



SUGAR PINE FISHER PROJECT FINAL REPORT

A Continuation of the
Sierra Nevada Adaptive
Management Project (SNAMP)

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EXECUTIVE SUMMARY

Fishers (*Pekania pennanti*) are medium-sized mammalian carnivores with a pre-European distribution encompassing the boreal forest zone of Canada, the Great Lakes region and northeastern United States, a relatively limited portion of the Rocky Mountains in the United States, and mountainous areas of Washington, Oregon, and California, USA (Powell 1993). Ecologically, fishers are a mature or old forest-obligate species (Zielinski et al. 2005) associated with closed canopy, multi-story forests and large, decadent structures. Range wide, their numbers were reduced historically by the combination of intensive trapping and loss of forest habitats (Powell 1993, Powell and Zielinski 1994). The species is uncommon to rare in the western United States, listed as a sensitive species by the Oregon Department of Fish and Wildlife and endangered by Washington State. Conservation planning has been underway in California since 2013 to develop an approach to maintaining viable populations of fishers in both northwestern California and in the southern Sierra Nevada. In July 2015, the California Fish and Game Commission voted to list the southern Sierra Nevada fisher population as threatened, though final confirmation of this listing is still pending. On April 14th, 2016, the US Department of Fish and Wildlife ruled that the West Coast Distinct Population Segment (DPS) of fishers did not warrant listing under the Endangered Species Act. This decision has been challenged, and as of the time of writing California fishers remain in candidate status.

In 2007, a collaborative research effort between the University of California Berkeley and the US Forest Service was initiated on the Bass Lake District of the Sierra National Forest. Named the Sierra Nevada Adaptive Management Project (SNAMP), the project was designed to quantify the various ecological impacts of fuel reduction activities and provide an independent evaluation of how vegetation management, as prescribed by the 2004 Sierra Nevada Forest Plan Amendment (USDA 2004), affected fire risk, wildlife, forest health and water.

As part of this effort, UC Berkeley implemented large-scale research program focused on understanding the ecology of fishers in the region, and the impact of vegetation management on fisher behavior, survival, population stability, and habitat selection. In 2013 the US Forest Service Pacific Southwest Research Station assumed day-to-day management of the SNAMP Fisher Project. In 2015, as the larger SNAMP effort was concluding, the Forest Service determined that due to delays in implementation of vegetation treatments, insufficient data had been collected to determine impacts to fisher. The project was therefore extended for an additional 2 years to collect post-treatment data, and renamed the Sugar Pine Fisher Project. A

major goal of the SNAMP/Sugar Pine Fisher Project was to determine whether current rates of survival and reproduction will allow fishers to persist in the Sierra Nevada in the context of active forest management to reduce fuels and the risk of severe wildfire.

A total of 143 individual fishers were captured, and 139 fishers were radio collared between Dec 2007 and Dec 2017 (78 females, 65 males). We collected 77 mortalities during the study (75 collared and 2 uncollared); annual survival rates ranged from 0.40 for juvenile males to 0.76 for adult females. The most common cause of death (61%) was predation, with felid carnivores (bobcats; *Lynx rufus*, and mountain lions; *Puma concolor*) as the primary predators. Disease accounted for 14% ($n = 6$) of documented mortalities, though it likely contributed to several other mortalities. Another 6 fishers (14%) died as a result of acute anticoagulant rodenticide exposure, linked with illegal marijuana grow sites in the Sierra National Forest, and three fisher deaths were associated with vehicular collision. Circumstantial evidence also indicates that these mortality factors are not necessarily independent; exposure to toxins associated with these grow sites also inflates the mortality rate associated with ‘natural’ sources such as predation and disease.

Eighty-one denning attempts were made by collared, breeding-age female fishers and we recorded 56 litters born, using a combination of direct observation and remote cameras. A total of 93 kits were observed, in an average litter size of 1.5 kits. Female fishers used seven different tree species for dens, though incense cedar, white fir, and black oak accounted for 93% of identified dens. Mean height and diameter at breast height (DBH) of hardwoods used as den trees was 19.8 m and 74.8 cm, respectively. For conifers, mean height and DBH was 34.6 m and 119.7 cm, respectively. Parturition occurred exclusively in tree cavities, though female fishers used a variety of structures to house dependent kits as they developed. While reproduction appeared relatively stable across years, fishers in the southern Sierra Nevada appear to produce fewer kits per litter than fishers in other parts of North America.

Camera surveys were a major aspect of the SNAMP/Sugar Pine Fisher Project in all nine years of the study. Between November 2007 and November 2016, camera stations were placed in 794 unique 1-km² grid cells. We detected fishers in 403 cells, resulting in annual naïve occupancy rates that ranged from 0.236 to 0.660 throughout the study. Across the entire study period and area and accounting for imperfect probability of detection, occupancy was 0.457 and appeared positively related to canopy cover, stand density, and the presence of large hardwoods.

An analysis of patterns of occupancy indicated that fishers remained in the vicinity of fuel reduction projects as long as they were able to move around thinned areas

Given the observed reproduction and survival rates, the population growth rate over all 10 years of the study was 0.99. We calculated the potential range of lambda values based on the observed 95% confidence intervals in female survival rates; results indicated that lambda could potentially range from 0.862 to 1.104. Actual annual variation, based on the observed annual variation in adult female survival, ranged from 0.917 in 2010 and 2013 to 1.062 in 2014. Given the length and intensity of the monitoring associated with calculating these estimates, and the lack of significant difference from zero, the SNAMP/Sugar Pine fisher population appears stable over the study period.

During the SNAMP/Sugar Pine Fisher Project, over 30,500 acres of the overall study area were treated for fuel reduction using a combination of commercial harvest, timber stand improvement efforts, and prescribed burning, however only a relatively small percentage of this acreage overlapped with active fisher remote camera monitoring efforts in both time and space (7, 87, and 127 grid cells for prescribed burning, timber harvest, and timber stand improvement (TSI) efforts, respectively). Fishers appeared to be less influenced by fuel reduction activities than by the initial site conditions, however the percentage of a grid cell impacted by harvest or TSI efforts significantly decreased the probability that a fisher was detected in that cell during the year of management activity. This impact did not appear to extend beyond the year in which management activity was conducted, as the time since treatment did not appear to influence occupancy, colonization, or extinction rate estimates.

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INTRODUCTION

Project Background

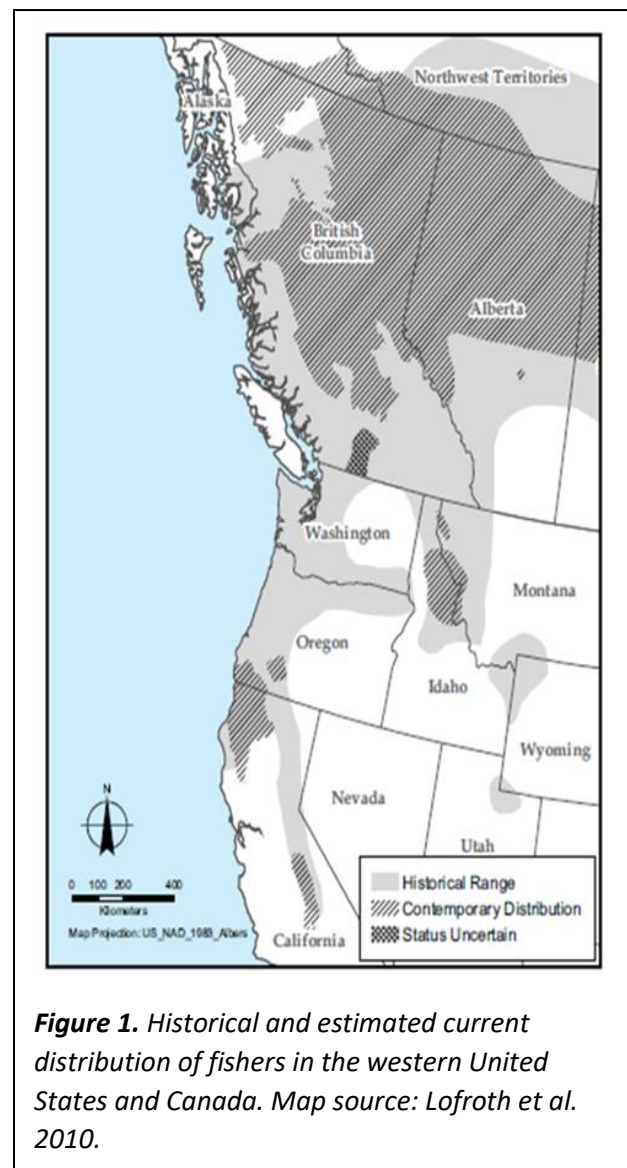
Fishers are a medium-sized mammalian carnivore with a historic distribution encompassing the boreal forest zone of Canada, the Great Lakes region and northeastern United States, a relatively limited portion of the Rocky Mountains in the United States, and mountainous areas of Washington, Oregon, and California, USA (Powell 1993). In the Pacific states, indigenous fishers currently exist in three remnant populations in southern Oregon, northern California, and the southern Sierra Nevada, California (Zielinski et al. 2005). In California the fisher occupies less than half of its historical range as described by Grinnell in the early 1900s (Grinnell et al. 1937), and the two remnant populations are separated by approximately 400 km. A third population, established via translocation starting in 2009, is now established in the central Sierra Nevada region. While most states banned fisher trapping in the 1930s, California was the last state to ban trapping. Despite evidence that fisher fur harvests were declining as early as 1924 (Hall 1942), trapping wasn't banned in California until 1946 and the population did not recover following cessation of trapping. Until recently, decreases in the range and numbers of fishers in California were considered to be a result of widespread timber harvest and fur trapping during the early to mid-1900s (Zielinski et al. 2005). Recent genetic research has complicated the picture, suggesting that the northern California and southern Sierra Nevada populations may have been genetically isolated prior to European settlement (Tucker et al. 2012). Whether this isolation stemmed from extirpation of fishers in the central Sierra Nevada, or from reduced genetic flow due to local topographic obstacles is unknown.

Notwithstanding uncertainty regarding the timing or cause of the population decline and range retraction, there may be fewer than 500 total fishers in the southern Sierra Nevada population (Spencer et al. 2011), where the species currently occupies approximately 4,400 km² of mid-elevation, mixed-coniferous forest (Spencer et al. 2015). In October of 2014, the US Fish and Wildlife Service proposed listing the West Coast Fisher Distinct Population Segment, meaning fishers in California, Oregon, and Washington, as threatened under the federal Endangered Species Act (<http://federalregister.gov/a/2014-23456>). In April of 2016, the US Fish and Wildlife Service determined that the West Coast Distinct Population Segment was collectively not threatened under the U.S. Endangered Species Act. This decision has been appealed, and pending a conclusion of the appeal, fishers are currently considered candidate

species under the Federal ESA. In August of 2015, the California Fish and Game Commission determined that the southern Sierra Nevada fisher population should be listed as Threatened in California. Fishers are also considered a sensitive species by the United States Forest Service and the Bureau of Land Management, and are listed as a species of Special Status in Yosemite National Park. Fishers are currently listed as state endangered by the Washington Department of Fish and Wildlife, and as a species of concern in Oregon.

Current Fisher Range and Population Trends

Grinnell et al. (1937) described the original range of the fisher in California as including the entire western slope of the Sierra Nevada, the southern Cascades, Klamath Mountains, and northern Coast Range, a total area of ~100,000-110,000 km² (Spencer et al. 2015) (Fig. 1). Lofroth et al. (2010) estimated that the current range of the fisher in California represents <50 percent of the historical range, with fishers absent from most of the northern and central Sierra Nevada, leaving a ~400-km gap separating the two remnant indigenous populations in the state (Zielinski et al. 1995) (Fig. 1), one in the northern Coast Range and one in the southern Sierra Nevada. In a recent statewide assessment, Spencer et al. (2015) used an updated fisher locality database to estimate their current geographic range in California at ~52,000 km², including ~40,000 km² in northern California and ~12,000 km² in the southern Sierra Nevada. While it is likely that the historical range estimates were inflated, this still represents an approximately 50% loss in occupied habitat (Spencer et al. 2015).



The southern Sierra Nevada fisher population is small, potentially including only ~500 total individuals and <300 adult fishers (Spencer et al. 2011), but appears to be stable over about the past decade (Zielinski et al. 2013a). Following substantial population contractions in the past (Knaus et al. 2011), fishers in this part of California may have expanded in the late 20th century (Tucker et al. 2014). The overall distribution of fishers in the southern Sierra Nevada has been monitored using a combination of track plates and motion detecting cameras since the mid-1990s (Truex et al. 1998, Zielinski et al. 2005, Jordan 2007). Zielinski et al. (2013a) analyzed occupancy records from this effort for the period 2002 to 2012, when a systematic survey design was in place, and found no detectable change in occupancy for the entire area or for any of the three subareas examined. Furthermore, the authors stated that “constant and positive persistence values suggested that sample units rarely changed status from occupied to unoccupied or vice versa” (Zielinski et al. 2013a). However, this evidence can be interpreted in two ways. The lack of a decreasing trend may indicate a small but stable population. Conversely, the lack of an increasing trend may indicate that despite being protected from fur harvest for over 60 years during a time when large scale clearing of forest habitat was diminished (Collins et al. 2010), the population is showing no evidence of recovery. Interpretation is further complicated by a combination of genetic patterns and survey data suggesting that the population north of the Kings River may have expanded during the 1990s, before the regional monitoring program was established (Tucker et al. 2014). Most recently, Zielinski and Gray (2018) determined that resting habitat availability remained constant in the southern Sierra Nevada between 1998 and 2010, and in 2017 a male fisher was observed north of the Merced River in Yosemite National Park, the first wild-born fisher observed in that area in nearly 100 years (S. Stock, personal communication).

Insight from previous research in the nearby High Sierra Ranger District of the Sierra National Forest suggests that fisher population densities range from 0.07 to 0.28 fishers/km² (Jordan et al. 2011, Thompson et al. 2012a). Earlier analyses on the SNAMP/Sugar Pine landscape estimated population densities ranging from 0.075 to 0.098 fishers/km², with a population growth rate of 0.966 (95% confidence interval between 0.786 and 1.155; Sweitzer et al. 2015). Results from research in northern California (Hoopa Study) indicate the potential for fisher densities to change rapidly. In the Hoopa Valley area of Northern California, fisher densities were estimated at 0.52 fishers/km² in 1998, but fell to 0.14 fishers/km² in 2005 (Matthews et al. 2013a). Due to the apparent variability in demographic estimates and the

potential for rapid change, developing precise estimates for different subpopulations and in different habitat types is critical for effective management.

Project Management and Conservation Planning

Federal and state resource agencies are currently developing strategies to aid in the maintenance of viable populations of fishers in both northwestern California and in the southern Sierra Nevada. It is possible that the isolated population of fishers in the southern Sierra Nevada will be impacted as the USDA Forest Service implements fuel reduction measures (Scheller et al. 2011). Fuel reduction treatments are becoming the dominant forest management activity in western forests in response to increases in the frequency of intense, stand-replacing forest fires over the past several decades (Mallek et al. 2013, Safford 2013). Advances in fire modeling have greatly improved managers' ability to plan and evaluate various landscape fuel treatment scenarios intended to reduce fire risks (Collins et al. 2010, Scheller et al. 2011). However, there remains a considerable gap between modeling landscape-scale fuel treatments and implementing them due to concern over the status of rare and uncommon species associated with multi-storied, late-seral stage forests, such as fishers and spotted owls (*Strix occidentalis occidentalis*) (Naney et al. 2012, Truex and Zielinski 2013). Presence of fishers has strongly influenced managers' ability to delineate landscape-scale fuel treatments in this fire-prone region (Collins et al. 2010, Thompson et al. 2011a). The Sierra Nevada Forest Plan Amendment (SNFPA; USDA 2004) represents the most recent attempt to reconcile the need to reduce fuel loadings in Sierra Nevada mixed-conifer forests and retain characteristics of late-successional forests important for these species. The strategy involves a network of "Strategically Placed Area Fuel Treatments" (SPLATS) that allow up to a 60% reduction in basal area and a 30% reduction in canopy cover in Sierra mixed conifer forest. In the long-term, this strategy may increase availability of important habitats for species such as the fisher and California spotted owl by reducing wildfire-induced losses (Spencer et al. 2011), but treatments may also impact habitat quality for fishers in the near-term (Thompson et al. 2011a).

To provide a framework for balancing the habitat needs of fishers with fuel treatments intended to reduce fire risks, the original Sierra Nevada Adaptive Management Project (SNAMP) initiated a coordinated effort to assess the effects of fuel treatments on many environmental features including the fisher, spotted owl, forest health, and water quality and quantity in the central Sierra Nevada. SNAMP was a collaborative effort between the University

of California Berkeley, the USDA Forest Service Region 5, and additional academic, agency, and public partners. SNAMP began in 2007 and was designed to evaluate the effects and effectiveness of fuel treatments implemented according to the revised Sierra Nevada Framework (USFS 2004) under a design that incorporated stakeholder participation. SNAMP was a landscape-scale, ecosystem-level experiment in natural resource management and involves a Before-After-Control (BACI) design developed specifically to assess the impacts of SPLATs on the overall forest ecosystem (Popescu et al. 2012).

In 2013, at the request of the USDA Forest Service Region 5 and UC Berkeley, the US Forest Service, Pacific Southwest Research Station (PSW) assumed the day-to-day management of the SNAMP Fisher Project. In 2015, active field monitoring for SNAMP ceased and the focus changed to analysis and report preparation. However, because implementation of vegetation management activities had been delayed 2-3 years throughout the lifespan of SNAMP, it was determined that insufficient post-treatment data had been collected to accurately represent the impacts of fuel reduction efforts on fishers. The SNAMP Fisher Project was therefore extended for an additional 2 years, under PSW management, and renamed the Sugar Pine Fisher Project. Primary objectives of the Sugar Pine project were to 1) continue to document reproductive rates in treated and untreated areas, 2) continue to document survival rates and causes of mortality in treated and untreated areas, 3) monitor population stability post-treatment, and 4) provide a better understanding of fisher response to the vegetation management guidelines outlined under the 2004 Sierra Nevada Forest Plan Amendment (USDA 2004). This report summarizes data collected to address these objectives during both the UC Berkeley and USDA Forest Service PSW research efforts.

SITE DESCRIPTION AND STUDY AREA

The SNAMP/Sugar Pine Fisher Project study area is located at the northern end of the southern Sierra Nevada fisher population in California, encompassing the area bounded by the Merced River in the north and the San Joaquin River in the south (Fig. 2). It consists of several nested landscapes. The core of the project area is the SNAMP key watershed region, four watersheds encompassing 128 km² (Fig. 2). These four watersheds (Sugar Pine, Nelder Creek, Ranier Creek, White Chief Branch) represent the two control and two treatment watersheds of the SNAMP research effort. Within the key watershed region, management activities were designed to facilitate evaluation of the impacts of fuel and vegetation management on fisher survival, habitat selection, and reproduction.

Due to the low density and large home ranges of fishers, a larger SNAMP Fisher Project study area was defined as the area needed to reach and maintain the project goal of monitoring 20 radio-collared animals at all times to obtain more precise estimates of demographic rates. This larger study area, 1300 km², encompassed a mix of public and private land and is topographically complex with elevations ranging from 758 m to 2652 m (Fig. 3). Administratively, the focal study area for the research is the non-wilderness region of the Bass Lake Ranger District in the Sierra National Forest, however it extends into the southern portion of Yosemite National Park. Field work was carried

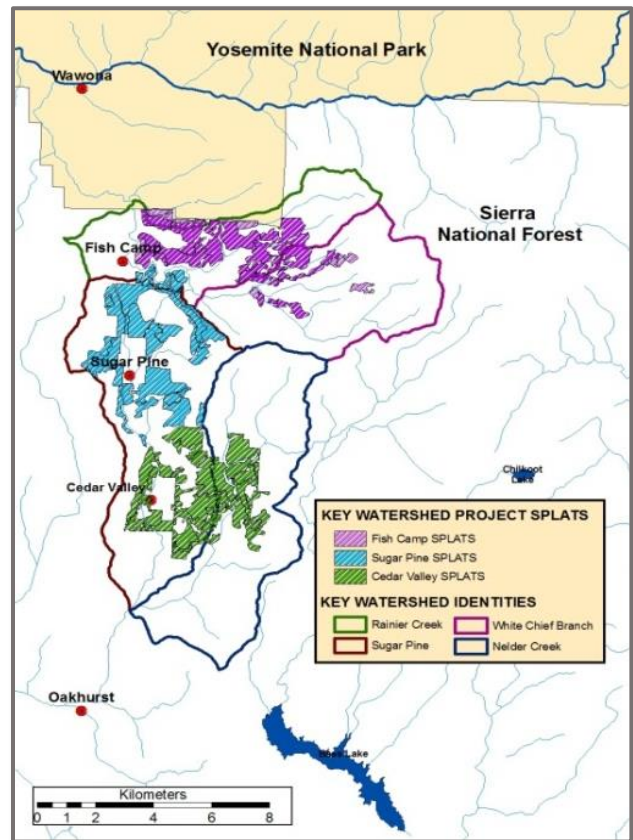


Figure 2. Placement of the four key watersheds within the SNAMP/Sugar Pine study area, and the distribution of associated fuel reduction treatments.

out between 1,000 m and 2,400 m in elevation, corresponding to fisher occurrence in the region. This elevation gradient includes California Wildlife Habitat Relationship (CWHR) system habitat types for Montane Hardwood (MHW), Ponderosa Pine (PPN), Montane Hardwood-Conifer (MHC), and Red Fir (RFR). Dominant tree species across the study area include a mix of conifers (ponderosa pine [*Pinus ponderosa*], Jeffrey pine [*P.*

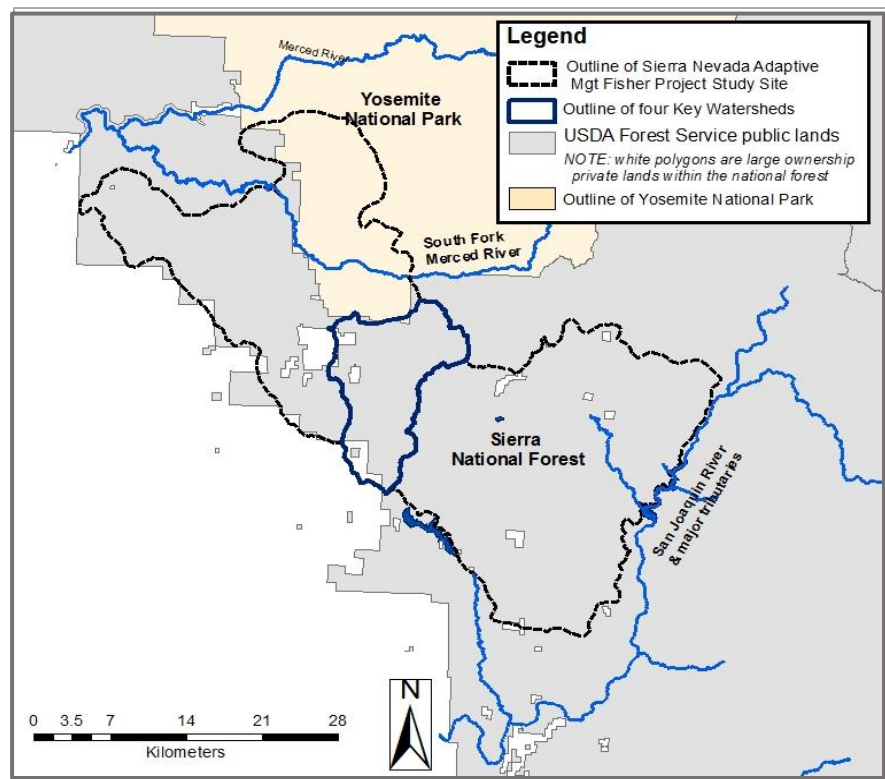


Figure 3. Location of the overall study area of the SNAMP Fisher Project, including administrative boundaries and the outer boundary of the key watersheds focal study area in the approximate center of the map region.

jeffreyi], sugar pine [*P. lambertiana*], lodgepole pine [*P. contorta*], white fir [*Abies concolor*], red fir [*A. magnifica*], and incense cedar [*Calocedrus decurrens*]), and hardwoods (California black oak [*Quercus kelloggii*], canyon live oak [*Quercus chrysolepis*], white alder [*Alnus rhombifolia*], and mountain dogwood [*Cornus nuttallii*]). Common shrubs and tree-like shrubs include whiteleaf manzanita [*Arctostaphylos viscida*], greenleaf manzanita [*A. patula*], mountain misery [*Chamaebatia foliolosa*], bush chinquapin [*Chrysolepis sempervirens*], mountain whitethorn [*Ceanothus cordulatus*], hazelnut (*Corylus cornuta*), mountain dogwood (*Cornus nuttallii*), and Sierra gooseberry [*Ribes roeslii*]. Giant sequoias [*Sequoiadendron giganteum*] are present, but primarily restricted to the Nelder Grove Historic Area within the Nelder Creek watershed. Permanent streams in the key watersheds are important for fishers and other wildlife and include Big Creek and Rainier Creek in the Rainier Creek watershed, Lewis Creek in the Sugar Pine watershed, and California Creek and Nelder Creek in the Nelder Creek watershed.

