

UNDERSTANDING A FISHER REINTRODUCTION
IN NORTHERN CALIFORNIA FROM 2 PERSPECTIVES

Annual Report for 2015

For the period October 2009 to December 2015

By

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to learn whether pertinent publications are now available and, if not, that the information in this report has not be superseded or updated.

Summary

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

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Introduction

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

Because of concern for the status of fishers in California, to understand how fishers in particular, and mammalian predators in general, respond to intensive forest management, and to understand better why some fisher reintroductions have succeeded while others have failed, the California Department of Fish & Wildlife (DFW; formerly Fish & Game), the US Fish & Wildlife Service (FWS), Sierra Pacific Industries (SPI) and North Carolina State University (NCSSU) started collaborating in 2007 to re-establish a fisher population in the northern Sierra Nevada and southern Cascade Mountains of California. In 2009, the California Department of Fish & Wildlife gave final approval for 40 fishers to be reintroduced over 3 years (autumn 2009 – autumn 2011) onto SPI's 648 km² Stirling Management Area (hereafter "Stirling"), which is managed intensively for timber production (Figure 1). The Memorandum of Understanding initiating the research states that released fishers and their progeny are to be studied intensively for the 8 years following the year-1 reintroductions.

In a related effort to understand the fisher population in the far northeastern extent of the fisher's range in California, we began independently in 2006 to use non-invasive methods to estimate population parameters for the fishers living on the managed, forested landscape centered on the Klamath River in northern California and southern Oregon (Figure 1). Combining the non-invasive, genetic surveys conducted in this study area with the research on reintroduced fishers on Stirling provided the opportunity to broaden conservation benefits for fishers (e.g., Seddon et al. 2007, Sarazin and Barbault 1996), to understand better the dynamics of fisher populations on managed landscapes, and to study a source population for a reintroduction. We moved fishers from this study area to Stirling in the winters of 2009-2010 and 2010-2011. These removals were

targeted to lands owned by Michigan-California Timber Company (formerly Timber Products Company), meaning that fishers were removed from managed, industrial timberlands and released on a different but also managed landscape. The removed animals were targeted to be adult members of the population with high reproductive potential.

The objectives of this research are:

- To estimate annual survival and reproduction of fishers on Stirling between 2010 and 2017.
- To evaluate habitat selection by reintroduced fishers and their offspring to test the predictions of available landscape-scale models of habitat quality and suitability for fishers.
- To evaluate fisher diet composition and prey distributions and abundances as a metric of fisher habitat quality.
- To quantify energy expenditure, energy balance, and overall body condition of fishers and relate these metrics to habitat quality and fisher conservation.
- To genotype genetic samples collected from reintroduced fishers and their offspring.
- To identify aspects of habitat associated with, and to test functional models for, natal dens, maternal dens and rest sites for fishers.
- To quantify disease prevalence and exposure in translocated fishers and their offspring to determine the influence of disease on short and long-term persistence on the landscape.
- To predict the placement, sizes, and shapes of home ranges of reintroduced fishers and their offspring using models of optimal home range choice and to test those predictions using data on actual use of space by those fishers.
- To predict patterns of breeding by Stirling males from home range placement and familiarity with landscapes and to test those predictions using data on paternity of fishers born in the study area.
- To evaluate the accuracy, precision and efficacy of a long-term fisher monitoring protocol during fall survey efforts 2013-2016.
- To estimate abundance, survival and recruitment, population growth rate and occupancy for the source population of fishers through 2016.
- To estimate the effects on abundance and population growth rate, if any, caused by removing 9 adult fishers from a source population (an estimated 17% of the population) in 2009-2010 for release on Stirling.
- To evaluate the original non-invasive study design, redesign the monitoring protocol as necessary, and test the redesigned protocol for use as a monitoring tool for the reintroduced fisher population on Stirling.
- To investigate the effects of intensive forest management for timber production and fire and associated salvage operations on fisher population dynamics.

Here we report on research activities that address these goals directly for year-6, January– December 2015 of the project. We review non-invasive research in the Klamath Region and the reintroduction activities to date.

Terms and Definitions

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

General condition, disease, and ectoparasites

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

Of 197 captures of fishers on Stirling through December 2015 where we evaluated the fishers, including recaptures of reintroduced fishers and captures of fishers born-on-site, we have graded none as being in poor condition. We have graded 23 (12%) as showing below average condition, 94 (48%) as average, and 80 (40%) as above average or excellent. The

average body condition may change through time on Stirling and, though the condition of animals to date gives us no cause for concern, we advocate continued monitoring of overall health and condition for as long as feasible.

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

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

Locations, movements and home ranges

The responses of fishers to being released onto Stirling, specifically their site fidelity after release, is an important measure of how those fishers perceived their environment and its habitat quality upon release (Berger-Tal and Saltz 2014). We have noted in previous reports that some released fishers did wander, or explore, and at times did settle into areas off the district (Powell et al. 2012). As of 2015, the majority of locations of fishers have occurred within the boundaries of Stirling or very near to it (Figure 3). Similarly, most den locations have occurred on Stirling. Because the majority of our research effort occurs on Stirling (Figure 4), these data are not representative of all fishers in the reintroduced population. We know that some fishers live on adjacent lands. Nonetheless, a minimum of 40 fishers remain on Stirling annually, representing a core population. Consequently, our data show that some fishers have found enough habitat of sufficient quality for them to stay on the study area.

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

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

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

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

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

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

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

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

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

We have attempted to monitor fishers during all times of day and night to ensure that our information is not biased to one time period. VHF transmitters are more difficult to locate at night, particularly in the winter when temperature, weather and road conditions hinder access to the study area. Thus, the vast majority of VHF telemetry locations have been collected during daylight hours (8 am to 4 pm; Figure 5). We have programmed Argos collars to be located during different times of day, leading the distribution of locations of fishers wearing those collars to be relatively even across all times. We programmed GPS collars to locate themselves across all times of day, leading to a very even distributions of locations.

We are collecting enough location data to estimate annual home ranges for most fishers. Thirty locations represent a reasonable minimum sample size for estimating annual home ranges with fixed-kernel methods, though having more locations is preferable (Fieberg and Börger 2012, Seaman et al. 1999, Noel 1993, Seaman and Powell 1996). We have more than 100 estimated locations per year for many fishers.

We define an animal's home range to be that part of the landscape in which it lives that it maintains updated within its cognitive map of the landscape (Powell and Mitchell 2012). For this report, for logistic reasons, we assume that 95% utilization distributions for fishers' use of space provide reasonable estimates of home ranges. We have estimated utilization distributions using a fixed kernel smoothing program. Such programs smooth data using a kernel and a smoothing parameter, "h", whose values are, ideally, related to aspects of the biology or management goals for the animals being studied. Silverman's (1990) kernel "k2" is a bell-shaped kernel with finite bounds, is leptokurtotic and, therefore, resembles the distribution of telemetry error for experienced researchers; we use "k2". Many researchers choose "h" to minimize internal error within a distribution

of location estimates, and we have advocated this approach in the past (Seaman and Powell 1996, Powell 2000). Such choice of "h", however, ignores the biology of the animals studied, chooses different values for "h" for different animals, and even for different random samples from a single data set, making comparisons between studies nearly impossible. For fishers, different values of "h" provide insight into different aspects of their biology. For our fishers, $h=750$ m appears to estimate reasonably well the probability of where a researcher will be able to find a given fisher using telemetry. Average daily movements of fishers suggest that 1500 m should estimate where a fisher can travel over the coming day. Average distances across distributions of location estimates suggest that 1000 m will estimate the overall range of space a fisher uses but not its small scale preferences. Values of "h" tailored to match the estimated error for each location estimate should provide the best estimates of fishers' habitat preferences. Table 3 shows mean estimates for 95% utilization distributions for 2010-2014 using $h = 750, 1000$ and 1500.

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

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

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

Population monitoring on Stirling

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

many fishers as possible and to outfit or re-outfit these fishers with functional transmitters. We spread our trapping effort across Stirling and adjacent lands focusing on areas where fishers were known to live, had been previously detected, or areas we considered likely to have fishers (Figure 4). To maximize efficiency, we split the study area into east and west of Butte Creek. We trapped the east side for 14 trap days (12 October - 25 October), then moved to the west side (26 October - 09 November) for 14 days. Logistical constraints precluded or curtailed trapping in some areas we thought may have resident fishers.

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

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

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

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

At the conclusion of trapping in 2015, the age structure of the known fishers on Stirling emphasized young fishers (Figure 6). Fishers < 2 years old comprised 50% of all fishers known to be alive at the end of 2015. Many fishers older than 2 years of age are still in the population but the young age structure suggests healthy reproduction and recruitment. The age distribution in Figure 6 is our best estimate of the true age distribution of the Stirling fisher population but is accurate only to the extent that our trapping results were representative for the population. We do not know why so few 1-year old fishers appear in Figure 6. Survival for that age group could have been low in 2015 but our survival analyses do not show low survival for that age class (Figure 7). Alternately, and more likely, we simply captured few 1-year old fishers during trapping in 2015.

At the end of our trapping effort in autumn, 2015, the minimum known population size for the fishers on Stirling was 50 (total fishers captured + non-captured fishers still wearing functional transmitter collars). We have retrospectively adjusted the minimum population sizes for previous years, accounting for fishers that were not known to be alive in those years but that since been captured, showing that they must have been alive in those past years (Figure 8). The minimum number alive values suggest a slight decrease in the population size in 2013, though less severe than estimated during that year, with a probable rebound in 2014 and into 2015. Calculations of the minimum number of females alive indicate that the female population size has been relatively stable or growing slightly since the final releases of fishers in 2011 (Figure 8). Thus, the observed decrease in minimum number alive size during 2013 appears to be related to changes in the number of adult males. Consistent with these numbers, we observed a relatively high number of male mortalities during 2013 and early 2014 ($n=5$).

Survival

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

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

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

The highest ranked hypothesis included the Age and Reproductive season with the sexes modelled as not equal (Table 6). The second highest ranked hypothesis included differences among ages and between sexes. Monthly survival estimates for females and males during reproduction 0.96 (95% CI = 0.93-0.98). These differences in monthly survival translate to relatively large declines in annual survival (Figure 9). During all other periods, estimated monthly survival rates were higher for both sexes (0.99; 0.97-0.99). We estimate that the annual survival rate for adult fishers, including breeding and non-breeding times of years, is 0.78 (0.53-0.81). This estimate of annual survival is marginally smaller than last years’ high estimate of 0.80. The pattern of survival has been consistent since we first released fishers and indicates that the time of year, and potentially the reproductive status of females and males, explains fisher survival better than other models we have explored. Additionally, in previous years we did not model age explicitly but age was an important variable in this year’s analysis. Strikingly, survival of adults is lower than that of both juveniles and 1-year old fishers (Figure 7). Perhaps, this occurs because juvenile survival for this analysis can be estimated only from the time of capture in the fall through to their first birthday and 1-year olds only remain in that age class for a single year. Old adults eventually die and this likely accounts for the relatively lower rate of estimated survival. Changes in the estimates of survival are not large through time but do change because of sample size and simple variation rather than an indication of a true change in survival. Annual survival does not show strong changes through time (year to year; Figure 9). Additional survival analyses with varied data sets where we used individual covariates such as weight were not strongly supported (Facka et al. In prep). Further, age and reproductive status were relatively well supported and indicate these variables are consistent across different analyses. Future analyses of survival will include covariates based on home range composition and estimates of habitat quality.

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

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

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

To date, a total of 26 carcasses (16 Translocates, 9 Stirling-bom, and 1 translocate candidate that died while in captivity prior to release in year-1) have been submitted for examination. Four of these carcasses were submitted in 2015 (2 translocated fishers and 2 Stirling-bom fishers). All but 2 carcasses (partial remains only submitted in 2015) have been examined. A complete set of formalin-fixed tissues from 4 fishers (2 translocated fishers, 1 Stirling-bom fisher, 1 candidate for translocation) that had partial remains. The remaining 14 carcasses examined were either too decayed or lacked tissues for histologic examination. In addition, liver samples from 5 and a muscle sample from 1 of the 24 necropsied carcasses

were tested for the presence of 7 different anticoagulant rodenticide compounds (ARs). The remaining 13 carcasses were not tested for ARs due to lack of suitable tissue for testing. Predation forensic analysis has been attempted on 6 carcasses; 5 partial carcasses are awaiting forensic testing pending fund availability.

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

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

brodifacoum and bromadiolone. Results for the fourth fisher (a year-2 female found in a river with unclear cause of death in 2015) showed a trace amount of brodifacoum. For the 3 reintroduced fishers, anticoagulant exposure could have occurred pre-capture or post-release, as the half-life of these 2 second generation anticoagulants in liver tissues is >150 days (Vandenbroucke et al, 2008). In contrast, for the fisher that died in 2015, positive result for exposure to anticoagulant rodenticides indicates exposure on the Stirling study area because she lived her entire life on the study area.

Anticoagulant rodenticides were not detected in the a Stirling-born fisher that died in April 2013 and had only muscle available for testing. Since anticoagulant rodenticides are stored long-term primarily in the liver, exposure cannot be definitively ruled out for this fisher.

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

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

Reproduction

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

Fishers on Stirling have denned and given birth at times similar to denning by fishers elsewhere (Powell 1993). Natal dens (dens in which females give birth) are most often found during the final two weeks in March or first week in April (mean = week 13.3; Figure 10A), with the earliest den found on 17 March and the last found 19 April. Because a female must localize movements before we even look for a den tree, our identification of den trees comes several days, maybe even weeks, after a female has given birth and thus our denning dates are biased late. We note in Figure 10B that we have generally found dens earlier and earlier each year. This trend may reflect our inexperience in finding den trees under difficult conditions during those early years (e.g., high snow fall, downed trees). Alternately, all females in 2010 and most in 2011 had recently been released to our study site, potentially causing the females that had been moved to give birth late. One of our early findings from this project is that the time

when we released females influences their probabilities of giving birth (Facka et al. 2016). Translocation may have also caused females to delay births. Females move their kits to maternal dens throughout the spring and summer with highly variable timing and without a pattern (Figure 10B). Some females never move their kits, which is not shown in Figure 10B. Though we attempt to locate females and their kits throughout the summer we consider the denning season to be effectively concluded by the end of June (week 27) and most females move kits often to rest trees that they use while foraging.

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

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

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

We have documented females denning across Stirling, on other private lands and on national forest lands (Figure 11). Of 148 natal and maternal den trees that we found in 2010-2015 (Table 8), black oaks (*Quercus kelloggii*) were most common for both natal and maternal dens (49%, Table 8). Female fishers used Douglas-firs (*Pseudotsuga menziesii*), incense-cedars (*Calocedrus decurrens*) and white firs (*Abies concolor*) in similar numbers (10%, 11% and 9%), and used other trees less commonly. Female fishers used live trees (35 of 46 dens) most often as natal dens but, later in the denning season, as kits began to travel with their mothers, females used snags slightly more often than live trees for maternal dens and even used hollow logs and piles of debris as dens or rest sites (48 live trees and 54 snags, logs, and debris). In 2010-2012, SPI committed

resources to collect data on vegetative and topographic characteristics within 90 m of den sites. Future analyses will examine patterns of female denning and movements (locations and timing) relative to topography (temperature related movements), time of year, predators and other factors that might influence female decisions to establish and move dens.

Population Viability Analysis

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

Habitat Relationships

During 2015 we distributed to all cooperators a report entitled "Fisher Habitat Selection on Stirling Management District 2010-2014: A Critical Test of Our Understanding of Fisher Habitat Needs". That report is appended to this Annual Report for 2015 as Appendix 2 and we refer readers to that report.

Food Habits and Prey Population Dynamics

During 2015 we distributed to all cooperators on this project a report entitled "Fisher (*Pekania pennanti*) prey availability and habitat use on managed timberlands in Northern Sierra Nevada". That report is appended to this Annual Report for 2015 as Appendix 3 and we refer readers to that report.

Non-invasive Sampling of Klamath Fishers

We monitored fishers in a 587 km² portion of the Klamath-Siskiyou ecoregion in northern California and southern Oregon (henceforth, "the Klamath"; Figure 12). The Klamath was predominantly comprised of conifer and mixed conifer/broadleaf forest. Elevation ranged from 472 to 2269 m. Non-invasive surveys of meso-carnivores have taken place in the Klamath annually since 2006 between mid-September and early December. In the summer of 2014, two wildfires burned areas on and adjacent to the Klamath (Figure 12): the Happy Camp Complex and the Beaver Fires. We have 8 years of data before the fires occurred, and at this point, 2 years of data following the fires. One objective of the current analysis is to determine the status of the population before these fires occurred. In the winters of 2009 and 2010, the fisher population in the Klamath also served as a source for a reintroduction of fishers to the northern Sierra Nevada and northern Cascade Mountains (Callas & Figura 2008). A total of 9 fishers (4F, 1M in 2009 and 3F, 1M in 2010) were removed, and of these, 8 were translocated to an area in the

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

Non-invasive sampling techniques and individual identifications with genetics

We attempted to deploy 100 survey stations in the Klamath at the same locations near streams and on ridge tops each year (Figure 12). At each survey station we placed a non-invasive sampling box made of corrugated plastic (25x25x75 cm). Each box was baited with a raw chicken drumstick and a can of wet cat food. The back of the sampling box was closed with hardware cloth, and the front was partially obstructed with 3 wooden slats. We fixed a glue strip to the underside of the bottom wooden slat so that a mammal coming into the box to obtain the bait was likely to leave a hair sample attached to the glue strip.

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

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

Spatial capture-recapture model

We developed a spatial capture-recapture model to investigate the population demography of fishers in the Klamath. In spatial capture-recapture models, the spatially-explicit locations of each individual i are assumed to be a

function of the distance to their latent activity center s_i (Efford 2004; Royle & Young 2008). Locations of s_i are then calculated by estimating individual encounter probabilities and the distribution of activity centers in the defined state-space (S).

It is important for S to be delineated as an area large enough to identify s_i for all individuals, even those where s_i is located outside of the vicinity of the study area. Home ranges of fishers in the Klamath have been documented to be (mean \pm SE) 9.6 ± 2.8 and 30.6 ± 8.6 km² for females and males, respectively (summarized from Lofroth et al. 2010). In the current research, S was defined as a 10-km buffer encompassing our sampling units (1836 km² in total) to accurately identify the locations of all activity centers of fishers. The locations of activity centers in the current research were modeled as an inhomogeneous Poisson point process in S (Royle et al. 2014). We divided S into a 1-km x 1-km grid and the location of each sampling unit was then assigned to the grid cell where it was located, and the probability of s_i in year t ($z(i, t)$) being at the center of grid cell j (prob $_j$) was modeled as an intensity function.

We used a Bayesian approach and fit our models using data augmentation (Royle et al. 2007). We introduced a sufficiently large number of all-zero encounter histories (m_i^0) to our population of observed individuals (n) to prevent any truncation of the number of individual fishers with activity centers located in S . We also introduced the partially latent variable $z(i, t)$ indicating population membership for observed or unobserved individual i in year t equal to M ($M = m_i^0 + n$). We set $z(i, t) = 1$ with certainty for individuals sampled in a given year, and estimated this parameter for all remaining years. For all years $z(i, t) \sim \text{Bernoulli}(\Psi)$, and the number of individuals N alive in year t was thus:

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

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

$$\log(p_{jt}) = \beta_0 + \beta_1 \cdot \text{sex}_i + \beta_2 \cdot \text{previous detection}_i$$

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

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

$$g_j = e^{(-d_{ij}^2 / 2\sigma^2)}$$

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

Model fitting and assessment

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

Results

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

The spatial capture-recapture model indicated the population of fishers in the Klamath was relatively stable

before the fires occurred and for the three years immediately following the removal of fishers for translocations (Figure 13; Table 9). Estimated locations of fisher activity centers varied each year on the landscape (Figure 14).

Current conclusions and future directions

The current modeling efforts indicate the population of fishers in the Klamath was relatively stable from 2006 to 2013. The abundance estimates are unchanged among years, with no statistically significant differences (95% credible intervals overlap; Figure 13, Table 9). The removals of 5 fishers in 2009 and 4 fishers in 2010 appear to go undetected; our results show that removing this quantity of fishers for translocation did not alter the abundance or density of fishers in the Klamath in the short-term.

Our estimates of fisher abundance in the Klamath are slightly smaller than, but do not differ significantly from, previous estimates (Swiers 2013). Both our current results and previous work indicate a stable population of fishers in the Klamath before the wildfires occurred in 2014, and for up to 3 years following the translocation efforts. Two major benefits to the current modeling approach over previous analytical techniques include more precise estimates of population sizes and spatially explicit densities.

Non-Invasive Sampling on Stirling

In the current report we present preliminary results from the non-invasive dataset accumulated in the fall 2013 field season. We monitored fishers in a 229 km² portion of the Stirling Management Area in this initial year of non-invasive sampling (Figure 15). The non-invasive sampling protocol designed for Stirling consisted of 16 sampling units (10.4 km²) with up to 3 station replicates placed within each sampling unit. Our sampling protocol was later expanded to a total of 27 sampling units and 78 survey stations in 2014 and 2015 (see previous annual report for further information).

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

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

the landscape, but their density was highest in the southwestern portion of the study area (Figure 15). The probability of detection non-significantly differed between the sexes; males and females had a 0.40 [0.12, 0.90] and 0.34 [0.17, 0.78] probability of being detected at a sampling unit at the center of their home range, respectively.

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

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Publications Related To Project

- Facka, A.N., J.C. Lewis, P. Happe, K. Jenkins, R. Callas, and R.A. Powell. *In review*: Timing of translocation influences birth rate and population dynamics in a forest carnivore. *Ecosphere* 7(1):e01223.10.1002/ecs2.1223
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- Clifford, D., L. Woods, V. Tkach, E. Hoberg, R. Callas, R. N. Brown, J. M. Higley, K. Haynes and M. W. Gabriel. 2012. Assessing disease risk from a novel parasite infection in Pacific fisher (*Martes pennanti*). The Western Section of The Wildlife Society 2012 Annual Conference, Sacramento, CA.
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Table 1. Mean numbers (+SD, N) of estimated locations per individual fisher per year across all years of study and 2014 organized by location method. Means are for individual fishers who were followed using each particular method. GPS collars were not used on females prior to 2015. The research was conducted on or near the Stirling Management area owned by Sierra Pacific Industries and located in the Northern Sierra and Southern Cascade Mountains of Northern California.

Sex	Year	All Locations	Triangulations	Walk ins	GPS	All Argos	Argos LC 2+3
Female	All Years	66 ± 95, 131	52 ± 49, 128	6 ± 4, 86	543 ± 152, 2		
	2015	138 ± 172, 29	96 ± 66, 29	5 ± 3, 19	543 ± 152, 2		
Male	All Years	262 ± 456, 69	14 ± 18, 17	2 ± 2, 11	734 ± 1073, 7	215 ± 237, 59	48 ± 21, 59
	2015	310 ± 402, 9	3 ± 0, 2		1332 ± 0, 1	181 ± 126, 8	24 ± 61, 8

Table 2. Classes for Argos locations of male fishers, error predicted by Argos services for locations in those classes, our mean observed error, standard deviation (St Dev), minimum error observed, the maximum error observed and the total number of location estimates for each location class across all years (2009-2012) on the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California. Data are from 17 tags at 26 locations.

Location Class	Predicted error	Mean (m)	Standard Deviation	Minimum (m)	Maximum (m)	n
3	<250 m	196	248	10	2482	431
2	250 – 500 m	458	461	10	3630	242
1	500 – 1500 m	1387	1227	34	6439	123
0	>1500m	2566	1730	58	7055	30
A	none	811	1128	10	6061	192
B	none	1289	1788	17	8744	349

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Table 3. Mean areas (\pm SD) for 95% fixed kernel utilization distributions (UD) of fishers followed with telemetry for \geq 6 months on Stirling in 2011-2015 using different smoothing parameters and Silverman's K2. All females followed were adults but 3 males in 2012 and 1 in 2014 were juveniles.

Smoothing parameter (m)	Year	Mean UD + SD (km ²), N	
		Females	Males
750	2010	17 \pm 7, 6	67 \pm 6, 3
	2011	28 \pm 6, 7	114 \pm 20, 3
	2012	17 \pm 5, 12	56 \pm 45, 9
	2013	15 \pm 5, 13	46 \pm 27, 3
	2014	16 \pm 4, 13	63 \pm 33, 3
	2015	15 \pm 3, 19	40 \pm 9, 3
1000	2010	22 \pm 8, 6	97 \pm 7, 3
	2011	37 \pm 8, 7	143 \pm 24, 3
	2012	22 \pm 8, 12	75 \pm 59, 9
	2013	18 \pm 6, 13	63 \pm 33, 3
	2014	19 \pm 5, 13	77 \pm 41, 3
	2015	18 \pm 4, 19	57 \pm 15, 3
1500	2010	32 \pm 12, 6	153 \pm 34, 3
	2011	56 \pm 11, 7	189 \pm 30, 3
	2012	30 \pm 14, 12	108 \pm 84, 9
	2013	24 \pm 8, 13	94 \pm 45, 3
	2014	25 \pm 7, 13	100 \pm 53, 3
	2015	23 \pm 5, 19	88 \pm 27, 3

Table 4. Total number of trap nights, fisher captures, individuals, females, males, new fishers, and capture percentage during fall trapping of years 2012-2015 on the Stirling district of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California.

Year	Trap Nights	# Total Fisher Captures	# Individuals Captures	Total Females	Total Males	# New Fishers Captured	Capture %
2012	2279	43	29	17	12	14	1.89%
2013	3172	34	22	17	5	8	1.07%
2014	2792	53	32	23	9	15	1.90%
2015	2865	84	46	33	13	25	2.93%

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Table 5. Number and percentage of total non-target carnivores captured during fall trapping of 2015 on the Stirling district of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California.

Species	Number	Percent of non-Targets
Ringtail (<i>Bassariscus astutus</i>)	22	15.5%
Opossum (<i>Didelphis virginianus</i>)	14	9.9%
Striped Skunk (<i>Mephitis mephitis</i>)	1	0.7%
Raccoon (<i>Procyon lotor</i>)	1	0.7%
Spotted Skunk (<i>Spilogale gracilis</i>)	50	35.2%
Grey Fox (<i>Urocyon cinereoargenteus</i>)	54	38.0%
Total	142	

Table 6. Model selection comparison for 14 models of survival from a known fates analysis in program MARK based on monthly fates of reintroduced fishers and their offspring in the Northern Sierra Nevada of California, December 2009 – December 2015.

Model	AICc	Δ AICc	w	likelihood	K	Deviance
Age + Reproductive Season ^a	231.304	0.000	0.543	1.000	4	136.913
Age \times Sex	233.730	2.426	0.161	0.297	6	135.306
Age	234.938	3.634	0.088	0.163	3	142.559
Age + Reproductive Season ^b	235.346	4.042	0.072	0.133	8	132.878
Control (Null)	235.382	4.079	0.071	0.130	1	147.018
Sex	236.892	5.589	0.033	0.061	2	146.522
Age by Year	237.427	6.123	0.025	0.047	6	139.004
Sex \times Year	241.819	10.516	0.003	0.005	14	127.146
Year	241.998	10.694	0.003	0.005	7	141.554
Month	244.197	12.893	0.001	0.002	12	133.604
Sex \times Month	249.243	17.939	0.000	0.000	24	113.984
Sex \times Age	252.096	20.792	0.000	0.000	22	120.979
Month \times Year	328.299	96.995	0.000	0.000	73	87.555
Sex \times Month \times Year	465.296	233.992	0.000	0.000	145	54.144

^aSexes Unequal

^bSexes Equal

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Table 7. The number of females that were radio-tracked, the number that denned, the percent of females that denned, the minimum number of kits known to have been produced (Min # kits), the mean minimum litter size (Litter Size \pm 95% CI), the ratio of kits known to have been produced to females (Kits/Female), the number of natal dens found, and the number of maternal dens found for females tracked in 2010-2015 on the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade Mountains of northern California.

Metric	2010	2011	2012	2013	2014	2015	Total
Females	8	9	10	11	7	18	63
Females denned	5	7	9	9	6	14	50
% Denned	63%	78%	90%	82%	86%	78%	79%
Min # kits	4	13	14	17	8	21	77
Kits in fall	1	8	17	13	12	21	72
Died in Den	2	1	2	1	2	1	9
Kits Died Den	2	3	3	2	2	1	13
Litter Size	1	2.2	1.8	1.9	1.6	1.9	1.7
Juvenile Spring:Fall	0.25	0.62	1.21	0.76	1.50	1.00	0.9
Kits/Female	0.5	1.4	1.4	1.5	1.1	1.2	1.2
Natal Dens	5	7	9	9	2	14	46
Maternal Dens	23	13	19	16	1	30	102

Table 8. Numbers of den trees by species for natal and maternal dens from 2010 to 2015, and by condition of the den tree (live tree, standing snag, or other [e.g., downed log or debris pile]) on the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California.

Tree Species	Natal			Maternal			Total
	Live tree	Snag	Other	Live tree	Snag	Other	
Big Leaf Maple (<i>Acer macrophyllum</i>)	0	0	0	0	1	0	1
Black oak (<i>Quercus kelloggii</i>)	23	1	0	35	12	1	72
Douglas Fir (<i>Pseudotsuga menziesii</i>)	1	1	0	3	10	0	15
Incense Cedar (<i>Calocedrus decurrens</i>)	2	4	0	3	7	0	16
Canyon live oak (<i>Quescus chrysolepis</i>)	0	0	0	1	0	0	1
Multiple trees	0	0	0	1	0	0	1
Ponderosa Pine (<i>Pinus ponderosa</i>)	1	2	0	2	1	2	8
Sugar Pine (<i>Pinus lambertiana</i>)	0	1	0	1	1	0	3
Tanoak (<i>Notholithocarpus densiflorus</i>)	4	0	0	1	2	0	7
White Fir (<i>Abies concolor</i>)	3	1	0	1	5	0	10
Unidentified Conifer	1	1	0	0	9	3	14
Total	35	11	0	48	48	6	148

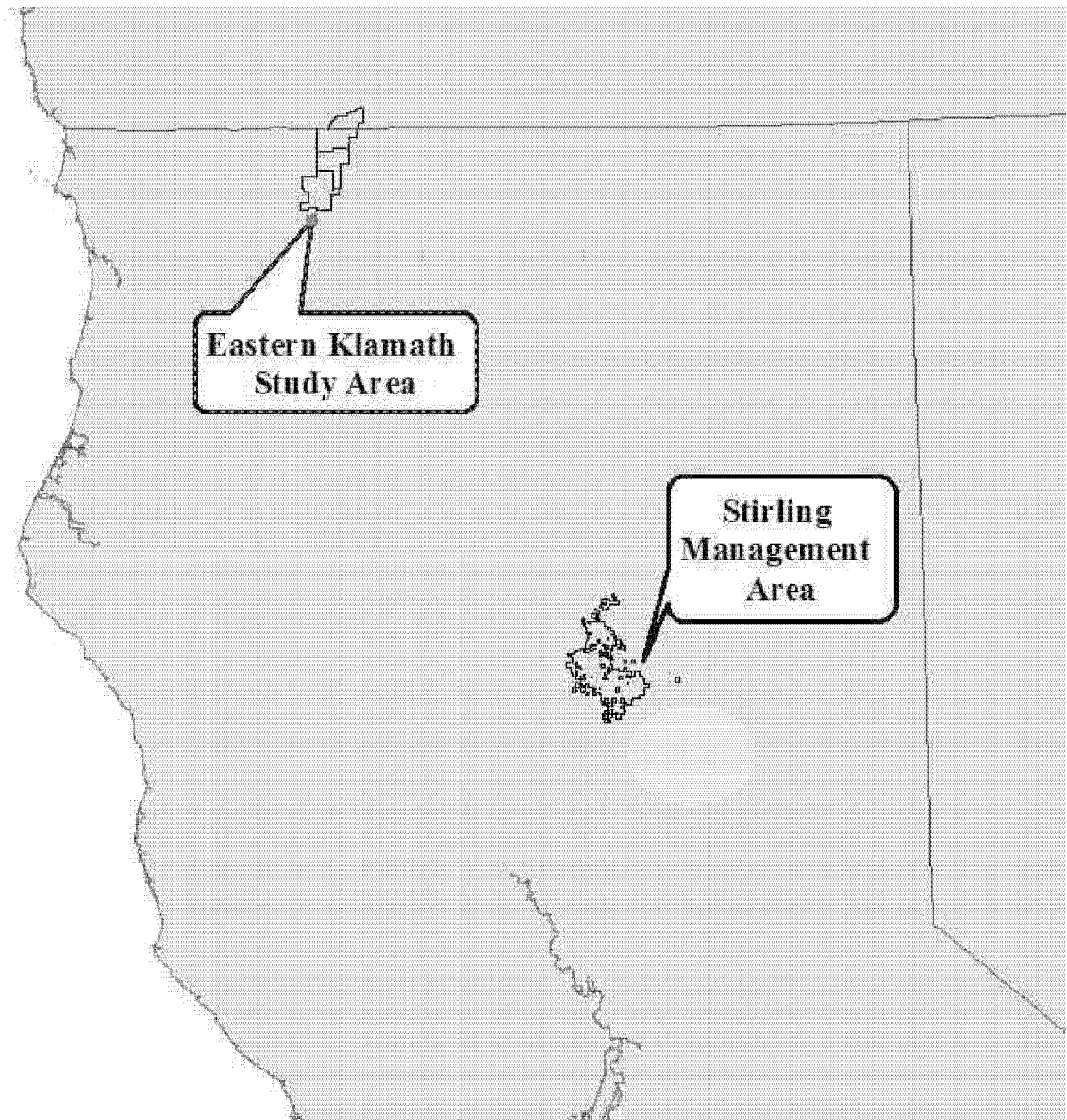
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Table 9 – Non-invasive. Derived posterior parameter estimates of annual population density, abundance, and population growth of fishers in the Klamath. Parameters are presented as median [95% credible interval].

Year	Density (fishers/100 km²)	Abundance	Lambda
2006	6.64 [4.94, 8.35]	39 [29, 49]	-
2007	6.64 [4.94, 8.18]	39 [29, 48]	1 [0.71, 1.35]
2008	6.99 [5.62, 8.69]	41 [32, 50]	1.06 [0.78, 1.4]
2009	6.47 [5.11, 8.18]	38 [29, 47]	0.92 [0.67, 1.2]
2010	5.79 [4.43, 7.33]	34 [26, 43]	0.91 [0.64, 1.21]
2011	6.47 [5.11, 8.18]	38 [28, 46]	1.09 [0.78, 1.45]
2012	6.3 [4.94, 8.18]	37 [27, 46]	0.98 [0.72, 1.33]
2013	6.99 [5.62, 8.69]	41 [32, 50]	1.11 [0.81, 1.49]

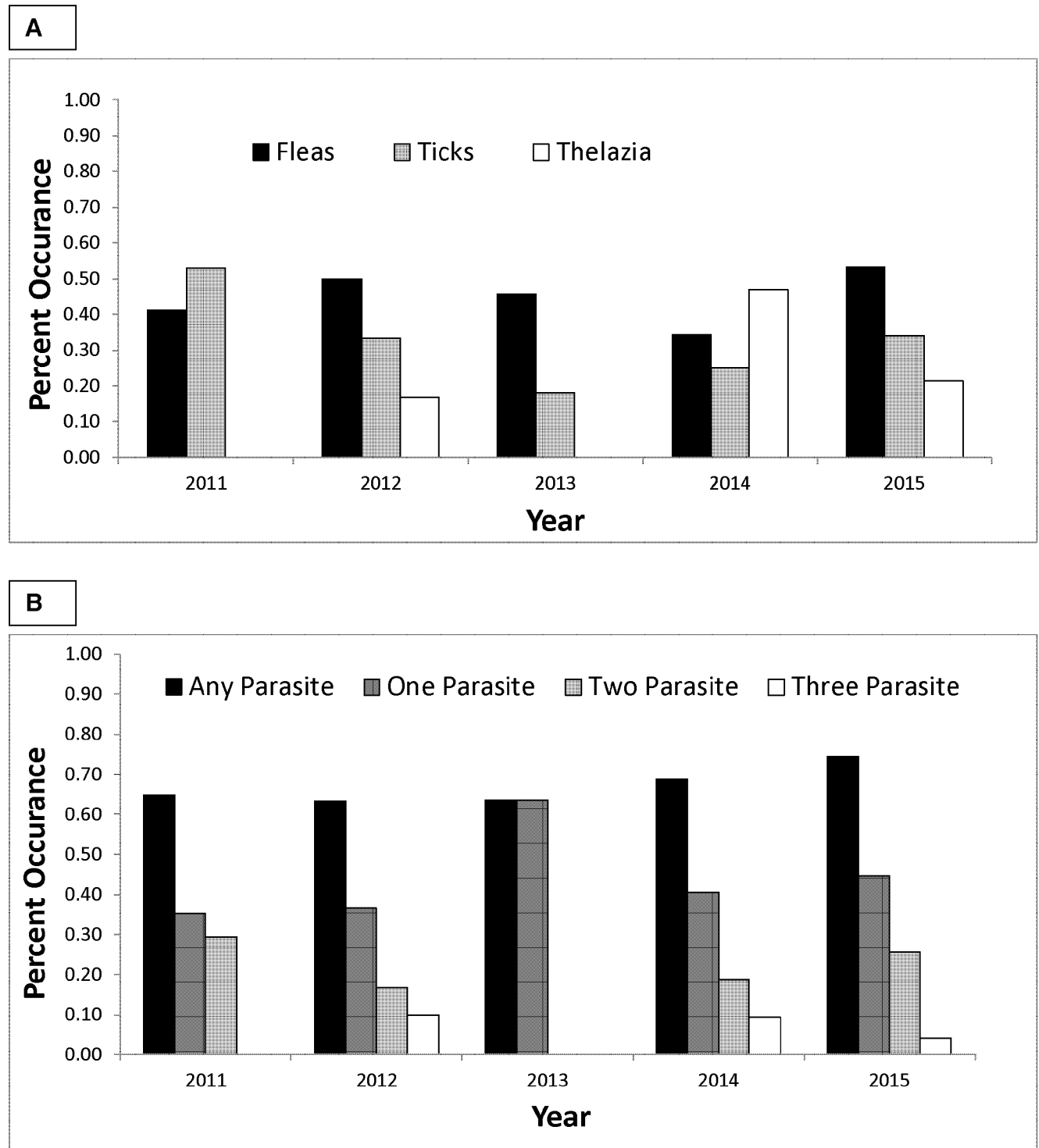
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Figure 1. Locations in northern California of the Stirling Management Area of Sierra Pacific Industries and the Eastern Klamath Study Area on the California-Oregon border.



