings (Knaus et al. 2011; Tucker et al. 2012, 2014) suggest that attempting to restore connectivity between the southern Sierra Nevada and northern California populations is not a reasonable conservation goal, and that the 2 populations should be managed independently to maintain local adaptations, as long as the southern Sierra Nevada population remains independently viable. However, as the distribution gap was once narrower than it is now, and with potential but unoccupied habitat north of the Merced River (Spencer et al. 2011), expanding the size and distribution of the southern Sierra Nevada population northward across the Merced River into these areas would increase long-term viability of the population (Spencer et al. 2011; Carroll et al. 2012).

3 Fisher Biology and Ecology

3.1 Life History

3.1.1 Reproduction

Fishers are polygynous and altricial, and males do not aid in rearing of young (Powell 1993; Aubry et al. 2013). Fishers have relatively low reproductive rates, because litters are small and it takes 2 or more years to reach reproductive maturity; this can slow population recovery following perturbations. Females may first mate at 12 months of age, but delayed implantation of the blastocyst leads to first litter production at 2 years of age (Hodgson 1937; Wright 1963; Powell 1993; Frost et al. 1997). Females older than 2 years are more likely to den and wean more offspring, probably due to increased experience (Matthews et al. 2013, R. Sweitzer et al., unpublished data). Male fishers compete for mating access to females, and most males do not reproduce until they attain sufficient body size to compete effectively, at 3-4 years of age (Wright and Coulter 1967; Lewis et al. 2012).

Population reproductive rates depend on the proportion of females giving birth, mean litter size, and the survival of kits to weaning age. Not all reproductive females give birth every year. Denning rates (proportion of females giving birth) in northwestern California and southern Oregon vary from ~51 to 87% (Aubry and Raley 2006; Higley and Matthews 2006; Matthews et al. 2013). In the SNAMP area, the mean annual denning rate (uncorrected for deaths during the denning season) was 85% (Table 2, R. Sweitzer, unpublished data). Litter size varies from 1 to 4, with means for west coast fishers ranging from 1.6 to 2.8 (Aubry and Raley 2006; Higley and Matthews 2006; Lofroth et al. 2010; Green et al. 2013). Mean litter size for fishers in the KRFP area is 1.6 (annual range 1.3-1.9); however, not all fishers in the litter survive to weaning age. One study found that the rate of successful weaning in northern California fishers was 65% (Matthews et al. 2013) of adult females. The mean weaning rate in the SNAMP area is 74% (annual range 65-82%), and the mean number of weaned kits per litter is 1.4 (annual range 1.2-1.6, R. Sweitzer, unpublished data).

Table 2. Annual reproductive rates in the KRFP and SNAMP study areas from 6 denning seasons (spring 2008-2013).

Parameter/year	KRFP			SNAMP		
	Mean	SD	Range	Mean	SD	Range
Adult females (#) ^a	13.2	4.4	6-19	15.5	2.3	11-17
Adult females denning (%) ^b	86	8	77-100	85	4	79-88
Adult females weaning (%)				74	6	64-100
Mean kits/denning female	1.6	0.2	1.3-1.9	1.6	0.2	1.3-1.8
Total kits produced	17	5.4	9-22	14.4	4.0	10-20
Mean weaned kits/female ^c				1.4	0.2	1.2-1.7

^aIncludes adult females captured after the denning season for which reproductive status could be assessed by measuring teats (Matthews et al. 2013)

^bMaximum known reproductive rate; does not correct for females that did not survive the denning season or that ceased denning behavior before June of each year

^cNot counting kits that did not survive to weaning

Fishers follow a fairly regular seasonal schedule of reproductive events. The following description is based on the Lofroth et al. (2010) review, with dates provided for the southern Sierra Nevada population based on data from the KRFP study from 2008 to 2013 (R. Green, UC Davis and KRFP,

unpublished data). Although there may be some geographic and annual variation in these dates, they are generally representative of the reproductive schedule throughout the assessment area.

Females give birth in natal dens in late March to early April (recorded dates at KRFP: Mar. 23-Apr. 11). Males visit females' dens to mate within about 10 days following parturition (recorded dates: Mar. 29-May 6, with a peak ~Apr. 6). Mothers move their young from the natal den to a maternal den in the weeks following birth (recorded dates: Apr. 6-Jun. 10) and may move them successively to other maternal dens throughout the spring and summer. Use of successional maternal dens may reduce exposure to predators that learn den locations, help accommodate kit growth by selecting larger cavities, avoid effects of prey depletion near the den, or possibly reduce exposure to feces and parasites that may accumulate in dens, as has been demonstrated in some burrowing mammals (D. Clifford, CDFW, personal communication). Kits are weaned after ~10 weeks (typically in late May), but the mother continues feeding them prey as they grow and become more mobile throughout the summer. Kits are mobile enough to travel with their mothers and kill live prey beginning ~4 months of age (late Jul.-early Aug.), are independent by ~7 months (Sep.-Oct.), and disperse and establish new home ranges starting at ~10 months (early Feb.) (Lofroth et al. 2010; R. Green, unpublished data; R. Sweitzer, unpublished data).

3.1.2 Survivorship

The lifespan of wild fishers is ~10 years (Powell 1993). On the Sierra National Forest, 1 female fisher lived 11 years, but the typical age at death is 6-7 years (R. Green, unpublished data). Annual survivorship estimates in the assessment area have ranged from ~0.61 (for females only, Truex et al. 1998) to ~0.94 (Jordan et al. 2011). The latter estimate was based on camera detections rather than radio-collared individuals, so the values have low precision due to tag loss and other factors. At the SNAMP and KRFP sites, which provide a longer and more detailed record of demographic rates than previous studies, annual survival appears higher for females (0.69-0.77) than males (0.60-0.66). Adult (females 0.76, males 0.65) and juvenile (female 0.77, male 0.66) survival is higher than that of subadults (females 0.69, males 0.60, Table 3), with ages defined as juvenile <12 months, subadults 12-24 months, and adults >24 months. While the data suggest that juvenile survival is similar to that for adult fishers (Table 3), this is likely an artifact of monitoring juvenile survival only after the first 6 months of life, when they are too small for radio collars. Table 3 therefore probably overestimates juvenile survival (R. Sweitzer, Great Basin Institute, and C. Thompson, PSW, personal communications).

Survival of radio-collared fishers in the SNAMP and KRFP studies is lower from March to August than September to February. Higher mortality from March through August, when prey are abundant and diverse, may be due to increased predation risk during spring and summer (G. Wengert, Integral Ecology Research Center, personal communication), exposure to anticoagulant rodenticides and other toxins around marijuana grow sites during the spring growing season (Gabriel et al. 2013; Thompson et al. 2014), increased physical stress on females while rearing kits (R. Green, UC Davis and KRFP, personal communication), or some combination of these factors.

3.1.3 Mortality causes

Documented sources of mortality for fishers in western North America include predation, disease, accidents, roadkill, and other human-caused factors (Lofroth et al. 2010). Contribution of "natural" mortality sources like disease and predation make up ~50-91% of all fisher mortality, and impacts vary by sex and age (Truex et al. 1998; Lofroth et al. 2010; R. Sweitzer et al. unpublished data). R. Sweitzer et al. (unpublished data) evaluated the deaths of 98 radio-collared fishers at the SNAMP

and KRFP sites from May 2007 to March 2013. Ultimate cause of death was determined for 81 of these, with predation being the primary cause (79%), followed by disease and injury/starvation (6.2% each), roadkill (3.7%), rodenticide poisoning (2.5%), starvation alone (1.2%), and entrapment in a human structure (1.2%) (Table 4, Gabriel 2013; Wengert 2013; R. Sweitzer et al. unpublished data). However, this analysis could not account for potential interacting factors (e.g., rodenticide poisoning) increasing predation risk, and growing evidence suggests that widespread exposure to rodenticides is compromising fisher health, possibly contributing to the high rate of predation observed as the ultimate cause of death (Section 3.1.3.3).

Table 3. Estimates of annual survival $s(t)$ for fishers at the SNAMP and KRFP sites. Based on Kaplan-Meier staggered entry analyses with data pooled by week, Mar. 18, 2007-Mar. 17, 2014.

Age group	SNAMP		KRFP		Sites combined	
	$s(t)$	95% CI	$s(t)$	95% CI	$s(t)$	95% CI
Juvenile females ^a	0.79	0.65-0.93	0.87	0.76-0.99	0.83	0.74-0.92
Juvenile males ^a	0.72 ^b	0.54-0.89	0.80	0.67-0.94	0.76	0.65-0.87
Subadult females	0.72	0.59-0.86	0.65	0.51-0.79	0.69	0.60-0.79
Subadult males	0.62	0.46-0.78	0.74	0.56-0.91	0.69	0.57-0.81
Adult females	0.72	0.62-0.82	0.72	0.64-0.81	0.72	0.64-0.81
Adult males	0.62	0.52-0.73	0.66	0.52-0.81	0.64	0.55-0.72

^aBased on a 20-week period (Oct. 18—Mar. 17)

Source: R. Sweitzer et al. unpublished data

Table 4. Cause-specific mortalities identified for fishers from SNAMP and KRFP, May 2007-Mar. 2013

Cause	SNAMP	KRFP	Total (%)
Predation	32	32	64 (79)
Canine distemper	4	1	5 (6.2)
Starvation or injury-induced ^a	4	1	5 (6.2)
Roadkill	3	-	3 (3.7)
Rodenticide toxicosis	2	-	2 (2.5)
Starvation	1	-	1 (1.2)
Human structures ^b	-	1	1 (1.2)
Unknown ^c	2	9	11
Pending	6	-	6
Total known cause	46	35	81 (100)

^aAnimals for which necropsies found evidence suggesting that injuries or wounding may have prevented foraging and contributed to starvation, and/or resulted in bacterial infection (septicemia), leading to death.

^bCombined field and necropsy data indicated this animal became entrapped in a PVC air sampling tube and subsequently died by starvation/dehydration.

^cAnimals recovered in decomposed/desiccated condition, or animals for which neither necropsy or DNA forensics could detect evidence definitively linked to death.

Source: R. Sweitzer et al. unpublished data.

3.1.3.1 Predation

In the southern Sierra Nevada, bobcats (*Lynx rufus*), mountain lions (*Puma concolor*), and coyotes (*Canis latrans*) prey on fishers, based on pathology and DNA forensics (Wengert 2013; Wengert et al. 2014). Felids are associated with >80% of all known or suspected predation deaths on SNAMP and KRFP from 2007 to March 2013 (N=50) (Wengert 2013). One fisher died after an encounter with a rattlesnake (*Crotalus viridis*; C. Thompson, unpublished data). Across California, predation accounts for 73% of female mortality and 45% of male mortality (Wengert et al. 2014). In the southern Sierra Nevada, bobcats and lions each account for 23% of all mortality, with bobcats killing female fishers and lions mostly responsible for killing males, probably due to the large body size differences of the sexes (Wengert et al. 2014). Bobcats and mountain lions have been detected with remote cameras at the bases of fisher den structures while mother and kits were known to be present (R. Sweitzer and C. Thompson, personal communications). At the SNAMP site, the carcass remains of 2 denning females were found within a few hundred meters of their den trees, and forensic analyses determined that both had been killed by bobcats. In 1 of those cases, images from remote cameras focused on the den tree included a bobcat with a fisher kit in its mouth, with the mother's carcass on the ground within a few meters of the tree.

3.1.3.2 Disease and infections

Viral, bacterial, and protozoal diseases, nutritional deficiency, and cancer have caused deaths in California fishers (Gabriel 2013). Necropsy shows that disease-caused mortalities in southern Sierra Nevada fishers include canine distemper, bacterial infections (some with interstitial pneumonia), and concurrent infection with the protozoal parasite *Toxoplasma gondii* and urinary tract blockage, leading to emaciation due to presumed malnutrition (Gabriel 2013). Four disease-associated deaths on the Sierra National Forest were linked to active infection with canine distemper virus, and another 5 to injury-related septicemia (Table 4; Keller et al. 2012; Gabriel 2013). One fisher on the SNAMP site died of complications after parasitic infection by *Toxoplasma gondii* (Gabriel 2013). Although exposure to *Toxoplasma gondii* was documented previously in California fishers (Brown et al. 2006) and elsewhere in North America (Larkin et al. 2011), this was the first case where complications from toxoplasmosis resulted in death.

Five fishers succumbed to various bacterial infections or starvation due to puncture wounds or other injury (Table 4). Death by bacterial infection or starvation after suffering wounds (1 fisher that died of septicemia had wounds consistent with failed predation) or debilitating injury (dislocated jaw and broken mandible of 1 fisher at the SNAMP site) is not surprising for animals as active as fishers. Other long-term studies of radio-collared fishers have reported similar circumstances (Aubry and Raley 2006; Weir and Corbould 2008). For more information, see Section 6.4.

3.1.3.3 Rodenticides and other pesticides

It is increasingly apparent that anticoagulant rodenticides (and other pesticides) represent a serious threat to fishers and other species in the southern Sierra Nevada and in California in general. A fisher death from rodenticide poisoning on the SNAMP study site during spring 2009 prompted wider testing for exposure to rodenticides among fishers in California and elsewhere using archived tissue samples (Gabriel et al. 2012a, b). Results from 3 long-term demographic studies (Hoopa, KRFP, SNAMP) indicate that >80% of fishers are being exposed to rodenticides and other toxins broadcast around illegal marijuana grow sites on California public and tribal lands (Gabriel et al. 2012a, b, 2013). Thompson et al. (2013) demonstrated that survival rates of female fishers are inversely correlated with the number of known marijuana grow sites in their home ranges in the KRFP area.

Of necropsied fishers, 91% at SNAMP and 85% at KRFP showed exposure to rodenticides (M. Gabriel, Integral Ecology Research Center, UC Davis, personal communication). Seven fishers in California have been documented as succumbing to rodenticide-linked toxicosis (Gabriel et al. 2013; Higley et al. 2013). Two fisher deaths on the Sierra National Forest between 2007 and mid-2013 were caused by toxicosis resulting in hemorrhaging after exposure to rodenticides (Gabriel et al. 2012a, b). In addition to direct mortalities, rodenticide exposure may increase mortality rates indirectly by inhibiting fishers' abilities to survive blood loss from minor injuries or an attempted predation event, or reducing its ability to carry out routine behaviors like predator avoidance, thermoregulation, and prey capture (Erickson and Urban 2004; Gabriel et al. 2012a, b; Thompson et al. 2014). Rodenticide exposure may also lower reproductive success (Mackintosh et al. 1988; Greaves 1993; Munday and Thompson 2003).

In addition to rodenticides, fishers are exposed to other toxicants at illegal marijuana sites, including over-the-counter insecticides and herbicides as well as compounds banned in the US. Not only are fishers exposed to these compounds through secondary poisoning, but they often are targeted directly through the use of poisoned bait. Consequences of exposure include reduced reflex time, increased susceptibility to disease and pathogens, reduced thermoregulatory capacity, and death. For more information on toxicants, see Section 6.2.

3.1.3.4 Roads and other human structures

Collisions with vehicles and entrapment in pipes or water tanks are also sources of fisher mortalities in the assessment area (Figure 5). Eight percent of necropsied fishers in the southern Sierra Nevada between 2007 and 2012 died from vehicular strikes (Gabriel 2013). There have been 10 documented fisher roadkill mortalities in Yosemite National Park over the past 2 decades (Figure 5, Table 5). Highway 41/Wawona Road, between the park boundary near Fish Camp and Yosemite Valley, is a hotspot for fisher roadkill: vehicles killed 4 fishers between 1992 and 2004 (Chow 2009) and 3 radio-collared fishers and 6 non-collared fishers between 2007 and 2013 (R. Sweitzer et al., unpublished data). Although no radio-collared fishers at the KRFP site were killed by vehicles between 2007 and 2013, 1 non-collared adult male was killed on Dinkey Creek Road. One road mortality occurred in Sequoia-Kings Canyon National Parks, between The Four Guardsmen and Commissary Curve landmarks in February 2011 (D. Gammons, Sequoia and Kings Canyon National Parks, personal communication).

Fishers also have died due to accidental entrapment in structures such as pipes and water tanks (Section 6.5). One male died in an air sampling tube on the KRFP site, and a female died in an open water tank at the SNAMP site. Truex et al. (1998) and Powell et al. (2012) both reported deaths of single radio-collared females in abandoned water tanks at research sites in north-central California, and Folliard (1994) recovered the skeletal remains of 8 fishers from an abandoned water tank on private timberlands in northwestern California. In the Cariboo-Chilcotin region of British Columbia, Canada, L. Davis (NPS, unpublished data) reported a dead radio-collared fisher that maneuvered itself into a relatively short section of an upright culvert.

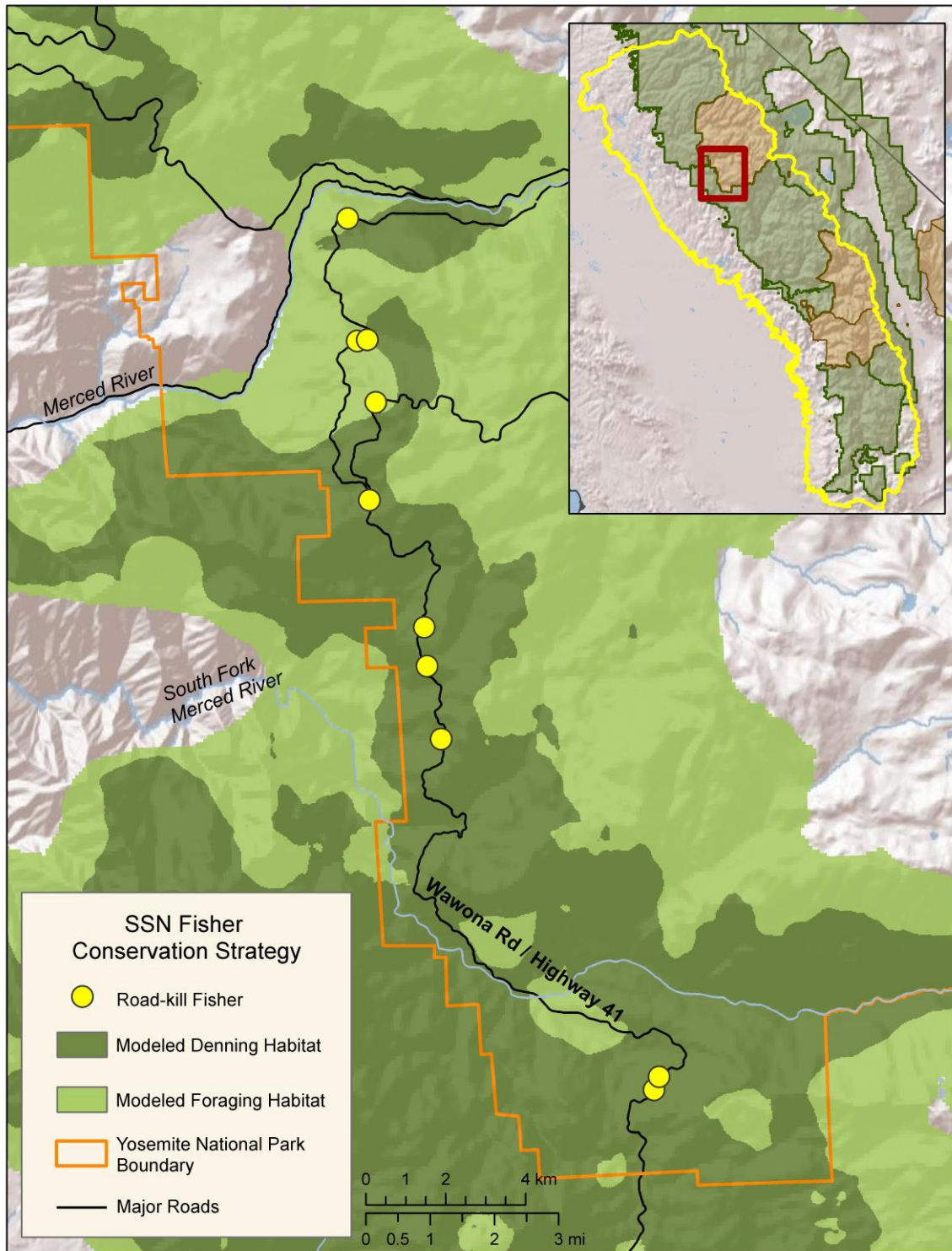


Figure 5. Road-killed fishers along Highway 41/Wawona Road, 2007-2013, in relation to modeled fisher foraging and denning habitat. See Appendix A for methods.

Table 5. Roadkill fisher history and anticoagulant exposure (AR) in Yosemite National Park, 1993-2012.

Date	Location	Sex	AR Exposure
May1993	Wawona Road at Indian Creek	Female	Not tested
Jun. 1994	Glacier Point Road at El Portal View	Unknown	Not tested
Dec. 1997	Wawona Road at Bishop Creek	Male	Not tested
Jun. 2000	Wawona Road at Grouse Creek	Unknown	Not tested
May 2008	1.4 mi north of South Entrance, Wawona Rd.	Female (lactating)	Positive
May 2009	2.5 mi south of Alder Creek, Wawona Rd	Male	Positive
Jun. 2009	south of Wawona Tunnel, Wawona Rd.	Female (lactating)	Positive
Oct. 2009	~1mi north of South Entrance, Wawona Rd.	Unknown ^a	Not tested
Jun. 2010	Wawona Road at Grouse Creek	Female	Positive
Apr. 2012	Wawona Road north of Bishop Creek	Male ^b	Pending

^aCarcass not collected; estimated location^bRadio-collared individual

Source: L. Cline, NPS, unpublished data

3.2 Ecology

3.2.1 Space use patterns

Understanding animal space-use is important for understanding population interactions and therefore conservation priorities. Individual fisher locality and movement data from the numerous fishers that have been monitored using radio-telemetry and GPS in the SNAMP and KRFP studies provide useful information on fisher space-use patterns in the assessment area.

3.2.1.1 Home ranges and home range core use areas

Fishers have large home ranges for their body size, and males' home ranges are larger than females' home ranges (Table 6). Adjacent home ranges may overlap significantly, especially between sexes, but individuals tend to have relatively exclusive intra-sexual home range core use areas, which may or may not represent defended territories (Powell 1993; Zielinski et al. 2004a). A core use area is a portion of the home range where the animal spends a disproportionate amount of time (Samuel et al. 1985). Core use areas within home ranges are sometimes arbitrarily defined as the 50% home range, but can also be determined analytically (Bingham and Noon 1997). Neither overall home ranges nor within-home range core use areas are fixed spatially—they may shift on the landscape with seasons and other changes as a fisher learns about resource distributions, neighbors, etc., and updates its “cognitive map” of the landscape (Spencer 2012).

On the Sequoia National Forest, Zielinski et al. (2004a) estimated mean 100% minimum convex polygon home range sizes of 5.275 km² (1,302.5 ac) for 8 females and 29.984 km² (7,409.2 ac) for 4 males. Due to the dynamic nature of home ranges, however, they are best delineated using a kernel-based utilization distribution (Worton 1989) of an animal's sample locations (usually from radio-telemetry or GPS techniques) over a given time period, such as a year or season. This assessment defines fisher home range area using the 95% fixed-kernel isopleth (Worton 1989)—i.e., enclosing 95% of an individual's location points during a year—and home range core use area using a fixed-kernel isopleth that minimizes intra-sexual overlap. Defining core use areas that minimize overlap between adjacent females is important for some spatially explicit population models such as PATCH

(Schumaker 1998), which simulates population processes for breeding-age females on exclusive breeding territories (Spencer et al. 2011). Annual home ranges overlap extensively among neighboring females, but overlap declines significantly using 70% or 60% fixed-kernel isopleths (Table 6, R. Sweitzer, unpublished data). These data suggest that females maintain exclusive intra-sexual territories within core use areas, within which they spend about ~60-70% of their time (C. Thompson, unpublished data, R. Sweitzer, unpublished data).

Table 6. Mean annual and core use home range sizes (km² and ac)^a for radio-tracked fishers at the SNAMP site, Dec. 2007-Mar. 2013.

Age/Sex	N	Mean annual \pm SE km ² (ac) ^b	Mean core use \pm SE km ² (ac) ^c
<i>Juvenile (<12 months)^d</i>			
Female	10	20.98 \pm 3.76 (5,184.26 \pm 929.11)	6.59 \pm 1.18 (1,628.42 \pm 291.58)
Male	4	35.68 \pm 3.83 (8,816.71 \pm 946.41)	11.86 \pm 1.02 (2,930.67 \pm 252.05)
<i>Subadult (12- 24 months)</i>			
Female	22	25.15 \pm 3.20 (6,214.69 \pm 790.74)	8.59 \pm 1.09 (2,122.63 \pm 269.34)
Male	18	51.85 \pm 4.76 (12,812.39 \pm 1,176.22)	18.15 \pm 1.66 (4,484.96 \pm 410.19)
<i>Adult (>24 months)</i>			
Female	56	22.93 \pm 1.36 (5,666.12 \pm 336.06)	7.78 \pm 0.59 (1,922.48 \pm 145.79)
Male	40	86.18 \pm 4.87 (21,295.51 \pm 1203.40)	30.23 \pm 1.78 (7,469.98 \pm 439.85)

^aEstimated using Home Range Tools in ArcGIS 9.3.1 and smoothed using the ad hoc bandwidth selection procedure in Kie et al. (2010).

^bAnnual home ranges based on the 95% fixed-kernel models estimated for fishers for which locations were available for ≥ 6 months Apr. 1-Mar. 31 (population year); number of location records used to estimate the annual and core use home models ranged from 77 to 326.

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

^dHome ranges for juvenile fishers monitored during ≥ 5 months of the Oct.-Mar. period; excludes home ranges for fishers that exhibited dispersal movement behavior.

Source: R. Sweitzer, unpublished data

Home ranges measured on the KRFP area are substantially smaller than those on the SNAMP area, which may reflect differences in methods (ground-based versus aerial telemetry) or differences in habitat quality or population density between the 2 areas. In the KRFP study, annual home ranges of adult males averaged 26.35 \pm 18.7 km² (6,510 \pm 4,620 ac), and annual home ranges of adult females averaged 10.96 \pm 6.37 km² (2,780 \pm 1,574 ac) (Thompson et al. 2010, 2013). In the SNAMP study, annual home ranges for adult males and females averaged 86.18 km² (21,300 ac) and 22.93 km² (5,670 ac), respectively (R. Sweitzer, unpublished data, Table 6). The aerial telemetry methods on SNAMP obtain more frequent locality data than is possible with ground-based telemetry and are more likely to detect fishers after long-range movements or in areas difficult to monitor from the ground.

Adult males move widely during the breeding season, resulting in widely overlapping use areas during spring (Popescu et al. 2014). In the southern Sierra Nevada, juveniles of both sexes disperse ~7-8 months of age. Once territories are established, female fishers exhibit high levels of site fidelity (Tucker 2013; R. Sweitzer and C. Thompson, unpublished data). Adult female home ranges are smallest during the spring, and reproducing females have smaller home ranges than non-reproducing females during spring and summer (Table 7) when mothers remain near the den and their dependent young. Seasonal home ranges of adult male fishers are smallest during the summer and largest during the spring, reflecting wide movement associated with mating during March and April (Table 7).

Table 7. Mean size and standard error (km² and ac) of fisher home ranges (95% fixed kernel)^a during each season^b for animals that were radio-tracked by SNAMP, Dec. 2007–Mar. 2013.

Age	Spring		Summer		Fall		Winter	
	N	Mean \pm SE km ² (ac)	N	Mean \pm SE km ² (ac)	N	Mean \pm SE km ² (ac)	N	Mean \pm SE km ² (ac)
<i>Juvenile^c</i>								
Female					11	16.24 \pm 2.52 (4012.99 \pm 622.70)	17	18.72 \pm 2.46 (4625.81 \pm 607.88)
Male					4	20.10 \pm 3.51 (4966.81 \pm 867.34)	9	48.92 \pm 12.73 (12088.38 \pm 3145.65)
<i>Subadult</i>								
Female	17	20.78 \pm 3.59 (5134.84 \pm 887.11)	21	17.19 \pm 2.73 (4247.73 \pm 674.60)	21	15.61 \pm 1.41 (3857.31 \pm 348.42)	22	22.87 \pm 2.28 (5651.29 \pm 563.40)
Male	12	36.48 \pm 4.26 (9014.39 \pm 1052.67)	13	30.49 \pm 3.31 (7534.23 \pm 817.92)	14	33.51 \pm 3.15 (8280.49 \pm 778.38)	19	58.90 \pm 8.61 (14554.48 \pm 2127.57)
<i>Adult</i>								
Female (denning)	59	8.18 \pm 0.64 (2021.32 \pm 158.15)	43	14.92 \pm 1.02 (3686.81 \pm 252.05)	50	19.70 \pm 1.37 (4867.97 \pm 338.53)	50	21.77 \pm 1.28 (5379.48 \pm 316.29)
Female (non- denning)	11	15.03 \pm 1.64 (3713.99 \pm 405.25)	8	12.48 \pm 1.47 (3083.87 \pm 363.24)				
Male	35	72.07 \pm 6.39 (17808.86 \pm 1579.00)	34	39.49 \pm 2.75 (9758.18 \pm 679.54)	32	49.25 \pm 4.04 (12169.92 \pm 998.30)	37	68.91 \pm 4.73 (17028.01 \pm 1168.81)

^aHome range models developed using Home Range Tools in ArcGIS 9.3.1 and smoothed using the ad hoc bandwidth selection procedure in Kie et al. (2010); seasonal home range models only developed if ≥ 25 location records were available during the season.

^bSpring: Mar. 21 – Jun. 20; Summer: Jun. 21 – Sep. 20; Fall: Sep. 21 – Dec. 20; Winter: Dec. 21 – Mar. 20

^cSeasonal home ranges not presented for juvenile and subadult fishers during seasons when dispersal-related exploratory movements occurred.

In contrast, seasonal home ranges of subadult males (likely non-reproductive) are largest during winter and relatively stable during spring, summer, and fall (Table 7). Excluding the spring season home range for adult males, home range size is largest for all age and sex classes in winter (Table 7), likely due to relative scarcity of prey.

3.2.1.2 Dispersal

Dispersal—when animals depart their natal areas and establish independent home ranges where they may eventually mate and produce offspring—is an important process for most vertebrates (Sweitzer and Berger 1998). 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. They also can contribute to losses of genetic diversity if individuals cannot disperse between subpopulations to breed.

For their size, fishers are relatively poor dispersers. The maximum known dispersal distance is 135 km (84 mi, Weir and Corbould 2008), but such long-distance movements are rare, and dispersal distances average much less (Lofroth et al. 2010). On the SNAMP site, 20 juvenile females dispersed an average of 4.9 km (3.04 mi), and 15 juvenile males dispersed an average of 6.93 km (4.41 mi, Table 8). One male dispersed ~36 km (22.4 mi) from the KRFP site (Core 4) to the SNAMP area (Core 5) in late winter 2013, representing the longest recorded dispersal in the southern Sierra Nevada and the only documented dispersal event between habitat cores (R. Sweitzer, unpublished data). Dispersal distances generally represent distances between established home ranges, while this might not best describe genetic connectivity (J. Tucker, personal communication). Larger movements of males during the breeding season that result in successful mating, but not establishment of a new home range, better depict genetic connectivity, but data on such movements are difficult to obtain.