A third study area was delineated following the conversion between the SNAMP and Sugar Pine projects. The primary intent of the project extension was to document fisher response to management actions, and to determine if management impacted fisher demographic parameters.

Therefore the 20 animal objective was relaxed, and research efforts focused on areas of recent or ongoing management activities. This intermediate-sized study area included the key watersheds and approximately 911 km² of the larger SNAMP Fisher Project study area, excluding the northwestern (Sweetwater Ridge), northeastern (Yosemite National Park), and southeastern (Mammoth Pool Reservoir) extremes. The additional acreage beyond the key watersheds allowed for the monitoring of animals in the vicinity of the Gray's Mountain, Sonny Meadows, Whiskey Ridge fuels management projects.

METHODS

Field Methods

Live Trapping

Individual fishers were live-captured in steel mesh traps (Tomahawk Live Trap Company, Tomahawk, WI) modified to include a plywood cubby box to provide the animals with a secure refuge where they were less likely to injure themselves (Wilbert 1992). Trapping to mark animals with radio collars was focused during the fall and winter seasons between December 2007 and March 2016. Animals were also trapped between December 2016 and March 2017 to remove collars upon the completion of the project. We did not trap during the spring denning period (late March to mid-June) to minimize disturbance to reproduction, with the exception of the first year of the study when we needed to capture fishers to initiate the study.

Live traps were baited with chicken, bacon, and venison, and lures such as Gusto and Fisher Red. Traps were checked daily, as early as possible, between the hours of 0630 and 1100. Captured animals were restrained in a handling cone, and sedated using a mixture of Ketamine hydrochloride (22.5 mg/kg) and Diazepam or Midazolam (0.125 mg/kg) injected intramuscularly. During handling, vital signs (temperature and respiration) were monitored closely (every 5-10 minutes). Sedated fishers were weighed, classified by age and sex based on examination of teeth, genitalia, and sagittal crest, and measured for standard morphological features. Small samples of ear tissue were collected for microsatellite DNA analysis using a sterile dermal biopsy punch. Several strands of hair were removed from the nape and rump region, also for DNA analysis. Hair samples were stored in a dry paper envelope, whereas tissue samples were stored in 95% ethanol until analysis at the USDA Forest Service Wildlife Genetics Lab (Rocky Mountain Research Station, Missoula MT). Teats on females were measured for base diameter and height using digital calipers (\pm 1mm), and those data were used to identify

females that weaned at least 1 kit when they had not been monitored during the denning period (Matthews et al. 2013b). Each animal was permanently identified by subcutaneous insertion of a passive integrated transponder (PIT) tag (Biomark, Boise, ID), and fitted with a very high frequency (VHF) radio collar (Holohil Systems Model MI-2M, Ontario, Canada). Custom breakaway devices were inserted into radio collars to reduce the risk of injury or strangulation between recaptures and to allow for growth (Thompson et al. 2012b). After processing, animals were returned to the cubby box until they were fully recovered from anesthesia, and then released in the same location where they were captured. Capture and handling procedures followed American Society of Mammologist guidelines (Sikes and Gannon 2011), and were approved by Institutional Animal Care and Use Committees of the University of California, Berkeley (protocol R139), the University of California, Davis (16302 and 18022), and the USDA Forest Service (2018-001).

Live-trapping is labor intensive, and the effort was designed to gain advantage from detections of non-collared fishers at cameras. Live traps were most frequently placed in the same area of camera stations after cameras had been removed (to prevent interference with camera surveys). Data from camera detections were used to design linear trap lines of 5-10 traps bracketing positive detection stations. Trapping success was enhanced in later years of the study by placing traps in locations where fishers had been captured in the past. In winter, snow falling from tree branches can ice up the treadle mechanism inside live traps and prevent the trap door from closing. To preclude this possibility, we used lightweight, canvas tarps or Coroplast (corrugated plastic sheets) to protect the inside of the live traps from falling snow, and debris,



Illustration 1. (A) Camouflaged live trap near the base of a white fir, and (B) a radio-collared fisher being released after processing.

and to provide some measure of thermal protection. Bark, branches and other debris were added to the tops and sides of traps for camouflage and to deter bears (*Ursus americanus*) from rolling the set traps (Illustration 1).

Telemetry Monitoring – Aerial and Ground

We used fixed-wing airplanes to monitor and relocate radio-collared fishers for the entirety of the SNAMP/ Sugar Pine Fisher Project. Beginning in December 2007, we worked with USDA Forest Service Supervisory Pilot to develop an aviation program in support of SNAMP Fisher Project and the program was continued with the Sugar Pine Fisher Project. This



Illustration 2. Cessna 185 and Piper PA-18 Super Cub, call sign “Weasel One”, used from December 2007 to November 2016 to locate fishers for the SNAMP/Sugar Pine Fisher Project.

program was fully established in August 2008 when a full time pilot was hired and the first of two dedicated aircraft were based at the Mariposa-Yosemite Airport in Mariposa, CA (Illustration 2). The two USDA Forest Service-owned aircraft acquired for supporting

the project were a Cessna 185 (Cessna Aircraft Co., Wichita, KS) and a Piper PA-18 Super cub (Piper Aircraft Inc., Vero Beach, FL). Two aircraft were necessary to maintain continuous monitoring of radio-collared fishers when routine maintenance or engine repair was necessary (John Litton, personal communication).

The optimal search procedure used when locating animals from light aircraft varies depending on the number of animals tracked, and the antenna configuration supported and approved for the airplane being used (Gilmer et al. 1981). Additional details are provided elsewhere (Thompson et al. 2012b), but we used two, 2-element H antennas (Telonics Inc., Mesa, AZ) mounted in a sideways configuration on each wing strut, and a single 3-element Yagi antenna (Advanced Telemetry Systems, Isanti, MN) mounted forward-facing on the right wing strut. This antenna configuration was effective in allowing the pilot and biologist to search for

radio-collared fishers using the Yagi antenna (detection range 5-20 km), then switching to the H-antennas to localize to a relatively precise location above the animal using a circling technique (Seddon and Maloney 2004).

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

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

Remote Camera Surveys

A 1 km² grid was used to define remote camera survey units across the entire study area (Fig. 4). Motion sensing cameras (Silent Image Professional, Rapidfire PC85; RECONYX Inc., Holmen, WI) were systematically deployed near the center of a grid cell in the study area beginning at the start of each camera survey year (October 15–October 14 the following year). Placement of cameras within 1 km² grid cells was determined based on the presence of habitat elements important for fishers (e.g., presence of mature or large diameter trees, moderate to steep slopes, canopy cover $\geq 60\%$, proximity to permanent streams; Purcell et al. 2009, Zielinski et al. 2004).

Cameras were focused on bait trees upon which we attached baits and applied scent lures as attractants (Illustration 3). Baits were pieces of venison or chicken in a sock, and 4–10 hard-shell pecans strung onto a wire (initial purpose was to index squirrel abundance, but were also

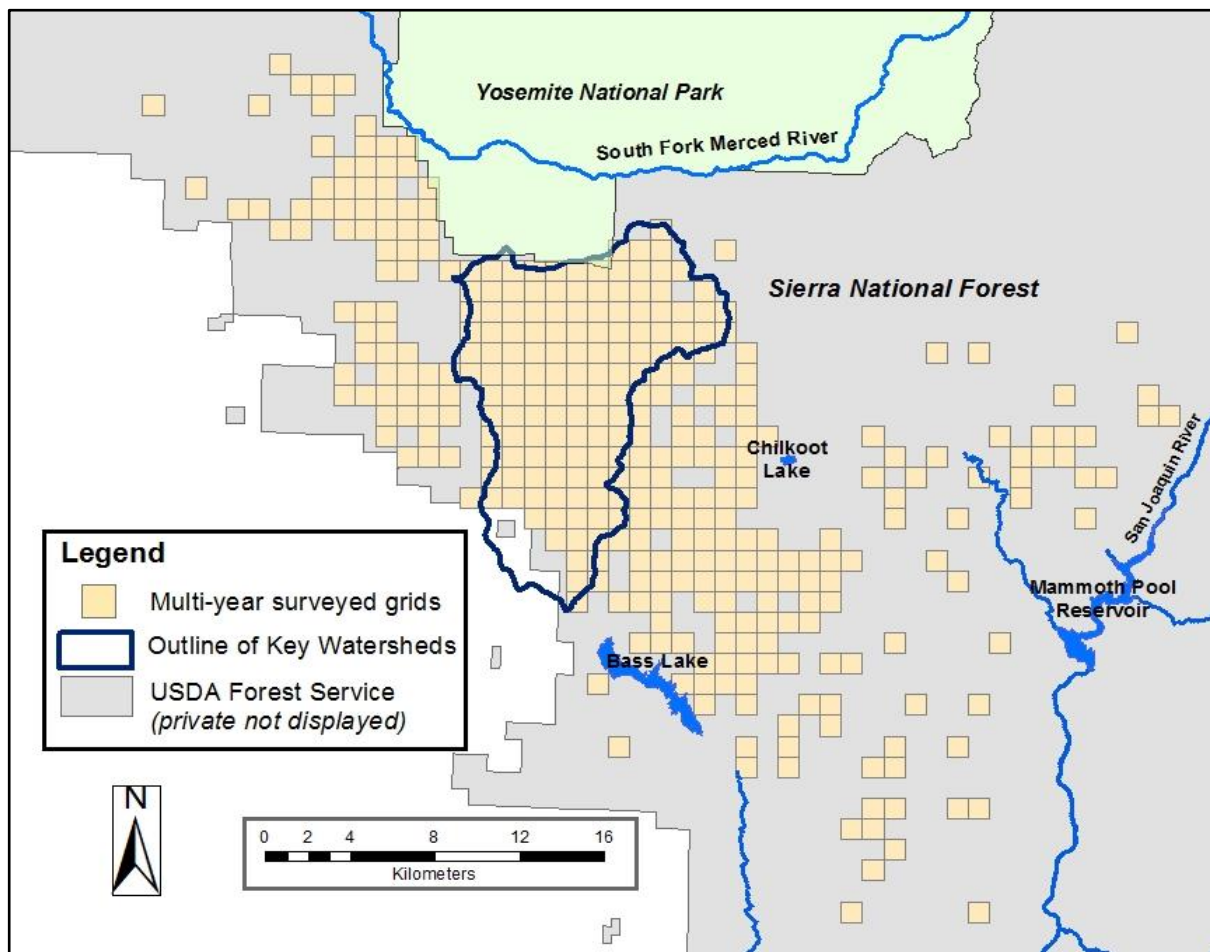


Figure 4. Distribution of 1-km² survey grids throughout the SNAMP/Sugar Pine Fisher Project study area on the Bass Lake District of the Sierra National Forest, CA.



Illustration 3. (A) Large DBH bait tree with nut ring and bait sock containing raw chicken and smeared with fisher scent lure, and (B) smaller DBH camera tree with game camera secured to the tree with a ratchet strap, cable and pad lock and encased in a protective metal bear box.

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

All camera images were processed and tagged according to protocols (see Appendix). Each image was viewed and given a set of “tags” depending on the type of camera survey (key watershed or 1k grid), the check number, and the subject of the image such as “fisher” or “bear.”

All tags and associated image data were filed by 1–km² grid cell ID and the study year. Image tagging data were summarized and compiled by study year for analysis.

Fisher Reproduction

Identifying Den Trees and Evidence of Reproduction

Female fishers exhibiting behavior consistent with denning were identified from late March through mid-April. Denning behavior was characterized by an abrupt change in behavior based on patterns of successive aerial radio telemetry locations. When the dispersal pattern within a female's home range changed from various widely distributed locations to spatially clustered locations (3-5 locations within 500 m over a 7-day period; Zhao et al. 2012), the female was considered to be potentially denning. When clustering of locations occurred, a biologist navigated to the area using a handheld Global Positioning System device. Standard ground-based radio telemetry techniques with a handheld receiver (model R1000; Communication Specialists, Inc. Orange, California) and an H-type antenna were used to home towards telemetry signals of radio-collared females. The biologist then identified the individual tree or snag that appeared to be used based on the strength of the signal

(Matthews et al. 2013a). Once a possible den structure was identified, 2 to 4 automatic “den cameras” that had been cleaned and de-scented were attached to nearby trees and focused on the bole of the den structure (scent and bait lures were not applied around den trees to avoid attracting other predators). We returned to these structures the day after initial placement of den cameras, and then every 3-5 days to

confirm use for denning based on regular occupancy and images indicating the fisher was ascending and descending the tree or snag (Illustration 4). Trees and snags used ≥ 3 times in succession and with camera-based evidence of movements up and down the bole were considered denning structures (Zhao et al. 2012).

Activities of known-denning female fishers were chronicled for the duration of each denning season through continuous monitoring with cameras and ground checks of den trees.



Illustration 4. Female fisher, caught on a remote camera, moving a dependent kit between dens.

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

Information on litter size was determined from images from den cameras, or by climbing den trees and using a video camera (Peep-A-Roo Video Probe System, Sandpiper Technologies, Manteca, CA) to count kits inside den cavities (Matthews et al. 2013a). We minimized disturbance to denning females by (1) restricting visits to den structures to service cameras to once every 3-5 days, (2) deploying multiple cameras at each den to maximize the probability of a successful count, (3) by not approaching den trees for climbing until kits were at least 3 weeks old and ground-based telemetry indicated the female was well away from the den structure, (4) minimizing the time spent and disturbance made at dens (e.g. climbers attempted to not break branches while ascending or descending the den tree, ground crews remained quiet), and (5) limiting den climbing activities to occasions where remote cameras failed to capture reliable images of females with kits.

Through the use of aerial telemetry between March 15th and July 1st and subsequent monitoring of dens, we were able to determine which adult females attempted to den each year and calculate the proportion of den opportunities that resulted in den attempts. The images from den cameras were used to obtain kit counts and those results were used to calculate the proportion of den attempts that were known to be successful. If a female dropped her collar or had a collar failure at any point between March 15th and July 1st the fate of her den was marked as unknown. Females who died during the denning season were considered to have failed dens due to mortality. In the cases of female mortalities during denning season biologists would attempt to climb den trees and recover any orphan kits. We followed terminology from Green et al. (2018; Table 1) to describe the results of fisher reproduction from the SNAMP/Sugar Pine Fisher Project.

Table 1. Glossary of terminology describing fisher denning ecology, adapted from Green et al. (2018).

Reproductive female	
	A female that shows signs of reproduction, demonstrated by location of a reproductive den, anatomical changes (e.g., teat measurements), or by observing or photographing a female in the presence of kits.
Proportion of females reproducing	
	The number of adult females (≥ 2 years) showing signs of reproduction per year divided by the number of adult females monitored or assessed that year. The method used (e.g., den location, teat measurements, cameras of females with kits), and age class (juvenile, subadult, adult) should be noted.
Den opportunity	
	For an individual female fisher, each spring den season during which she is an adult (≥ 2 years) and sufficiently monitored to assess reproductive status (e.g., Powell 1993, Matthews et al. 2013a).
Den attempt	
	A den attempt is an opportunity during which a female exhibits signs of parturition by localizing to a single structure in early spring. Localizing is defined as using the same structure for at least 3 days in a 7-day period (ground telemetry) or being located with a 500m radius at least 3 times in a 7-day period (aerial telemetry).
Den success	
	A successful den occurs when a female produces and rears ≥ 1 kit over a full den season. The den season ends as females discontinue localization at individual structures for extended periods, kits begin eating solid food, and kits increase in mobility.
Den failure (early, late, or due to mortality)	
	A den failure occurs when a female stops attending a den structure where kits are thought (or known) to occur. An early den failure occurs when a female localizes to 1 structure for ≥ 2 days near the time of parturition (i.e., a den attempt), then stops attendance; as presence of live or dead kits may not be verified, there may be uncertainty about whether the female actually gave birth. A late den failure occurs when a female stops den attendance after

	approximately ≥ 3 weeks of regular den attendance. A den failure due to female mortality occurs when a female dies during the den season.
Reproductive den	
	A reproductive den is the location used by a reproductive female as a refuge to conceal herself and her kits while they are young, vulnerable, and typically still nursing during spring and early summer. Den structures generally are live trees or snags with microsites (cavities) large enough to hold a female and her kits. A reproductive den can be either a natal den (where parturition occurs) or a maternal den (structure used subsequent to the natal den).
Parturition date	
	The date when a pregnant adult female gives birth. In field studies, parturition is assumed to occur when a female first localizes to a natal den; the date is estimated based on den attendance behavior not direct observation.
Litter size	
	The number of kits produced by an individual female in a given year. To facilitate comparisons, noting the time frame of counts and methods is useful as some loss of kits may occur over the denning season. If data on kit mortality or survival in the den season are available, a late season litter size can be estimated.

Habitat Characteristics for Fisher Denning Structures and Surrounding Areas

We developed a standardized protocol (Appendix) for measuring biotic and abiotic characteristics female fishers may be selecting/using for denning habitat. The protocol was designed to collect data comparable to both the SNAMP Forest Health Project core plots in the Sugar Pine area and the concurrent USDA Forest Service PSW Kings River Fisher Project at den trees used by fishers in the High Sierra District, Sierra National Forest. Full details on how habitat data were assessed are provided in the appendix. Briefly, we used an 18m radius circular plot centered on the denning structure (Appendix Fig. A1) to organize collection of data on (1) canopy cover, (2) litter, duff, and coarse woody debris (associated abundance of fuels), (3) cover and height of herbaceous plants and understory woody shrubs (concealment cover; Illustration 5), (4) topography (slope and aspect), and (5) size, number, and height of trees and snags (3 size classes, 4 crown classes). Data on habitat characteristics for den trees were typically collected

during late spring or summer, and always when the den trees were no longer in use for denning.

Fisher Survival

Monitoring to Detect Mortality

All radio-collars fitted to fishers on the SNAMP/Sugar Pine Fisher Project were equipped with either mortality or activity sensors, internal mercury switches that change the pulse rate when the collar is stationary for over 8 hours (mortality switch) or active (activity sensor). These sensors allowed us to detect inactive signals, investigate fisher mortalities, and recover carcasses soon after death in most cases. When a mortality signal was detected, immediate efforts were made to locate the collar and recover either the carcass or the dropped radio collar, with a goal of locating the carcass within 12 to 24 hours. Decomposition begins immediately after death, which can prevent identification of underlying disease processes (Gabriel 2013, Keller et al. 2012), and scavenging can mask both the cause of death and the responsible predator (Wengert et al. 2014). Because of this, our primary rationale for consistent and frequent monitoring of radio-collared fishers by fixed-wing aircraft was to recover carcasses of animals as soon after death as possible. When an inactive signal was detected the biologist in the airplane would (1) transmit the estimated location coordinates to the research office via Forest Service radio, whereupon (2) a staff biologist would immediately be dispatched to investigate the location and recover the carcass following an approved forensic protocol (see below), (3) transport the carcass to the research office where the carcass was placed in -20°C freezer for storage until (4) a necropsy could be scheduled at the UC Davis School of Veterinary Medicine.

Determining Cause of Mortality

When fisher carcasses were discovered we followed a standardized forensic protocol for collecting samples and documenting evidence at mortality sites using photographs and diagrams (Wengert et al. 2013; Illustration 6). When predation was suspected as the cause of death (e.g. visible puncture wounds, partial consumption), we recorded information on the characteristics of the predation event including patterns of consumption and evidence of caching or burying. Samples collected included swabs of visible bite wounds, clipped fur from near the bite wounds (clipped to avoid fisher DNA in root bulbs), swabs of the claws and teeth, and any non-fisher hairs left on or near the carcass (Wengert et al. 2013). Carcasses were double-bagged in plastic bags, labeled, and transported back to the field office where they were frozen in a -20°C freezer.

We submitted all carcasses for necropsy and disease and DNA assessment to cooperating pathologists at the University of California Davis, Veterinary Medical Teaching Hospital, and California Animal Health and Food Safety Laboratory in Davis, CA. When possible, the team of pathologists determined cause of death for each fisher using all available information, including a detailed necropsy examination, disease and



Illustration 6. Carcass of a female fisher found on the Sierra National Forest, CA. Inactivity triggered a telemetry ‘mortality’ signal, which initiated recovery efforts by field personnel.

toxicological results, DNA forensic results based on submitted samples potentially containing saliva and other biological evidence of the predator, evidence recovered or identified as important from the mortality site, and habitat characteristics around the carcass. During necropsy, liver samples were collected and subsequently tested for the presence of anticoagulant rodenticide residues; liquid chromatography-tandem mass spectrometry was used to screen for presence of anticoagulant rodenticides and high-performance liquid chromatography was used to quantify positive samples (Gabriel et al. 2015). When predation was determined to be the cause of death, all lesions attributed to predation were described in detail. To distinguish between ante and post-mortem wounds (i.e. between predation and scavenging), the pathologist noted whether the lesions had associated hemorrhage and edema. Age-class at time of death was estimated as adult (≥ 24 months), subadult (12-23 months), or juvenile (< 12 months) based either on tooth wear or cementum annuli counts.

Forensic samples were processed and analyzed for predator DNA (felid or canid) following the methods of Wengert et al. (2013). Because multiple polymerase chain reaction (PCR) products were occasionally obtained when the products were visualized on an agarose gel, an appropriately-sized fragment was gel-excised (200–300bp for felids and 400 for canids) and extracted DNA using Qiagen Qiaquick Gel Extraction kit according to the manufacturer’s

instructions. The PCR products were sequenced, then aligned using RidomTraceEdit (Ridom GmbH, Würzburg, Germany). Sequences were cross-referenced on GenBank using Basic Local Alignment Search Tool (BLAST) to match them to the most closely aligned sequence to identify species of predator DNA.