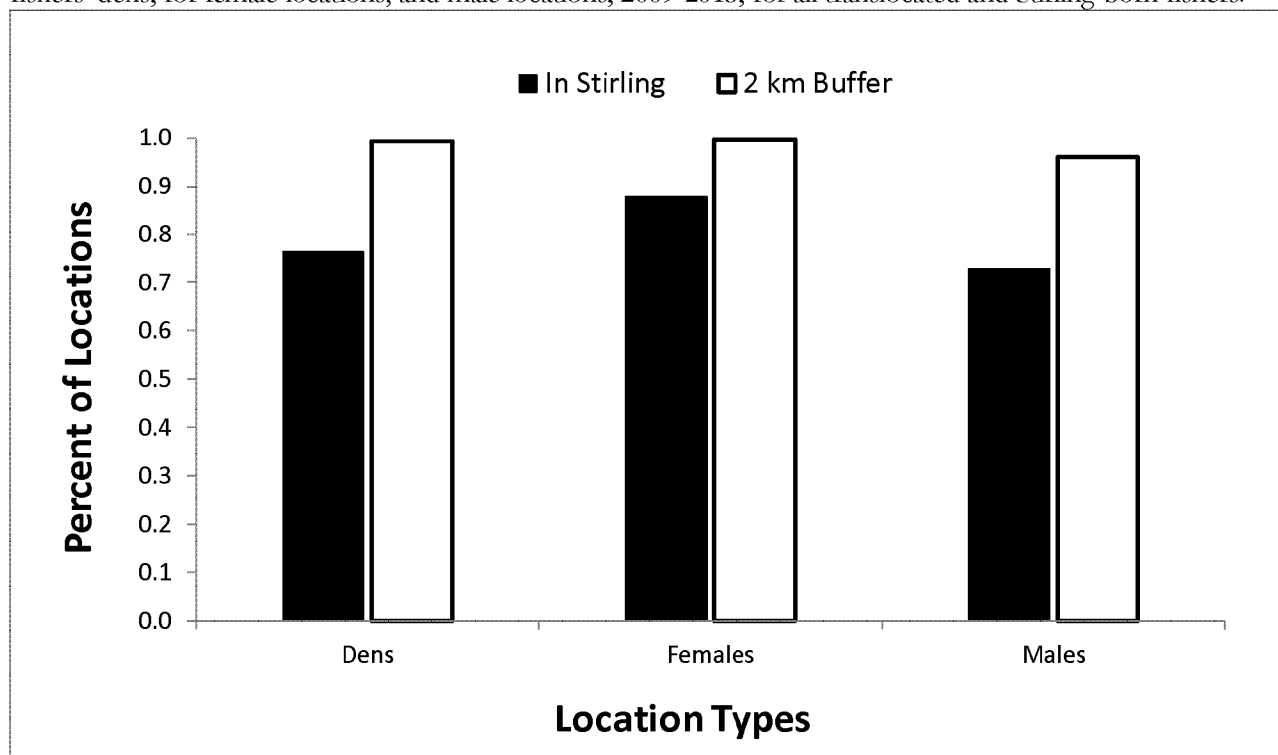
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Figure 2. Occurrence of 3 taxa of ectoparasites found on fishers from 2011-2015. Data includes only fishers captured on the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California. (a) Occurrence of parasites on fishers on Stirling by year and taxon. (b) Percentage of fishers on Stirling by year infected with any (at least one parasite of any species), 1, 2, or 3 of the most common ectoparasites by year.



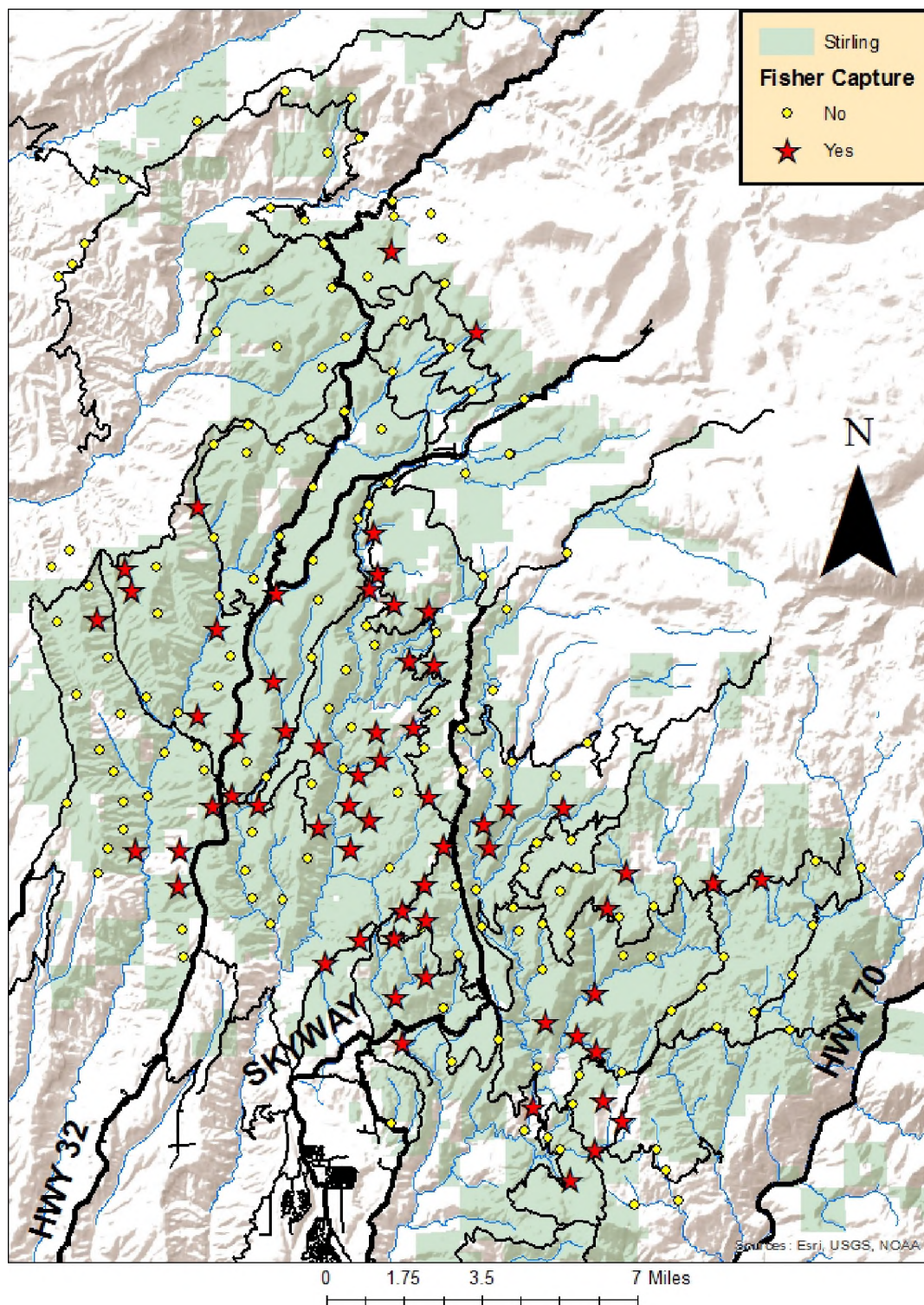
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Figure 3. The percentage of all validated locations that have occurred within the boundary for the Stirling Management Area of Sierra Pacific Industries (In Stirling) or within a 2-km buffer of the boundary (2 km Buffer) for fishers' dens, for female locations, and male locations, 2009-2015, for all translocated and Stirling-born fishers.



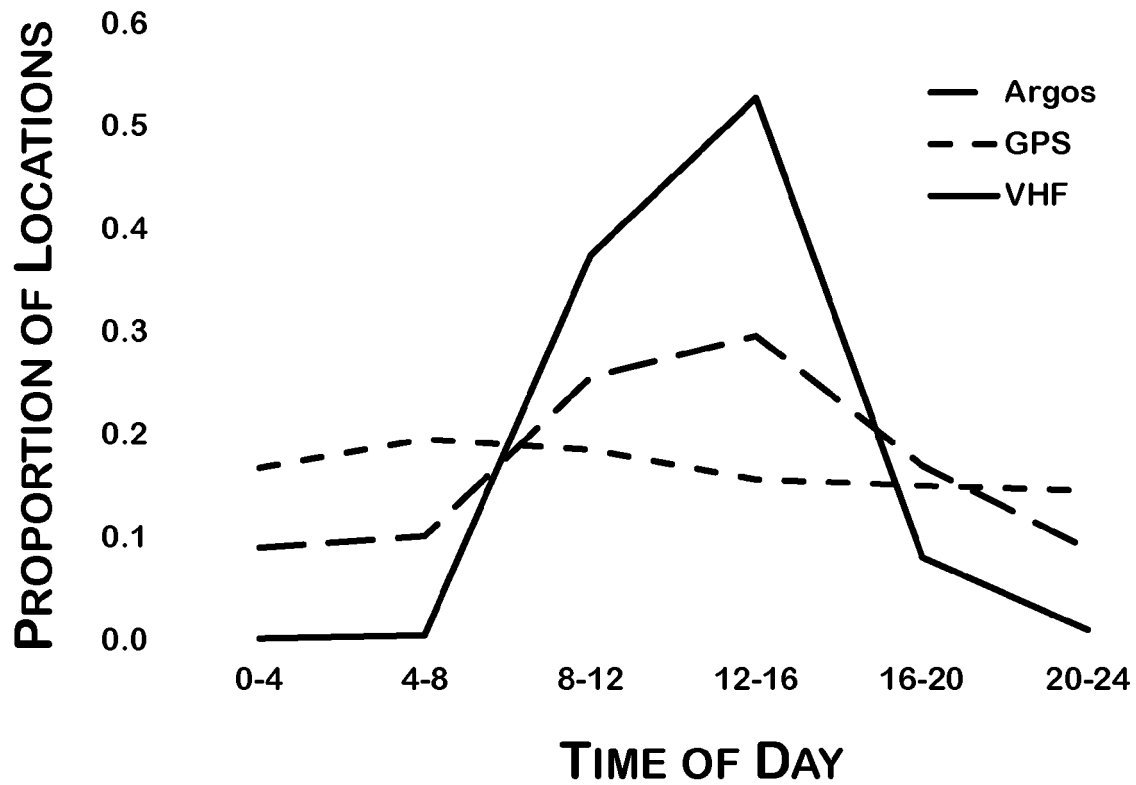
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Figure 4. Map of the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade mountains of California (green shading) and the locations of all traps set during October – November, 2015. Yellow dots represent traps that did not capture a fishers and red stars represent traps that captured at least 1 fisher.



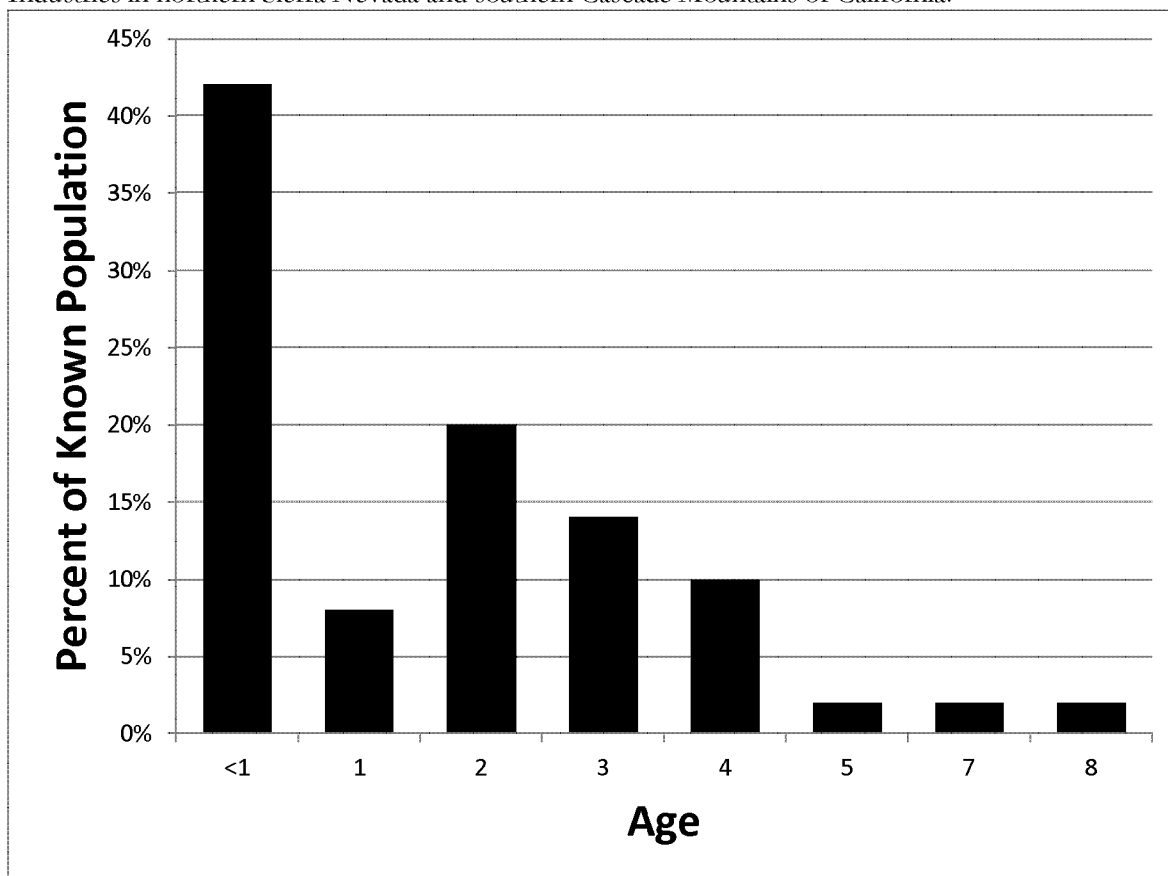
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Figure 5. Percent of estimated locations of fishers obtained via Argos, GPS and VHF telemetry at different times of day across all years of study (2009-2015) on the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California.



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Figure 6. Percent of fishers by age distribution based off cementum annuli estimates (ages for new fishers captured in 2015 were estimated based on body size, and development) on the Stirling Management Area of Sierra Pacific Industries in northern Sierra Nevada and southern Cascade Mountains of California.



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Figure 7. Annual estimates \pm 95% confidence intervals of survival for adult fishers during the non-reproductive season, juvenile fishers (<1 year old), 1-year old fisher (>1 < 2- years old) and adult fishers during the reproductive season (April-August) for reintroduced and non-reintroduced fishers from 2009 to 2015 in northern California.

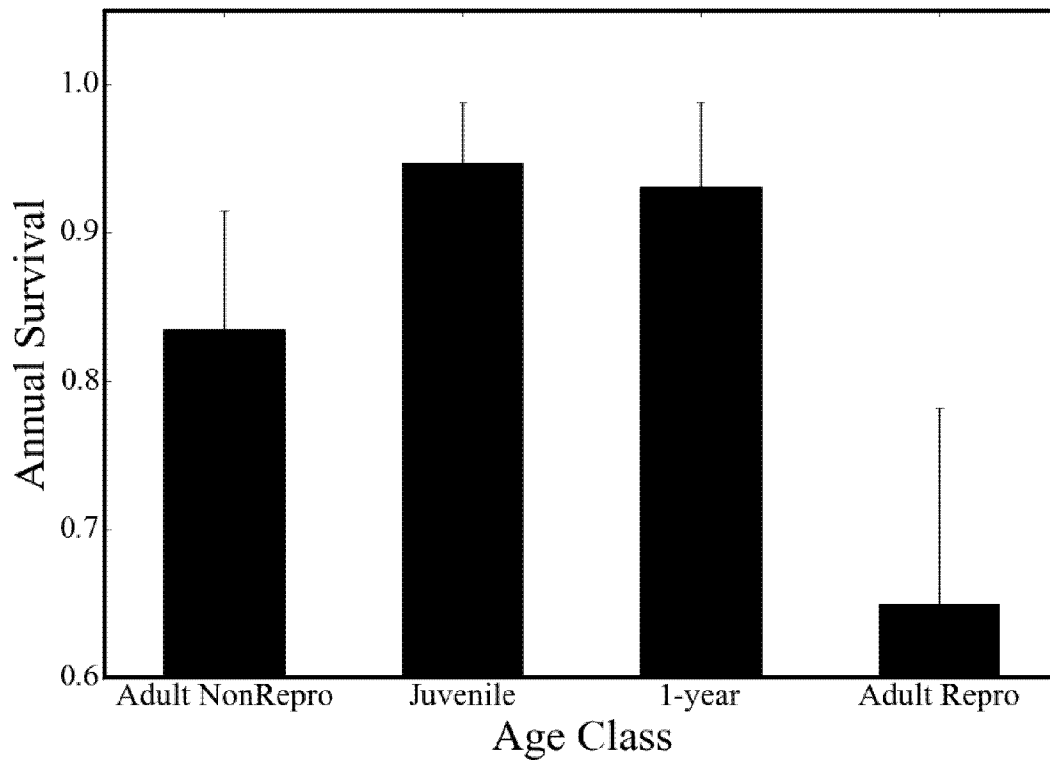
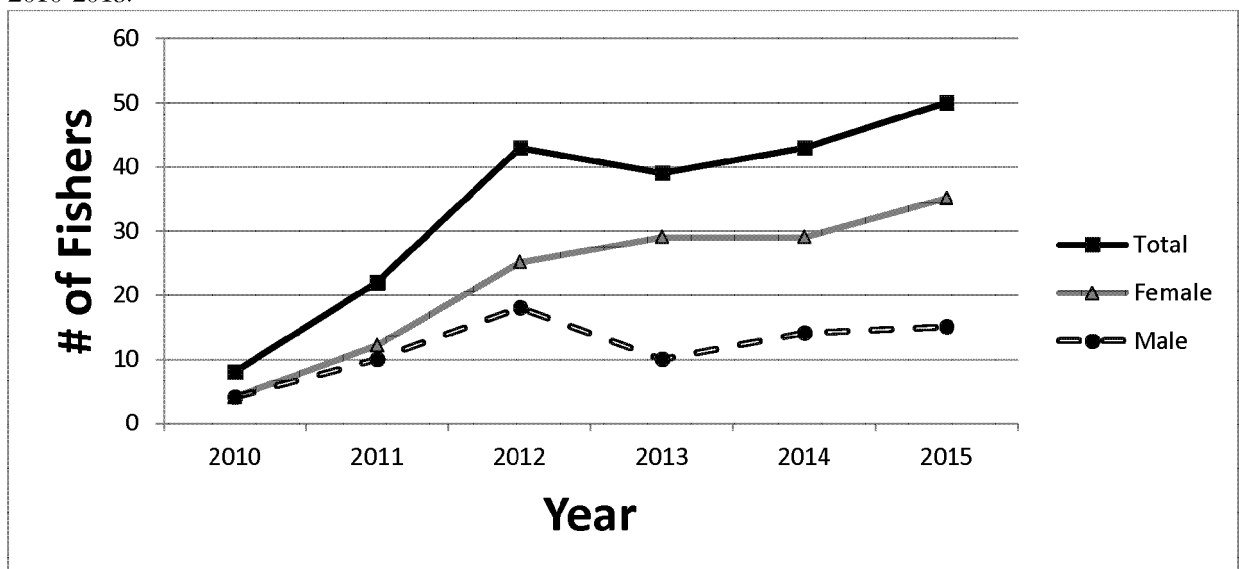
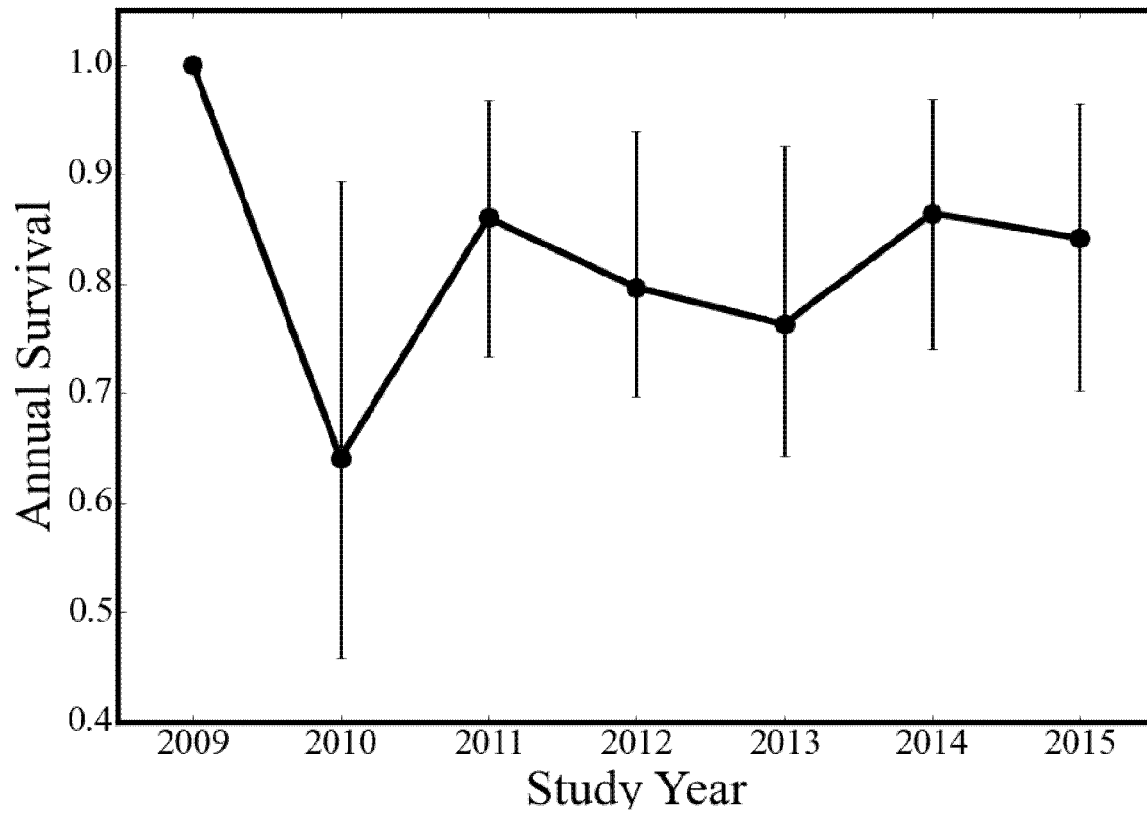


Figure 8. The minimum number of females (triangles and gray solid line), males (circles and a dashed line) and total (squares with solid black line) known to be alive for fishers living on or near the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade Mountains of California during the autumns of 2010-2015.



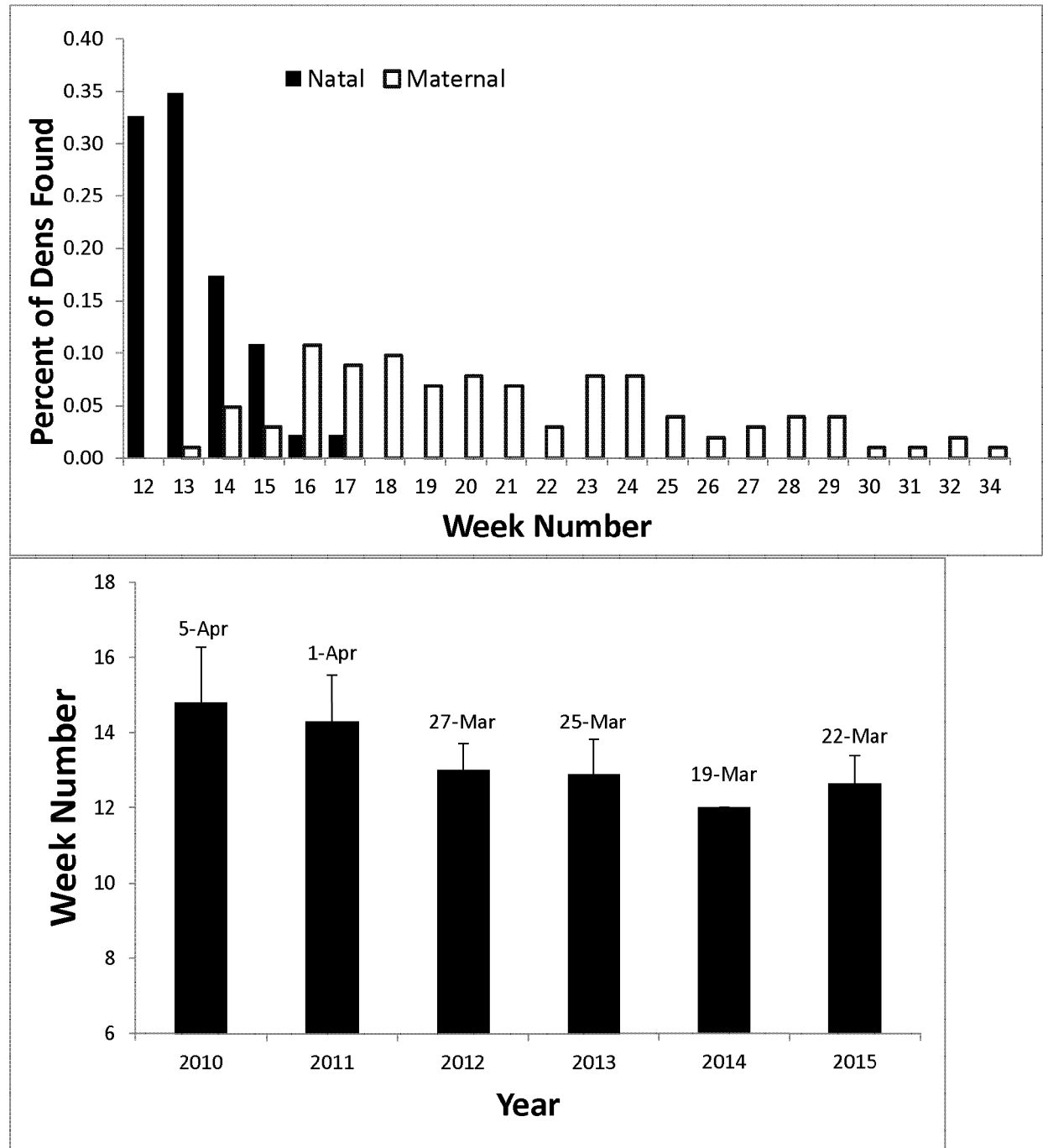
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Figure 9. Annual estimates \pm 95% confidence intervals of survival for reintroduced and non-reintroduced fishers from 2009 to 2015 in northern California. Note 2009 estimates derived from only December of that year.



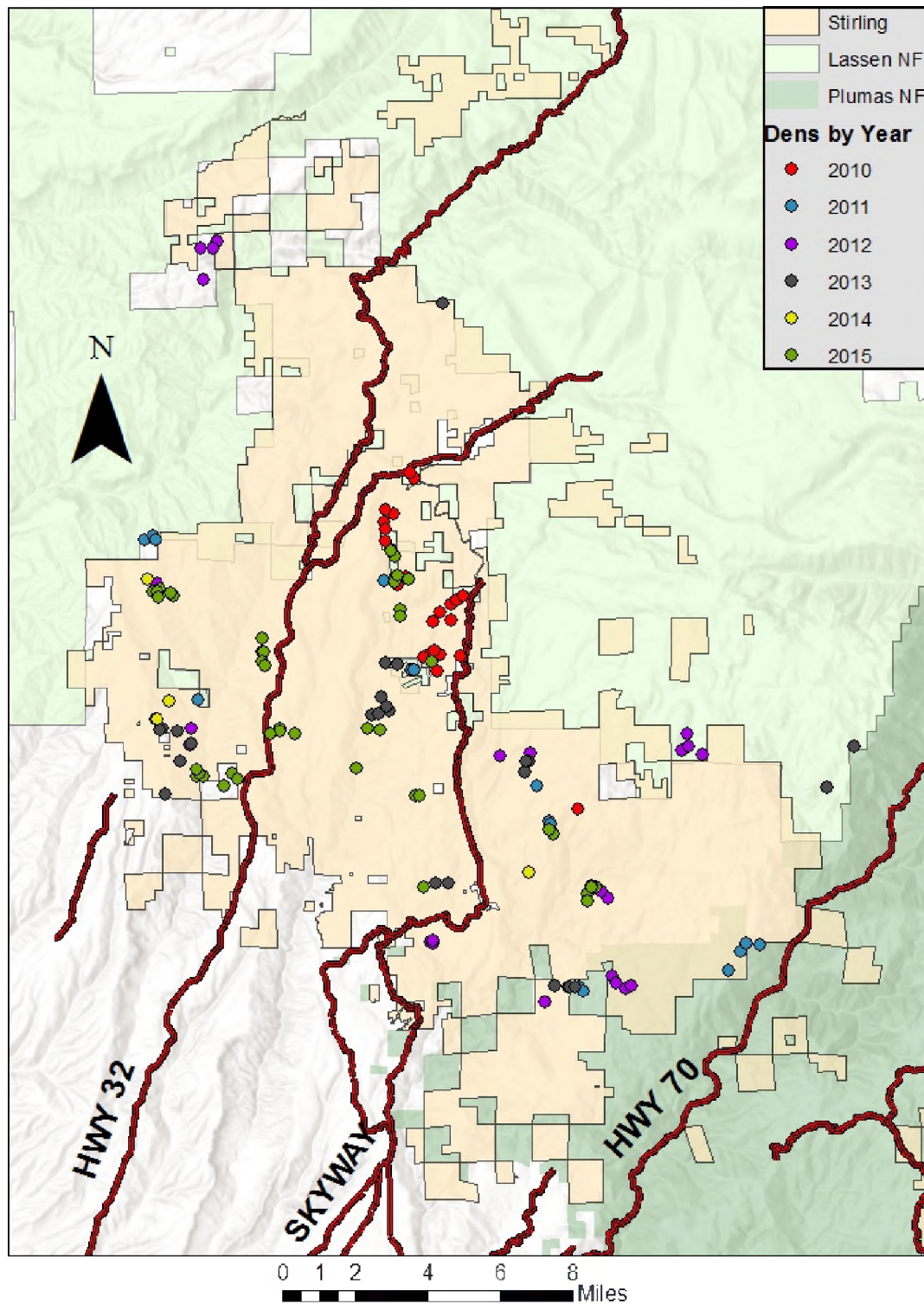
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Figure 10. (A) The percent of all natal (black bars) and maternal (open bars) fisher dens found by week in 2010-2015 and (B) The mean week number \pm 1 SD (mean date above bar), by year, that natal dens were found on and near the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade Mountains of California.



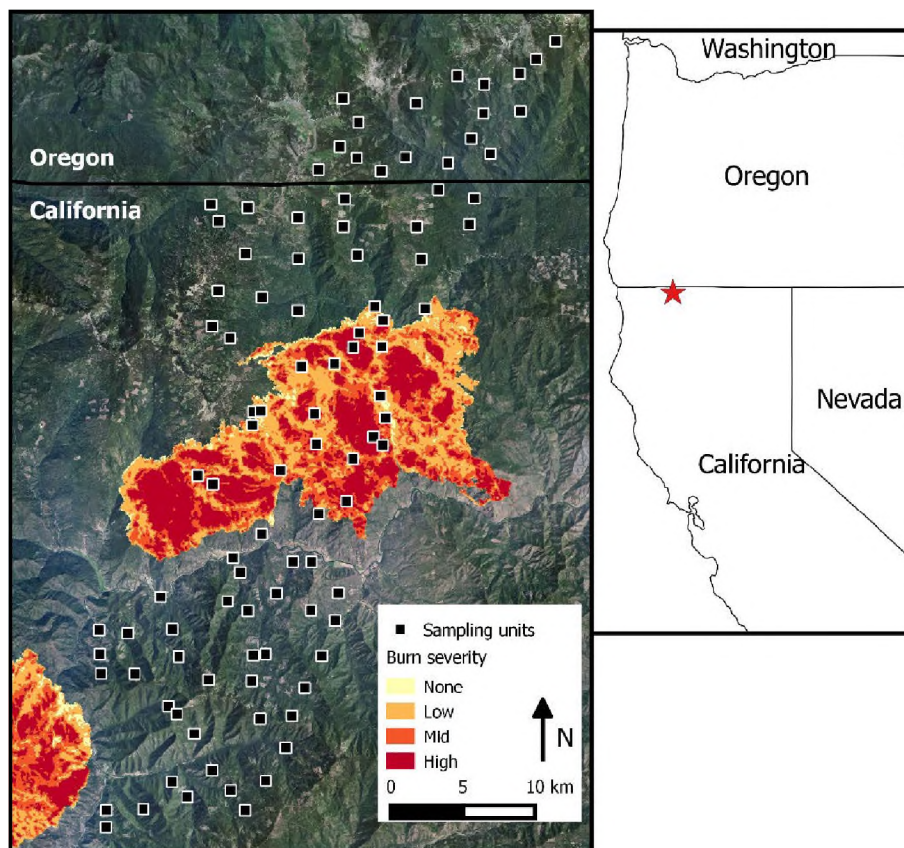
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Figure 11. Locations of female fishers' dens located during the springs and summers of 2010-2015 on, or near, the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California.



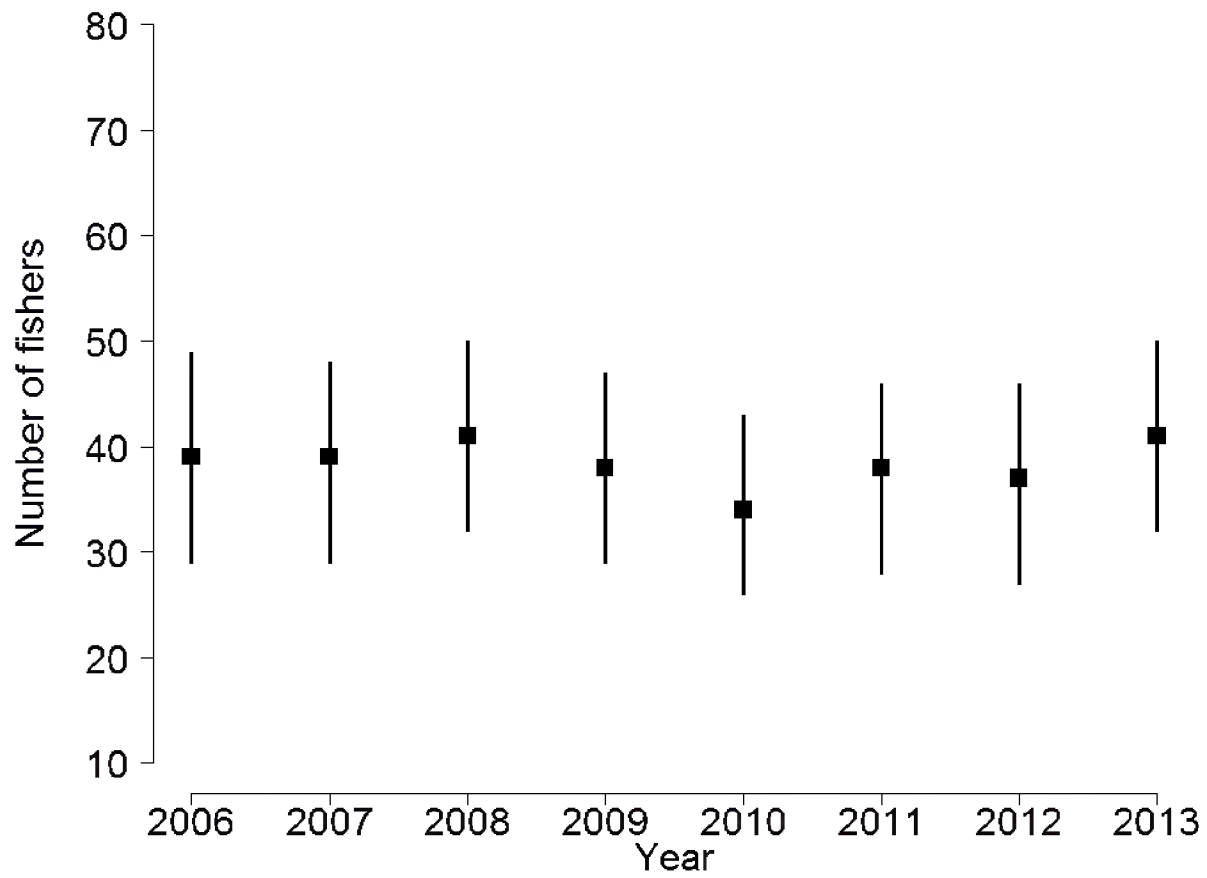
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Figure 12. Study site for monitoring fisher population dynamics in response to two wildfires in Northern California. Both wildfires occurred in the summer of 2014. The Beaver Fire is the northern fire and the Happy Camp Complex Fire is the southern fire depicted in this figure. The black squares show the locations of our survey sites. The light-colored area of low elevation bisecting the study site from east to west in approximately the middle is the valley containing the Klamath River. This satellite image is courtesy of Microsoft Bing basemaps.