Table 8. Minimal straight-line dispersal^b distances (km and mi) for fishers at the SNAMP site, Oct. 2008-Dec. 2013.

Sex	Sample size	Mean \pm SE km (mi)	Range km (mi)
Females	20	4.89 \pm 1.36 (3.04 \pm 0.86)	0.24-22.26 (0.15-13.83)
Males ^a	17	8.48 \pm 2.39 (5.27 \pm 1.48)	0.94-36.17 (0.58-22.48)

^aIncludes ~36 km (22 mi) dispersal by a male fisher that moved north from the KRFP area into the SNAMP study area during 2013

^bMeasured between centroids of 95% adaptive kernel home ranges before and after dispersal events.

Source: R. Sweitzer, unpublished data

Landscape genetic patterns demonstrate that female dispersal is more limited than male dispersal (Tucker 2013) and suggest that female movements between core habitat areas are rare. This is consistent with movement data from the SNAMP and KRFP radio-tracking and GPS studies, which have not recorded female dispersal between cores; all dispersal events described by R. Sweitzer (unpublished data) have been within Core 5, and most female dispersers establish new home ranges adjacent to their natal range. Eleven of 20 juvenile females (55%) and 6 of 15 juvenile males (40%) showed no or very limited dispersal movements, but rather they established adult territories near their natal home ranges (R. Sweitzer, unpublished data). Regional monitoring data also suggest that females generally don't move as far as males: 12 of the 13 individuals detected at more than 1 sample

unit (spaced ~4 km [2.5 mi] apart) between 2006 and 2012 were males (Tucker et al. 2014; J. Tucker, personal communication).

Female dispersal seems more limited by water bodies and roads, and is more consistently associated with dense forest cover and large trees, than male dispersal (Tucker 2013); females appear to disperse primarily within high quality fisher habitat, generally remaining within core habitat areas with dense forest and large trees. Males, in contrast, appear more tolerant of a range of landscape features and are more likely than females to disperse between core habitat areas through less suitable habitat conditions (Tucker 2013). This evidence suggests that most fishers occasionally observed north of the Merced River are likely dispersing males, and that improving connectivity for females could, in theory, help establish a breeding population north of the Merced River (Cores 6 and 7).

Although dispersal can occur in any season, most dispersal events during the SNAMP study occurred in late winter to mid spring (Table 9).

Table 9. Seasonality of juvenile and subadult dispersal events on the SNAMP site, Oct. 2008-Dec. 2013. Dispersal defined by period of transition from a distinct natal home range to a distinct adult home range.

Period	Female	Male	Total (%)
Fall to mid-winter: Oct. 15–Feb. 4	2	3	5 (20.8)
Late winter to mid-spring: Feb. 5–May 5	7	7	14 (58.3)
Late spring and summer: after May 5	2	5	5 (20.8)

Dispersal is generally considered a risky period for mammals, because young, naïve animals are exposed to predators and other dangers, food deprivation, and intraspecific competition when moving through unfamiliar habitats and establishing new home ranges (Chepko-Sade and Halpin 1987). However, all but 1 of the 19 juvenile or subadults that dispersed away from their natal areas during the SNAMP study survived the transition. One subadult female fisher was killed by a bobcat in May 2009 during her “transitional movement” phase of dispersal (R. Sweitzer, unpublished data).

Likely dispersal impediments, or filters, associated with the Yosemite Valley (steep, granitic slopes and cliffs, open vegetation, the Merced River, and roads and associated traffic) may be inhibiting northward expansion of the population across the Merced River (Spencer et al. 2011; Carroll et al. 2012; Spencer and Rustigian-Romsos 2012a). Tucker (2013) found that major roads and rivers impede gene flow of females, and steep slopes impede gene flow of males, although the influence of these landscape features on gene flow is much greater for females than for males. Juvenile and subadult males appear to explore before settling on a home range, and males that may disperse north of the Merced River but fail to find any females will likely return in search of females (R. Green, personal communication). High mortality rates in the occupied regions just south of the Merced River may further reduce fisher colonization potential by limiting the number of potential dispersers in the area (Spencer et al. 2011; Carroll et al. 2012; Spencer and Rustigian-Romsos 2012a).

3.2.2 Activity patterns

Although fishers were once thought to be nocturnal (Coulter 1966), they can be active nearly any time of day, with a tendency toward crepuscular (morning and evening) activity peaks. Activity patterns vary with sex, season, and other factors (Zielinski 2000). Fishers in the SNAMP study exhibit an “expanded crepuscular” activity pattern year-round (Figure 6, R. Sweitzer and C. O’Brien,

unpublished data). Activity peaks do not appear closely associated with sunrise or sunset; rather, fishers appear to avoid activity during the warmest or brightest period of the day, especially during summer (Figure 6).

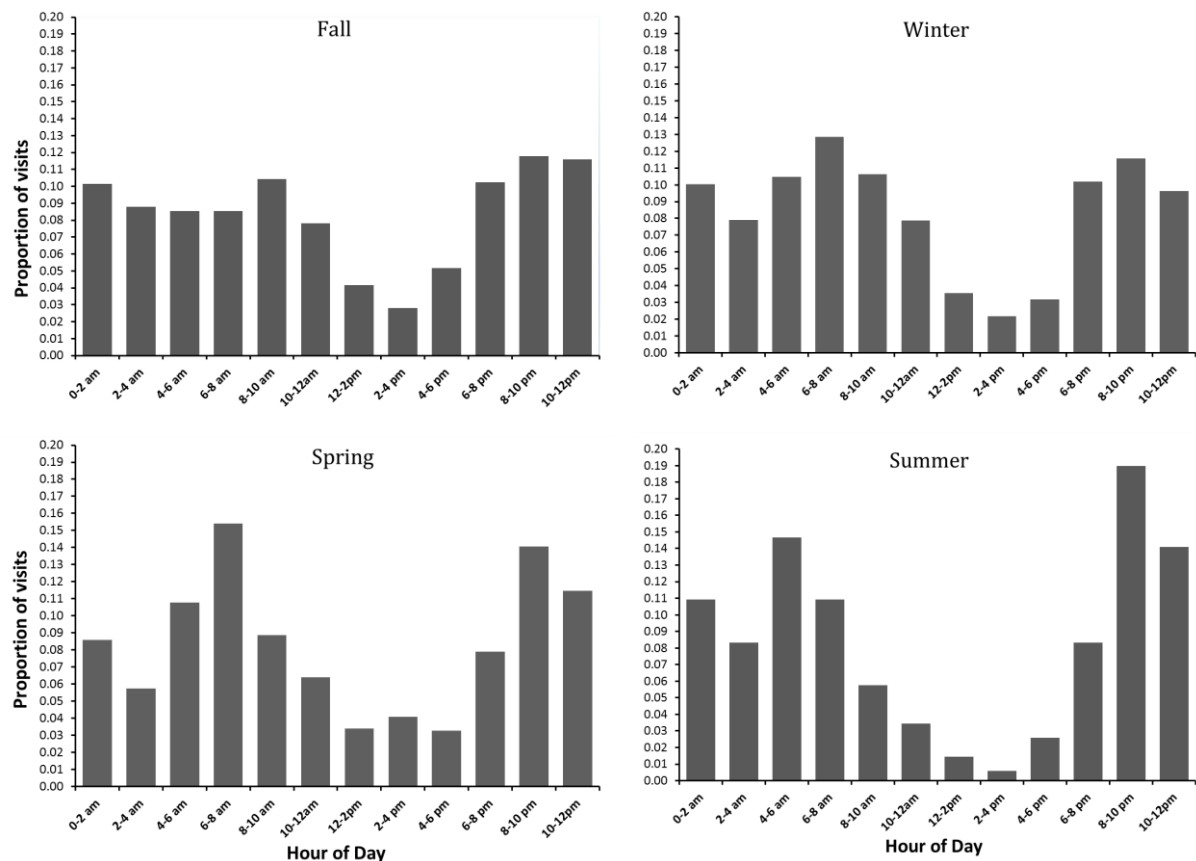


Figure 6. Proportion of fisher visits to baited camera survey stations by 2-hr periods of the day during 4 seasons on the SNAMP site. Data from fisher detections at baited camera stations on the SNAMP study, Oct. 2007-Oct. 2013 (R. Sweitzer unpublished data); seasons defined by the annual solar cycle.

Fishers are often active (presumably looking for food, mates, etc.) for 2-5 hours at a time, separated by variable periods of inactivity for resting or feeding (Arthur and Krohn 1991; Powell 1993). Males are more active than females during the breeding season; both sexes are more active in summer than winter; and snow cover and ambient temperatures can affect activity patterns (Raine 1983; Arthur and Krohn 1991; Weir et al. 2004; Weir and Corbould 2007). Denning mothers are probably more active than non-reproductive females during late spring, because mothers must feed their kits (Arthur and Krohn 1991; Paragi et al. 1994). Although there is individual variation, unbaited camera stations at den sites within the SNAMP study area photographed denning females at the den tree most often around sunset (R. Sweitzer and C. O'Brien, unpublished data).

3.2.3 Diet

Fishers are dietary generalists, consuming a variety of small- and medium-sized mammals, birds, insects, reptiles, and hypogeous fungi (Zielinski et al. 1999; Zielinski and Duncan 2004; Golightly et al. 2006). Southern Sierra Nevada fishers are smaller than fishers outside California and consume a wider variety of small prey and fewer large prey (Zielinski et al. 1999; Thompson et al. 2014). In the assessment area, small- to medium-sized mammals are found in >70% of fisher scat (Martin 1994; Zielinski et al. 1999). However, most dietary data are based on frequency of occurrence of prey

remains (e.g., hair, bones) in fisher scat, which can underestimate the relative contribution of larger prey and overestimate the contribution of smaller prey (Klare et al. 2011)⁶.

Recent analyses (K. Slauson, PSW, unpublished data) suggest that fishers may specialize on mammal species that weigh >200 g (7 ounces [oz]) and that the proportion of these prey in the diet is inversely related to fisher home range size. Moreover, an increase in the frequency of medium-sized mammals (75-200 g [2.65-7 oz]) in the diet is related to increased home range size, whereas the proportion of smaller mammals (<75 g [2.65 oz]) has no relationship to home range size. Fishers in the assessment area eat a high proportion of large prey (>200 g [7 oz]), especially western gray squirrels (*Sciurus griseus*) and Douglas squirrels (*Tamiasciurus douglasii*)—species typically associated with mature forest conditions—as well as California ground squirrels (*Otospermophilus beecheyi*)—which are associated with early seral vegetation communities. Fisher diets in the assessment area appear to include a low proportion of brush rabbits (*Sylvilagus bachmani*) and woodrats (*Neotoma* spp., K. Slauson, unpublished data). The fisher's diet changes seasonally with the availability of food items. The diet is most diverse during summer and fall and least diverse in winter and spring, when it is mostly composed of mammals (K. Slauson, unpublished data). California ground squirrels and chipmunks (*Tamias* spp.) are presumably available only during the warmer months (summer, late spring, and early fall), due to hibernation.

Abundance of larger fisher prey items may be a limiting food source for fishers in the southern Sierra Nevada (Slauson and Zielinski, unpublished data,). In other regions, snowshoe hares (*Lepus americanus*) and porcupines (*Erethizon dorsatum*) are major dietary components (Coulter 1966; Powell 1993). Hares and porcupines are uncommon in the assessment area, although historically porcupines were common throughout the Sierra Nevada, especially in open to moderately dense montane conifer forests with an understory of herbs, grasses, and shrubs (Taylor 1935; Woods 1973; Verner and Boss 1980). Grinnell et al. (1937) reported, based on stomach examinations and field observations, that porcupines were in California fisher diets; however, the authors did not elaborate on numbers or locations of these observations, so the historical importance of porcupines in their diet in the assessment area is unknown. Porcupines appear to be declining in California and many portions of western North America (Allen and Casady 2012). Although there is little historical data on porcupine populations in the assessment area, numerous anecdotes suggest that they declined substantially during the 20th century and are continuing to decline (Weiser 2012; L. Myers and J. Buckley, Central Sierra Environmental Resource Center, unpublished data 2012, 2013⁷). One reason for porcupine declines was systematic poisoning and shooting during the 20th century to reduce porcupine damage to trees (Keyes 1934; Anthony et al. 1986). According to several members of the California Forest Pest Council, in the Sierra Nevada, forest managers and animal damage control specialists regularly poisoned and shot porcupines in the 1950s and continuing into the 1980s in some places (R. Sweitzer, personal communication). Porcupine populations do not appear to be recovering since systematic persecution stopped, probably in part due to the very low reproductive rates of porcupines (a female can have only 1 pup per year). It is also possible that continuing threats, such as rodenticide poisoning at marijuana grow sites, is contributing to the lack of recovery, although there is no direct evidence for

⁶ Fishers and other mesocarnivores usually swallow small prey, like mice, whole but avoid swallowing bones and skin of larger prey. Because scat analysis uses bones, hair, or other physical evidence to infer what was eaten, frequency of occurrence in scats therefore underestimates the contribution of larger prey and inflates the importance of smaller prey in the diet.

⁷ http://cserc.org/main/news/news_briefs/2012-04_porcupinesurveys_outreach.html
http://www.cserc.org/main/news/news_briefs/2013_porcupine_survey.html

this hypothesis. The relationship between prey availability and fisher ecology needs further research, including whether increasing porcupine and squirrel abundance might benefit fishers.

3.2.4 Interspecific competition

Fishers may compete for prey with other species, including coyotes, foxes (*Urocyon*, *Vulpes*), skunks (*Mephitis*, *Spilogale*), bobcats, lynx, martens, weasels (*Mustela frenata*), and wolverines (*Gulo gulo*) (Powell and Zielinski 1994; Dark 1997; Campbell 2004). In the Sierra Nevada, Campbell (2004) found that mesocarnivores with similar body sizes were less likely to co-occur. In particular, gray fox (*Urocyon cinereoargenteus*) and striped skunk (*Mephitis mephitis*) were significantly less likely to occur where fishers were common, which may be due to exclusion by fishers or differences in habitat preferences between the species.

The dynamics of competition between martens and fishers are complex. There is extensive dietary overlap between fishers and martens in the southern Sierra Nevada (Zielinski and Duncan 2004). Fishers, with their size advantage, appear to excel in interference competition (Krohn et al. 1995), but in regions with deep winter snow, fishers are disadvantaged by their higher foot-loading, which reduces their ability to move efficiently on snow (Krohn et al. 1997).

Marten and fisher ranges in the Sierra Nevada are largely parapatric, with martens at higher elevation, but with some overlap at intermediate elevations (Zielinski et al. 1997; Zielinski and Duncan 2004). R. Sweitzer (unpublished data) evaluated range overlap between martens and fishers using camera detections among nearly 900 survey grids from October 2007 to January 2012. Fishers were detected in 405 grids, and martens in 126 grids. Both species were detected at only 35 grids between 1,597 and 2,496 m (5,240–8,189 ft) elevation (mean = 2,044 m [6,706 ft]). Fishers were rarely detected above 2,290 m (7,500 ft), and marten detections were rare below 1,830 m (6,000 ft). The relatively narrow zone of overlap (marten distribution extends to ~3,200 m [10,500 ft]) supports the hypothesis that martens avoid habitats occupied by fishers due to interspecific competition and that fishers avoid areas with deeper snow (Krohn et al. 1997).

However, J. Tucker (unpublished data) found extensive overlap in fisher and marten elevation ranges on the Sequoia National Forest (Figure 7). Martens are apparently absent from the Kern Plateau—a high elevation area in the eastern portion of the Sequoia National Forest where fishers are regularly detected at elevations typically associated with martens. The lack of deep snows on the Kern Plateau may allow fishers to occupy a higher elevation range there than on the Sierra National Forest (Powell and Zielinski 1994); this may also explain the lack of marten detections there, because martens are highly snow-adapted (Krohn et al. 1995, 1997) and rest almost exclusively below snow during winter (Spencer 1987). In the western portion of the Sequoia National Forest, both species are detected at multiple survey locations in the Greenhorn Mountains, a very steep, narrow range that drops rapidly from nearly 2,530 m (8,300 ft) to <1,220 m (4,000 ft) over just a few miles. This terrain, which receives deep snows at higher elevations, may not provide martens an opportunity to avoid range overlap with fishers at the resolution of the regional sampling data (J. Tucker, personal communication).

Zielinski et al. (in press) found higher detection rates for martens during winter than summer, with martens restricted primarily to upper elevation red fir (*Abies magnifica*) and lodgepole pine (*Pinus contorta*) forests in summer and apparently spreading down to also use lower elevation mixed-coniferous forests during winter. R. Sweitzer et al. (unpublished data) also detected a down-elevation shift in habitat use by fishers between snow and non-snow periods of the year, coincident with martens being detected at lower-elevation survey stations during winter compared to summer. A

reduction in snowpack at transitional zones within the southern Sierra Nevada is expected in the next century, potentially leading to increased encounters between the species (Purcell et al. 2012).

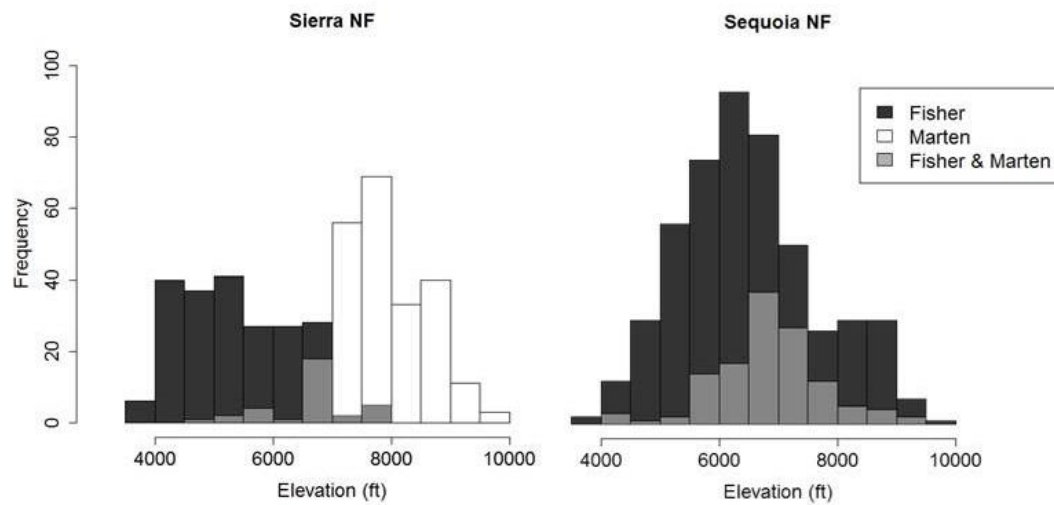


Figure 7. Elevation of fisher and marten detections and overlap at survey stations on the Sierra and Sequoia national forests. Source: J. Tucker, unpublished data, regional monitoring program.

4 Habitat Associations

Fisher habitat has been described at multiple, hierarchical scales, consistent with the 4 orders of habitat selection described by Johnson (1980): (1) selection of the geographic range at a landscape scale by 1 or more populations; (2) selection of home range areas by individuals within the geographic range; (3) use of habitat components or areas within the home range; and (4) selection of specific habitat elements or resources within an area, such as a denning cavity (Johnson 1980, Sawyer and Brashares 2013). Conservation and management actions must consider all 4 scales to ensure that individuals can obtain all their life requisites (e.g., food, cover, mates) within their home ranges, and home ranges must be distributed in relatively contiguous and connected blocks of habitat to maintain a healthy, interbreeding population.

The landscape scale, measured for fishers in thousands of square kilometers (or miles), is the appropriate scale for considering how fisher populations are distributed across a region (e.g., the southern Sierra Nevada) in relation to environmental conditions, such as elevation, climate conditions, and forest types. Understanding selection at this scale can help prioritize where specific conservation or management actions are warranted to conserve or enhance a population, or to connect multiple populations or subpopulations.

The home range scale ($\sim 10 \text{ km}^2$ or 4 mi^2) considers the distribution of environmental conditions (e.g., forest composition and structure) within areas selected by individual fishers for home ranges. Understanding selection at this scale helps us understand the habitat needs of individual fishers and how individuals, and populations of individuals, may be affected by environmental changes.

Studies of habitat selection at the sub-home range scale ($< 10 \text{ km}^2$ or 4 mi^2) evaluate ecological conditions within fisher home ranges—such as forest stand characteristics surrounding resting, denning, or foraging locations (sites)—as well as habitat elements such as cavities or other structures used for resting, denning, or capturing prey. Understanding selection at this scale can highlight particular types of habitat features to enhance or protect with management actions.

4.1 Landscape Scale

In western North America, fishers are associated with late-successional conifer or mixed-conifer-hardwood forests characterized by an abundance of dead and downed wood, dense, often multi-layered canopies, and large trees (Buskirk and Powell 1994; Zielinski et al. 2004a, b; Purcell et al. 2009; Lofroth et al. 2010; Raley et al. 2012; Lewis 2014). In the Sierra Nevada, fishers occur primarily in dense, mature mixed-conifer and ponderosa pine forests at elevations that support the greatest above-ground forest biomass (many large trees) and do not accumulate as much deep and persistent snow as higher elevations.

Landscape scale selection typically has been examined using statistical species distribution models using fisher detection-nondetection data from field surveys. These data are statistically related to environmental GIS variables within a $5\text{-}20 \text{ km}^2$ area ($1.9\text{-}7.7 \text{ mi}^2$, roughly the size of a female home range) around each survey site, using multivariate modeling algorithms, such as generalized additive models (GAM) or maximum entropy models (Maxent). The resulting statistical equations correlating fisher occurrence with environmental variables can be expressed as a map showing relative habitat suitability (or more accurately, probability of detecting fishers) across a landscape. Large areas of predicted suitable habitat (i.e., with probability of fisher occurrence higher than an appropriate probability threshold) can be interpreted as areas where fishers may establish home ranges.

A variety of landscape-scale habitat models have been developed for fishers in the southern Sierra Nevada (Davis et al. 2007; Zielinski et al. 2010; Spencer et al. 2011; Spencer and Rustigian-Romsos 2012a), using an array of environmental data layers and different sets of fisher survey data. Although the specific statistical algorithms, resolutions, and environmental variables vary among these models, the predicted fisher distribution or habitat quality patterns on the landscape are remarkably similar, increasing confidence in model predictions. The best predictors of suitable habitat are elevation, climate (precipitation and temperature), and various measures of forest composition and structure, such as total above-ground forest biomass, tree canopy cover, and average tree size, averaged over a 5-10 km² (1.9-3.9 mi²) area.

Figure 8 shows the best recent fisher distribution model developed by CBI for this assessment using Maxent (Phillips et al. 2006), the 2011 fisher monitoring and survey database, and some environmental data layers that were not available to previous modeling efforts (details described in Appendix A). Predictors included in the model were averaged over a 10-km² (3.9 mi²) moving window (roughly female fisher home range size) and included basal area-weighted canopy height (a measure of tree size), proportion of the moving window with >60% canopy cover, minimum temperature of the coldest month, and tassel-cap greenness (a measure of lush, green vegetation based on satellite imagery). The landscape pattern of predicted habitat suitability is generally consistent with previous models and provides excellent statistical fit to the fisher locality data. As with previous models and on-ground habitat assessments, the variables indicate that fishers are closely associated with forests at intermediate elevations with moderate climate conditions that support many large trees within stands having dense, green canopies.

Landscape habitat models show that fisher habitat in the southern Sierra Nevada is arranged in a roughly north-south trending collection of narrow habitat patches (or habitat cores, as described in Section 5) in the elevation zone that supports Sierran Mixed Conifer, Ponderosa Pine, and Mixed Hardwood-Conifer CWHR forest types, and separated by major river canyons (e.g., San Joaquin, Kings, and Kern rivers). The narrow, undulating distribution of habitat makes it susceptible to fragmentation into even smaller patches, for example, by large, stand-replacing wildfires. Predicted habitat north of the Merced River is not currently occupied by a reproducing fisher population. Fishers have never been documented in the isolated patches of predicted habitat south of the Kern River (e.g., Piute Mountains), and the FTT considers this area unlikely to support fishers in the future due to isolation, small size, and marginal habitat conditions.

4.2 Home Range Scale

Fisher home ranges comprise a mosaic of vegetation types and successional stages, but are consistently associated with larger, more contiguous patches of dense and mature forests having few or small open areas (Corbould 2010; Lofroth et al. 2011; Raley et al. 2012; Sauder and Rachlow 2014; Weir and Lewis 2014). In the northern Rockies, Sauder and Rachlow (2014) found that fisher home ranges included a higher percentage of mature forest, closer proximity among mature forest patches, and a lower percentage of open areas than randomly located “pseudo” home ranges. In a comprehensive review of fisher habitat ecology in western North America, Raley et al. (2012) found that moderate to dense forest canopy is one of the strongest and most consistent predictors of fisher habitat use at all scales, and that fisher home ranges generally contain ≤5% open canopy areas. Sauder and Rachlow (2014) and Weir and Corbould (2010) predict that a 5-10% increase in the amount of open area at the home range scale reduces the probability of fisher occupancy by 39-60%.

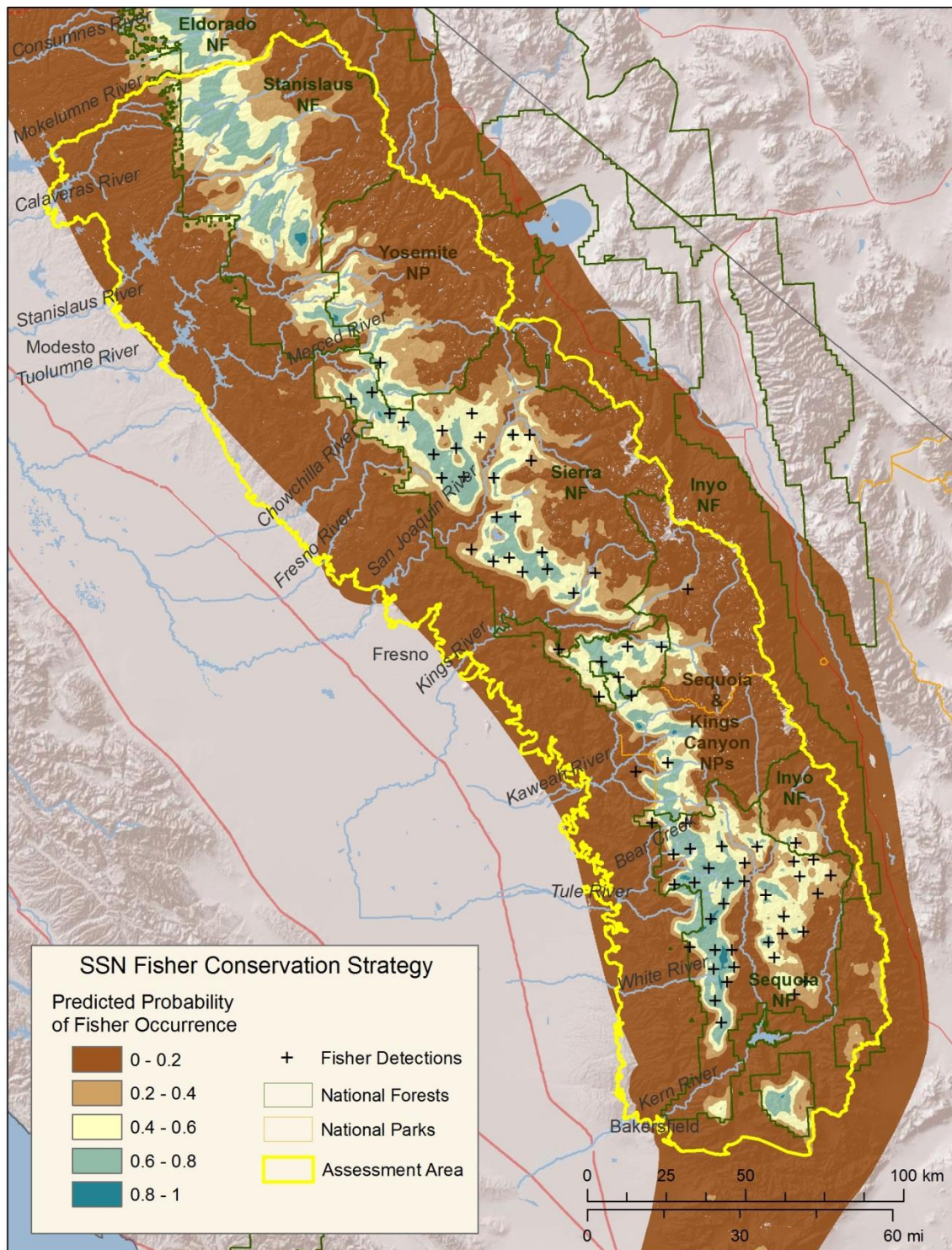


Figure 8. Predicted probability of fisher occurrence (or predicted habitat value) at the landscape scale within the assessment area. See Appendix A for methods.

However, reference data suggest that yellow pine (Jeffrey and ponderosa in this area) and mixed conifer habitats of the Sierra Nevada historically had a higher proportion of open and early seral stages (15-25% in herbs, shrubs, seedlings, and saplings) than today (~10%), and that vegetation had more fine-grained patchiness than we see today (Safford 2013). We don't know how this may have affected fisher distribution, abundance, or habitat selection.

Several studies have described forest composition and structural characteristics of fisher home ranges in the assessment area, including Zielinski et al. (2004b) in the SSN fisher and marten study on Sequoia National Forest and Thompson et al. (2010) on the Sierra National Forest. On the Sequoia National Forest, the dominant CWHR types within home ranges were Sierran Mixed Conifer (~40%), followed by Ponderosa Pine (~30%), and Montane Hardwood (~12%). About 60% of all types were size class 4 (28-61 cm [11-24] dbh) and about 12% were size class 5 (>155 cm [61 in]). The overwhelming majority of stands within home ranges were characterized as “dense canopy” (60-100% canopy cover); <2% of home range areas were in stands that were “sparse” (10-24% canopy cover, see Table 3 in Zielinski et al. 2004a).

Thompson et al. (2010) characterized female home ranges on the basis of vegetation composition and configuration. Canopy cover averaged 63% (SD=3.2), basal area averaged 162.4 ft²/ac (37.3 m²/ha; SD=23.3), and there was an average of 2.55 large trees (>89 cm [35 in] diameter at breast height (dbh) per acre across female home ranges. Home ranges were characterized as having high cohesion of habitat patches, small but connected areas of CWHR size class 4 habitat, and a moderate degree of edge (see Table 2 in Thompson et al. 2010). Because of their large sizes, fisher home ranges include a mosaic of forest successional stages, including significant patches of young stages; however, the patches of mature successional stages are necessary to support prey, resting, and denning resources.