Determination of Survival Rates

We monitored the status (alive, dead, or missing) of radio-collared fishers from time of first capture until death, censorship (due to dropped or failed collars on animals that were not quickly recaptured), or the end of the study. Breakaway devices in the radio-collars occasionally resulted in premature detachment, requiring efforts to re-collar animals that were missing for relatively short periods of time (1-2 months). Because it was relatively common for animals to drop collars and be missing for less than one month, we evaluated survival on monthly intervals rather than weekly or bi-weekly. Overall patterns of survival were determined using the Kaplan-Meier (KM) staggered entry method (Koen et al. 2007, Pollock et al. 1989, Price et al. 2010). KM models were used to produce estimates for annual survival and combined year survival (data pooled by month across all years). The population year was defined as April 1 to March 31 based on the timing of reproduction for female fishers in the southern Sierra Nevada, where mean parturition was March 30 (range March 17 to April 12, $n = 76$; Green et al. 2018). Because our live-trapping activities were focused on fall and winter months, first captures of young-of-the-year juveniles generally occurred from mid-October to February (a few juvenile fishers were captured before October or in early March). Kaplan-Meier models to estimate “annual” survival for juveniles were typically initiated in December, resulting in survival estimates for juveniles based on a 3-4 month period from December or January to March. When data for juveniles were pooled across population years, however, the dataset allowed for evaluating juvenile survival for the 6 month period from October to March. Z-tests were used to compare estimates for combined year survival for all possible age and sex combinations. Significance levels (α) for multiple comparisons were adjusted for Type I error rates using the Bonferroni procedure (McCann et al. 2010).

Population Structure

Occupancy Modelling

We used data from remote camera surveys across the study area to examine patterns in fisher occupancy in relation to vegetation structure and road density information. Camera data

were organized into 8-day sampling sessions (closed sampling period), with a total of 6 sampling sessions per year (open survey period). Surveys were conducted annually, starting immediately following the denning period and running into winter. Because all grid cells were not sampled every year, and some survey periods were extended due to external factors such as weather or bear disturbance, we used an effort covariate to indicate whether a particular cell was surveyed during a particular closed (8-day) sampling period. We used nine years of sampling data (2007 – 2016), and compared occupancy patterns to 11 landscape variables:

- Total canopy cover (%)
- Reineke stand density index
- Density of snags >25cm DBH (trees/ha)
- Stand density diversity
- Cover of downed wood >25 cm diameter (%)
- Volume of live trees >2.5cm dbh (m³/ha)
- Basal area weighted mean diameter of all trees
- Basal area weighted mean diameter of conifers
- Basal area weighted mean diameter of hardwoods
- Road density (m/km²)
- Off-road trail density (m/km²)

Vegetation structure variables were obtained from the Gradient Nearest Neighbor (GNN) dataset, developed and maintained by the Landscape Ecology, Modelling, Mapping, and Analysis (LEMMA) group at Oregon State University (<https://lemma.forestry.oregonstate.edu/data>). Current GNN data are based on 2012 imagery, so this analysis was not sensitive to changes in forest cover from 2013-2016, however only 5% of grid cells received treatment during that period intensive enough to influence structural variables, and of those only 5 cells both received treatment and were surveyed for fisher occupancy during that time. Road and trail density data were provided by the Bass Lake Ranger District of the Sierra National Forest. Occupancy modelling was conducted using Program PRESENCE (Hines 2006).

Population Growth Rates and Leslie-matrix Modeling

Intensive research as part of the SNAMP/Sugar Pine Fisher Project study has generated information on all key vital rates needed to evaluate the population growth rate (λ) in the area, critical for understanding whether the population has the potential to persist, or if it is in decline. We developed an age-structured matrix model to estimate a deterministic population growth rate (λ) for the SNAMP/Sugar Pine fisher population using observed data on reproduction, fecundity, and survival. We defined three age classes: juvenile (<12 months), subadult (12-24 months), and adult (>24 months) (Powell et al. 2003). Only adult females were presumed to breed (Mead 1994). Individual fertility (F_i) was calculated for adult-age female fishers as:

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

where fecundity of individual i , $b(i)$, was the mean number of female kits weaned per reproductive female (sex ratio at birth assumed = 0.5), and P_i was the age-specific survival rate (Gotelli 2001). Fertility for juveniles (F_1) and subadults (F_2) was fixed at 0. We estimated age-specific survival rates for radio-collared juvenile (P_1 : 6-11 months), subadult (P_2 : 12-23 months) and adult-age (P_3 : ≥ 24 months) female fishers in the study area using monthly encounter histories in Kaplan-Meier staggered entry model analyses (KM survival).

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

$$\begin{bmatrix} n_1(t+1) \\ n_2(t+1) \\ n_3(t+1) \\ n_4(t+1) \\ n_5(t+1) \\ n_6(t+1) \\ n_7(t+1) \\ n_8(t+1) \end{bmatrix} = \begin{bmatrix} F_1 & F_2 & F_3 & F_4 & F_4 & F_4 & F_4 & F_4 \\ P_1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & P_2 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & P_3 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & P_3 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & P_3 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & P_3 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & P_3 & 0 \end{bmatrix} \times \begin{bmatrix} n_1(t_0) \\ n_2(t_0) \\ n_3(t_0) \\ n_4(t_0) \\ n_5(t_0) \\ n_6(t_0) \\ n_7(t_0) \\ n_8(t_0) \end{bmatrix} \quad \text{Equation 2}$$

Numbers of fishers in age classes n_2 , n_3 , and n_4 for the \mathbf{n} vector at time t_0 were based on the number of radio-collared female fishers present at the start of population year 3 on April 1, 2009 ($n_2 = 5$, $n_3 = 6$, $n_4 = 5$, $n_5 = 5$), whereas n_1 was the number of juvenile females in the radio-collared population on Dec 31, 2009 ($n_1 = 4$). We multiplied the Leslie matrix by the new vector

of abundances for N_{t+1} for 20 successive years, and summed the number of individuals in each age class each year to obtain a total N , and the population growth rate (λ) for year $t+1$ was calculated as N_{t+1}/N_t . After several years a stable age distribution was achieved and λ converged to a constant value, which was the estimate of λ for the set of demographic parameters evaluated. We calculated a lower and upper range for λ based on the 95% lower and 95% upper C.I.s for age-specific survival and age-specific fertility.

Home Range Dynamics

Fisher location estimates from fixed-wing aerial telemetry flights and known locations of den and rest structures were used to define home range area (km^2) for all adult (> 2 years old) male and female fishers from the SNAMP/Sugar Pine Fisher Project. Home range estimates were limited to animals with a minimum of 30 locations, and were calculated using R Statistical Software, package *adehabithrHR* to create 95% kernel density estimates. Aerial telemetry locations were less accurate (\pm an average of 339m) than locations obtained from homing to dens and rest structures using ground based telemetry (± 15 m; Sweitzer et al. 2015). Locations within the aerial telemetry dataset that diverged greatly from the main core of location points were omitted from the home range analysis.

Fisher Response to Fuel Management

Occupancy modeling

Using the US Forest Service Forest Activities Tracking Software (FACTS) database, we compiled information on fuel management activities conducted on the Bass Lake Ranger District between 2007 and 2016 into three broad categories; harvest, timber stand improvement, and prescribed fire (Illustration 7). Insufficient acreage was impacted each year to further refine these



Illustration 7. Fuel reduction management treatments in the SNAMP/Sugar Pine Fisher Project study area. Panels show examples of commercial thinning (harvest), mastication (timber stand improvement), and prescribed fire.

categories by management activity (i.e. salvage cut vs. commercial thin). For each grid cell surveyed by remote cameras, we calculated the percent of the grid cell impacted by management activity per year to quantify the impact of treatment intensity on fisher use. We also calculated the number of years since the last management activity for each cell by year, to quantify any temporal relationship in fisher response.

The FACTS database is notoriously difficult to work with due to variations in reporting requirements over time, as well as the nature forest management activities. An individual stand, for example, may be listed multiple times in the database over 1-2 years as it was first prepped for treatment, then non-commercially thinned, slash piled, then underburned. Because each component of a fuel reduction treatment is listed separately, there is a high risk *commercial thinning, control burning, and mastication/mowing* of double counting acreage in any given year. Therefore the database was carefully screened, fuel reduction activities likely to impact fisher were identified, and references to other activities on the same acreage were removed.

These values were incorporated into the multi-season occupancy analysis as survey covariates in program PRESENCE. Because we were unable to identify exactly when, within each year, a management activity took place, we assumed that the impact of activities remained constant across a year (within and between sampling sessions), but could change between annual surveys.

RESULTS

Sample Size & Monitoring History

Trapping

A total of 143 individual fishers (78 females and 65 males) were captured as part of the SNAMP/Sugar Pine Fisher Project between December 2007 and December 2016. Of the 143 total captured fishers, 139 were fitted with radio collars. Three juvenile fishers were not collared because they were too small (< 2.0 kg) and one fisher was not collared because it was caught at the end of the study in December 2016.

Radio-collared fishers were located as often as possible through the use of ground based and aerial telemetry, and their movements were monitored. Any lapse in the knowledge of a fisher's location was caused either by the mortality of that individual, a failure in the radio collar,

a dropped collar, or in some cases the fisher had moved so far from its original home range that it was undetectable by aerial telemetry.

Throughout the initial phase of the project, the objective was to consistently maintain at least 20 collared fishers in order to gather sufficient data on the health and stability of the regional population. Following the transition to the Sugar Pine Project, the research focus shifted to documenting individual fisher response to vegetation management resulting in a reduction in the overall study area. During this transition, trapping efforts were continued across the larger study area to remove radio collars from animals no longer deemed necessary to monitor. Once that effort was concluded, both the number of traps set and number of animals monitored declined accordingly (Table 2, Figs. 5, 6).

Table 2. Summary of fisher trapping and monitoring per population year from the SNAMP/Sugar Pine Fisher Project, December 2007 through February 2017. Fisher population years begin April 1st and end March 31st of the following year.

Study Year	Population Year	# Fishers Captured (new/recapture)	# Collared Fishers ^a	Mortalities	# Failed or Dropped Collars	Year End # Collared Fishers
1	2007-08	10 / 0	10	3	0	7
2	2008-09	34 / 6	41	8	2	31
3	2009-10	20 / 18	51	10	5	36
4	2010-11	18 / 16	54	13	6	35
5	2011-12	21 / 18	58	8	5	45
6	2012-13	8 / 11	53	11	7	35
7	2013-14	2 / 12	37	10	13	14
8	2014-15	19 ^b / 15	33	2	9	22
9	2015-16	8 / 11	27	6	7	14
10	2016-17	1 / 2 ^c	16	4	12	0

a - Numbered of collared fishers monitored for ≥ 1 day.

b - Three fishers captured but not collared due to small size.

c - New individuals and recaptured fishers in 2016-17 were either not collared or collars were removed because study was ending.

Telemetry results

Aerial telemetry was a large and unique part of the SNAMP/Sugar Pine Fisher Project. Technicians began locating radio-collared fishers in December of 2007 and depended on these aerial missions until the last flight in November 2016. A total of 37,728 fisher locations were collected during 1,998 aerial telemetry flights. The average number of locations per fisher was 273 (range = 1 – 1,373). Forty-seven technicians were trained to gather fisher locations by aerial

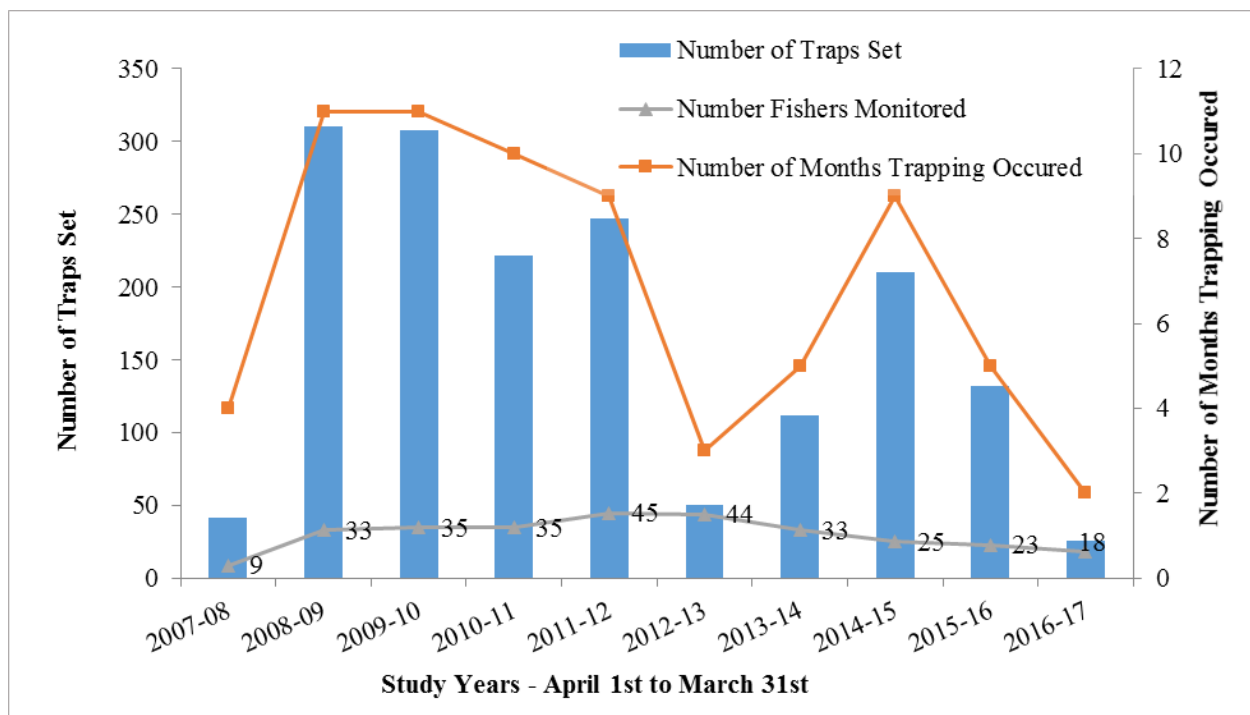


Figure 5. Summary of SNAMP/Sugar Pine trapping effort, including the number of months each year that trapping occurred, the number of traps set, and the maximum number of fishers collared and monitored per study year.

telemetry methods, and 737 test collars were located so that technicians could track their error rates and improve their telemetry skills.

Remote camera survey results

Camera surveys were a major aspect of the SNAMP/Sugar Pine Fisher Project in all nine years of the study (Illustration 8). Between November 2007 and November 2016, camera stations were placed in 794 unique 1-km² grid cells, 133 of which were located within the four key watersheds focal area of the study (Fig. 14). One hundred and twenty-eight of the key watershed grid cells are at least 50% USDA Forest Service ownership, and were surveyed in all nine survey years. Although other camera work was done in collaboration with California Department of Fish and Wildlife and Yosemite National Park, the focus of this report is on the 794 grid cells that were surveyed as part of this study.

Fishers were detected in 403 of the 794 unique 1-km² grid cells, accounting for a 51% detection rate across all of the surveyed grid cells. Within the key watersheds focal area of the study, 116 grid cells were positive for fisher detections (87%), while the other 287 fisher detections were located in grid cells outside of our core study area. Accounting for all 794 1-km² camera grid cells, all camera checks, and years of the study, fishers were detected ~ 2,003 times.

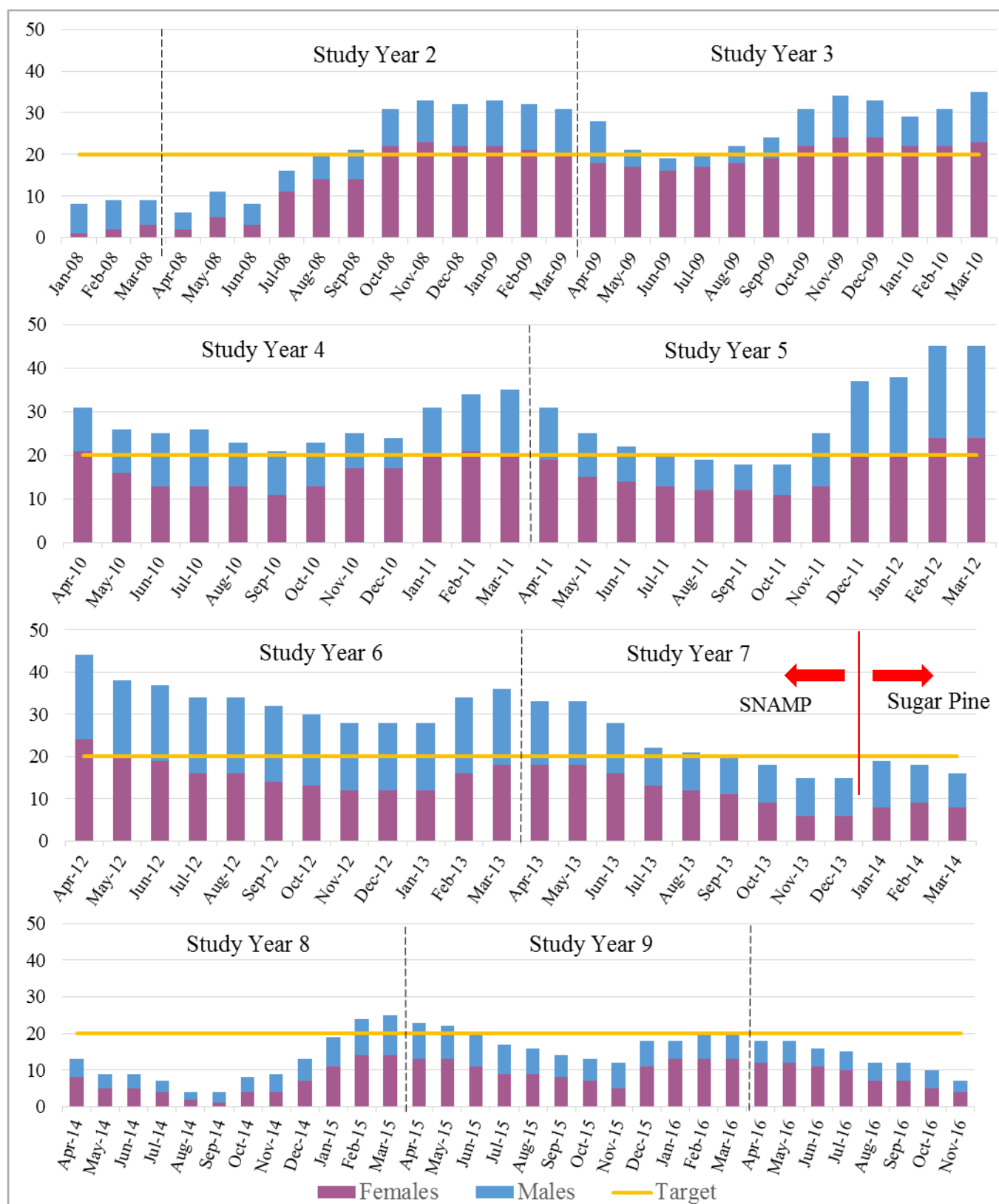


Figure 6. Number of radio-collared male and female fishers monitored per month, based on aerial telemetry detections from the SNAMP/Sugar Pine Fisher Project. The target number of fishers to be monitored was 20.

Fisher Reproduction

Reproductive Rates

During the nine spring denning seasons studied, a total of 79 female fishers were known to have denned based on spatially clustered aerial telemetry locations confirmed with images from den cameras (Table 3). On average females were located in their natal dens by the third week in March. Den cameras were deployed as soon as a female's telemetry locations indicated she might be denning.

The latest observation of kits being moved by the mother was July 5th, although cameras revealed kits climbing down the den tree on their own by the middle of June (Table 3, Illustration 9). After mid-June female fishers began to show a more diverse pattern of movements, but it is likely that kits continued to rely to a large extent on the female for food and travel with her through August (Green et al. 2018).

During the course of nine denning seasons between 2008 and 2016, there were 104 denning opportunities (Table 4) of which 81 denning attempts were identified. We determined that 69% ($n = 56$) of the denning attempts were successful, resulting in the female producing one or more kits and rearing them for at least 10 weeks or to the end of denning season. A total of 80 individual fisher kits were counted in 62 litters, with an average of 1.5 kits per litter. Ten female fishers died during the denning season between 2008 and 2016; eight deaths were attributed to predation, one female was hit by a vehicle but also tested positive for high levels of anticoagulant rodenticides, and the results of one case are still pending. One of the



Illustration 8. Two radio-collared fishers detected via remote camera monitoring on the Sierra National Forest, CA. The reflective bands on the tree trunk allowed for estimation of body length.



Illustration 9. Female fisher carrying a kit down the bole of a den tree on the Sierra National Forest, CA.

Table 3. Summary of data on denning female fishers from the SNAMP/Sugar Pine Fisher Project between March 2008 and June 2016.

Denning Season	# Denning Females	# Den Trees	Earliest Den Detection ^a	Latest Observed Kit Movement ^b
2008	1	3	9-Apr-08	4-Jun-08
2009	12	31	27-Mar-09	7-Jun-09
2010	13	34	24-Mar-10	9-Jun-10
2011	8	16	31-Mar-11	5-Jul-11
2012	11	27	29-Mar-12	20-Jun-12
2013	12	24	23-Mar-13	6-Jun-13
2014	6	14	19-Mar-14	1-May-14
2015	8	18	21-Mar-15	29-May-15
2016	8	22	27-Mar-16	20-Jun-16
Totals	79	189	Mean = 26 Mar	Mean = 7 Jun
a -Earliest detection of a natal den based on spatially clustered telemetry locations and confirmed with den camera images.				
b - Latest date kits were observed on den camera images.				

predation cases also tested positive for an active canine distemper infection, and two others had trace amounts of anticoagulant rodenticides in their livers. For eight of the ten deaths of denning females, the locations of den trees were known and were being monitored. In one case den camera images included a bobcat with a kit in its mouth, and the partial carcass of the denning female was recovered nearby. In two cases the den structure was a large, unstable snag, and we did not attempt to climb the tree to determine litter size due to safety considerations. In each of the other five cases we climbed the den tree to attempt to recover orphaned kits in accordance with California Department of Fish and Wildlife policy. Two deceased kits were found in one den cavity and we failed to find kits in another. From the remaining three dens, a total of seven live kits were recovered; two litters were from the 2010 denning season (litter sizes 2, 3), and the other litter was from the 2015 denning season (litter size 2).

The seven orphan kits rescued were raised in captivity by the Fresno Wildlife Rescue and Rehabilitation Service approved by the California Department of Fish and Wildlife, and under the care and supervision of the Fresno-Chaffee Zoo chief veterinarian. In 2010, one of the orphan kits died in captivity due to urinary tract blockage attributed to a parasitic nematode, whereas the

other four survived captive rearing. Two kits from one litter were released within their mother's home range, and the two kits from the second litter were released into an area with suitable fisher habitat in the Kaiser Wilderness, close to the southern margin of the study area. Both orphan kits from the 2015 litter survived captive rearing and were later released in Yosemite National Park.

We were unable to adequately monitor 11 breeding-age females known to have denned, but whose premature collar loss prevented us from determining the fate of their dens. In seven of these 11 cases, kit counts had been confirmed prior to the females losing their collars. Another five dens were assumed to have failed when females that had initiated denning, ceased to localize to den trees, potentially related to the death of kits.

We tested differences in weights of female fishers (ascertained at capture prior to the denning season) relative to denning status and litter size. There were no statistically significant different differences between females that denned and those that did not, or between females with one kit versus those with two kits (Fig. 7).