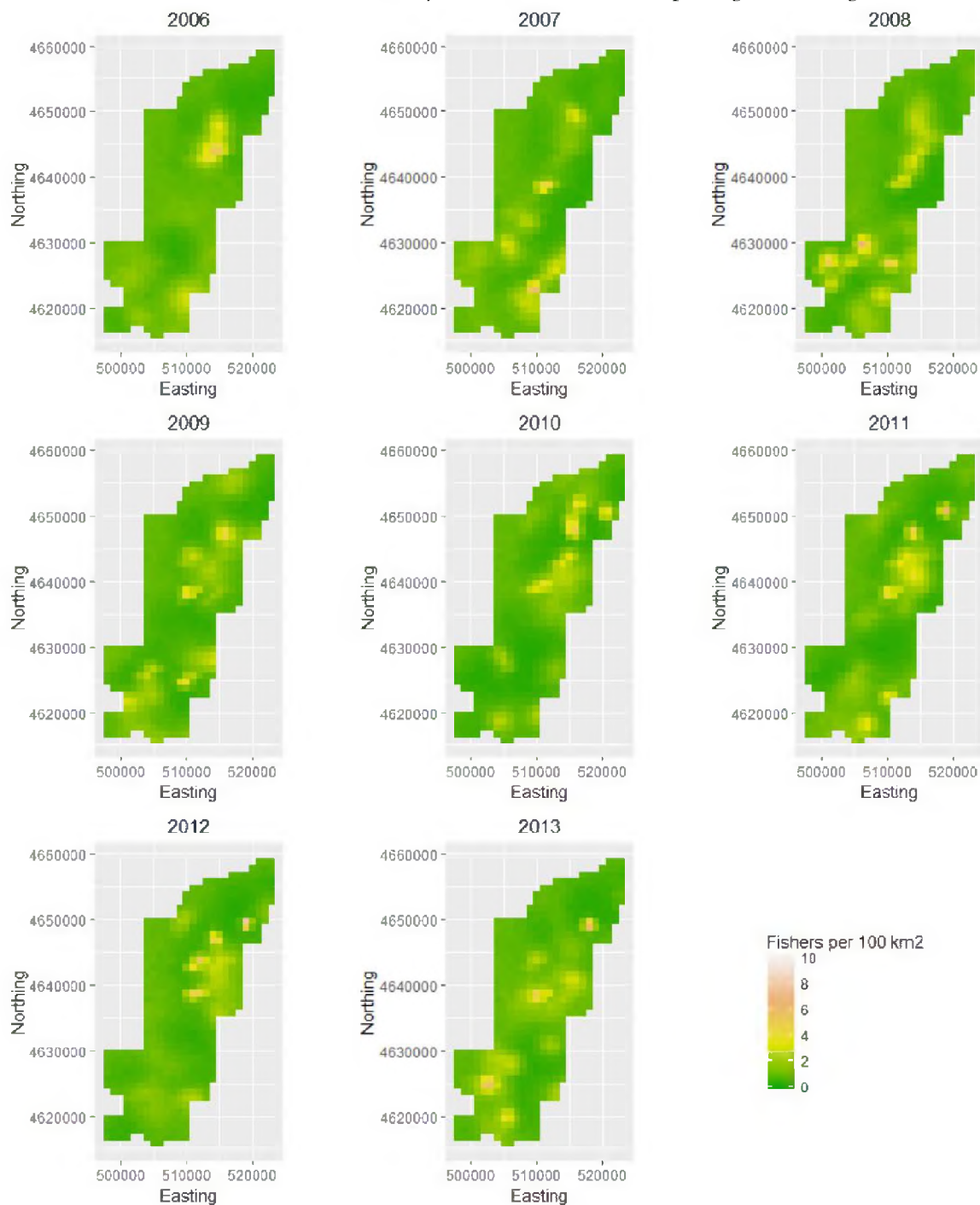
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Figure 13. Estimated annual abundance (■) and 95% credible intervals of fishers in the Klamath.



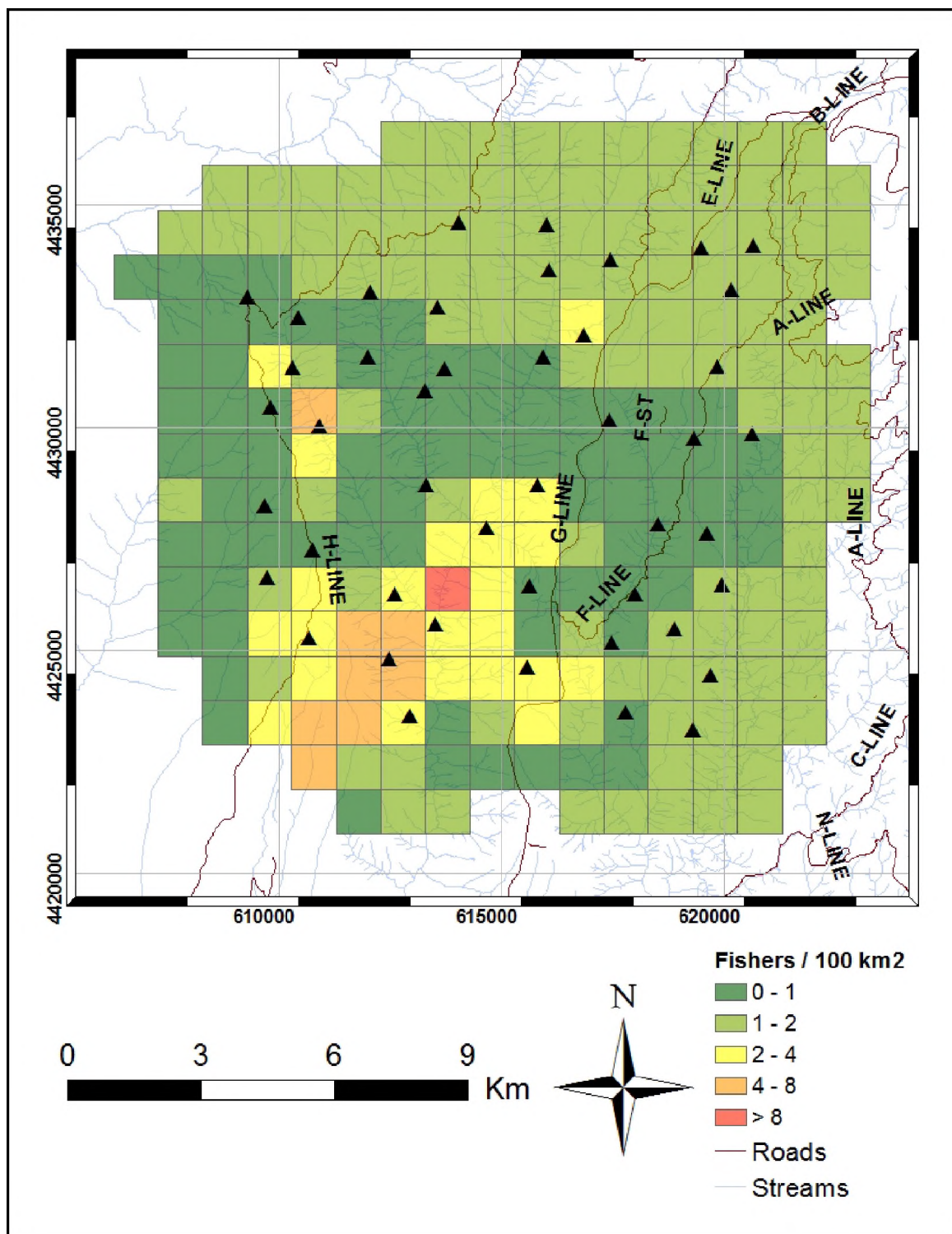
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Figure 14. Density maps indicating the locations of fisher activity centers in the Klamath. The scale bar and color scheme indicate the number of estimated activity centers/100 km² encompassing the 1 km² grid cell.



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Figure 15. Density map indicating the locations of fisher activity centers in a portion of the Stirling Management Area in 2013 relative to the non-invasive sampling units (▲). The color scheme indicates the number of estimated activity centers/100 km² encompassing the 1 km² grid cell.



Appendix 1. For fishers (*Pekania [Martes] pennanti*) released onto the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada, California, December 2009 – December 2015 identification (PIT5), sex, year of birth, age at the time of initial capture, month and year released onto the study site, weight at initial capture, whether the individual has been recaptured (# of recaptures), weight at recapture (value is the mean if individual was recaptured more than once), the month and year of death (if dead), and the cause of mortality if known.

PIT5	Sex	Year of Birth	Age at initial Capture	Cohort	Release Date (Mon-Yr)	Initial Wt (kg)	Recap (#)	Re-Cap Wt (kg)	Death Date (Mon-Yr)	Mortality Cause
199B9	F	2007	2	Year-1	Dec-09	2.2				
F6280	F	2008	1	Year-1	Dec-09	2.1			Jun-10	Bobcat predation or scavenge
17ECC	F	2007	2	Year-1	Dec-09	2.4	Yes(1)	2.3		
D00B0	F	2004	5	Year-1	Dec-09	2.5	Yes(1)	2.5	Jul-10	Drowning (water tank), No ARs
17582	F	2008	2	Year-1	Jan-10	2.1				
19316	F	2006	4	Year-1	Jan-10	2.2				
F8B8D	F	2008	2	Year-1	Jan-10	2.0	Yes(3)	2.2		
168F2	F	2007	3	Year-1	Feb-10	2.0				
F65B6	F	2008	2	Year-1	Feb-10	1.5			Jun-10	Bobcat predation or scavenge
93B5A	F	2005	5	Year-2	Nov-10	2.2	Yes(3)	2.2	Jul-14	Suspect raptor predation
18FFF	F	2007	3	Year-2	Nov-10	2.1			Aug-11	Unknown
18871	F	2008	2	Year-2	Dec-10	2.1	Yes(3)	2.4		
17FD8	F	2009	1	Year-2	Jan-11	2.2	Yes(4)	2.5		
182F4	F	2009	1	Year-2	Jan-11	2.0	Yes(1)	2.3	Jul-12	Unknown, possible felid
1E003	F	2007	3	Year-2	Jan-11	2.2	Yes(10)	2.2	Dec-15	Pending
21FB6	F	2009	2	Year-2	Jan-11	2.3	Yes(2)	2.5	Jun-13	Traumatic injury head, neck
23775	F	2008	3	Year-3	Nov-11	2.1				
2189C	F	2009	2	Year-3	Nov-11	1.8			Oct-12	Unknown
1E03E	F	2011*	0	Year-3	Nov-11	2.1			Feb-12	Systemic disease, unknown pathogen
21DFE	F	2011	0	Year-3	Nov-11	2.2				
20058	F	2010	1	Year-3	Nov-11	2.1	Yes(1)	2.3		
1F111	F	2009	2	Year-3	Dec-11	2.0			Jun-12	Bobcat predation
252FD	F	2008	3	Year-3	Dec-11	2.1			Dec-11	Systemic disease + Brodifacoum (0.17 ppm)
714C2	F	2008	3	Year-3	Dec-11	2.0				
1F955	F	2011	0	C2011	Oct-11	2.0	Yes(4)	2.4		
21392	F	2011	0	C2011	Oct-11	2.1	Yes(12)	2.4		
23955	F	2011	0	C2011	Oct-11	2.2	Yes(4)	2.5	Sep-14	Drowning (water tank)
209DD	F	2011*	0	C2011	Jan-12	2.3				
20950	F	2011	0	C2012	Jan-12	1.9	Yes(10)	2.1		
1EA8D	F	2012	0	C2012	Oct-12	2.1	Yes(8)	2.3	May-15	Pending
6178F	F	2012	0	C2012	Oct-12	2.2				
36A8B	F	2012	0	C2012	Oct-12	2.0	Yes(4)	2.3		
242DB	F	2012	0	C2012	Oct-12	1.8	Yes(4)	2.2		
614DF	F	2011	1	C2012	Oct-12	2.3	Yes(3)	2.4		
3828E	F	2012	0	C2012	Nov-12	2.2	Yes(5)	2.3		
35978	F	2012	0	C2012	Nov-12	1.9				
397A3	F	2012	0	C2012	Nov-12	2.0	Yes(2)	2.2		
76482	F	2013	0	C2013	Oct-13	2.1	Yes(6)	2.5		
15852	F	2012	1	C2013	Oct-13	2.3	Yes(3)	2.4		

16026	F	2013	0	C2013	Oct-13	2.2	Yes(2)	2.4		
69940	F	2013	0	C2013	Oct-13	2.1	Yes(2)	2.4		
23737	F	2013	0	C2013	Oct-13	2.0	Yes(1)	2.2		
34786	F	2013	0	C2013	Oct-13	2.4				
35232	F	2013	0	C2013	Nov-13	2.2	Yes(1)			
33860	F	2014	0	C2014	Oct-14	2.1				
35006	F	2014	0	C2014	Oct-14	2.2	Yes(5)	2.5		
31237	F	2013	1	C2014	Oct-14	2.2	Yes(3)	2.2		
36775	F	2014	0	C2014	Oct-14	2.4	Yes(1)	2.3		
36651	F	2014	0	C2014	Oct-14	2.3	Yes(1)	2.4		
43476	F	2014*	0	C2014	Oct-14	2.2				
36069	F	2014	0	C2014	Oct-14	2.2	Yes(2)			
36752	F	2013	1	C2014	Nov-14	2.1	Yes(1)	2.2		
51812	F	2013	1	C2014	Nov-14	2.3	Yes(2)	2.4		
0DE63	F	2015	0	C2015	Oct-15	2.2				
136BC	F	2015	0	C2015	Oct-15	2.1				
27FB2	F	2015	0	C2015	Oct-15	2.0	Yes(3)			
16FDD	F	2015	0	C2015	Oct-15	1.8				
178B2	F	2012	3	C2015	Oct-15	2.2	Yes(1)			
17105	F	2015	0	C2015	Oct-15	2.2	Yes(1)	2.0		
27E2F	F	2015	0	C2015	Oct-15	2.0			Dec-15	Pending
29B29	F	2015	0	C2015	Oct-15	2.0	Yes(1)	2.0		
2741A	F	2015	0	C2015	Oct-15	1.6	Yes(3)			
194F6	F	2015	0	C2015	Oct-15	2.2	Yes(1)	2.0		
11720	F	2013	2	C2015	Nov-15	2.3				
19277	F	2015	0	C2015	Nov-15	2.0				
0A3CA	F	2015	0	C2015	Nov-15	2.0	Yes(1)	2.1		
0D8F1	F	2015	0	C2015	Nov-15	2.3				
16848	M	2007	3	Year-1	Jan-10	4.5				
18308	M	2007	2	Year-1	Jan-10	4.2	Yes(3)	5.1	May-12	Unknown
596E2	M	2004	5	Year-1	Jan-10	4.1	Yes(3)	4.0	Sep-13	Possible avian predation or scavenge
181F9	M	2009	1	Year-1	Jan-10	3.4	Yes(4)	4.4		
F0858	M	2006	4	Year-1	Jan-10	4.1	Yes(2)	4.3	Jun-13	Unknown
FB7DA	M	2007	3	Year-1	Jan-10	4.2	Yes(1)	4.4		
F605B	M	2008	2	Year-2	Nov-10	3.3	Yes(1)	4.1		
58985	M	2007	3	Year-2	Nov-10	4.5	Yes(4)	4.5		
18CC8	M	2006	4	Year-2	Nov-10	3.6	Yes(2)	4.1	Apr-15	Pending
18C3E	M	2006	4	Year-2	Dec-10	4.5			Mar-11	Roadkill, + Brodifacoum Bromadiolone
18AA5	M	2004	6	Year-2	Jan-11	4.2	Yes(3)	4.1		
22526	M	2006	4	Year-2	Jan-11	5.3				
1E14C	M	2009	2	Year-3	Nov-11	3.9				
24315	M	2011	1	Year-3	Nov-11	3.5	Yes(1)			
1E10F	M	2011	0	Year-3	Dec-11	3.7	Yes(9)	4.4		
1EC04	M	2010	1	Year-3	Dec-11	4.1	Yes(1)	4.8		
24033	M	2011	0	C2011	Oct-11	3.6				
2305B	M	2010	1	C2011	Oct-11	4.0				
24101	M	2011	0	C2011	Oct-11	4.1	Yes(3)	4.3	Apr-13	Unknown, No ARs in muscle
1FE60	M	2011	0	C2011	Oct-11	3.5	Yes(3)	4.2		

24B09	M	2012	0	C2012	Oct-12	3.5	Yes(1)	4.2	Jan-14	Unknown
64311	M	2012	0	C2012	Oct-12	3.2	Yes(2)			
39A7E	M	2012	0	C2012	Oct-12	3.3	Yes(2)			
3AD54	M	2012	0	C2012	Oct-12	3.0				
1E613	M	2011	1	C2012	Nov-12	3.8	Yes(1)	4.0	Mar-14	Unknown
38908	M	2012	0	C2012	Nov-12	3.4				
70926	M	2013	0	C2013	Oct-13	3.5	Yes(1)	4.0		
36623	M	2014	0	C2014	Oct-14	3.4				
34589	M	2014	0	C2014	Oct-14	3.2	Yes(3)	4.0		
32976	M	2014	0	C2014	Oct-14	3.2	Yes(2)			
33890	M	2014	0	C2014	Oct-14	3.5	Yes(2)			
12834	M	2014	0	C2014	Oct-14	3.1				
36056	M	2014	0	C2014	Nov-14	3.3	Yes(1)			
36103	M	2012	2	C2014	Jan-15	4.8	Yes(3)	4.8		
33276	M	2013	1	C2014	Jan-15	4.3	Yes(1)			
32402	M	2015	0	C2015	Oct-15	2.7	Yes(1)	3.0		
33785	M	2015	0	C2015	Oct-15	3.0	Yes(3)			
35734	M	2015	0	C2015	Oct-15	3.1	Yes(8)			
36226	M	2015	0	C2015	Oct-15	2.8	Yes(1)			
18F2C	M	2013	2	C2015	Oct-15	4.4				
34046	M	2015	0	C2015	Oct-15	3.2	Yes(1)			
34509	M	2015	0	C2015	Oct-15	2.9				
34104	M	2015	0	C2015	Oct-15	3.3	Yes(1)			
33850	M	2015	0	C2015	Nov-15	3.6				

* No tooth extracted, year of birth estimated from physical exam.

Fisher Habitat Selection on Stirling management district from 2010-2014: A critical test of our understanding of fisher habitat needs

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Executive Summary

Understanding the habitat requirements and habitat selection of fishers (*Pekania pennanti*) on lands managed primarily for timber production is a key concern for their management and conservation throughout the western portion of their range. Several models of fisher habitat need have been developed, but most have never been tested with data independent of those used to construct those models. Independently testing habitat models provides an important way of assessing our basic understanding of the needs of fishers, both generally and specifically on managed landscapes. Fishers are generally associated with forests of high canopy closure and large mean diameter trees. Most fisher models incorporate measures of these two components, but there is wide variation in other attributes of models including environmental variables, such as precipitation and terrain, other vegetation-based attributes like canopy layers, stand types (e.g., hardwoods vs conifers), and relationships to human-caused disturbance (e.g., road density, edge density). We tested 12 models of fisher habitat quality to evaluate how well they predicted use of space by a population of fishers reintroduced to the Stirling district owned by Sierra Pacific Industries. We tested the models of Allen, Carroll (1999 & 2005), California Wildlife Habitat Relationships (CWHR), Davis (2007), SPI and a composite mean model. With the exception of the Davis models we used vegetation data from SPI's forest inventory to create annual habitat models. We evaluated these models across multiple scales of selection including across the landscape (Stirling and surrounding areas), within female home ranges, and within the short-term areas of movement for 4 fishers with GPS collars. We found that the Allen model predicted the highest overall habitat quality, followed by the SPI habitat form model, the CWHR model, Carroll (1999), Carroll 2005, and the David models. Across all models, fishers avoided areas predicted to have low habitat quality. These areas are characterized by low (<20%) canopy

closure, and trees with mean quadratic mean diameter at breast height (QMD) of 30 cm (11''). Thus, there is strong support that fishers avoid areas with little over story tree component. Additionally, there was high agreement among models that fishers used areas of canopy closure >60% and QMD > 60cm (24''). At moderate levels of habitat quality there was little agreement among models. An index of the Carroll 2005 (simple model) which included Terrain Ruggedness and canopy closure predicted fisher selection of habitat best with the highest correlation coefficients as well as the strongest indices of habitat avoidance at low values and selection at habitat values >0.50. Nevertheless, we find that terrain ruggedness was correlated to other vegetative metrics including percent of a stand in hardwoods, tree diversity and logs per acre. Further, areas of high terrain ruggedness are areas likely most unaffected by management, and hence the terrain ruggedness index captured multiple aspects of habitat quality in one metric. That is, areas with high terrain ruggedness were also likely to have more tree diversity, hardwood trees, and less active management. We also find that the Allen model displays relatively high indices of avoidance and selection at the highest and lowest values; however, fishers consistently avoid areas within the predicted habitat quality range of 0.8-0.9. Further, we found that these areas, on average, have a low component of hardwoods within these stands though they have relatively large trees and overstory canopy closure. Thus, the Allen model seems to have been diagnostic to detecting areas that fishers avoided despite conforming to what might be considered high habitat quality without inclusion of hardwoods. Additionally, we found that fisher avoided values within the CWHR model that were similar to the Allen model. Further analysis revealed high overlap between the CWHR and Allen model at these 'moderate' habitat values that fishers avoided. Thus, suggesting fishers avoided similar places that were low in hardwoods. The SPI habitat form model, which relies on canopy, QMD and counts of trees over >22'', accurately predicted fisher avoidance at low and high values and with indices of selection similar to other models. Fishes avoided all other areas classified by the SPI model other than the top category (Habitat Form 4). Evaluation of the SPI territory opportunity model indicated that female fishers established home ranges in all 3 hexagon types proposed by SPI. Logistic regression analysis suggests that hexagon classified as support cores or territory cores are more to contain a female home range compared to hexagons classified as currently below threshold. Nevertheless, there is not statistical difference in the probability of a hexagon containing a home range, or more than one home range, based on the hexagon classification. Across all models we

tested, the highest habitat quality are areas that have undergone little clear-cut logging within the last 50 years. Thus, areas of highest quality have commensurately high canopy and QMD, but also are correlated with components such as hardwoods and tree diversity not seen at lower classification levels. Thus, we urge additional work to evaluate the importance of hardwoods as well as other vegetative components such as tree diversity and understory diversity. Additionally, we find no statistical difference in habitat use and avoidance based on the sex of fishes or among the years we evaluated habitat use. Thus, we find no strong evidence that fishers we originally reintroduced used habitat differently than did fishers born on the study site in later years.

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INTRODUCTION:

To understand how fishers (*Pekania pennanti*) respond to intensive forest management the California Department of Fish & Wildlife (DFW), the US Fish & Wildlife Service (FWS), Sierra Pacific Industries (SPI) and North Carolina State University (NCSU) started collaborating in 2007 to re-establish a fisher population in the northern Sierra Nevada and southern Cascade Mountains of California. How fishers use and respond to habitat alteration, particularly alterations occurring because of industrial logging, is a central concern for fisher management ((USDI 2014). Thus, a major objective of our project is to evaluate models of habitat for fishers, those already existing and ones we develop. No less than 12 models of habitat have been developed, including those of Allen (1983), the California Wildlife Habitat Relationship model (Airola 1988), Carroll (1999, 2005), Davis et al. (2007), Zielinski et al. (2010), and a model developed by Sierra Pacific Industries for its Candidate Conservation Agreement with Assurance with the FWS. We view the applicability, or generality, of models to predict use of space by

fishers in multiple and novel areas as most important. Unfortunately, none but the Allen model have been tested with independent data or at locations aside from those where the models were developed and, thus, their abilities to predict fisher use of habitat generally is unknown. Since the Allen model has been validated at 2 independent locations (Thomasma et al. 1991, Powell & Zielinski 1994a, Thomasma et al. 1994b, Powell 2004), it represents an important model to test and refine first, as a baseline. Therefore, we present here an examination of the use of habitat by fishers and in particular we evaluate several habitat models using data from the Stirling reintroduction and location data for fishers for 2009-2014.

Allen (1983) developed the first quantitative model of fisher habitat from the literature and from expert knowledge and indexed how use of space by fishers varied with 4 variables tied to fishers' resources: number of canopy layers, canopy closure, overstory tree diameter at breast height, and the ratio of deciduous to conifer trees in the canopy (Figure Allen). Despite its simplicity, this model has been tested 3 times in 2 vastly different forest ecosystems (the Laurentian forests of the northern Great Lakes region and the mixed conifer-hardwood forests of the northern Sierra Nevada of California) and shown to predict use of space by fishers (Powell 2004, Thomasma 1991, 1994). Over the decades, models of fisher habitat have employed ever more sophisticated multivariate statistical algorithms to produce maps of fisher probability of occurrence or use of space. Recent models have used diverse, GIS-based environmental variables, including traditional land cover data, climate variables, and remotely sensed variables, along with fisher location data to map use of space by fishers (Scheller et al. 2011, Zielinski et al. 2012) across spatial scales. Nevertheless, most models of fisher habitat include identical or very similar metrics to the Allen model. In particular, canopy cover, tree size and hardwood/conifer components are common among many models (see Table 1).

In general, field observations and habitat models agree that fishers are associated with forests with high overhead canopy and large trees (Powell 1993, Powell & Zielinski 1994b, Carroll et al. 1999, Sauder & Rachlow 2015). Hardwood and deciduous tree components, though commonly considered in the literature, have variable and inconsistent effects on predicted fisher habitat quality. Snow-tracking of fishers suggested that, though fishers travel through deciduous stands, they concentrate their foraging within isolated pockets of conifers with complex structure within forest stands that are otherwise largely deciduous (Powell 1978, McCann et al. 2014). Thus, Allen viewed an abundance, or prevalence, of deciduous trees as a negative effect on overall habitat quality, but viewed few or even no deciduous trees as having only marginal negative influence on habitat quality. In contrast, Thomasama (1994) suggested that stands with no deciduous component have low overall habitat value. Often studies using telemetry and available raster GIS databases have mistakenly concluded that fishers used deciduous stands extensively (Gilbert et al. 1997) but may be stands that they actually avoid (McCann et al. 2014, Powell 1994a).

Habitat models can be too specific, reducing their generality. Some habitat models for fishers developed for northern California emphasize terrain ruggedness (Carroll 2005, Davis et al. 2007) because fisher locations were correlated with steep terrain. Yet, such correlations may be spurious or may reflect other processes. For example, steep terrain may simply hinder logging or road building and thus provide for forests with few disturbances. Fishers exist elsewhere in California and east of the Rocky Mountains where steep terrain does not exist. Hence, terrain is unlikely a limiting factor of itself.

Testing models with independent data has provided feedback and new insight into why habitat characteristics are important to fishers (Powell 2004, Thomasama et al. 1991, 1994). Research on

fishers has also shown where caution is needed with habitat modelling (Gilbert et al. submitted, McCann et al. 2014, Zielinski et al. 2010). Habitat models are being developed from the bottom up (e.g., Mitchell & Powell 2003)), starting with fishers' requirements, establishing how the conditions for requirements can be represented by environmental variables in such a way as to be applicable throughout the fisher's range. Testing such a model will show how well we understand how and why habitats meet fishers' requirements.

Information on the locations, home ranges, and habitat selection of fishers on a managed landscape are particularly relevant to development of a Candidate Conservation Agreement with Assurances (CCAA) between the USFWS and SPI. Recent discussion between the USFWS Yreka Field Office staff and the research team at NCSU have identified several key information gaps relative to habitat selection and habitat avoidance of fishers on managed forests. Thus, the research team at NCSU is in a singular position to evaluate fisher habitat associations and their needs on managed landscapes.

Here we perform analyses aimed at evaluating and predicting habitat quality for fishers using several existing habitat models. We will test the appropriateness of each of these models by evaluating what fishers have selected since their initial release in 2009 through 2014. Further, we will quantify which, if any, habitat model is appropriate for describing fisher habitat and rank them in terms of their abilities compared to one another.

Tested Models

A wide variety of fisher habitat models have been proposed and constructed with various statistical methods and techniques. Some habitat models address specific requirements or life-history needs for fishers such as denning or resting habitat (Zielinski et al. 2012, Niblett et al. 2014). Other models aim to describe and predict the likelihood of occurrence of fishers over

specific landscapes (Airola 1988, Carroll et al. 1999, Carroll 2005, Davis et al. 2007, Zielinski et al. 2010, Sauder & Rachlow 2014, 2015). Fewer fisher habitat models have been developed, and tested, that are general to all aspects of fishers habitat associations throughout their range (e.g., Allen 1983, Thomasma et al. 1991, Thomasma et al. 1994a). For our analyses and tests we did not include, or test, models that were specific to habitat, denning or other specific fisher needs or to other life-history traits or events. We chose to focus on models that had explicit or implicit goal of creating an integrated (or complete) representation of fisher habitat quality. Additionally, we chose models that were both specific to Stirling and California as well as any that were spatially and temporally general (Table 1). We built models put forth by Allen (1983), Carroll et al (1999) and amended in 2005, the most recent California Wildlife Habitat Relationships model (CWHR; Airola 1988), and habitat relationship models proposed by Sierra Pacific Industries (Ed Murphy personal communication). For these aforementioned models we used SPI forest inventory data in all years from 2010 to 2015 to construct annual habitat models for the Stirling tract. Additionally, we used the Gradient Nearest Neighbor (GNN) data sets downloadable at (<http://lemma.forestry.oregonstate.edu/data>)(Ohmann et al. 2011), to construct some, but not all, models from a different data set for testing and evaluation. The GNN data set is fundamentally different then stand-level inventory data provided by SPI. The GNN data is pixel-based (30 ×30m) and compiled from various sources. Additionally, these data are not temporally consistent since the field data that are used to construct it are available from different sources and potentially collected at disparate times.

MODEL CONSTRUCTION AND DATA

Allen HSI

The Allen model is relatively simple and is based on 4 vegetative metrics: 1) percent tree canopy closure, 2) quadratic mean diameter at breast height (QDBH) of overstory trees, 3) tree canopy diversity (*i.e.*, number of canopy layers), and 4) percent of overstory trees that are hardwoods. Allen's original conception was to use the percentage of deciduous trees (Figure Allen; Allen 1983). Nevertheless, we included all hardwood trees in our models because Allen's original models considered deciduous trees as important components for denning habitat. Therefore, we assumed that all hardwood trees provide increased opportunities for denning habitat (among other possible ways they can contribute to habitat quality). The canopy layer component of this model is somewhat nebulous based on what is considered important to fishers as well as the needed data to quantify this metric. Layering could potentially include only vertical layers that are off the ground and in this conception stands with shrub (or sapling), mid-aged, and dominant canopies could be considered to have high values for fishers. Additionally, layering could be important for foraging and could include structure (layers) that occur on the ground including logs, rocks and low hanging tree limbs. In our development of the Allen model with SPI-based data we chose to parameterize the layer component with data based on the overall area covered by dominant, codominant, intermediate and suppressed trees. For all 4 components of Allen's model we used mean stand-level forest inventory data from SPI, we were able to approximate the 4 components of Allen's model and created an overall habitat suitability index (HSI) for Stirling for 2010 to 2015 (equation 1). The Allen model constructed with SPI-based data is explicit to land within the Stirling boundary (Figure AllenSPI).

Eq. 1:

$$\text{Allen HSI} = (\text{Canopy Closure} + \text{QMD} + \text{Canopy Layer})^{\frac{1}{3}} \times \text{Percent Hardwood}$$

In addition to developing the Allen model with SPI-based inventory data we also constructed a model with GNN/Lemma data. We used 4 variables found with the GNN dataset to parameterize the Allen model: quadratic mean diameter of all dominant and codominant trees (QMD_DOM), canopy cover (CANCOV), number of canopy layers (CANCOV_LAYERS), and proportion of hardwood basal area (BAH_PROP). These layers were classified and implemented in the Allen HSI exactly as was the model using SPI-inventory data. Because the GNN data set covers Stirling and the surrounding area it was implemented for the entire region around Stirling as well as Stirling proper (Figure AllenRegion).

Carroll 1999 – adjusted

The Carroll (1999) habitat model is a logistic regression model based on the probability of detecting fishers at a location. Similar to Allen, the Carroll model uses measures of canopy closure and quadratic mean diameter of trees (Table 1). Additionally, the Carroll model uses the percent of conifers (Per_Conifer) within a stand, mean annual precipitation (AnnPrecip), the north-south position of locations (UTM_N) and north-south position raised to the second power (UTM_N²). The equation also includes two interaction terms between canopy and percent conifer as well as QMD and annual precipitation. Data concerning tree size, canopy closure and percent conifer was taken from SPI-inventory data and UTM position was extracted from ArcGIS 10.3. Mean annual precipitation data from 1999 to 2010 were taken from the ClimateWNA website at

<http://climatemodels.forestry.ubc.ca/climatebc/downloads/libraryfiles.zip> (Wang et al. 2012).

These data are on 1-km² resolution that covers the entire study area and surrounding areas. These variables were used with the beta-parameter estimates found in Carroll (1999; equation 2).

Eq.2

$$\begin{aligned} \text{Carroll} = & (1/(1+\exp(-(-2401+(0.6023 \times \text{canopy}) + (-12.07 \times \text{QMD}) + (0.4911 \times \text{Per_Conifer}) \\ & + (-0.01307 \times \text{AnnPrecip}) + (0.001059 \times \text{UTM_N})+(-0.0000000001176 \times \text{UTM_N}^2)+(- \\ & 0.006251 \times (\text{canopy} \times \text{Per_Conifer}))+ (0.005004 \times (\text{QMD} \times \text{AnnPrecip})))) \end{aligned}$$

In Carroll's original model the vegetative parameters were averaged across a 10-km² moving window (Carroll 1999). We did not average vegetative parameters because we were interested in fisher use at the scale of individual locations and home ranges; whereas, the scale of the original paper was across the state of California. Our first implementation of the Carroll model provided results that were contrary to our understanding of the original model's overall pattern. Areas that should have low habitat values (e.g., recently logged areas with 0 canopy closure and QMD) were estimated with values approaching 1 (Fig. CarrollA). The misrepresentation of habitat quality at these low canopy and QMD values was likely an aberration that never occurred in the original model because those values would likely never have appeared in the averaged landscapes. Thus, we corrected this model by replacing all cells with habitat quality > 0.90 and that had canopy values < 0.10 to have habitat quality = 0. This adjustment produced results that were visually consistent with our interpretation of the original model (Figure Carroll B). We constructed the Carroll model with this adjustment for all year from 2010 to 2015 on Stirling. Nevertheless, we recognize that the model we produced is fundamentally different from that of Carroll 1999, but incorporates the same variables and patterns of habitat quality. We did not construct the Carroll model based on GNN data.

Carroll index

Carroll (2005) revisited the analysis from the 1999 paper and amended it with new data that generated new beta-parameters estimates. Additionally, Carroll constructed additional logistic regression models. We attempted to implement these models, but could not because the Carroll 2005 report lacked all parameters (the y-intercepts) from all models. Preliminarily, we constructed these models with specific assumptions about the y-intercepts but could never find a solution that was satisfying. Nevertheless, Carroll created and reported a model (model 3; Carroll 2005) that included only 2 parameters: Terrain Ruggedness Index (TRI; Riley 1999) and canopy cover. Though we could not find a satisfactory solution based on the beta-parameters for this model we created an index that incorporated both parameters. We calculated TRI within ArcGIS 10.2 using a 30×30 m digital elevation model (DEM) following the methods of Riley (1999; the same method as Carroll 2005). This resultant raster produced values between 0 and 160 with the highest values being the most rugged locations. We then scaled the TRI raster to values between 0 and 1 by dividing each cell by 160. We then calculated a mean value between the canopy raster and the TRI raster. Because the beta parameters for both TRI ($\beta = 0.0226603$) and canopy ($\beta = 0.0456$) were positive (Carroll 2005), the resultant value simply quantifies areas that have high, or low, values for both canopy and TRI and thus indexing the model results. We calculated this index using SPI-inventory data from 2010 to 2014 for Stirling. Additionally, we constructed this index for the entire area on and around Stirling using GNN data for canopy in addition to the TRI values for the same area.

SPI habitat classification

These analyses were performed based on the classification (hereafter “habitat model” or “model”) of forest types provided by Sierra Pacific Industries through a series of white papers

and in conversations with Ed Murphy. The model, as constructed for use in this analysis, is comprised of three variables: Overhead canopy closure, quadratic mean diameter at breast height (QMD [reported in inches]), and the number of trees per acre over 22 inches (TPA).

Sierra Pacific Industries uses canopy closure, QMD, and TPA to classify their forests in 4 categories: Habitat Form 1 (HF1), Habitat Form 2 (HF2), Habitat Form 3 (HF3), and Habitat Form 4 (HF4). Additionally, SPI creates a sub classification of forests in the HF2 class known as Habitat Form 2 heavy (HF2hv). These classifications are briefly described below with the exception of HF3 which has very limited distribution on Stirling and is therefore not considered further (Ed Murphy personal communication). For the purposes of this report we use those classifications as a discrete variable with a predicted hierarchy of importance to fishers. The forest classification we assume is least important, or most avoided, by fishers is HF1. It is defined as all other forest types not defined by the 3 remaining classifications. The next classification, Habitat Form 2 (HF2), is defined by forests with canopy cover greater than 40% and $QMD \geq 6''$. A subset of HF2 is Habitat Form 2 heavy (HF2hv) which are forest stands with canopy closure greater than 50% and $QMD \geq 11''$. The final classification level, and the one considered most appropriate for fishers, is Habitat Form 4, which is defined by forests with canopy closure greater than 60% and QMD greater than 13''. For the purposes of analysis we arbitrarily assigned the habitat form categories the values of $HF1 = 0$, $HF2 = 0.50$, $HF2hv = 0.75$, and $HF4 = 1.0$. The classification described above where implemented in ArcGIS with the following raster algebra implementation:

Eq. 2.

$$\text{SPI HabitatForm} = \text{Con}((\text{canopy} \geq 60) \ \& \ (\text{QMD} \geq 13) \ \& \ ((\text{TPA_22''}) \geq 9), 1,$$

$$\text{Con}((\text{canopy} \geq 50) \ \& \ (\text{QMD} \geq 11), 0.75, \text{Con}((\text{canopy} \geq 40) \ \& \ (\text{QMD} \geq 6), 0.50,$$

$$\text{Con}((\text{canopy} < 40) \ | \ (\text{QMD} < 6), 0))))$$

Where “Con” is a conditional function requiring the stated arguments (conditions) to be met to include in the classification. The “canopy” and “QMD” arguments are generic arguments that are substituted with canopy cover and mean QMD rasters for each specific year. We used forest inventory data provided by Sierra Pacific Industries for each year to construct spatial representations of their forest classification system on Stirling. We used these data to create a separate model for forest classification in each calendar year from 2010 to 2014. We did not create the SPI habitat form classification model using GNN data or other sources beyond SPI inventory data.