4.3 Sub-home Range Scale

Throughout the western US, forest structure seems to be more important than tree species composition for within-home range fisher habitat selection (Powell and Zielinski 1994; Raley et al. 2012). Both active (foraging) and inactive (resting and denning) fishers are associated with complex forest structure (Lofroth et al. 2010; Zhao et al. 2012)—i.e., understory vegetation, a diversity of tree sizes, and snags and other coarse woody debris. Less is known about foraging habitat selection than resting and denning selection, but it appears that fishers are less selective for foraging habitat, perhaps because the diverse suite of suitable prey occurs across a variety of habitat conditions. Fishers also appear to select generally cooler, more mesic, and less variable microclimates, such as near canyon bottoms versus ridgetops. Fisher telemetry locations are disproportionately on mid-slopes of canyons, where forest density is greater than on ridges (Underwood et al. 2010).

4.3.1 Foraging

Foraging habitat selection remains 1 of the least well-understood aspects of fisher ecology, due to the challenge of observing fishers while foraging. Snow-tracking of fishers elsewhere in their range suggests that they spend most of their time foraging in habitats with the highest abundance of high-value prey (Powell 1993). In the assessment area, high-value prey (>200 g [7 oz], K. Slauson, unpublished data) typically includes tree and ground squirrels. Tree squirrels tend to be found in mature forests with moderate to high canopy cover, large live and dead trees, and in or near riparian areas (Meyer et al. 2005, 2007; Kelt et al. 2014). Key food resources for tree squirrels include hard mast and truffles—the fruiting bodies of ectomycorrhizal fungi. Mast of conifers and oaks is most abundant and reliable when produced by large-diameter, old trees in older stands (McDonald 1990). Truffle abundance is positively correlated with conditions that promote soil moisture, such as large

coarse woody debris, deep organic soil layers, and proximity to riparian areas (Meyer and North 2005).

The wide variety of food items in the fisher diet (Zielinski et al. 1999; Golightly et al. 2006) suggests that fishers forage in a diversity of vegetation types, though they likely concentrate their activities in mature forests where they have the highest probability of securing relatively large prey (e.g., >200 g [7 oz] tree squirrels). Important prey include species associated with oaks, such as dusky-footed woodrat (*Neotoma fuscipes*), brush mouse (*Peromyscus boylii*), and western gray squirrels (Innes et al. 2007; Roberts et al. 2008). Other species preyed on by fishers, such as broad-footed mole (*Scapanus latimanus*), pocket gophers (*Thomomys bottae* and *T. monticola*), voles (*Microtus longicaudus* and *M. montanus*), western jumping mouse (*Zapus princeps*), and mountain beaver (*Aplodontia rufa*) are associated with canopy gaps, meadows, and riparian areas (Verner and Boss 1980; Beier 1989; Anthony et al. 2003). Other seral forest stages, edges of shrub fields, and chaparral support more generalist prey species such as deer mice (*Peromyscus* spp.), moles, and pocket gophers.

R. Sweitzer (unpublished data) evaluated the distribution of squirrels in the SNAMP area using the same camera detection stations used to monitor fishers on 1-km² (0.39-mi²) grid cells (Figure 9). Stations were baited with peanut butter and pecans and monitored October 2007–October 2010. Ninety-five percent of surveyed grids detecting fishers also detected at least 1 species of tree or ground squirrel. Douglas squirrel detections are most common between 1,219 and 2,134 m (4,000–7,000 ft), whereas western gray squirrel detections are most common below ~1,676 m (5,500 ft; Figure 8). Gray squirrels are most commonly found in CWHR Montane Hardwood habitats, while Douglas squirrels are most frequently in cells dominated by CWHR Sierran Mixed Conifer. Northern flying squirrels (*Glaucomys sabrinus*) were found in grids dominated by CWHR Ponderosa Pine and Sierran Mixed Conifer, and California ground squirrels were most frequently found on grids dominated by the same habitats.

Truex and Zielinski (2013) developed a predictive fisher “foraging” habitat model, using 8 primary variables linked to important prey resources: canopy cover, size of hardwood trees, maximum tree dbh, presence of conifer snags, size of all trees, basal area of hardwoods, distance to water, and slope. The results suggest that fisher foraging is not closely tied to particular habitat types because fisher prey occupy a diversity of habitats. However, if home range size is inversely related to foraging habitat quality, the smallest home ranges are expected to occur in places where the fisher’s diet includes the largest proportion of western gray and Douglas squirrels, species that favor mature hardwood and conifer forests, respectively.

In keeping with the assumptions of Truex and Zielinski (2013), this assessment considers the landscape-scale fisher habitat model (Figure 8) to be equivalent to a fisher foraging model, because fishers baited to detection stations were presumably foraging, and because essentially all habitat types within fisher home ranges support some prey species and may be used for foraging, with the possible exception of larger meadows or other open habitats.

4.3.2 Resting habitat and structures

Within their home ranges, fishers need a variety of suitable resting habitats and structures (e.g., cavities or platforms in trees or snags) to avoid bad weather conditions, conserve energy, avoid predators, and safely survey for and consume prey (Zielinski et al. 2004a, b; Purcell et al. 2009; Aubry et al. 2013). They use multiple resting sites to minimize travel between foraging and are rarely documented reusing rest sites (Kilpatrick and Rego 1994; Seglund 1995; Zielinski et al. 2004b; Aubry et al. 2013). Resting structures and the surrounding conditions are probably the best studied aspect of

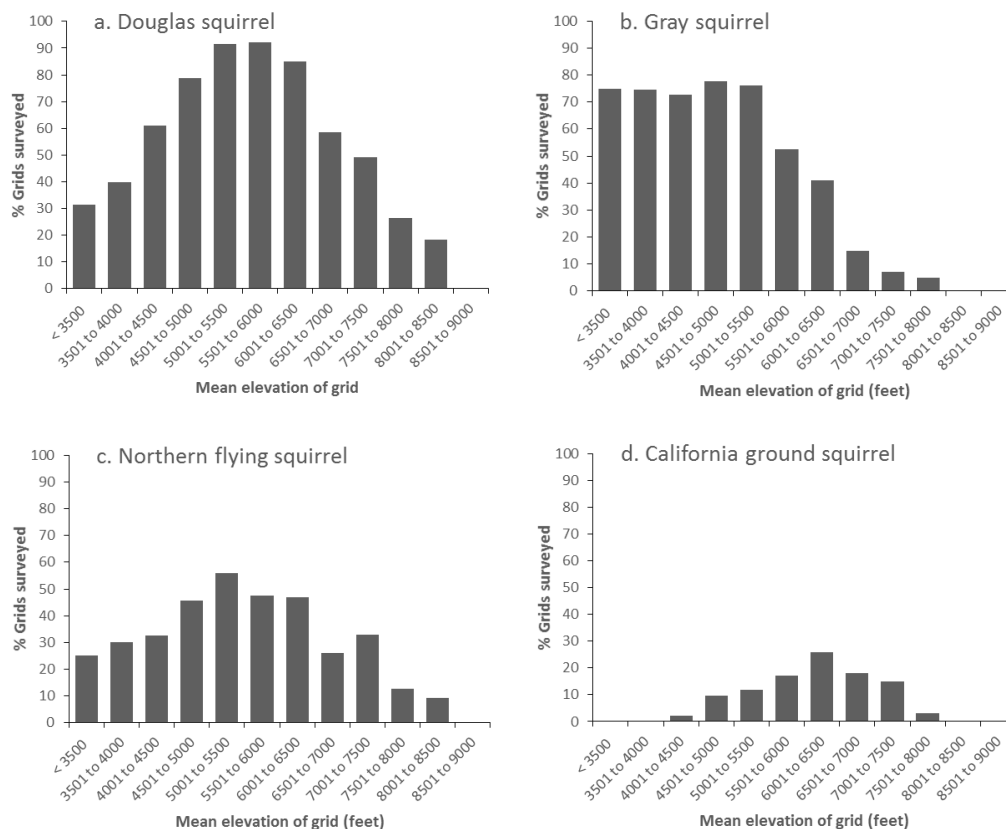


Figure 9. Proportion of camera stations that detected 4 species of squirrels as a function of elevation on the SNAMP area, Oct. 2007-Oct. 2013.

Source: R. Sweitzer, unpublished data

fisher habitat, in part because they can be located precisely using telemetry, and in part because they are considered a limiting resource for fishers, as compared to potential foraging or movement habitats (Zielinski et al. 2004b).

Resting structures used by fishers in western North America are primarily live trees (64-83%), followed by snags (7-26%), and coarse downed wood (2-20%, Lofroth et al. 2010). By comparing use and availability by tree species for rest sites, Purcell et al. (2009) found that ponderosa pine was used for resting more often than expected, given availability, incense cedar less often than expected, and there was some evidence that California black oaks (*Quercus kelloggii*) were also selected (14% of rest trees versus 9% of available trees). Table 10 shows the resting structures used by fishers at the KRFP site (R. Green, unpublished data). Resting sites are usually in forest stands with dense canopy cover, large trees, a high basal area of small and medium-sized trees, high abundance of medium- and large-sized snags and hardwoods, close proximity to water, and steep slopes (Zielinski et al. 2004a, b, 2006; Purcell et al. 2009; Aubry et al. 2013; Truex and Zielinski 2013). Resting typically occurs in live trees, snags, and logs that are in the largest available diameter classes (Lofroth et al. 2010; Aubry et al. 2012; Raley et al. 2012), and these structures are typically deformed or in a state of decay (Weir et al. 2012). Resting trees or snags are typically among the largest structures in the rest-site vicinity, averaging 1.5-1.7 times larger in diameter than available trees (Zielinski et al. 2004b; Purcell et al. 2009). Zielinski et al. (2004b) reported the following average diameters of resting trees and snags on the Sequoia National Forest: live conifer 110.2 cm (range 31-433 [43.3 in (range 12.2-170.4 in)]), live

hardwood 64.9 cm (range 30-145 [25.5 in (range 11.8-57 in)]), conifer snag 120.3 cm (range 45-328 [47.4 in (range 17.7-129.1 in)]), log 131.5 cm (range 45-500 [51.8 in (range 17.7-196.8)]). Fishers use cavities, platforms, witches' brooms, old squirrel nests, and large branches for resting. Cavities are mostly created by heartwood decay, whereas witches' brooms are created by mistletoe and rust fungi (Purcell et al. 2012).

Table 10. Summary of rest structures used by fishers at KRFP (summer 2007–fall 2013).

Rest site structure type	No.	Percent	Tree species ^a
Live tree (conifer)	267	38	White fir, ponderosa pine, incense cedar, sugar pine, Jeffrey pine, red fir, Douglas fir
Live tree (hardwood)	163	23	California black oak, canyon live oak, white alder, big leaf maple
Snag (conifer)	156	22	White fir, incense cedar, ponderosa pine, sugar pine, red fir, Jeffrey pine
Snag (hardwood)	18	3	California black oak
Log	45	6	Incense cedar, California black oak, ponderosa pine, sugar pine, white fir, giant sequoia
Ground burrow	2	<1	
Rock pile (often associated w/ ground burrow)	36	5	
Snow burrow	3	<1	
Stump	6	1	
Other	9	1	
Total	705		

^aTree species listed in general order of frequency of use for each structure type: white fir (*Abies concolor*), ponderosa pine, incense cedar (*Calocedrus decurrens*), sugar pine (*P. lambertiana*), Jeffrey pine (*P. jeffreyi*), red fir, Douglas fir (*Pseudotsuga menziesii*), California black oak, canyon live oak (*Q. chrysolepis*), white alder (*Alnus rhombifolia*), big leaf maple (*Acer macrophyllum*), giant sequoia (*Sequoiadendron giganteum*).

Source: R. Green, unpublished data.

Canopy cover appears to be the most important variable in resting site habitat selection throughout the western US and within the assessment area (Zielinski et al. 2004b, 2006; Purcell et al. 2009; Truex and Zielinski 2013). In the KRFP study area, Purcell et al. (2009) found canopy cover to be the most important variable in distinguishing resting sites from random sites; canopy cover (measured using moosehorn) averaged 73.7% at resting sites compared to 55.3% at random sites. Purcell et al. (2009) also found that the probability of a site being used for resting decreased with decreasing canopy cover, from an optimal canopy cover of 72% to a lower threshold of ~50%, below which resting sites are rarely found (97% of resting sites had >53% canopy cover). The decrease in the probability of use was more pronounced for sites that lacked other key characteristics of resting habitat (e.g., larger trees), compared to higher quality sites. Based on sensitivity analyses, Purcell et al. (2009) concluded that management actions should maintain a minimum of 61% canopy cover while also growing and retaining large trees and snags and complex horizontal and vertical forest structure.

Throughout the Pacific coastal region, Aubry et al. (2013) found that fishers select resting sites with lower heat load indices relative to available sites. In the assessment area, resting sites tend to be on steep slopes, in canyons rather than on ridges, and close to water (Zielinski et al. 2004b; Purcell et al. 2009; Underwood et al. 2010)—all factors that, in addition to dense canopies, contribute to low heat loads and reduced temperature variability. Fisher resting sites in the southern Sierra Nevada are

characterized by more variable tree sizes than random sites, including numerous smaller trees. The abundance of smaller trees surrounding resting sites may be due to the history of logging and fire suppression in the assessment area, which has resulted in most forest stands being characterized by fewer large trees and more small trees. Abundant smaller trees may provide the requisite canopy cover and vertical structure needed for good resting habitat, so long as there is a suitably large resting tree or snag within the stand (Zielinski et al. 2004b; Purcell et al. 2009).

Seasonal variation in rest site use has had little study in the assessment area. In British Columbia, fishers used subnivean (under snow) rest sites more frequently during winter and arboreal sites more frequently in spring (Weir et al. 2004; Weir and Corbould 2008). In the Sierra, fishers tend to use cavities in trees and snags more than platforms for resting in colder temperatures (K. Purcell, PSW, unpublished data).

CBI prepared a fisher resting habitat model at a 2-km² (494-ac) resolution (Figure 10). The model used 1,248 rest-site localities provided by the KRFP (C. Thompson, unpublished data) and SSN marten and fisher studies (Zielinski et al. 2004b), filtered to 237 localities for model training (Appendix A). The model provides good statistical fit to locality data and appears to extrapolate well over the entire assessment area. It uses 4 variables averaged over 200 ha (494 ac) to predict resting habitat quality: proportion in appropriate forest types, proportion dominated by trees >25.4 cm (10 in) dbh, proportion with canopy cover >70%, and proportion having a hardwood component (see Appendix A).

4.3.3 Denning habitat and structures

Den sites, where mothers bear and raise their kits, are probably the most limiting habitat element for the fisher population. Mothers typically use >1 den during the denning season (late Mar.-late Jun.): they give birth and initially care for young in natal dens, and later may move kits to 1 or a succession of several maternal dens until they are weaned (Powell et al. 2003; Aubry and Raley 2006; Matthews et al. 2013; R. Sweitzer, unpublished data). Like resting structures, reproductive dens are usually cavities in large, live or dead trees, in forest stands with dense canopy cover and complex structure. Suitable denning sites are probably a subset of suitable resting sites, because the requirements are more stringent: (1) the cavities must be large enough to shelter both mother and kits; (2) each den may serve as shelter for weeks rather than days; (3) the mother must provision her young while they are restricted to the den, so dens must be located within high-value foraging areas; and (4) denning begins in late March and early April, when snow may make foraging more difficult at higher elevations.

Using LiDAR to characterize the habitat structure surrounding den trees, Zhao et al. (2012) found that tall trees and steep slopes ($\geq 17\%$) are important predictors of denning habitat at fine scales (10-20 m, 33-66 ft). At larger scales (30-50 m, 98-164 ft), forest structure and complexity are more important. Together, the results indicate that fishers select den sites in clusters of large, mature trees in stands with variable tree heights and dense canopies, on fairly steep slopes. In the SNAMP area, R. Sweitzer (unpublished data) found habitat features within an 18-m (59-ft) radius of den trees to include mean canopy cover of 72%, mean shrub cover of 19%, mean slope of 37%, and mean elevation of 1,591 m (5,220 ft).

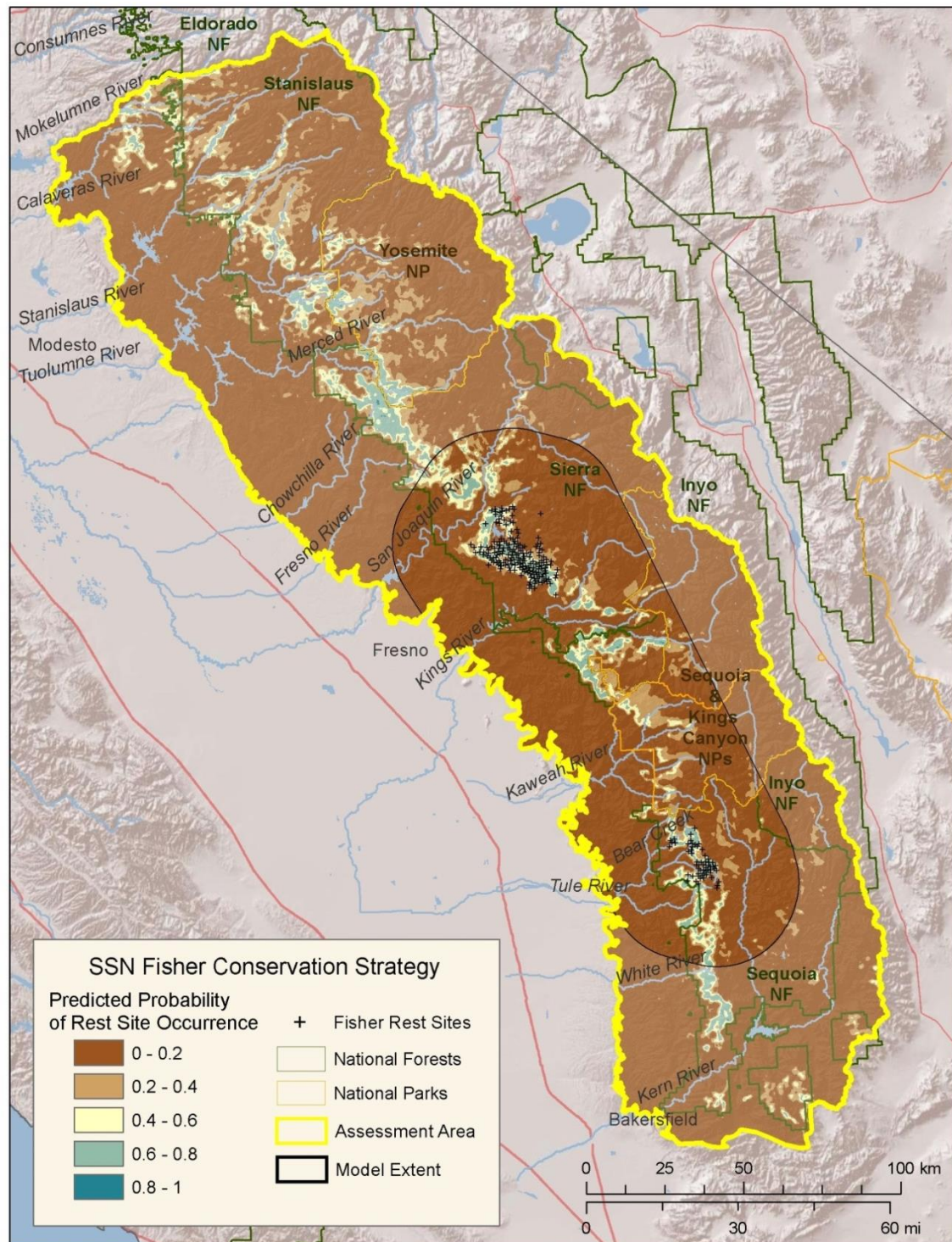


Figure 10. Fisher resting habitat modeled using Maxent and 237 resting localities from the KRFP and SSN fisher and marten telemetry studies. Highest confidence in model predictions is within the model extent (darker region); see Appendix A for methods.

Spencer and Rustigian-Romsos (2012b) used natal and maternal den localities from the SNAMP and KRFP studies to analyze fisher denning habitat on the Sierra National Forest at 2-km² (494-ac) resolution. They found that dens are concentrated at lower elevations relative to overall predicted fisher habitat, in the densest available mixed coniferous stands (CWHR density class D) having the largest trees (CWHR size classes ≥ 4). Dens were also concentrated where a moderate proportion of the landscape supports hardwood trees such as black oaks. Ninety-eight percent of fisher dens are in CWHR density class D (>60% canopy cover), and 98% are within size classes 4 (28-61 cm [11-24 in] dbh) and 5 (>61 cm [24 in] dbh).

Figure 11 shows a denning habitat model using a den locality dataset from the SNAMP and KRFP studies (unpublished data). The model predicts denning suitability, averaged at 2-km² (494-ac) resolution, using the following variables: proportion of area in an appropriate CWHR forest type, proportion of area with canopy cover $\geq 60\%$, proportion of area supporting hardwoods as a dominant component, percent slope, and August maximum temperature. Dens are primarily in mixed-coniferous and coniferous-hardwood stands with dense canopy cover, a moderate intermix of California black oaks, on steep slopes (~20-50%), and in areas with relatively low summer temperatures.

Confidence in the mapped denning habitat predictions is highest in the highlighted region of Figure 11, where the den data were collected, and predicted quality outside this model extent should be interpreted with caution. Almost no denning habitat potential is mapped on the Kern Plateau, which may reflect (1) poor model extrapolation or (2) true differences in denning habitat value due to environmental differences between the Kern Plateau and west-slope fisher habitat. For example, California black oaks, which appear to contribute to denning habitat value in other areas, are lacking on the Kern Plateau. If the model predictions are reliable, lack of denning value may explain, at least partially, the low fisher occupancy rates on the Kern Plateau (Zielinski et al. 2013a).

Figure 12 compares the elevation distribution of 321 dens on the SNAMP and KRFP areas (R. Sweitzer and C. Thompson, unpublished data) versus elevation of random points in fisher core habitat within the model extent shown in Figure 11. Dens are clustered at lower elevations than habitat in general, because dens are used only by females (which don't use higher elevation forests as much as males) during the denning season, which is initiated when snow is still on the ground.

Figure 13 shows modeled denning habitat overlaid onto modeled landscape-scale (or foraging) fisher habitat on the Sierra National Forest (both habitat layers are shown as binary, habitat versus non-habitat, using thresholds that optimize discrimination between presence and absence). The map illustrates that denning habitat represents a limited subset of available habitat, is selected at finer resolution than foraging habitat, and biased toward westerly, lower-elevation portions of available habitat where there is less snow and greater intermix with hardwoods, especially black oak.

Mothers typically move the young from the natal den to a succession of several maternal dens until they are weaned (Powell et al. 2003; Matthews et al. 2013; R. Sweitzer, unpublished data). In the southern Sierra Nevada, the total number of observed dens used each year thus varies from 1 to ~6 (mean=3.4 on the KRFP, R. Green, unpublished data). Multiple den trees may be used within a single denning season to (1) accommodate kit growth by moving to larger cavities, (2) reduce predation risk, as bobcats and mountain lions discover a den location (possibly due to odors from the accumulation of urine and feces) and may return to prey on the mother or young, and (3) avoid an accumulation of feces and parasites in a cavity. Although not analyzed in detail, there are no obvious differences in the types (tree species, live or snag) or size (mean dbh) of den trees used as natal compared to maternal den trees in the SNAMP area.

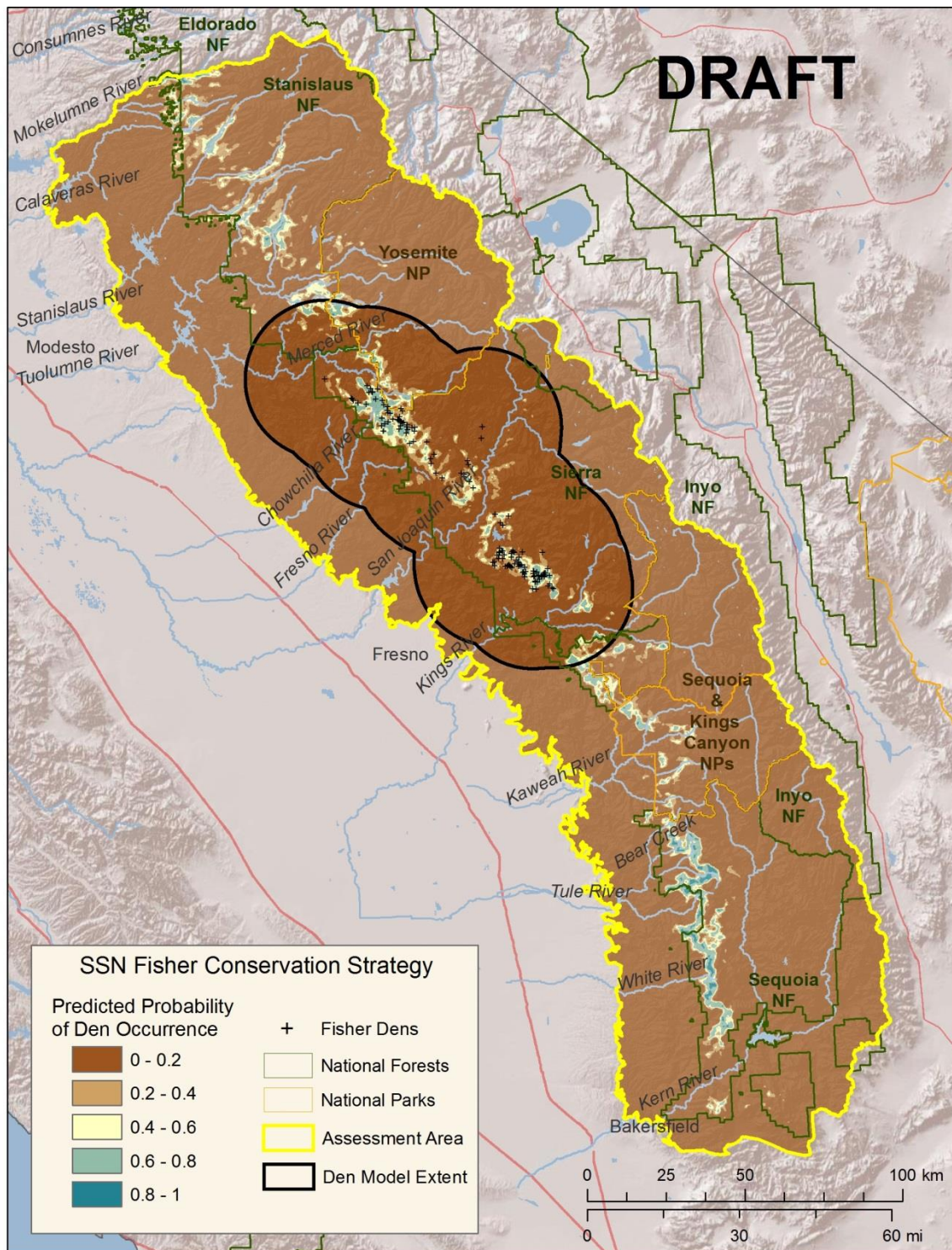


Figure 11. Fisher denning habitat modeled using Maxent and 154 den sites on the Sierra National Forest. Highest confidence in model predictions is within the den model extent (darker region). Note lack of predicted denning habitat on the Kern Plateau (southeast portion of Sequoia National Forest). See Appendix A for methods.

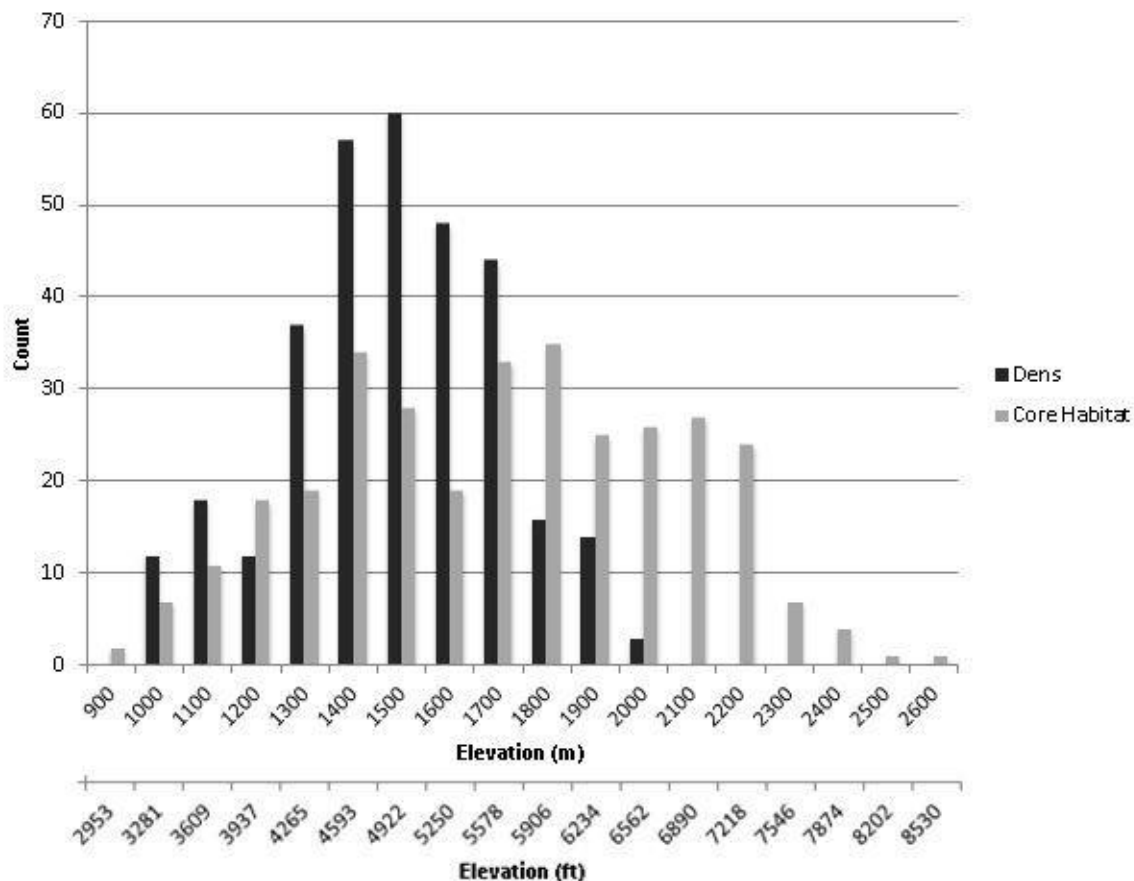


Figure 12. Elevation distribution of 321 den sites and core habitat on Sierra National Forest.