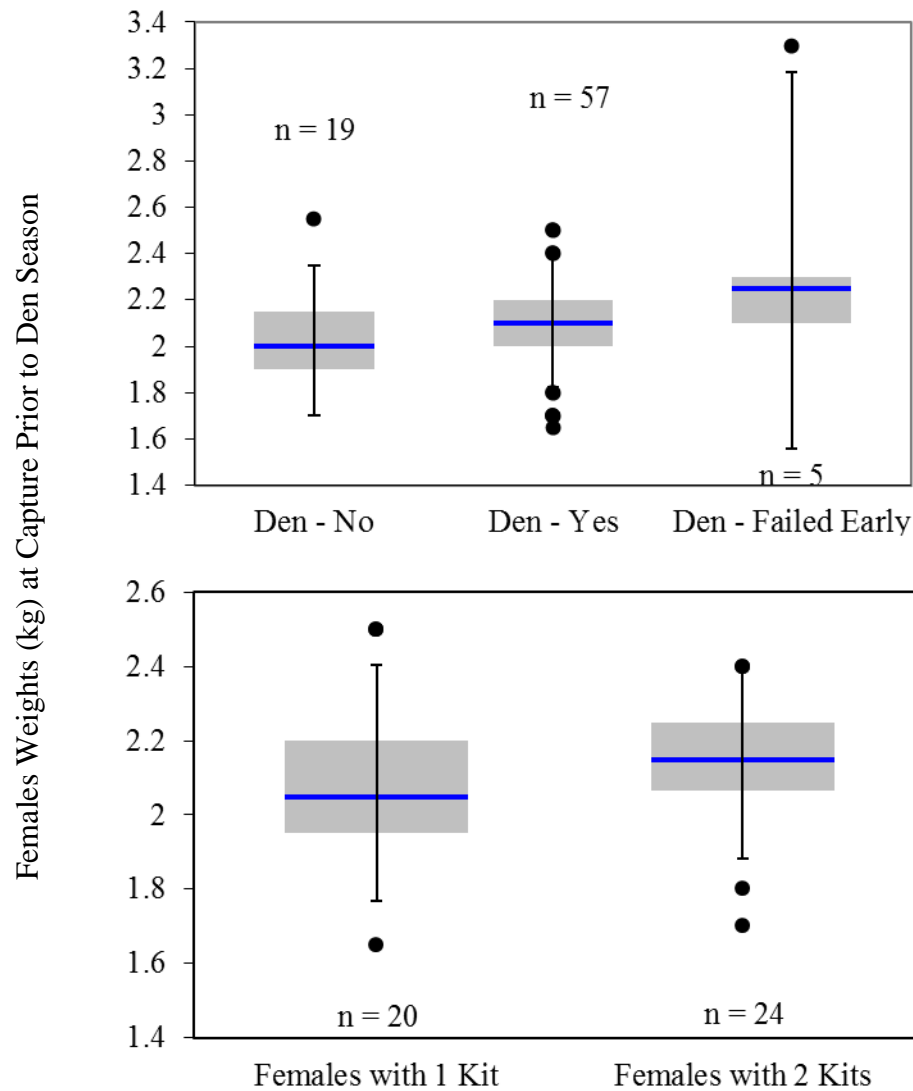


Figure 7. Weights of adult and subadult female fishers as a function of (a) denning status and (b) litter size. The top panel (a) shows that weights of females that denned and those that didn't did not differ significantly. The bottom panel (b) shows that weights of females who had one kit did not differ compared to those with two kits.

Table 4. Results of fisher den monitoring on the SNAMP/Sugar Pine Fisher Project across 9 reproductive seasons (2008 – 2016). Standard deviation was generated from annual data. See Table 1 for standardized terminology definitions.

Metric Association with Fisher Reproduction	2008	2009	2010	2011	2012	2013	2014	2015	2016	Total	Mean (\pm SD)
Denning opportunities	1	15	16	13	15	14	8	11	11	104	11.6 (\pm 4.7)
Denning attempts	1	12	15	8	11	12	6	8	8	81	9.00 (\pm 4.1)
Denning successes	0	10	10	5	10	7	4	5	5	56*	6.22 (\pm 3.4)
Early den failure (within 3 weeks of den initiation)	0	1	1	2	0	1	0	0	0	5	0.60 (\pm 0.7)
Den failure due to female mortality	0	1	3	1	1	2	0	1	1	10	1.11 (\pm 0.9)
Unknown fate of den due to collar loss	1	0	1	1	0	2	2	2	2	11	1.22 (\pm 0.8)
Proportion of adult females attempting to den	1	0.8	0.93	0.62	0.73	0.86	0.75	0.73	0.73	81/104	0.78 (\pm 0.1)
Proportion of adults females with known successful dens	0	0.67	0.63	0.38	0.67	0.5	0.5	0.45	0.45	56/104	0.54 (\pm 0.2)
Parturition date (mean)	4/9	4/5	4/1	4/10	4/5	3/31	3/24	3/26	4/1	-	4/2
Parturition date (\pm SD in days)	-	\pm 6	\pm 6	\pm 8	\pm 5	\pm 7	\pm 5	\pm 4	\pm 4	-	\pm 6
Fisher kits counted	1	14	14	10	16	10	6	8	14	93	-
Litters counted	1	8	9	7	10	8	3	8	7	61*	-
Litter size produced (mean)	1	1.75	1.55	1.5	1.42	1.25	2	1	2	-	1.50 (\pm 0.4)

* Fifty six females were known to have produced and reared ≥ 1 kit over a full denning season (denning success), however another five females denned and produced kits but died before their kits were reared, attributing the higher number of litters counted.

Denning Structures and Denning Habitat Characteristics

Between the denning seasons of 2008 and 2016, 182 trees were identified as natal or maternal dens, including 61 incense cedars, 55 white firs, 53 California black oaks, 7 sugar pines, 4 ponderosa pines, one canyon live oak and one white alder. Twenty-one den trees were reused, either by the same female or by a different female (12%), similar to den reuse rates reported on the nearby Kings River Fisher Project (12.6% reuse of natal dens / 14% reuse of maternal dens; Green et al. 2018). In three cases females used den trees that had been used by a different individual in different denning season. One female used the same live incense cedar (DBH = 140 cm) as her natal den for three consecutive denning seasons, then as her first maternal den in the fourth denning season. Another female reused 5 of her previous den trees, choosing to return to those specific trees during the same time and in the same order, across different denning season. Yet a third female used a tree as her natal den and returned to that same tree later in the denning season to use as her third maternal den.



Illustration 10. *California black oak used for denning in spring 2011 from a distance (left), and view of the fork where the den entrance was located (right).*

Female fishers monitored from den initiation through the end of June ($n = 57$) used an average of 2.4 den trees per season (range 1-5). Successive dens of females that used more than one den structure were located on average 383 m apart ($n = 84$, range 11-1487 m). Fifty-five percent of the unique trees used for denning in the SNAMP/Sugar Pine area were live trees ($n = 88$), whereas 45% were snags ($n = 73$). California black oak was the most common live tree used for denning (Illustration 10), followed by incense cedar and white fir (Table 5). Among snags

used as denning structures, white fir and incense cedar were both commonly used, whereas use of black oak was less common (Table 5). Hardwoods used as den trees averaged 20m in height (SD = 6.1) and 75.1 cm DBH (SD = 18.4), while conifers averaged 28.2 m in height (SD = 10.4) and 111.7 cm DBH (SD = 29.4) (Fig. 8).

Table 5. Number, mean DBH, and mean height of live and dead trees used by female fishers during denning seasons between 2008 and 2016 in the SNAMP/Sugar Pine Fisher Project. Some trees were used more than once by the same fisher or by other fishers (reuse den structures) and were only counted once in this analysis.

Tree Species	Live Trees			Dead Trees		
Unique Den Structures	n	Mean DBH (cm)	Mean Height (m)	n	Mean DBH (cm)	Mean Height (m)
Black Oak	41	75.2	20.0	8	75.9	18.1
Incense cedar	24	134.6	33.7	29	106.7	16.8
White Fir	15	101.5	36.3	32	102.8	26.5
Pines	6	105.4	33.7	4	100.6	23.3
White alder	1	84.0	20.7	0	0	0
Canyon live Oak	1	61.0	12.6	0	0	0
Reuse Den Structures	n	Mean DBH (cm)	Mean Height (m)	n	Mean DBH (cm)	Mean Height (m)
Black Oak	4	81.6	21.9	0	0	0
Incense cedar	4	135.7	33.1	4	97.6	13.2
White Fir	2	100.5	34.8	6	84.8	25.1
Pines	1	156.0	44.6	0	0	0

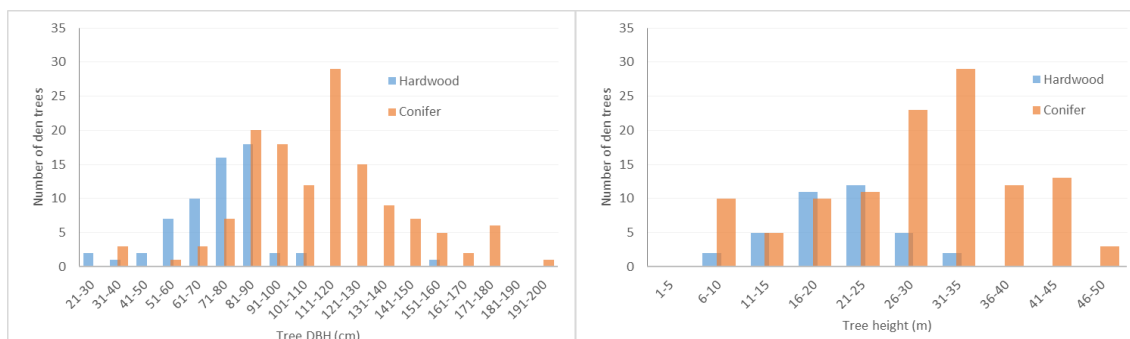


Figure 8. Distributions of den tree diameter (left) and height (right), on the Bass Lake Ranger District of the Sierra National Forest, showing differences in DBH and height of hardwoods and conifers.

The majority of denning structures (85%) used by fishers in the SNAMP/Sugar Pine study area were in the elevation range between 1370 and 1830 m (4500 - 6000 ft.; Fig. 9). Circular habitat plot assessments were completed for 127 den trees (70% of the total 182 den trees). Summary of den tree habitat data indicated the use of areas with high canopy cover, limited herbaceous cover and relatively low shrub cover (Table 6). Concealment cover was 72% for low ground cover (0-0.3 m), 47% for high ground cover (0.3-1 m), 58% for low shrub (1-2 m), and 36% for high shrub cover (2-3 m). On average, belt transects within the circular habitat plots around den trees included an average of 6.5 down logs (coarse woody debris; logs/branches with a minimum large end diameter of 15cm, ≥ 1 m total length). Many denning structures were on steep slopes (Table 6) with a slight preference for the northerly and easterly aspects (Fig. 10).

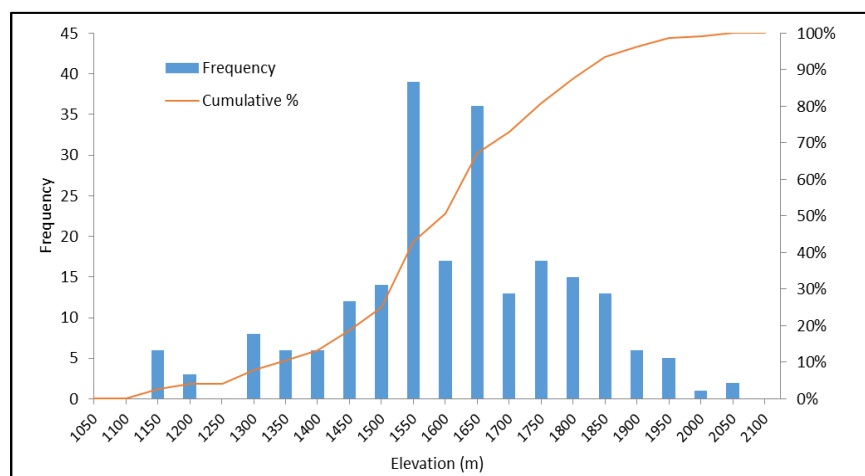


Figure 9. Distribution of den trees across an elevational gradient in the Bass Lake Ranger District of the Sierra National Forest, CA.

Table 6. Basic habitat attributes around fisher den trees for the SNAMP/Sugar Pine Fisher Project from spring 2008 to spring 2016.

Attribute ^a	Mean	Range
Canopy cover	74%	30-98%
Shrub cover	19%	0-83%
Herbaceous cover	6%	0-29%
Prevailing slope	32%	1-75%
^a Habitat attributes are from circular plots (18m radius) centered on fisher den trees (n = 127)		

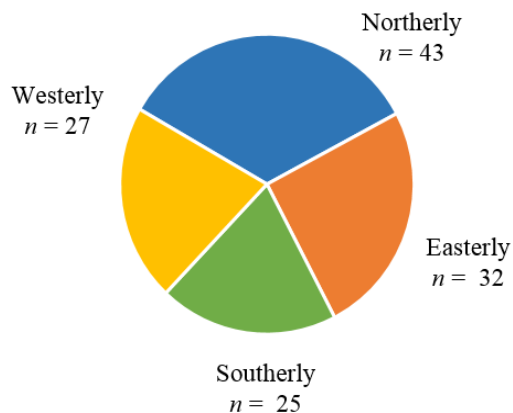


Figure 10. Aspect of fisher den structures (n = 127) from the SNAMP/Sugar Pine Fisher Project.

Fisher Survival

Kaplan-Meier survival rates showed that overall survival was the highest for adult females (76%) and lowest for juvenile males (40%; Table 7). All adults had a higher survival rate (71%), than subadults (64%), and juveniles had the lowest survival rate (55%). Female fishers, across all age groups, exhibited a consistently higher survival rate

than did males (Table 7). On an annual basis, subadults and juveniles were advanced to the next age category at the beginning of the fisher population year on April first. Because too few new subadults and juveniles were added to the population each year, we were unable to determine annual survival rates for these age groups. Annual variability for adults was high. Male survival rates ranged from a low of 0.49 in 2015-16 to a high of 1.0 in 2014-15, while females survival rates ranged from 0.61 during 2010-11 to 0.91 in 2014-15 (Table 8). Despite the fact that both

Table 7. Survival rates (s(t)) and 95% confidence intervals for all fisher age and sex combinations from the SNAMP/Sugar Pine Fisher Project.

Age/Sex	s(t)	95% CI
All adult	0.71	0.65 – 0.77
Adult female	0.76	0.67 – 0.84
Adult male	0.66	0.57 – 0.75
All subadult	0.64	0.53 – 0.74
Subadult female	0.72	0.58 – 0.85
Subadult male	0.50	0.35 – 0.65
All juvenile	0.55	0.45 – 0.67
Juvenile female	0.71	0.57 – 0.85
Juvenile male	0.40	0.24 – 0.57

Table 8. Annual Kaplan Meier survival rates (s(t)) and 95% confidence intervals for adult male and female fishers from the SNAMP/Sugar Pine Fisher Project.

Study Year	Adult female		Adult Male	
	s(t)	95% CI	s(t)	95% CI
2008-09	0.77	0.56 - 0.97	0.5	0.22 – 0.78
2009-10	0.86	0.67 – 1.04	0.75	0.45 – 1.05
2010-11	0.61	0.37 – 0.85	0.88	0.63 – 1.12
2011-12	0.9	0.73 – 1.07	0.6	0.39 – 0.82
2012-13	0.83	0.62 – 1.05	0.71	0.51 – 0.91
2013-14	0.61	0.28 – 0.95	0.65	0.35 – 0.96
2014-15	0.91	0.74 – 1.07	1	1.00 – 1.10
2015-16	0.9	0.71 – 1.09	0.49	0.10 – 0.89
2016-17	0.76	0.39 – 1.12	0.75	0.33 – 1.17

male and female survival rates peaked in 2014-15, survival rates of the sexes were not correlated (Pearson's correlation coefficient = -0.097).

Kaplan-Meier plots of variation in survival throughout the fisher year (April 1 through March 31 of the following year) indicated that adult and subadult female survival was relatively consistent across all seasons (Figs. 11a and c). Adult male survival was relatively consistent across 10 months of the year, but dropped at the onset of the breeding season in February and March (Fig. 11b). Increased male mortality during this time period is likely associated with their extensive movements and explorations in search of breeding females. Subadult male survival dropped steeply between April and October, but stabilized during the fall and winter (Fig. 11d). The stabilization of subadult male survival rates likely corresponds with their physical development and reaching adult stature but being a subadult male appears to entail risks not faced by subadult females. Both male and female subadult survival was more variable than the adult rates, as evidenced by wider confidence intervals (Fig. 11). Data on juvenile survival are limited but results suggest that male juvenile survival is low, especially compared to that of juvenile females (Figs. 11 e and f).

Causes of Mortality

Seventy-five monitored fishers died between December 2007 and October 2016. These deaths include 37 female and 38 male fishers, representing 52% of the total number of fishers collared during these studies (Fig. 12). An average of eight fishers died per year, excluding the first few months of the study in population year 2007-08. This number is a slight decrease from the previous report by Sweitzer et al. (2015) who reported an average of 10.5 deaths per fisher population year but was also reporting on the initial phase of the project when more animals were being monitored each year. Proportionally, the mortality rate of collared fishers ranged from 0.06 in 2015-16, when only 2 of 33 collared fishers died, to 0.27 in 2013-14, when 10 of 37 collared fishers died. Though the greatest number of fishers died in 2010-11 ($n = 13$), 54 total fishers were collared and monitored for some part of that year, resulting in a 0.24 mortality rate.

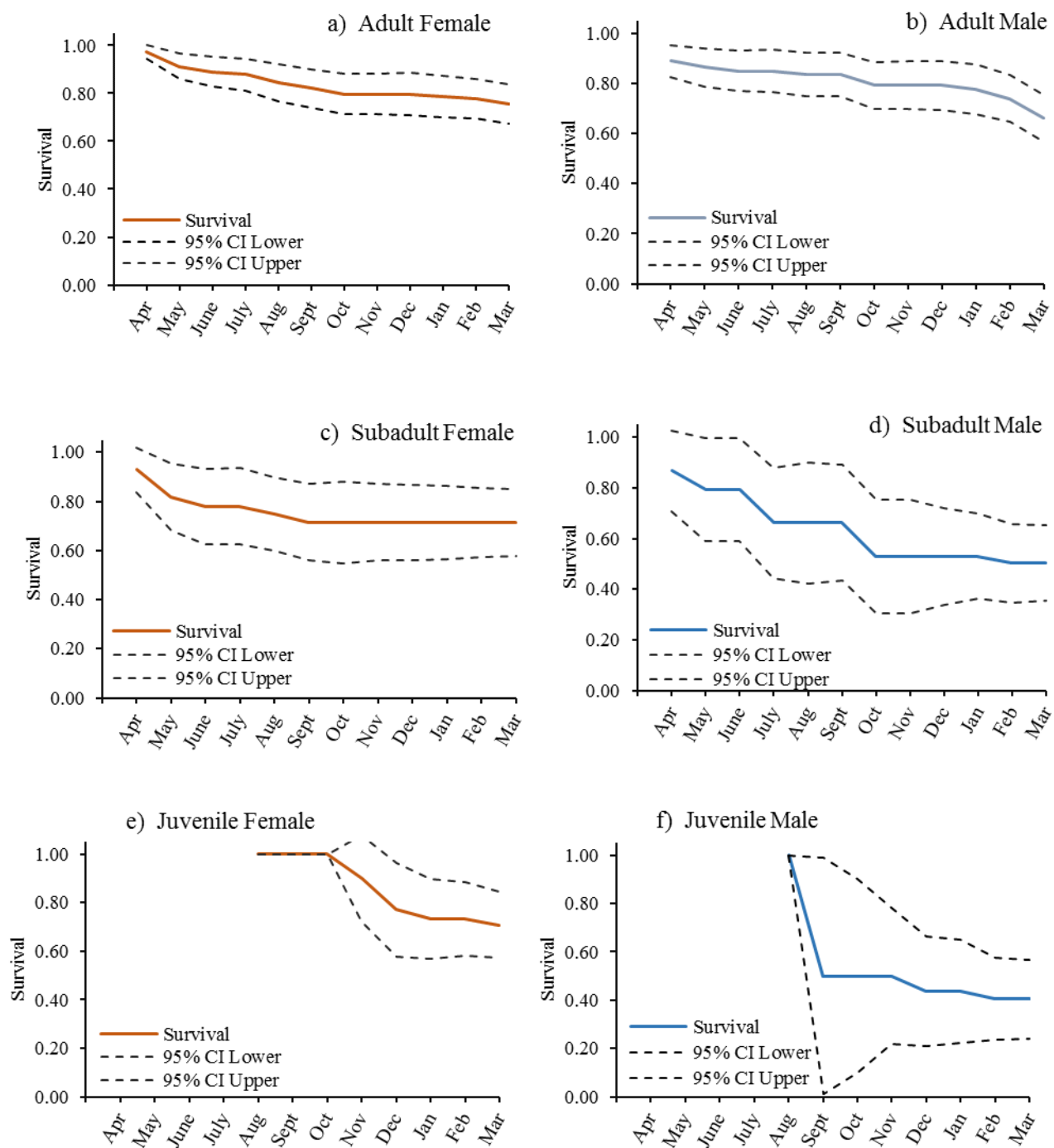


Figure 11. Survival curves showing the monthly Kaplan-Meier survival rates for a) adult female, b) adult male, c) subadult female, d) subadult male, e) juvenile female, and f) juvenile male fishers, across all years from the SNAMP/Sugar Pine Fisher Project. Data on juveniles early in the year are lacking as they were captured between July and March.

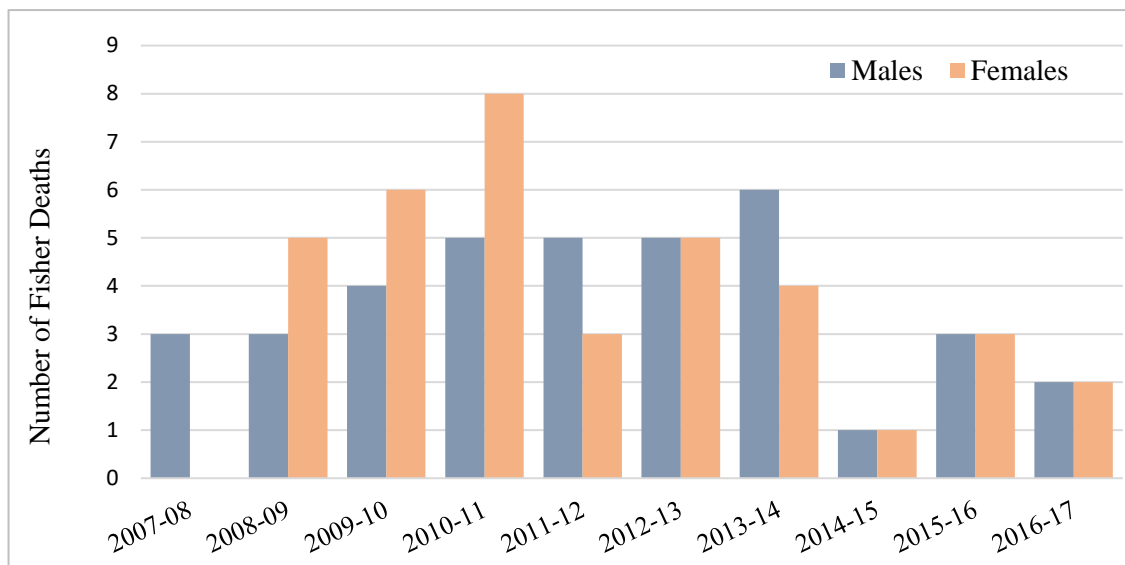


Figure 12. Number of known male and female fisher mortalities between December 2007 and February 2017 from the SNAMP/Sugar Pine Fisher Projects.

Of the 75 mortalities collected, in 52 cases we collected sufficient remains to request a necropsy. Of those submitted for necropsy, 44 provided sufficient evidence to determine the cause of death, though the level of detail varied according to the condition of the remains. The remaining 8 were characterized as ‘unknown cause of mortality.’ Documented primary causes of death included predation, disease, injury/infection, poisoning, and vehicular strike.