CWHR

The California Wildlife Habitat Relationships (CWHR) models exist for most animal species occurring in California. The CWHR model for fishers are based on the dominant woody vegetative types (e.g., Pondersosa pine, Sierran Mixed Conifer) as well as classification of canopy closure and average tree size (Airola 1988; Figure CWHR). Canopy closure is classified into 5 bins: none (0-10% canopy), sparse (10-24% canopy), open (25-39% canopy), moderate (40-59% canopy) and dense (>60%). We arbitrarily gave these bins the index values 0,1,2,3, and 4. Tree size (QMD) is classified into 6 bins: 1) seedling (<1” QMD), 2) Sapling (1”-6” QMD), 3) Pole (6”-11” QMD), 4) Small (11”-24” QMD), 5) Medium/large tree (>24”), and 6) Multi-layered trees which are stands with class 5 trees over a layer of 4 or 3 trees. We gave these classification bins the value 10,20,30,40, and 50, and 60. The classified values for canopy and

tree size were added together to produce a raster (code10) with discrete values ranging from 0 to 65. We did not to include stands classified as 60 (or multi-layered trees), because preliminary analysis demonstrated fewer than 2 km² across Stirling that could be potentially be classified (mean tree QMD > 24'') in this category. The values from the 'code10' raster were assigned a habitat classification value with equation 4 (Figure CWHR).

Eq. 4:

CWHR Habitat Quality =

$$\begin{aligned} &Con("code10" \leq 31, 0, Con(("code10" == 32) | ("code10" == 33) | ("code10" == 41) | \\ &("code10" == 51), 0.11, Con(("code10" == 34) | ("code10" == 51), 0.22, Con(("code10" == 42) | \\ &("code10" == 52), 0.33, Con(("code10" == 43), 0.66, Con(("code10" == 44), 0.89, Con(("code10" == \\ &53), 0.77, Con(("code10" \geq 54, 1.0))))))))) \end{aligned}$$

We used SPI-inventory data to classify canopy and tree size according to those rules. Habitat quality within the majority of tree stand types is identical with the highest habitat values occurring at the highest classification of both canopy and tree size (Figure CWHR). We constructed our final habitat model to follow these rules exactly as in the CWHR model with the exception of stands classified as Ponderosa pine dominated. The only differences between other stand types and Ponderosa pine stands occur in low quality stands (low canopy and tree class). Thus, we assigned habitat quality values to Ponderosa pine stands the same as implemented for all other stand types (equation 4, Figure CWHR).

Mean values

We recognized that most of the models in our tests have some variables in common (e.g., canopy and tree size). Generally, there is agreement among those models about how canopy and

tree size affect fisher habitat quality (positively). Nonetheless, the final predictions about habitat quality among habitat models are variable. We hypothesized that fisher use of habitat might be well predicted by identifying areas of agreement among models. Thus, we calculated the arithmetic mean habitat quality produced from the Allen, Carroll (1999), Carroll 2005 (index), CWHR, and SPI values. We considered this a 6th model that we chose to include in our analysis. We used only models constructed using SPI-inventory data to calculate this mean value. Additionally, because we constructed annual versions of each of the models used we likewise constructed a mean model for each year across Stirling. We also calculated the standard deviation among the models to evaluate areas, or components of habitat quality, that were consistent among models and those that were different.

Davis

We included the original 6 habitat models from Davis et al. (2007). The Davis models were directly downloaded from <ftp://ftp.biogeog.ucsb.edu/pub/org/biogeog/users/fd/fisher/> and were used as originally constructed by Davis et al. The Davis models were not constructed using either SPI or GNN datasets. Rather, they are the exact models produced by Davis et al. and based on the vegetation, terrain, and environmental data available during their original analysis. Clearly, these models are flawed with respect to predicting current habitat quality because of changes that occurred since their analysis. Changes within stands of trees and forests, including logging and wildfires, that occurred since 2006 are most obvious and important for the scope of our analysis. Additionally, the vegetation data used to construct these models are unlikely as accurate as those produced by SPI's forest inventory data. Nevertheless, the Davis models to incorporate data that would have been consistent across the region of our study area and thus provides insight into how well those models predict habitat quality at a landscape scale. We did

not attempt to reconstruct these models, as well as other models that used Generalized Additive Modelling (GAM) (e.g., Zielinski et al. 2010) because those models were not published with parameters that could be used to construct models with new data and new places (J. Dunk and B. Zielinski, personal communication).

Methods and Analysis

Analysis: Fisher Use and Resource Selection Functions

Individual Locations Analysis

We used locations of both female and male fishers from 2009 through 2014. Because we constructed habitat models for both area bounded by the Stirling boundary (hereafter Stirling) as well as the area on and around Stirling (hereafter ‘area’) we also used data from fisher locations at two different scales. For habitat models on Stirling we used fisher locations that occurred only on Stirling as well (10,011 total). This data set included a total of 4025 locations for females from estimated triangulations, walkins, flights, and captures. For males we used 5986 locations deriving from Argos locations of classes 1, 2, and 3, Global Positions Systems (GPS) collars, capture, and walkins. We generated 10,011 random points distributed throughout Stirling to estimate the proportion of habitat occurring on Stirling. We calculated the habitat quality of all locations for each habitat model and for the calendar year that the location occurred (Figure LocYear). Similarly, the habitat value for each model was calculated for all random points by year. Thus, allowing direct comparison of the habitat fishers used and that was available in each specific year. We quantified the proportion of both used and available habitat occurring within 0.10 bins for all models. We used the proportion of used and available habitat to calculate a resource selection function (RSF) based on the Ivlev electivity index (equation 5) and modified Ivlev electivity index (equation 6; Manly et al. 2007).

Equation 5:

$$Ivlev\ electivity\ index = \frac{proportion\ used - proportion\ available}{proportion\ used + proportion\ available}$$

Equation 6:

$$modified\ Ivlev = \frac{2 \times (proportion\ used - proportion\ available)}{1 + (proportion\ used + proportion\ available)}$$

We calculated the mean values for both the Ivlev and modified Ivlev across years, for each model, to generate a mean resource selection function. Habitat models that capture important aspects of what fishers are selecting should have a positive relationship between the predicted habitat quality values and the selection index of used vs available habitat. Thus, we used ordinary least squares linear regression in the statsmodels package for Python to evaluate the relationship between predicted habitat quality and the resource selection functions. We performed separate analyses for males and female as well as for all locations combined.

In our analysis we also constructed models for the region including Stirling and the surrounding area. Because SPI-inventory data is only available for Stirling but fisher selection occurred both on and off Stirling we needed data sets that were comparable. For these analyses we used all locations collected from males and females on both Stirling and off. In total, we used 12,591 points from fishers for this analysis to calculate locations and habitats that fishers used. To represent available habitat in the study area we calculated a Minimum Convex Polygon (MCP) in the minimum bounding geometry tool within ArcGIS 10.2 around all locations (see Figure AllenGNN). Within this MCP we generated 12,591 random points and the calculated the habitat value for each of those points based on the Allen (GNN data type), Davis (all six

models), and Carroll Index models. We then calculated both the Ivlev and modified-Ivlev resource selection functions (eq 5 & 6) and used regression to evaluate the relationship of predicted habitat quality to fisher use and avoidance for each model tested.

Female Utilization Distribution Analysis

Analysis of individual locations provides one measure of fisher use and avoidance of habitat. More specifically, analyses that focus on all points as equal, compared to the entire landscape, provide insight into where fishers settled after reintroduction and which habitat model describes these general patterns best. Nevertheless, fishers, and all animals, select home ranges because they provide important resources for their lives. In particular, female fisher habitat use is important to understand because they may have the most restrictive habitat requirements (e.g, appropriate dens and foraging requirements during reproduction). Thus, analysis of where female fishers establish home ranges, and their associated habitat quality, as well as the specific locations used within those home ranges provides a level of precision that goes beyond our more broad analyses. To evaluate home range habitat selection we created utilization distributions for female fishers using Kernel Density Estimation (KDE) approaches. We used a fixed kernel approach with a bandwidth of 750 m to smooth female locations during each calendar year from 2010 to 2014. We did not estimate a utilization distribution for a female if she had fewer than 20 locations for a single year. We categorized each utilization distribution by isopleths that bounded 25%, 50%, 75%, 90%, and 95% of the cumulative probability of occurrence. The smaller isopleths hypothetically represent areas of highest use, and ostensibly importance, for females. Larger isopleths are likely to envelope points on the periphery of a home range and should have the weakest level of habitat selection. To estimate the proportion of available habitat we extracted the habitat value at each cell within the estimated utilization distribution of a female

(by year). We then calculated the resource selection functions (eq 5 & 6) for each female by year and isopleth. We calculated a mean yearly mean RSF (both Ivlev and modified Ivlev) across females and regressed this value on predicted habitat quality. We next calculated a mean RSF values across years but by isopleth. Finally, we calculated a mean RSF score across isopleth. Analyses for the utilization distributions of females focused on the habitat models exclusively on Stirling and built with SPI-inventory data.

GPS Data Analysis

Most fisher locations used in previous analyses come from either Argos (male) or VHF telemetry (Powell et al. 2014, Facka et al, in preparation). Both these types of location data give accurate though relatively imprecise locations for fishers (>200 m error). Additionally, both these types of locations usually provide a maximum of 1 or 2 locations per day. As such, examining fine-scale spatial use or habitat use over fine scales is difficult. In 2012, 2013, and in the fall of 2015 we have opportunistically placed GPS collars on 2males and 2 females. These collars have been programed to record data at short intervals (<10 minute), but over a short total duration (<10 days) and on average provide relatively high precision for location estimates. Thus, we performed habitat selection analyses using GPS by itself for these four individuals. The GPS location data for both male (2012 & 2013) are included in our other location data; however, these data for females became available only in the fall of 2015.

We visually examined the GPS data from all four fishers and eliminated all data points that were obvious errors. For example, locations that was aberrantly distant from all other locations but within a similar time window was eliminated. As with our other analyses we then calculated the habitat value across for the Allen, Carroll (1999), Carrol index, SPI model, and the mean habitat models. We then quantified the proportion of points that fell within specified 0.1 habitat

quality bins. To quantify the proportion of random points we created a minimum convex polygon around all points for each individual fishers GPS locations. We then generated a number of random points within the MCP equal to the total points collected by the GPS collar for those fishers. We then used these proportions to calculate the Ivlev and modified Ivlev RSF and used regression to evaluate the general relationship between predicated habitat quality and habitat selection.

SPI Territory Opportunity

A proposed component of SPI's habitat designation and planning is the identification of "Territory Opportunities" for fishers. Territory Opportunities rely on the classification of forests into habitat forms, as described above, and then to classify hexagons on the landscape. Hexagons are classified into one of three categories: 1) A territory core (TC; hexagons with the highest habitat value for fishers), 2) A support core (SC: moderate habitat value), and 3) hexagons currently below threshold (CBT; hexagons with little or no value to fishers. The three categories are delineated by quantifying the amount and proportion of each hexagon and then identifying whether the hexagon meets the following three conditions:

1. Hexagon $\geq 30\%$ HF4
2. Hexagon $\geq 50\%$ HF4 and/or HF2H
3. At least one contiguous 30 ac. stand of HF4, which is also contiguous to at least an additional 20 ac. of HF2Hv (HF4 + HF2Hv ≥ 50 ac)

Hexagons that meet all three criteria are considered Territory cores whereas hexagons that only meet conditions 1 and 2, 2 and 3, or 3 are classified as support cores. Hexagons that do not meet the criteria for a territory core or support core are classified as currently below threshold (CBT).

To evaluate this approach we constructed a series of hexagons on and around Stirling that were 1.95 km² (481.5 acres). We then quantified the proportion and amount of HF1, HF2, HF2Hv, and HF 4 within each of those hexagons and used Zonal Statistics tools in the Spatial Analyst toolbox in ArcGIS 10.3. Each hexagon was then classified as a territory core (index value 2), support core (index value 1) or CBT (index value 0) for each year from 2010 to 2014.

Initially, we simply described the basic composition of female home ranges by hexagon value (type). We overlaid the utilization distributions of female fishers with the territory opportunity values for each hexagon. For each utilization distribution we calculated the proportion and value for each hexagon. We calculated the mean values across years and female utilization distribution. We calculated the types of hexagons that comprised female home ranges by examining the variety (range) of hexagons used and which type were most common.

The purpose of the territory opportunity method is to identify areas (hexagons) that can support fisher home ranges. Thus, we used two different, but similar analytic techniques to evaluate if a hexagon's classification was related to the likelihood that a hexagon contained a female's home range and number of female home ranges. We used logistic regression to determine if a hexagon's territory opportunity classification increased the likelihood that the hexagon intersected at least one female home range. Additionally, we used polytomous logistic regression to evaluate if a hexagon's territory opportunity classification increased the likelihood of a hexagon intersecting 1, 2, 3, or ≥ 4 female home ranges. For both analyses we used the portion of a utilization distribution that bounded 50% of the total probability distribution. The 50% isopleth is conservative in eliminating areas that are less often utilized and thus stresses areas that are ostensibly more important to females. Additionally, the 50% isopleth minimized intersections among multiple females within a specific hexagon that were attributable to short-

term forays or that occurred after major events (e.g., the death of a female). For both the logistic and polytomous logistic regression we also evaluated how differences among years, and geographic position may have influenced the probability of encountering a home range within a hexagon. Both forms of logistic regression rely on comparison between positive events (a home range within a hexagon) and negative events. Thus, in all years we randomly selected an equal number of hexagons that did not have a female home range to those that did. We performed logistic and polytomous logistic regression in Statistical Applications Software (SAS; Cary, NC) using Proc Logistic.

RESULTS AND DISCUSSION

Allen

Stirling locations

For locations occurring only on Stirling, the Allen model consistently predicted the highest amount of high quality habitat across Stirling. Mean available habitat quality on Stirling across all years was 0.71 ± 0.38 SD. Mean habitat quality decreased from 0.71 ± 0.38 in 2010 to 0.70 ± 0.39 in 2014. The distribution of habitat quality was similar among years with the approximately 50% of Stirling occurring at the highest habitat quality values (Figure AllenRandom). Habitat quality values below 0.2 comprised >0.15 percent of Stirling in all years with a slight increase from 2010 to 2014. Habitat quality values from 0.2 to 0.8 were a relatively small proportion of the landscape.

The average habitat quality of fisher locations was 0.83 ± 0.31 . Mean habitat quality used by fishers was constant and showed no strong patterns in change from 2010 to 2014 (range = $0.82 - 0.83$; Figure AllenSPI). The relationships between predicted habitat quality and both the Ivlev and modified Ivlev resource selection function were not statistically different from 0 with an R^2

value of 0.06 (Table 3; Figure AllenRSFalla). Nevertheless, the beta-estimate for this relationship was positive ($\beta = 0.14$) and both the Ivlev and modified-Ivlev RSFs indicate fishers avoided habitat quantified at low (<0.20), but selected habitat values greater than 0.90 (Figure AllenRSFalla). Habitat values from 0.80 – 0.90 were most obviously deviant from the predicted relationship. Fishers used areas quantified in the 0.80 – 0.90 far less than would be predicted by the amount of those habitats available indicating avoidance of those areas despite having relatively high predicted values according to the Allen model.

Female and male fishers appear to have used habitat similarly based on the Allen model. Mean used habitat for females (0.82 ± 0.31) and males (0.83 ± 0.30) was nearly identical across all years. Mean female habitat use was similar from across years with the minimum value occurring in 2012 (0.74), but in all other years this value ranged from 0.82-0.88 with no obvious trend through time indicating that fishers used habitat markedly different among years. Mean male habitat use was also relatively consistent from 2010 to 2014 (range = 0.75 – 0.88). Generally, male mean habitat use was slightly lower than females, but within the 95% confidence intervals. The mean Ivlev and modified Ivlev resource selection functions showed similar patterns among females and males (Table 3). Both demonstrated positive relationships between predicted habitat quality and patterns of usage, but neither were significantly different from zero. As with the more general analysis, female and males both avoid habitat that is quantified below 0.1 and select habitat over 0.90. Additionally, both females and males show a pattern of avoidance for habitat values from 0.80 – 0.90 (Figure Allensex).

Area locations

The Allen model, constructed using GNN data for the area on and around Stirling, had a mean available habitat value of 0.54 ± 0.39 (Figure AllenGNN). The highest predicted habitats

with GNN data were within the 0.8 – 0.90 value bins (36%). Habitat values less than 0.1 represented 27% of the total area whereas habitat with the highest values (0.9 – 1.0) comprised 21% of the area sampled. Moderate values of available habitat quality (0.1 to 0.79) totaled 17% of the total area (Figure AllenGNN). High quality habitat was distributed throughout much of Stirling as well as areas with similar elevation to Stirling. Low habitat quality was associated with elevations below Stirling.

Fishers selected higher quality habitat (0.78 ± 0.29) on average than was available. Fishers also appeared to avoid areas with low predicted habitat quality and selected for the highest quality habitat (Figure AllenGNNb). At moderate values of habitat quality fishers seems indifferent and used habitat in approximately the same proportion as was available on the landscape. Similar to the Allen model with SPI-inventory data, fishers used areas in the 0.8 – 0.9 habitat quality bin less than would be expected (Figure AllenGNNb). The relationship between predicted habitat quality and both the Ivlev and modified-Ivlev RSF were significantly different from zero ($\beta = 0.35$; Table 3). The results between the GNN and SPI-based models were similar in pattern but different with respect to how well predicted habitat quality related to either resource selection functions. Obviously, the models were produced using different vegetation information and different scales (stand vs cell based). Thus, differences between using 30×30 m pixels compared to entire stands may be important to the observed differences. Additionally, the GNN-based model sampled a wider area than just Stirling and resulted in different proportion of available habitat. The estimated differences in available habitat likely reflected a strong signal of habitat avoidance at very low habitat quality (e.g., more negative RSF scores). Differences between the estimation of vegetative components of the habitat model (e.g., percent hardwoods) may also explain differences between the model results. Importantly, the pattern of avoidance of

the 0.8 – 0.9 habitat quality is apparent across all analyses (model types, years, sex) and suggested an actual avoidance by fishers rather than a statistical anomaly or error in the way the models were constructed.

Female Home Ranges

The available habitat within female home ranges as quantified by the Allen model was similar to that observed for the entire study area (Fig. Allenrandom). Across all years the highest amounts of habitat (range = 0.56 – 0.67) were found in the highest habitat quality bins (0.9-1.0). Female home ranges incorporated the lowest habitat quality within their home ranges in low proportions compared to the highest (range across years = 0.10 to 0.13). The proportions of other habitat quality were also low compared to the highest habitat quality with values from 0.80 – 0.90 relatively common within female home ranges (range = 0.08 – 0.12).

The mean habitat value selected by females within their home ranges was 0.82 ± 0.31 . The mean habitat quality selected by females across years was nearly identical to the values from all female locations (range = 0.73 – 0.89) with no obvious trend through time. The relationship between female selection (resource selection functions) and predicted habitat quality was similar, though variable, across years (Figure AllenHRbyyear A). For the Allen model a consistent pattern emerged at every level of analysis – female, year, isopleth, and mean values (Table HRresults; Figure AllenHRbyYear). The pattern mirrors the pattern seen for other analyses with the Allen model – avoidance at low levels of habitat quality with relatively strong selection at the highest levels of habitat quality (Figure AllenHRbyYear). As with our previous analyses there was a strong avoidance in the range of habitat values from 0.8 to 0.9. This deviation appears to be the main reason that the Allen model predicted habitat quality does a poor job in explaining patterns of selection (Table HRresults). In fact, the RSF scores for this relatively high habitat

quality are often more strongly avoided than locations with the lowest habitat quality. At moderate habitat quality values (0.2 – 0.79) fishers show neither strong avoidance nor selection. Rather, fishers appear indifferent to these areas or perhaps use them only within the context of other high quality habitats. Because none of the models we tested is spatially explicit by taking into account distance between patches of different habitat quality we cannot dismiss that some areas are used simply because they are close to patches of high quality habitat.

GPS

The proportion of available habitat within the convex polygons of fishers with GPS collars was high variable (Figure GPSallen). Few fishers (n=4) were included in the GPS analysis and they occurred over different areas of the district. Thus some fishers had high proportions of high quality habitat (Figure GPSallen A) whereas other fishers higher proportions of moderate or moderately high habitat quality (Figure GPSallen B). As a result the relationship between habitat quality and RSFs for individual fishers showed high variability as well (Figure GPSallen). In some instances fisher selection was positively related to the habitat quality predicted by the Allen model whereas in other cases it was negative. The mean relationship, across the 4 collared fishers, was negative for the Allen model (Table GPSresults). This negative relationship, in part, relates to the strongly avoidance for habitat values in 0.8-0.9 ranges of predicted habitat quality. Additionally, fishers showed apparent indifference to habitat values that were very low. Fisher use of these areas in proportion to their availability is also related to the GPS data which collected points continuously through time. When fishers spend time stationary (perhaps resting for long periods) the collar continues collecting points. If the location of the fisher is in or near a boundary between low and high quality habitat the result is many locations occurring in poor

habitat (Figure Femharvest). In future analyses we will identify areas where fishers stationary and use a mean value for that location and thus remove this bias.

Though harvest or logging activity is not considered a model of itself we decided to include the age when a unit was logged as a variable to test with other GPS data. Females appeared to have used areas that have not been logged in the last 50 years. The year a stand was logged explained fisher use better than did other models we tested. The beta-value for this model is somewhat arbitrary because we set the value for stands harvested before 1970 to 0. This decreased the magnitude of the beta-value because the explanatory variable had a range of over 2000 units. Nevertheless, though a small parameter estimate, harvest year and intensity may be a value metric to examine more closely in future analyses.

Allen Overview

We emphasize for the Allen model that at every level of analysis the available habitat is biased toward the highest habitat quality bin. Yet, fishers use these areas disproportionately more than would be expected at random (Figure HR-compare). The fact that the Allen models predicts an abundance of high quality habitat but fishers use it preferentially suggests that something important to fishers is being captured by this part of the model. Indeed, among all habitat models the Allen model shows among the strongest selection values for this top bin. Similarly, the strong avoidance of low habitat qualities also suggests the model captures something important about places fishers avoid. At moderate levels of habit quality there may be confounding influences. In particular, the consistent avoidance of habitat values from 0.8 to 0.9 is interesting. Because we observed consistent avoidance of habitats within this range of values we explored the attributes of areas with those values compared to lower and higher habitat quality values (Figure AllenCompare).

We subdivided Stirling into three habitat types: habitat values < 0.79 , habitat values ≥ 0.80 & ≤ 0.90 , and habitat values > 0.90 . Within each of those three categories we calculated the mean values for each component of the Allen habitat model. The mean values for canopy, QMD, and canopy layers were all distributed as we would expect from the predictions of the Allen models (e.g., the highest canopy values occur in the highest predicted habitat quality). Yet, the percentage of hardwoods in a stand was lowest for habitats values from $0.80 - 0.90$. The original Allen model considered stands dominated by hardwoods as relatively poor stands. Forests, or stands of forest, that have no hardwoods, but have high QMD and canopy, are classified relatively high habitat quality (> 0.80). Thus, though not definitive, the results across the different Allen analyses indicate that fisher are avoiding stands with relatively high values for QMD, canopy and canopy layers, but with low proportions of hardwoods. Thomasma et al. (1994b) presented data suggesting the stand with low hardwoods were relatively poor habitat. In future refinements of the Allen model we will model the effect of hardwoods on the final habitat quality values similar to that of Thomasma et al. and evaluate differences between the two versions of the Allen model.

Carroll 1999

Stirling location

In contrast to the Allen model, the Carroll 1999 model predicted the highest amount of low quality habitat across Stirling. Mean available habitat quality on Stirling across all years was 0.30 ± 0.30 SD. Mean available habitat quality increased steadily from the low in 2010 of 0.29 ± 0.35 to 0.31 ± 0.36 in 2014. The distribution of available habitat quality was skewed towards low values with the approximately 50% of Stirling occurring at the lowest predicted habitat quality (Figure CarrollRand). Habitat quality values above 0.9 comprised roughly 10% of Stirling in all

years with a slight increase from 2010 to 2014. Habitat quality values from 0.2 to 0.8 were individually a relatively small proportion of the landscape, but were roughly equivalently distributed (Figure CarrollRand). Spatially, the areas with highest quality habitat were found along the lower portions of the West Branch of the Feather River, Butte Creek, and Big Chico creek (Figure Carroll).

The average habitat quality for locations fishers used was 0.50 ± 0.35 . Mean habitat quality used by fishers consistent across years, but did show a trend towards increased values in later years (range = 0.50 – 0.52). The relationships between predicted habitat quality and both the Ivlev and modified Ivlev resource selection function demonstrated a positive relationship statistically different from 0 with R^2 values over 0.50 (Table 3; Figure CarrollUsed). Both the Ivlev and modified-Ivlev RSFs indicate fishers avoided habitat quantified as low (<0.20), but selected habitat values greater than 0.90 (Figure AllenRSFalla). Additionally, habitat predicted by the Carroll model to be of moderate value was selected more than expected by fishers. Habitats with the highest values were selected, on average, more than expected by chance. Nevertheless, the highest habitat values were not the most strongly selected by fishers.

Female and male fishers appear to have used habitat similarly based on the Carroll model. Mean used habitat for females (0.51 ± 0.37) and males (0.51 ± 0.35) was similar in pattern but did display differences across years. Mean female habitat use varied across years with the minimum value occurring in 2010 (0.31), but increased steadily to 0.60 by 2014. Mean male habitat use displayed a less obvious temporal trend with the lowest mean used values occurring in 2011 (0.37) and the highest in 2013 (0.61). Generally, male and female mean habitat use was similar with the exception of increased values for females in the latter years. The mean Ivlev and modified Ivlev resource selection functions showed similar patterns among females and males

(Table 3). Both demonstrated positive relationships between predicted habitat quality and patterns of usage, but neither was significantly different from zero. As with the more general analysis, female and males both avoid habitat that is quantified below 0.1 and select habitat over 0.90.

Female Home ranges

The available habitat within female home ranges as quantified by the Carroll model was similar throughout most of the distribution to that observed for the entire study area. A notable exception occurred at the highest predicted habitat values. Within female home ranges the mean proportion of habitats with values from 0.9- 1.0 increased to 0.20 (range = 0.15 – 0.24). Across all years the highest proportions of available habitat were found in the lowest and highest habitat qualities. Thus, the distribution of available habitat was nearly bimodal with peaks at the ends of the range of habitat quality.

The mean habitat value selected by females within their home ranges was 0.46 ± 0.37 . The mean habitat quality selected by females across years with their home ranges showed a similar pattern to mean values from all female locations with the highest mean values occurring in 2014 (0.54 ± 0.35). The relationship between female habitat selection (resource selection functions) and predicted habitat quality was highly variable across years (Figure CarrollHRbyyear). The pattern of female habitat selection we observed within home ranges was markedly different from that seen at the scale of locations (Table HRresults). Female's within their home ranges appear to have selected high quality habitat in proportions lower than expected. In fact, the mean relationship between predicted habitat quality and RSFs for the 25% and 50% isopleth of a female utilization distribution were negative (Table HRresults). Because the Carroll model indicated the lowest amount of high quality habitat this result is somewhat surprising. Yet,

females apparently incorporated relatively high amounts of high quality habitat within home ranges and thus the proportions available within home ranges was different then the landscape generally.

GPS

The proportion of available habitat, as predicted by the Carroll model, within the minimum convex polygons of fishers with GPS collars was highly variable (Figure Carroll GPS). Similar to the Allen model the results of the Carroll analysis with GPS data were highly variable (Table GPSresults). The mean relationship between predicted habitat quality and fisher selection of habitat was positive though not significantly different from 1. In comparison to the Allen model analysis for GPS data we found no strong results. Mean RSF values were intermittently high throughout the distribution and generally there appeared to be no actual pattern. Additionally, fishers showed apparent indifference to habitat values that were very low.

Carroll Overview

The Carroll model was the most restrictive with respect to predicting high quality habitat across Stirling (Figure Carroll). Thus, the Carroll model also predicted the highest proportion of low quality habitat of all models tested (that were built with SPI inventory data). The overall utility of the Carroll model is difficult to assess because of conflicting patterns seen through different analyses (Tables 3 and 4). At the scale of individual locations the Carroll model did a reasonable job of predicting locations that fishers avoided. Locations above a predicted habitat quality of 0.3 were used by fishers disproportionate to their availability (Figure CarrollUsed). Nevertheless, beyond the habitat value of 0.3 the Carroll model did not show strong selection at the highest habitat values compared to the moderate values. The Carroll model performed very poorly at the scale of home ranges and with data collected from GPS collared fishers (Figure

CarrollhomerangebyYear). Within these analyses the predicted habitat quality was unrelated to the pattern of avoidance and selection by fishers. Spatially, we identify areas that the Carroll model predicted as low quality but used by fishers and in particular female fishers.

Because the Carroll model has many variables including two interaction terms we had difficulty understanding which, if any, variable, is consistently lacking or over-valued by this model. The Carroll model has the canopy closure QMD variables in common with the Allen model. Additionally, the Carroll model incorporates the percent of conifer in a stand and this metric is negatively linearly related to the percent hardwood (used in the Allen model). Nevertheless, the mathematical relationship among these variables to habitat quality is different from the Allen model and creates highly divergent results. The Carroll model was constructed as a descriptive model of fisher detection throughout California. Thus, Carroll et al. 1999 used variables such as precipitation, northing and northing² to explain fisher detection. The northing variables likely contributed to explaining fisher presence in the original model but reduce the generality of using the Carroll models (both 1999 and 2005) for predicting fisher habitat quality.

Carroll Index

Stirling locations

The Carroll Index, based on the Carroll 2005 model 3, had the second lowest predicted mean available habitat quality across Stirling. Mean available habitat quality on Stirling across all years was 0.41 ± 0.19 SD. Mean available habitat quality was effectively constant from 2010 to 2014 (range = 0.41 – 0.42). The distribution of available habitat quality was skewed towards moderate values but very little of Stirling occurred in habitat values over 0.70 (Figure CarrollIndex). Habitat quality values above 0.7 comprised less than 1% in all years. Habitat quality values from 0.2 to 0.5 were roughly equivalently distributed (Figure CarrollRand).

Because the Carroll index weighted terrain roughness and canopy cover equally the majority of high quality habitat occurred in location along drainages with steep terrain. Consequently, locations with relatively flat topography were predicted to have low quality habitat compared to areas with similar canopy cover but that were steep.

The average habitat quality for locations fishers used was 0.52 ± 0.15 . Mean habitat quality used by fishers was consistent across years, and did not show a trend towards increased through time (range = 0.51 – 0.52). The relationships between predicted habitat quality and the both the Ivlev and modified Ivlev resource selection function demonstrated a positive relationship that was statistically different from 0 with R^2 values of 0.51 (modified-Ivlev) and 0.72 (Ivlev) (Table 3). Both the Ivlev and modified-Ivlev RSFs indicate fishers avoided habitat quantified below 0.50, but selected habitat values greater than 0.50 (Figure CarrollRSF). The pattern of avoidance and selection as stark with a clear divide at habitat values as 0.50. Habitat above 0.50 was not necessarily selected concordant with the predicted habitat quality. Nevertheless, those differences may emerge simply because of sampling error because few locations on Stirling were available with the highest values.

Female and male fishers used habitat similarly based on the Carroll index model. Mean used habitat for females (0.53 ± 0.14) and males (0.51 ± 0.18) and did not display differences across years. The mean Ivlev and modified Ivlev resource selection functions showed positive though beta estimates that were not significantly different from zero (Table 3).

Area locations

The Carroll index model, constructed using GNN data for the area on and around Stirling, had a mean available habitat value of 0.29 ± 0.21 (Figure AllenGNN). The highest predicted habitats with GNN data were within the 0.4 – 0.70 value bins (71%). Habitat values less than 0.4

represented 27% of the total area whereas habitat with the highest values (0.9 – 1.0) comprised less than 2% of the area sampled. Moderate and high quality habitat was distributed along areas with steep terrain and thus major drainages (e.g., Feather River, and Butter creek).

Fishers selected moderate to high quality habitat (0.43 ± 0.12) on average than was available. Fishers also appeared to avoid areas with low predicted habitat quality and selected for the highest quality habitat (Figure Carrollin_GNN). Fishers selected for locations in the area with predicted habitat values above 0.60. There were differences observed between the Ivlev and modified-Ivlev RSF (Figure Carrollin_GNN). At habitat values above 0.80 the Ivlev RSF indicated avoidance by fishers whereas the modified-Ivlev RSF shows indifference. There were relatively few cells within these high habitat values and these results are likely due to sampling errors. The relationship between predicted habitat quality and both the Ivlev and modified-Ivlev RSF were not significantly different from zero (Table 3). Similar to the comparison with the Allen model, the two versions of the Carroll index were produced using different data and different scales (stand vs cell based). The Carroll index is different because it has fewer parameters (TRI and canopy). Canopy should be a relatively robust measure between data sets (data types) and TRI was the same in both models. Thus, the scale of the analysis (individual cells as well as a broader region) is likely the major contributor between the results of the two models.

Female Home ranges

The available habitat within female home ranges as quantified by the Carroll index model was similar throughout most of the distribution to that observed for the entire study area (Fig CarRand). The highest proportion of habitat occurred in the values from 0.5 to 0.6 with moderate

amount at lower values. There were annual differences in the proportion of available habitat but minor in magnitude.

The mean habitat value selected by females within their home ranges was 0.58 ± 0.14 . The mean habitat quality selected by females across years within their home ranges was relatively invariant (range = 0.51 – 0.53) with no trends through time. The relationship between female habitat selection (resource selection functions) and predicted habitat quality was variable across years but consistent with other analyses with this model (Figure CarrollHRbyyear). In all years, and for the mean across years, female fishers avoided areas with low predicted habitat quality but selected locations with high predicted habitat quality. The pattern of female habitat selection we observed within home ranges was similar to that seen at the scale of locations on Stirling (Table HRresults). Female's within their home ranges appear to have selected high quality habitat in proportions lower than expected. There was positive relationship between predicted habitat quality and RSFs for all isopleths of a female's utilization distribution (Table HRresults). Generally, the Carroll index model explained home range selection well at a broad scale of poor habitat (<0.50) and good habitat (>0.50). Additionally, the selection scores were among the highest of all models tested (Figure HR-compare). Similarly, selection scores for avoidance were also strong compared to other models with the exception of the Allen model at the 0.8 to 0.9 habitat quality range.

GPS

The proportion of available habitat, as predicted by the Carroll index model, within the minimum convex polygons of fishers with GPS collars was highly variable (Figure CarIndexGPS). The pattern of fisher response to the predicted habitat quality at the scale of GPS data was different from that seen at other levels (Table GPSresults). The mean relationship

between predicted habitat quality and fisher selection of habitat was effectively zero ($\beta = -0.010$), with no strong trend showing avoidance or selection. In comparison to the Allen model analysis for GPS data we found no strong results. Mean RSF values were generally low than expected, given the proportion of habitat, for high habitat values. As we noted with GPS data and the Allen model, these results may have something to do with more points collected at resting sites that border poor quality habitat. Additionally, because the GPS data is collected over a short amount of time the results are much more prone to sampling error than are other analyses. The nature of the Carroll index model may simply be too general to provide precision for the types of locations that are provided by GPS collars.

Carroll Index overview

The Carroll index model consisted of only 2 variables – TRI and canopy. Our index of the original model used a scaled mean value of both variables. TRI ranges from 0 to 160 and thus the highest values within our scaled TRI variable would occur at values of 160. Yet, the mean value for TRI on Stirling was 22 ± 13 (Table TRI_coar). Thus, the mean value for TRI in our model was only 0.14. The average for canopy and TRI would rarely approach 1.0 and hence there were few locations with predicted habitat value ≥ 0.80 . Consequently, the Carroll index cannot be thought of in the same way as other models we tested in terms of the values that represent good vs poor habitat. Nevertheless, the pattern of fisher use for this index was surprisingly consistent at the level of all locations across Stirling and within female home ranges (Tables 3 & HRresults). Fishers consistently avoided areas with low habitat quality values (low canopy and flat terrain). Locations with steeper terrain and more canopy appear favored which supports the earlier model of Carroll 2005 although we emphasize this is only an index of that model.

Regardless, the inclusion of only canopy and TRI demonstrated better resource selection values by fishers than most other models regardless of the data type or scale of analysis.

The canopy metric is nearly omnipresent in the models we tested and for fisher habitat models generally. Locations without canopy from trees appear avoided by fishers on average. Measurement of terrain ruggedness have appeared in other models (e.g, Davis et al. 2007). Nevertheless, terrain ruggedness, of itself, likely offers very little to fisher requirements. On Stirling TRI is correlated with several other variables that are potentially important for fishers (Table TRI_coor). The percent hardwood is positively associated with TRI while percent conifer is negatively related. The results from the Allen model indicate that at Stirling, hardwoods may be important for fisher selection even when other attributes, such as QMD or canopy, are present. Additionally, canopy itself is correlated with TRI as are the number of logs per acre, tree diversity, and tree height (negative relationship). Obviously, many of these variables are themselves correlated. For example, areas with more hardwoods are likely to have high tree diversity; moreover, places with high tree diversity could be expected to have lower average tree height because of a greater proportion of sub-story trees. Steep terrain may reduce the amount of road building, logging and other management activities that alter many of the variables we mentioned. Thus, we hypothesize that TRI serves as an indicator of areas with low management, and high forest complexity (diversity) rather than being important itself. Within the Carroll index model TRI is likely accounting for many variables simultaneously that could be modeled individually. The inclusion of a terrain-related variable will likely always outperform models with the inclusion of multiple variables because terrain will explain similar variation with fewer estimated parameters. Nevertheless, future work should seek to understand which variables are truly important to fishers rather than relying on variables such as TRI that are unlikely directly

related to the requirements of fishers. At broad scales identifying areas of steep terrain may be a way of refining early assessments of habitat. This, of course, is only viable if our hypotheses concerning management and terrain ruggedness are valid.