Moving kits to another tree is a physical challenge that is repeated for each kit. Consequently, natal and maternal den trees are usually relatively close together and clustered within a home range core use area. Successive den trees used by female fishers in the SNAMP fisher study area were 363 m (1,191 ft) apart on average (SE=37.8 m [124 ft], range=31-1,505 m [102-4,938 ft], N=71), and the distance between natal and first maternal den averaged 419 m (1,375 ft, R. Sweitzer, unpublished data). Successive maternal den sites average closer together than this (mean=287 m, $t=1.75$, $df=69$, $P=0.04$), likely because kits are larger following the first move from the natal den.

Both the SNAMP and KRFP studies have observed repeat use of dens between years. The SNAMP study identified 143 dens, spring 2008-spring 2013, of which 119 were unique and 24 were reused (Table 11). In 23 of these cases, the same individual reused 1 or several dens in successive years; in 1 case a female used a den used by a different female in a previous year (R. Sweitzer, unpublished data). Den reuse appears less frequent in the KRFP study area, which may reflect differences in availability of suitable denning structures (R. Green, personal communication).

California black oaks comprise the largest proportion (54%) of den species in the KRFP study area, with 91% of those dens in live trees (R. Green, unpublished data). The most common tree species used for denning in the SNAMP study area are California black oak (43%) and incense cedar (36%), followed by white fir and pine species (Table 11). Fifty-five percent of trees used for denning in the SNAMP area were live and 45% were dead (Table 11). Most white firs used for denning were live trees (74%), whereas only 44% of black oaks used for denning were snags (Table 11).

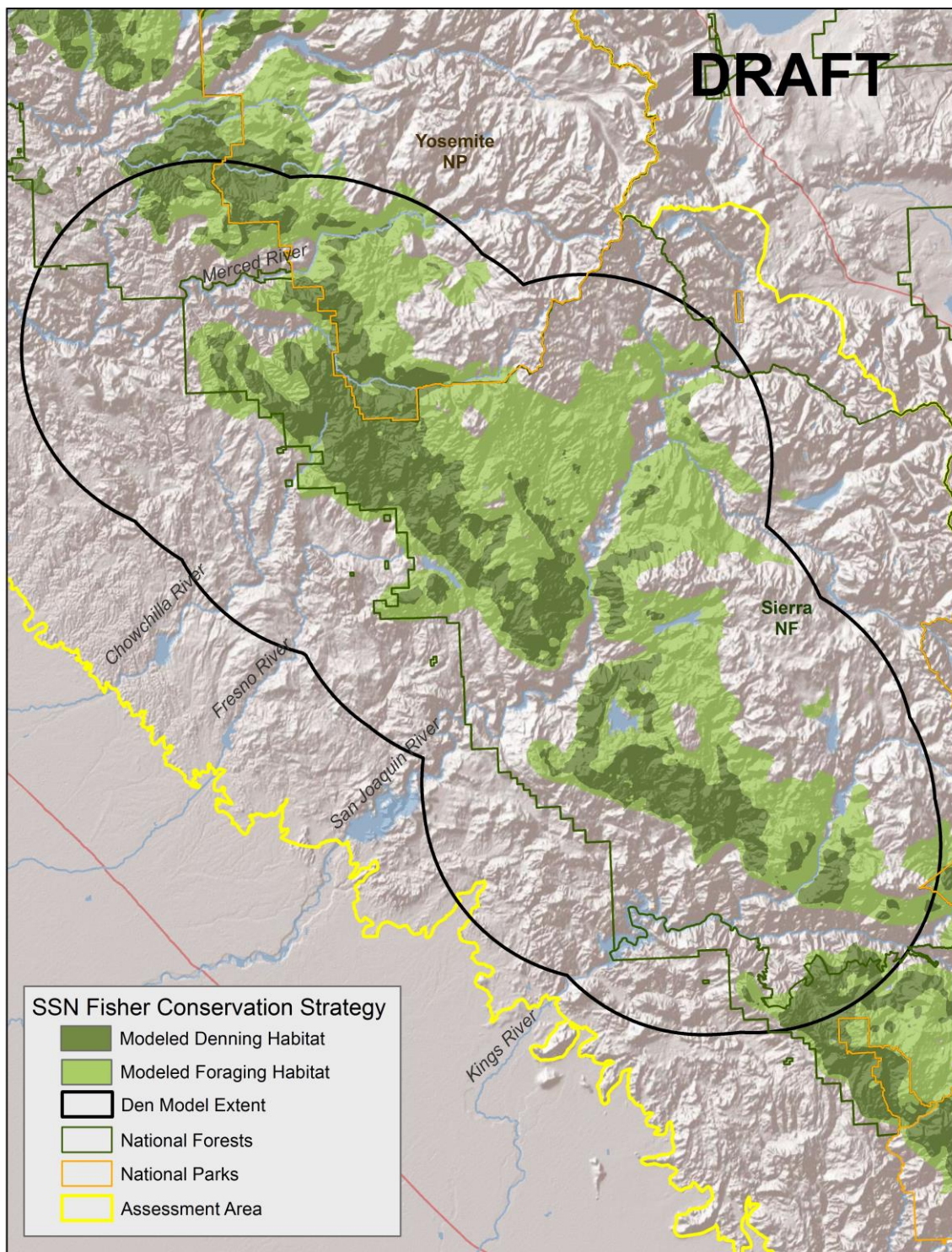


Figure 13. Modeled denning habitat overlaid on modeled foraging habitat on the Sierra National Forest. Maxent models shown in Figures 8 and 11 are here thresholded into binary habitat/non-habitat using the maximum sum of sensitivity versus specificity to optimize discrimination between presence and absence (see Appendix A). Denning habitat tends to be restricted to a generally lower-elevation (more westerly) subset of foraging habitat or overall suitable habitat in this area.

Table 11. Den use by tree and snag species in the SNAMP area.

Tree Species	Denning events (repeat use)^a	Percent within group	Unique structures^b
<i>Live trees</i>			
Black oak	34 (3)	43	31
Incense cedar	25 (4)	32	19
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 (4)	42	22
White fir	5 (5)	8	5
Pine species ^c	5	8	5
<i>Live tree or snag</i>			
Black oak	61 (3)	43	54
Incense cedar	52 (8)	36	41
White fir	19 (5)	13	19
Pine species	10	7	10
Canyon oak	1	1	1
Total	143 (16)		125
Burrow ^d	2		2

^a Number of individual trees used more than twice for denning; one live cedar tree was used by the same female in four successive denning seasons; all other repeat-use dens were used in two denning seasons.

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

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

^d Both burrows were used very briefly (1-2 days) and appeared to be openings into or beneath large decaying logs on the forest floor.

Source: R. Sweitzer, unpublished data

Trees and snags used for denning tend to be among the largest available, at least in part because the den cavities must be large enough to accommodate a female fisher and her kits. Conifers used for dens in the SNAMP study averaged >100 cm (39 in) dbh, and oaks averaged >70 cm (28 in) dbh (Table 12). In both the SNAMP and KRFP studies, 75% of oaks used for dens were ≥ 63 cm (25 in) dbh. Conifers were even larger, with 75% of them ≥ 89 cm (35 in) dbh in the KRFP study and ≥ 94 cm (37 in) dbh in the SNAMP study. Most cavities are in large-diameter live trees having some form of defect that facilitates access by decay organisms. This may be in the form of frost cracks, fire scars, branch scars, or woodpecker holes (Weir et al. 2012). Activities of primary cavity excavators seem particularly important in the southern portion of the fisher range (Raley et al. 2012; Weir et al. 2012). Live black oaks are susceptible to internal decay and probably last longer on the landscape than conifer snags (McDonald 1990).

Table 12. Physical characteristics of fisher den structures in the SNAMP area, Mar. 2008-Jun. 2013. Includes both natal and maternal dens.

Live trees				Snags		
Tree species	<i>n</i>	Mean dbh (cm/in)	Mean height (m/ft) ^a	<i>n</i>	Mean dbh (cm/in)	Mean height (m/ft)
Black oak	30	74.2/29.2	21.7/71.2	5	69.5/27.4	8.8/28.9
Incense cedar	18	127.2/50.1	32.5/106.6	22	105.1/41.4	16.4/53.8
White fir	14	110.8/43.6	33.9/111.2	22	103.7/40.8	27.4/89.9
Pines	5	112.8/44.4	37.4/122.7	5	109.6/43.1	27.6/90.6

^aData from a subset of den trees for which detailed measurements have been completed (*n* = 84). Source: R. Sweitzer, unpublished data.

5 Habitat Core and Linkage Areas

The distribution of fisher habitat in the southern Sierra Nevada has been described as a “string of sausages” (R. Barrett, UC Berkeley, personal communication)—with fishers distributed in a series of subpopulations separated by steep river canyons and other unsuitable habitat areas. Figures 14 and 15 illustrate this distribution pattern, based on results of spatially explicit habitat models (Section 4.1) and the landscape genetic and occupancy patterns described previously. The “core” habitat areas shown in Figure 14 are contiguous polygons of modeled suitable fisher habitat large enough to support at least 5 adult females (see Appendix A for methods). Gaps between some core areas occur where major river canyons (San Joaquin, Kings, and Kern) separate suitable habitats by at least 1 fisher home range radius. The connectivity (or linkage) areas shown in Figure 15 are 50-km (31-mi) normalized least-cost corridors (McRae and Kavanagh 2011), or swaths of habitat across which a fisher is expected to experience relatively low cumulative costs while passing between core areas (see Appendix A for methods).

Figure 15 refines the core delineation modeled in Figure 14 to show individual core areas and linkage habitats between them (including connectivity before and after the 2013 Rim Fire). Figure 14 also subdivides a long, contiguous habitat polygon south of the Kings River to reflect the genetic subdivision identified by Tucker et al. (2014) at Mountain Home Demonstration State Forest (using Bear Creek as the geographic split). The resulting map of core and linkage areas is consistent with data on fisher space-use patterns in the SNAMP and KRFP areas as well as landscape genetic patterns across the assessment area: fishers are expected to comeingle, interbreed, and establish home ranges relatively freely within each occupied habitat core, but dispersal between cores is relatively rare (especially by females; Tucker 2013; Tucker et al. 2014). The map of core areas and linkage habitats between them provides a biologically relevant subdivision for planning and management purposes. It provides a template for developing spatially explicit management and monitoring goals and objectives as part of the Conservation Strategy.

5.1 Fisher Core Areas

Table 13 summarizes key characteristics of the 7 fisher core areas, described below from south to north. Cores 1-5 (4,198 km² [1,621 mi²] total area) are occupied by breeding fisher populations; Cores 6 and 7 (1,677 km² [647 mi²] total area) currently are not occupied by breeding fisher populations, although fishers are detected occasionally in Core 6.

Table 13. Characteristics of delineated fisher core habitat areas.

Core	Occupied	Area, km ² (mi ²)	Mean (SD) predicted habitat quality	Area of denning habitat km ² (mi ²)	Primary (secondary) jurisdiction
1	yes	430 (166)	0.504 (0.072)	0 (0)	Sequoia NF (Inyo NF)
2	yes	936 (361)	0.622 (0.110)	466 (180)	Sequoia NF
3	yes	985 (380)	0.564 (0.937)	464 (179)	Sequoia NP (Sequoia NF)
4	yes	751 (290)	0.551 (0.090)	334 (129)	Sierra NF
5	yes	1,096 (423)	0.574 (0.097)	611 (236)	Sierra NF (Yosemite NP)
6	no	321 (124)	0.542 (0.103)	172 (66)	Yosemite NP (Stanislaus NF)
7	no	1,357 (524)	0.573 (0.094)	587 (226)	Stanislaus NF (Yosemite NP)

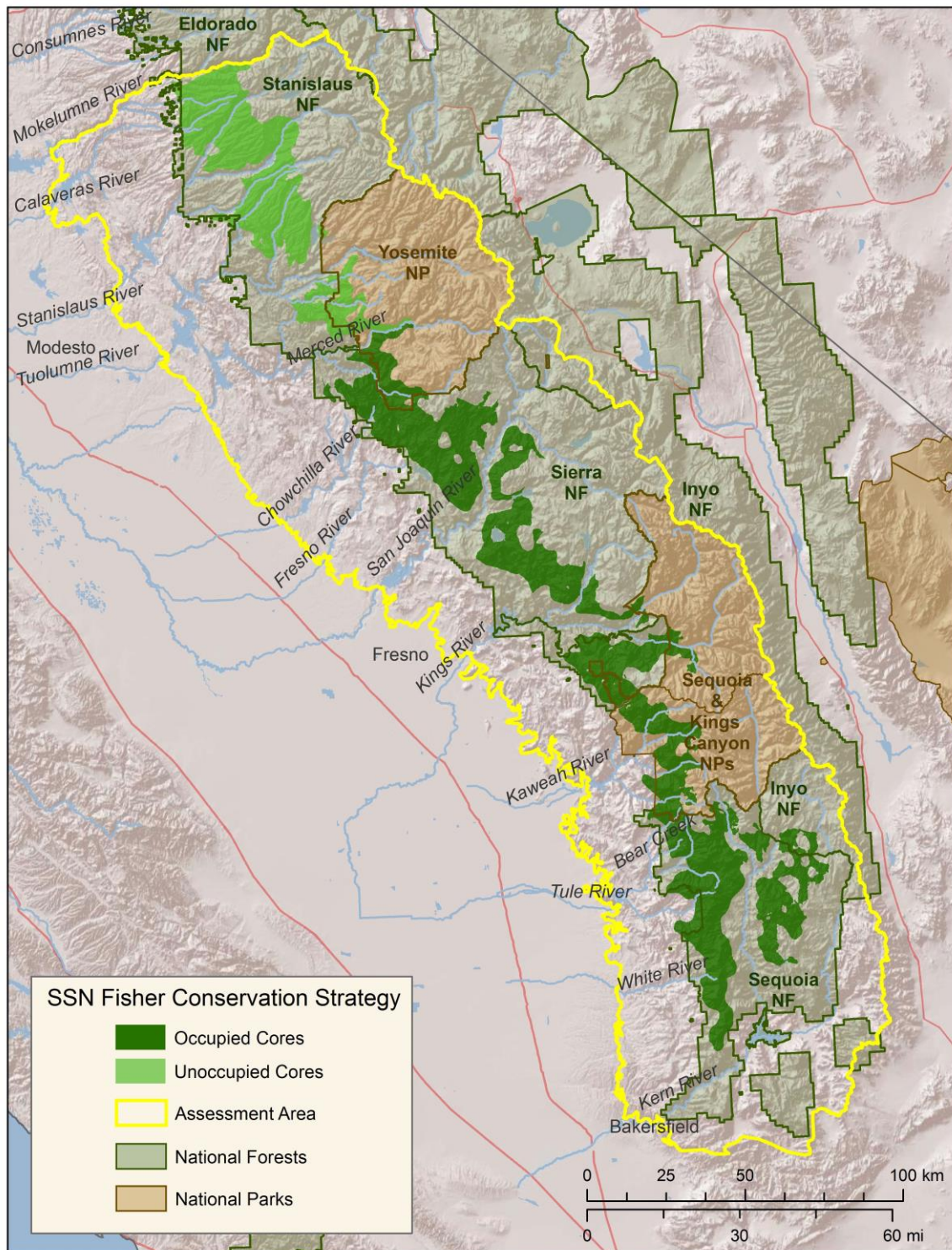


Figure 14. Distribution of occupied and unoccupied fisher habitat core areas in Southern Sierra Nevada Fisher Conservation Assessment area. Potential habitat cores were mapped as contiguous polygons having a predicted probability of fisher occupancy >0.41 (“strongly selected for”) large enough to support ≥ 5 adult females (38.75 km^2 or 14.96 mi^2) based on a Maxent distribution model and regional fisher monitoring data (methods described in Appendix A).

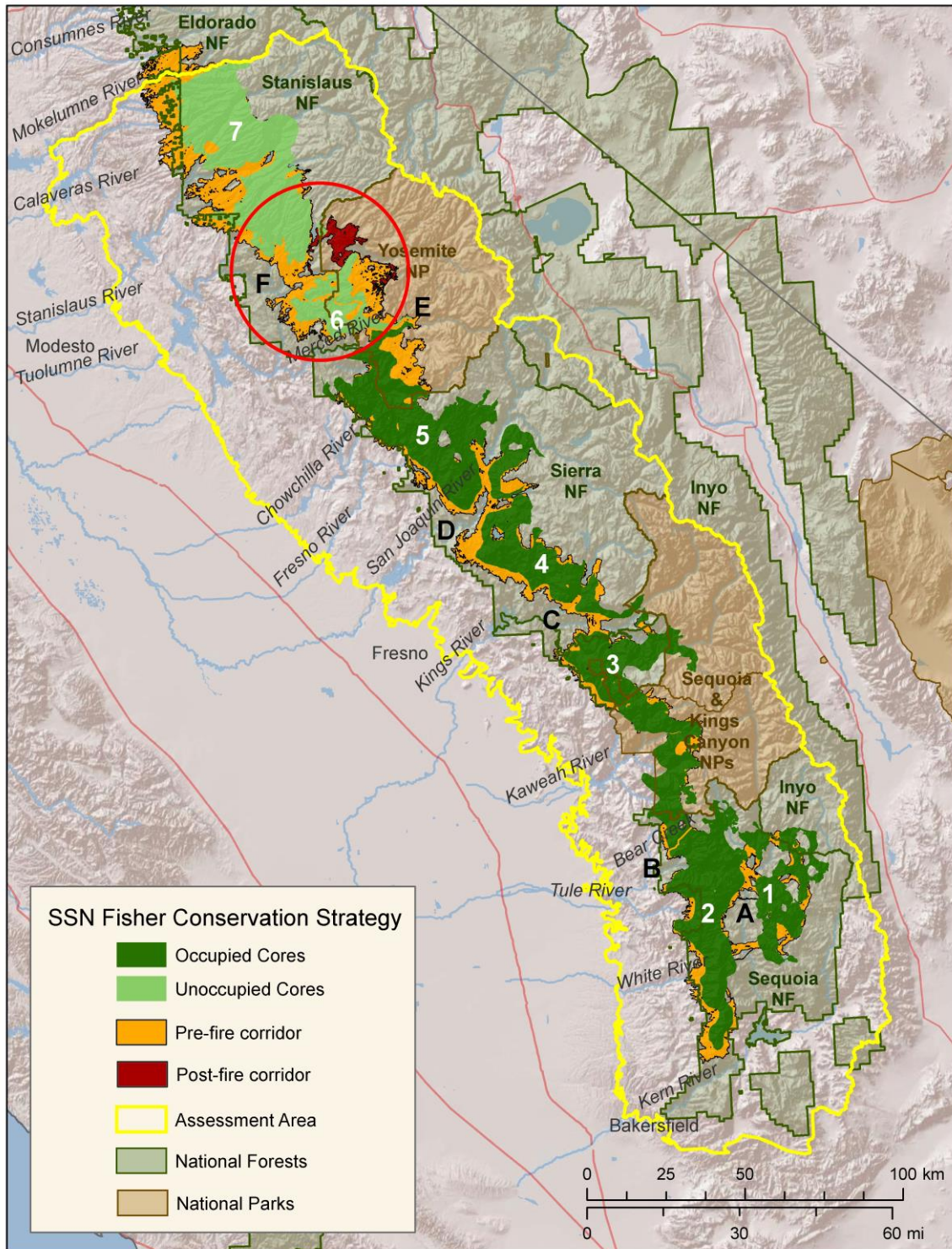


Figure 15. Modeled fisher connectivity or dispersal habitat. Represented as the union of 50-km (30-mi) normalized least-cost corridors produced using 3 different assumptions about fisher movements between cores (see Appendix A). The red circle indicates a significant shift in predicted dispersal corridors due to the 2013 Rim Fire, which burned much of the predicted pre-fire dispersal habitat at high severity. Habitat core areas are numbered; linkages between cores are lettered.

5.1.1 Core Area 1

- Geography.** This core (Figure 16) is mostly on the Kern Plateau in the southeastern portion of the assessment area and is the only core not on the west slope of the Sierra Nevada. It is largely within Sequoia National Forest, with a small portion on the Inyo National Forest. The Kern Plateau has unique environmental conditions, due to differences in climate, geology, and vegetation, compared to the west-slope cores (Miles and Goudey 1998). It receives less annual precipitation (~25-76 cm or ~10-30 in.) than forests in other cores (~102-152 cm or ~40-60 in.), and the vegetation is somewhat more open. The lesser accumulation of snow in this core may explain why fishers occupy higher elevations here than elsewhere in the assessment area and why martens (which are more snow-adapted than fishers) are absent (J. Tucker, unpublished data).
- Habitat Condition.** This is the smallest occupied core area, has the lowest predicted habitat value of any core (Table 13), and appears to lack potential resting and denning habitat. However, models may not predict habitat value here as well as in other cores due to the unique geography and dearth of fisher data from Core 1 with which to calibrate models. Consequently, habitat condition should be evaluated with caution in Core 1, as fishers experience a different range of environmental conditions compared to other cores and may select habitat based on different criteria. This core has experienced frequent fires in recent decades, including some large, severe fires, and some mosaic fires within the natural range of variability that were managed for resource values (M. Meyer, USFS, personal communications). The resulting vegetation is a patchy mosaic of forest stand ages and sizes intermixed with open areas and shrublands. Pinyon-juniper woodlands, canyon oak woodlands, and birch-leaf mountain mahogany are a greater component of the vegetation of the Kern Plateau than other portions of the assessment area, and California black oak, an important component of fisher habitat where it occurs, is rare or absent from the Kern Plateau.
- Population Condition.** Occupancy modeling using data from the regional fisher monitoring program shows this core to have the lowest fisher occupancy rates in the region (Zielinski et al. 2013a), suggesting lower population densities here than in other cores. Tucker et al. (2014) did not find evidence of population subdivision between Cores 1 and 2, despite the apparent break in habitat contiguity across the Kern River valley, suggesting that connectivity may be relatively good through the Golden Trout Wilderness Area, across the Little Kern and Kern River canyons above their fork (Linkage A on Figure 16).
- Management Implications.** Core 1 needs additional research and more intensive monitoring to better understand habitat selection and population characteristics. Management should favor tree growth, increased canopy cover, and essential fisher habitat elements.

5.1.2 Core Area 2

- Geography.** This core includes the southwestern tip of the Sierra Nevada and Greenhorn Mountains—between the Kern River on the east and south and Bear Creek in the Tule River watershed in the north (Figure 16). It is mostly on Sequoia National Forest and Giant Sequoia National Monument, but also extends north and east through the Golden Trout Wilderness to the Kern River, upstream of the Kern/Little Kern fork. Very steep terrain creates relatively abrupt environmental gradients and concentrates the dense mixed-coniferous forest conditions favored by fishers into a relatively narrow band on the west slope and near the crest of the Greenhorn Mountains.

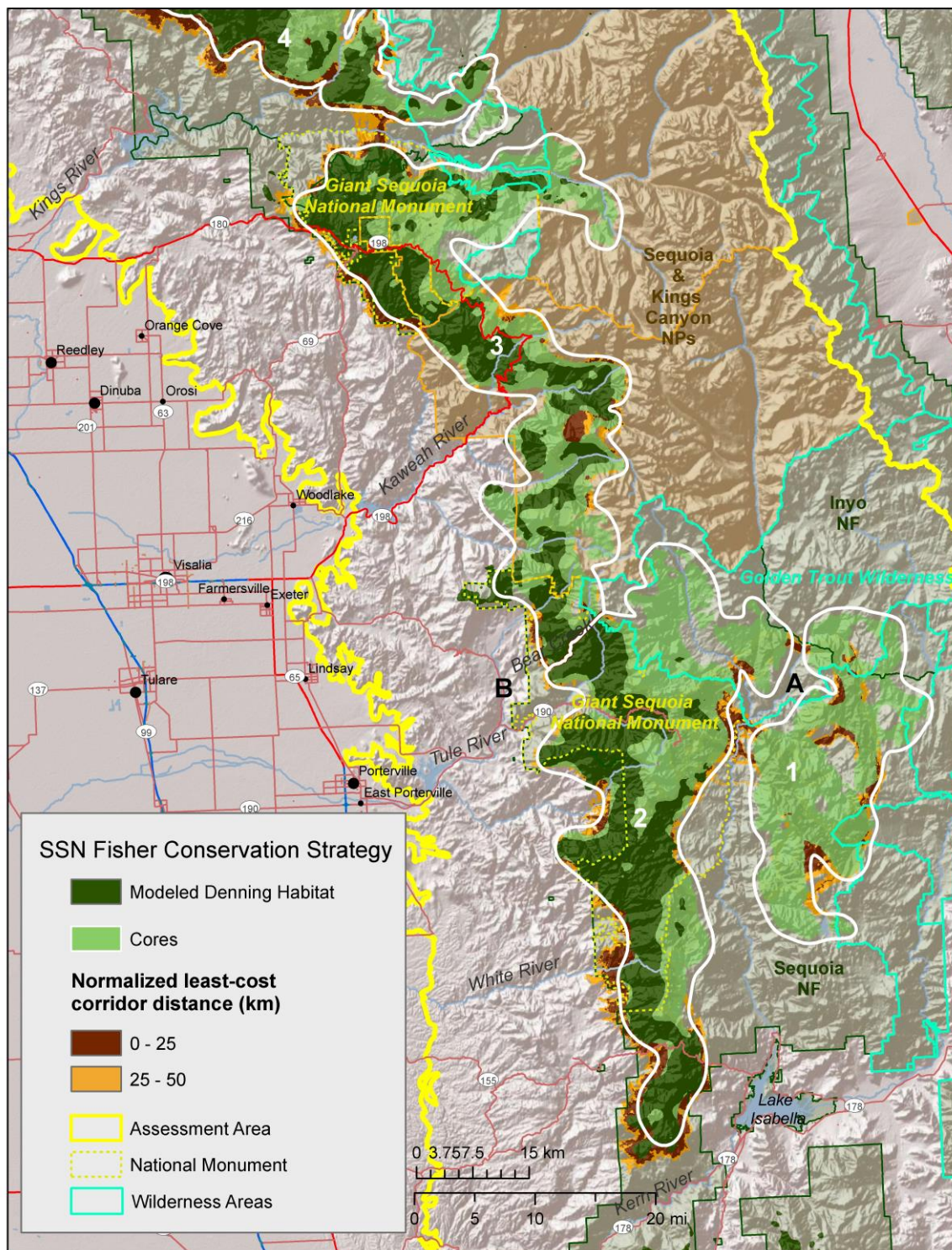


Figure 16. Fisher Cores 1, 2, and 3 showing modeled denning and dispersal habitat. Dispersal (or connectivity) habitat was modeled as normalized least-cost corridors of 0-25 km (highest value) and 25-50 km (secondary value). Note the absence of modeled denning habitat in Core 1. See Appendix A for methods.

- **Habitat Condition.** Core 2 has the highest predicted average habitat quality in the assessment area (Table 13), with high-quality denning habitat distributed as a relatively narrow band on the west slope of the Sierra Nevada and Greenhorn Mountains. Vegetation in the northeastern portion of Core 2, in and near the Golden Trout Wilderness Area, is generally more open than that on the steeper west slope forests and lacks denning habitat (based on modeling). Due to less intensive historic logging compared with other cores, Core 2 supports relatively abundant old-growth mixed-conifer forest, scattered giant sequoia groves, and large sugar pines, with high basal areas, high diversity of tree diameter classes, and dense canopy cover (>70%).
- **Population Condition.** This core, combined with Core 3, has the highest recorded fisher occupancy rates (Zielinski et al. 2013a) and highest genetic diversity (Tucker et al. 2014) in the assessment area. Genetic patterns suggest this area may have served as a refuge for fishers following European settlement—perhaps due to the steep terrain that limited human impacts (Beesley 1996). Genetic evidence and survey results both suggest that the fisher population expanded northward from this area during the late 20th century. Zielinski et al. (2004a) found smaller home ranges in the western portion of Core 2 than in other regions, possibly due to higher quality habitat (dense mixed-coniferous forests, large trees, and abundant black oak). Tucker et al. (2014) did not find evidence of population subdivision between Cores 1 and 2, suggesting good connectivity between Cores 1 and 2 across the Golden Trout Wilderness Area, the Great Western Divide, and the Little Kern and Kern River canyons, despite the somewhat more open forest conditions.
- **Management Implications.** Much of Core 2 is within the Giant Sequoia National Monument and Golden Trout Wilderness, where management calls for restoring ecological processes and patterns that enhance forest ecosystem resilience to stressors (e.g., uncharacteristic wildfire, climate change) and protecting and enhancing wildlife habitat through prescribed fire and limited mechanical treatments.

5.1.3 Core Area 3

- **Geography.** Core 3 (Figure 16) is largely within Sequoia National Park and adjacent portions of Sequoia National Forest and Giant Sequoia National Monument. It is on the west slope of the Sierra Nevada, including some very steep slopes in Sequoia National Park. Cores 2 and 3 were originally delineated as one continuous core (Figure 14), but the FTT split them into 2 cores at Bear Creek (Linkage B on Figure 16) to reflect the genetic discontinuity identified there by Tucker et al. (2014). It is possible that this genetic subdivision is an artifact of the historical population contraction-expansion across the region, rather than reflective of a current dispersal impediment (J. Tucker, personal communication).
- **Habitat Condition.** This core has relatively high habitat value, but the habitat band is fairly narrow in Sequoia National Park due to the steep elevation gradient. Modeled denning habitat is also narrow and somewhat fragmented by steep canyons and open ridges. Nevertheless, much of Core 3 has very high habitat value due to mature forest conditions and numerous giant sequoia groves and other mixed-coniferous forests supporting high basal area, dense canopies, and abundant black oaks.
- **Population Condition.** Fishers have not been studied intensively in this core. Together, Cores 2 and 3 have the highest measured fisher occupancy rates in the southern Sierra Nevada (Zielinski et al. 2013), which suggests that they support relatively high population densities.

- **Management Implications.** Much of this core is within Sequoia National Park and Sequoia National Monument, where management calls for restoring natural ecological processes and patterns to enhance forest ecosystem resilience to stressors (e.g., uncharacteristic wildfire, climate change) and protect or enhance high-value wildlife habitat. Management treatments include prescribed fire, wildfire managed for resource objectives, and limited mechanical treatments. Other areas on Sequoia National Forest should be managed to restore and maintain old forest conditions while enhancing resiliency to fires, climate change, and other disturbances.