Predation was the leading cause of mortality and accounted for 27 of the 44 mortalities for which necropsy evidence was available (61%) (Fig. 13). Of these 27 confirmed predations, 22 were genetically identified to

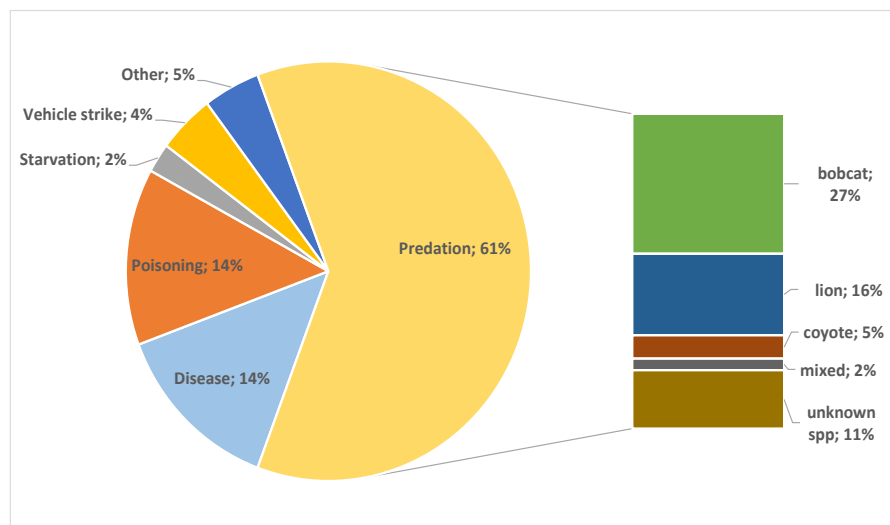


Figure 13. Summary of mortality data showing percentage of known causes of mortalities based on necropsy results from the SNAMP/Sugar Pine Fisher Project between December 2007 and December 2016.

predator class (20 felid and 2 canid). Of those 22, 21 could be further identified to species: 12 were confirmed as bobcat predation, 7 as lion predation, and 2 as coyote predation. One mortality tested positive for both bobcat and lion DNA, and we were unable to determine which species was responsible for the mortality. The remaining 5 mortalities were confirmed as predation based on gross necropsy results, however genetic testing was unable to determine the predator. In addition, predation contributed to one additional mortality where a female appeared to escape a predation attempt, however a canine puncture in her shoulder became infected and she subsequently died from bacterial septicemia. This mortality was officially classified as disease, however the failed predation and associated injury were the ultimate cause. Bobcats preyed only on female fishers; no males were known to have died from bobcat predation. Mountain lions, on the other hand, preyed primarily upon male fishers, accounting for six male fisher deaths and only one female fisher death (Illustration 11).



Illustration 11. Mountain lions and bobcats accounted for the majority of fisher mortalities due to predation on the Sierra National Forest, CA. Bobcats killed exclusively female fishers, while lions killed primarily males.

Of the 44 carcasses for which necropsy evidence was available, anticoagulant rodenticide (AR) poisoning was determined to be the primary cause of death for 6 fishers; 5 male and 1 female (14%). Further analysis of AR exposure was possible for 39 of the 44 necropsied animals, where sufficient liver tissue remained to facilitate testing for specific compounds as well as quantify the amount present. This included a combination of 36 collared animals and an additional 3 uncollared fishers recovered by Yosemite National Park staff. Thirty-six of 39 fishers (92%) showed evidence of exposure to between one and four AR compounds (Table 9). The most common compound detected was brodifacoum, with quantification rates as high as 3.4 ppm. Other compounds detected included bromadiolone, difethialone, chlorophacinone, and diphacinone. It is worth noting that the three unexposed fishers were all females captured early in the project. No animals captured after October 2009 tested negative for AR exposure.

Table 9. Anticoagulant rodenticide exposure rates for fishers collected during the SNAMP/Sugar Pine Fisher Project on the Sierra National Forest, CA, 2007-2016. Quantification rates are in parts per million (ppm), (+) indicates the presence of that compound at levels too low for quantification. Compounds include brodifacoum (brd), bromadiolone (brm), difethialone (dif), chlorophacinone (chl), and diphacinone (dip). * denotes fishers that died of anticoagulant rodenticide exposure.

Fisher ID	SEX	Cause of mortality	AR exposure (+) or (-)	# of AR	BRD	BRM	DIF	CHL	DIP
F01	F	infection	-	0					
F09	F	predation	-	0					
F10	F	starvation	-	0					
F14	F	predation	+	1					+
F19	F	predation	+	1	+				
F20	F	predation	+	1	0.01				
F24	F	predation	+	1	0.10				
F25 *	F	AR exposure	+	1	0.02				
F28	F	predation	+	1	+				
F35	F	unknown	+	1	+				
F44	F	predation	+	1	0.02				
F48	F	predation	+	1	+				
F56	F	infection	+	1	+				
FK-2 CHO	F	unknown	+	2	0.05				+
M01	M	predation	+	2	0.13	0.20			
M02	M	vehicle	+	2	0.03				+
M03	M	unknown	+	2	0.01	+			
M04	M	vehicle	+	2	1.30	0.10			
M05	M	disease	+	2	+	0.10			
M06	M	disease	+	2	0.13	+			
M07 *	M	AR exposure	+	2	0.10	+			
M13 *	M	AR exposure	+	2	+		+		
M14	M	disease	+	2	+	+			
M18	M	old age	+	2	3.4	0.24			
M19	M	predation	+	2	0.65	0.54			
M20	M	unknown	+	2	0.32				+
M21	M	predation	+	2	0.46	0.19			
M22	M	unknown	+	2	0.10	+			
M23	M	predation	+	2	0.14	+			
M24	M	unknown	+	2	0.25	+			
M28	M	predation	+	3	0.34	+			0.48
M40 *	M	AR exposure	+	2	0.53	0.10			
M41	M	unknown	+	3	+		+		+
M42 *	M	AR exposure	+	2	0.42	0.06			
M44	M	predation	+	3	0.08	0.33			+
M45 *	M	AR exposure	+	3	0.17	0.07			+
YOS 1	F	vehicle	+	3	0.60	0.17			+
YOS 2	M	vehicle	+	3	0.38	0.11		+	
YOS 3	F	vehicle	+	4	1.20	+		+	+

Disease, including both viral and bacterial infection, was the direct cause of death for 6 fishers (4 male, 2 female) and likely contributed to the death of two additional female fishers. Two male fishers died of canine distemper in April 2009, likely victims of a local CDV outbreak (Keller et al. 2012). A female also died during the same CDV outbreak in 2009; while the direct cause of death was predation, she was actively infected with CDV which likely contributed to her death. A third male also died in 2009 during capture and handling, however the cause of death was respiratory depression caused by severe CDV infection exacerbated by anesthesia during handling. Two additional fishers died of infection by *Toxoplasma gondii*, one male in 2008 and one female in 2011. One female died in 2012 of bacterial infection of injuries sustained while escaping predation. One female fisher died of starvation in 2009, however she was actively infected with *Toxoplasma gondii* at the time, which likely contributed to her death. Additional animals tested positive for these and other infectious pathogens, though infection was not associated with mortality. Based on blood samples tested for 112 fishers, population-level exposure rates included canine distemper (9% positive), canine parvovirus (7% exposure), canine adenovirus (7% exposure), *Toxoplasma gondii* (68% exposure), canine herpesvirus (3%).

Population Rates

Occupancy Rates

Annual survey efforts ranged from 172 to 420 grid cells surveys each year (Table 10; Fig. 14). Naïve occupancy rates throughout the study ranged from 0.24 to 0.66 (Table 10). Average

Table 10. Summary of yearly data from 1-km² camera grid cells, fisher detections within cells, and naïve occupancy rates from the SNAMP/Sugar Pine Fisher Project between November 2008 and November 2016.

Study Year	1	2	3	4	5	6	7	8	9
	2007-08	'08-09	'09-10	'10-11	'11-12	'12-13	'13-14	'14-15	'15-16
Grid cells surveyed	233	265	341	420	354	147	188	178	172
Grid cells with detection	55	148	158	165	196	97	90	92	77
Naïve occupancy	0.24	0.56	0.46	0.39	0.55	0.66	0.48	0.52	0.45
Total # of detections	55	250	255	301	375	231	190	186	160

occupancy across the study area was 0.457 (95% CI: 0.438 – 0.475, median = 0.453, SD = 0.174, range = 0.082 to 0.982). GNN data for the nine habitat covariates we considered were available for only 361 of our 1-km² grid cells, therefore we based estimates of the influence of vegetation structure on occupancy and colonization rates on observations of fisher activity in these 361 cells.

The most strongly supported model to explain occupancy patterns relied on stand density and mean hardwood dbh as covariates on occupancy, canopy cover as a covariate explaining colonization, and included annual variation in colonization rates (Table 11). This model indicated that fishers were more likely to occupy grid cells with higher stand density and greater hardwood basal area. The probability of colonization, meaning the likelihood of a fisher being found in a cell if the cell was unoccupied the previous year, was greater in cells with higher canopy cover, and colonization rates appeared to vary year by year while extinction rates, the probability that a cell would be unoccupied if it had been occupied the previous year, remained constant across years. This model was highly supported, with an Akaike weight of 1.0. The next most supported model, identical to the top model but without the canopy cover covariate to explain colonization rates, resulted in a ΔAIC value of 34.92, also indicating strong support for the top model. Other variables such as stand diversity, presence of large snags, and road density appeared unrelated to occupancy rates.

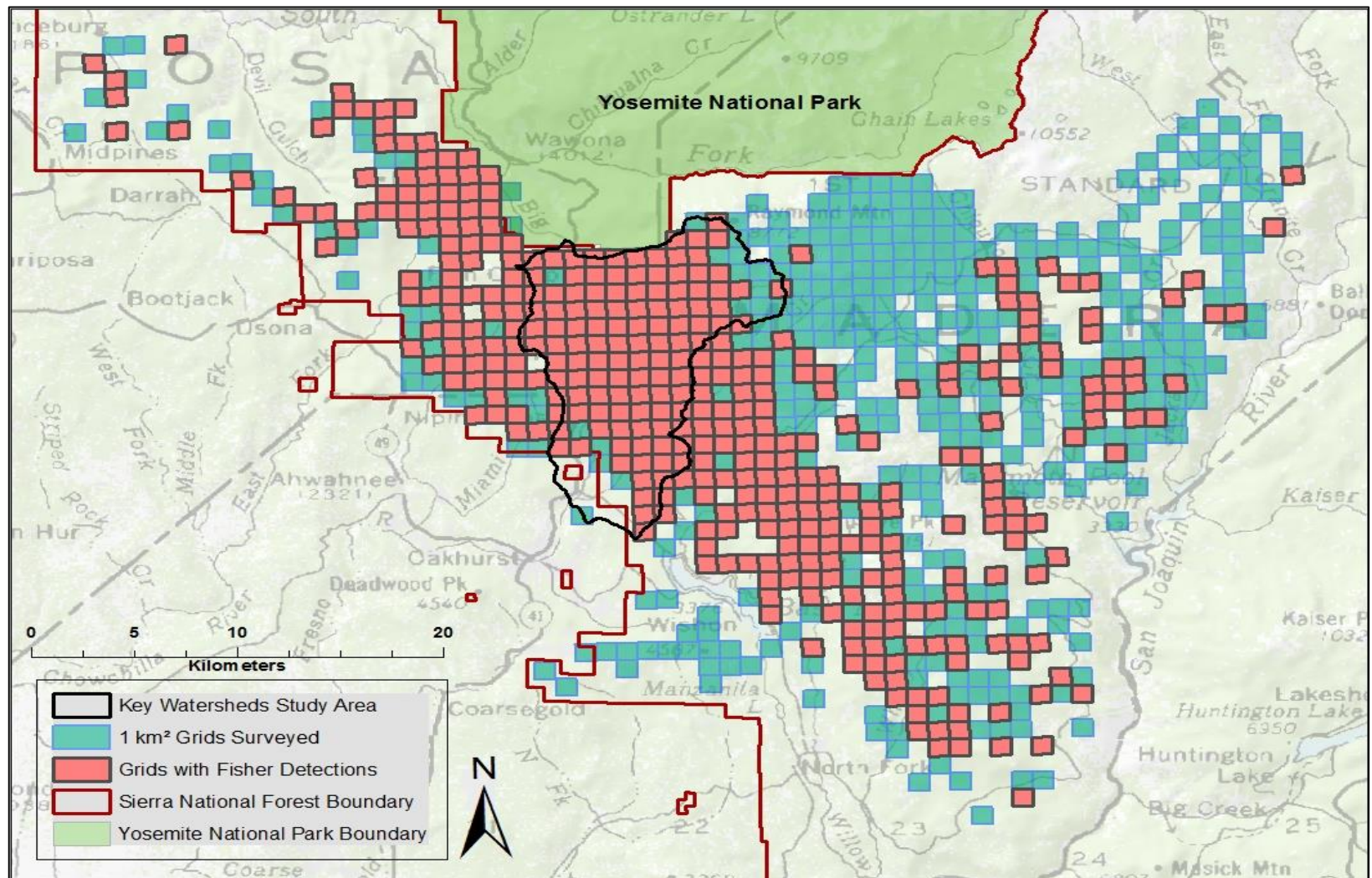


Figure 14. Map of 1km grid cells surveyed using camera surveys as part of the SNAMP/Sugar Pine Fisher Project. Orange squares represent the grid cells where fishers were detected.

Table 11. Summary of occupancy modeling results from the SNAMP/Sugar Pine Fisher Project. Symbols represent occupancy (Ψ), colonization rate (Υ), extinction rate (ϕ), and probability of detection (ρ). SDI is the Reineke stand density index, HDW is basal area based on the weighted mean diameter of hardwoods, can_cov is total canopy cover (%), and annual represents annual variation.

Model structure	# Parameters	AIC	Δ AIC	AIC weight
Ψ (SDI + HDW), Υ (annual + can_cov), ϕ (.), ρ (.)	14	7982.04	0.00	1.0
Ψ (SDI + HDW), Υ (annual), ϕ (.), ρ (.)	13	8016.96	34.92	0.0
Ψ (can_cov + HDW), Υ (annual), ϕ (.), ρ (.)	13	8025.40	43.36	0.0
Ψ (SDI), Υ (annual), ϕ (.), ρ (.)	12	8046.01	63.97	0.0
Ψ (HDW), Υ (annual), ϕ (.), ρ (.)	12	8060.89	78.85	0.0
Ψ (SDI + HDW), Υ (can_cov), ϕ (.), ρ (.)	7	8087.20	105.16	0.0
Ψ (.), Υ (annual), ϕ (.), ρ (.)	11	8099.95	117.91	0.0
Ψ (SDI + HDW), Υ (.), ϕ (.), ρ (.)	6	8107.22	125.18	0.0
Ψ (can_cov + HDW), Υ (.), ϕ (.), ρ (.)	11	8116.14	130.68	0.0
Ψ (SDI), Υ (.), ϕ (.), ρ (.)	6	8125.66	134.10	0.0
Ψ (.), Υ (.), ϕ (.), ρ (.)	4	8174.07	192.03	0.0

Population Growth Rates

We based Leslie matrix modelling on the following demographic estimates as reported above: adult female survival: 0.76 (95% CI: 0.84 – 0.67), subadult female survival: 0.72 (95% CI: 0.85 – 0.58), juvenile female survival: 0.0.71 (95% CI: 0.85 – 0.57), adult female fertility: 0.554 female kits produced per year. The resulting model indicated that the population growth rate (λ) would stabilize at 0.989 at 20 years, with a 95% confidence interval between 1.1044 and 0.862, suggesting no difference from a stable population (Fig. 15). Projecting these values resulted in a gradually declining population over the next 50 years (Fig. 15), however due to the high variability in female survival rates and the large 95% CI values, results are inconclusive (Fig. 15).

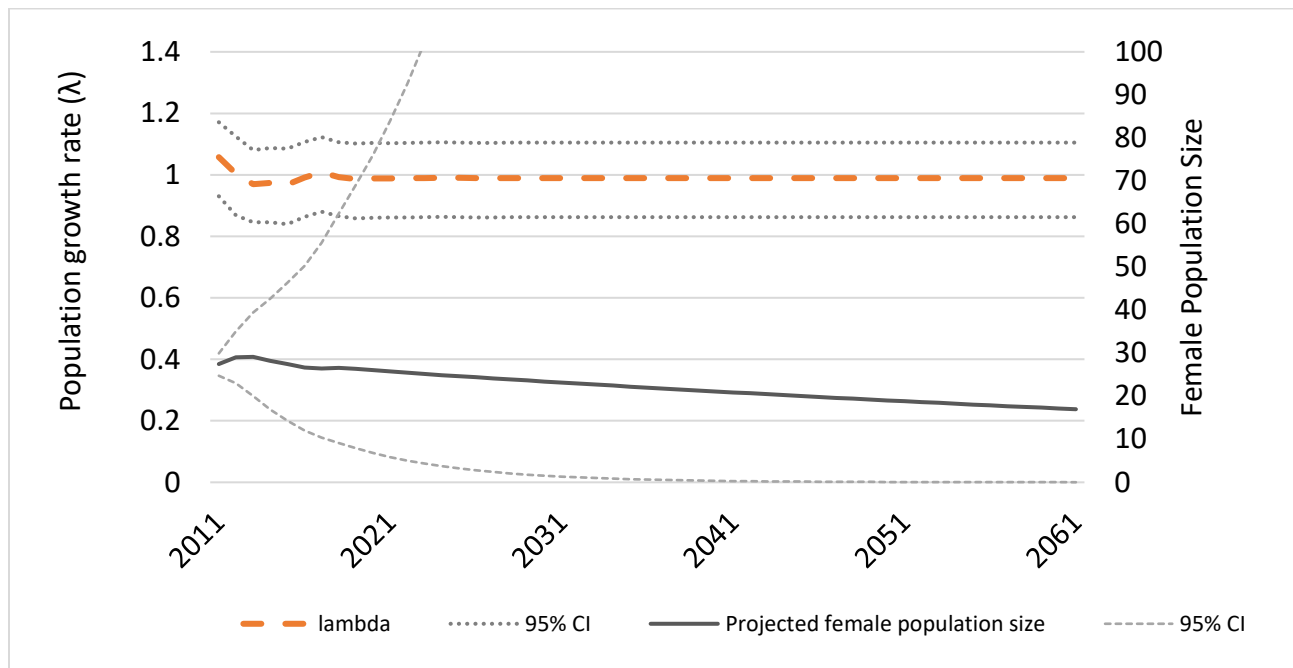


Figure 15. Population trajectory, including growth rate and population size, for female fishers based on estimated fisher demographic values, as well as associated upper and lower 95% confidence intervals.

Home Range Dynamics

We processed ~38,099 fisher locations from aerial telemetry data gathered between October 2007 and November 2016 as part of the SNAMP/Sugar Pine Fisher Project. We also included ~300 den and rest structure locations in our calculations of home ranges for both male and female fishers. Female home ranges were substantially smaller than male home ranges. The average annual home range size for female fishers was 31 km² (range 9 – 78 km²; Fig. 16). The average annual home range of male fishers, including movements during breeding season, was 123 km² (range 53 – 231 km²). Male home range size decreased by 13% to 107 km² (range 41 – 252 km²) when breeding season locations (March 1st to June 30th) were omitted from the analysis (Fig. 16). These home range sizes are large compared to other published estimates (Fig. 17). The higher error rate of locations based on aerial telemetry could be partially responsible for the larger home ranges, but for the most part, the larger home ranges are likely due to the large number of locations we were able to obtain through nearly daily flights. We were able to locate animals that had made movements outside their core area and into more remote areas, which would likely have been missed with ground telemetry.

One male, M02, whose home range size was not included in the male averages, exhibited a home range size of 300 km² outside of breeding season, and 400 km² with breeding season included. M02 had two distinct clusters of telemetry locations. The southern cluster of locations, in the vicinity of Gray's Mountain, was acquired between January and May of 2008. From May 6 through August 7, 2008, M02 disappeared and was not located via aerial telemetry. On August 7, 2008, M02 was found again, this time in the southeastern portion of Yosemite National Park. In the first few days

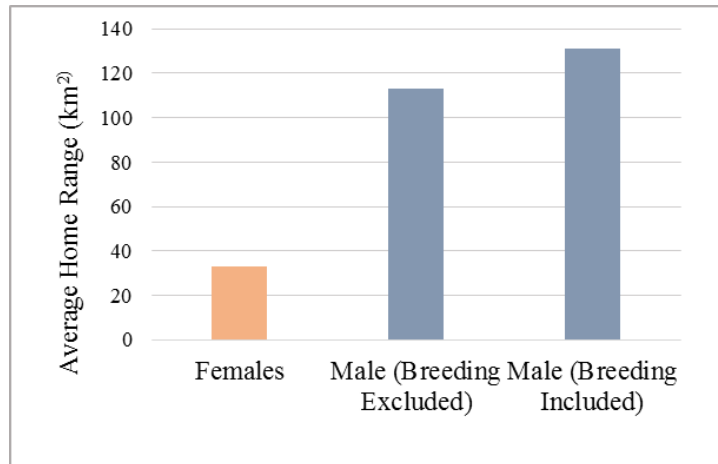


Figure 16. Average home range size (km²) for female (n = 48) and male fisher (n = 42) with breeding season movements included and excluded.

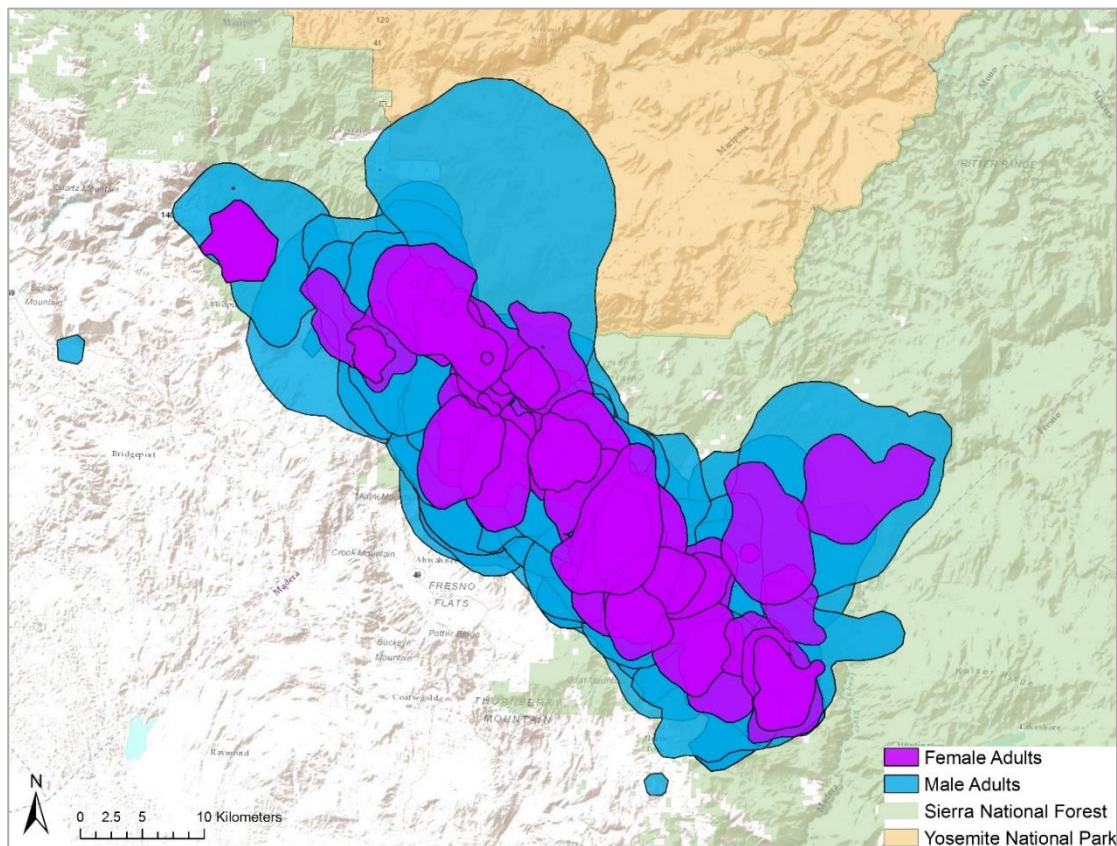


Figure 17. Map of all male and female fisher home ranges from the SNAMP/Sugar Pine Fisher Project between December 2007 and November 2016.

of November 2008, M02 traveled back to the southern portion of his previous home range only to return to Yosemite, traveling over 60 km in four days. Large movements such as this are not uncommon for highly mobile species such as fishers, particularly for young males, however the space use does not reflect territorial behavior or an established home range. M02 then remained within the southern area of Yosemite National Park, periodically venturing north to Yosemite Valley, until his death on Highway 41 in December, 2011.