CWHR

Stirling locations

The CWHR model predicted modest, relative to other models, amounts of available habitat on Stirling. The mean available habitat quality on Stirling across all years was 0.57 ± 0.37 SD. Importantly, the mean value for the CWHR model is not a value that is actually possible since this model allows for only 7 actual habitat values (Figure CWHR). There was almost no change in mean available habitat from 2010 (0.58) to 2014 (0.57) although 2012 was the lowest year of predicted available habitat at 0.56. The distribution of available habitat quality was skewed towards the highest values with > 50% occurring at the highest predicted habitat quality (Figure CWHRrand). No value higher than 0.89 was described on Stirling although a peak value of 1.0 is possible. Predicted values greater than 0.89 did not occur on Stirling because no stand has sufficient mean QMD to be classified in the ‘medium/large’ tree category needed for the top value in the CWHR system (Figure CWHR). Habitat quality for other habitat values was roughly uniformly distributed in most years. Spatially, areas with the highest quality habitat were found throughout most of Stirling. Low habitat quality occurred in recent clear cuts, areas with fires in the last 10 years and in some locations dominated by hardwoods.

The average habitat quality for locations fishers used was 0.67 ± 0.33 . The mean CWHR habitat quality used by fishers was roughly equal across years (range = 0.67 – 0.68). The relationships between predicted habitat quality and both the Ivlev and modified Ivlev resource selection function demonstrated a positive relationship but statistically indifferent from zero

(Table 3; Figure CWHR_RSf). The Ivlev and modified-Ivlev resource selection functions indicated slightly different relationships between CWHR habitat quality and fisher selection. The Ivlev RSF showed fishers strongly avoiding habitats quantified as less 0.89 (the top value) whereas the modified-Ivlev showed avoidance at the lowest habitat values but relative indifference in moderate values. Both RSFs indicated selection for habitats predicted as the highest quality.

Female and male fishers used habitat classified by the CWHR model virtually identically. Mean used habitat for females (0.68 ± 0.33) and males (0.68 ± 0.34) was indistinguishable as was mean habitat use across years (results not shown). Similarly, male and female selection and avoidance were similar to the pattern observed with the combined data set (Figure CWHR_RSf)

Female Home ranges

The available habitat within female home ranges based on the CWHR habitat model tended was similar to that of habitat available across Stirling. Habitat in the highest quality interval was higher in each year within home ranges (range = 0.60-0.67) compared to across Stirling generally. Concordantly, the proportion of available habitat at lower values was reduced.

The mean CWHR habitat value selected by females within their home ranges was 0.69 ± 0.32 . The mean habitat quality selected by females across years (range = 0.76 – 0.67) was generally similar with the lowest value occurring in 2012. The relationship between female habitat selection (resource selection functions) and predicted habitat quality showed a similar pattern of avoidance at low habitat quality values across years (Figure CWHR_HR A). Similarly, fishers appeared to show selection for habitat in the highest quality habitats disproportionate to their availability (Figure CWHR_HR). The RSFs functions were better explained by habitat quality values at low ($\leq 50\%$) isopleths with highest R-squared values and significantly positive

beta-values (Table HRresults). On average, female fishers appeared to avoid habitat with a CWHR value of 0.66 which are delineated as stands with trees 28 to 60 cm (11-24") and canopy cover of 40-59%. In comparison to other habitat models at the home range scale, the CWHR showed a consistent pattern of selection by fishers. Nevertheless, fishers did not avoid areas with the lowest CWHR values as strongly as other models (Figure HR-compare). Moreover, fishers showed relatively weak selection for the highest CWHR habitat values compared to models such as the Allen, and Carroll index (Figure HR-compare).

GPS

The proportion of available habitat, as predicted by the CWHR fisher model of habitat, within the minimum convex polygons of fishers with GPS collars demonstrated similar variability as other habitat models (Figure CWHR_GPS). Similar to the Allen model the results of the Carroll analysis with GPS data were highly variable (Table GPSresults). The mean relationship between predicted habitat quality and fisher selection of habitat was positive though not significantly different from 1. Additionally, a consistent pattern emerged among all models with the GPS data in which individuals that used poor habitat in high amounts did so across models (Table GPSresults).

CWHR Overview

The conception of the CWHR model is straightforward and places the highest value on locations with large trees, of all types, and high canopy cover. Generally, this approach seems to predict roughly fisher selection and avoidance though not at all scales. The overall pattern of selection vs predicted habitat quality is positive across sexes, locations, and home ranges (Table

3 and 4). Nevertheless, in contrast to the Carroll index model and the Allen model the mean selection and avoidance scores are relatively weak (Figure HR-compare). Additionally, the available evidence suggests that fishers avoid areas with moderate habitat quality characterized by trees less than 60 cm (24'') and moderate canopy closure (<60%). The CWHR model does not incorporate attributes relative to the types or proportions of trees within a stand (e.g., deciduous). The Allen model suggested strong avoidance of areas with similar sized trees and canopy as those avoided at the CWHR 0.66 value but that lacked hardwoods. Thus, the attributes associated with size in the CWHR model may also reflect lack of other components such as hardwoods. Values for the CWHR model from 0.6-0.7 overlapped with the Allen model values of 0.8-0.9 on 10.4% of Stirling (Figure ModOverlap). These areas of overlap between the Allen and CWHR model had nearly no overlap with female utilization distributions at the 50% isopleth. These areas of moderate value occur primarily in the northern and eastern portions of Stirling where geology and elevation are different compared in comparison to much of Stirling. Future use and testing of the CWHR model may place added emphasis on understanding why fishers seem to avoid areas in classified by moderate values.

SPI Habitat Form

Stirling locations

The SPI habitat form model predicted the second highest mean available habitat of all models tested on Stirling. The mean available habitat quality on Stirling across all years was 0.59 ± 0.37 SD. The SPI habitat model was restricted to 4 values representing the four classes and thus the mean value is not directly comparable to other models. Available habitat was distributed roughly evenly across the 4 classes with the lowest values occurring at the 0.50 value (HF2; Figure SPIrand). Habitats in the two highest categories were roughly equivalent and represented

the majority of predicted habitat in all years. There was almost no change in mean available habitat from 2010 (0.60) to 2014 (0.59). Low habitat quality occurred in recent clear cuts and areas with fires in the last 10 years.

The average SPI habitat fishers used (mean 0.72 ± 0.30) was greater than that available. The mean SPI habitat quality used by fishers was roughly equal across years (range = 0.71 – 0.73). The relationships between predicted habitat quality and both the Ivlev and modified Ivlev resource selection function demonstrated a positive relationship that is statistically different from zero (Table 3; Figure SPI_RSf). The Ivlev and modified-Ivlev resource selection functions showed very similar relationships between SPI habitat quality and fisher selection (Figure SPI_RSf). Both functions indicated fishers avoided habitats classified as HF0 (0) but selected HF2hv and HF4 disproportionate to their availability. Nevertheless, the strength of the selection index was weaker than seen in the Allen and Carroll index and similar to that of the CWHR model.

Female and male fishers used habitat classified by the SPI habitat form model virtually identical (Figure Meanhab_sex). Similarly, male and female selection and avoidance were similar to the pattern observed with the combined data set (Table 3) and showed positive relationships with relatively high r-squared values.

Female Home ranges

The available habitat within female home ranges based on the SPI habitat form model was similar to that of habitat available across Stirling. The distribution of available habitat was also similar across years from 2010 to 2014. Nonetheless, the proportion of habitat in HF1 increased slightly from 0.14 in 2010 to 0.16 in 2014. The highest proportion of habitat in the HF4 category occurred in 2012 (0.34) and was lowest in 2014 (0.30).

The mean SPI habitat value selected by females within their home ranges was 0.73 ± 0.28 . The mean habitat quality selected by females across years (range = 0.72 – 0.79) was generally similar with the lowest value occurring in 2012. The relationship between female habitat selection (resource selection functions) and predicted habitat was positive in regardless of the isopleth where we evaluate selection (Figure HRresults). We observed an apparent avoidance of the HF2hv classification at the scale of the home range (Figure HRresults) within the SPI habitat form classification. Thus, indicating the females preferentially used only the highest classification habitat quality even with HF2hv was within their home ranges.

GPS

The proportion of available habitat, as predicted by the SPI fisher model of habitat, within the minimum convex polygons of fishers with GPS collars demonstrated similar variability as other habitat models (Figure SPI_GPS). The mean relationship between predicted habitat quality and fisher selection of habitat was positive though not significantly different from 1. Additionally, a consistent pattern emerged among all models with the GPS data in which individuals that used poor habitat in high amounts did so across models (Table GPSresults). Additionally, fisher avoidance of areas classified as HF2hv was apparent in the GPS data as well as the home range analyses.

SPI Overview

The SPI habitat form model is similar to other models of fisher habitat in that it predicts areas the fishers use reasonably well. Additionally, it consistently predicts that fisher select areas of relatively high QMD and canopy closure. The SPI model performs similarly to the CWHR model and the Allen model for predicting the high class of fisher habitat (Figure HR-compare).

At moderate values (specifically the HF2hv category) fishers appear to have weak selection for this class at the scale of the entire district. Within female home ranges this category was avoided consistently (Figure SPI_HR, SPI_GPS). Habitats described by the HF2hv are not identical to other values produced in other models, but is similar to that of the CWHR model values of 0.66-0.77. In both the CWHR and SPI model stand within this moderate and moderate-high values reflect medium sized tree QMD (>11" but often less than 24") and canopy closures below 60%. Additionally, we demonstrated overlap with these moderate values of the CWHR and the Allen models. Consequently, much overlap exists between the HF2hv classification and the Allen and CWHR models that were avoided by fishers over the period of observation.

Mean model

Stirling locations

The Mean habitat model we created from other models tested predicted median available habitat value (Figure Mean_rand) of all models tested on Stirling. The mean available habitat quality on Stirling across all years was 0.54 ± 0.29 SD. The Mean habitat model showed peaks in available habitat at the lowest habitat values at those in range of 0.7-0.8 (Figure Mean_rand). Generally, available habitat not at the lowest bin increased steadily until values of 0.75 and then declined again. Mean available habitat was invariant from 2010 to 2014 with a change of 0.004 seen among years.

The average Mean habitat fishers used (mean 0.68 ± 0.24) was greater than that available. Indeed, across all the models used to calculate the mean model fishers always selected habitat value higher than available at random. The mean Mean habitat quality used by fishers was roughly equal across years (range = 0.67 – 0.69). The relationships between predicted habitat quality and both the Ivlev and modified Ivlev resource selection function demonstrated a

strong positive relationship that was statistically different from zero (Table 3; Figure Mean_RSF). The relationship between habitat predicted by the Mean model and the Ivlev function was nearly linear and exhibited high avoidance at low levels with high selection at high values. The relationship between predicted habitat and the modified-Ivlev resource selection functions appeared non-linear with relatively weak avoidance at low habitat values followed by relative neutrality at moderate values and then weak selection at high values. Nevertheless, both patterns are consistent with a model that explains fisher avoidance and selection as we predicted.

Female Home ranges

There was little difference between the distributions of available habitat seen across Stirling compared to within female home ranges. Additionally, because we built the Mean model by taking a mean value from other models there was little change in the distribution of available habitat through time.

The mean SPI habitat value selected by females within their home ranges was 0.73 ± 0.28 . The mean habitat quality selected by females across years (range = 0.65 – 0.70) was similar with the lowest value occurring in 2012. The relationship between female habitat selection (resource selection functions) and the predicted mean habitat was consistently positive but the magnitude of these relationships was weak and not statistically different from 0 (Table HRresults). We observed an apparent avoidance of areas with low predicted habitat qualities, but areas with habitat values >0.2 were generally used in proportion to their availability.

Mean Overview

There was strong agreement among models in quantifying very and very high habitat for fishers (Figure MeanSTD). Locations with habitat values near 0 or 1.0 showed the lowest amounts of variation (standard deviation). Indeed, the strongest disagreement among models in

predicted habitat comes at the moderate values from 0.3-0.7 (Figure MeanSTD). Indeed, this is also where we observe high discrepancy among models in predicting fisher selection of habitats. Moreover, at this moderate values is also where the much contention concerning fisher and forest management converge. Among the models we have tested many of the moderate values for habitat are actually avoided consistently and often strongly. The reasons fishers chose to avoid such areas is not completely known, but most models rely on metric for only QMD and canopy closure. The Allen Carroll (1999) model incorporate stand type more explicitly but still yield, but those parameters may need adjustment. In particular, we must retest the Allen model after incorporating a penalty for stands that lack all hardwoods.

The Mean habitat produced results that explained fisher habitat selection well at the landscape (Stirling) scale (Figure Mean_RSF). Nevertheless, within home ranges Mean habitat provided poor explanatory power (Table HRresults). The Mean model may provide a good metric for identifying areas of poor habitat the fishers tend to avoid, but selection of the habitat may simply create home ranges that are already proportioned in ways that fishers will use that area. More work is needed to support this conjecture, but using Mean model outputs may be an appropriate way to identify important places across landscapes for fishers. We used a simple arithmetic mean value across a series of models, but more refined approaches may yield better results. We encourage additional work that incorporates consistency among models as a way to identify important areas for conservation and management.

Davis

Area locations

The Davis models all were highly skewed towards low values of predicted available habitat in the area on and around Stirling (Figure DavisAvailable). Five the 6 models predicted the

proportion available habitat as over 60% (Figure Davis Map). Because we used the models as initially developed we did not create models by years and thus do not account for changes through time or that occurred since the original model was published.

All the Davis models resulted produced low values for mean used habitat by fishers (range = 0.009 – 0.17). Further, the relationship between predicted habitat quality and fisher selection was consistently negative suggesting fishers studied from 2010 to 2014 used high quality habitat at rates lower than expected by random (Table 3). In all models fishers did avoid areas predicted as the lowest quality habitat (both Ivelev and modified-Ivlev). Yet, fishers used areas of moderately low habitat quality (0.2-0.6) at rates higher than expected by random (Figure Davis_RSF).

Territory Opportunity

We evaluated 300 individual hexagons from 2010 to 2014 for overlap with female 50% utilization distributions (Figure TerritoryCore). Of these 300, 150 (50%) never overlapped with the utilization distribution of a female, 91 (30%) overlapped with 1 female utilization distribution, 28 (9%) with 2, 19 (6%) with 3 and 12 (4%) with 5. The total number of hexagons with at least one utilization distribution increased in each year with a slight increase in the number per hexagon (Table HexNum). Hexagons that never overlapped with a female utilization distribution were on average of lower quality than were hexagons with ≥ 2 , but with high variation (Figure MeanHexVal).

Female fishers had 50% utilization distributions that intersected with a diversity of hexagon types. The majority (86%) of females had utilization distributions with hexagons that were classified in all three categories. On average female utilization distributions were composed of 29% hexagons of the CBT, 34% SC and 36% TC (Figure Terr_dist). Nevertheless, female utilization distributions were composed of a wide variety of different hexagon values. On

average the 50% utilization distribution was composed of $4.98 \pm 4.2 \text{ km}^2$ identified as a support core.

Our logistic regression analysis indicated that the value of a hexagon increased the probability that a home range (utilization distribution) would occur within it (Table TerrOppResult). Models that incorporated the classification of the hexagon as an explanatory variable were more well supported ($\text{AIC}_c \geq 10$) than models that incorporated yearly differences, location (easting and northing) or a random variable (Table TerrOppResult). The model that best explained the probability of a hexagon having a home range incorporated hexagon value as a discrete variable. A model treating hexagon value as a continuous variable was second best supported. Both models indicate a positive relationship between the likelihood of finding a home range within a hexagon and the assigned value of the hexagon. Nevertheless, the discrete model explained more variation and offers an important insight. A hexagon classified as below threshold (CBT) will be 0.3 (95% CI: 0.2-0.5) times less likely to have a female home range as will hexagons classified as SC or TC (Table TerrOppResult). Yet, a hexagon classified as TC is not statistically any more likely to have a home range than one classified as SC. The continuous model for hexagon value predicts an increased likelihood at all levels. Hexagons classified as CBT were have probability of holding a home range of 0.35, whereas one classified as TC had a probability of holding a home range of 0.66.

The polytomous logistic regression had very similar results to the logistic regression. The value, or classification, of a hexagon increased the likelihood that the hexagon would hold a female home range. This model better explained the probability of encountering a home range than did year effects, random effects or changes in direction east or north (results not shown). The probability of a hexagon having 1 (Odds ratio = $1.8 \pm 1.49\text{-}2.34$ 95%CI) or 2 (Odds ratio =

2.37 ± 1.6 -3.5) home ranges increased with hexagon value. Having 3 or 4 home ranges did not increase with hexagon value, but there were few hexagons with that many home ranges and thus we had little statistical power to detect an effect. Future analyses should bin all values over 2 home ranges together.

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Fisher (*Pekania pennanti*) prey availability and habitat use on managed timberlands in Northern Sierra Nevada

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Summary

The relationship between predators and their prey is among the most important but least studied aspects of carnivore ecology. While many studies document where animals live to develop habitat associations there are few studies that identify spatial distributions and abundances of prey species to understand better carnivore habitat needs. Human-caused disturbance through industrial loggings is hypothesized to diminish fisher (*Pekania pennanti*) habitat quality. Yet, the direct effects of logging on fisher prey species are unknown and generally unconsidered. We investigated potential fisher prey occupancy, relative abundance and diversity on an industrial landscape where fishers were recently reintroduced. We sampled 250 locations using live-trapping and remote cameras over 3 years (2012, 2014, and 2015) to identify where prey species were most likely to occur and what habitat attributes best predicted their presence, relative abundance and general diversity. We quantified vegetation metrics including canopy cover, basal area, mean quadratic mean diameter of trees, number of logs, and density of roads across 3 scales of measures (the microsite, the stand, and the neighborhood) to identify how forest management influenced prey species. We detected 44 genera of mammals, birds and herpetofauna and did detailed analysis on the most widely distributed and detected taxa (*Peromyscus* spp, *Neotoma* spp, *Tamiasciurus douglasii*, *Sciurus griseus*, *Otospermophilus beecheyi*, and *Glaucomys sabrinus*). We identified complex relationships between forest metrics and management and species occupancy and diversity. Nevertheless, we found consistent support for metrics associated with average trees size and the number of downed logs as consistent drivers of occupancy, abundance and

diversity. Generalist species including deer mice and woodrats exhibit avoidance or indifference to areas of high mean tree diameter whereas tree squirrel species exhibit selection for such sites. Other metrics associated with mature forests including down logs were also important drivers of occupancy for many tree species which appear important to fishers.

Introduction

The patterns and mechanisms that relate disturbances to species presence and biodiversity are critical to understand in ecology and management (Gunther et al. 1983, Pacala & Rees 1998, MacDougall et al. 2013, Dirzo et al. 2014). Many explanations contribute to our understanding of disturbance, particularly human-disturbance, and species presence and community assemblages (Pacala and Rees 1998). Fundamentally, disturbance affects animal presence and abundance by increasing local extinction rates in a location which is subsequently recolonized by animals at different rates and times (Kondoh 2001, Fox 2013). Organisms recolonize a disturbed patch based on their ability to disperse from source locations and such abilities are often mediated by the habitats between habitat patches, populations or sub-populations (MacArthur 1967, Bowman & Fahrig 2002, Hargreaves & Eckert 2014). If organisms successfully disperse to a previously disturbed area they will only persist if sufficient resources occur at the location post-disturbance (Hargreaves et al. 2014). Often studies that examine disturbance and species diversity examine only a single aspect of the community without regard for higher trophic interactions. Predatory species may be affected by both initial disturbances, but also by the response of their prey species to those same disturbances. Much effort has been put forth to understand how and where predators should forage and what species they should eat. Generally, we anticipate the predators will seek the largest prey they can catch because it increases net benefit (Emlen 1966, Charnov 1976) and they may not forage in locations with low ranking prey. In cases where one prey type is

highly abundant, or preferred, predators may forage only where this species occurs (Emlen 1966, Powell 1975). Additionally, particularly for generalist predators, they may utilize a suite of prey that are locally available, and thus locations with diverse prey may be favored because they maximize the likelihood of success for at least one prey type (Pyke et al. 1977, Powell 1979). The complex interactions between disturbance, prey availability, and predator response are poorly investigated but are ecologically interesting and important for conservation and management (Moorcroft et al. 2006). Attempts to understand and manage predator's responses to disturbance depend on knowledge of both the needs of the predator as well as members of the prey community or the community as a whole. Some species may be unable to disperse to a location though the habitat would support them (Bowman & Fahrig 2002, Bowman et al. 2002, Hargreaves et al. 2014), whereas some may successfully colonize a disturbed location but cannot persist because of inadequate habitat. These factors confound our observations about the predators habitat use and response to disturbances because animals may use or avoid an area that is sufficient in one way (e.g., denning or resting habitat) but insufficient in others (abundant prey). Without understanding how disturbance limits prey presence and abundance there is no tractable way to predict and manage the concordant response of predators.

Industrial clear-cut logging creates multiple patches of disturbance across landscapes by cutting and removing live trees and structural components. This may lead to localized extinction of some, or all, prey species (Carey & Johnson 1995, Mitchell et al. 1997, Carey et al. 1999, Carey 2000, Carey & Harrington 2001). Conversely, generalist species, including some small mammals, may show increases or only moderate declines in abundance and diversity following clearcutting in some locations (Kirkland 1977, Gunther et al. 1983, Macquire et al. 2008). In commercially managed landscapes clearcut areas are planted with trees that regenerate for many

decades before being harvested, and the resulting landscape becomes a series of patches that were disturbed at different times (a 'chronosequence'; Urban & Swihart 2011). Patches of regenerating timber are of different ages and reflect the forestry and management regimes of that time resulting in differences in vegetative composition (e.g., the ratio of hardwoods to conifer trees), tree density, and structural characteristics. These differences in vegetation, structure and overall habitat components may preclude persistence of some species in particular areas with large numbers of trees but few other elements that foster abundance animal populations. Further, as a high percentage of the landscape is disturbed fewer patches may serve as sources for animals that exist on the landscape. Years of human-caused disturbance may simply alter how animals respond to forest structure and type until there are no discernable patterns or responses that occur generally (Kelt et al. 2013). Thus, species may respond to disturbances or overall habitat conditions at multiple scales (Orrock et al. 2000, Bowman et al. 2001, Coppeto et al. 2006). Consequently, landscapes that are heavily logged provide a unique opportunity to examine how animals are effected across multiple spatial and ecological scales and to investigate the dynamics and habitat needs of prey species caused by intense disturbance at multiple spatial scales.

Small and medium sized mammals are the major prey species of fishers (*Pekania pennanti*), a mid-sized carnivore that lives in coniferous forests in North America (Powell 1993). Fishers are opportunistic predators (Powell 1993, Zielinski et al. 1999, Golightly et al. 2006), but they generally rely on small mammals including lagomorphs (rabbits and hares) and rodents (e.g., sciurids, murid rodents, and porcupines). Yet, fishers also preferentially select specific animal types (e.g., porcupines) that will maximize their energetic benefit while minimizing the effort needed to obtain them (Powell 1979, 1993, Powell et al. 2016). Specific forest elements, including downed woody debris, may further increase the ability of fishers to capture prey (Andruskiw et al. 2008). The availability,

and spatial distribution and density, of these resources are therefore important to fisher survival and reproduction. To understand better the relationships between fishers, prey and habitat we investigated presence, abundance, and diversity of fisher prey on a landscape where active industrial logging coincides with a burgeoning fisher population.

Beginning in late 2009, extending through 2011, fishers were reintroduced onto an industrial logged landscape in California creating an ideal environment to study the response of a mammalian predator to short and long-term disturbance (Callas & Figura 2008, Powell et al. 2012). Inference to the locations fishers use, and consequently, their response to logging, require an understanding of their prey and how they respond to logging. Without this knowledge, or patterns that govern them, the locations they select, or avoid, are confounded among many variables and thus reduces our ability to resolve aspects of habitat that are important for fishers.

We studied potential prey species of fishers during the summers of 2012, 2014, and 2015 to estimate occupancy (presence), abundance and overall diversity of taxa identified as fisher prey. In general, our methods were focused on small mammals but were meant to detect any abundant taxa present in the study area. Moreover, we investigated how specific vegetation elements across 3 multiple scales directly influence metrics of prey presence, abundance and diversity. The USFWS recognizes the fisher in the Pacific northwest as a candidate species for federal Endangered Species Act (USDI 2014) and is a candidate species under the California Endangered Species Act (CESA; California Fish and Game Commission Nov 2012). Compared to the fisher's historic range mapped by Grinnell et al. (1937), the range in the state has decreased substantially (Kucera et al. 1995, Zielinski et al. 2005). The fisher's status as a federal and state candidate species and concern over the absence of fishers from a substantial portion of their historic range

in California led U.S. Fish and Wildlife Service (USFWS) and California Department of Fish and Wildlife (CDFW) to consider reintroduction as a way to increase the species range in the state.

Study Area

Sampling for fisher prey species occurred on SPI's 64,800 ha Stirling District which is located in portions of Plumas, Butte and Tehama counties, approximately 20 air miles NE of Chico California (Figure 1). Stirling was managed by previous owners between 1903 and 1992 using single tree selection (personal comm. Ed Murphy, SPI). Beginning in the late 1990's SPI introduced even-aged management (clearcuts). For each clearcut unit two percent of the area is uncut with the goal of retaining structures and diversity in a stand while it regenerates with planted tree seedlings. SPI annually logs an average of 600 ha on the Stirling District with the majority (90%) as clear-cuts since 2008. These figures exclude salvage logging from wildfires (personal comm. Steve Roberts, SPI Stirling District Manager). The climate on Stirling is temperate with the majority (>85%) of precipitation coming in late fall and winter, as snow fall and rain, and little precipitation occurring in other seasons (Pandey et al. 1999). Vegetation on Stirling is typified by the Sierra Nevada mixed conifer forest with 6 dominant trees species: Ponderosa pine (*Pinus ponderosa*), Sugar pine (*Pinus lambertinia*), Incense cedar (*Calocedrus decurrens*), white fir (*Abies concolor*), Douglas fir (*Pseudotsuga menziesii*), and California black oak (*Quercus kelloggii*). Tanoak (*Lithocarpus densiflorus*) and Canyon live oak (*Quercus chrysolepis*) are notable as well, as they may form dense stands at these same elevations (Griffin & Critchfield 1972, Beesley 2007). We chose Stirling to conduct fisher prey research because radio-marked fishers occur throughout the district (Figure 1). Trapping locations occurred in areas with female fishers tend to frequent and specifically in or near female home ranges (Figure 1).

Methods and Analysis

Field Methods

Fisher prey sampling

We attempted to detect fisher prey species and in particular small mammals by live-trapping 30×30 m plots using two types of small live-traps. Additionally, we used infrared camera to observe species that we were unlikely or unable to capture (e.g., black-tailed jackrabbits [*Lepus californicus*]). Trapping was conducted using a stratified-random sampling design to capture wide diversity of forest composition and management based on dominant tree types and tree age class categories. Our aim was to sample locations that had variable amounts of key vegetative components such as canopy closure, basal area of dominate trees, number of downed logs, and percentage of conifer vs hardwood trees. Additionally, plots were clustered within ‘neighborhoods’ (areas of several kilometers) that were similar in environmental conditions and recent management activity. Generally, plots within the same neighborhood were spaced 200 – 4000 m from one another. In some instances plots were placed closer together if there were dramatic differences in the habitat between them.

Sampling plots consisted of five Sherman-style aluminum traps (3 x 3 x 25.4 cm) and five wire-mesh box traps (15.2 x 15.2 x 40.6 cm) baited with peanut butter and oats. At each plot we affixed one wire-mesh box trap (15.2 x 15.2 x 40.6 cm) to trees at chest height (~1.5 m) and baited with apples to target squirrels particularly northern flying squirrels (*Glaucomys sabrinus*). Placement of the cameras and traps in trees was contingent on the availability of a suitable tree. At locations without suitable trees we did not put a trap in trees but placed the trap on the ground. We placed all traps in shaded areas (e.g., beneath trees or shrubs) to minimize heat stress. If shade were not available from trees or other sources we placed pieces of bark or vegetation to shade the trap. We placed cotton batting

in traps for nesting material. All traps were checked a minimum of one time per day and traps located on unshaded or warm sites were checked 2 times (morning and evening). We placed a single camera (Reconyx PC800, Reconyx Inc, Holmen, WI) near the center of each plot and from 1 to 7 meters away but facing the center traps.

We processed all captured animals within 30 m of their capture location. Animals that could be safely handled were processed immediately and with as little distress as possible. Animals that posed significant health risks (e.g., all mesocarnivores or venomous snakes) were released without handling. We restrained animals in a small handling bag and recorded the species, sex, estimated age, breeding status, relative numbers of ectoparasites, and weight. Individuals that weighted less than 50 g (e.g., *Sorex* spp, *Peromyscus* spp) were temporarily marked with commercial hair-dye or colored markers. Animals weighing more than 50 g and that could be safely mechanically restrained were permanently marked with a unique passive integrated transponder (PIT) tag (Biomark, Inc., Boise, ID). PIT tags are advantageous over ear tags because individuals can be identified without being handled (once they are marked), accuracy of identification is improved, and identification time is shortened (Schooley et al. 1993, Morley 2002). Additionally, Schooley et al. (1993) found no known mortalities cause by tagging and little occurrence of infection at the implant site.

We could fail to detect animals at a plot because animals avoided specific plots even if they existed in close proximity. This would indicate avoidance of a plot and the conditions of management and environment that typified that plot. Conversely, animals might not be found at a specific plot because that type of animal was absent from the entire area (a neighborhood). In those situations avoidance of plots would not reflect avoidance of a plot. Thus, while we went about our daily sampling on, and between, specific plots in their respective neighborhoods we noted any species that we saw, heard or identified from tracks or other sign (incidental

detections) to document animals that could occur within a neighborhood. Previous research with remote-cameras has demonstrated differences in detection rate based on body size (Marcus Rowcliffe et al. 2011). Additionally, we knew from our own previous trapping experiences that animals like shrews (*Sorex* spp) and moles (*Scapanus* spp) would be less likely detected in traps or on camera. Hence, we attempted to identify prominent or regular species in the areas we studied. Nevertheless, we recognize that groups or types of animals were likely undetected.

Timeframes for Sampling

We deployed traps and cameras at each plot site for five days (four nights) to determine occupancy and document capture rates and minimum population sizes on the plots (MacKenzie et al. 2002). After we sampled a location for 5 full days all traps and cameras were moved to a new area and no plot was re-sampled in any single year. The plots we sampled during any given 5 days period were clustered in close proximity spatially and temporally and we describe these areas as ‘neighborhoods’. We sampled 7 neighborhoods in 2012 and 10 in both 2014 and 2015. Within each of these neighborhoods 6-12 sampling plots were deployed (Figure 1). In each year sampling commenced in late May and continued through mid-August (approximately 15 weeks). We conducted our study during the summer months because it coincides with when female fishers are actively lactating and have the highest energetic needs (Powell 1979, 1993). Additionally, sampling during the summer reduced the likelihood that individual animals would go undetected because of hibernation or other inactivity. We sampled individual plots and neighborhoods at roughly the same time each year to reduce potential effects of sampling differences among years.

Vegetation sampling

We estimated vegetative and abiotic factors on plots that we predicted *a priori* could influence fisher prey presence, abundance, and overall diversity (Table Veg_defs). On individual trapping plots, referred to as the micro-site scale [MS], we estimated vegetation from the center of the plot using 4 x 15 m (width × length; area = 60 m²) transects in each cardinal direction. Thus, we sampled 27% of the total plot area within our total transect area. At the terminus of each transect we estimated overhead canopy cover with a spherical densiometer. Within each transect we identified all species of trees and shrubs and measured their diameter at breast height (DBH). Trees with a DBH of < 2.54 cm (1 inch) were not measured but were counted. Additionally, all downed and dead trees (logs) and dead and standing trees (snags) ≥ 2.54 cm were counted within each transect. At the center location we estimated the aspect and slope of the dominant topographic feature (e.g., a hillside) with a hand-held compass. Additionally, we estimated the distance to the nearest maintained road and noted if the plot were within 50 m of a permanent or perennial stream or water source. At the center of the plot we used a spherical densitometer to estimate the overhead canopy cover. From these raw data collected in the field data we estimated the mean canopy cover, quadratic mean diameter at breast height, mean basal area, number of logs and snags, percentage of stand in conifer and hardwood trees, and a shrub diversity index using the Simpson diversity index.

Analysis

Vegetation

We estimated vegetation metrics for the stand of trees where a plot occurred using a Geographic Information System and the inventory data of Sierra Pacific Industries (see Table Veg_defs). We mapped all plot locations with ArcGIS 10.3 and corrected the

coordinates of plots that were erroneous. We extracted forest inventory data for the appropriate plot and year of sampling to a database for later analysis. To quantify the vegetative characteristics of each neighborhood we created minimum convex polygons around all plots within a neighborhood using the “minimum bounding geometry” function in ArcGIS 10.3 (Figure 1). To this minimum convex polygon we created a 200 m buffer to ensure we captured all vegetation that contributed to animal abundance and diversity in the neighborhood. We then calculated the mean value for canopy cover, quadratic mean diameter at breast height, basal area, logs per acre, snags per acre and percent conifer and hardwood species. Additionally, we summed the total basal area for a neighborhood and calculated a mean azimuth and elevation. In each neighborhood we calculated the total length of all permanent or maintained paved and unpaved roads and divided the total road length by the total area to estimate road density.

To assess the direction and magnitude of any potential correlations among vegetation metrics and their respective scales of measurement we used ordinary least squares regression. We used the “statsmodels” package available for Python (Python Software Foundation, available at <https://www.continuum.io/downloads>). From the statsmodels package we used linear regression and ran regressions on all pairwise continuous vegetation metrics.

Model Construction and Comparison

In our analyses we evaluated several different metrics to understand how fisher prey may respond to management actions, specifically logging, and to forest composition and structure generally (see Table Veg_Defs). For these different metrics we used a similar approach in that we first compared and tested all of the single vegetation metrics that we thought might influence prey

presence, abundance and diversity. We knew *a priori* that fisher prey were likely to respond to many different variables simultaneously and often in complex ways. Yet, developing models that incorporated multiple variables ($n > 5$) in different permutations would quickly become an enormous analysis that would be tantamount to data mining and difficult to understand and present. Thus, we developed additional models that incorporated numerous variables, or the same variable measured at a different scale, only if they explained a reasonable amount of variation by themselves and aligned with our hypotheses and predictions. This approach limited the total number of models we tested but still incorporated our *a priori* logic that we deemed vital to the analyses. We tested all competing hypotheses that explained our metrics of presence, abundance, and diversity with an Information Theoretic approach (e.g., Burnham and Anderson 2002). We ranked our hypotheses using Akaike's Information Criteria corrected for small sample size (AICc; Burnham & Anderson 2002). We calculated the ΔAICc and from this the AICc weight (w) and AIC likelihood. We used the top model to provide parameter estimates for the beta-estimates in those analyses when there was clear support for a model or variable. In cases where there was no clear support ($\Delta\text{AICc} \leq 4$) we used model averaging of the beta parameters. To understand which variables were most influential in any analysis we summed (w) for all models containing variables of interest. Those with the highest summed weight were deemed most influential.

Occupancy

We estimated naïve estimates of occupancy for the individual plots as well as the neighborhoods. Naïve estimates of occupancy were simply the sum of plots where at least one individual was detected divided by the total number of sampled plots. We repeated this for all genera detected. We also estimated occupancy using the Occupancy estimation procedure within program MARK

(White & Burnham 1999). We defined occupancy of a plot as the presence of at least one individual of a species on at least one day of sampling. We used both trapping and camera data to estimate occupancy. When an individual of a species was detected on a plot within a neighborhood we considered that animal to occupy the neighborhood as well. As with all estimates of abundance or occupancy there are biases associated with how detectable (e.g., trappable) an animal is at any one time. Most species have detection rates below 1 which may vary in relationship to time or some attribute of the environment (MacKenzie et al. 2002, MacKenzie 2006). Hence, we structured our occupancy analyses to consider each day, of our 5 day sampling period, as a repeated sample within one survey period. We considered the same plot that we resampled from one year to the next as independent samples because animal population change through time and occupancy at one time does not ensure the site will remain occupied in the future. To address the dynamic responses different species have to forest composition, structure, and management we created separate analyses for each species where we had sufficient data. We were most interested in how rates of occupancy changed relative to vegetation and abiotic metrics; nevertheless, we also considered the possibility that species, or local communities, were subject to idiosyncratic events that would confound other relationships (Kelt et al. 2013). Thus, we grouped all plots by neighborhoods and tested if those groupings better explained our data than did other environmental or vegetation measures. For occupancy analysis we first attempted to find a suitable model that explained the probability of detection (p) before we modeled occupancy (Ψ). Only after we evaluated and settled on a model for p did we proceed to occupancy. As with our other analyses we used AICc rankings to infer the model best able to explain our data and used estimates from these top models for each species.