5.1.4 Core Area 4

- **Geography.** This core (17) is between the Kings and San Joaquin river valleys on Sierra National Forest (High Sierra Ranger District). It is on the west slope of the Sierra, on slightly less steep terrain than Cores 1-3. Core 4 is partially segmented, with a relatively broad habitat area in the central portion and somewhat narrower and more fragmented conditions near the southern end (between the main stem and the North Fork of the Kings River) and the northern end (between the San Joaquin River and Big Creek/Huntington Lake).
- **Habitat Condition.** This core has moderate predicted fisher habitat value (Table 13). Denning habitat is concentrated at lower elevations (western portions) of the core. It is relatively contiguous and broadly distributed in the central portion, but occurs in smaller and more fragmented patches in the northern and southern “tails” of the core near the San Joaquin River (vicinity of Shaver Lake, Huntington Lake, and Kaiser Wilderness Area) and near the Kings River. In 2013, the 9,300 ha (23,000-ac) Aspen Fire burned much of the northern segment of the core and potential dispersal habitats connecting Cores 4 and 5 in a mosaic of mostly low to moderate severity, with some high-severity patches. Portions of the burned area were salvage-logged. Monitoring fisher habitat use in the fire’s aftermath could provide valuable information on effects of fire and salvage operations.
- **Population Condition.** This area has moderate fisher occupancy rates (Zielinski et al. 2013a) and genetic diversity (Tucker et al. 2014). Tucker et al. (2014) found a genetic subdivision at the Kings River (Linkage C on Figure 17), suggesting that the Kings River Canyon is a significant impediment to dispersal between Cores 3 and 4. Although the regional monitoring program has detected fishers throughout this core, female fisher home ranges are concentrated within modeled denning habitat and lacking in the somewhat more open, higher-elevation eastern portions of the core (Figure 17), which are used by males. Fisher ecology has been studied intensively in the central portion of Core 4 since 1995 (Boroski et al. 2002; Mazzoni 2002; Jordan 2007), including studies for the KRFP (Purcell et al. 2009; Thompson et al. 2010, 2012). Extrapolating a modal population density calculated for the KRFP study area by Thompson et al. (2012) provides a population estimate of about 78 fishers in Core 4. The northern area in and around the Kaiser Wilderness is partially separated from the rest of the core near Big Creek/Huntington Lake. Genetic evidence suggests some slight genetic differentiation there, but it is unclear whether this is due to dispersal impediments or is a founder effect from population expansion (Tucker et al. 2014).

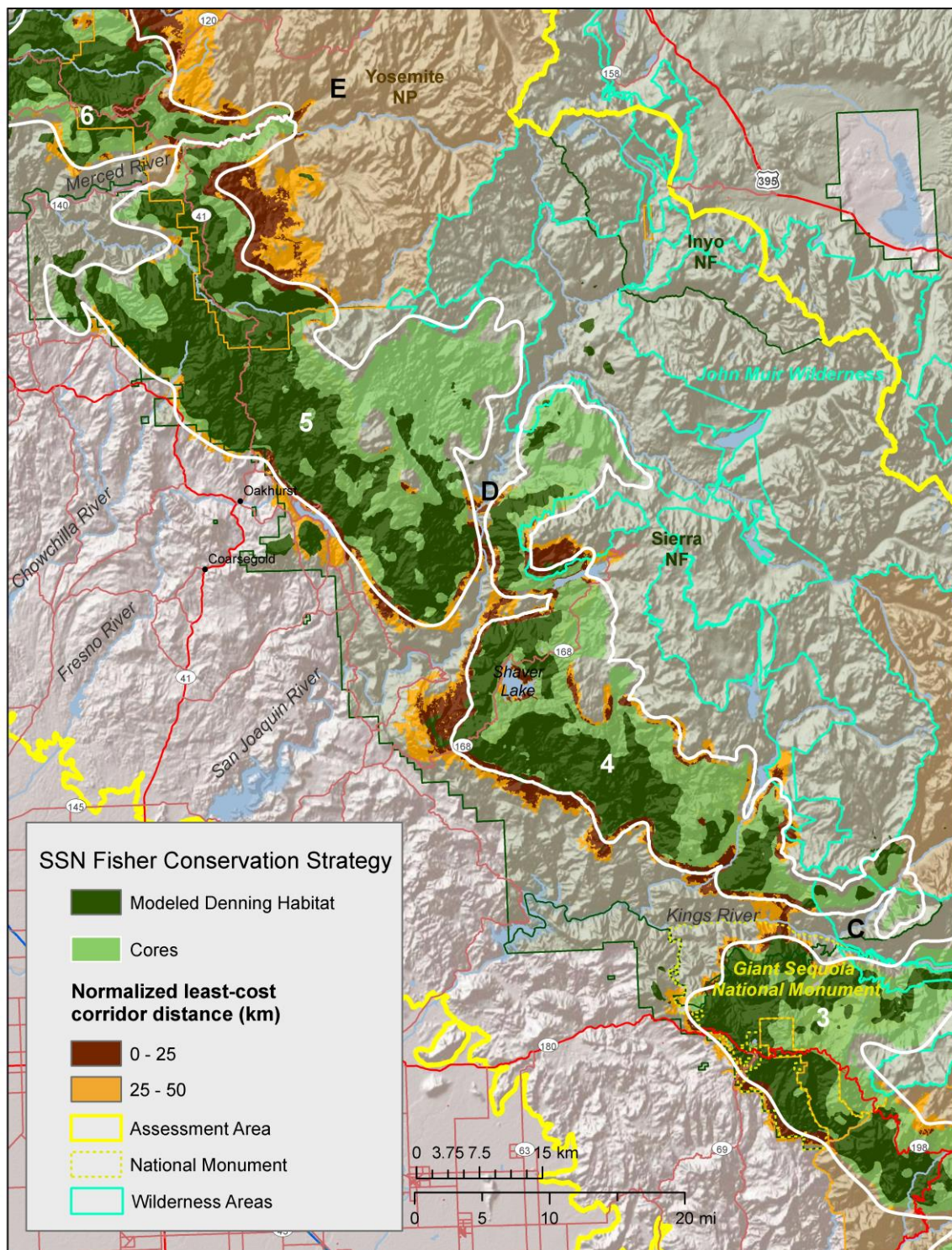


Figure 17. Fisher Cores 4 and 5 showing modeled denning and dispersal habitat. Dispersal (or connectivity) habitat was modeled as normalized least-cost corridors of 0-25 km (highest value) and 25-50 km (secondary value). See Appendix A for methods.

- **Management Implications.** Vegetation in much of the central portion of this core is being treated by a variety of mechanical and prescribed fire treatments, coupled with monitoring of fisher responses to the habitat changes. Adaptive management is being used to make iterative improvements in project siting and design. The rich data set and ongoing research associated with the Dinkey Collaborative Forest Landscape Restoration project and the Kings River Fisher Study suit this core for testing new tools and approaches to fisher habitat management. Near the two “tails” of the core, south of the North Fork of the Kings River and north of Shaver Lake, minimize further habitat loss or fragmentation and maintain or improve dispersal potential between Cores 3 and 4 to the south (across Kings River Canyon, Linkage C), and Cores 4 and 5 to the north (across San Joaquin River, Linkage D).

5.1.5 Core Area 5

- **Geography.** This core (Figure 17) lies between the San Joaquin and Merced rivers, primarily on Sierra National Forest (Bass Lake Ranger District) and the southwestern portion of Yosemite National Park. It is one of the broadest cores due to generally less steep terrain compared with the southern 3 cores. Core 5 has a fairly extensive wildland-urban intermix (WUI) and numerous human dwellings at risk from wildfires, especially along the Highway 41 corridor, which runs north-south through the heart of the highest-value fisher denning habitat. High value resources include year-round and vacation residences, recreational facilities including motels and camps, and the communities of North Fork, Bass Lake, Sugar Pine, Cedar Valley, Wawona, and Fish Camp. The south rim of Yosemite Valley appears to be the northern boundary of currently occupied fisher habitat in the southern Sierra Nevada.
- **Habitat Condition.** Core 5 has relatively high predicted habitat quality and is the largest and broadest of the occupied cores. This area was heavily railroad logged from the late 1890s until 1932. Resulting second growth stands are mostly 90-110 years old, with a relatively high proportion of shade-tolerant incense cedar and white fir. It has relatively abundant and contiguous denning habitat at lower elevations on the western portions. Denning habitat is broader in the northern half and narrower and slightly more fragmented in the southern half. Heavily traveled Highway 41/Wawona Road passes north-south into Yosemite National Park through the heart of fisher denning habitat. Recent and ongoing mechanical thinning and prescribed fire have altered forest structure in significant portions of this core, and the SNAMP is monitoring fisher responses to these treatments. The easternmost portion of Core 5 and connecting habitats to Core 4 (Linkage D on Figure 17) burned in the 2014 French Fire, directly across the San Joaquin River from the 2013 Aspen Fire. Effects of these fires on habitat value and inter-core dispersal potential should be studied. Ponderosa pines, sugar pines, and black oaks have been dying in recent years due to drought, disease, and insect attack. This mortality is likely to continue and potentially accelerate with climate change.
- **Population Condition.** Core 5 is the northernmost core occupied by a breeding fisher population. It has moderate occupancy rates (Zielinski et al. 2012) and lower genetic diversity than the other occupied cores, probably due to the relatively recent northward expansion of fishers from the southern cores (Tucker et al. 2014), coupled with low dispersal rates across the San Joaquin River watershed, between Cores 4 and 5 (Linkage D on Figure 17). Female fisher home ranges are concentrated within modeled denning habitat and lacking in the somewhat more open and higher-elevation eastern portions of the core (Figure 17). R. Sweitzer (unpublished analysis) estimated the mean population size in Core 5 in recent years at ~87 (range 77-97) fishers. The SNAMP fisher project studied this core intensively in 2007-

2014, with some aspects of the study continuing as the Sugar Pine study. Survey results and genetic evidence suggest that fishers re-colonized this area from the south in the latter half of the 20th century and that the population has expanded in size and distribution in recent decades (Zielinski et al 1995; Tucker et al. 2014). However, the northward expansion appears to have stalled at the Merced River (Yosemite Valley; Linkage E on Figure 17) since then, possibly due to a combination of high mortality rates (reducing the potential number of dispersers) and dispersal impediments associated with Yosemite Valley (e.g., steep slopes, sparse forest, and heavily traveled roads).

- **Management Implications.** Improve habitat connectivity and potential for fishers to disperse north across Yosemite Valley to Core 6 and south across the San Joaquin River to Core 4, and reduce mortality factors (e.g., roadkill on Highway 41/Wawona Road, rodenticide poisoning) to facilitate population expansion. Detecting and cleaning up trespass marijuana grow sites should be a high priority. Manage vegetation to maximize integrity of female home ranges and potential denning habitat while restoring more resilient forest conditions.

5.1.6 Core Area 6

- **Geography.** This core (Figure 18) straddles the western portion of Yosemite National Park and the southeastern portion of Stanislaus National Forest between the Merced River (Yosemite Valley) and the Tuolumne River. It is the smallest of the delineated cores and is separated from Core 5 by Yosemite Valley (Linkage E).
- **Habitat Condition.** This core has moderate predicted habitat quality (Table 13), including some modeled denning habitat. Although portions of this core support dense, mature forest stands, including scattered giant sequoia groves within the park, other portions are of low to moderate habitat quality, in part due to a complex recent fire, tourism, and management history that has replaced mature forest cover with early seral vegetation, shrublands, and plantations. Historic railroad logging impacted large areas, as have more recent large, severe fires, including the 1990 Arch Rock Fire, 1990 Steamboat Fire, and 1996 Ackerson Fire. The 105,200-ha (260,000-ac) Rim Fire in 2013 burned the western and northern portions of this area, much of it in high-severity, stand-replacing fire.
- **Population Condition.** Fishers were historically present in this core (Grinnell et al. 1937) and have been observed occasionally in recent decades (Chow 2009); however, systematic monitoring studies have not detected fishers here, and there is no evidence of an established breeding population. Because females are apparently much more dispersal-limited than males, it is likely that most recent observations of fishers in this core are lone males dispersing from Core 5 that fail to find mates in Core 6.
- **Management Implications.** Because this core currently does not support a breeding fisher population, it may represent an opportunity to test management actions that could otherwise harm fishers or their habitat in the short term with the objective of restoring more resilient and higher-value habitat conditions in the longer term. However, this potential opportunity to use more intensive, extensive, or experimental management actions must be balanced against the goal of fisher dispersing into and breeding here as soon as possible. Sensitivity tests using a spatially explicit fisher population model could inform decisions about how to balance management to promote population expansion versus actions to restore more resilient forest condition. Decreasing mortality factors in Core 5 and managing for habitat connectivity between Cores 5 and 6 (Linkage E) may facilitate natural re-establishment of a breeding population in Core 6 via dispersal across Yosemite Valley.

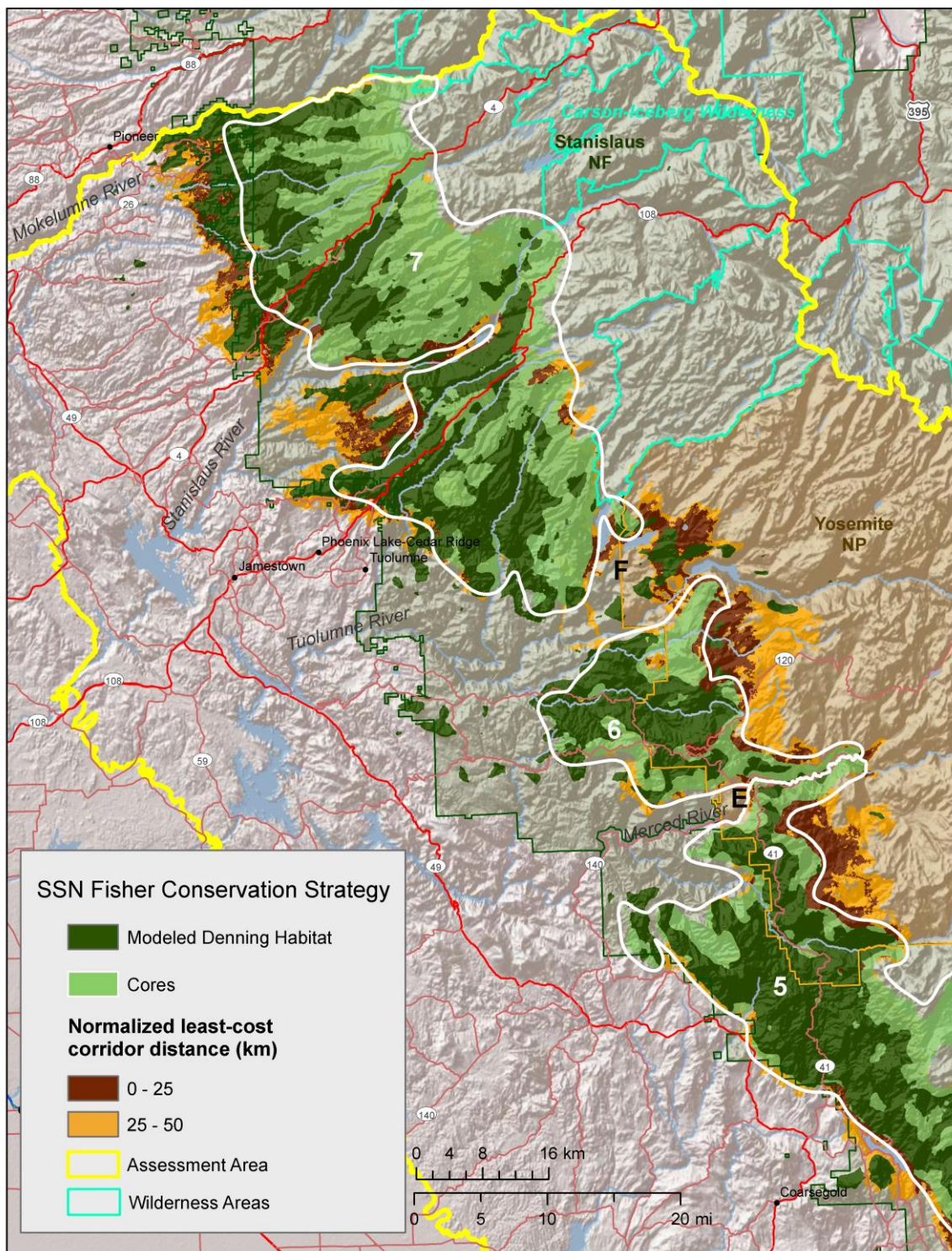


Figure 18. Fisher Cores 6 and 7 showing modeled denning and dispersal habitat. Unlike Cores 1-5, Cores 6 and 7 are not currently occupied by breeding fisher populations. Dispersal (or connectivity) habitat was modeled as normalized least-cost corridors of 0-25 km (highest value) and 25-50 km (secondary value) after the 2013 Rim Fire. See Appendix A for methods.

5.1.7 Core Area 7

- **Geography.** Core 7 (Figure 17) is the largest and broadest of the delineated cores, with generally gentler terrain than the southern cores. It is currently unoccupied, though it supports potential habitat, mostly on the Stanislaus National Forest, along with various other state, local, and private lands (e.g., Sierra Pacific Industries) and Yosemite National Park. Two east-west highways cross the Sierra here.
- **Habitat Condition.** There is high uncertainty about true habitat potential here, as this area has not supported a fisher population for many decades and it is unclear how well available habitat models extrapolate to this region. This area has experienced more large fires and more intensive forest management in the form of harvest, post-fire salvage, and tree planting in the recent past than other cores. The 2013 Rim Fire burned the southern end of this core, and the large 1987 Stanislaus Fire Complex burned a significant area, replacing many ponderosa pine stands with chaparral and leaving patches of hardwoods and scattered pines. The Rim Fire also burned with high severity across modeled dispersal habitat between Cores 6 and 7, shifting potential dispersal habitats upslope in the Tuolumne River watershed.
- **Population Condition.** Although fishers have not been observed in this core in many decades, it was almost certainly occupied by fishers during the early 20th century (Grinnell et al. 1937). The FTT believes it could be occupied again given appropriate management actions.
- **Management Implications.** Because this core is not currently occupied by a breeding population, and it likely will take several decades before natural re-establishment of a population here, Core 7 may represent an opportunity to experiment with relatively intensive, extensive, or experimental vegetation treatments without short-term harm to fishers. The goal should be to restore forest resiliency and increase fisher habitat potential within a few decades, in anticipation of continued northward expansion of the population. Post-fire management on Forest Service lands within the Rim Fire includes some salvage of dead trees, but retaining the largest diameter snags and isolated clusters of trees within salvage units. Reforestation should promote spatial heterogeneity of trees using diverse and fire-resilient species mixes, GTR-220/237 principles that consider topographic position and soils, and wildlife habitat values.

5.2 Linkage Areas

Linkage areas were modeled as representing the least costly or risky dispersal corridors between core habitat areas using least-cost corridor techniques (McRae and Kavanagh 2011) and varying assumptions about fisher dispersal decisions (see Appendix A for methods). Linkage areas contain dispersal habitat, which is unlikely to provide all requirements found in live-in habitat, but which may provide prey and cover for limited periods, especially during dispersal. In general, some overhead cover, whether from trees or shrubs, seems necessary for dispersal habitat; and open vegetation types, barren areas, and large water bodies (lakes, reservoirs) may impede or prevent movement. Females are more constrained in their movements than males, and genetic evidence and telemetry data suggest that females are highly constrained to dispersing within forests with dense canopies and large trees (Tucker 2013). Female dispersal is critical to expanding the population into currently unoccupied areas, and dispersal by both sexes helps maintain genetic diversity.

The following text describes the nature of each linkage area (as labeled on Figures 16-18) and implications for management actions to increase dispersal potential.

5.2.1 Linkage A

- **Description.** Linkage A is a generally east-west, multi-strand connection across the Kern River watershed between Cores 1 and 2, with several strands near the forks of the Kern (Kern River/Little Kern River junction) and the Golden Trout Wilderness. Lack of genetic subdivision found here by Tucker et al. (2014) suggests that dispersal across the Kern River watershed may not be overly constrained, although time lags to detect barriers from genetic data make this uncertain (J. Tucker, personal communication).
- **Management Implications.** Evaluate potential linkages between Cores 1 and 2 in the field to refine model predictions. Because the most likely dispersal areas are in or just downstream of the Golden Trout Wilderness, and genetic results suggest that dispersal may be adequate between Cores 1 and 2 (Tucker et al. 2014), little if any management changes are recommended. Wildfire is expected to continue playing its natural role in maintaining vegetation heterogeneity.

5.2.2 Linkage B

- **Description.** Linkage B represents the genetic subdivision identified by Tucker et al. (2014) near the Mountain Home Demonstration State Forest, for which Bear Creek in the Tule River watershed was used as the geographic break between Cores 2 and 3. There is no obvious reason for the genetic discontinuity here, as there are no clear dispersal impediments. The genetic subdivision may be a historical legacy of a southward population contraction followed by northward re-expansion across this area, rather than a reflection of current dispersal impediments (J. Tucker, personal communication).
- **Management Implications.** Because there are no obvious dispersal impediments in this area, no specific restoration actions are recommended. Maintain natural, mature forest conditions in and near the Mountain Home Demonstration State Forest, and avoid creating major canopy breaks that could fragment the linkage.

5.2.3 Linkage C

- **Description.** Linkage C crosses the Kings River Canyon between Cores 2 and 3. Kings Canyon appears to be a significant dispersal impediment due to steep, mostly unforested slopes, consistent with landscape genetic patterns (Tucker et al. 2012, 2014). Connectivity models and aerial imagery show few good options for fishers to traverse the canyon. Narrow riparian strips along tributary creeks may offer sufficient cover for dispersal across the main stem of the Kings River Canyon (e.g., Rough Creek, Converse Creek), or fishers may cross farther upstream where there are smaller canyons and more continuous forest cover.
- **Management Implications.** Evaluate in the field the most likely crossings and impediments to fisher dispersal. Consider whether restoration actions may increase tree, shrub, or log cover in key locations. Manage to maintain high-quality fisher live-in habitat (especially denning habitat) on either side of the canyon (e.g., around Converse Mountain/Converse Basin to the south and Spanish Mountain/Rodgers Ridge to the north).

5.2.4 Linkage D

- **Description.** Linkage D connects Cores 4 and 5 across the San Joaquin River. Connectivity models suggest the most likely river crossing is at or below the Mammoth Pool dam, which is consistent with observations of repeated crossings there by 1 radio-collared male (R. Sweitzer,

unpublished data). The 2013 Aspen Fire burned much of the potential dispersal habitat on the east side of the San Joaquin River, and the 2014 French Fire burned the west side (Figure 19). High-severity burn areas were patchy in both fires, probably reducing but not totally disrupting dispersal potential between the cores. Reevaluate the linkage when vegetation burn severity data are available for the French Fire, ~1 year post-fire.

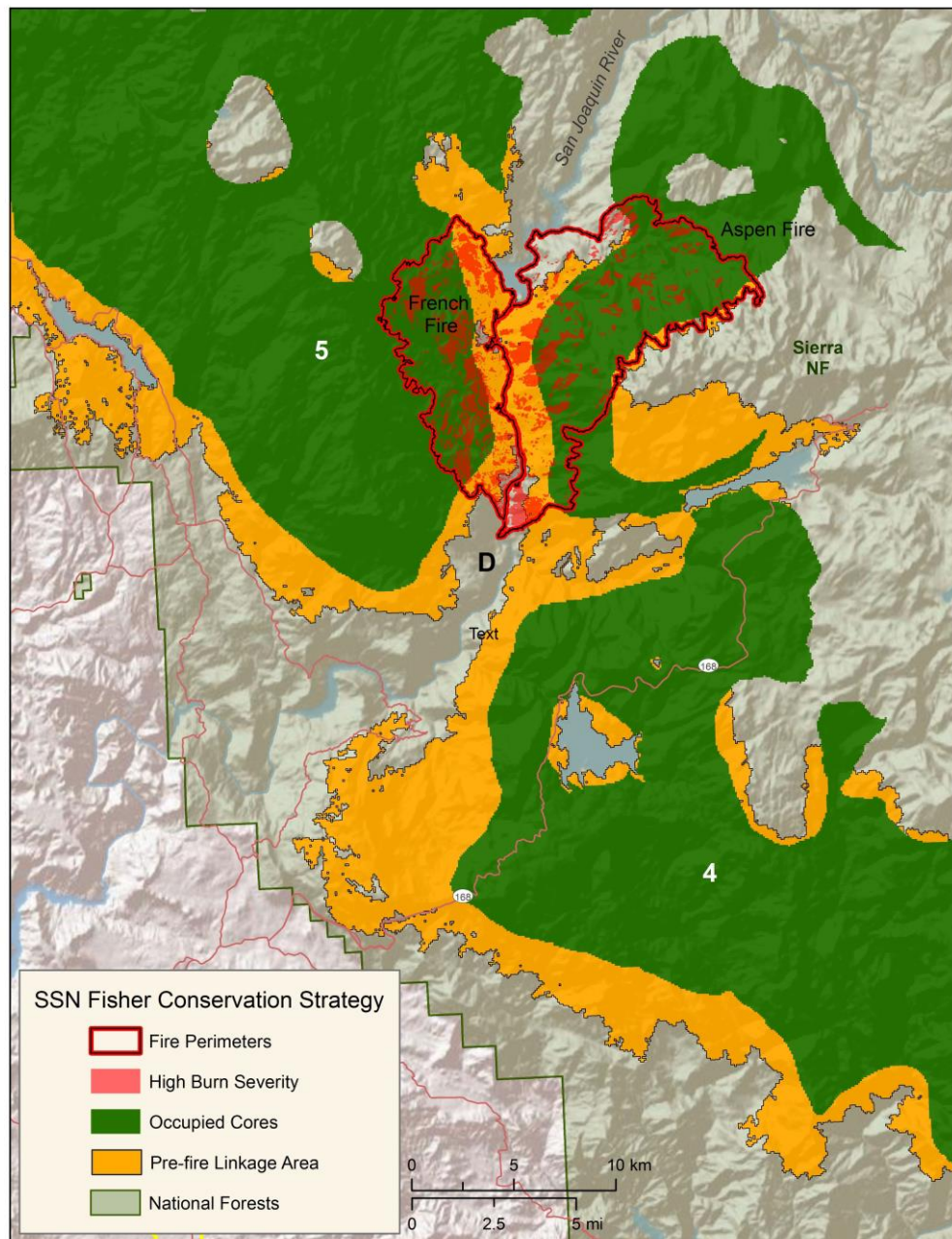


Figure 19. The 2013 Aspen Fire and 2014 French Fire in Linkage D and Cores 4 and 5 showing severely burned patches. Together the fires affected the entire width of likely fisher dispersal habitat and may have moderately fragmented the linkage due to patches of stand-replacing fire.

(Source: USFS, Pacific Southwest Region, 2014, Vegetation Burn Severity, 1984-2013 for the 2013 fire, <http://www.fs.usda.gov/detail/r5/landmanagement/gis/?cid=STELPRDB5327833>; USFS, 2014, RAVG data bundle for the 2014 fire (CA3726811933420140728) <http://www.fs.fed.us/postfirevegcondition/>.)

- **Management Implications.** Using post-fire habitat modeling, field assessment, and radio-tracking, evaluate how the two recent fires and post-fire management actions (e.g., salvage logging) may affect Linkage D function. Avoid removing fisher habitat elements within the linkage area to favor structural complexity of recovering forest cover. Promote high quality fisher habitat (especially denning habitat) within and adjacent to burned areas on either side of the linkage.

5.2.5 Linkage E

- **Description.** Linkage E connects occupied Core 5 and unoccupied Core 6 across Yosemite Valley and the Merced River. Connectivity models and field inspection suggest the most likely fisher crossing here would be lower reaches of Yosemite Valley, just east of the Wawona Tunnel overlook in Yosemite National Park. This appears to be about the only portion of Yosemite Valley where fishers would not have to navigate extremely steep, unvegetated granite slopes and cliffs and could stay within relatively dense coniferous forest while crossing the valley. Alternatively, fishers might move along various tributary streams entering the Merced River downstream of the park and move upstream to cross the valley, or they might cross branches of the Merced upstream of Yosemite Valley. A mosaic of recent fires, including the 1990 Arch Rock and Steamboat fires, 2009 Big Meadow Fire, and 2014 El Portal Fire and Meadow Fire impacted portions of this linkage and adjacent habitat. Most of the anecdotal observations of fishers in the park, including multiple road-killed fishers, are along Wawona Road and other western portions of the park.
- **Management Implications.** Management should favor retention of forest cover across lower Yosemite Valley and reduce risks of severe, canopy-replacing fire in the western portion of the Park and adjacent portions of Sierra National Forest. Maintain overstory cover in all significant drainages ending in a culvert under Wawona Rd. Protect riparian areas and modify culverts as needed to facilitate fisher movement (e.g., install wildlife shelving above waterline in large box culverts). Explore opportunities for additional crossing improvements along the Highway 41/Wawona Road corridor, including retrofitting existing structures, installing new undercrossing structures (particularly where topographic conditions may funnel animals toward an undercrossing), or constructing a large vegetated overcrossing to facilitate movement of fishers and larger mammals. Monitor fisher use of road-crossing improvements in the park to inform further improvements. Monitor roadkill and use of road-crossing structures.

5.2.6 Linkage F

- **Description.** Linkage F connects unoccupied Cores 6 and 7 on the Stanislaus National Forest and the western edge of Yosemite National Park. The high-severity 2013 Rim Fire burned much of this potential linkage; as a result, the modeled post-fire least-cost corridor is shifted significantly eastward (upslope) (Figure 20), increasing the total distance and cost of travel between core areas. The pre-fire least cost path was 16.7 km (10.4 mi) between core edges, whereas the post-fire least cost path is 26.5 km (16.5 mi), with a nearly 3-fold increase in the cumulative cost of travel.
- **Management Implications.** Retain and promote fisher habitat elements, within both the pre- and post-fire linkage areas, and recovery of forest canopy, hardwoods, and shrubs, depending on site conditions. Much of the pre-fire linkage burned at high severity. Portions of the Rim Fire in Yosemite National Park are in congressionally designated wilderness areas and will not be treated with anything other than prescribed fire or managed wildfire. The park will consult ecologists and specialists to determine when and where to burn to promote forest succession in high severity patches. Post-fire management on Forest Service lands within the Rim Fire area

includes some salvage of dead trees, but retaining the largest diameter snags and isolated clusters of trees within salvage units. Restoration will likely include reforestation of some salvaged areas. Design planting schemes to promote future tree spatial heterogeneity using diverse and fire-resilient species mixes, GTR-220/237 principles that consider topographic position and soils, and wildlife habitat values. A carnivore connectivity plan is underway as part of the reforestation effort.