Fisher Response to Fuel Management

Across the SNAMP/Sugar Pine fisher study area, fuel reduction activities were conducted on 30,530.6 acres, with annual totals ranging from 6963.8 to 1962.0 acres (Table 12). It is important to note that annual totals are not independent; often individual stands were treated in multiple years and may therefore be included multiple times in the total. Therefore the total acreage represents the overall amount of treatment conducted over the study period, however it is incorrect to assume that 30,531 unique acres were treated during that time. Furthermore, camera surveys were not conducted on each grid every year. Only grids where surveys overlapped with management activity provided information for subsequent analyses. For example, only 7 active survey grids overlapped with a prescribed burn during the period of the research effort, compared to 87 and 127 grids for harvest and timber stand improvement projects, respectively.

Table 12. Acreage of fuel reduction activities, by year, conducted within the SNAMP/Sugar Pine Fisher Project study area on the Bass Lake Ranger District, Sierra National Forest. Data compiled from the US Forest Service Forest Activity Tracking Software (FACTS) database.

Year											
Management Activity	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	<i>Total</i>
Harvest	3705	2848	1502	0.0	2371	154	1058	2796	3705	606	17684
Prescribed Fire	0.0	622	609	522	504	319	310	322	134	0.0	2476
Timber Stand Improvement	3259	1625	2398	1483	1699	1489	1516	2091	1476	547	10371
<i>Total</i>	6964	5094	4508	2005	4574	1962	2884	5208	5314	1153	30531

Fishers appeared to be less influenced by fuel reduction activities than by initial site conditions (Table 13). An evaluation of the impact of either the percent of a grid cell treated or the number of years since treatment on occupancy, colonization, and extinction rates resulted in

models that differed from the null model by 1.2 AIC points or less, however combining management impact variables with the vegetation structural variables resulted in significantly better model performance. In general, models performed better when the percentage of each grid thinned or harvested was used to predict extinction rates. These variables did not appear to influence occupancy or colonization rates, and time since management did not appear to influence any of the three derived estimates. Due to the limited overlap between prescribed fire and active fisher survey units, this variable was excluded from analyses.

The top performing model was a combination of the vegetation structure model identified in the occupancy analysis combined with the percent of a grid treated as a covariate on the extinction rate estimate (Table 13). Percent harvest appeared to have a greater impact on fisher behavior than percent thinned and addition of percent of the grid cell impacted by timber harvest generated the greatest increase in model performance. Altering the structural variables

Table 13. Top five fisher occupancy models, plus the null model, based on forest vegetation structure and management activity variables, on the Bass Lake district of the Sierra National Forest, CA. Symbols indicate occupancy (Ψ), colonization rate (Υ), extinction rate (ϕ), and probability of detection (ρ). Vegetation structure variables shown here include SDI (Reineke stand density index), HDW (basal area based on the weighted mean diameter of hardwoods), and can_cov (total % canopy cover). Forest treatment variables include %thin and %harvest, which refer to the percentage of each grid cell impacted by timber stand improvement or harvest activities in a given year, respectively, and annual reflects time since treatment. The null model is specified as Ψ (.), Υ (.), ϕ (.), ρ (.).

Model structure	# Parameters	AIC	Δ AIC	AIC weight
Ψ (SDI + HDW), Υ (annual + can_cov), ϕ (annual+%thin+%harvest), ρ (.)	23	7935.25	0.00	0.5227
Ψ (SDI + HDW), Υ (annual +can_cov), ϕ (annual + %harvest), ρ (.)	22	7935.75	0.50	0.4071
Ψ (SDI + HDW), Υ (annual +can_cov), ϕ (annual + %thin), ρ (.)	22	7940.51	5.26	0.0721
Ψ (SDI + HDW), Υ (annual + can_cov), ϕ (annual), ρ (.)	21	7940.80	5.55	0.0623
Ψ (SDI + HDW), Υ (annual), ϕ (annual+%thin+%harvest), ρ (.)	22	7961.29	26.04	0.0
Ψ (.), Υ (.), ϕ (.), ρ (.)	4	8174.07	238.82	0.0

consistently reduced model performance, and addition of time since management had minimal impact. Graphing the results of the top model (Fig. 18) shows a slight, but statistically significant ($P = 0.05$) downward trend in occupancy associated with increasing the percentage of a grid cell treated for timber harvest and/or fuel reduction.

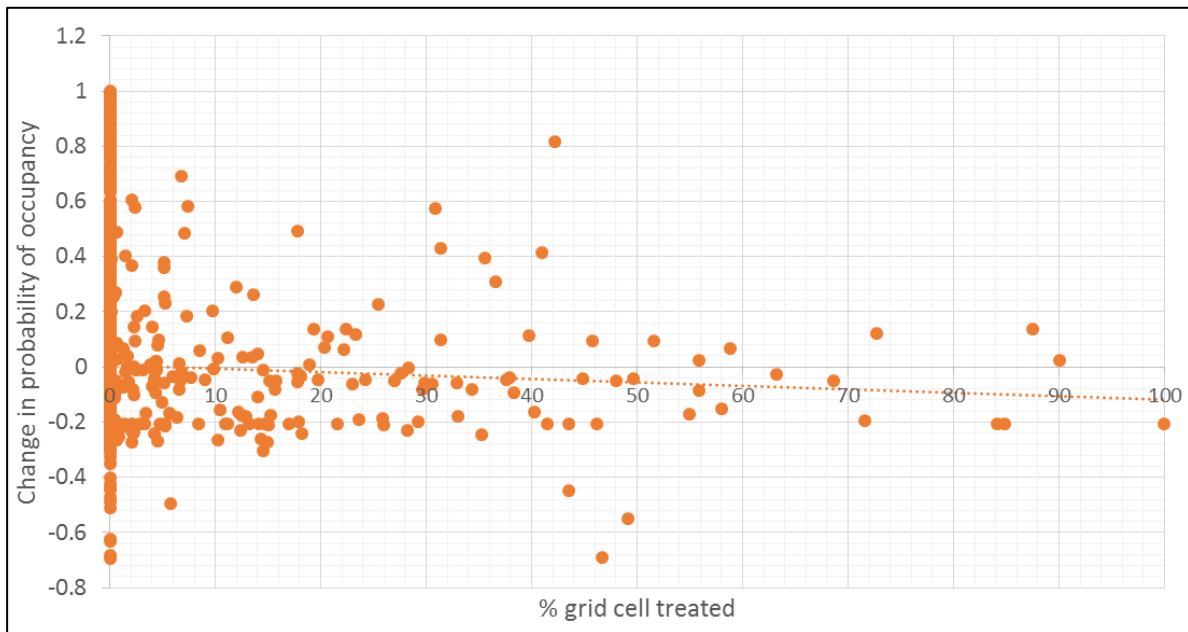


Figure 18. Relationship between the change in fisher occupancy from year (t) to ($t+1$), compared to the percent of that grid cell treated (combined harvest and timber stand improvement) in year ($t+1$). Change in occupancy was estimated using the top model, combining vegetation structure and management intensity $\geq \psi$ ($SDI + HDW$), γ (annual + can_cov), ϕ (annual+%thin+%harvest), ρ (.)

DISCUSSION

Reproductive Rates

Empirical data on reproduction and fecundity are important for understanding the ability of a population to withstand challenges to survival, and to produce realistic estimates of population size and trend in landscape level population models being developed for conservation planning (Lofroth et al. 2010, Spencer et al. 2011). The basic life history of fishers with regards reproduction is generally well known. Fishers exhibit delayed implantation, in which a fertilized eggs develops into a blastocyst before entering a period of suspended development. Blastocyst development, probably prompted by day length, resumes following a delay of nearly 10 months following fertilization (Mead 1994, Powell et al. 2003). Once the blastocyst implants in the

uterine wall, embryonic development resumes and ~ 36 days later 1-4 kits are typically born (Powell et al. 2003).

Across all nine reproductive seasons monitored by SNAMP/Sugar Pine personnel, 78% of all adult females attempted to den. Of all den opportunities, 69% of adult females were successful. The remaining 31% of denning efforts failed for a variety of reasons, both known and unknown. These reproductive rates are low compared to other fisher reproductive parameters reported both in California and throughout the fisher range in North America (Green et al. 2018). For example, 87% of adult female fishers on the Hoopa Reservation in Northern California, and 86% of adult female fishers on the nearby Kings River Fisher Project in the Southern Sierra Nevada Mountains attempted to den (Matthews et al. 2013a, Green et al. 2018). The only report of a lower denning rate in California comes from industrial timberland in the north-central Sierra Nevada (53%; James et al. 2008).

Similarly, our observed litter size of 1.5 kits per litter ($n = 62$ litters), based on direct observation of kits via cameras, is the lowest reported in the literature. In general, litter sizes reported within California are lower than in other parts of North America. Green et al. (2018) reported 1.57 kits/litter in the Southern Sierra Nevada, and Matthews et al. (2013a) reported 1.9 kits per litter on the Hoopa Reservation. Across other parts of their range, mean litter sizes for wild fishers ranged from 1.9 in Southern Oregon (Aubry and Raley 2006) to 2.8 in British Columbia (Simpson et al. 2013) and Massachusetts (York 1996). Higher values have been reported in captive or fur farm populations (Green et al. 2018).

The lower reproductive output observed in the Southern Sierra Nevada region may stem from a variety of both methodological and ecological factors. Researchers have employed a variety of methods in estimating fisher litter size, including counts of blastocysts or placental scars, and observations of kits within natal cavities. Our method, emphasizing non-invasive camera monitoring, is likely the most conservative and would not include kits that died in the womb or soon after birth. At the same time, estimates of litter size generated when the kits were old and robust enough to be moved to a new den are likely more representative of the actual population fecundity rate. For a more detailed summary of the role of methodology in calculating fisher litter size estimates, see Green et al. (2018).

However the low litter sizes reported by both the SNAMP/Sugar Pine and the Kings River fisher projects, as well as the large sample sizes of both studies, suggest an ecological driver. Green et al. (2018) found that fishers in areas with greater seasonality and variability in

temperature had lower litter sizes and, conversely, areas with warmer mean annual temperatures and higher mean annual precipitation, as found in the southern Sierra Nevada, have smaller litters. The reasoning is that in areas with harsh winters, larger litter sizes might compensate for high winter mortality (Tökölyi et al. 2014). By extension, higher survival rates in areas with milder winters could offset smaller litter sizes. The extent to which these factors balance each other out is unknown. Alternative explanations for small litter sizes in the southern Sierra Nevada include smaller body size, diet limitations, and lower genetic variability (Zielinski et al. 1999, Zielinski and Duncan 2004). As a result, females may be more challenged to provide sufficient nutrition during the denning season. Fisher habitat in the southern Sierra Nevada also occurs in a relatively narrow, linear band along the west slope of the mountains, with periodic areas of constriction along river canyons. Sierra Nevada fishers may therefore be more spatially constrained, and smaller litter sizes may be an evolutionary adaptation to historic population density. Regardless of the cause, the low reproductive output observed indicates that fishers in the southern extent of their range may be more sensitive to stochastic events, and less resilient to population fluctuations, than fishers in other areas.

Denning Structures and Habitats

Female fisher reliance on cavities for reproductive dens has been well established across North America (Paragi et al. 1996, York 1996, Raley et al. 2012, Weir et al. 2012, Zhao et al. 2012, Matthews et al. 2013a, Green et al. 2018). Cavities provide protection from predators and thermal insulation in early spring to late spring when females are rearing their young (Lofroth et al. 2010, Weir et al. 2012). Females often move their kits throughout the denning season, using between 1 and 8 dens each year (Matthews et al. 2013a, Niblett et al. 2015); female fishers on the SNAMP/Sugar Pine Fisher Project used an average of 2.4 different denning structures per den season (range 1-5), compared to 3.4 on the Kings River Fisher Study (Green et al. 2018), and 3.1 in northwestern California (Matthews et al. 2013a). Female fishers may use more than one denning structure in a season for several reasons: to accommodate kit growth by moving to larger cavities, to reduce predation risk, as bobcats and mountain lions may discover a den location due to odors from the accumulation of urine and feces, to move closer to unexploited foraging areas, and to avoid exposure to feces and parasites that may accumulate in den cavities. The lower number of dens found in this study could indicate that suitable alternative cavities are limited in this area.

Green et al. (2018) currently provides the most extensive analysis of fisher denning structures and habitat in the Sierra Nevada region. Based on a sample size of 83 natal and 192 maternal dens, they documented that natal dens occurred most often in live hardwoods (45.8%), followed by live conifers (33.7%), and conifer snags (15.7%). Maternal dens occurred most often in live hardwoods as well (55.2%), followed by conifer snags (26.6%) and live conifers (20.3%). California black oak was the most common species used for denning (55%), followed by white fir and incense cedar.

Comparatively, fishers in the SNAMP/Sugar Pine region used incense cedar, white fir, and California black oak in relatively equal proportions (33.5%, 30.2%, and 29.1% respectively). Live trees accounted for 54.4% of dens, and snags accounted for the remaining 45.6%. However the proportions become more skewed when looking at den types; 70% of natal dens occurred in live trees while only 40% of maternal dens did. Fishers rely on the den site characteristics to keep young kits warm and secure (Lofroth et al. 2010), therefore the higher thermal insulation and stability of live trees compared to snags may be particularly important in the selection of natal dens (Green 2017). This pattern, the selection of large trees with sufficient decay to produce cavities but which do not facilitate air flow and reduce thermal insulation, has been observed in other areas as well (Weir et al. 2012, Raley et al. 2012, Thompson et al. 2016).

Forest characteristics in the immediate vicinity of the den site may also play a role in moderating environmental conditions, providing sufficient prey resources, and security from predators. Den sites on the SNAMP/Sugar Pine study generally occurred in areas characterized by high overhead cover (mean 74%, range 30-98%) and low to moderate shrub cover. However when observed horizontally from ground level, as most potential predators would be doing, understory concealment cover was relatively high (72% for 0-0.3m, 47% for 0.3-1m, and 58% for 1-2m). Comparatively, Green (2017) reported concealment cover estimates of 63.2% and 53.5% for 0-1m and 1-2m, respectively.

The limited use of hardwoods for denning on the SNAMP/Sugar Pine Fisher Project is at odds with use reported by other research projects throughout California. Green (2017) reported that 51.3% of the 301 den sites identified were located in live hardwoods, primarily California black oak (*Quercus kelloggii*). Similarly, of the 111 dens located between 2005 and 2008 on the Hoopa Reservation in Northern California, 69% occurred in hardwoods, primarily tanoak (*Notholithocarpus densiflorus*) and California black oak. And fishers reintroduced to the central

Sierra Nevada used hardwoods for 56% of the dens located, again primarily California black oak (Thompson et al. 2015).

The repeated use of individual structures as both natal and maternal dens that we observed, the limited use of hardwoods compared to other studies, along with fewer total structures used compared to other studies, indicates that these structures may be limiting on the landscape. Not only were certain trees used repeatedly by individual females for denning, on three occasions a different, unrelated female selected the same tree for denning that had been used by a previous female in that area. Given the long-term ecological investment required to create a suitable den structure, including the age of the tree, presence of disturbance and tree pathogens to initiate cavity creation, and activity by primary cavity excavators (Raley et al. 2012), it is not surprising that suitable sites are reused by multiple individuals.

Fisher Survival and Cause-specific Mortality

Across North America, published reports of fisher annual survival rates range from 0.88 for females in British Columbia to 0.39 for males in Ontario (Koen 2007). On the whole, lower survival rates appear to correspond to regions where fishers are still legally harvested (Lofroth et al. 2010). Females generally have higher survival rates than males, with the only published exception occurring in southern Oregon, where Aubry and Raley (2006) observed a male survival rate of 0.85 and a female survival rate of 0.78. Our finding of an overall adult survival rate of 0.71 (0.76 female / 0.66 male) is similar to other studies, and nearly identical to estimates generated by the nearby Kings River Fisher Project (0.77 female / 0.65 male; Thompson unpublished data).

Mortality rates fluctuated widely throughout the study, ranging from 2012/2011 when 13 animals (24% of monitored population) died to 2014/2015 when only 2 animals (6% of monitored population) died. While overall survival of females was higher than for males, differences between male and female survival rates appear to be exaggerated in the younger age classes. Juvenile and subadult females exhibited annual survival rates similar to adult females (71% and 72% respectively compared to 76% for adult females), however the annual survival of males was only 40% for juvenile and 50% for subadult, compared to 66% for adult males. Variations in survival rates by age is common in mesocarnivores (Farias et al. 2005, Murdoch et al. 2010); more surprising is the relatively high survival of juvenile and subadult females in the

SNAMP/Sugar Pine study area. For a detailed assessment of the possible reasons underlying variation in survival rates, both annually and seasonally, and by sex, see Sweitzer et al. (2015).

In general, the addition of three years of post-treatment monitoring data did not significantly alter the age-specific fisher survival results initially reported in the SNAMP Final Report (Sweitzer et al. 2015). Estimates of adult female, adult male, and subadult female survival rates increased slightly while subadult male and juvenile female survival rates decreased slightly. The only significant difference was a decrease in juvenile male survival, from 0.6 to 0.4, however due to the high annual variation in survival rates across all age categories this does not appear to reflect any specific impact of management activities.

Predation has been firmly documented as the primary source of fisher mortality in California (Gabriel et al. 2015) as well as across the western United States and western Canada (Lofroth et al. 2010). Continent-wide, primary predators include primarily mountain lion, coyote, bobcat, Canada lynx, and wolverine (Raley et al. 2012). In the eastern United States and Canada, sources of fisher mortality are often more anthropologically-based, such as trapping and vehicle collisions (Krohn et al. 1994, York 1996, Koen et al. 2007, Belant 2007). While we observed limited vehicle and entrapment-related mortality, 61% of our observed mortality was attributed to predation. This is higher than the averages reported by Lofroth et al. (2010), who reported that 43% of all mortality observed in 8 fisher research projects in the western United States and Canada could be attributed to predation.

Anticoagulant rodenticide exposure, associated with illegal marijuana cultivation on public and tribal land, has emerged as a significant factor in fisher population dynamics in the western United States over the past decade (Spencer et al. 2015). In 2009, the first documented AR/marijuana-related fisher mortality was observed on the SNAMP Fisher Project. Since then, the issue has gained international attention yet exposure rates continue to increase (Gabriel et al. 2015, Thompson et al. 2017). While it is only the third leading cause of mortality on the SNAMP/Sugar Pine Fisher Project, 14% additive mortality may be enough to limit population expansion (Spencer et al. 2011). Adding to this concern is the fact that correlative evidence of interactive effects between sublethal poisoning and predation rates has been documented (Thompson et al. 2014), meaning that it is likely that natural rates of predation are inflated by exposure to toxicants (Thompson et al. 2017). Sublethal exposure rates also have the capacity to limit reproduction and increase the probability of dying from minor injuries (Erickson and Urban 2004, Gabriel et al. 2018).

Infectious disease is a significant concern for isolated mesocarnivore populations that persist in a landscape supporting a diverse carnivore community and for species with limited distributions. Canine distemper virus (CDV) in particular is a conservation concern, having caused near extirpation of species such as the black-footed ferret (*Mustela nigripes*; Williams et al. 1988) and the island fox (*Urocyon littoralis*; Timm et al. 2009). In 2009, we documented the impacts of a local canine distemper outbreak on SNAMP/Sugar Pine fishers when four collared fishers died of CDV (3 SNAMP animals, 1 KRFP animal) (Keller et al. 2012). An additional three fishers showed evidence of exposure to CDV, but apparently recovered and were not actively infected at the time of testing, and two foxes were observed suffering from what was likely advanced CDV though infection was not pathologically confirmed. The death of one fisher, a juvenile male, was likely related to an active infection of canine parvovirus, and an additional male fisher died due to complications after parasitic infection by *Toxoplasma gondii* (Gabriel 2013). Although exposure of fishers to *Toxoplasma gondii* has been previously documented for fishers in North America (Larkin et al. 2011), this was the first case where complications from toxoplasmosis resulted in death (Gabriel 2013).

The role of disease and poisoning as contributing factors in facilitating predation mortality is difficult to interpret. While the two sources are each known to have directly accounted for 14% of observed mortality, it is likely that fishers suffering the symptoms of infection or exposure may be less vigilant, slower to respond, and therefore at greater risk of predation (Keller et al. 2012, Thompson et al. 2014). Of the 39 carcasses submitted for necropsy that included sufficient liver tissue to test for AR exposure (Table 9), 36 (92%) showed exposure to between 1 and 4 toxicants. Of the 27 animals that were determined by necropsy to have died as a result of predation, 14 provided sufficient liver tissue to test for AR exposure. Of these 14 animals that died of predation, 13 (93%) tested positive for AR exposure. Of the same 27 predation cases, 10 were collected quickly enough to test for evidence of exposure to pathogens known to infect fishers. Of those 10, 6 showed evidence of exposure to one or more pathogens (3 - canine distemper, 4 - *Toxoplasmosis gondii*, 1 - canine parvovirus, 1- canine adenovirus). While only circumstantial, the high percentage of predation mortalities that show exposure to these additional, potentially contributing factors, suggests that predation rates on fishers by larger carnivores in the southern Sierra Nevada may be inflated by these additional factors. Such an interaction would be one potential explanation for the higher predation rates we observed compared to those reported by Lofroth et al. (2010). While the interaction between disease and

predation is likely a natural phenomenon, it may be inflated by the number of domestic animals present in the area and the additional disease vectors these animals represent. Furthermore, any interaction between AR exposure and predation is clearly an artificial, anthropogenic influence that could represent a threat to fisher persistence in the region.

Population Size and Growth

An earlier analysis of SNAMP Fisher Project data indicated a population density ranging between 0.072 to 0.097 fishers/km², and a population size of between 48 and 62 individuals in the overall fisher study area (Sweitzer et al. 2015). Leslie matrix modelling at that time indicated a population growth rate of 0.90 (95% confidence interval: 0.77-1.22). Because the overall project footprint was reduced in 2014, we were unable to generate comparable population density and size estimates. However our estimated population growth rate of 0.989 (95% CI: 0.862 – 1.104), while similar, is slightly higher and may represent an improvement over the previous estimate. The confidence intervals of both estimates overlap 1.0, due to the high annual variation in survival rates, therefore no conclusions can be drawn regarding the population trajectory beyond the appearance of fragile stability.

Tucker et al. (2014) suggested that a combination of recent genetic analyses and survey data indicate that the SNAMP/Sugar Pine fisher population may be the result of a recent northward population expansion. They found a high degree of fine-scale genetic subdivision in the area immediately north of Shuteye Peak, indicative of up to four recent founder effects. The recent discovery of a native, subadult male fisher north of the Merced River in Yosemite National Park supports this theory, as does our observation of slight improvements in the SNAMP/Sugar Pine Fisher Project demographic rates over time. Despite these indications of expansion, the fisher population immediately south of Yosemite National Park remains small, prone to annual fluctuations in survival, and is subject to numerous threats such as habitat loss to wildfire, periodic disease epizootics, and increased mortality associated with illegal toxicant exposure.

Fisher response to fuel management

While the survival rates we observed varied substantially across age classes and years, there did not appear to be a discernable pattern associated with vegetation management. Similarly, annual variation in fisher reproductive rates did not appear related to management activities.