Vegetation Threshold Index

Specific models from our occupancy analyses incorporated multiple vegetative metrics to understand their influence on occupancy. Nevertheless, potential interactions between these variables coupled with insufficient data to estimate precisely multiple variables in one model were potentially problematic. We hypothesized that locations with the lowest potential value to important fisher prey species (e.g., tree squirrels) would exist at the early stages after logging. Such locations would lack attributes that existed above specific thresholds across multiple attributes and locations. Hence, we created a composite variable we called the vegetation threshold index to evaluate how species responded to locations with fewer attributes above specific thresholds. We calculated the vegetation threshold index by assigning a plot a value of 0 if it fell below our threshold for a specific attribute and a value of 1 if it was above this threshold (see Table 4). We then summed the values for all 6 metrics to give a score to that plot location. Plots with a value of 0 had no attribute above our threshold whereas plots with a score of 6 were above the threshold for each attribute. We calculated a threshold for canopy cover (> 10% cover), QMD (> 12.7 cm), mean basal area (>15.2 m²), mean logs per acre (>1), mean snags per acre (>2), and percent of trees in hardwoods (>20%).

To evaluate the response of different species, or groups of species, to the vegetation threshold index we created a resource selection function based on the Ivlev electivity index (equation 3; Manly et al. 2007).

Equation 3:

$$Ivlev\ electivity\ index = \frac{proportion\ used - proportion\ available}{proportion\ used + proportion\ available}$$

where the proportion used was defined as the number of detections at each value of the vegetation threshold index divided by the total detection for that species. The proportion available was defined as the total number of plots at each value of the vegetation threshold index divided by the total number of sampled plots. The output of the resource selection function demonstrated if animals used locations, classified by the vegetation threshold index, more or less than expected as a function of their availability.

Abundance

We wished to understand how the abundance of potential fisher prey changed in abundance among sites and based on specific vegetative metrics. In particular, we had hypothesized that clearcutting would reduce the overall abundance of mammals that fishers would use as prey. Further, we anticipated that the abundance of prey animals would slowly increase through time and that plots with moderate to high amount of tree coverage and average size would have correspondingly high numbers of prey animals. We indexed abundance of animals with 2 metrics: catch per unit effort (CPUE) and minimum number alive (MNA). Catch per unit effort is a commonly used metric to estimate abundances of wildlife species (Krebs et al. 1994, Lancia et al. 1996). Generally, in CPUE all captures and recaptures of organisms are summed and divided by the total capture effort. In effect, this method makes counts of captures and recapture comparable when there is uneven trap effort. Nevertheless, CPUE is a compound metric that incorporates both the abundance of organisms as well as their willingness, or ease, of capture (Tracey et al. 2005) and can therefore give biased or

misleading results if there are relatively few animals that are recaptured often. Hence, we also utilized the MNA metric to index abundance.

The MNA accounts for only the minimum number of individuals that can be identified as alive at a location during a specific time. Many studies have used MNA and though the metric is known to be conservative, it is generally highly correlated with more rigorous estimates of population size (Efford 1992, Fryxell et al. 1998, Facka et al. 2008). Because MNA does not use recaptures it is a fundamentally a different metric than CPUE and only reflects the number of animals alive and hence indexes primarily abundance. Both CPUE and MNA are potentially biased by differences in detectability among site and year (time periods) and some have questioned their utility (Anderson 2001, 2003).

Catch per unit effort (CPUE)

We calculated CPUE from our data by summing the total number of captures and recaptures for a location, by year, and dividing the sum by the total trap nights for that location. We used only numbers of captures and recaptures from trapping events and excluded all detection of animals from cameras. Because deer mice (*Peromyscus* spp) were the dominant species we captured we calculated one CPUE from captures of all individuals including deer mice and another where deer mice were we excluded deer mice from the calculation. We calculated both versions of CPUE for each plot and in each year. Estimates of CPUE were used as the dependent variable in a generalized linear model that assumed an underlying binomial distribution. We used the GLM module in the

“statsmodels” package available in Python. We first evaluated the relationship between each of *a priori* variables and CPUE as single factor models. Subsequently, we evaluated models with more explanatory variables.

Minimum number alive (MNA)

We calculated the MNA metric for all species we trapped by summing the total number of uniquely identified animals for a specific plot during the appropriate year. Recaptures were not considered in this calculation. We did not incorporate data from cameras because we could not guarantee that animals seen on the camera were not already captured or that one animal would appear at multiple plots. Additionally, because we could not identify an individual we could not differentiate between the same animal visiting multiple times or multiple individuals visiting a few times. Nevertheless, including camera data would have increased the minimum number alive on some plots and in particular for species that were not often captured including tree squirrels (*Sciurus* and *Tamiasciurus*). We used generalized linear models (GLM) where the dependent variable (minimum number alive) was modeled on the Poisson distribution in program R version 3.2 (R Core Team, Vienna Austria, <http://www.R-project.org/>).

Diversity

We estimated the diversity of fisher prey at any site you metrics of diversity from our capture data and excluded data cameras. Because we were interested in both the total species (richness) present on a plot as well as their relative abundances we use the Simpson’s diversity index (Simpson 1949; Equation 1, Buckland et al. 2005).

Equation 1.

$$D = 1 - \frac{1}{N(N-1)} \sum_{i=1}^s (n_i(n_i - 1))$$

Where n is the number of individuals belonging to the i th species and N is the total number of individuals in the dataset (at the individual plot). The Simpson diversity index is most easily interpreted as the probability that any 2 individuals sampled (in our case captured) on a plot belong to the same species. Hence, high values of D indicate the presence of relatively few species when considering the total abundance of organisms. Therefore, we also calculated the reciprocal of Simpsons' index (Equation 2) which indicates the true diversity of a location with high values representing high diversity relative to the total abundance of organisms.

Equation 2.

$$\frac{1}{D} = \frac{1}{1 - \frac{1}{N(N-1)} \sum_{i=1}^s (n_i(n_i - 1))}$$

Values for both Simpson's index and the reciprocal of Simpson's index were non-normal and did not conform to other statistical distributions available. Thus, we used a square-root transformation of the data to approximate normality (Zar 1999). We then fit the square-root transformed variable using a GLM model in program R.

Results and Discussion

Vegetation

Across all three years we sampled 250 total plots with 30 plots having a common location in all three years. Ninety-eight plots had the same location in both 2014 and 2015. Sampling plots were distributed across much of Stirling (Figure 1) and spanned a wide diversity of elevations, environmental, and management conditions. We collected vegetation at the microsite (plot) level for all but 9 plots that we sampled for fisher prey in 2012. Across our plots we were successful in sampling plots that roughly represented the distribution and range of vegetation conditions across the landscape. Plots were placed in locations that created as wide diversity and range of values for specific vegetative metrics as well as interactions among variables and scales.

The mean values at the level of the microsite were similar as those for both the stand and neighborhood for metrics that were comparable (Table 2). Several vegetative metrics showed high levels of correlation (Table 3). In particular, metrics that related tree volume (basal area) or canopy showed positive relationships to one another and other metrics that were indicative of tree number and tree size (Table 3). Additionally, and expectedly, measure of the percent of conifer trees to hardwood trees were negatively correlated (Table 3). Nevertheless, relatively few vegetation metrics were correlated even within the same scale of measurement (Table 4). Stand-level measurements were most often positively correlated with other stand-level measurements (33%). Measurements at the neighborhood scale were correlated with one another 28% of the time whereas measures within the microsite were only correlated for 7% of variables. Estimates of vegetation metrics occurring across scales were correlated less often than were those occurring within the scales (Table 4). Measurements at the two broadest scales were more often related to one another than were either were to those taken that plot level.

Occupancy

We detected 44 genera of mammals, birds and herpetofauna (snakes, lizards, and amphibians) using all methods (trapping, cameras and incidental occurrences) across all sampled neighborhoods and years (Table 5). We documented 23 genera of mammals, 15 genera of birds, and 5 genera of herpetofauna (Table 5). Deer mice (*Peromyscus maniculatus* and *boylii*) were the most commonly encountered genera of mammal and were detected at 100% of the neighborhoods we sampled indicating they are widespread throughout Stirling and areas adjacent. Mule deer (*Odocoileus hemionus*), woodrats (*Neotoma* spp.), and black bears (*Ursus americanus*) were also detected at > 85% of all neighborhoods we sampled. Other types of mammals including tree squirrels (genera *Sciurus*, *Tamiascirus*, and *Glaucomys*) were detected at $\geq 50\%$ of neighborhoods (Table 5). Gray squirrels are noteworthy at this level because in 2012 and 2015 they were detected at 70% and 90% of plots respectively. In 2014 they were detected at only 40% of neighborhoods. Yet, squirrels were almost undoubtedly persistent at similar levels in all years. Different types of herpetofauna, and specifically lizards, were widely distributed across neighborhoods we sampled (> 50%) but they were not often captured or detected by cameras. Rather, most lizards and snakes were observed at the plots and neighborhoods when we checked traps. Birds were also prevalent within most neighborhoods and we detected jays (*Cyanocitta* spp), woodpeckers (*Colaptes* spp) and creepers (*Certhia* spp) at > 35% of neighborhoods (Table 5). Birds were most often detected through cameras or by seeing or hearing them on plots and we know that we failed to identify many species of forest birds that were common. Nevertheless, a wide diversity of animals species were detected across Stirling and several genera were detected in nearly all areas we conducted research.

We detected a similar assortment of animals on our sampling plots as we did within neighborhoods (Table 5 and 6). Across all plots and years we detected deer mice at 81% (205) of all plots (Table 6). Woodrats were also commonly detected at plots (113; 45%) whereas other species of potential fisher prey (small mammals and birds) were detected $\leq 20\%$ of plots (Table 6). In addition to deer mice and woodrats only chickarees (50), ground squirrels (28), and gray squirrels (19) were detected at sufficient plots to warrant individual occupancy analysis (Table 6). Deer (56), black bears (57) and gray fox (40) occurred and $< 25\%$ of plots, but were detected often enough to estimate occupancy. Nevertheless, we did not conduct individual occupancy analyses on these three genera because they were not our primary research goal.

Other species of meso-carnivores including fisher, ringtail (*Bassariscus astutus*), bobcat (*Lynx rufus*), and skunks (*Mephitis mephitis* and *Spilogale gracilis*) were detected infrequently through all years of the study (2%; Table 6). Additionally, mesocarnivores were detected by remote-camera in all but one instance when we captured a young ringtail which we promptly released from the trap uninjured.

Small-bodied species (deer mice and woodrats) were detected via capture more often than by camera (Figure Trap_Bias). Indeed, captures accounted for the majority of detections for both deer mice and woodrats and had we used cameras exclusively we would have estimated a much lower rates of naïve occupancy than if we had not used both methods. Cameras only detected large-bodied species (e.g., deer and bear) because the traps were far too small to capture them. Animals including the tree squirrels were also rarely captured and were most often detected using cameras. We collected photographs of gray squirrels and chickarees on or

near our traps that refused to enter. California ground squirrels (*Otospermophilus* [*Spermophilus*] *beechyi*) were often captured in contrast to other squirrel types (Figure Trap_Bias).

Peromyscus

The probability of detecting deer mice on plots was best described using a model that incorporated differences among neighborhoods (Table Occupancy). Models that did not contain a neighborhood effect were very poorly supported ($\Delta AICc > 10$) compared to those that contained this effect. Further, the probability of detection was also influenced by the year when sampling occurred. We attempted to model neighborhood effects with similar changes among year, predicting a similar positive or negative trend, similar among all neighborhoods. Yet, a model for detection that incorporated a neighborhood \times year interaction was most supported (Table Occupancy). This model indicates that probability of detecting deer mice was dynamic in both time and space across neighborhoods and year but not always in the same way. Indeed, some neighborhoods showed marked increases in detection from 2012 to 2015 while others showed the opposite pattern. On average the probability of detection was relatively high (0.50; range = 0.16 – 0.87), but highly variable depending on neighborhood. Models that incorporated other temporal (week or month) or vegetative measures performed poorly compared to a model with neighborhood by year effects. Generally, the high rates of detection for deer mice indicated we could estimate rates of occupancy reasonably well.

Several models explained deer mice occupancy rates well (Table Occupancy). Models where occupancy was explained by the variable QMD across the neighborhood were in 6 of the top ten models. Other variable that showed reasonable support in these top models were the number of snags at the microsite, percent of trees as hardwoods, and our index of shrubs (Table Occupancy). Across

all models the QMD variable was most well supported as the single variable influencing deer mice occupancy (summed AICc = 0.68; Table Occ_effects). Quadratic mead diameter had a consistent negative influence on rates of occupancy for deer mice ($\beta = -0.643$; 95%CI = -1.11, -0.174). This relationships indicates that deer mice occupied most plots (>90%) at QMD values below 30 cm (12 inches), but the rate of occupancy declined to <60% onlywhen QMD values were above 36 cm (14 inches). Other variables that influenced the rates of occupancy had estimates with 95% confidence that bounded 0 indicating relatively weak relationships or perhaps insufficient statistical power (plots). Nevertheless, both percentage of a plot in hardwood trees ($\beta = 4.1$, 95%CI = -1.34, 9.56) and the number of snags ($\beta = 1.0$, 95%CI = -0.43, 2.47) had positive relationships to occupancy of deer mice. Explanatory variables for time (sample week, month, and year) and general topography including slope and elevation were poorly supported compared to QMD (Tables Occupancy & Occ_effects).

Neotoma

The probability of detecting woodrats across plots was apparently not contingent upon what neighborhood we sampled because a constant (null) model explained detection rates better than those with a neighborhood effect (Table Occupancy). Across all plots and years the average rate of detection was 0.48 (95%CI = 0.43, 0.52). Models that contained a neighborhood effect were very poorly supported ($\Delta\text{AICc} > 10$) compared to those with a simple null effect. Models that incorporated temporal effects or those of topography (elevation, aspect, or slope) were also relatively poorly supported compared to those with vegetation metrics. Rates of detection for woodrats were best explained by models using the total basal area of a neighborhood, the number of logs and hardwoods at the plot. In fact, we could not easily distinguish a best model when we added or removed number of hardwoods from total basal area and number

of logs (Table Occupancy). Across all models which used those variables total basal area had a negative influence on detection rates ($\beta = -0.71E-6$, 95%CI = -0.13E5, -0.12E6). Similarly, the number of logs on a plot negatively influenced rates of detection for woodrats ($\beta = -0.065$, 95%CI = -0.11, -0.023). Thus, on average woodrats were most likely to be detected in locations with relatively low total basal area and fewer downed logs of all sizes.

Several models explaining woodrat occupancy rates were well supported (Table Occupancy). Models where occupancy was explained by QMD across either the stand or the neighborhood were in all but 3 of the top 15 models (Table Occupancy). In fact, the best selected model included both QMD variables from the stand and neighborhood level of sampling. As single models either QMD variable were effectively tied with other models incorporating more variables, but the value of QMD within a stand was slightly better at explaining occupancy than was QMD at the neighborhood. Other variables that showed reasonable support in these top models were the snags per acre within the stand, the distance to the nearest road, logs per acre, and mean elevation (Table Occupancy). Both QMD variables had high support as variables influencing woodrat occupancy and had the highest summed AICc weights of all variables (Table Occ_effects). Quadratic mean diameter had a consistent negative influence on rates of occupancy for woodrat ($\beta = -0.057$; 95%CI = -0.11, -0.001; Figure WoodratOcc). This relationship indicates that woodrat occupied 60% of plots when QMD was approaching 0 but decreased to 35% at QMD values grew large. Other variables that influenced the rates of occupancy had estimates with 95% confidence that bounded 0 indicating relatively a weak relationship or perhaps insufficient statistical power (plots). Woodrats, were found at moderate levels throughout the study area but appear most likely to occur at low to moderate levels of tree

size and volume. Other variables such as downed wood debris and percent hardwoods may further influence occupancy rate but presently we lack sufficient statistical power to sufficiently develop such models (Tables Occupancy & Occ_effects).

Chickaree (Douglas squirrel)

The probability of detecting chickarees across plots was not strongly different among neighborhoods and a constant (null) model explained detection rates better than those with a neighborhood effect (Table Occupancy). Across all plots and years the average rate of detection was 0.19 (95%CI = 0.13, 0.26). Models that explained detection of chickarees with a neighborhood effect were very poorly supported ($\Delta AICc > 8$) compared to those with a simple null effects. Models that incorporated temporal effects or those of topography (elevation, aspect, or slope) were also relatively poorly supported compared to those with vegetation metrics. Models of detection with a term for logs per acre across stands and neighborhoods had overwhelming supported (Table Occupancy). Indeed, logs per acre at either the stand or neighborhood level explained detection better than other models tested. The probability of detecting chickarees increased with the number of logs per acre at the stand ($\beta = 0.31$, 95%CI = 0.37, 0.60) and neighborhood ($\beta = 1.06$, 95%CI = 0.26, 1.86). Most other variable that we tested to understand rates of detection had beta estimates bounding zero. Mean elevation did positively relate to detection rate ($\beta = 0.10$, 95%CI = 0.096, 0.122) indicating that chickarees were more easily detected at higher elevation. Because models of elevation were ranked low compared models with just logs per acre we did not include it in additional models of chickaree detection rate. Nevertheless, elevation may be a useful explanatory variable in future analyses. On average chickarees were most likely to be detected in locations with the highest numbers of logs per acre.

Chickaree occupancy was well explained by incorporating a neighborhood effect. Nevertheless, we realized we had insufficient replication and detections of chickarees to model occupancy with neighborhood \times covariate models. Hence, though there was strong support for a neighborhood effect we removed this constraint and modeled occupancy with no neighborhood effect (Table Occupancy). Our top models explaining chickaree occupancy were those that contained an effect from the mean percentage of a neighborhood in coniferous trees and the quadratic mean diameter at breast height within the sampled stand (Table Occ_effects). A diversity of other variables including logs per acre, canopy cover, snags per acre, and road density appeared in the top 10 models but always in conjunction with either percent coniferous trees or QMD (or both). As both QMD and the percentage of trees that were conifer increased so too did the probability that chickarees occupied a site (Figure Chickaree). Other variables that apparently influenced the rates of occupancy had estimates with 95% confidence that bounded 0 indicating relatively a weak relationship or perhaps insufficient statistical power (plots). Generally, chickarees occupied areas with large mean tree size and a high percentage of coniferous trees (Figure Chickaree).

Gray Squirrel

The probability of detecting gray squirrels across plots was not strongly affected based on the neighborhood where they were sampled (Table Occupancy). Across all plots and years the average rate of detection was 0.10 (95%CI = 0.05, 0.24). Models that explained detection of gray squirrels within a neighborhood effect were very poorly supported ($\Delta AICc > 10$) compared to those with a simple null effect. Models that incorporated temporal effects or those of topography (elevation, aspect, or slope) had less support than that of a simple null model with the constant detection rates across plots. Models of detection with a term for logs per acre across

neighborhoods had better support ($\Delta\text{AICc} = 1.53$) than did a model with constant detection rate. Additionally, logs per acre performed better than other vegetation metrics at explaining gray squirrel detection including the percent of stand in conifer trees, the number of logs at the plot, and the number of snags. Because there was a real possibility that a model with no covariate or an alternative covariate was more appropriate to model detection we evaluated the influence of different models on the estimation of detection rate. The variation in detection probability across all models was 39.15% suggesting relatively little change regardless of the structure of our models. We used logs per acre as the best model structure for additional analyses on occupancy. The probability of detecting gray squirrels increased with the number of logs per acre at the neighborhood ($\beta = 0.31$, 95%CI = 0.37, 0.60; Figure Gray_detection.). Below 1.5 logs per acre there was nearly no probability of detecting gray squirrels but it increased sharply after this value (Figure Gray_detection).

As with chickarees there was initial support for differences in occupancy among the different neighborhoods. Nevertheless, the relatively few detections of gray squirrels coupled with low rates of detection (0.10) caused us to remove all models with additive or interactive effects with neighborhood. The most supported models for gray squirrel occupancy were those that included the QMD at the plot and the mean number of logs per acre within the neighborhood (Table Occupancy). Our top models explaining gray squirrels occupancy were those that contained an effect from the mean percentage of a neighborhood in coniferous trees and the quadratic mean diameter at breast height within the sampled stand (Table Occ_effects). The only other variables with any appreciable influence (summed $\text{AICc} > 0.10$) on occupancy were the basal area of conifer and hardwood trees (these metrics being highly negatively correlated; Table Occ-effects). The estimated relationships between QMD, logs per acre, and percent hardwoods were surprising in

that all three were negative (contrary to our expectations). The estimated occupancy of gray squirrels was nearly 1.0 at all values of QMD below 45 cm (18 inches) and then fell to nearly 0 for all values above. Logs per acre showed a similar pattern where the estimated occupancy for all values below 1.6 logs/acre was 1.0 and value above 2.0 logs/acre was 0.

California Ground squirrel

The probability of detecting ground squirrels across plots was strongly contingent upon what neighborhood we sampled (Table Occupancy). Ground squirrels were never detected in some areas but were commonly detected in other areas (Figure SPBEE_Detec). Across all plots and years the average rate of detection was 0.18 (95%CI = 0.11, 0.27). In neighborhoods that ground squirrels apparently occupied the rates of detection were highly variable but were relatively high in several locations. Our best models for explaining detection rate included a neighborhood effect and an interactive term with the number of snags per acre within the stand where sampling was conducted (Table Occupancy). Models with additive terms between the neighborhood and snags per acre and other metrics were poorly supported indicating that ground squirrel detection rate was dynamic across locations. In general, ground squirrel rates of detection were negatively correlated with the number of snags per acre within each neighborhood (mean β = -2.34), but only 34% of variation in estimates came from the choice of models suggesting robust estimates of occupancy regardless of how we modeled detection. Models that incorporated temporal effects or those of topography (elevation, aspect, or slope) were relatively poorly supported compared to those with vegetation metrics. On average ground squirrels detection rates were primarily driven by differences within the neighborhood that could be attributable to multiple factors including idiosyncratic mechanisms we did not test.

Several models had similar support for explaining ground squirrels occupancy rates but the basal area of conifer trees had far and away the most support (Table Occupancy). Additionally, including variables related to basal area of total trees (mean basal area and QMD) improved the explanatory power of models that already included basal area of conifer species. Basal area of conifer trees at the plot level was the 3rd highest ranked model and had similar levels of support to models with more complex structures (Table Occupancy). Basal area of conifers was clearly the most influential variable explaining ground squirrel occupancy and had the highest summed AICc weights of all variables (Table Occ_effects). Basal area of conifer trees was negatively related to occupancy of ground squirrels ($\beta = -0.010$; 95%CI = -0.02, -0.0002). Indeed, all of the most influential models from this analysis were negatively related to probability of a ground squirrels occupying an areas suggesting that ground squirrels were most likely to occur in locations with few trees. Locations where the basal area of conifers was less than 30 ft² had an estimated rate of occupancy of 70% with decreasing rates of occupancy as basal area increased (Figure SPBEE_OCC). Ground squirrels occupied areas somewhat idiosyncratically being found in some locations but not in others. Nevertheless, they also appear to occupy areas with fewer trees or open areas such as locations that may have been recently clear cut or are open by nature.

Vegetation threshold index

Our analyses of multiple vegetative thresholds was supportive of our results from our occupancy analyses. The highest proportion of our sampling plots (38%) were plots that met 5 of the 6 vegetation thresholds we delineated (Table VIT; Figure VTI2). Plots meeting 4 (19%) and 6 (16%) total thresholds were the second and third most common types we sampled and those meeting < 4 thresholds comprised $\leq 10\%$ of plots each. Most plots met the thresholds for sufficient canopy cover and QMD and no plot with value

of greater than 2 did not meet these thresholds. Most plots that did not meet all thresholds were deficient in logs per acre or snags per acre.

Because the landscape was comprised primarily of plots that met 5 vegetation criteria the majority of animals were also detected in these same plots. The proportions of animal types detected in different plots were variable depending on the species of animals. For example, 29% of California ground squirrels were detected in plots meeting 5, 40% for deer mice detections and roughly 80% of gray and flying squirrels detections occurred in these plots. Deer mice showed little avoidance or selection for plots that met any number of vegetation thresholds. Indeed, Ivlev values for deer mice neared 1 at every vegetation threshold value indicating they were likely to occur in places regardless of the specific vegetation metrics. Woodrats showed selection for plots that met 0 or 1 thresholds (Figure VTI_Wood). Additionally, there was apparent indifference or selection against plots with the highest number of thresholds. Ground squirrels showed a stronger selection for plots meeting few vegetation thresholds and avoidance or indifference of those meeting more (Figure FTI_Ground). Tree squirrels (gray squirrels, chickarees, and flying squirrels) were captured in relatively high proportion in plots meeting at least 5 vegetation thresholds. They showed relatively strong avoidance of plots meeting 0 or 1 vegetation thresholds and selection for those meeting at least 5 (Figure TreeSquirrel). Nevertheless, the pattern of selection or avoidance at most value above 1 was not strong and appeared random in places. Plots that met 0 or 1 vegetation thresholds were characteristically lacking overhead canopy and relatively large trees whereas all other plots had these traits (Table VTI). The results from the analyses of tree squirrels indicates they may select for canopy and average trees size > 12 cm first and other traits that

incorporated into our index were less important. We did not analyze other groups of animals within this analysis because we had insufficient detections of other species to create meaningful results.

Abundance

Catch per unit effort

We captured primarily small mammals weighting less than 1 kg (Table CPUE2). Deer mice accounted for 71% (907) of our total 1274 captures across 3 years. Woodrats accounted for the next highest percentage of animals captured (22%; Table CPUE2). California ground squirrels, chickarees, and shrews, chipmunks and gray squirrels accounted for the majority of our other captures (Table CPUE2). Total captures increased in each successive year concordant with trapping effort in those years (Table CPUE). Nevertheless, catch per unit effort increased in each year indicating that we more effectively captured animals through time. Mean CPUE across all years was 0.13 ± 0.14 . There were marked differences in mean CPUE across neighborhoods indicating strong spatial differences that were not attributable to processes occurring only at the microsite level or with specific management practices or forest structure at those sites (Figure CPUE). Across all years 26 total animals died as the result of capture. Small species including shrews and deer mice died most often ($n = 18$). Generally, these animals appear to have died for unknown reasons but most likely from stress incurred while be trapped. One ground squirrel and one chipmunk died because of apparent heat stress and once these mortalities occurred we added shade and checked traps more often at those locations. In 2015, 4 woodrats were killed on successive days at the same plot. The traps were moved several 20 to 300 m from their original placement and the remains of the woodrats were found inside often nearly completely devoured. We removed these trap from this location because we could not ensure repeated behavior. The

remains of all mortalities from 2012 and 2014 were sent to the Wildlife Investigations Lab for anti-coagulant rodenticide (AR) analysis. Presently, none of those samples have resulted in positive detections for AR (Deana Clifford unpublished data). Mortalities from 2015 are still in our possession at our SPI office in Chico, CA. We will provide these to WIL at their earliest convenience.

The number of captures of deer mice was strongly correlated with the CPUE ($\beta = 0.022$, $R^2 = 0.51$, $p < 0.0001$, $n = 249$; Figure CPUE_{Ex}). Nevertheless, when we removed captures of deer mice from the CPUE the recalculated number of total captures and catch per unit effort values were still strongly positively correlated with CPUE ($\beta = 1.34$, $R^2 = 0.45$, $p < 0.0001$, $df = 249$; Figure CPUE_{Ex}). Though deer mice were clearly the most important species captured the positive relationship between CPUE and non-deer mice captures suggests, on average, that plots has similar capture rates in spite of species. Additionally, to understand how plots may have varied from year to year we regressed CPUE from 2014 with CPUE from 2015. Catch per unit effort between the two years was positively correlated ($\beta = 0.67$, $R^2 = 0.34$, $p < 0.0001$, $df = 95$; Figure CPUE_Reg).

Generalized linear models that attempted to explain CPUE through temporal, topographical or vegetative metrics were largely uninformative. We ran single-variable models using most of the variables we predicted *a priori* as important (see Table Veg_Defs). All tested models explained similar levels of variation and no model had a $\Delta AICc$ score > 2.6 . We incorporated a single randomly generated variable that was ranked similarly to all other tested models ($\Delta AICc = 2.2$). Additionally, the estimate beta-parameters for all tested models had 95% confidence intervals bounding zero suggesting little, if any, explanatory power. Of all tested models the mean logs per acre at both the neighborhood and stand scale were most highly ranked. Nevertheless, the lack of differentiation among

variables and models and their uninformative estimated parameters indicate that no model explained CPUE effort well and thus we did not investigate more parameterized models.

Minimum number alive

We characterized the abundance of 8 different genera of mammals and one genera of herpetofauna using trapping data collected over the course of the three years of study. Using parametric analysis of variance (ANOVA) we detected differences in the mean MNA across both local (plot, $F=2.127$, $p<0.001$) and neighborhood scales (line, $F= 5.731$ $p<0.001$), but not temporally (year, $F=2.417$, $p=0.121$) (week, $F=0.351$, $p=0.554$). The top twenty ranking models included distance to water and distance to road (Table_MNA_1). In addition to these distance parameters, we had strong support that the additive effects of slope and basal area (BA) strongly influenced MNA at the local level based on the sum of Akaike's weight (Table_MNA_2).

The most influential variables effecting the abundance of animals captured alive at the local level were distance to water and distance to road (Tables_MNA_1 and 2). Both of these variables had the highest summed Akaike weights indicating high importance. The inclusion of the distance to road parameter in all of our top models indicates that there is a difference in abundance of animals found on a particular plot as a result of proximity to road (Table_1_MNA). Plots established further away from the road had a greater abundance of capture animals as quantified by MNA (Table_2_MNA). Distance to water was characterized as a discrete, binomial variable in the data with distance from the plot to a water source $<50\text{m}$ away receiving a 1 and $>50\text{m}$ away receiving a 0. In contrast, other variables included in the top twenty models (Table_1_MNA) were recorded as continuous variables. The difference in these methods of measuring these data could explain the inflated AIC Model Average for the distance to water parameter in comparison to

distance to road (Table_2_MNA). Nonetheless proximity to a water source was strongly positively correlated with MNA and was a consistently significant variable in all the top models (Table_2_MNA).

Slope and basal area were also found to be important covariates in explaining the variance in MNA at the local level. Slope had a higher summed AIC weight and greater AIC model average (Table_2_MNA), thus describing our data better, than basal area. Higher abundances of animals were found on plots as slope and average basal area increased. Using the beta estimates for the variables included in the top 20 models we generated 95% confidence intervals to measure the precision of the relationship of the covariates to the MNA at the local level (Table_2_MNA). We found that the confidence intervals for the beta estimates did not bound zero for 4 of the 18 models slope was included in and 2 of the 13 models that included mean basal area. In comparison, we found that the confidence intervals generated for the beta estimates for distance to road and distance to water did not bound zero indicating an informative relationships to MNA.

Diversity

The square-root transformed Simpson's reciprocal index of diversity demonstrated variation of diversity both spatially and temporally. The sampling week was among the most important variables influencing our measure of diversity at the local level (Fig_Div_1). Simpson's reciprocal index of diversity was negatively correlated with week but positively related to year (Table_2_Div). Week had the highest summed Akaike weights and a high average proportion of importance of significance values across models (Fig_2_Div). Looking at 95% confidence intervals generated from the beta estimates for the top 20 models we found that week bounded zero less frequently than any other covariate, including year (Table_2_Div). Year also had a strong effect on model

selection, with a large AIC model average and high summed Akaike's weight (Table_1_Div). This provides good support that Simpson's reciprocal index was strongly influenced by week with an additive effect of year

The vegetative covariate QMD was included in most of our top twenty models, and had the third highest summed Akaike's weight (Table_2_Div). Models that removed QMD had the highest fold change in AICc and the smallest difference in deviance from the top model, receiving nearly no support (Table_1_Div). Percentage of hardwood on plot and canopy cover also had high summed Akaike's weights as well as a higher average proportion of values of significance across the top twenty models. QMD and percentage of hardwood trees across the plot were both strongly correlated with Simpson's reciprocal index of diversity. The very weak negative correlation with Simpson's Reciprocal index and the estimated confidence intervals for the beta estimates bounding zero indicated a weak relationship (Table_1_Div).

Discussion

The interactions between predators and prey to their respective habitats are complex and dynamic processes that require additional and refined investigations. In particular, for animals like fishers, human mediated habitat disturbance is becoming the norm and we must better understand how such disturbances influence prey and then subsequently their predators. Nevertheless, our analyses do reveal important associations with forest structure and management that can be useful in better understanding and conserving species even in the face of resource utilization and resource extraction. We find that specific measures of forest composition and structure influence the distribution, occurrence and abundance of species that fishers require as their critical prey. In particular, the

mean tree diameter (QMD) consistently appears in models of occupancy and diversity. Secondly, other metrics of mature forests including mean basal area and number of logs also appear as influential in multiple analyses. Additionally, proximity to water may be important because of the correlation to water and stands of relatively complex forests. Large mean tree size appears to affect animals differently with deer mice, woodrats and ground squirrels occurring in higher probability at low values whereas tree squirrel species occur at the opposite end of the spectrum. Other elements appear important for some species. The number of downed logs appears important to tree squirrel species detection which may be a proxy or correlate of abundance of those species.

We can conclude that scale is an important consideration for evaluating vegetation and habitat metrics for fisher prey which comports with other findings (Bowman et al. 2001, Coppeto et al. 2006). Vegetation measures at the plot, stand and neighborhood scale all influenced rates of detection and occupancy. Though we did not specifically identify assemblages by habitat of mammals or prey species generally we can suggest that consideration of the forest structure and management at small scales improves our understanding of how animals will respond even when we consider measurements at broad scales. Moreover, these metrics were not often correlated on our study site suggesting measurement at one scale may be insufficient to appropriately understand the response of prey species. Through time the relationships between vegetation types or composition relative to prey occurrence or abundance may break down with increased intensity of management (e.g., Kelt et al. 2013). We have not yet included the overall composition of fragmentation within a neighborhood. Further, we have not quantified how the size, composition or distance to neighboring stands influences occupancy, abundance or diversity for any of the groups we've investigated. We hypothesize such metrics would be

informative and useful to better understand future implications of management on prey and ultimately fisher prevalence on the landscape.

We realize that our study has underrepresented some, and perhaps many, groups of mammals. In particular, shrews and moles were almost never detected by our methods of capture and are unlikely to be detected by cameras (Marcus Rowcliffe et al. 2011). Pitfall trapping would likely yield better rates of detection for both groups and should be considered in future studies. Nevertheless, shrews and moles are not considered highly ranked fisher prey. Other groups including chipmunks (*Tamias* spp) were not detected often and maybe a function of our trapping methods or biases in elevation. Our primary focus was on mammals but fishers also eat many types of birds. Our study was largely insufficient to detect and understand occupancy of bird species; nevertheless, some of the patterns we observed for mammals are consistent with studies on birds (Furnas & Callas 2014, Furnas & Barrett 2015).

Our use of CPUE and MNA to index abundance may be insufficient to detect important relationships between abundance and forest structure and practices. In particular, CPUE was uninformative perhaps because we combined all genera into one metric rather than splitting them into more detailed groups. Future analyses will look more closely at the abundance of specific genera or species to better evaluate abundance. The MNA metric did appear to give promising results that were in line with our *a priori* hypotheses. Abundance correlating to distance to water makes sense because areas close to water usually remain uncut or lightly managed. Thus, tree densities and size are often large in these areas as are other elements including downed logs and hardwoods species. Plots closer to water are thus more likely to have dense stand of large trees with dead woody elements as well as having different microclimates (cooler and more moist during summer sampling). As with CPUE we will further investigate abundance with refined metrics and

evaluate groups of animals individually rather than collectively. Presently our analyses of occupancy provide our best information about individual groups of animals and their response to forest management and composition.

Deer mice appear to be in all habitats and locations throughout Stirling. Future management activities are unlikely to extirpate deer mice at any significant scale or time. Overall deer mice appear indifferent to different forest measures except, perhaps, when forest are at their most mature with large QMD and downed logs and standing snags. These findings are consistent through both the occupancy and vegetation threshold analyses suggesting deer mice showed high occupancy in most habitat with only modest declines in occupancy at the highest values of average tree diameter (size).

The occurrence of woodrats in nearly all neighborhoods suggests that plots where we did not find them are most likely a result of some level of avoidance. Woodrats were cosmopolitan on Stirling and occurred in most locations in spite of types, abundance or size of trees or forest composition generally. Nevertheless, woodrats appear to do well in locations with few or small trees and locations with small QMD have higher rates of occupancy. These results do not preclude other variables from being important for woodrat presence and indeed several other variables improved explanatory power of models including QMD. Our results do indicate the woodrats are likely to persist and recover for disturbance related to clearcuts relatively well.

The results from the chickaree analysis are informative and interesting in identifying QMD as an important driver of occupancy. Additionally, detection rates that are driven by logs per acre at two scales indicates may suggest that chickarees are more abundant in areas with large numbers of logs since detection rates may increase with abundance (Royle & Nichols 2003). More work is needed to fully understand if chickarees are most abundant in areas with logs. Indeed areas with high log densities may simply be

correlated with other processes important to chickarees. Nevertheless, the relationship between detection rate and abundance is important for all species investigated in this study. Chickarees are also notable because they are apparently most likely to occupy sites with high percentages of conifer trees and QMD. Regenerating forests, or plantations, are often characterized by a high proportion of conifers and given enough time large trees and other red squirrel species appear tolerant of heavy timber management (Cote et al. 2001). Thus, chickarees may do better than some other species of tree squirrels in those conditions. We recommend that forest managers focus on maintaining other attributes in addition to average large trees including a diversity of sized of downed (Herbers et al. 2007).

Our analysis of gray squirrels is perhaps the least informative of all analyses because of the low detection rates. Negative relationships between QMD and log numbers are unexpected because gray squirrels are associated with medium to large trees in the CWHR model (Airola 1988). We caution that these results are likely a product of poor statistical power to estimate those relationships. The selection of QMD and logs is nonetheless interesting because it comports to the findings from other species in this analysis. As with chickarees the high detection rate at high levels of logs is interesting. In contrast to occupancy, the rate of detection, is a more robust estimate. That both chickarees and gray squirrel detection rates are high with high numbers of logs seems to be an unusual coincidence. We hypothesize that areas with many downed trees and logs are actually very good habitat for trees squirrels of all types and that their densities are highest in these areas. Squirrels may be occasionally present in areas with low QMD and logs but may simply exist at low densities and thus explain why we do detect them in low QMD locations but that they are so infrequently detected

and re-detected there. Of course, other factors influence detection rate and we cannot simply assume that abundance is the only variable of importance (Krebs et al. 1994).