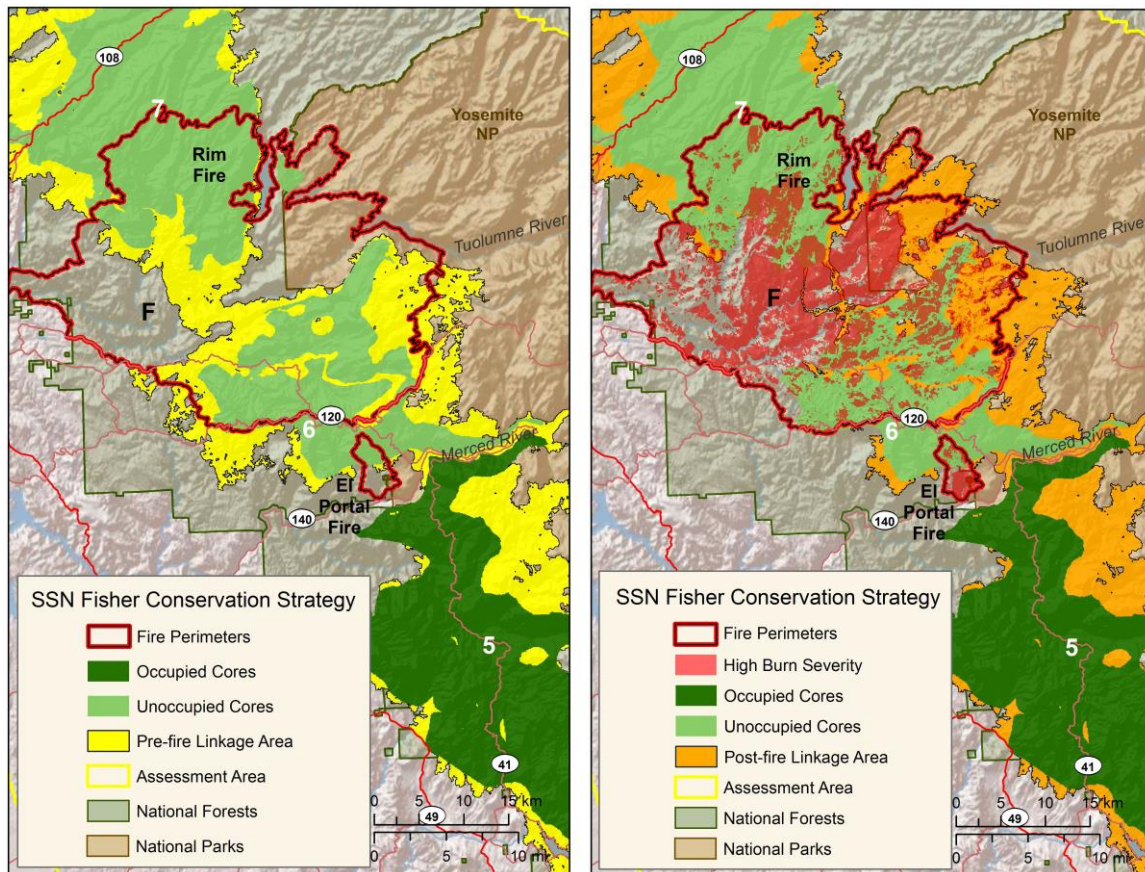


Figure 20. Modeled effects of the 2013 Rim Fire and 2014 El Portal Fire on potential dispersal habitat between fisher Cores 6 and 7 (Linkage F). The pre-fire least-cost corridor (left, yellow) was affected by large areas of severe (crown-replacing) wildfire (right, red), shifting the predicted corridor eastward (right, orange) to higher-elevation forests that did not burn at high severity. (Source: USFS, Pacific Southwest Region, 2014, Vegetation Burn Severity, 1984 to 2013, for the 2013 fire, <http://www.fs.usda.gov/detail/r5/landmanagement/gis/?cid=STELPRDB5327833>; USFS, 2014, RAVG data bundle for the 2014 fire (CA3726811933420140728) <http://www.fs.fed.us/postfirevegcondition>.)

6 Threats Assessment

Small and isolated populations are at heightened risk of extirpation (Shaffer 1981), and at <500 individuals the southern Sierra Nevada fisher population faces a variety of threats that can shrink, fragment, or degrade the quality of habitat, or directly impact the population via increased mortality or decreased reproduction. Low genetic diversity (Tucker et al. 2012) exacerbates the risks inherent to small population size. The specific spatial arrangement of fisher habitat in the assessment area—a long, narrow sequence of mid-elevation forests separated by major river canyons—elevates the potential for further population fragmentation, which can exacerbate other threats to the population. Although no single factor is likely to cause extinction in the near term, the cumulative effect of multiple factors could extirpate the population.

6.1 Habitat Loss and Fragmentation

Fisher habitat can be fragmented or reduced in quality, at least temporarily, by disturbances that change forest structure and remove essential fisher habitat elements. Historically, logging coupled with grazing and hydraulic mining had major adverse effects on fisher habitat in the Sierra Nevada (Lofroth et al. 2010). Currently, large, severe wildfires, in concert with drought, climate change, and insect outbreaks, are generally considered the largest threat to fisher habitat (Scheller et al. 2011; Lawler et al. 2012). Vegetation management, including tree harvest and thinning to reduce wildfire risks, can also adversely affect fisher habitat, but this risk may be offset if vegetation treatments reduce the risk that large, severe wildfires will affect habitat over larger areas and longer periods (Scheller et al. 2011). Vegetation changes due to management actions, fires, or other disturbances may have either positive or negative effects on fisher habitat depending on location, spatial and temporal scale, and their specific effects on forest structure and habitat elements.

6.1.1 Large and severe wildfires

Fire is a natural ecological process, and fires within the natural range of variation (Safford et al. 2012; Safford 2013) are generally considered beneficial to fisher habitat, especially over the long term, because they recruit essential habitat elements (e.g., snags, den cavities), increase abundance of some fisher prey species, and contribute to habitat resiliency. In contrast, very large and severe fires, outside the natural range of variation, can remove forest cover and fragment fisher habitat over large areas and long time periods. It is therefore important to clearly articulate the potential effects of different types of fires on fisher habitat—from prescribed fires and wildfires managed to benefit fishers, forest resiliency, or other resource values, to large, severe fires that can remove and degrade fisher habitat over large areas.

Terms like “high-severity” and “mixed-severity” fires are used in different ways in the literature. In this assessment and in the fisher conservation strategy, we use the following definitions, which are generally consistent with Barrett et al. (2010):

- **Low-severity fire**—Any surface fire replacing <25% of the dominant upper canopy layer in a succession class (or up to 25% basal area mortality of trees); as a result, low severity fires can open or maintain a given succession class.
- **Moderate-severity fire**—A fire replacing 26-75% of the dominant upper canopy layer in a succession class (or 26-75% basal area mortality of trees).

- **High-severity fire**—A fire replacing >75% of the upper canopy layer in a succession class (or >75% basal area of trees); sometimes referred to as a canopy-replacing fire or stand-replacing fire, especially if >95% of canopy or basal area is removed.
- **Mosaic fire**—A fire that burns with a mix of severity classes resulting in a post-fire mosaic of mostly low- to moderate-severity canopy effects, often with some smaller severely burned and/or unburned patches. This document uses the term mosaic fire rather than mixed-severity fire to avoid confusion with moderate-severity fire (see above) or mixed-severity fire regime (below), which is a more general landscape-scale pattern of variation in burn severity across time and multiple fires.
- **Mixed-severity fire regime**—A general pattern in which fires tend to be of mixed severity, cause selective mortality in the upper canopy layer (depending on different species' susceptibility to fire), or vary in time or space between low-severity and high-severity (stand-replacing) fires (Smith et al. 2000). Note that mixed-severity fire regime has a broader definition than mixed-severity fire and may include a general pattern of mosaic fires or of fires with varying severities over time.

A dynamic mix of fire sizes and severities within the natural range of variation is desirable to maintain forest resilience and fisher habitat values. Mosaic fires that include some smaller areas of severe fire increase vertical and horizontal habitat heterogeneity, reduce fuel loads, create dead-wood structures needed by many species, including fisher and black-backed woodpecker (*Picoides arcticus*, Saracco et al. 2011), and may increase some prey populations and foraging opportunities (Roberts et al. 2008; Kalies et al. 2010; Swanson et al. 2010). In contrast, large and severe wildfires that kill the majority of standing trees can negatively affect fisher habitat by removing canopy cover and essential habitat elements (Scheller et al. 2011; Thompson et al. 2011); these effects may persist for many decades until canopy cover and large trees regrow sufficiently (Collins and Roller 2013). In light of the precarious nature of the southern Sierra Nevada fisher population and its narrowly distributed habitat, fires that burn with high severity over large areas pose a significant risk to the population by reducing and fragmenting habitat.

Vegetation treatments focused primarily on reducing fire spread rates or severity by breaking up canopy continuity, reducing ladder fuels, or removing woody structures can also reduce fisher habitat quality, at least locally and temporarily (Scheller et al. 2011; Thompson et al. 2011; Truex and Zielinski 2013). However, over the long term (measured in decades), fuel treatments may indirectly benefit fishers and other wildlife associated with dense, mature forests by reducing the probability that very large, severe fires will reduce and fragment forest habitat even more, over larger areas, and for a longer time (Scheller et al. 2011). Fuel treatments that also incorporate wildlife habitat goals—such as retention of tree clumps, multi-storied canopies, and dead-wood structures—may influence fire behavior and increase forest resiliency, but such treatments probably need to be larger than more intensive treatments to effectively reduce fire severity (Kennedy and Johnson 2014).

Average fire size in California mixed-conifer forests before Euro-American settlement has been estimated at <300 ha (750 ac), while the average over the last 25 years is closer to 1,500 ha (3,750 ac); and recent fires on USFS lands in California are much larger than that (Show and Kotok 1923; Taylor and Skinner 1998; Minnich et al. 2000; Taylor 2000; Beaty and Taylor 2001; Taylor and Solem 2001; Collins and Stephens 2007; Miller et al. 2012; Safford 2013; A. Taylor, Pennsylvania State University, unpublished data). Fire severity is also higher today than under pre-settlement conditions, with the average fire in modern mixed-conifer and yellow pine forests on USFS lands supporting 5 to

7 times more area of stand-replacing fire than fires before Euro-American settlement (Miller et al. 2009; Miller and Safford 2012; Mallek et al. 2013; Safford 2013). Fire size and fire severity have been trending up in low and mid-elevation forests on USFS lands over the last 20 to 30 years, and these trends have been linked to increasing forest fuels from historical forest management actions, fire suppression, and climate change (Miller et al. 2009; Miller and Safford 2012; Safford et al. 2012; Mallek et al. 2013). Recent fires in the Sierra Nevada have included some huge patches of stand-replacing fire, extending for thousands or even tens-of-thousands of acres. This is in direct contrast to the size of stand-replacing patches from active fire regime forests in reference landscapes of the Sierra Nevada (areas where the fire regime is minimally influenced by humans), where mean stand-replacing patch size is <4 ha (10 ac) and maximum patch size generally is ≤ 100 ha (250 ac) (Collins and Stephens 2010; Miller et al. 2012; Safford 2013). The 2013 Rim Fire included contiguous areas of stand replacement >12,140 ha (30,000 ac)—the largest yet recorded in the Sierra Nevada (USFS, unpublished data; <http://www.fs.fed.us/postfirevegcondition/index.shtml>). Large, contiguous areas of severe fire can result in the long-term replacement of conifer forest by shrubs, which are maintained by subsequent fires (Willken 1967; Biswell 1974; Bock and Bock 1977).

There is a dearth of scientific information on fisher use of burned areas; however, the evidence from habitat selection and long-term demographics studies suggests that fishers cannot meet all life requisites (e.g., establish home ranges or find sufficient resting and denning habitats) within large areas burned by high-severity fires. Hanson (2013) used locations of fisher scats, located by scat-detecting dogs, to evaluate the use of forested areas burned by a mosaic of fires during 2000-2009 on the Kern Plateau, including the 2000 Manter Fire, 2002 McNally Fire, and several smaller fires. Fisher scats were found inside the perimeters of fires that had burned 10-12 years previously, suggesting that fishers did not categorically avoid or select areas that experienced some tree mortality from fires that burned a decade or so previously. However, due to idiosyncratic definitions of “moderate” and “high” severity classes used by Hanson (2013)⁸, and subsequent combining of the 2 categories for statistical analyses, no conclusions can be drawn concerning the effects of moderate or severe fires on fisher habitat use. From inspection of fire history maps and Figure 1 in Hanson (2013), most fisher scats were found in unburned or lightly burned areas, and scats inside fire perimeters were mostly near edges rather than fire interiors. Hanson’s survey transects did not adequately sample large areas burned at moderate to high severity to draw any conclusions about their use by fishers; and evidence that fishers sometimes use post-burn habitats does not imply that they can establish home ranges and reproduce in such areas. Precisely how the McNally Fire and other fires on the Kern Plateau have affected fisher occupancy and abundance therefore warrants more study. Nevertheless, the regional occupancy results indicate that fishers have persisted in a landscape that has experienced a mosaic of low- to high-severity fires, albeit with the lowest recorded occupancy rates in the assessment area (Zielinski et al. 2013a).

6.1.2 Lack of fire as a natural disturbance process

The combination of effective fire suppression, tree harvest, and tree planting has substantially changed the composition and structure of forests over large areas of the Sierra Nevada. These changes affect habitat conditions for fishers and other wildlife as well as the way fires now burn on the

⁸Hanson (2013) defined “moderate” and “high” severity fires using lower mortality thresholds (>15% and >50% basal area mortality, respectively) than traditionally used to define severity classes (>25% for moderate and >75-90% for high, depending on the classification system used). Hanson then combined these two categories for analysis—statistically comparing areas with <15% basal area mortality of trees (no or very low severity) to areas with >15% basal area mortality.

landscape (McKelvey and Johnston 1992; SNEP 1996; Sugihara et al. 2006; Barbour et al. 2007; Safford 2013). Fire suppression began in earnest in the early 20th century, though some watersheds experienced decreased fire frequencies decades earlier, and fire suppression policies were effective at nearly erasing fire as an ecological force across much of the Sierra Nevada during the 20th century (Sugihara et al. 2006). Today, most of the fisher core areas have had almost no fire for >100 years, after experiencing centuries of frequent, mostly low-severity and mosaic fires (Safford and Van de Water 2013) (Figure 21). The long-term lack of fire as a natural disturbance process, combined with historical selective removal of large, fire-resistant trees, have led to increased fuel loads and densities of small, shade-tolerant, fire-intolerant trees (e.g., incense cedar and white fir, Agee and Skinner 2005). This also increases competition on less shade-tolerant species, such as ponderosa pine and black oak, which are adapted to a more frequent fire regime (Agee 1993).

Variability in forest composition and structure has been reduced by the absence of low- and moderate-severity fire effects, combined with previous even-aged forest management strategies (North et al. 2009; Purcell et al. 2012), resulting in more continuous, homogeneous forest than occurred historically (Skinner 1995; Agee and Skinner 2005; Hessburg et al. 2005). This reduction in spatial habitat complexity may reduce the diversity and abundance of some fisher prey (Roberts et al. 2008; Roberts et al. in press; Fontaine and Kennedy 2012; Kelt et al. 2014). Fire suppression also reduces specific, fire-related habitat elements, such as snags and fire scars that develop into cavities used by fishers for resting and denning. Lawler et al. (2012) found that moderate to high frequencies of low- and moderate-severity fire create and maintain snags and coarse woody debris over the long term. Roberts et al. (2008, in press) also found that a mix of burn severities, especially low to moderate severity, and large patches of unburned refugia, are important for maintaining the diversity and abundance of small mammal assemblages in forests of Yosemite National Park.

Finally, lack of fire as a natural disturbance process is a temporary phenomenon, because fire is inevitable in Sierra Nevada forests. The fine-scale mosaic pattern created by frequent fire is a key component of dry forest resilience (Churchill et al. 2013). Excluding fire from the mixed-conifer landscape for long periods has decreased this resilience, increasing the risk that, when an area finally does burn, it is likely to do so at greater-than-normal severity and scale, due to the buildup and increasing continuity of fuels. Thus, the absence of increased forest management actions to prepare the landscape for fire, and a dearth of managed fire, are likely to increase the probability that large, severe fires eventually will remove, degrade, and fragment fisher habitat over large areas. North et al. (2012) estimated that, at current rates of mechanical and prescribed fire treatments and wildfire, fuels are annually reduced on only 18-48% of the acreage that historically burned on USFS land and 17-46% on NPS land. This deficit in the acreage that burns or is treated annually is steadily increasing the amount of land susceptible to very large and severe wildfires when they do occur.

6.1.3 Post-fire habitat management

Vegetation management in burned forests varies widely with land ownership and management goals, and may include salvage logging, tree planting, and use of herbicides to reduce shrub growth and enhance tree growth. Salvage logging is intended to recover the economic value of burned timber, reduce hazards to human safety and structures, and reduce fuel accumulation (e.g., by removing standing dead trees). Tree planting and herbicide application may be used to speed the regrowth of commercial timber or to facilitate regeneration of desired habitat elements.

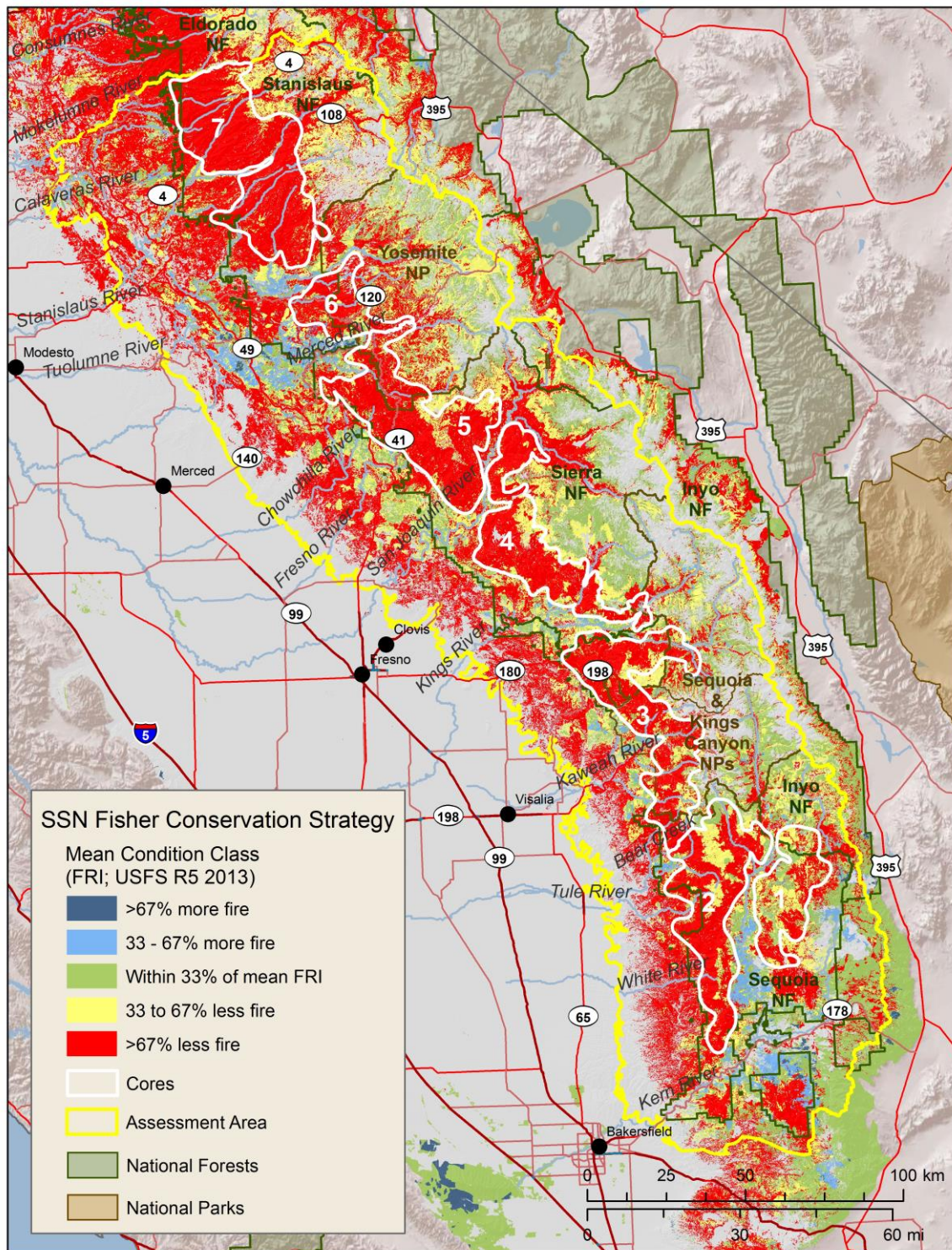


Figure 21. Fire Return Interval Departure (FRID) in the assessment area. Most fisher core habitat (white border) has experienced significantly less fire during the period 1908-2012 than during pre-Euro-American settlement, thus elevating fuel loads and risks of large, severe fires (Safford and Van de Water 2013: <http://www.fs.usda.gov/main/r5/landmanagement/gis>).

Such post-fire management actions can impact fisher habitat both negatively and positively. Post-fire salvage operations remove large dead-wood structures that contribute to forest structural complexity and are used for shelter by fishers and other species. However, snags created by fire fall sooner than those created in other ways, regardless of tree species (Morrison and Raphael 1993). Ritchie et al. (2013) found that 81% of snag biomass had fallen to the ground by 8 years post-fire in a mixed-conifer forest. Snags in large, severely burned areas are more exposed to wind and therefore likely to fall more quickly than those in other situations (Morrison and Raphael 1993).

In ecosystems with infrequent high-severity fire regimes, salvage logging has been shown to prevent development of complex early seral vegetation communities favored by many species (Lindenmayer et al. 2008; Swanson et al. 2010; Hanson 2013). Although effects of salvage logging in Sierran Mixed-Conifer forests have not been well studied, some studies suggest that salvage may help modify subsequent fire behavior to help maintain desired habitat conditions. Large amounts of dead wood remaining after a fire can contribute to future fire intensity and spread due to embers from burning snags (van Wagtenonk 2006) and torching of trees preheated by burning of heavy fuels on the forest floor (Ritchie et al. 2013). Decomposing snags and logs provide a receptive surface for ignition of spot fires from embers (Stephens 2004). In addition, a substantial portion of the fuels consumed by fire may be in the form of large logs, especially in areas that have experienced high tree mortality in the recent past (Ritchie et al. 2013). Large fuels increase burnout time, and prolonged heat exposure affects soil porosity and structure (McNabb and Swanson 1990; Brown et al. 2003).

As many tree species in the Sierra Nevada, such as most pines, evolved with generally low- to moderate-intensity fire, they have limited capacity to recover rapidly following extensive stand-replacing fire (Barton 2002; Goforth and Minnich 2008; Keeley 2012; Collins and Roller 2013). Tree planting and thinning may promote more rapid development of large trees and dense canopy cover that fishers favor, especially where natural regeneration may be slow, such as in large, severely burned patches devoid of natural re-seeding sources (Cal Fire 2010; Collins and Roller 2013). However, these actions may also result in unnaturally homogeneous forest conditions and inhibit shrub and herbaceous communities that contribute to habitat value for fishers and other species. Traditional reforestation strategies tended to produce dense, uniformly spaced stands that were intensively managed with herbicides to limit the growth of herbs and shrubs. Regular thinning of such densely stocked tree plantations is costly, and lack of funding has resulted in plantations, at least on national forest lands, that are overly dense and uniform, lack legacy structures, are unlikely to provide desirable habitat conditions for fishers, and are likely to increase fire hazard (Kobziar et al. 2009). More recent reforestation projects (e.g., Moonlight/Antelope and Chips Fire reforestation projects, Plumas National Forest) use a more variable tree planting pattern in an effort to increase spatial heterogeneity early in stand development and avoid the need for costly thinning projects.

Following stand-replacing fire, many sites in the southern Sierra Nevada can support a diversity of herbaceous, shrub, and hardwood species if they are not planted and managed for rapid recovery of conifers. These areas may provide suitable movement and foraging habitats for fishers some years following fire and, over time, may provide adequate resting and denning habitat. Thus, management decisions for enhancing fisher habitat should consider the spatial extent of severe, stand-replacing fire and the proportion of remaining late-seral habitat. Large post-fire restoration projects, including for the Rim and Aspen fires, may inform and refine science-based post-fire management guidelines.

6.1.4 Vegetation management

Vegetation management projects in the Sierra Nevada, especially those on federal lands, are usually designed to achieve a variety of objectives, including reducing hazardous fuels, improving growth of selected trees, changing the age structure of the stand, increasing spatial heterogeneity, and harvesting timber. Treatment practices vary in intensity, extent, and pace over time and across the landscape. Commercial timber harvest designed primarily for economic objectives, especially on private lands, is generally more intensive and may have fundamentally different impacts on forest composition and structure than multi-objective forest management projects.

In general, the effects of different types and intensities of vegetation treatments on fishers and fisher habitat are not sufficiently understood, although the SNAMP and KRFP fisher research teams are conducting before-after/control-impact studies and modeling studies (e.g., Thompson et al. 2011) to examine potential impacts and tolerance of fishers to management actions. Zielinski et al. (2013b) found that fishers on the Sierra National Forest appear to tolerate some level of treatments within their home ranges. Although thresholds in how much treatment fishers will tolerate are not well established, Zielinski et al. (2013b) found fishers using some areas that had experienced up to ~6.5% of their area treated annually, on average. They also found evidence of greater tolerance to treatments designed primarily for forest restoration goals than those with more emphasis on commercial timber harvest.

In their evaluation of threats to the West Coast fisher population, Naney et al. (2012) identified fisher habitat conditions that could be adversely affected by vegetation management (Table 14). The magnitude and duration of such changes depend on site productivity and the specific management actions applied.

Table 14. Outcomes of vegetation management identified as potential threats to fishers (Source: Naney et al. 2012).

Threat	Definition	Duration of Impact
Overstory reduction	Loss of dominant and co-dominant trees; decrease in canopy closure	Depends on degree and extent of change
Understory reduction	Loss of shrubs, saplings, suppressed trees, intermediate trees, and structural diversity	Shorter term, 1-2 decades
Reduction of structural elements	Reduction in large live and dead trees with mistletoe, broom rusts, heart rot, cavities, or pest and disease damage; reduction in large down wood	Longer term, many decades
Reduced vegetation diversity	Loss of floristic or tree species diversity	Depends on degree and extent of change
Increased fragmentation	Increased fragmentation of the pattern, distribution, and patchiness of the environment	Depends on degree and extent of change

Reduction in canopy cover, understory vegetation, and structural elements such as large snags may adversely affect fisher habitat quality (Table 14). Reduced canopy cover increases fisher exposure to weather extremes and predators (Powell and Zielinski 1994; Weir and Corbould 2008; Naney et al. 2012); reduction in canopy cover and structural elements increases travel distances between safe sites

and decreases presence of thermal refugia (Green et al. 2008; Naney et al. 2012). Reduction in understory and ladder fuels may increase fisher exposure to predation and decrease habitat quality for prey (Naney et al. 2012), while opening up forest stands and introducing more vegetation edges in and between stands may increase access by fisher predators, such as mountain lions, bobcats, and coyotes (Wengert 2013). Mechanical thinning reduces the density of small and medium-sized trees, a key component of fisher resting habitat (Zielinski et al. 2006). Truex and Zielinski (2013) found that predicted (modeled) fisher resting habitat is affected negatively by mechanical thinning followed by prescribed fire, although the duration of these effects is unknown. As habitat data used for this study were collected 1 year after treatment, the predicted negative effects were likely greatest at the time of evaluation and likely would diminish over time. Fall applications of prescribed fire at Sequoia–Kings Canyon National Park also significantly reduced predicted fisher resting habitat compared to spring applications, due primarily to greater reductions in canopy cover (Truex and Zielinski 2013).

Logging practices that remove the largest trees significantly reduce fisher resting and denning structures, likely the most limiting habitat elements for fishers (Weir and Corbould 2010; Weir et al. 2012; Aubry et al. 2013; Schwartz et al. 2013). Clear-cutting, in addition to removal of large trees, also removes canopy, the most important feature of fisher resting and denning habitat (Powell 1993; Buskirk and Powell 1994; Powell and Zielinski 1994; Purcell et al. 2009; Raley et al. 2012). Based on forest patterns in fisher home ranges, Sauder and Rachlow (2014) and Weir and Corbould (2010) predicted that a 5–10% increase in the amount of open area at the home range scale, e.g., due to timber harvest, reduces the probability of fisher occupancy by 39–60%, because fishers avoid establishing home ranges in areas with numerous openings (Weir and Corbould 2010).

Fishers appear to tolerate some degree of fuel-reduction or restoration vegetation treatments in their home ranges, despite their short-term and localized effects (Garner 2013; Zielinski et al. 2013b). Garner (2013) found that, although fishers avoid using areas treated for fuel reduction (including mechanical thinning and prescribed fire), their home ranges tend to include larger proportions of treated areas than in the landscape as a whole, and they do not shift home ranges in response to treatments. Garner (2013) concluded that treatments do not render the habitat unsuitable and may, in fact, increase fire resiliency, provided management focuses on surface and ladder fuels. In their modeling study, Truex and Zielinski (2013) found no significant effects of either solely mechanical or solely prescribed fire treatments on predicted resting habitat value, and no effects of any treatment type or combination on predicted foraging habitat.

Zielinski et al. (2013) sampled fisher home range-size areas for fisher scats, using scat detector dogs, and found that the areas with the most abundant scats had an average of 2.6% of their area disturbed per year (equivalent to 13% over a 5-year period) by a combination of vegetation management treatments. This exceeds the minimum proportion of treated landscape that may be necessary to reduce the size or severity of future fires (e.g., 8% treated per a 5-year period, Syphard et al. 2011; 2% annually, Finney et al. 2007). The degree of disturbance within sample units varies widely, and fishers may in some circumstances tolerate even higher rates of disturbance. In 1 of 5 high-use units and 1 of 3 moderate-use units, ~6.5% of the area was disturbed annually on average (Zielinski et al. 2013b). Zielinski et al. (2013b) found no statistically significant difference in the mean area of treatment per year across 3 fisher use categories (high, medium, and low), indicating that vegetation disturbance is only 1 of many factors affecting fisher habitat quality.

Collectively, these studies suggest restoration and maintenance of heterogeneous forest conditions may increase resilience to fire while enhancing habitat for fishers in the longer term. Currently, many forests are fairly homogeneous, either from the effects of fire suppression or from high-severity

wildfires resulting from suppression's increased fuel loads. In the near term, efforts to reduce fuels and increase fire resilience can reduce some of the forest structural features associated with fisher use, such as canopy cover and stem density. However these treatments increase unburned and post-burn habitat heterogeneity and therefore may have longer-term benefits to fishers. We currently lack information on both the temporal and spatial scales at which habitat heterogeneity is optimized for fishers in fire-prone forests.

Management actions may also impact fishers by disrupting behavior and movement patterns due to noise and activity associated with mechanical treatments. Anecdotal evidence in the Ashland Watershed in southern Oregon suggests that individual responses may be related to intensity of the disturbance; fishers appeared tolerant of hand-thinning crews, but immediately abandoned an area subjected to helicopter logging (C. Thompson and D. Clayton, USFS, unpublished data).

Timing of prescribed fire may be important for fishers. Fishers prefer denning in large, live trees, so loss of the den structure to fire is unlikely, especially as dens appear well-insulated to temperature increases associated with under-burns; however, smoke accumulation in the cavity immediately before and after parturition could negatively affect neurological development in kits (C. Thompson, personal communication). Low to moderate mixed-intensity fires during other seasons may not pose the same hazard, as adults are probably less susceptible to hazards from smoke. Due to heavy surface and ground fuel accumulations, first-entry fires after long-term fire exclusion may create more smoke than repeat burns. Fires in early spring or after precipitation in the fall may produce relatively more smoke than summer burns due to fuel moisture levels. Additional research should inform whether a spring-prescribed fire influences a female's ability to raise kits successfully.