Overall, fisher occupancy was not negatively impacted by management activity (Illustration 12). Fishers did appear to be temporarily displaced by management activities, as evidenced by the relationship between treatment intensity and local persistence. However this effect was limited to the year during which the management activity was conducted and did not represent a longer-term shift in fisher behavior. Instead, fisher occupancy and colonization rates were more associated with site conditions, including structural complexity and canopy cover.



Illustration 12. Image of a fisher taken by remote camera as part of occupancy monitoring efforts on the Sierra National Forest, CA.

Our results are similar to those reported by Sweitzer et al. (2016). In an earlier examination of SNAMP data, the authors concluded that fisher occupancy was not negatively associated with either extractive or restorative fuel reduction, though disturbances from restorative fuel reduction had a negative effect on local scale persistence. By examining telemetry data in conjunction with the occupancy modelling, the authors determined that fishers appeared to be avoiding the immediate vicinity of fuel treatment activities, yet moving around the impacted areas and maintaining stable home ranges. Therefore the pace and scale of treatments are critical considerations in minimizing impacts of fuel reduction on fishers. The tolerance of fishers to fuel reduction activities was investigated by Zielinski et al. (2013b), who reported that fishers on the Sierra National Forest continued using landscapes exposed to 2.6% treatment/year. While individual cells in our study may have had up to 80% of the cell treated in any given year, because these treatments were well dispersed, less than 1% of the overall study area was treated in any given year.

Garner (2013) examined the relationship between fisher space use and forest

management activities on the Sierra National Forest, and found that fishers did not appear to orient their home ranges or core use areas to avoid past treatments. Instead, treated stands were included in fisher territories in proportion to their availability on the landscape. Within their territories, fishers did avoid treated areas, generally limiting their use of areas within 200m of a past forest management project.

The general pattern suggested by these studies is similar to what we observed; fishers avoid management activities in the short-term but generally do not abandon territories. Instead, they move around treated areas and may begin reusing treated landscapes soon after management is completed if sufficient forest structure remains. As noted in Sweitzer et al. (2016), the scale of monitoring on the SNAMP/Sugar Pine Fisher Project (1km²) was relatively fine compared to the size of resident female (31 km²) and male (123 km²) fisher home ranges. Therefore as long as treatments are dispersed across the landscape and not locally concentrate, the vast majority of a fisher's home range would remain undisturbed.

Collar Removal and Project Wrap Up

A considerable amount of time was needed to bring the SNAMP/Sugar Pine Fisher Project to a close. The most important part of closing down the study was to put forth as much effort as possible to remove radio collars from the fishers that were still on air when the project ended in December 2016. At the time of the last telemetry flight on November 10th, 2016, eight fishers (four females and four males) had collars that were still emitting signals, although many of the signals were losing strength indicating the batteries were dying. Although all Sugar Pine fisher collars were constructed to eventually drop off, it was our priority to remove as many of the collars as possible at the end of the study to ensure the comfort and safety of the animals. With limited resources and personnel we were able to trap in two different locations, targeting four of the collared fishers that were known to use the areas where our traps were placed. Ten traps were open and baited for 22 trap nights between November 20th and December 15th, 2017. During this time period one known adult male was trapped and his collar was removed and one new juvenile male was trapped and processed but not collared. Poor weather conditions and a lack of sufficient personnel caused the trapping effort to be cut short.

An aerial telemetry mission was flown over the study site on February 14, 2017 to locate any fishers whose collars may still be working. One female fisher was heard on that flight and was located within the previous trapping area. Ten traps were set for another nine trap nights

between February 22nd and March 3rd 2017, in an effort to catch F17, one of the oldest fishers on the SNAMP/Sugar Pine Fisher Project. Although F17 was not successfully captured, an adult male was captured and his collar was removed. Traps were closed in early March due to the onset of fisher denning season.

Management Implications

Predation was the largest source of fisher mortality observed, with the majority of mortality being caused by mountain lions and bobcats. Due to the rarity of fishers across the landscape, it is unlikely that either predator was intentionally targeting fishers as a primary prey source. More likely, chance encounters between two predator species resulted in the death of the smaller one. Therefore, management activities that promote intraguild carnivore interactions should be minimized. For example, fishers utilize dense forest stands while both bobcats and lions are known to travel roads and trails extensively (Nichols 2017). Therefore the construction of trails, skid roads, or permanent firebreaks through dense habitat could increase the rate of interactions between the species. All three species are also likely to use the edges of clearings or treated stands as hunting opportunities. Therefore dispersing treatments across the landscape, without connecting them, is likely to minimize carnivore interactions while concentrating treatments would have the opposite effect.

Natural mortality rates are also likely inflated by the presence of illegal toxicants across the landscape. While the 14% mortality we observed due to anticoagulant rodenticide poisoning is sufficient to limit population growth (Spencer et al. 2015), the potential for indirect effects makes the effect far more concerning. Sublethal exposure to anticoagulant rodenticides and other toxicants has been shown to cause lethargy, partial paralysis, reduced immune system function, and reduced healing capacity (Thompson et al. 2014). These interactions may be one potential explanation for the increased predation rates we observed compared to other fisher populations in the western United States and Canada (Lofroth et al. 2010).

Sweitzer et al. (2016) reported a nonsignificant but slightly negative population growth rate for the SNAMP fisher population. The addition of three years of monitoring data raised the estimated lambda to 0.989 and closer to a lambda of 1, representing a stable population. Ten years of monitoring therefore indicate that the fisher population on the Bass Lake Ranger District has remained stable while experiencing high annual variation in reproductive and survival rates. This evidence of tentative stability supports the conclusions of Zielinski et al. (2013a), who

reported consistent occupancy of fishers across the Sierra Nevada region. However the high annual variation in demographic rates indicates that the population is at risk of stochastic events such as disease outbreaks or the loss of large portions of habitat due to wildfire.

Accumulating evidence also suggests that fishers may be tolerant to the levels of forest management necessary to change fire behavior in the Sierra Nevada (Zielinski et al. 2013b). Our results, as well as those of Sweitzer et al. (2016) suggest that as long as treatment activity is limited in scope and duration, and critical structures such as large oaks and damaged or decaying conifers are retained, fisher avoidance of the area is relatively brief. Given the scale of fisher habitat use compared to the scale of fuel reduction efforts, dispersing management activities across a fisher home range to the degree where impacts are minimized, appears to be a viable option.

The scope of the drought and insect-caused tree mortality, which began as the SNAMP/Sugar Pine Fisher Project was nearing completion, highlights this concern. Large portions of the Bass Lake Ranger District, and the Sierra National Forest, have experienced unprecedented levels of conifer mortality since 2015. While the effects of this dramatic change on fisher habitat use and population persistence is unknown, the loss of large conifers and canopy cover, two forest structural elements known to be important to fishers, are likely to reduce habitat quality in many areas. Ongoing fisher monitoring on the nearby Kings River Fisher Project may help illuminate how fishers are responding to this landscape-level change, but until more data are available management should be conservative with respect to the management of existing habitat patches.

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APPENDIX

Key protocols used for assessing vegetation characteristics at fisher den trees for the SNAMP/Sugar Pine Fisher Project.

The purpose of this protocol is to begin to understand the combination of biotic and abiotic characteristics female fishers are likely selecting/using for denning habitats. Our goal is to collect similar types of data as those being recorded by the Forest Health Team on the Core Plots in the Sugar Pine area, while also capturing the same types of data being recorded by the Kings River Fisher Project at their den trees.

For each natal and maternal den tree we will define a circular plot centered on the den tree with a fixed radius of 18 m (area ~ 1 hectare) (Fig. A1). Each circular plot will include four 18 m transects, one each oriented N (0/360°), E (90°), S (180°), and W (270°), originating from small nails in the base of each den tree (removed at completion of plot). We will next define four 4m x 18m belt transects as the sampling space extending 2m from either side of each 18 m line transect. Finally, within the circular plot centered on the den tree, we will also define 4 quadrants ordered A, B, C, and D clockwise from the N transect.

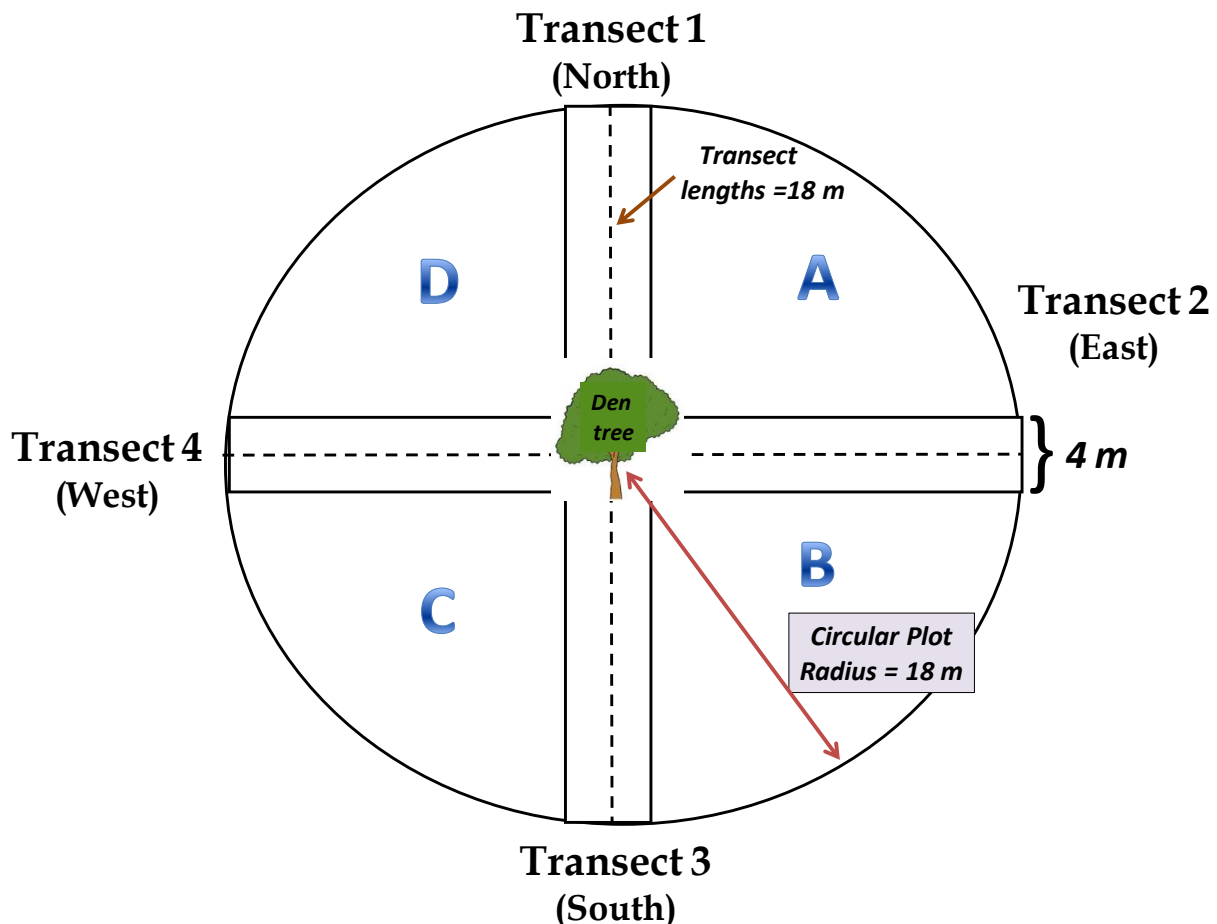


Figure A1. Diagram illustrating the layout of circular plots and associated measurement transects, including 4 quadrants A, B, C, and D, for vegetation data collected at dens.

The diagram shows a circular plot with four transects labeled Transect 1 (top), Transect 2 (right), Transect 3 (bottom), and Transect 4 (left). A central sun-like symbol represents the Plot center. A legend box titled "Transect measurements" is located in the top right corner, containing the following items:

- Canopy cover (blue plus sign)
- Duff & litter layers (green circle)
- Herbaceous veg height (yellow pentagon)
- Fuels height (red triangle)
- Plot center (sun symbol)

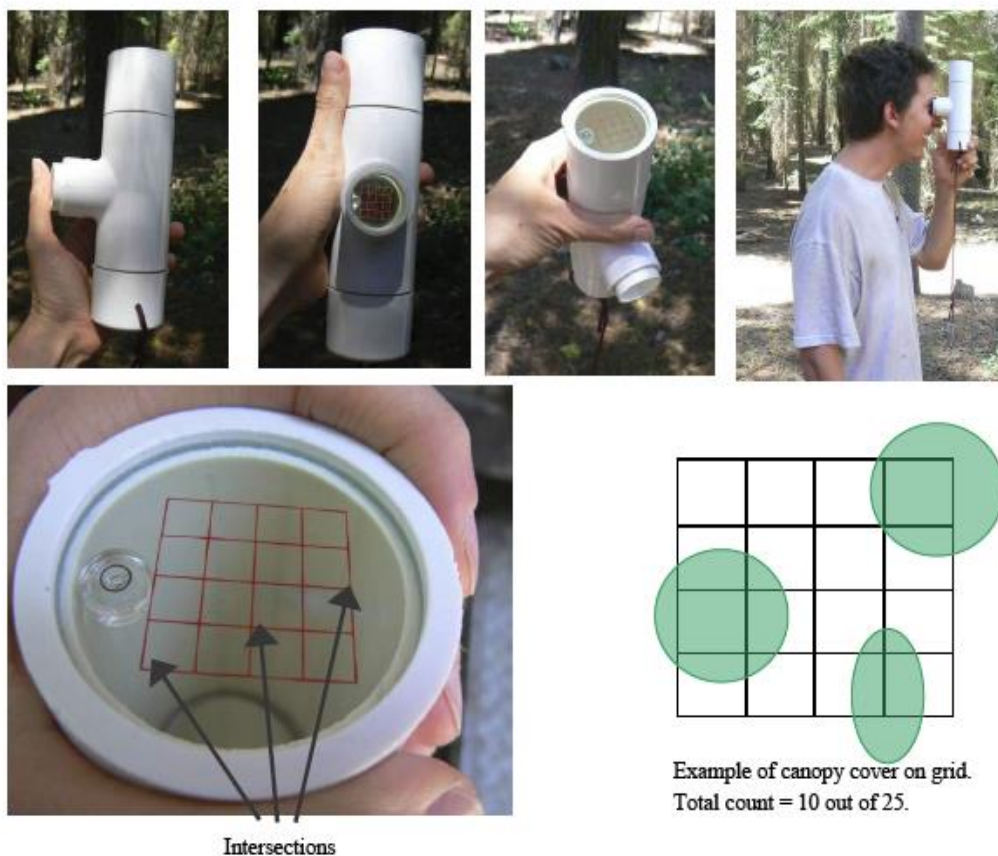
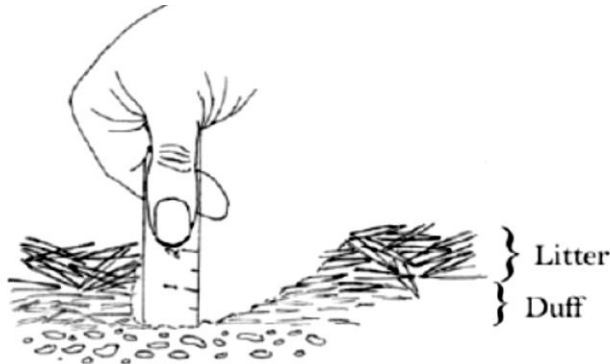


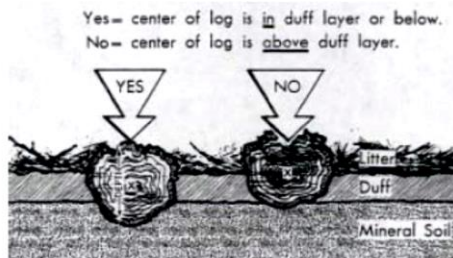
Figure A3. Overview of use of the moosehorn coverscope. The coverscope should be held with the longest length perpendicular to the ground. The small side tube should be held up to the observer's eye and leveled with the bubble level. The user will then count the number of intersections covered by canopy. The lower left photo illustrates what the intersections look like (corners, side intersections, and middle intersections). The diagram on the lower right provides an example of what canopy cover might look like on the grid. In this example, the count would be 10 out of a total of 25.

Assessment of Litter, Duff and Fuel Height: We will measure the thickness of the duff and litter layers, herbaceous vegetation, and fuel height at several positions along each 18m transect. Measurements are done at the 2m, 10m, and 18m positions on each transect (Fig. A4). Height of herbaceous vegetation (herbs or grasses), if present within a 10 cm radius of the point, should be measured first, prior to excavating holes for measuring the duff and litter layers. Use a hand trowel to dig three small vertical holes down through the litter+duff to mineral soil at 2m and 10m and 18m along each 18m transect. If a tree or stump occurs at the position, offset your digging 30cm to the right. Measure the thickness of the duff ($\pm 0.5\text{cm}$) from the mineral soil to top of duff, and the litter layer ($\pm 0.5\text{cm}$) from the duff to top of litter, not including woody debris (branches/sticks/logs). Fuel measurements start at the bottom of the litter layer and end at the top of the tallest fuel up to 1.83m (6 feet). Fuels include any woody twig ($> 0.64\text{ cm}$ diameter), branch, or log that is severed from the original source of growth with its central axis lying above the duff layer (excluding needles, grass, bark or pine cones). Basically, if the fuel object isn't more than 1/2 buried in the duff, its height up to 1.8 m/6 feet is measured. If no fuel exists, record the height of the litter at the point.

Figure A4. Diagram illustrating how to properly discriminate between the litter and duff layers for taking measurements.



When stumps, logs, and trees occur at the point of measurement, offset 30 cm perpendicular to the right side of the sampling plane. Measure through rotten logs whose central axis is in the duff layer.



Duff depth is measured through a rotten log when its central axis lies in or below the duff.

- Duff thickness (mineral soil to top of duff) (see diagram)
- Litter layer thickness (duff to top of litter; $\pm 0.5\text{cm}$) (see diagram)
- Fuel height (bottom of litter layer to top of tallest fuel up to 1.8m/6 feet); if no fuel at point record the depth of the litter layer)
- Herbaceous/grass vegetation height (nearest 1 cm)

Shrub level vertical cover (i.e., concealment): Shrub level vertical cover will be measured using a cover-board/drop-cloth design (Fig. A5). As we are concerned with concealment in general, not just shrub cover or foliage density, anything that provides potential cover to the fisher as it is coming or going from the tree should be counted (e.g., tree trunks, sapling foliage, shrub cover, boulders). The drop-cloth is 3 m x 0.5 m and is composed of 0.1 x 0.1 m squares separated into 4 categories: 0-0.3 m (15 squares, low ground), 0.3-1m (35 squares, high ground), 1-2m (50 squares, low shrub), and 2-3 m (50 squares, high shrub). One observer stands with their back to the den tree while a second person holds the cloth at the 10 m position along each transect. The observer counts the number of squares within each height interval at least 50% obscured by ‘cover’ and records this number. The observer should squat while reading the two lowest sections (low ground and high ground) and stand while reading the upper sections (low shrub and high shrub). The technician holding the cover-board should try to hold it as straight and steady as possible, taking care to make sure the bottom dowel is hitting the ground and that the cloth is taut in all directions.

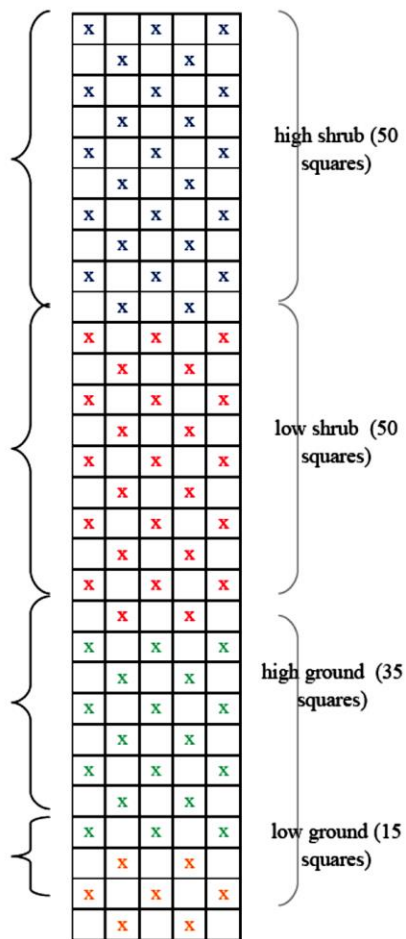


Figure A5. Design and use of coverboard for estimating concealment cover around the base of den trees. The coverboard is divided into four sections representing low ground cover, high ground cover, low shrub cover, and high shrub/small tree cover.

Measuring Topographical/Landscape Features: The prevailing aspect for the slope upon which the den tree is located will be measured using a clinometer. Walk to a position on the slope adjacent to the den tree with your back to the hillside behind you. Use the clinometer to estimate the prevailing aspect (Hint: if you spilled your water, which way would it flow?). Slope will also be measured for the hillside upon which the tree is located. Use a clinometer to take two measures of slope, which will be averaged for the recorded measurement on the data form. To obtain an appropriate measure of slope look directly uphill and locate a tree or object about 15-25 meters away. Estimate where your eye height would be on that tree/object (around 1.5 m), look through the viewfinder on the clinometer and read the % slope on the right hand side.

Assessment of Sizes, Numbers of Trees and Snags within Circular Plot: We will enumerate and measure all medium and large size trees and snags (see below) within each area of the circular plot. All medium and large size trees will be characterized by Vigor Class and Crown Class.

Trees and snags will be grouped into three size classes for measurements:

- Size class 1: greater than or equal to 19.5cm dbh
- Size class 2: 5.0cm to 19.4cm dbh
- Size class 3: ≤ 5.0 cm dbh

Live trees and snags are further broken down into 6 vigor classes (stages; Fig. A6) as follows:

Live trees:

- Class 1: Healthy tree with no visible defects
- Class 2: Healthy trees with minimal damage or defects (broken/dead tops, abnormal lean, etc.)
- Class 3: Live trees that appear near death or likely to die within 5 years

Dead trees

- Class 4: Recently dead tree with little decay (retains bark, branches, top, even some needles)
- Class 5: Tree showing some decay including loss of some bark, broken off branches/top
- Class 6: Tree shows extensive decay including loss of most bark, branches, broken top

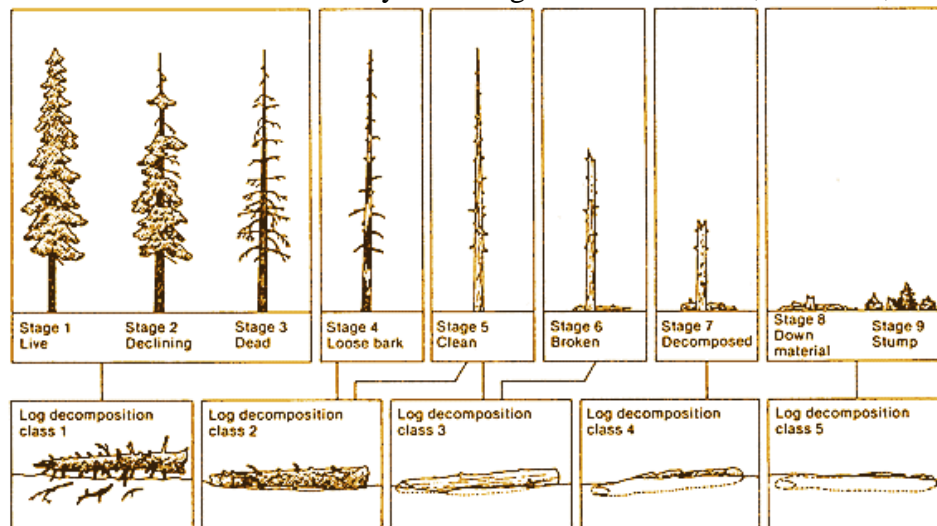


Figure A6. Illustration of possible vigor classes for trees and snags. Vigor class 3 represents trees that are near death, vigor class 4 includes recently dead snags, vigor class 5 snags are those with loss of limbs and loose bark, and vigor class 6 snags are those with broken tops, few if any limbs and visibly decomposed.