The high bias of detecting tree squirrels through remote cameras is an important point that provides for further opportunity. Remote cameras are becoming ubiquitous because of their relative ease of use and their ability to function without intensive monitoring (Pimm et al. 2015). Indeed, remote cameras are used by nearly all fisher projects and are often used throughout the year and are spatially well distributed. Because trapping seem relatively ineffective at detecting squirrels further work using the rich resource of remote camera data can better quantify the relationship between squirrels and specific forest elements. Further, we recommend that future studies focus on precisely estimating squirrel abundance by habitat type.

California ground squirrels had a highly variable spatial distribution across Stirling. They were common in some locations but not found in others. We had anticipated that variables such as slope would predict areas of ground squirrel occupancy because they were rocky and steep and afforded burrowing opportunities (Leger et al. 1983, Fehmi et al. 2005). Nevertheless, these variables were unsupported as good explanatory variables. We hypothesize that soil types or corridors to other groups of ground squirrels may be important to explaining broad-scale occupancy. California ground squirrels appear less likely to occupy areas with high basal area or large trees. Other studies note that California ground squirrels are more frequent in mixed forest and pine forests similar to those on our study site (Coppeto et al. 2006). Additionally, the California Wildlife Habitat Relations models indicate preferred habitat in stands characterized by small trees (Airola 1988). Generally, we predict that ground squirrels will respond favorably to logging and disturbance.

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TABLES.

Table Veg_Defs. Description of explanatory variables used to model the abundance, occupancy, and catch-per-unit-effort for potential fisher prey species on Stirling district in northern California.

Scale	Variable	Description
Temporal	Week	The week that sampling was conducted
	Month	The month that sampling was conducted
	Year	The year that sampling was conducted
Microsite	Slope	Percentage grade (steepness) at the sampled plot
Plot (30x30m)	Canopy[MS]	Percentage of the forest floor covered by overstory trees
	Distance to Road	Distance (m) to nearest road
	BA[MS]	Total cross-sectional area (ft ²) of tree trunks of all trees (DBH>12 cm) on plot
	Num_logs[MS]	Total number of downed and dead logs (DBH>1 cm) on plot
	Mean_logs[MS]	Mean numbers of downed and dead logs across 4 sample transects
	Num_snags[MS]	Total number of standing dead trees (DBH>12 cm) on plot
	Mean_snags[MS]	Mean numbers of standing dead trees (DBH>12 cm) across 4 sample transects
	Num_con[MS]	Total number of conifer trees (DBH > 12 cm) on plot
	Num_hard[MS]	Total number of hardwoods trees (DBH > 12 cm) on plot

	BACon	Mean Basal area of all conifers trees (DBH > 12 cm) on plot
	BA Hard	Mean Basal area of all hardwoods trees (DBH > 12 cm) on plot
	pCon[MS]	Percent Basal area of all conifer trees on plot
	pHard[MS]	Percent Basal area of all hardwood trees on plot
	Shrubs Index	Index of shrub diversity on plot calculated using Simpsons' Diversity Index
	Shrubs Index R	Reciprocal (1/Shrubs Index) of diversity on plot
	QMD[MS]	Quadratic Mean Diameter at breast height of all trees > 12 cm on plot
Stand	BA[St]	Mean basal area of all trees (DBH>12 cm) within a stand
	QMD [St]	Quadratic Mean Diameter at breast height of all trees > 12 cm within stand
	Canopy[St]	Mean percentage of the forest floor covered by overstory trees in stand
	LPA[St]	Mean number of logs per acre (DBH > 30 cm) within stand where plot occurred
	pCon[St]	Percent Basal area of all conifer trees on plot
	pHard[St]	Percent Basal area of all hardwood trees on plot
	SPA[St]	Mean number of snags per acre (DBH > 30 cm) occurring within a stand
Neighborhood	TBA[N]	The total basal area of all trees (DBH > 12 cm) occurring within a sampled neighborhood
	MBA[N]	The mean basal area of all trees (DBH > 12 cm) occurring within a sampled neighborhood
	pConr[N]	Percent Basal area of all conifer trees in stand
	pHard[N]	Percent Basal area of all hardwood trees in stand
	QMD[N]	Quadratic Mean Diameter at breast height of all trees > 12 cm in stand
	RoadsLength	total length of all roads within a neighborhood

	RoadDensity	Total length of all roads within a neighborhood divided by total area of neighborhood
	SPA[N]	Mean number of snags per acre (DBH > 30 cm) occurring within neighborhood
Landscape	Easting	Universal Transverse Mercator coordinate for the Easting of a plot
	Northing	Universal Transverse Mercator coordinate for the Northing of a plot

Table 2. The mean \pm 1 standard deviation for each variable at each scale of measurement at fisher prey sampling plots during 2012, 2014, and 2015 on Stirling in northern California.

Variable	Plot	Stand	Neighborhood
Slope	21.55 \pm 12.47
Canopy (%)	76 \pm 31	78.28 \pm 35.98	78.72 \pm 8.11
Distance to Road (m)	37 \pm 20.01	..	^a 0.01 \pm 0
Basal Area (m ²)	141.94 \pm 91.61	166.99 \pm 87.91	166.46 \pm 16.48
Mean Num Logs	4.28 \pm 6.12	1.19 \pm 1.28	1.16 \pm 0.58
Mean snags	3.62 \pm 5.03	3.06 \pm 3.56	2.69 \pm 0.9
Percent Conifer	0.62 \pm 0.34	55.85 \pm 37.26	67.38 \pm 13.16
Percent Hardwoods	0.33 \pm 0.32	24.92 \pm 24	29.26 \pm 13.93
Shrubs Index	1.85 \pm 1.11
Shrubs Index Reciprocal	1.55 \pm 1.54
Quadratic Mean Diameter	11.92 \pm 4.35	11.35 \pm 4.89	11.25 \pm 1.23

^aRepresents total road length in across neighborhoods rather than the distance to roads collected for plots.

Table_3_Summary statistics for 32 linear regression analyses between vegetation metrics showing the variable used as the explanatory variable, the dependent variable (Y_Variable), the estimated y-intercepts and slope parameters, the 95% confidence limits around the estimate of slopes, the coefficient of determination (R^2), and p-value of each regression.

X_Variable	Y_Variable	Y-Intercept	Slope	95low	95high	R^2	p-value
BA[MS]	Num_Hard[MS]	-0.593	0.048	0.039	0.056	0.319	<0.001
BA[MS]	Num_Con[MS]	2.019	0.033	0.024	0.043	0.165	<0.001
BAPA[St]	Canopy[St]	20.932	0.343	0.316	0.371	0.704	<0.001
BAPA[St]	QMD[ST]t	4.232	0.043	0.038	0.047	0.587	<0.001
BAPA[St]	SPA[St]	-0.012	0.018	0.014	0.023	0.207	<0.001
BAPA[St]	LPA[St]	0.216	0.006	0.004	0.007	0.159	<0.001
MBA[N]	Canopy[N]	38.580	0.241	0.188	0.295	0.240	<0.001
MBA[N]	TBA[N]	-1001623.1	9633.4	7329.0	11937.9	0.213	<0.001
TBA[N]	RoadLength	1793.229	0.023	0.022	0.024	0.915	<0.001
TBA[N]	Canopy[N]	71.883	<0.001	<0.001	<0.001	0.232	<0.001
TBA[N]	SPA[N]	3.374	<0.001	<0.001	<0.001	0.188	<0.001
Canopy[MS]	PCon[St]	64.359	-0.668	-0.790	-0.547	0.319	<0.001
Canopy[MS]	Num_logs[MS]	0.853	0.022	0.015	0.028	0.150	<0.001
Canopy[N]	PerConN	158.084	-1.152	-1.294	-1.010	0.504	<0.001
Canopy[N]	PerHardN	-66.153	1.212	1.061	1.364	0.498	<0.001
Canopy[N]	SPA[N]	6.260	-0.045	-0.058	-0.033	0.166	<0.001
Canopy[N]	RoadLength	-16448.9	407.9	291.3	524.5	0.159	<0.001

Canopy[St]	PHard[St]	-2.939	0.356	0.286	0.426	0.285	<0.001
LPA[St]	LPA[N]	0.916	0.210	0.160	0.260	0.214	<0.001
Num_logs[MS]	Mean_Snag[MS]	2.245	1.218	0.901	1.536	0.185	<0.001
Num_Hard[MS]	pHardMS	0.157	0.028	0.024	0.032	0.457	<0.001
Num_Hard[MS]	PCon[MS]	0.781	-0.025	-0.030	-0.021	0.333	<0.001
pCon[MS]	pCon[St]	24.022	51.008	39.038	62.977	0.220	<0.001
pHard[MS]	pCon[MS]	0.919	-0.893	-0.964	-0.821	0.705	<0.001
pHard[MS]	pHard[St]	15.083	29.752	21.248	38.255	0.159	<0.001
PHard[St]	pConN	73.619	-0.250	-0.311	-0.190	0.209	<0.001
PHard[St]	Canopy[N]	75.002	0.149	0.111	0.187	0.195	<0.001
PHard[St]	pHard[N]	23.043	0.249	0.184	0.315	0.185	<0.001
pConN	pHard[N]	96.384	-0.996	-1.041	-0.952	0.886	<0.001
QMDn	pCon[N]	13.197	4.816	3.627	6.005	0.203	<0.001
QMD[ST]t	Canopy[St]	19.512	5.177	4.527	5.827	0.496	<0.001
QMD[ST]t	SPA[St]	-1.070	0.364	0.285	0.442	0.250	<0.001
SPA[N]	RoadLength	30051.498	-5351.5	-6279.21	-4423.8	0.340	<0.001
SPA[N]	RoadDensity	0.008	-0.001	-0.001	-0.001	0.320	<0.001

Table 4. Summary results of regressions conducted between pairwise comparisons of forest metrics at 3 levels of scale (MicroSite, Stand, Neighborhood) with the number of regressions between that resulted in p-values below 0.05 for metrics at specified scales (Num regression), the total number of regressions performed (Total Regression) and the proportion where the p-value was

Scale Comparison	Num Regressions	Total Regressions	Proportion P < 0.05
MicroSite-MicroSite	8	108	0.074
MicroSite-Stand	3	112	0.027
MicroSite- Neighborhood	0	192	<0.001
Stand-Neighborhood	4	84	0.048
Stand-Stand	7	21	0.333
Neighborhood -Neighborhood	15	53	0.283

below 0.05 for plots (n= 250) location on Stirling district in northern California.

able 6. The number of fisher prey sampling plots in each year where at least one individual of each species was detected with the combined number of plots (total) across all 3 years of the study and the overall percentage of sampled plots that were estimated to be occupied by the given species on Stirling in northern California.

Genus	Common	2012	2014	2015	Total	Percent
Peromyscus	Deer Mice	37	82	86	205	0.81
Neotoma	Woodrats	17	48	48	113	0.45
Ursus	Black Bear	4	24	29	57	0.23
Odocoileus	Deer	12	26	18	56	0.22
Tamiasciurus	Chickaree	5	25	20	50	0.20
Urocyon	Gray Fox	7	19	14	40	0.16
Otospermophilus	Ground Squirrel	4	11	13	28	0.11
Sciurus	Gray Squirrel	5	4	10	19	0.08
Sorex	Shrews	0	6	3	9	0.04
Tamias	Chipmunks	2	1	4	7	0.03
Glaucomys	Flying Squirrel	0	2	5	7	0.03
Lepus	Jackrabbit	2	2	2	6	0.02
Microtus	Vole	0	0	1	1	0.00
Bassariscus	Ringtail	4	0	1	5	0.02
Mephitis	Striped Skunk	0	0	2	2	0.01
Procyon	Raccoon	1	1	0	2	0.01
Lynx	Bobcat	0	1	0	1	0.00
Pekania (Martes)	Fisher	0	1	0	1	0.00
Spilogale	Spotted Skunk	1	0	0	1	0.00
Pipilo	Towhee	3	1	0	4	0.02
Turdus	Thrush	0	1	2	3	0.01
Picoides	Woodpecker	0	0	2	2	0.01
Colaptes	Woodpecker	0	0	1	1	0.00
Cyanocitta	Jay	0	1	0	1	0.00

Juncus	Junco	1	0	0	1	0.00
Mimus	Mockingbird	0	0	1	1	0.00
Piranga	Tanager	1	0	0	1	0.00

Table 5. The number of neighborhoods by year where at least one individual of the respective species was identified on or adjacent to sampling plots with the total number of neighborhoods (Neigh) visited across each year and the percentage of those neighborhoods occupied on Stirling district in northern California.

Type	Genera	Common Name	2012	2014	2015	Neigh.	Total	Percent
Mammal	Peromyscus	Deer mouse	7	10	10	26	26	1.00
Mammal	Odocoileus	Mule deer	6	10	9	26	25	0.96
Mammal	Neotoma	Woodrat	5	9	10	26	24	0.92
Mammal	Ursus	Black bear	3	9	10	26	22	0.85
Mammal	Tamiasciurus	Chickaree	4	8	9	26	21	0.81
Mammal	Urocyon	Gray fox	3	6	8	26	17	0.65
Mammal	Sciurus	Gray squirrel	4	4	9	26	13	0.65
Mammal	Otospermophilus	ground squirrel	4	6	5	26	15	0.58
Mammal	Lepus	Jackrabbit	2	2	8	26	12	0.46
Mammal	Sorex	Shrew	3	6	3	26	12	0.46
Mammal	Tamias	Chipmunk	1	1	5	26	7	0.27
Mammal	Glaucomys	Flying squirrel	0	2	3	26	5	0.19
Mammal	Oreortyx	Quail	0	0	5	26	5	0.19
Mammal	Pekania	Fisher	0	1	3	26	4	0.15
Mammal	Canis	Coyote	0	0	2	26	2	0.08
Mammal	Mimus	Mockingbird	0	0	2	26	2	0.08
Mammal	Procyon	Raccon	1	0	1	26	2	0.08

Mammal	Bassariscus	Ringtail	2	0	1	26	1	0.04
Mammal	Lynx	Bobcat	0	1	0	26	1	0.04
Mammal	Mephitis	Striped skunk	0	0	1	26	1	0.04
Mammal	Microtus	Vole	0	0	1	26	1	0.04
Mammal	Scapanus	Mole	1	0	0	26	1	0.04
Mammal	Spilogale	Spotted Skunk	1	0	0	26	1	0.04
Herp	Sceloporus	Fence lizard	^a 0	8	10	26	18	0.69
Herp	Elgaria	Alligator lizard	6	0	8	26	14	0.54
Herp	Thamnophis	Garter snake	0	2	1	26	3	0.12
Herp	Crotalus	Rattlesnake	0	1	1	26	2	0.08
Herp	Pituophis	Gopher snake	0	2	0	26	2	0.08
Avian	Cyanocitta	Jay	0	7	10	26	17	0.65
Avian	Colaptes	Woodpecker	0	5	6	26	11	0.42
Avian	Certhia	Creepers	0	0	9	26	9	0.35
Avian	Corvus	Raven	0	3	6	26	9	0.35
Avian	Accipiter	^b Cooper's Hawk	0	0	6	26	6	0.23
Avian	Junco	Junco	1	0	5	26	6	0.23
Avian	Poecile	Chickadee	0	0	5	26	5	0.19
Avian	Picoides	Woodpecker	0	0	5	26	5	0.19
Avian	Pipilo	Towhee	2	1	0	26	3	0.12

Avian	Meleagris	Turkey	0	2	1	26	2	0.08
Avian	Buteo	Redtail Hawk	0	1	7	26	1	0.04
Avian	Callipepla	Quail	0	1	0	26	1	0.04
Avian	Piranga	Tanager	1	0	0	26	1	0.04
Avian	Strix	Owl	0	0	1	26	1	0.04
Avian	Turdus	Blackbird	0	0	1	26	1	0.04

^a All lizards were mistakenly identified as genus *Elgaria* (alligator lizard) in 2012.

^b Included both Sharp shinned and Cooper's hawk but excludes Goshawks.

Table Occupancy. Comparison of the top ranked models ($\Delta AICc \leq 4$) hypothesized to explain occupancy for 5 genera of mammals sampled on Stirling in northern California in 2012, 2014, and 2015. The -2log likelihood (not shown), the number of estimated parameters (k), number of observations ($n = 250$ for all are used to calculate Akaike's information criterion corrected for small sample size (AICc) and the differences between the top model and other competing models ($\Delta AICc$), the model likelihood, and the model weight (w). Bolded portions of model descriptions emphasize the model structure for Ψ (occupancy).

Genera	^a Model Description	AICc	$\Delta AICc$	w	Likelihood d	k
Peromyscus	p(g × Year) $\Psi(QMD[N]+Num_snags[MS]+pHard[MS])$	1584.0	0.00	0.26	1.00	27
	p(g × Year) $\Psi(QMD[N]+shrubs[MS]+pHard[MS])$	1586.4	2.35	0.08	0.31	27
	p(g × Year) $\Psi(QMD[N]+shrubs[MS])$	1586.6	2.51	0.08	0.29	26
	p(g × Year) $\Psi(QMD[N]+pCon[MS])$	1587.0	3.00	0.06	0.22	26
	p(g × Year) $\Psi(pHard[MS])$	1587.3	3.20	0.05	0.20	25
	p(g × Year) $\Psi(QMD[N]+pHard[MS])$	1587.4	3.33	0.05	0.19	26
	p(g × Year) $\Psi(pCon[MS])$	1587.8	3.73	0.04	0.16	25
	p(g × Year) $\Psi(QMD[N])$	1587.8	3.74	0.04	0.15	25
	p(g × Year) $\Psi(Num_snags[MS])$	1588.4	4.36	0.03	0.11	25
Neotoma	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(QMD[N]+QMD[St])$	1083.8	0.00	0.12	1.00	7
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(QMD[St])$	1084.4	0.63	0.09	0.73	6
	p(TBA[N]+Num_logs[MS]+NumHard[MS]+pCon[S]) $\Psi(QMD[St])$	1084.7	0.90	0.08	0.64	7
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(QMD[N]+SPA[St])$	1085.0	1.25	0.07	0.54	7
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(QMD[St]+Dist2Road)$	1085.5	1.77	0.05	0.41	7
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(QMD[N]+LPA[St])$	1086.0	2.27	0.04	0.32	7

	p(TBA[N]+Num_logs[MS]) $\Psi(\text{QMD}[\text{St}])$	1086.1	2.33	0.04	0.31	5
	p(TBA[N]+Num_logs[MS]) $\Psi(\text{QMD}[\text{N}])$	1086.2	2.45	0.04	0.29	5
	p(TBA[N]+Num_logs[MS]) $\Psi(\text{QMD}[\text{N}]+\text{SPA}[\text{St}])$	1086.4	2.68	0.03	0.26	6
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(\text{QMD}[\text{N}]+\text{SPA}[\text{St}])$	1086.5	2.70	0.03	0.26	7
	p(TBA[N]+Num_logs[MS]+mQMD[St]) $\Psi(\text{QMD}[\text{St}])$	1087.1	3.31	0.02	0.19	6
	p(TBA[N]+Num_logs[MS]) $\Psi(\text{Year} + \text{QMD}[\text{St}])$	1087.1	3.33	0.02	0.19	6
	p(TBA[N]+Num_logs[MS]+pHard[MS]) $\Psi(\text{QMD}[\text{St}])$	1087.1	3.36	0.02	0.19	6
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(\text{Dist2Road})$	1087.4	3.61	0.02	0.16	6
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(\text{Year})$	1087.5	3.68	0.02	0.16	6
	p(TBA+Num_logs) $\Psi(\text{Elevation})$	1087.7	3.93	0.02	0.14	5
	p(TBA[N]+TBA^2+Num_logs[MS]) $\Psi(\text{QMD}[\text{St}])$	1087.8	4.00	0.02	0.14	6
Tamiasciurus	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}]+\text{QMD}[\text{MS}]+\text{LPA}[\text{N}])$	526.4	0.00	0.32	1.00	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}]+\text{QMD}[\text{MS}])$	528.0	1.56	0.15	0.46	6
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}]+\text{QMD}[\text{MS}]+\text{Canopy}[\text{St}])$	528.8	2.34	0.10	0.31	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}]+\text{QMD}[\text{MS}]+\text{SPA}[\text{N}])$	529.3	2.86	0.08	0.24	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}]+\text{BA}[\text{St}]+\text{LPA}[\text{N}])$	529.3	2.89	0.07	0.24	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}]+\text{QMD}[\text{St}]+\text{LPA}[\text{N}])$	529.5	3.11	0.07	0.21	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}]+\text{Canopy}[\text{MS}])$	529.6	3.17	0.07	0.20	6
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}]+\text{QMD}[\text{MS}]+\text{RoadDensity})$	530.0	3.57	0.05	0.17	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon}[\text{N}])$	530.0	3.58	0.05	0.17	5

	p(LPA[N]+LPA[St]) Ψ (pCon[N]+pCon[N] ²)	532.1	5.67	0.02	0.06	6
Sciurus	p(LPA[N]) Ψ (QMD[MS]+LPA[N]+pHard[St])	184.6	0.00	0.34	1.00	6
	p(LPA[N]) Ψ (QMD[MS]+LPA[N]+BACon[MS])	186.1	1.47	0.17	0.48	6
	p(LPA[N]) Ψ (QMD[MS]+LPA[N]+pCon)	186.2	1.54	0.16	0.46	6
	p(LPA[N]) Ψ (QMD[MS]+LPA[N]+BAHard[MS])	186.8	2.16	0.12	0.34	6
	p(LPA[N]) Ψ (QMD[MS]+LPA[N]+BAHard[MS])	186.8	2.16	0.12	0.34	6
	p(LPA[N]) Ψ (QMD[MS]+LPA[N])	188.9	4.26	0.04	0.12	5
Otosperm.	p(g × SPA[St]) Ψ (BACon[MS]+MBA[N])	332.1	0.00	0.25	1.00	21
	p(g × SPA[St]) Ψ (BACon[MS]+Bacon[MS] ²)	332.2	0.08	0.24	0.96	22
	p(g × SPA[St]) Ψ (BACon[MS])	334.2	2.07	0.09	0.36	20
	p(g × SPA[St]) Ψ (BACon[MS]+BA[St])	334.6	2.46	0.07	0.29	21
	p(g × SPA[St]) Ψ (QMD[St])	335.2	3.09	0.05	0.21	20
	p(g × SPA[St]) Ψ (BACon[MS]+QMD[N])	335.9	3.73	0.04	0.15	21
	p(g × SPA[St]) Ψ (BACon[MS]+Shrubs)	336.1	4.02	0.03	0.13	21

^a All models with “g” in either portion of the model have a group neighborhood effect reflecting differences among neighborhood

Table Occ_effects. The summed AICc weights for explanatory variables for each species from occupancy models of 250 fisher prey plots in 2012, 2014, and 2015 on Stirling in northern California.

Variable	Peromyscus	Neotoma	Tamiasciurus	Sciurus	Otospermophilus
QMD[MS]	0.0066	0.0072	0.7029	0.9978	0.0000
QMD[N]	0.6836	0.3263	0.0005	0.0003	0.0484
LPA[N]	0.0057	0.0023	0.0010	0.9900	0.0000
BACon[MS]	0.0000	0.0000	0.0000	0.1650	0.7740
QMD[St]	0.0031	0.5569	0.0674	0.0001	0.0529
pHard[MS]	0.4857	0.0000	0.0000	0.0002	0.0053
MBA[N]	0.0032	0.0046	0.0000	0.0000	0.2480
shrubs[MS]	0.1902	0.0000	0.0002	0.0000	0.0128
BAHard[MS]	0.0212	0.0000	0.0014	0.1167	0.0268
BA[St]	0.0031	0.0053	0.0753	0.0000	0.0726
pCon[MS]	0.1516	0.0000	0.0004	0.0000	0.0000
SPA[St]	0.0068	0.1387	0.0000	0.0002	0.0000
Canopy[St]	0.0035	0.0050	0.0993	0.0000	0.0000
Canopy[MS]	0.0031	0.0132	0.0665	0.0000	0.0096
Dist2Road	0.0031	0.0701	0.0000	0.0000	0.0000
RoadDensity	0.0032	0.0047	0.0535	0.0001	0.0058
Num_snag[MS]	0.0488	0.0000	0.0005	0.0001	0.0000
Year	0.0016	0.0428	0.0001	0.0000	0.0028
Canopy[N]	0.0031	0.0000	0.0000	0.0000	0.0380
Slope	0.0093	0.0107	0.0000	0.0000	0.0131
TBA[N]	0.0203	0.0067	0.0005	0.0003	0.0037
BA[MS]	0.0031	0.0111	0.0000	0.0000	0.0076
Elevation	0.0044	0.0170	0.0004	0.0000	0.0000
SPA[N]	0.0043	0.0063	0.0000	0.0001	0.0000

Aspect	0.0000	0.0000	0.0000	0.0011	0.0090
num_logs[MS]	0.0031	0.0000	0.0000	0.0000	0.0056
NumLogs[N]	0.0000	0.0063	0.0000	0.0000	0.0000
pCon[N]	0.00704	0.0000	0.9860	0.0002	0.0000
pHard[N]	0.00438	0.0276	0.0000	0.0019	0.0000

Table VTI. Number of plots within each vegetation index category (0 lowest predicted quality) and their respective proportions of total sampled plots. For each vegetative category the proportion of plots falling below discrete thresholds.

Veg Index	Plot Nums	Percent	Canopy ^a	DBH ^b	Basal Area ^c	LPA ^d	SPA ^e	pHardwood ^f
0	13	0.05	1.00	1.00	1.00	1.00	1.00	1.00
1	24	0.10	1.00	0.42	1.00	0.58	1.00	1.00
2	7	0.03	0.00	0.00	1.00	1.00	1.00	1.00
3	22	0.09	0.00	0.00	0.00	1.00	1.00	1.00
4	48	0.19	0.00	0.00	0.04	0.96	0.69	0.31
5	95	0.38	0.00	0.00	0.00	0.39	0.21	0.40
6	40	0.16	0.00	0.00	0.00	0.00	0.00	0.00

^a Threshold $\leq 10\%$ canopy

^b Threshold $\leq 12.7\text{cm}$ QMD

^c Threshold $\leq 15.2\text{ m}^2$

^d Threshold ≤ 1 logs/acre

^e Threshold ≤ 2 snags/acre

^f Threshold $\leq 20\%$ hardwoods

Table CPUE2. Total number of captures for 7 common genera of mammals captured in 2012, 2014, and 2015 on fisher prey sampling plot in northern California.

Genera	2012	2014	2015	Grand Total
Peromyscus	207	345	355	907
Neotoma	34	137	121	292
Otospermophilus	6	12	18	36
Tamiasciurus	10	2	4	16
Sorex	3	5	1	9
Tamias	0	0	7	7
Sciurus	2	0	0	2

Table CPUE1. The total number of captures, the mean capture per unit effort, standard deviation of capture per unit effort (Std CPUE), total number of plots sampled, and total trap effort in each year (2012, 2014, and 2015) on Stirling in northern California.

Metric	2012	2014	2015	Total
Total Captures	265	501	508	1274
Mean CPUE	0.092	0.115	0.173	0.135
Std CPUE	0.072	0.100	0.181	0.140
n CPUE	48	98	104	250
Total Effort	2939	4409	4591	11929

Table_MNA_1. The top 20 models hypothesized to explain the minimum number of species alive across three years of small mammal sampling data on Stirling property in 2012-2015. We display Akaike's information criterion corrected for sample size (AICc), the difference between the top model and other models (Δ AICc), the model likelihood, the model sum AIC weight and the difference in the null and residual deviance (Δ Deviance).

Models	AICc	Δ AICc	Likelihood	w	Δ Deviance
DistH20+DistRoad+Slope+BA+QMD	1246.13	0.00	1.000	0.114	247.2
DistH20+DistRoad+Slope+BA	1246.90	0.77	0.680	0.077	217.7
DistH20+DistRoad+Slope+BA+Week	1247.08	0.945	0.623	0.071	237.6
DistH20+DistRoad+Slope+BA+No_Hard	1247.20	1.062	0.588	0.067	236.4
DistH20+DistRoad+BA+QMD	1247.28	1.144	0.564	0.064	213.9
DistRoad+DistH20+Slope	1247.33	1.194	0.550	0.063	191.7
DistH20+DistRoad+Slope+QMD	1247.38	1.243	0.537	0.061	212.9
DistH20+DistRoad+No_DS+Slope	1247.49	1.36	0.507	0.058	211.7
DistH20+DistRoad+BA	1248.09	1.951	0.377	0.043	183.8
DistRoad+DistH20+Year+Slope+BA	1248.10	1.97	0.373	0.042	227.2
Slope+No_DS+DistRoad+DistH20+BA	1248.17	2.031	0.362	0.041	226.6
DistH20+DistRoad+Per_Con+Slope	1248.29	2.151	0.341	0.039	203.5
DistH20+DistRoad+Slope+BA+Week+Year	1248.38	2.243	0.326	0.037	246.2
DistH20+DistRoad+Slope+Mean_DS	1248.41	2.273	0.321	0.037	202.3
DistH20+DistRoad+Slope+Year	1248.50	2.37	0.306	0.035	201.3
Slope+Per_Con+DistRoad+DistH20+BA	1248.51	2.373	0.305	0.035	223.1
DistH20+DistRoad+Slope+BA+Shrub_Index	1248.66	2.523	0.283	0.032	221.5
DistH20+DistRoad+Slope+BA+Mean_DS	1248.67	2.534	0.282	0.032	221.4
DistH20+DistRoad+Slope+BA+Per_Hard	1249.00	2.862	0.239	0.027	218
DistH20+DistRoad+Slope+Shrub_Index	1249.11	2.978	0.226	0.026	195

Table_MNA_2. Summary statistics for variables included in the GLMs including the summed Akaike weight (ΣW), sum of the product of the likelihood measures and the covariates (AM_{avg}), proportion of models with a significant p-value ($\alpha = 0.05$), and the upper and lower bounds of the 95% confidence intervals around the beta estimates of the covariates.

Variable	ΣW	AM_{avg}	Σp -values	$CI_{AvgLower}$	$CI_{AvgUpper}$
DistH2O	1.000	2.621	1.00	0.448	2.341
DistRoad	1.000	0.267	1.00	0.161	1.042
Slope	0.893	0.286	0.39	-0.002	0.065
BA	0.683	0.024	0.15	-0.001	0.008
QMD	0.276	-0.014	0.00	-0.029	0.004
Year	0.114	-0.201	0.00	-0.628	0.227
Week	0.108	0.085	0.00	-0.041	0.219
NumSnag	0.099	0.130	0.00	-0.109	0.401
pCon	0.074	-0.372	0.00	-1.856	0.718
Num_Hard	0.067	-0.026	0.00	-0.070	0.012
Shrubs	0.058	-0.031	0.00	-0.463	0.255
MeanSnag	0.032	0.007	0.00	-0.051	0.118
<u>pHard</u>	0.027	-0.028	0.00	-1.491	1.254

Table_Div_1. Top 20 models generated to explain Simpson's transformed reciprocal index of diversity alive across the course of the study. We display Akaike's information criterion corrected for sample size (AICc), the difference between the top model and other models (Δ AICc), the model likelihood, the model sum AIC weight and the difference in the null and residual deviance (Δ Deviance).

Models	AICc	Δ AICc	Likelihood	w	Δ Deviance
Year+Week+QMD+Slope	542.2	0.000	1.000	0.112	20.340
Week+QMD+CanopyCover+Per_Hard	542.4	0.217	0.897	0.101	20.230
Year+Week+QMD+CanopyCover+Per_Hard	542.5	0.267	0.875	0.098	21.280
Week+QMD+Per_Hard+CanopyCover+Slope	542.6	0.442	0.802	0.090	21.190
Year+Week+QMD+Per_Hard+CanopyCover+Slope	542.7	0.458	0.795	0.089	22.250
Week+Per_Hard+QMD+No_DD+CanopyCover	543.6	1.419	0.492	0.055	20.700
QMD+Year+Week	543.7	1.446	0.485	0.055	18.540
Week+QMD+CanopyCover+Slope	543.7	1.485	0.476	0.054	19.590
Year+Week+QMD+Slope+CanopyCover	543.8	1.587	0.452	0.051	20.610
Week+Per_Hard+QMD+No_DS	544.2	1.967	0.374	0.042	19.350
Year+Week+QMD+No_DS	544.7	2.509	0.285	0.032	19.070
Year+Week+QMD+CanopyCover	545.0	2.819	0.244	0.027	18.910
Week+Per_Hard+QMD+No_DD+No_DS	545.3	3.079	0.214	0.024	19.860
Year+Week+QMD+No_DD	545.63	3.381	0.184	0.021	18.630
Year+Week+QMD+No_DS+No_DD	546.7	4.491	0.106	0.012	19.140
Week+Per_Hard+QMD+Slope	547.2	5.042	0.080	0.009	17.770
Week+Per_Hard+QMD+No_DD+Slope	547.7	5.476	0.065	0.007	18.640
Week+Per_Hard+QMD+No_DD	548.1	5.844	0.054	0.006	17.360
Year+Week	550.7	8.471	0.014	0.002	13.790
Week+Per_Hard+No_DS+No_DD	551.6	9.442	0.009	0.001	15.490

Table_Div_2. Summary statistics for variables included in the GLMs including the summed Akaike weight (ΣW), sum of the product of the likelihood measures and the covariates (AM_{avg}), proportion of models with a significant p-value ($\alpha=0.05$), and the upper and lower bounds of the 95% confidence intervals around the beta estimates of the covariates.

Variable	ΣW	AM_{avg}	$\Sigma Pvalues$	$CI_{AvgLower}$	$CI_{AvgUpper}$
Week	1.003	-0.39	0.90	-0.107	0.116
QMD	0.953	0.24	0.90	0.0655	0.122
Canopy	0.639	-0.02	0.55	-0.008	0.033
pHard	0.591	1.20	0.50	-0.001	0.0154
Year	0.560	0.63	0.44	0.007	0.014
Slope	0.455	0.02	0.00	-0.237	0.338
NumSnag	0.125	-0.05	0.25	0.0398	0.072
NumLog	0.107	-0.05	0.00	-0.049	0.065

Figures

Figure 1. Locations of fisher prey sampling plots by year (Plots) and bounded neighborhood around those plots in 2012, 2014, and 2015 with the home ranges of female fishers during 2012, 2013, and 2014 on the Stirling district owned by Sierra Pacific Industries in northern California.

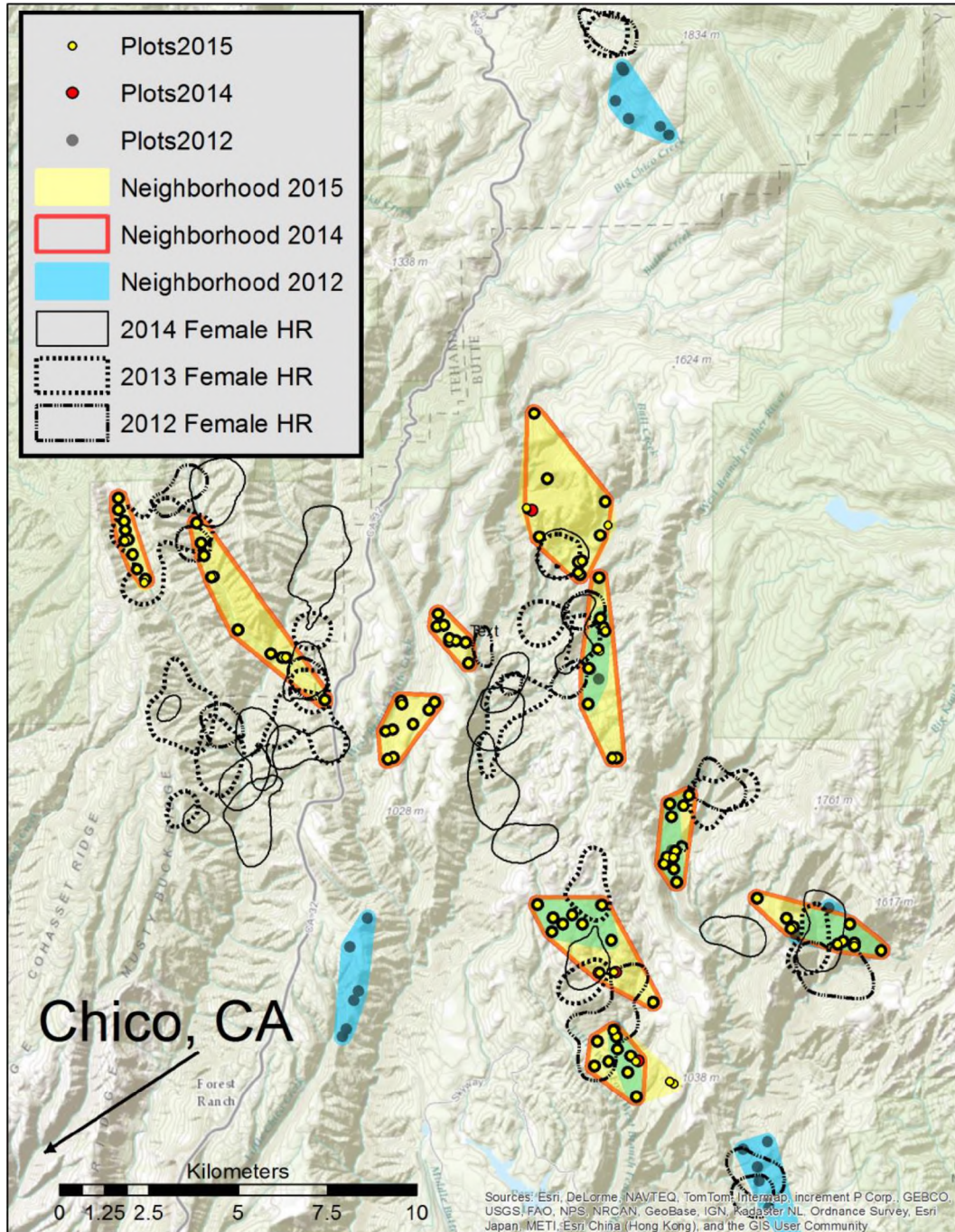


Figure Trap_Bias. Naïve occupancy estimates for 9 genera of mammal using both trapping and camera data (blue bars) and camera data exclusively (gray bars) at 98 fisher prey sampling plots during 2014 on Stirling in northern California.