6.2 Rodenticides and Other Poisons

Fishers throughout California are exposed to and sometimes die from rodenticides and other poisons, often associated with illegal marijuana grow sites on public lands. Toxicants cause mortality directly and indirectly through increased susceptibility to parasites, pathogens, and predation (Grue et al. 1997; Fournier-Chambrillon et al. 2004; Berny 2007; Relyea and Diecks 2008). Sub-lethal exposure to pesticides impairs anti-predator behavior, inhibits healing, and hinders thermoregulation (Cooke 1971; Farr 1977; Hunt et al. 1992). Evidence from laboratory and field studies in other species supports the premise that pesticide exposure can reduce immune system function (Li and Kawada 2006; Janeway et al. 2007; Zabrodskii et al. 2012), increase the prevalence of infectious disease (Riley et al. 2007; Vidal et al. 2009), and cause transient hypothermia (Gordon 1984; Grue et al. 1991), which may lower the effective lethal dose and increase mortality (Ahdaya et al. 1976, Martin and Solomon 1991). Multiple studies demonstrate that sub-lethal exposure to anticoagulant rodenticides or organophosphates may impair an animal's ability to recover from physical injury via clotting abnormalities and hemorrhaging (Townsend et al. 1984; Eason et al. 2001; Bailey et al. 2005; Webster 2009; Rattner et al. 2012). Predators with liver concentrations of rodenticides as low as 0.03 ppm (ug/g) have died as a result of excessive bleeding from minor wounds inflicted by prey (Erickson and Urban 2004). Accordingly, fishers exposed to rodenticide may be at risk of prolonged bleeding if they incur a wound inflicted by prey, conspecifics (e.g., bite wounds inflicted during mating), or predators. This finding has particular relevance, because predation is (ultimately) the leading cause of mortality for fishers in California (Gabriel 2013; Wengert 2013; Wengert et al. 2014). Sub-lethal pesticide exposure may exacerbate responses to other stressors (Jaques 1959; Newton et al. 1999; Erickson and Urban 2004), predisposing individuals to death from other causes (e.g., predation, collisions with automobiles, accidents, starvation, dehydration).

More than 300 illegal marijuana cultivation sites have been located within the SNAMP and KRFP areas since 2002, and Thompson et al. (2013) found that female fisher mortality rates in the KRFP study were directly correlated with the number of known marijuana cultivation sites within their home ranges. Ninety-one percent of necropsied fishers at the SNAMP site and 85% of those at KRFP showed exposure to anticoagulant rodenticides (M. Gabriel, Integral Ecology Research Center, UC Davis, unpublished data), and 3 mortalities have been attributed directly to rodenticide (Gabriel et al. 2012b; C. Thompson, unpublished data). Gabriel et al. (2012b) observed that fisher anticoagulant rodenticide mortalities occur mainly in April and May, when mothers are rearing dependent young, which will also die if they lose their mother. Increased exposure in spring also increases the potential for impacts on kits through reduced thermoregulatory ability and contamination in utero or through mother's milk (Gabriel et al. 2012b; Thompson et al. 2014). A dependent kit—exposed to rodenticide either placentally or through milk—died after its mother was killed (Gabriel et al. 2012b). Low birth weight, stillbirth, abortion, and bleeding, lack of appetite, and lethargy of neonates have been documented for other species exposed to rodenticides (Mackintosh et al. 1988; Greaves 1993; Munday and Thompson 2003).

Marijuana cultivation sites may be localized population sinks for small mammals and, as a result, pesticides used at these sites can reduce fisher prey populations. Rodents that ingest rodenticides may live for several days (Erickson and Urban 2004) and exhibit behavior that increases their predation risk (Cox and Smith 1992). Rodenticide exposure may contribute to the current absence of porcupines from most of the assessment area, despite the cessation of targeted porcupine persecution decades ago. However, no research has assessed rodenticide exposure in porcupine populations.

In addition to anticoagulant rodenticide effects, 3 mortalities in the northern California population have been attributed to different toxicants—carbamate, cholecalciferol, and bromethalin (M. Gabriel, unpublished data). While all known marijuana grow sites on Sierra National Forest were “reclaimed” (removal of all discovered plants, trash, equipment, and associated poisons) as of the end of 2011 (C. Thompson, personal communication), new and undetected sites pose continued threats.

6.3 Predation

Predation is the single greatest ultimate cause of mortality in fishers in California (Wengert et al. 2014). While predation may be a natural cause of death, it is a population-level threat because human actions may lead to increased predation rates and may alter where predation occurs. For example, high exposure to rodenticides and other poisons may elevate the apparent rate of predation and roadkill due to changes in fisher behavior and physiology. Habitat alteration may also contribute to predation rates, because forest edge habitats, open areas, and fragmented forest landscapes can increase access by bobcats, coyotes, and mountain lions. Wengert (2013) found that the risk of encounter between bobcats and fishers is greater in areas with more open habitat, such as meadows or pasturelands, and that predation risk increases with decreasing distance to open or brushy habitats. Most of the brushy habitats in the study were associated with recent clear-cuts or heavily-thinned stands, illustrating a potential link between human activities and fisher predation (Wengert 2013).

6.4 Disease and Infections

Several diseases have contributed to the mortality of southern Sierra Nevada fishers, including canine distemper virus (CDV), bacterial infections (some with interstitial pneumonia), concurrent infection with the protozoal parasite *Toxoplasma gondii* and urinary tract blockage, and emaciation due to presumed malnutrition (Gabriel 2013). Disease apparently is not a major threat to the southern Sierra

Nevada population, but an epizootic sweeping through 1 or more subpopulations could have significant negative consequences to the larger fisher population.

Canine distemper virus was associated with the death of 4 radio-collared fishers in the assessment area in 2009 (Keller et al. 2012). Three of these fishers died within a 2-week period from April 22 to May 5, 2009 and were found within 20 km (12.4 mi) of each other, while the 4th died during immobilization by researchers 4 months later, ~70 km (43.5 mi) away from the initial case locations. The source and transmission routes of the infections are unclear, but the temporal and spatial distribution of the CDV mortalities, and the similarity of the virus isolates, suggest 2 spillover events from sympatric carnivore species. In California, CDV mortalities in gray foxes and raccoons (*Procyon lotor*), both generalist carnivores in fisher habitat, are common (D. Clifford, CDFW, unpublished data). Although the fisher's solitary nature may lower disease transmission and thus large-scale risk of outbreak, CDV has been responsible for the near extirpation of other small carnivore populations, including black-footed ferrets (*Mustela nigripes*, Williams et al. 1988) and Santa Catalina Island foxes (*Urocyon littoralis catalinae*, Timm et al. 2009). Furthermore, highly virulent biotypes of CDV can be transmitted and cause mortality in multiple carnivore species (Origgi et al. 2012). Although CDV can cause mortalities in fishers, antibodies against the virus have been detected in a small number of apparently healthy live-captured fishers in California, indicating that some fishers can survive infection. From 2007 to 2009 in the southern Sierra Nevada, 14% (5 of 36) of sampled fishers on KRFP and 3% (1 of 36) in the SNAMP area had been exposed to CDV (Gabriel 2010). As a precaution against disease transmission during the Stirling project translocation, fishers exposed to CDV were not translocated, and those individuals selected for translocation were vaccinated against CDV using a recombinant vaccine proven safe in ferrets and other sensitive wildlife species (Wimsatt et al. 2003; Timm et al. 2009).

Deaths from rabies and canine parvovirus (CPV), both potentially significant pathogens for *Martes* and *Pekania* spp. (Gabriel et al. 2012c), have not been documented in California fishers, but clinical illness (lethargy and diarrhea) from confirmed CPV infection was observed in a fisher from northern California (D. Clifford and M. Gabriel, unpublished data). Fishers inhabiting the Hoopa Valley tribal reservation in northwestern California have been exposed to and infected with CPV: 13 of 31 (42%) fishers tested in 2004 and 2005 had CPV antibodies (Brown et al. 2006), while 13 of 66 (20%) fishers sampled between 2003 and 2007 were infected (Gabriel 2010). CPV exposure also has been documented in the southern Sierra Nevada fisher population, where the exposure prevalence was 47% (9 of 19) in sampled KRFP fishers and 4% (1 of 24) in sampled SNAMP fishers (Gabriel 2010). Although CPV has caused clinical disease, which could contribute to or cause mortality of individual fishers, testing to date suggests that the disease is circulating in California fishers without having population-level effects. Related parvoviruses that could cause greater disease impacts in fishers, including American mink (*Neovison vison*) enteritis virus and Aleutian disease virus, have not been detected in California fishers (M. Gabriel and D. Clifford, unpublished data). However, continued surveillance for parvoviruses is warranted, as Aleutian disease virus infection was diagnosed recently in free-ranging striped skunks in California (F. Giannitti, California Animal Health and Food Safety Laboratory System, UC Davis, personal communication).

Fishers in California are commonly exposed to *Toxoplasma gondii*, an obligate intracellular parasite that has caused mortality in captive black-footed ferrets (Burns et al. 2003), American minks (Pridham and Belcher 1958), and southern sea otters (*Enhydra lutris nereis*, Cole et al. 2000). Both the northern California and southern Sierra Nevada fisher populations have been exposed, with 66% (22 of 33) of tested fishers being exposed on SNAMP and 54% (21 of 39) being exposed on KRFP (Brown et al. 2008; Gabriel 2010). California fishers have been exposed to 2 vector-borne pathogens,

Anaplasma phagocytophilum and *Borrelia burgdorferi sensu lato*, the bacterium that causes Lyme disease (Brown et al. 2008), but mortalities from these diseases have not been reported. Fishers are likely susceptible to *Yersinia pestis*, the agent of plague, but exposure has not been assessed, and no cases have been documented as a source of mortality in California fishers (Gabriel et al. 2012c). Plague is a serious zoonotic risk (Williams et al. 1994) known to cause mortality in other mustelids and is endemic in many parts of California.

Fishers and *Martes* spp. harbor numerous ecto- and endoparasites. Although some parasites can serve as vectors for diseases, infections and infestations are usually associated with minimal morbidity and mortality (Gabriel et al. 2012c). Two new parasite infections have been documented in northern California fishers, but not in the southern Sierra Nevada. D. Clifford (unpublished data) found eyeworms, *Thelazia californiensis*, under the eyelids of multiple individuals from northern California. Although these worms may cause irritation and eye damage, there were no vision deficits or eye damage noted in affected fishers. *T. californiensis* most often infects livestock and is transmitted by flies, which mechanically transport eyeworm eggs among animals while feeding on eye secretions (Weinmann et al. 1974). In 2010, trematode flukes and eggs were recovered from 5 fishers from Humboldt County that had severe perianal swellings and subcutaneous abscesses (D. Clifford, unpublished data). Retrospective analysis of field observations reveals that perianal swelling and abscesses are noted occasionally on fishers immobilized as part of the Hoopa fisher project (J. Higley, Hoopa Tribal Forestry, unpublished data). No mortalities have been attributed to this novel trematode infection to date (L. Woods, California Animal Health and Food Safety Laboratory, UC Davis, unpublished data), but it is not known if fishers with severe disease suffer morbidity or reduced long-term survival. D. Clifford (unpublished data) is investigating trematode species identity, life history, and geographic distribution. Gabriel et al. (2012c) include a comprehensive list of pathogens and parasites associated with fishers and *Martes* spp.

6.5 Roads and Other Human Structures

Vehicle collisions are a significant mortality factor for fishers in some portions of the assessment area, especially where moderate- to heavily-traveled roads traverse high-quality habitat, such as along Highway 41/Wawona Road on the Sierra National Forest/Yosemite National Park. The SNAMP study recorded 9 vehicle collision mortalities between December 2007 and January 2013 (R. Sweitzer, unpublished data), and KRFP and Sequoia–Kings Canyon National Parks each have recorded 1 mortality (D. Gammons, Sequoia and Kings Canyon National Parks, personal communication). Ten road-killed fishers have been found in Yosemite National Park over the past 2 decades—9 on Wawona Road and 1 nearby on the Glacier Point Road (Table 5, Figure 5). Of the 10 road-killed fishers, 8 were killed during the denning season (Mar.-Jun.); of these, 4 were females, 2 were males, and 2 were of unknown sex. Of the 4 females killed during denning season, 2 were lactating. Highway 41/Wawona Road passes through high-quality fisher denning habitat (Figure 5); females provisioning young must forage daily near their dens, probably increasing the frequency of crossing a nearby road during denning season. Males also may be at elevated risk during the denning season, when they roam widely in search of females.

Fishers can minimize risk of roadkill by using culverts to cross under roads. In September 2012, remote cameras detected fishers using culverts to cross Highway 41 near Fish Camp in the Sierra National Forest, just south of Yosemite National Park. During fall 2012, cameras in the Sierra National Forest detected at least 4 individual fishers using 4 separate culverts to cross Highway 41 and adjacent USFS roads (A. Otto, USFS, personal communication). As of November 2013, 5 fishers were detected (L. Cline, NPS, personal communication). Remote cameras have detected fishers at 3

drainages along Wawona Road in Yosemite National Park since monitoring began in fall 2011; in September 2013, cameras documented a fisher using a culvert to cross underneath that road (L. Cline, personal communication). During fall (and possibly other seasons), culverts have little or no water flow, which facilitates passage. However, spring snowmelt may force fishers to cross the paved road surface rather than below the road, increasing the probability of vehicle collisions. Eight of the 10 known road-killed fishers in Yosemite National Park were found during the spring denning season (Mar. 1-Jun. 30), when spring snowmelt often fills culverts with water. The radio-collared male killed by a vehicle along Wawona Road in April 2012 had a large home range that encompassed much of Wawona Road; he was detected at several camera stations in the park from 2009 to 2011. The loss of breeding animals during the denning season may significantly affect population viability.

In addition to the risk of vehicular collisions on paved roads with higher vehicle speeds, forest roads and trails may elevate fisher predation by mountain lions, bobcats, and coyotes using these trails as travel and hunting corridors (Naney et al. 2012). Predation sites are closer to roads, on average, than sites where live fishers were detected (Wengert 2013). Additionally, bobcat and fisher interactions are more likely to occur near roads, hard edges (i.e., abrupt changes between closed and open habitat), open areas, and bushy areas (Wengert 2013).

Fishers have also died after becoming trapped in human structures such as water tanks and pipes. Folliard (1994) recommended that abandoned water tanks be covered, drained, or modified by inserting branches, poles, or bars to allow fishers or other wildlife to escape.

6.6 Climate Change Effects

Changes in climate may have both direct (e.g., thermal stress) and indirect (e.g., changes in species interactions and vegetation) effects on wildlife distribution and abundance (Martin 2007; Rubidge et al. 2011). Direct effects of climate warming may force species to higher altitudes or latitudes, while indirect effects may be far more complex.

6.6.1 Physiological

Warmer and drier climates may directly impact fishers due to physiological tolerance limits. At the landscape scale, fisher distribution in the Sierra Nevada is highly correlated with annual and seasonal climate variables, indicating that fishers select habitats having moderate or intermediate temperature and precipitation ranges, and lower temperature variability, compared to what is available in the region (Section 4.1, Spencer and Rustigian-Romsos 2012a, b; Spencer et al., unpublished analyses performed for the Yale Framework Climate Adaptation Project: <http://yale.databasin.org/pages/cbi>). Although it is unclear to what degree these correlations reflect fisher physiological tolerances versus climate effects on vegetation, Spencer et al. (unpublished analyses: <http://yale.databasin.org/pages/cbi>) found that habitat models combining both vegetation and climate variables significantly outperformed models based on vegetation or climate variables alone. Furthermore, within their home ranges, fishers tend to select locations with cooler, more mesic, and less variable microclimates, especially for resting and denning (Section 4.3). They select resting sites that have lower heat load indices relative to available sites (Aubry et al. 2013), and in the southern Sierra Nevada, resting sites tend to be on steep slopes, in canyons rather than on ridge tops, and close to water (Zielinski et al. 2004b; Purcell et al. 2009; Underwood et al. 2010). Dens are in areas with relatively low summer temperatures (Section 4.3.3). These factors, along with fisher preferences for areas with dense and often multi-storied tree canopies, suggest that fishers select sites for reduced heat loads and temperature variability, and may have a physiological intolerance for warmer temperatures. In colder seasons, fishers tend to use cavities in trees and snags more than platforms for resting (K. Purcell, unpublished data), probably to

stay warm. Taken together, these data suggest that fishers have a relatively narrow temperature tolerance, and increases in temperature and temperature variability may negatively impact them physiologically or cause them to become even more selective for micro-climatic refugia, such as shaded canyons and slopes and dense, multistoried tree canopies. Fishers may also shift upslope to cooler elevations, especially if snow packs decline with climate change.

6.6.2 Species interactions

Climate change will affect species idiosyncratically (Moritz et al. 2008) and has the potential to alter species interactions and create novel communities. Lawler et al. (2009a, b) projected moderate vulnerability of California's mammalian fauna under a high greenhouse gas emissions scenario, leading to a 10-40% change in the mammalian fauna by the end of the century. Shifting species' distributions will create novel species assemblages, leading to new competition or predation pressures for some species (Stralberg et al. 2009). A recent resurvey of small mammal communities in Yosemite National Park showed that, over the last century, some species range limits have contracted or shifted upwards, some have expanded, and some have not changed (Moritz et al. 2008). Of potential fisher prey species, 3 showed range contraction ranging from 159 to 1,007 m (522-3,304 ft) on the lower end and 0 to 334 m (1,096 ft) on the upper end of their ranges (voles, western jumping mouse, and Allen's chipmunk [*Tamias senex*]). Four fisher prey species showed no significant change in range over the last century (big-eared woodrat [*Neotoma macrotis*], brush mouse [*Peromyscus boylii*], montane vole [*Microtus montanus*], and Douglas squirrel). Those species exhibiting range contraction or upward shifting are likely limited by thermal tolerance and contracting suitable vegetation distributions (e.g. *T. alpinus* and *T. senex*), while those with stable or expanding distributions (e.g., *T. speciosus*) may have been released from interspecific competition by retreating species (Rubidge et al. 2011).

Little is known about how fisher competitors and predators are being influenced by climate change. Lawler et al. (2012) suggested that martens, like fishers, will be highly sensitive to climate change. A reduction in snowpack in the southern Sierra Nevada over the next century may lead to increased encounters between fishers and martens (Purcell et al. 2012), but fishers are unlikely to be adversely affected by this change.

6.6.3 Habitat

Climate change is affecting wildlife habitats in the Sierra Nevada, and fishers may be particularly at risk due to changes in temperature, moisture availability, and forest fires, with implications for vegetation composition and structure. Projections of future climate and vegetation conditions, using the MC1 vegetation change model (Bachelet et al. 2001, Lenihan et al. 2008), suggest a major decrease in suitable fisher habitat over the next 50 years (Spencer et al., unpublished analyses performed for the Yale Framework Climate Adaptation Project: <http://yale.databasin.org/pages/cbi>), although these models may not adequately account for topographic effects on local microclimate and vegetation, which may partially mitigate the changes in mountainous terrain. Climate and CO₂-driven changes in fire regimes are projected to increase wildfire intensity, area, and frequency (Price and Rind 1994; Miller and Urban 1999; Flannigan et al. 2000, 2013; Fried et al. 2004; McKenzie et al. 2004; Lenihan et al. 2008; Spracklen et al. 2009; Westerling et al. 2011; Purcell et al. 2012; Yue et al. 2013). Climate-driven increases in fire size and severity are already apparent in the western US and are strongly linked to increasing temperatures and earlier spring snowmelt (Westerling et al. 2006). Modeling studies project increased fire activity to persist and possibly accelerate under most future climate scenarios, due to the increased growth of fuels under higher CO₂ (and in some cases greater precipitation), decreased fuel moisture from higher dry season temperatures, and possible increased

thunder cell activity causing lightning strikes (Price and Rind 1994; Miller and Urban 1999; Lenihan et al. 2003, 2008; Westerling and Bryant 2006; Westerling et al. 2011; Yue et al. 2013). Mid-elevation sites on the west side of the Sierra Nevada are likely to show the greatest increases in burned area in the next few decades (Westerling et al. 2011). Increased fire frequencies, sizes, and intensities are likely to drive changes in tree species compositions (Lenihan et al. 2003, 2008) and reduce the extent of late-successional forests (USFS and BLM 1994; McKenzie et al. 2004), which could alter the extent, abundance or occurrence of species associated with these habitats (McKenzie et al. 2004; Purcell et al. 2012).

In addition to fire-driven vegetation changes, changes in moisture regimes affect important fisher habitat components. Lenihan et al. (2003, 2008) predict that, under wetter future scenarios, broadleaf trees (especially oak species) will replace conifer-dominated forests in many parts of the low- and mid-elevation Sierra Nevada in the next century. Under drier future scenarios, Lenihan et al. (2003, 2008) predict that shrublands or grasslands will expand into conifer types, due to drought and increases in fire frequency and severity, thus further reducing fisher habitat. Van Mantgem et al. (2009) document widespread increases in tree mortality in old-growth forests across the western US—with the highest mortality rates in the Sierra Nevada and in mid-elevation forests (1,006-2,042 m [3,300-6,700 ft])—probably due to increased drought stress. Lutz et al. (2009) conclude that, between the mid-1930s and mid-1990s, both the density and diversity of large-diameter trees in Yosemite National Park declined, due to water stress.

Lawler et al. (2012) investigated the possible effects of climate change on selected species of the genus *Martes* (including fishers, now *Pekania*) and found that macroclimate conditions closely correlated with Pacific fisher presence in California are likely to change greatly over the next century, which, along with changing fire regimes, will decrease the amount of suitable habitat. Their results suggest that martens and fishers will be highly sensitive to climate change and will probably experience the largest climate impacts at the southernmost extent of their ranges (i.e., in the southern Sierra Nevada). To protect fisher habitat over the long term, Lawler et al. (2012) recommend using targeted forest fuel treatments and fire management policies that allow more naturally ignited fires to burn during moderate weather conditions.

Genetic evidence suggests that fishers have survived climate-driven range contraction in the past, and that the southern Sierra Nevada may have acted as a climate refugium during this time (Tucker et al. 2014). Loarie et al. (2008) identify the southern Sierra Nevada as a potential climate refugium—an area projected to sustain species with otherwise shrinking ranges. Favorable impacts of climate change on fishers may include decreased snow levels and an increase in hardwood species. Thus, while fisher habitat and conditions are likely to be negatively impacted by long-term climate change, there may be some resilience in the population.

7 Conclusions

Data from diverse studies over multiple scales suggest that the southern Sierra Nevada fisher population is a relatively small but stable population in recent years and that it expanded northward to the Merced River from somewhere south of the Kings River beginning several decades ago. Fisher demographic rates in the SNAMP and KRFP study areas are generally consistent with those measured in other regions. Nevertheless, these encouraging data must be tempered by considering the various risks faced by the population, including stochastic risks like severe wildfires or disease outbreaks, as well as controllable risks, such as vegetation management actions. In particular, the persistent threat of exposure to poisons associated with illegal marijuana cultivation increases the direct risk to fisher survival, as well as the risk posed by other factors. Management should attempt to balance complex and competing risks, especially the potential for large severe wildfires to reduce and fragment fisher habitat versus the more localized and short-term risks of vegetation management intended to mitigate these fire risks (Scheller et al. 2011).

Based on the information summarized in this Fisher Conservation Assessment, the Fisher Conservation Strategy should include and expand on the following recommendations:

- Plan and implement vegetation treatments (e.g., based on concepts in General Technical Report 220/237 [North et al. 2009; North 2012] and other best available science) that favor a return to more sustainable wildfire conditions and resilient forest conditions on the landscape while protecting essential fisher habitat conditions and elements. Strive for naturally heterogeneous conditions (e.g., denser forest in canyons, basins, and on mesic slopes, more open forest on ridgetops and south slopes) that favor patchy, mosaic fires (mostly low to moderate severity with some unburned and high severity patches) on the landscape, and reduce the risks of very large and severe fires that can fragment fisher core and linkage areas. Maximize recruitment and retention of potential fisher resting and denning structures and microhabitats, and reduce conifer canopy immediately around some black oaks to encourage growth.
- Investigate ways to increase population connectivity between core habitat areas, and expand the population into currently unoccupied areas. Identify and improve movement potential through corridor pinch points, and manage to retain or increase tree canopy cover where open conditions may impede fisher dispersal. Remove or mitigate potential barriers or filters to movement in critical locations (e.g., improved road-crossing structures).
- Prioritize management actions that counter specific threats in locations where they are most likely to increase population resilience, such as improving road-crossing structures along Highway 41/Wawona Road and other areas where vehicle collision mortalities are a problem; fortifying law enforcement to locate and remove trespass marijuana grow sites and clean up associated pesticides and trash; and closing and remediating unneeded roads or trails that may facilitate increased use by bobcats, mountain lions, and coyotes in fisher habitat, especially in or near resting and denning habitat.
- Investigate ways to enhance habitat for larger fisher prey species (e.g., squirrels), and determine if recovering the porcupine population in the assessment area is feasible and desirable. Porcupines are large and important prey for fishers in other regions, and porcupine recovery in the assessment area has the potential to shrink fisher home ranges and increase

fisher population density if they comprise an important part of the diet in the assessment area as well. In addition, porcupines serve as “ecosystem engineers” that create deformities in trees used as fisher denning and resting structures, such as cavities and platforms.

- Consider direct population intervention as a contingency option to expand the current population or reduce extinction risks, such as assisted migration across the Merced River into Cores 6 and 7. Research and discuss whether contingency plans for raising and releasing orphaned fisher kits may be a useful population intervention. Develop contingency plans for translocation from occupied to unoccupied core habitat areas if natural colonization does not occur by a given period (e.g., within 10-15 years), or if dramatic changes in habitat conditions or threats within occupied cores (e.g., massive tree death or a fisher epizootic in a particular core area) suggest that reintroduction to other areas will decrease extinction risks.

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Appendix A. Methods

This appendix briefly summarizes the Conservation Biology Institute's (CBI) methods for unpublished analyses included in the Conservation Assessment.

A-1 Study Area Delineation

The study area (or assessment area) boundary was delineated based on watersheds to include all occupied fisher habitat in the southern Sierra Nevada plus a substantial amount of potential but unoccupied habitat. The boundary comprises 17 watershed sub-basins⁹ clipped at the 152 m (500-ft) elevation contour of the Central Valley. The southernmost sub-basin (Middle Kern–Upper Tehachapi–Grapevine) extends well outside the fisher's geographic range, so was cropped at the boundary used by Spencer et al. (2011). The northern boundary was adjusted northward from a watershed boundary to the Mokelumne River and North Fork Mokelumne River to include all of Stanislaus National Forest.

A-2 Landscape-scale Habitat Model

CBI modeled predicted probability of fisher occurrence (PPO, often interpreted to represent potential habitat quality) using Maxent (version 3.3.3k, Phillips et al. 2006), 72 fisher detections, and an array of 22 environmental data layers (Table A-1). Potential environmental predictors included vegetation, climate, elevation, terrain, and Landsat-derived reflectance variables at 30-m and 1-km resolutions. Environmental variables were averaged over a 10-km² moving window, which roughly approximates female fisher home range size, and then resampled to 90 m. Urban and open water land covers were masked out. CBI tested predictor correlation for each model calibration region using ENMTools 1.3 (Warren et al. 2008).

Verifiable fisher locality data (e.g., captures, camera detections) spanning 1991–2011 were provided by multiple sources, including universities, non-profit organizations, and agencies, in addition to the USFS regional fisher monitoring data. To avoid non-independence of fisher detection data, we randomly removed (“filtered”) localities, using a 5-km minimum nearest-neighbor distance and retaining the most reliable and recent detections. Filtering reduced sample size from 514 total detections to 72. Because fishers were extirpated historically from regions north of the Merced River, and all detections were south of the Merced River, we trained models south of the Merced River to avoid biasing against potential but unoccupied habitat, and then projected the results over the entire study area.

We ran models using 10-fold cross-validation and default Maxent parameters, initially using all 22 environmental predictors. We eliminated correlated variables ($|r| \geq 0.7$) by retaining the one that yielded the maximum decrease in training gain when excluded from the model. We then systematically removed variables that provided the minimum decrease in training gain when excluded, using a stepwise procedure, until obtaining a model with the fewest predictors having an average training gain not significantly different than the full model (Table A-2). Significance was

⁹ The 17 sub-basins (HUC-8; Calwater 2.2.1, California Interagency Watershed Map of 1999, California Interagency Watershed Mapping Committee) are Mill, South Fork Kern, Upper Calaveras, Upper Chowchilla–Upper Fresno, Upper Deer–Upper White, Upper Dry, Upper Kaweah, Upper Kern, Upper King, Upper Merced, Upper Mokelumne (southern portion only), Upper Poso, Upper San Joaquin, Upper Stanislaus, Upper Tule, Upper Tuolumne, and Middle Kern–Upper Tehachapi–Grapevine.

Table A-1. Potential environmental predictors used in landscape modeling averaged within a 10-km² (4-mi²) moving window.

Variable	Description
Vegetation cover (source: USGS LANDFIRE US_110EVT Refresh 2008)	
Dense forest	Proportion classed as forest with $\geq 60\%$ canopy cover
Vegetation size (source: NACP Aboveground Biomass and Carbon Baseline Data (NBCD 2000))	
Canopy height	Basal area-weighted canopy height (weights contribution of trees to stand height by their basal area); m*10
Biomass	Aboveground live dry biomass; kg/m ² *10
Forest age	NACP Forest Age Maps at 1-km Resolution for the US
Forest stand age	NACP Forest Age Maps at 1-km Resolution for the US
Vegetation type (source: USGS LANDFIRE US_110EVT Refresh 2008)	
Conifer forest	Proportion classed as conifer forest
Hardwood	Proportion classed as hardwood forest
Mixed forest	Proportion classed as mixed conifer-hardwood forest
Tasseled cap indices (source: Landsat Ecosystem Disturbance Adaptive Processing System Landsat-7 ETM+ data products)	
Tasseled cap greenness	Transformation to condense Landsat spectral data into component associated with vegetation characteristics
Tasseled cap wetness	Transformation to condense Landsat spectral data into component associated with vegetation characteristics
Topographic (source: USGS National Elevation Dataset)	
Latitude-adjusted elevation	Mean latitude-adjusted elevation (0.625 m added to elevation for every km north from southernmost point in study area)
Insolation index	Mean solar insolation index
Local relief	Standard deviation of elevation in 5 pixel x 5 pixel moving window
Ruggedness	Vector ruggedness measure, calculated in 5 pixel x 5 pixel moving window
Slope	Mean percent slope
Climate (source: Hijmans et al. 2005)	
Max temp warmest month	Average maximum temperature warmest month ($^{\circ}\text{C} \times 10$, 1960-1990)
Min temp coldest month	Average minimum temperature coldest month ($^{\circ}\text{C} \times 10$, 1960-1990)
Mean temp coldest quarter	Average minimum temperature coldest quarter ($^{\circ}\text{C} \times 10$, 1960-1990)
Mean diurnal range	Mean of monthly max temp - min temp ($^{\circ}\text{C} \times 10$, 1960-1990)
Isothermality	Mean diurnal range / temperature annual range * 100 ($^{\circ}\text{C} \times 10$, 1960-1990)
Temperature seasonality	Standard deviation * 100 (1960-1990)
Precipitation coldest quarter	Average precipitation during coldest quarter (mm, 1960-1990)
Precipitation	Average annual precipitation (mm, 1960-1990)

Table A-2. Landscape model selection; R1 is the full model, and R7 is the selected model.