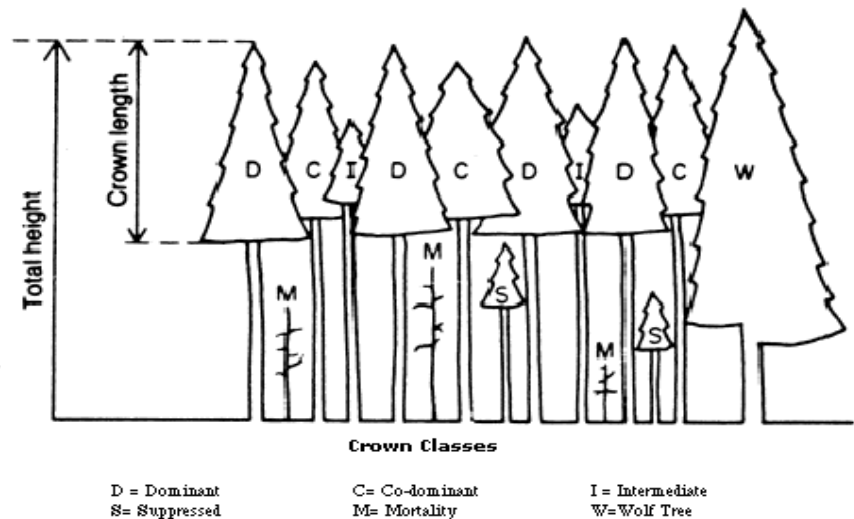
Relascope: A relascope or prism is used to determine which trees will be measured. The default Basal Area Factor is 40, but 20 can be used if needed. (We will always use a factor of 20). Stand holding the relascope 5 ft (1.524 m) north of the den tree. Keeping your feet in the same place, rotate 360 degrees, counting all trees and snags that are “in” based on the 20 factor. To do this, the person holding the relascope will look through it, lining up the left edge of the left white bar with the left side of a tree, and if the right side of that tree is outside of the right side of the second white bar (from the left) that tree is considered “in.” For each tree or snag that is in, record the following, based on the minimum DBH requirements:

- **Live trees-** must have a DBH ≥ 12.7 cm (5 in). Determine the species (Table A2) and measure the DBH for all trees and snags. The height of a representative tree from each Crown Class should be measured, as well as the den or rest tree. Estimate % live crown, assign a position/dominance class [See Crown Class requirements and Fig. A7 below], determine snag decay (rating 3-9), and look for woodpecker holes, other cavities, and mistletoe.
- **Snags-** must have a DBH > 38.1 cm (15”). Determine the species, measure the DBH (estimate if snag is unsafe) and snag height, assign a decay class. Also note presence of woodpecker holes, other cavities, or mistletoe.
- On the datasheet, record Tree tag number (if present); Tree status (live/dead); Tree count (which will always be 1), and any Tree remarks

Live trees only will be characterized by Crown Class as follows (Fig. A7):

- **D (Dominant):** trees with crowns extending above the general level of the crown cover and receiving full light from above and some from the sides
- **CD (Co-dominant):** trees with crowns forming the general level of the crown cover and receiving full light from above, but little light from the sides
- **I (Intermediate):** trees that are shorter than the class D and CD trees but with crowns either below or extending into the above crown cover and receiving little direct light from above or the sides
- **S (Suppressed):** trees with crowns entirely below the general level of the crown cover, receiving no direct light from above or the sides

Figure A7. Diagram illustrating trees of different crown classes, and associated crown height measurements.



Trees/Snags within Size Class 1 and 2: We will measure the straight line distance from the den tree to every tree or snag within size classes 1 and 2 in each of the four quadrats (A, B, C, D) of the circular plot around the den tree. In addition, we will take a bearing from the den tree to each of the Size Class 1 and 2 trees in order to create a stem map once back in the office. After measuring the distance (± 5 cm) and bearing to each of these trees/snags, record/assess the vigor class, crown class (live trees only), height to live crown base (live trees only) and total tree/snag height for each of these trees/snags.

Small Trees/Snags (Tree Size Class 3): All live and standing dead trees less than 5.0 cm dbh and a minimum height of 1.37 m will be counted in each of the four quadrats around the den tree. Track on the data form the live/dead status of these trees.

Assessment of Woody Shrubs: Woody shrubs will be sampled along each of the four 18 m transects using the line-intersect technique. For all woody shrubs that occur along and intersect the transect line record/measure:

- Shrub species (Table A2)
- Length of shrub intersect (nearest 10 cm)
- Average height (± 5 cm)
- We will also record an ocular estimate of overall shrub cover by species within each of the 4 quadrats

Assessment of Cover by Herbaceous Plants: In addition to measuring the height of herbaceous cover (± 1 cm) at the 2 m, 10 m, and 18 m positions along the four line transects, we will also assess and record an ocular estimate of the percent cover of herbs and grasses combined in each of the four quadrats around the den tree.

Assessment of Course Woody Debris (CWD): Course woody debris is defined as dead tree boles, large limbs, and other large wood pieces either lying on the ground or elevated off the ground up to 45° , but no longer supported by roots (i.e., dead trees hung up or leaning on other vegetation). CWD does not include live material, standing dead trees, stumps, dead foliage, and separated bark, non-woody pieces, roots, or the part of the bole below the root collar. We will assess/measure CWD along two of the four 4 X 18m belt transects, either transects 1 and 3, or transects 2 and 4. Use a coin flip to choose between Transects 1 and 2; if Transect 1 was selected then CWD will be assessed/measured along Transects 1 and 3, whereas if Transect 2 was selected then CWD will be assessed along Transects 2 and 4. Each piece of CWD sampled within the belt transects must have a large end diameter (LED) of 15cm and be at least 1m in length.

Core variables to be measured include:

- Species (if determinable)
- Small end diameter (SED) in cm
- Large end diameter (LED) in cm
- Total length in meters to nearest cm
- Length of CWD within transect in meters to nearest cm
- Whether or not the midpoint of the CWD object fell within the belt transect
- Decay class of the CWD object (see Table A1)

Other sampling rules for CWD pieces and logs:

- If a log is partially suspended by other logs or tipped against other trees, measure only the portion of the log that is within 2 m of the ground.
- Pieces that are tipped must have an angle $< 45^\circ$ with the ground to qualify as CWD.
- For logs with their root wad still attached, the large-end diameter is measured just above the butt swell, but the length is taken to extend into the mass of wood within root wad
- When a log is forked or has a very large branch attached to the main bole and both segments intersect the transect, they are tallied as two separate pieces if each meets the required minimum dimensions (Fig. A8). Forked trees are examined to identify one fork as the main bole by measuring both diameters at the fork location. The forked segment with the largest diameter is considered the main bole and the length is measured from the tip of the fork to the end of the log. The smaller segment is recorded as a second piece with a length measured from the fork tip down to the point where the fork joins the main bole.

Table A1. Overview and definitions of five Course Woody Debris (CWD) decay classes based on structure integrity, texture and overall condition.

Decay Class	Structural Integrity	Wood Texture	Condition of branches, twigs, and bark.
1	Sound	Intact, no rot.	Branches, twigs, and needles still attached with tight bark. Log solid.
2	Heartwood sound; sapwood somewhat decayed.	Mostly intact; sapwood soft-starting to decay. Wood cannot be pulled apart by hand.	Branches present, fine twigs or needles gone. Loose or peeling bark, 75-100% remaining.
3	Heartwood sound; log supports its weight. Sapwood decaying.	Large hard pieces of sapwood can be pulled apart by hand.	Branches not present; stubs will not pull out. Bark loose but 50-75% remaining.
4	Heartwood rotten; does not support its weight but shape maintained.	Soft, small blocky pieces can be pulled apart.	Branch stubs pull out easily. 0-50% of bark remaining.
5	No structural integrity. Log circumference flattened.	Soft and powdery when dry.	Bark gone.

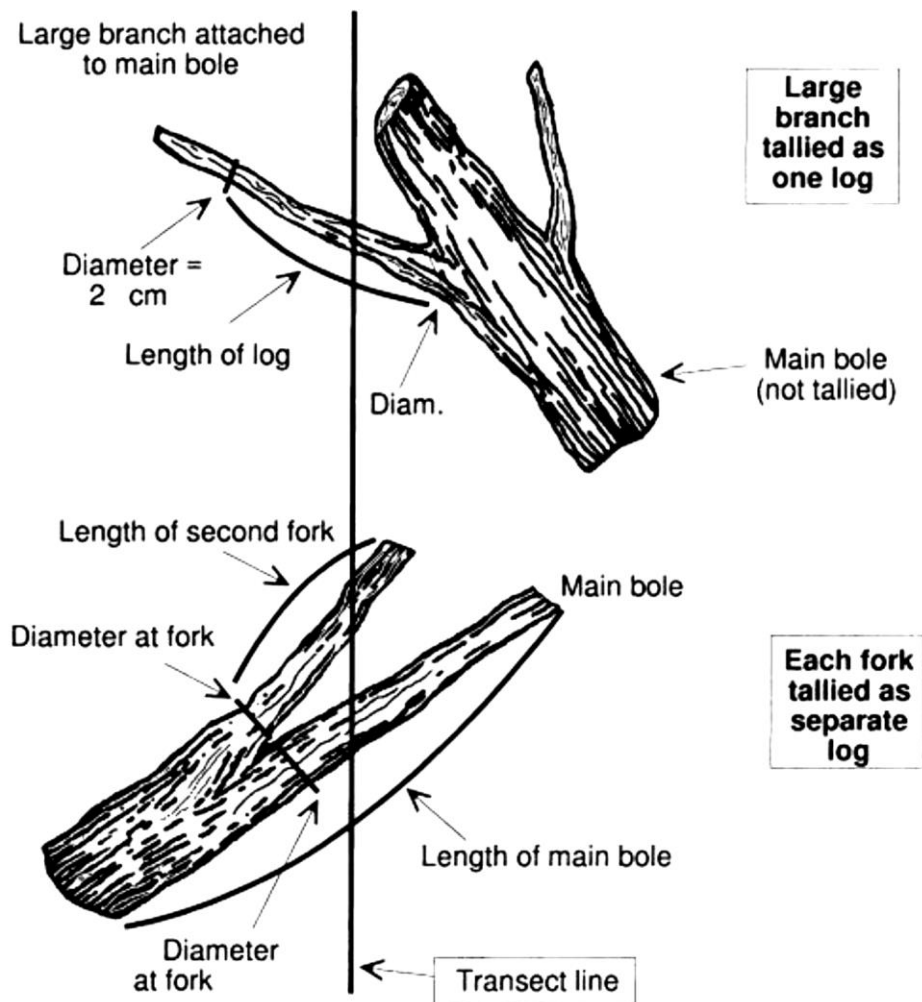


Figure A8. Illustration of how branched logs are counted and measured during CWD assessments.

Table A2. List of the species of conifers, hardwoods, and woody shrubs present in our study area and around den tree structures.

Type and common name	Genus and species	Data code
Conifer trees		
White fir	<i>Abies concolor</i>	ABCO
Incense cedar	<i>Calocedrus decurrens</i>	CADE
Sugar pine	<i>Pinus lambertiana</i>	PILA
Jeffrey pine	<i>Pinus jeffreyi</i>	PIJE
Ponderosa pine	<i>Pinus ponderosa</i>	PIPO
Western white pine	<i>Pinus monticola</i>	PIMO
Douglas fir	<i>Pseudotsuga menziesii</i>	PSME
Giant sequoia	<i>Sequoiadendron giganteum</i>	SEGI
Red fir	<i>Abies magnifica</i>	ABMA
Lodgepole pine	<i>Pinus contorta</i>	PICO
Foothill pine	<i>Pinus sabiniana</i>	PISA
Hardwood trees		
California black oak	<i>Quercus kelloggii</i>	QUKE
Canyon oak	<i>Quercus chrysolepis</i>	QUCH
Interior live oak (uncommon)	<i>Quercus wislizenii</i>	QUWI
Tan oak (uncommon)	<i>Lithocarpus densiflorus</i>	LIDE
White alder	<i>Alnus rhombifolia</i>	ALRH
Mountain dogwood	<i>Cornus nuttallii</i>	CONU
Woody shrubs		
Pine mad manzanita	<i>Arctostaphylos nevadensis</i>	ARNE
Greenleaf manzanita	<i>Arctostaphylos patula</i>	ARPA
Whiteleaf manzanita	<i>Arctostaphylos viscida</i>	ARVI
Birchleaf mountain mahogany	<i>Cercocarpus betuloides</i>	CEBE
Mountain whitethorn	<i>Ceanothus cordulatus</i>	CECO
Buckbrush	<i>Ceanothus cuneatus</i>	CECU
Deerbrush	<i>Ceanothus integerrimus</i>	CEIN
Hazelnut	<i>Corylus cornuta</i>	COCO
Mahala mat	<i>Ceanothus prostratus</i>	CEPR
Mountain misery	<i>Chamaebatia foliolosa</i>	CHFO
Bush chinquapin	<i>Chrysolepis sempervirens</i>	CHSE
Mountain dogwood	<i>Cornus nuttallii</i>	CONU
Bitter cherry	<i>Prunus emarginata</i>	PREM
Huckleberry oak	<i>Quercus vaccinifolia</i>	QUVA
Western azalea	<i>Rhododendron occidentale</i>	RHOC
Sierra gooseberry	<i>Ribes roezlii</i>	RIRO
Currant	<i>Ribes</i> spp.	Ribes
California rose	<i>Rosa californica</i>	ROCA
Thimbleberry	<i>Rubus parviflorus</i>	RUPA
Blackberry	<i>Rubus ursinus</i>	RUUR
Willow	<i>Salix</i> spp.	Salix
Snowberry	<i>Symphoricarpos mollis</i>	SYMO

Details on measuring dbh (diameter at breast height) in different situations:

DBH is outside bark diameter at 4.5 ft above the forest floor on the uphill side of the tree. To determine breast height, the forest floor includes the duff layer that may be present, but does not include unincorporated woody debris that may rise above the ground line. If a snag is missing bark, measure the dbh without the bark and record that measurement.

Forked tree: In order to qualify as a fork, the stem in question must be at least 1/3 the diameter of the main stem and must branch out from the main stem at an angle of 45 degrees or less. Forks originate at the point on the bole where the piths intersect. Forked trees are handled differently depending on whether the fork originates above or below 4.5 feet.

Trees forked below 4.5 ft are treated as distinctly separate trees. DBH is measured for each stem at 4.5 ft above the ground.

Trees forked at or above 4.5 ft count as one tree. If a fork occurs at or immediately above 4.5 ft, measure diameter below the fork just beneath any swelling that would inflate dbh.

Stump sprouts originate between ground level and 4.5 ft on the boles of trees that have died or been cut. Stump sprouts are handled the same as forked trees, with the exception that stump sprouts are not required to be 1/3 the diameter of the dead bole. Stump sprouts originating below 1.0 ft are measured at 4.5 ft from ground line. For multi-stemmed woodland species, treat all new sprouts as part of the same new tree.

Tree with irregularities at dbh: On trees with swellings, bumps, depressions, and branches at DBH, diameter will be measured immediately above the irregularity at the place it ceases to affect normal stem form. If this is not possible, because of the vertical extent of the irregularity, then adjust the DBH measurement to better reflect the diameter of a regular bole.

Tree on slope: Measure diameter at 4.5 ft from the ground along the bole on the uphill side of the tree.

Leaning tree: Measure diameter at 4.5 ft from the ground along the bole.

Independent trees that grow together: If two or more independent stems have grown together at or above the point of DBH, continue to treat them as separate trees.

Missing wood or bark: If 50% or more of the circumference of the bole is intact, reconstruct the diameter at DBH.

Figure A9. Proper use of a diameter tape to obtain diameter at breast height (DBH). Height of the tape is 4.5' measured vertically from the uphill side of the bole. The tape is wrapped perpendicular to lean of tree. Do not place tape at abnormal location on bole of tree.

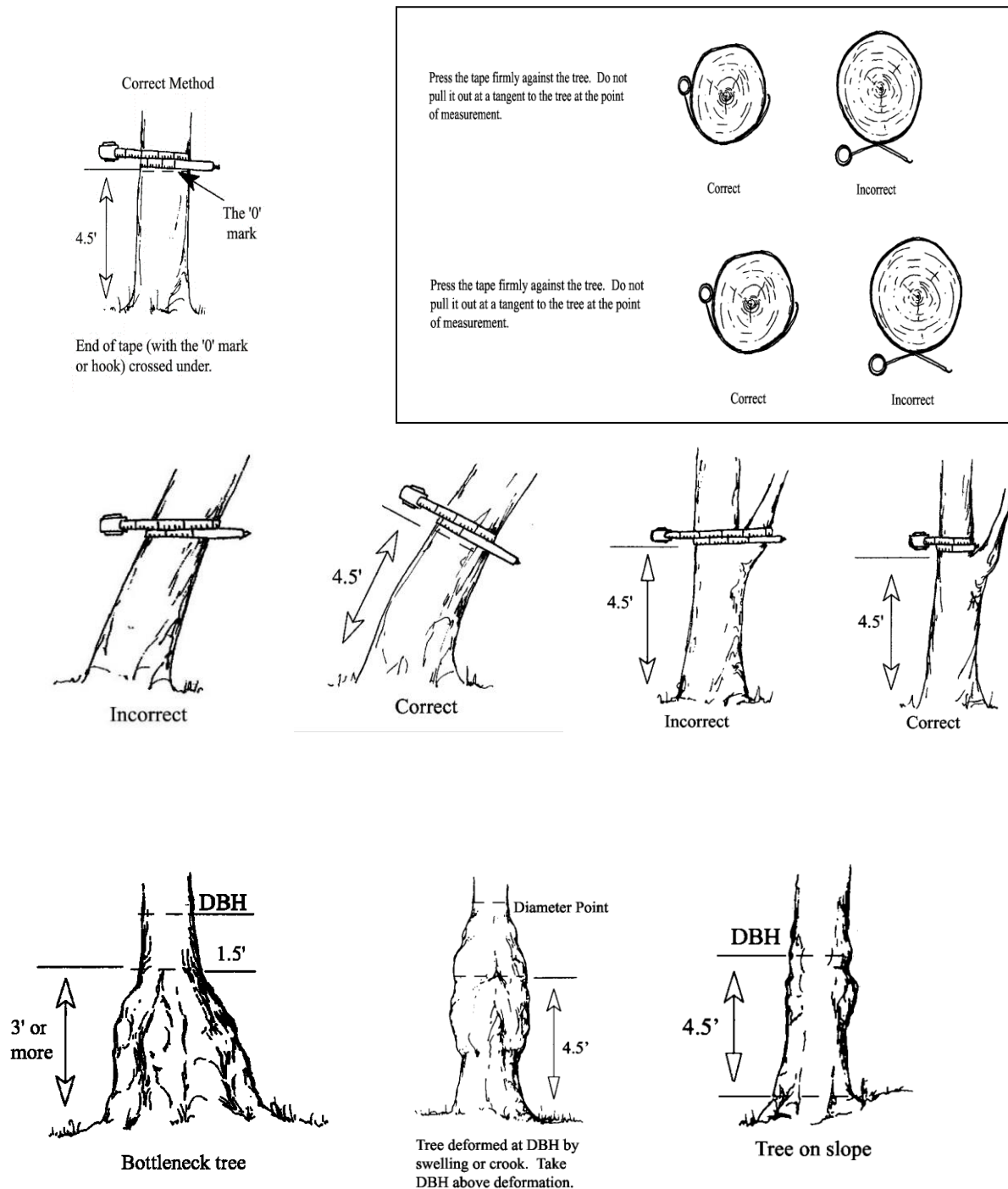
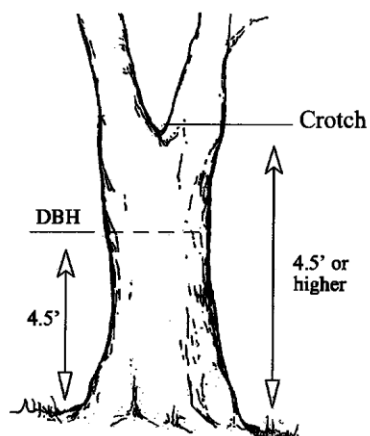
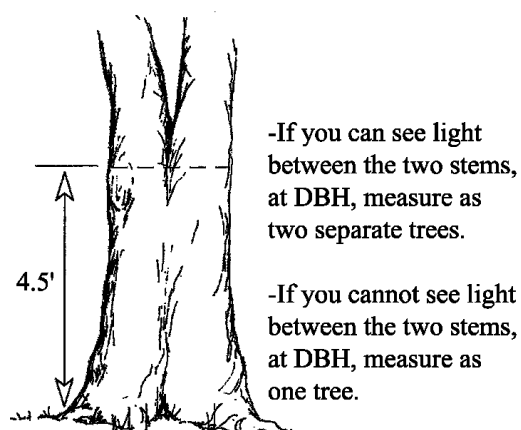
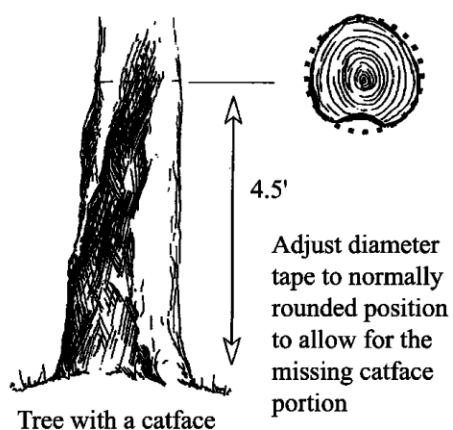
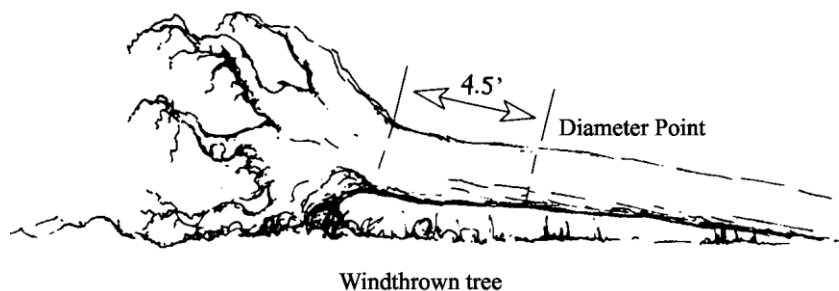
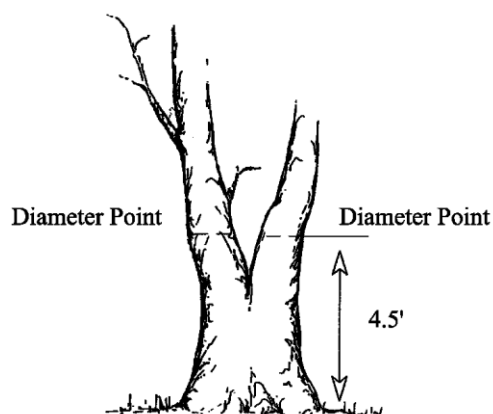


Figure A9 (contd.). Proper use of a diameter tape to obtain diameter at breast height (DBH). Height of the tape is 4.5' measured vertically from the uphill side of the bole. The tape is wrapped perpendicular to lean of tree. Do not place tape at abnormal location on bole of tree.



Tree forked at 4.5 feet or higher. Record as one tree and consider only the main fork. Take DBH below the swell of the fork.



Tree forked below 4.5 feet. Record each fork that is "in" as a separate tree. Measure diameter at 4.5 feet.