Model	Predictors	AUC ¹	RTG ²	95% CI ³
R1	Canopy height, min temp coldest month, tassal-cap greenness, dense forest, hardwood forest, precip, insolation index, mixed forest, isothermality, stand age	0.891	1.359	1.340-1.378
R2	Canopy height, min temp coldest month, tassal-cap greenness, dense forest, hardwood forest, precip, insolation index, mixed forest, isothermality	0.893	1.357	1.338-1.376
R3	Canopy height, min temp coldest month, tassal-cap greenness, dense forest, hardwood forest, precip, insolation index, mixed forest	0.896	1.353	1.334-1.373
R4	Canopy height, min temp coldest month, tassal-cap greenness, dense forest, hardwood forest, precip, insolation index	0.898	1.350	1.330-1.370
R5	Canopy height, min temp coldest month, tassal-cap greenness, dense forest, hardwood forest, precip	0.900	1.347	1.326-1.366
R6	Canopy height, min temp coldest month, tassal-cap greenness, dense forest, hardwood forest	0.900	1.337	1.316-1.358
R7	Canopy height, min temp coldest month, tassal-cap greenness, dense forest	0.902	1.317	1.293-1.342
R8	Canopy height, min temp coldest month, tassal-cap greenness	0.903	1.287	1.264-1.309
R9	Canopy height, min temp coldest month	0.896	1.226	1.199-1.252
R10	Canopy height	0.867	1.009	0.984-1.034

¹Mean 10-fold cross-validated²Mean 10-fold cross-validated regularized training gain³95% confidence interval of regularized training gain

defined as lack of overlap between 95% confidence intervals for training gain averages (calculated in R version 2.15.3, R Core Team 2013). We produced 2 versions of the final—a continuous logistic output and a binary output. The binary output was created by reclassifying the continuous logistic output using the maximum training sensitivity plus specificity threshold (Liu et al. 2013), which can be interpreted as separating habitat from non-habitat.

We used both fisher detections and non-detections to evaluate models. To create an evaluation dataset from the compiled fisher detections dataset, we removed fisher detections used in model calibration and all detections within 1 km (3,280 ft) of those; we then filtered non-detections and the remaining detections using the same rules as for the calibration data. A total of 51 detections and 16 non-detections remained after filtering for model evaluation. We also generated a random sample of 1,000 background points for model evaluation. We used both threshold-dependent measures and independent measures to evaluate the final model. For threshold-dependent measures, the maximum training sensitivity plus sensitivity (Liu et al. 2013) threshold was used for binary data classification. The R statistical package (version 2.15.3, R Core Team 2013) and the R package PresenceAbsence (version 1.1.9, Freeman and Moisen 2008) and POC code (Phillips and Elith 2010) were used to calculate several model evaluation measures appropriate for presence-background data (Table A-3).

Table A-3. Model evaluation metrics.

Name	Description
Threshold-dependent	
Sensitivity	Proportion of actual presences correctly predicted
Omission error (false negative)	Predicting absence where species is detected
Positive predictive power	Probability that a site predicted as present is actually present
AUC	Ranges from 0.5 to 1.0; indicates the proportion of cases in which a random selection from the positive group will score higher than a random selection from the negative group
Threshold-independent	
Boyce index (BI) (Boyce et al. 2002)	Divide habitat suitability into bins; calculate area in each bin, number of detections in each bin, and predicted-to-expected ratio. Plot of P/E against habitat suitability class should show an increase in P/E as suitability increases. Spearman rank correlation coefficient between ratio and class varies from -1 to 1. Positive values indicate predictions are consistent with presences in the test dataset, values close to 0 indicate model not different from chance, negative values indicate incorrect model which predicts poor quality areas where presences are more frequent.
POC plots	Presence-only calibration plots (Phillips and Elith 2010). Avoid the sensitivity of BI to binning. Evaluate model calibration: whether predictions are proportional to conditional probability of presence. Presence-only analogue of traditional (presence-absence) calibration plots

A-3 Core Delineation

We delineated fisher habitat “core areas” for connectivity analyses and for planning and management purposes (e.g., to establish area-specific conservation and management goals and standards) as follows:

1. Used the CBI Maxent model of fisher predicted probability of occurrence described in Section A-2 as the base.
2. Used the “strength of selection” analysis (Hirzel et al. 2006) performed by D. LaPlante, Natural Resource Geospatial) for the USFWS west coast fisher evaluation to delineate habitat that is “strongly selected for” by fishers (probability threshold = 0.41) (CBI and D. LaPlante, unpublished analyses).
3. Removed patches with area $<7.75 \text{ km}^2$ (1,915 ac; one female core use area, based on analyses of SNAMP and KRFP telemetry data by R. Sweitzer and C. Thompson).
4. Aggregated polygons within 1.57 km (1 mi; radius of a 7.75-km^2 [1,915 ac] female home range core use area) of one another.
5. Delineated habitat patches $\geq 38.75 \text{ km}^2$ (9,575 ac; ~5 female core use areas) as potential core areas.

6. Removed areas with elevation $\leq 1,250$ m along the Merced River to split a habitat polygon that spanned the Merced River (to recognize that there is currently no breeding population in or north of Yosemite Valley).
7. Divided a core at Bear Creek through Mountain Home State Forest to show the genetic division found by Tucker et al. 2014.

A-4 Resting Habitat

We modeled predicted probability of fisher resting site occurrence using Maxent and similar model selection and evaluation procedures described for the landscape-scale habitat model (Section A-2). Default Maxent parameters with 10-fold cross-validation were used with the following exception: linear, quadratic, and product feature types were used rather than the default auto-features, thus creating more parsimonious models with smoother response curves.

The KRFP fisher study team (C. Thompson, unpublished data) provided resting site locality data (N=903, spanning 2007-2013) from Sierra National Forest, and the Southern Sierra Nevada Marten and Fisher study (Zielinski et al. 2004) provided resting site locality data (N=345, spanning 1994-1996) from Sequoia National Forest. We defined the model extent using a minimum convex polygon encompassing 25-km (15.5 mi) buffer envelopes around resting site localities in these two study areas. To avoid non-independence of resting site locality data, we randomly removed (“filtered”) localities so that no more than 1 resting site fell within a 2-km² (500 ac) moving window used to smooth environmental layers. The 2-km² (500 ac) resolution was used based on earlier elasticity analyses performed to determine the resolution that provided the best fit (AUC value) to den locality data (Spencer and Rustigian-Romsos 2012). Filtering reduced total sample size from 1,248 total resting site localities to 237.

An array of 17 potential environmental predictor layers was created, including vegetation, topography, climate, and hydrology variables at 30-m (100 ft) resolution (Table A-4). Vegetation data were derived from merged and rasterized USFS R5 existing vegetation tiles (attributes TOTAL_TREE_CFA, OS_TREE_DIAMETER_CLASS_1, and WHRTYPE). We tested multiple measures of tree cover (using 10% cover class bins available in the vegetation tiles) for inclusion in the model (Table A-4). We obtained elevation data for topographic variables from a 30-m (100 ft) digital elevation model (National Elevation Dataset, USGS) and extracted hydrology data from the National Hydrography Dataset (USGS). Climate data were derived by resampling 270-m (885 ft) downscaled historic climate data (Flint and Flint 2012, California Basin Characterization Model, <http://climate.calcommons.org/dataset/10>). All environmental variables were averaged over a 2-km² (500 ac) moving window except for distance to nearest perennial and intermittent water features.

Potential full models were created by individually substituting in multiple measures of tree cover, tree size, and climate with forest cover type, topographic, and hydrographic predictors. In a few cases, we dropped models whose variables produced idiosyncratic response curves indicative of model overfitting or that were biologically uninterpretable, potentially due to inaccurate or inconsistent data coverage. We evaluated the remaining candidate full models using Akaike’s Information Criterion corrected for small sample size (AICc) calculated with ENMTools 1.3 (Warren et al. 2008). The model with the highest AICc weight was selected as the best full model. Variables that provided the minimum decrease in training gain when excluded were then systematically removed from this selected full model using the stepwise procedure described in Section A-2. The resulting model with the highest AICc weight was selected (Tables A-5 and A-6). We repeated this process 3 times to create and contrast alternative types of models: using both biotic and abiotic variables, biotic variables

only, and biotic variables excluding hardwood as a potential predictor (Tables A-7 and A-8). These different alternative models may be useful for different purposes and can help evaluate uncertainties in model projections beyond the model training extent. For example, while a model that includes biotic, abiotic, and hardwood variables may provide the best overall fit to fisher data, biotic-only models may be more useful for management purposes (because abiotic variables like elevation cannot be managed); and including/excluding hardwoods allowed us to contrast how models map habitat value in regions lacking significant hardwood communities, such as the Kern Plateau.

Table A-4. Potential environmental predictors used in resting habitat modeling averaged within a 2-km² (500 ac) moving window (except distance from water).

Variable	Description
Vegetation cover (source: USFS R5 existing vegetation tiles)	
Total tree cover $\geq 70\%$	Proportion with total tree cover from above $\geq 70\%$
Total tree cover $\geq 60\%$	Proportion with total tree cover from above $\geq 60\%$
Total tree cover $\geq 50\%$	Proportion with total tree cover from above $\geq 50\%$
Total tree cover $\geq 40\%$	Proportion with total tree cover from above $\geq 40\%$
Total tree cover $< 10\%$	Proportion with total tree cover from above $\leq 10\%$
Total tree cover $< 20\%$	Proportion with total tree cover from above $\leq 20\%$
Vegetation size (source: USFS R5 existing vegetation tiles)	
Overstory dbh ≥ 25 cm (10 in)	Proportion with mean diameter at breast height > 25 cm (10 in) for trees forming the uppermost canopy layer
Overstory dbh ≥ 50 cm (20 in)	Proportion with mean diameter at breast height > 50 cm (20 in) for trees forming the uppermost canopy layer
Vegetation type (source: USFS R5 existing vegetation tiles)	
Forest type	Proportion with CWHR type as aspen, eastside pine, Douglas fir, montane hardwood conifer, montane hardwood, Sierran mixed conifer, ponderosa pine, Jeffrey pine, or white fir
Hardwood	Proportion with hardwood cover from above > 0 and CWHR type as montane hardwood conifer or montane hardwood OR regional dominance type alliance as riparian mixed hardwood, interior hardwood, canyon live oak, black oak, interior live oak, black cottonwood, or montane mixed hardwood
Topographic (source: USGS National Elevation Dataset)	
Percent slope	Mean slope (%)
Insolation index	Mean insolation index
Climate (source: Flint and Flint 2012, California Basin Characterization Model)	
Precipitation	Mean annual precipitation (mm; 1971-2000)
Snowfall	Mean annual snowfall (mm; 1971-2000)
August max temperature	Average August maximum temperature (C°; 1971-2000)
January min temperature	Average January minimum temperature (C°; 1971-2000)
Hydrology (source: USGS National Hydrographic Dataset)	
Distance from water	Distance (m) to nearest perennial or intermittent stream or lake (not averaged within moving window)

Table A-5. Resting habitat model selection using both abiotic and biotic variables; R1 is the full and selected model.

Model	Predictors	AUC ¹	Parameters	RTG ²	95% CI ³	AICc Weight
R1	January_min_temperature, overstory dbh <u>>25 cm</u> (10 in), hardwood, percent slope, total tree cover $\geq 60\%$, forest type	0.934	19	1.681	1.670-1.693	1.000
R2	January_min_temperature, overstory dbh <u>>25 cm</u> (10 in), hardwood, percent slope, total tree cover $\geq 60\%$	0.931	15	1.652	1.640-1.664	0.000
R3	January_min_temperature, overstory dbh <u>>25 cm</u> (10 in), hardwood, percent slope	0.928	12	1.612	1.600-1.625	0.000
R4	January_min_temperature, overstory dbh <u>>25 cm</u> (10 in), hardwood	0.925	8	1.579	1.565-1.593	0.000
R5	January_min_temperature, overstory dbh <u>>25 cm</u> (10 in)	0.915	4	1.444	1.430-1.459	0.000
R6	January_min_temperature	0.818	2	0.741	0.733-0.749	0.000

¹Mean 10-fold cross-validated²Mean 10-fold cross-validated regularized training gain³95% confidence interval of regularized training gain**Table A-6. Resting habitat model (abiotic and biotic) variable percent contribution (%) and permutation importance (PI); R1 is the selected model.**

Variable	R1		R2		R3		R4		R5		R6	
	%	PI	%	PI	%	PI	%	PI	%	PI	%	PI
Jan_min_temp	9.7	20.2	14.5	30.1	20.4	23.8	20.4	23.3	35.6	33.6	100	100
Overstory dbh ≥ 10 in	22.4	38.0	46.6	58.2	63.3	64.5	64.5	69.4	64.4	66.4		
Hardwood	5.3	4.1	9.7	5.0	15.5	9.8	15.0	7.3				
Percent slope	1.1	2.2	1.2	2.5	0.8	1.9						
Total tree cover $\geq 60\%$	10.1	4.4	28.1	4.2								
Forest type	51.5	31.0										

Table A-7. Resting habitat model selection (biotic only); R1 is the full and selected model.

Model	Predictors	AUC ¹	Parameters	RTG ²	95% CI ³	AICc Weight
R1	Overstory dbh ≥ 25 cm (10 in), hardwood, forest type, total tree cover $\geq 70\%$	0.916	10	1.478	1.465-1.491	0.991
R2	Overstory dbh ≥ 25 cm (10 in), hardwood, forest type	0.913	8	1.459	1.446-1.472	0.009
R3	Overstory dbh ≥ 25 cm (10 in), hardwood	0.905	5	1.392	1.378-1.406	0.000
R4	Overstory dbh ≥ 25 cm (10 in)	0.851	2	0.918	0.907-0.929	0.000

¹Mean 10-fold cross-validated²Mean 10-fold cross-validated regularized training gain³95% confidence interval of regularized training gain**Table A-8. Resting habitat model (biotic only) variable percent contribution (%) and permutation importance (PI); R1 is the selected model.**

Variable	R1		R2		R3		R4	
	%	PI	%	PI	%	PI	%	PI
Overstory dbh ≥ 25 cm (10 in)	27.0	29.0	27.9	32.6	71.6	73.0	100	100
Hardwood	9.5	8.0	10.9	10.3	28.4	27.0		
Forest type	59.1	60.8	61.2	57.1				
Total tree cover $\geq 70\%$	4.4	2.1						

After training the model within the model extent, we projected the results over the entire assessment area. We evaluated uncertainty in projections by inspecting the multivariate environmental similarity surface produced by Maxent. This analysis compares environmental conditions in projection areas to those in the training area; predictions become less certain as variable values differ significantly from values observed in the training area (Elith et al. 2010).

Two versions of the final model were produced—a continuous logistic output and a binary output (0.128 threshold for abiotic and biotic model and 0.193 threshold for biotic only model) using the criteria described in Section A-2. To create a model evaluation dataset, fisher resting site locations not used in model calibration (filtered to a minimum nearest neighbor distance of 800 m (0.50 mi), N=129) were combined with random points (N=130, minimum nearest neighbor distance of 800 m [0.50 mi]) generated within a 10-km (6.2 mi) buffer of the calibration resting sites. We used both threshold-dependent and independent measures to evaluate the final model using the same procedures described in Section A-2.

A-5 Denning Habitat

We modeled predicted probability of fisher den occurrence using Maxent, 154 fisher den (maternal and natal) localities, and the same model selection and evaluation procedures described in Section A-4. The SNAMP and KRFP study teams (C. Thompson and R. Sweitzer, unpublished data) provided den locality data (N=350, spanning 2008 to 2013). We defined the model extent using a 25-km (15.5 mi) buffer envelope around den localities. To avoid non-independence of den locality data gathered from the same female during the same year (because mothers may be limited to moving kits relatively

short distances between dens), we randomly removed (“filtered”) localities so that no more than 1 den used by the same female in the same year fell within the moving window used to smooth environmental layers, using an 800-m (0.50 mi) minimum nearest-neighbor distance (approximate radius of a 2-km² circular moving window). Any den known to have been reused over multiple years was protected against removal on the assumption that multi-year reuse reflects high den site quality. Filtering reduced total sample size from 350 den localities to 154.

An array of 19 potential environmental predictor layers was created, including vegetation, topography, climate, and hydrology variables at 30 m resolution (Table A-9). We derived vegetation data from merged and rasterized USFS R5 existing vegetation tiles (attributes “TOTAL_TREE_CFA,” “OS_TREE_DIAMETER_CLASS_1,” and “WHRTYPE”). Multiple measures of tree cover and size were tested for inclusion in the model. We obtained elevation data for topographic variables from a 30-m (100 ft) digital elevation model (National Elevation Dataset, USGS) and extracted hydrology data from the National Hydrography Dataset (USGS). Climate data were derived by resampling 270-m (885 ft) downscaled historic climate data (Flint and Flint 2012, California Basin Characterization Model, <http://climate.calcommons.org/dataset/10>). All environmental variables were averaged over a 2-km² (500 ac) moving window, except for distance to nearest perennial and intermittent water features. The 2-km² (500 ac) resolution was used based on earlier electivity analyses performed to determine the resolution that provided the best fit (AUC value) to den locality data.

We created potential full models by individually substituting in multiple measures of tree cover, tree size, and climate with forest cover type, topographic, and hydrographic predictors. Variables with idiosyncratic response curves were dropped, and the remaining candidate full models were evaluated using AICc calculated with ENMTools 1.3 (Warren et al. 2008), selecting the one with the highest AICc weight as the best full model. The same stepwise variable removal and model evaluation procedures described in Section A-4 were used for den modeling (Tables A-10, A-11, A-12, A-13).

We projected the final model results onto the entire study area and evaluated uncertainty in projected areas by inspecting the multivariate environmental similarity surface, as described in Section A-4. Continuous logistic and binary (0.143 threshold for abiotic and biotic model and 0.182 threshold for biotic only model) versions were produced using the same criteria as described in Section A-2.

To create a model evaluation dataset, fisher den locations not used in model calibration (N=186) were combined with random points (N=190) generated within a 10-km (6.2 mi) buffer of the calibration dens (caveat: while test dens were not used in model calibration, they are not truly independent). We used both threshold-dependent and independent measures to evaluate our final model. For threshold-dependent measures, the maximum training sensitivity plus sensitivity threshold (Liu et al. 2013) was used for binary data classification. The R statistical package (version 2.15.3, R Core Team 2013) and the R package PresenceAbsence (version 1.1.9, Freeman and Moisen 2008) and POC code (Phillips and Elith 2010) were used to calculate several model evaluation measures appropriate for presence-background data (Table A-3).

Table A-9. Potential environmental predictors used in denning habitat modeling averaged within a 2-km² (500 ac) moving window (except where noted).

Variable	Description
Vegetation cover (source: USFS R5 existing vegetation tiles)	
Total tree cover $\geq 40\%$	Proportion with total tree cover from above $\geq 40\%$
Total tree cover $\geq 50\%$	Proportion with total tree cover from above $\geq 50\%$
Total tree cover $\geq 60\%$	Proportion with total tree cover from above $\geq 60\%$
Total tree cover $\geq 70\%$	Proportion with total tree cover from above $\geq 70\%$
Total tree cover $\leq 10\%$	Proportion with total tree cover from above $\leq 10\%$
Total tree cover $\leq 20\%$	Proportion with total tree cover from above $\leq 20\%$
Vegetation size (source: USFS R5 existing vegetation tiles)	
Overstory dbh ≥ 25 cm (10 in)	Proportion with mean diameter at breast height ≥ 10 in for trees forming the uppermost canopy layer
Overstory dbh ≥ 50 cm (20 in)	Proportion with mean diameter at breast height ≥ 20 in for trees forming the uppermost canopy layer
Vegetation type (source: USFS R5 existing vegetation tiles)	
Forest type	Proportion with CWHR type as aspen, eastside pine, Douglas fir, montane hardwood conifer, montane hardwood, Sierran mixed conifer, Ponderosa pine, Jeffrey pine, or white fir
Hardwood	Proportion with hardwood cover from above >0 and CWHR type as montane hardwood conifer or montane hardwood OR regional dominance type alliance as riparian mixed hardwood, interior hardwood, canyon live oak, black oak, interior live oak, black cottonwood, or montane mixed hardwood
Topographic (source: USGS National Elevation Dataset)	
Percent slope	Mean slope (%)
Insolation index	Mean insolation index
Climate (source: Flint and Flint 2012, California Basin Characterization Model)	
Precipitation	Mean annual precipitation (mm; 1971-2000)
Snowfall	Mean annual snowfall (mm; 1971-2000)
Spring snowpack	Average spring snowpack (mm; 1971-2000)
Winter snowpack	Average winter snowpack (mm; 1971-2000)
August max temperature	Average August maximum temperature (C°; 1971-2000)
January min temperature	Average January minimum temperature (C°; 1971-2000)
Hydrology (source: USGS National Hydrographic Dataset)	
Distance from water	Distance (m) to nearest perennial or intermittent stream or lake (no moving window averaging)

Table A-10. Denning habitat model selection (abiotic and biotic); R1 is the full model, and R2 is the selected model.

Model	Predictors	AUC ¹	Parameters	RTG ²	95% CI ³	AICc Weight
R1	Forest type, hardwood, total tree cover $\geq 60\%$, percent slope, August max temp, overstory dbh ≥ 25 cm (10 in)	0.946	15	1.892	1.879-1.904	0.255
R2	Forest type, hardwood, total tree cover $\geq 60\%$, percent slope, August max temp	0.946	16	1.887	1.875-1.899	0.745
R3	Forest type, hardwood, total tree cover $\geq 60\%$, percent slope	0.943	11	1.843	1.832-1.854	0.000
R4	Forest type, hardwood, total tree cover $\geq 60\%$	0.937	5	1.772	1.748-1.797	0.000
R5	Forest type, hardwood	0.931	4	1.684	1.658-1.710	0.000
R6	Forest type	0.929	1	1.626	1.600-1.653	0.000

¹Mean 10-fold cross-validated²Mean 10-fold cross-validated regularized training gain³95% confidence interval of regularized training gain**Table A-11. Denning habitat model (abiotic and biotic) variable percent contribution (%) and permutation importance (PI); R2 is the selected model.**

Variable	R1		R2		R3		R4		R5		R6	
	%	PI	%	PI	%	PI	%	PI	%	PI	%	PI
Forest type	62.0	66.7	74.1	69.4	77.2	86.6	81.1	87.8	94.4	95.7	100	100
Hardwood	4.8	5.0	6.4	5.8	5.0	2.7	5.5	4.0	5.6	4.3		
Total tree cover $\geq 60\%$	15.6	11.5	17.2	12.1	16.8	8.9	13.4	8.2				
Percent slope	1.2	3.6	0.9	2.8	1.0	1.8						
August max temp	2.9	9.0	1.5	9.9								
Overstory dbh ≥ 25 cm (10 in)	13.6	4.2										

Table A-12. Denning habitat model selection (biotic only); R1 is the full model and R2 is the selected model.

Model	Predictor	AUC ¹	Parameters	RTG ²	95% CI ³	AICc Weight
R1	Forest type, hardwood, total tree cover $\geq 60\%$, overstory dbh ≥ 25 cm (10 in)	0.936	7	1.780	1.756-1.805	0.461
R2	Forest type, hardwood, total tree cover $\geq 60\%$	0.937	5	1.772	1.748-1.797	0.539
R3	Forest type, hardwood	0.931	4	1.684	1.658-1.710	0.000
R4	Forest type	0.929	1	1.626	1.600-1.653	0.000

¹Mean 10-fold cross-validated²Mean 10-fold cross-validated regularized training gain³95% confidence interval of regularized training gain

Table A-13. Denning habitat model (biotic-only) variable percent contribution (%) and permutation importance (PI); R2 is the selected model.

Variable	R1		R2		R3		R4	
	%	PI	%	PI	%	PI	%	PI
Forest type	70.9	87.1	81.1	87.8	94.4	95.7	100	100
Hardwood	4.8	4.1	5.5	4.0	5.6	4.3		
Total tree cover $\geq 60\%$	10.9	7.3	13.4	8.2				
Overstory dbh ≥ 25 cm (10 in)	13.4	1.4						

A-6 Connectivity and Dispersal Habitat

We used Linkage Mapper (McRae and Kavanagh 2011) to analyze connectivity between predefined core habitat areas. Linkage Mapper uses user-defined core habitat areas and a resistance to movement (“cost”) surface layer to delineate normalized least-cost corridors. A previous cost surface layer was developed at 90-m (300 ft) resolution based on the opinions of fisher experts about how land cover, topography, roads, and other environmental factors are likely to affect the costs or risks of fishers dispersing across the landscape (CBI Sierra Nevada Carnivores project, Spencer and Rustigian-Romsos 2012). We updated this previous layer to encompass the current study area and to include the most recent available GIS data layers. We slightly modified previous costs in consultation with the Fisher Technical Team (FTT) to better reflect landscape genetic patterns found by Tucker et al. (2014), simplify land cover divisions, and increase the overall cost range to make least-cost corridor models more discriminating (Tables A-14, A-15). Total resistance value of each 90-m (300 ft) pixel was scored as the sum of the land cover resistance cost plus any additional feature costs in the pixel (Table A-15). To account for the effects of high-severity fires subsequent to vegetation map updates converting dense forest to open habitat, areas that burned at high fire severity between 2008 and 2012 were given an additional feature cost (50) to account for reduced vegetation cover following fire (Table A-15). To examine the potential impact of two large 2013 fires on connectivity (the Aspen and Rim fires), we updated the cost raster again by applying the 50-point severe-fire feature cost (Table A-15) to pixels burned at high severity in 2013 and compared least-cost corridor results before and after the fires. For the final cost layer, areas mapped as urban or open water were assigned maximum resistance values (325).

Table A-14. Resistance values (costs) for land cover types used for fisher least-cost corridor delineation.

WHR type	WHR size	WHR density	Cost
DFR, WFR, PPN, JPN, EPN, MHC, SMC, ASP, OR MHW	4 or 5	D	1
DFR, WFR, PPN, JPN, EPN, MHC, SMC, ASP, OR MHW	3	M or D	25
DFR, WFR, PPN, JPN, EPN, MHC, SMC, ASP, OR MHW	4 or 5	M	25
RFR, LPN, SCN, or MRI	3, 4, or 5	M or D	25
DFR, WFR, PPN, JPN, EPN, MHC, SMC, ASP, MHW, RFR, LPN, SCN, or MRI	2	M or D	50
DFR, WFR, PPN, JPN, EPN, MHC, SMC, ASP, MHW, RFR, LPN, SCN, or MRI	3, 4, or 5	S or P	50
MCP or MCH		D	50
All others			100

Table A-15. Additional feature costs used for fisher least-cost corridor delineation.

Feature	Additional Cost
Major roads	100
Slopes	
<50%	0
50-75%	25
>75%	50
Elevation	
<999 ft (305 m)	50
1,000-1,999 ft (305-610 m)	25
>1,000 ft (610 m)	0
Severe fires (high severity, 2009-2013/2008-2012)	50

While our cost surface and resulting least-cost corridors were not formally tested with independent data or subjected to a sensitivity analysis, we compared the resulting corridors with locations of 37 dispersal events recorded by the SNAMP study (See Section 3.2.1; R. Sweitzer, unpublished data). All but 1 dispersal events were within Core Area 5 reinforcing our assumption, and the genetic results of Tucker et al. (2014) that mature, dense forest are less costly to move through than other vegetation types. The 1 male disperser that crossed the San Joaquin River several times between Cores 5 and 4 appeared to do so within our modeled linkage area. In addition, we inspected aerial imagery within modeled corridors, which strongly fit with biological intuition about likely fisher movement habitats, such as by primarily following riparian corridors and other thickly vegetated areas and avoiding extreme slopes, barren areas, etc. While a quantitative comparison with the genetic results of Tucker et al. (2014) was not feasible, our predictions were generally consistent with Tucker et al. (2014) results that water and roads were associated with increased resistance to gene flow for females, and that they primarily disperse within dense, mature forest. Note that it may not actually be the water (i.e., stream) or road that creates resistance to gene flow, but that water and roads in the Assessment Area tend to be in canyons, which also feature steep slopes, open vegetation and other dispersal-resistant characteristics that we assigned high cost in our analyses.

We ran Linkage Mapper on the cost surfaces using 3 different approaches for defining what to connect: edge-to-edge connections between cores, core centroids, and nodes subjectively placed at select termini. These represent different assumptions about fisher dispersal: edge-to-edge analyses assume that a fisher will decide where to cross between adjacent cores while at the edge of 1 core. This approach is more likely to identify corridors with shorter Euclidean distances between cores than other approaches. Centroid-to-centroid analyses assume that fishers will disperse from somewhere near the heart of 1 core to the heart of the next. This approach provides more flexibility (mathematical “room to roam”) and is less likely to select the shortest crossing as the least costly, compared to edge-to-edge runs. For the select termini approach, we used the northern terminus of the northernmost core (Core 7), the southern terminus of the southernmost core (Core 2), and 4 points of maximum predicted habitat value within the southeastern core (Core 1). Cores 2 through 7 form a quasi-linear alignment along the west slope of the Sierra Nevada, such that running a single model from the northern to the southern tip captures the least costly connections between each pair of neighboring cores with a single modeled corridor, while providing maximum “room to roam” between each pair of cores. Because of the sequential alignment of Cores 2 through 7, all 3 approaches provide very similar least-cost corridors between core pairs, increasing confidence in the predictions. However, compared

with other cores, Core 1 is a highly convoluted polygon of relatively low value with several pockets of higher-value habitat. Fishers may be more likely to disperse from or to these higher-value areas, such that east-west connections between Cores 1 and 2 could take multiple potential routes because of their geographic dispersion. We therefore placed termini in each of 4 higher-value habitat areas within Core 1 to investigate how this influenced predicted dispersal corridors across the Kern River watershed. For each approach, we ran the model using normalized least-cost corridors (NLCC), with cost-weighted distances of <25 km (15.5 mi), <50 km (31 mi), and <75 km (46.5 mi). After reviewing results with FTT fisher experts, we used the union of the 50-km (31 mi) NLCCs produced by all 3 approaches to comprehensively represent likely fisher dispersal habitat or movement corridors between cores.

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