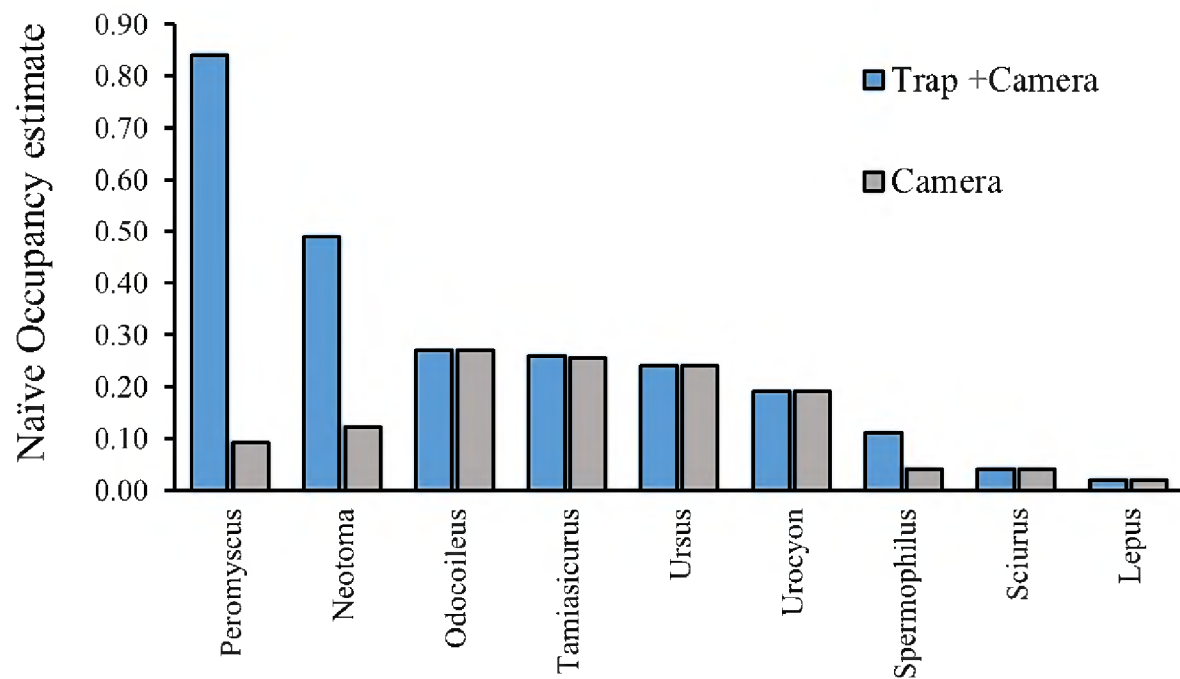


Figure WoodratOcc. The predicted relationship of quadratic mean diameter at breast height of trees at the stand-level to the probability that a plot was occupied by woodrats for 250 fisher prey sampling plots in northern California during 2012, 2014, and 2015.

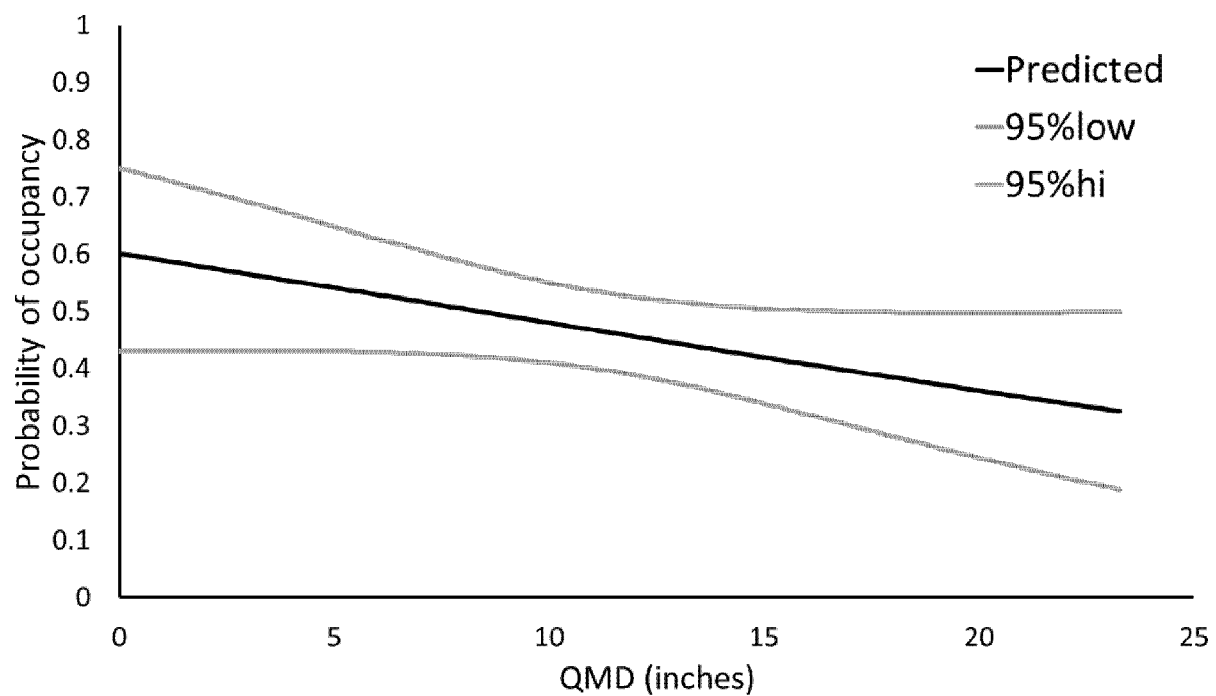


Figure Chickaree. A. Estimated relationship between the percent of trees that were coniferous within a neighborhood to the probability of chickaree occupancy and B. Estimated relationship between quadratic mean diameter of trees at a plot to the probability of chickaree occupancy for 250 fisher prey sampling sites in northern California in 2012, 2014, and 2015.

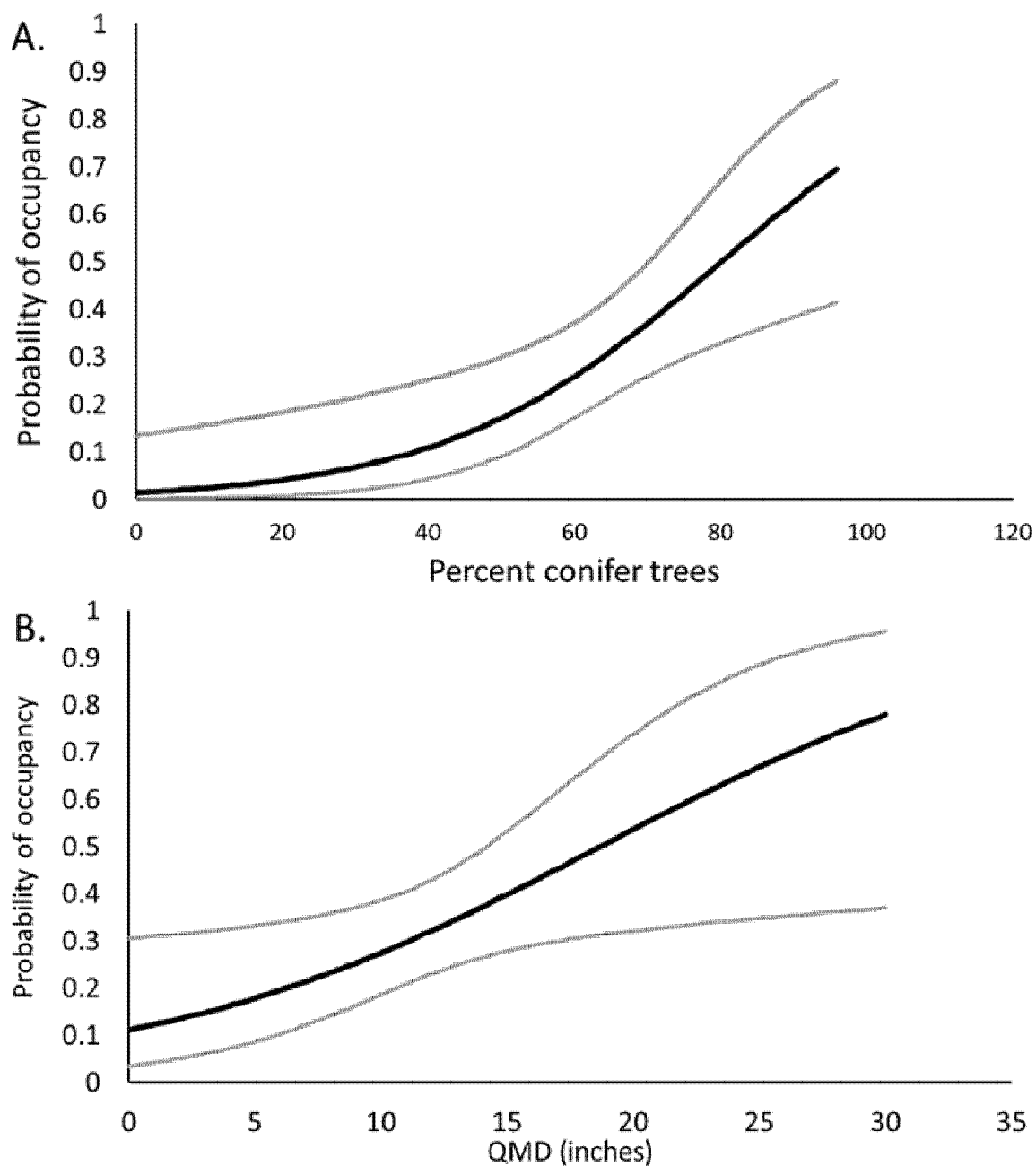


Figure Gray_Detection. The estimated probability of detecting gray squirrels as a function of the mean number of logs per acre within a neighborhood changes for 250 fisher prey sampling plots in northern California in 2012, 2014, and 2015.

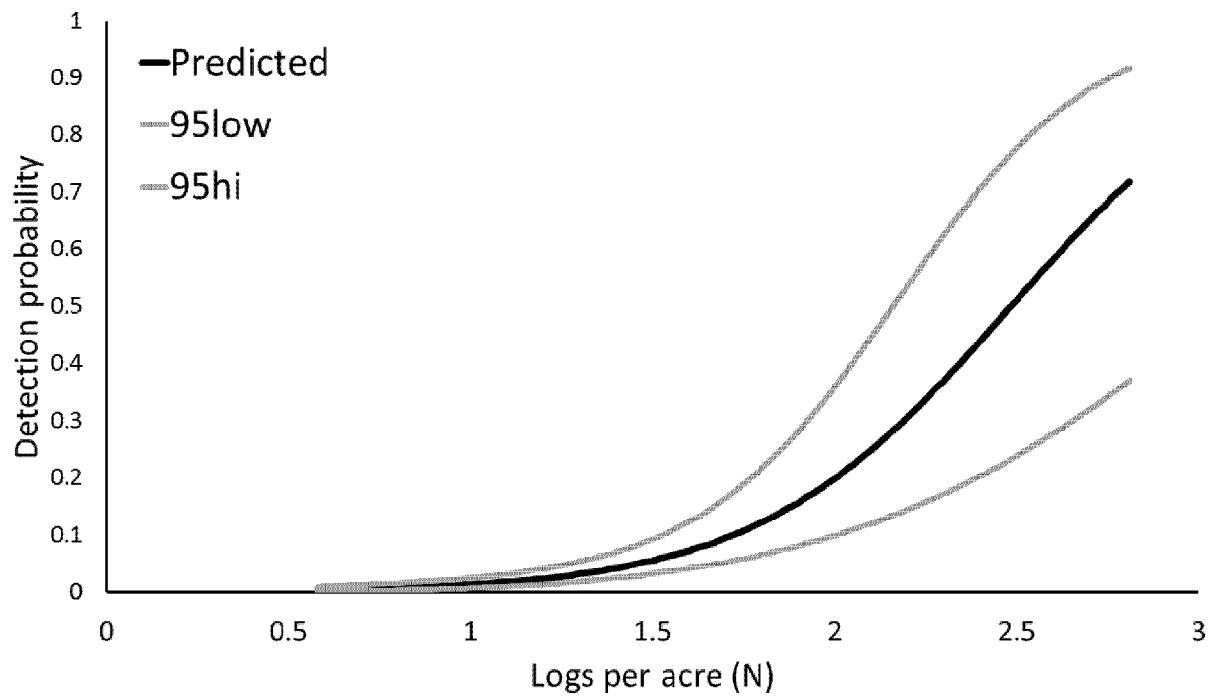


Figure SPBEE_Detec. Estimated detection rates of California ground squirrel (*Otospermophilus beecheyi*) across all neighborhoods on Stirling in northern California during 2012, 2014, and 2015.

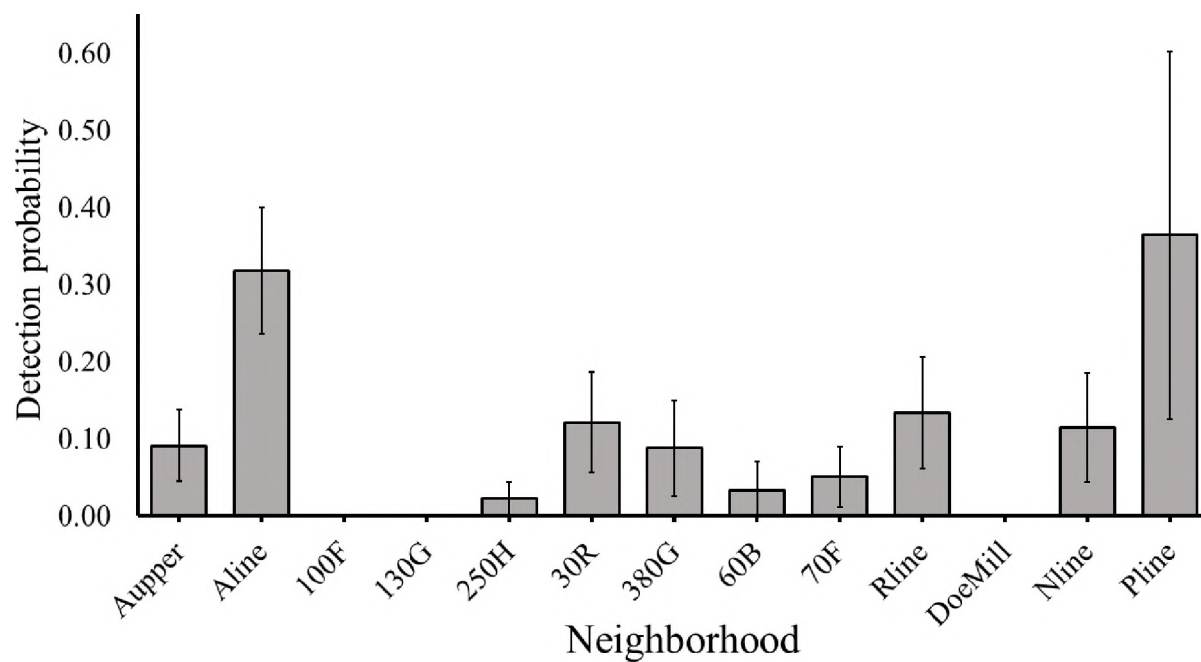


Figure SPBEE_OCC. The estimated probability of detecting California ground squirrels as a function of the basal area of conifers at a plot for 250 fisher prey sampling plots in northern California in 2012, 2014, and 2015.

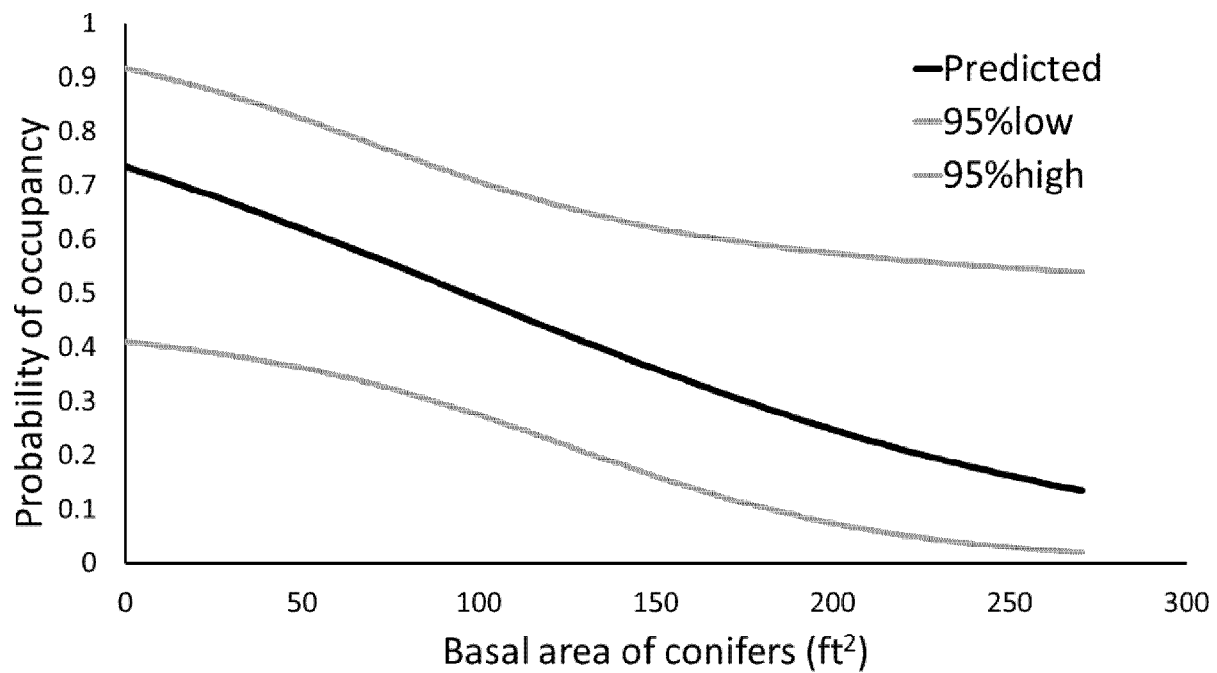


Figure VT12. The total value of a locations for 6 vegetation metrics (canopy cover > 10%, QMD > 12 cm, mean basal area > 15m², logs per acre >1, snags per acre > 1, and percent hardwoods > 20%) above a specific threshold on Stirling in northern California.

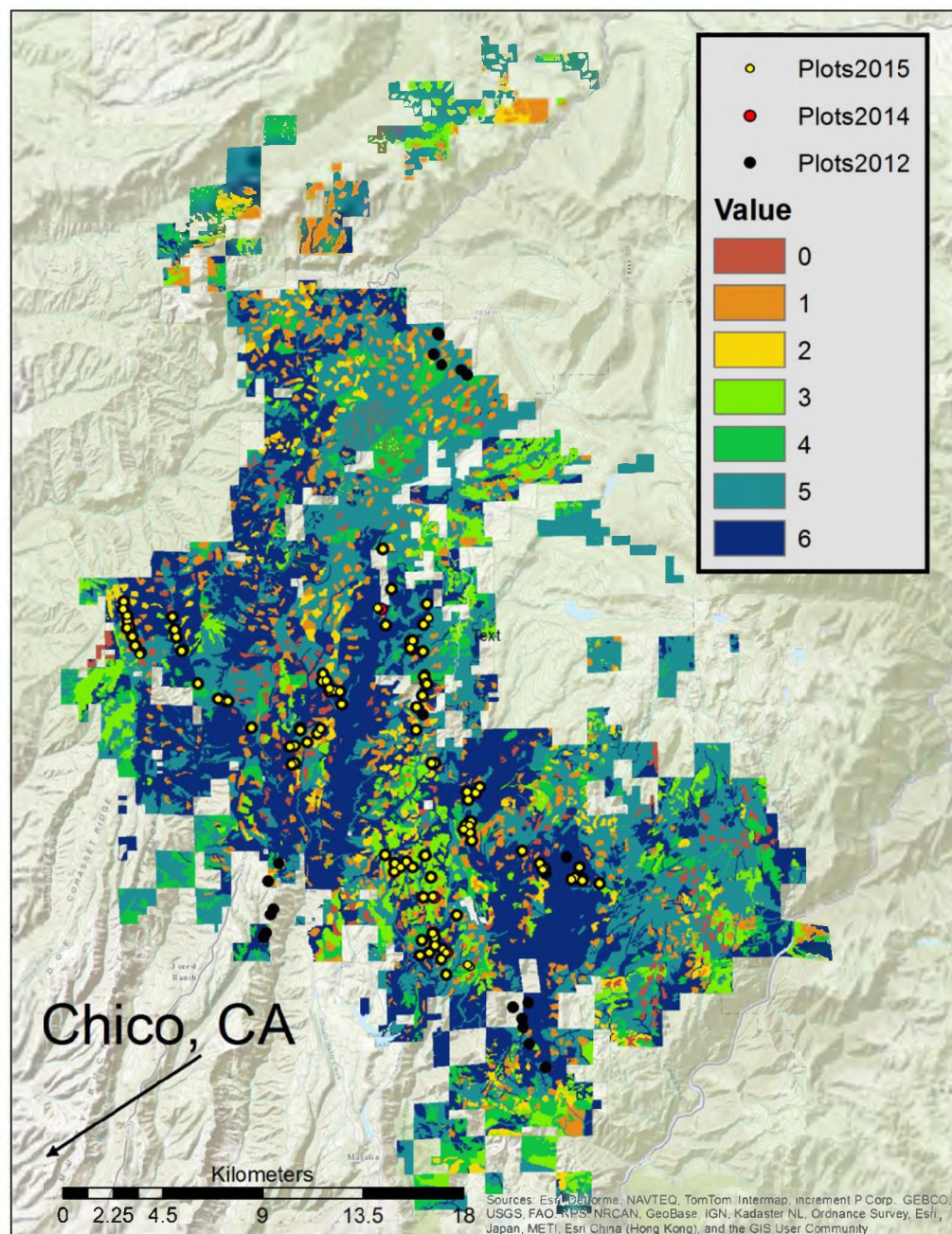


Figure VTI_Wood. Top) The proportion of detections for all woodrats (*Neotoma* spp) at plots meeting 0-6 threshold metrics for canopy cover (>10%), quadratic mean diameter (>12.7 cm), basal area (>15.2 m²), mean logs per acre (>1), mean snags per acre (>2), and percent hardwood trees (>20%) and Bottom) The Ivlev electivity index value for woodrats at each threshold value for 250 plots in northern California in 2012, 2014, and 2015.

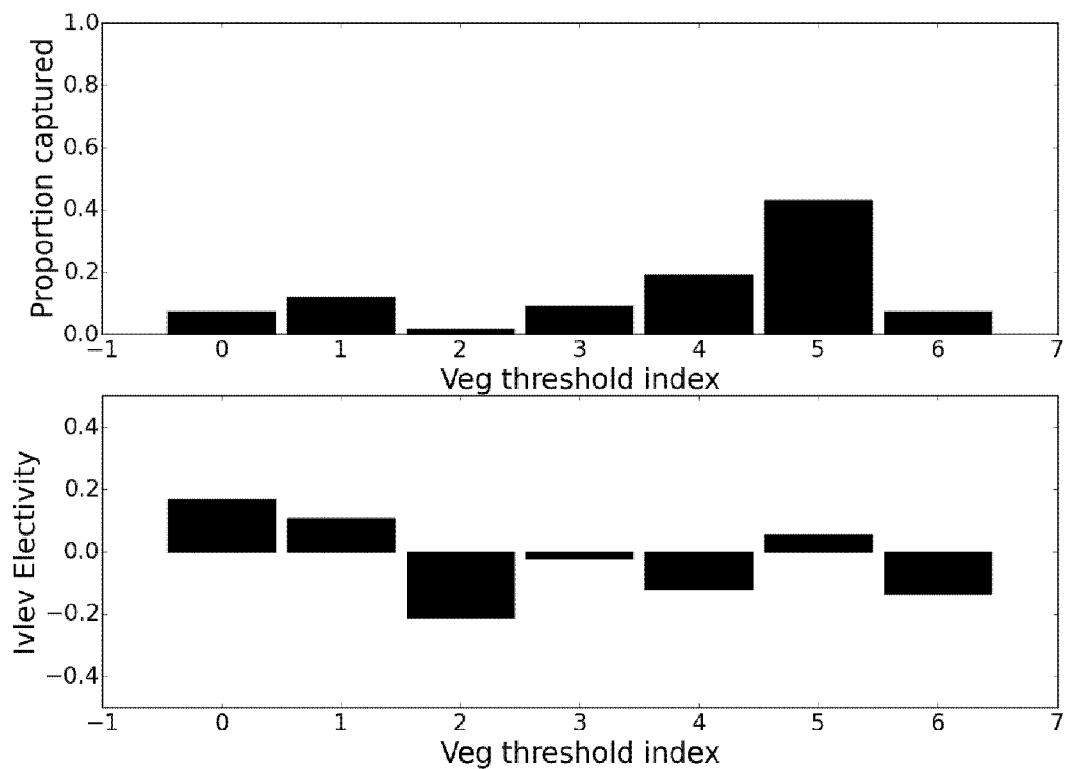


Figure FTI_Ground. Top) The proportion of detections for all California ground squirrels (*Otospermophilus beecheyi*) at plots meeting 0-6 threshold metrics for canopy cover (>10%), quadratic mean diameter (>12.7 cm), basal area (>15.2 m²), mean logs per acre (>1), mean snags per acre (>2), and percent hardwood trees (>20%) and Bottom) The Ivlev electivity index value for woodrats at each threshold value for 250 plots in northern California in 2012, 2014, and 2015.

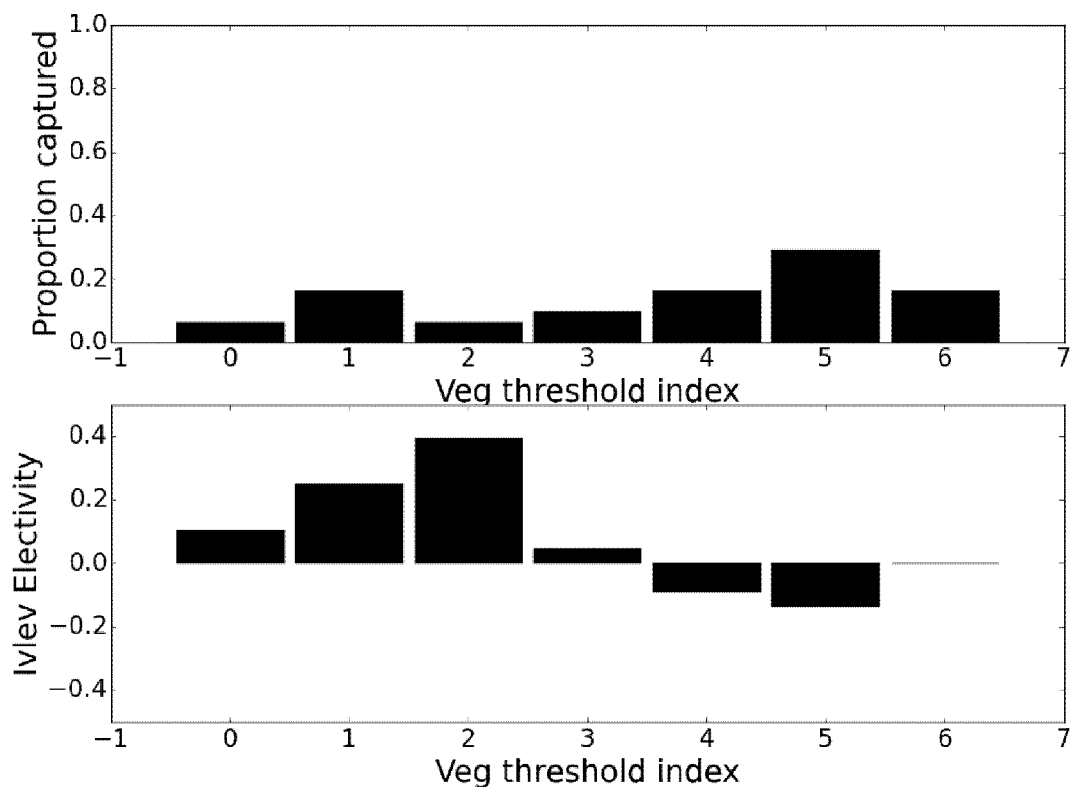


Figure TreeSquirrel. Top) The proportion of detections for all tree squirrels (*Sciurus griseus*, *Tamiasciurus douglasii*, and *Glaucomys sabrinus*) at plots meeting 0-6 threshold metrics for canopy cover (>10%), quadratic mean diameter (>12.7 cm), basal area (>15.2 m²), mean logs per acre (>1), mean snags per acre (>2), and percent hardwood trees (>20%) and Bottom) The Ivlev electivity index value for woodrats at each threshold value for 250 plots in northern California in 2012, 2014, and 2015.

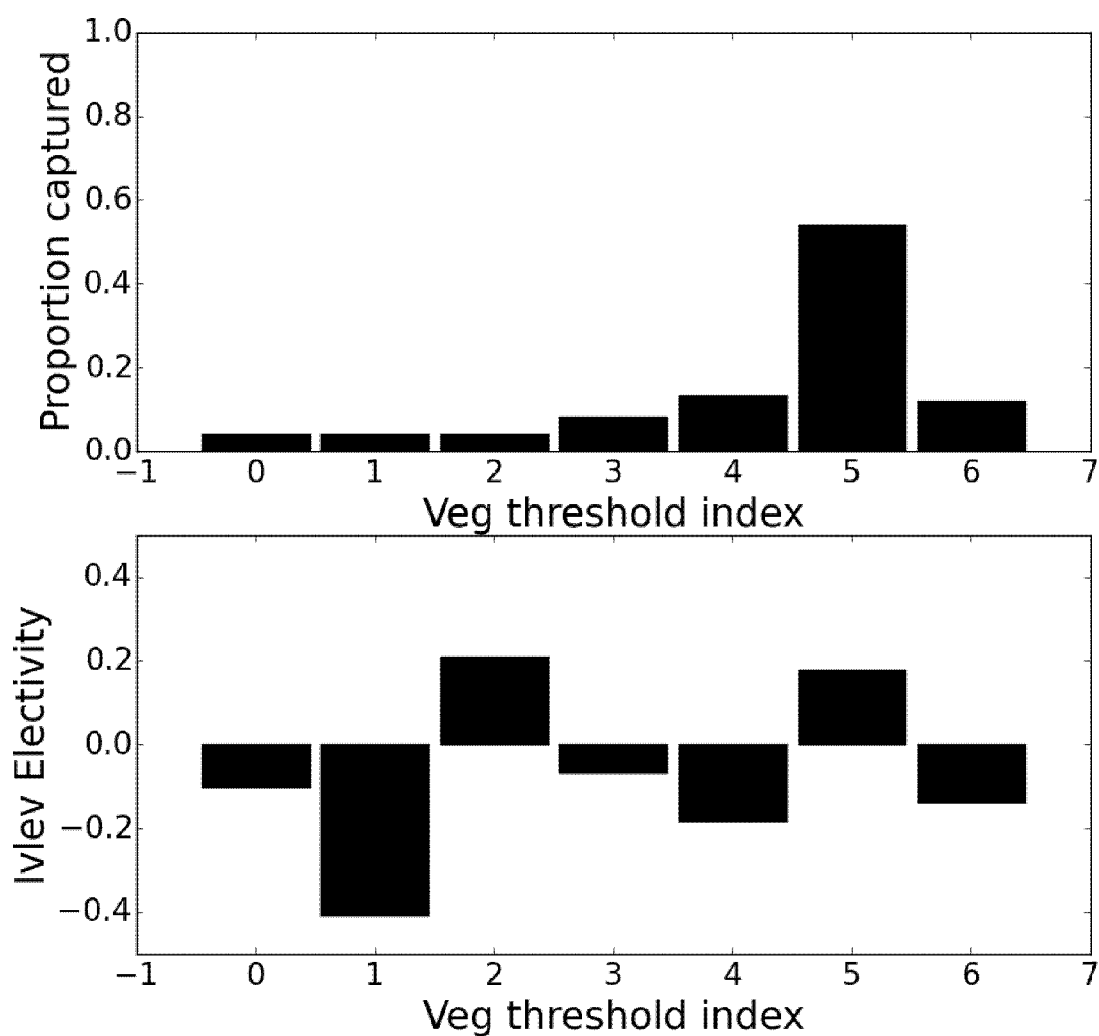


Figure CPUE. Mean catcher per unit effort (CPUE) for all fisher prey species captured or recaptured at all plots found within a neighborhood for years 2012, 2014, and 2015 on Stirling district in northern California.

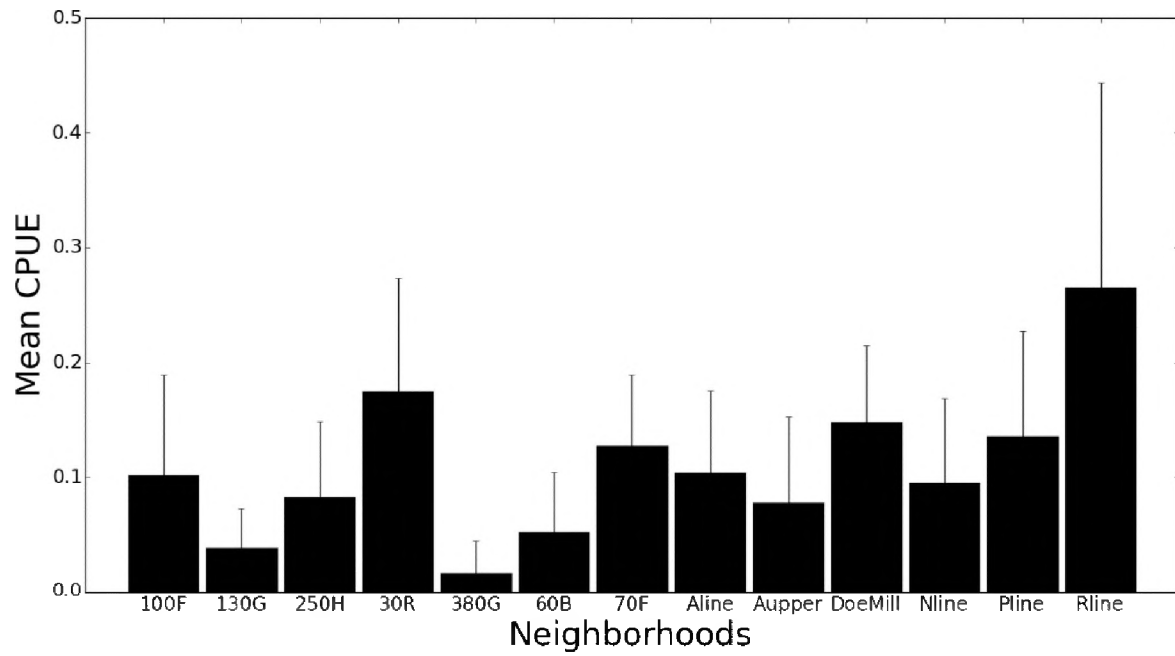


Figure CPUEx. Top) Regression catch per unit effort (CPUE) on the number of deer mice (*Peromyscus* spp) and Bottom) Regression of CPUE on the catch per unit effort rates excluding all deer mice captures for 250 fisher preys sampling plots in 2012, 2014, and 2015 on Stirling in northern California. Solid lines represent the predicted mean response and dashed lines are the 95% confidence limits.

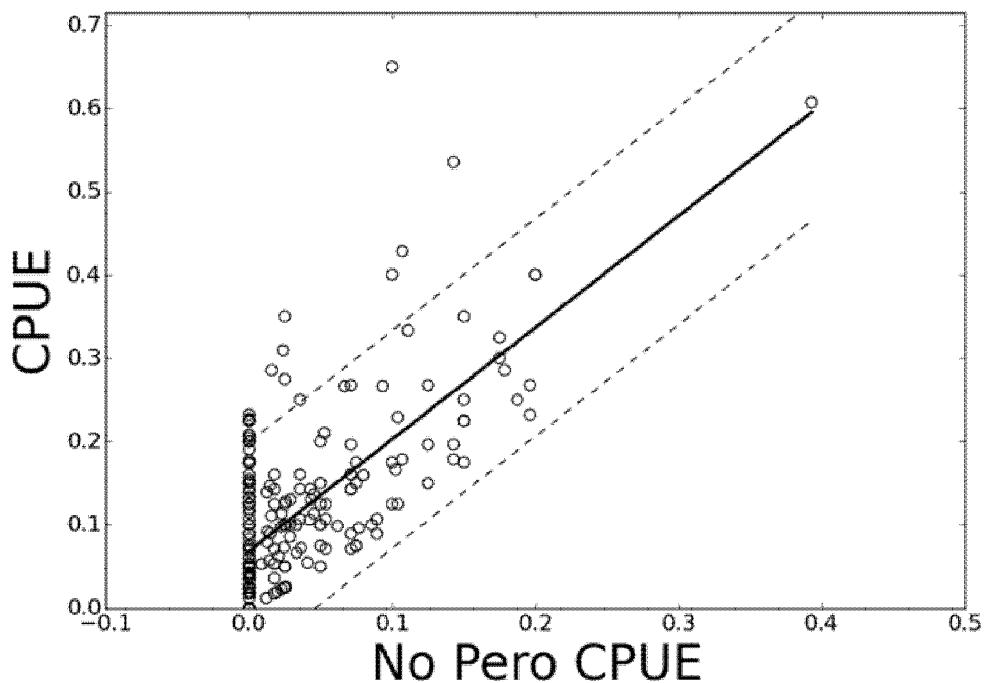
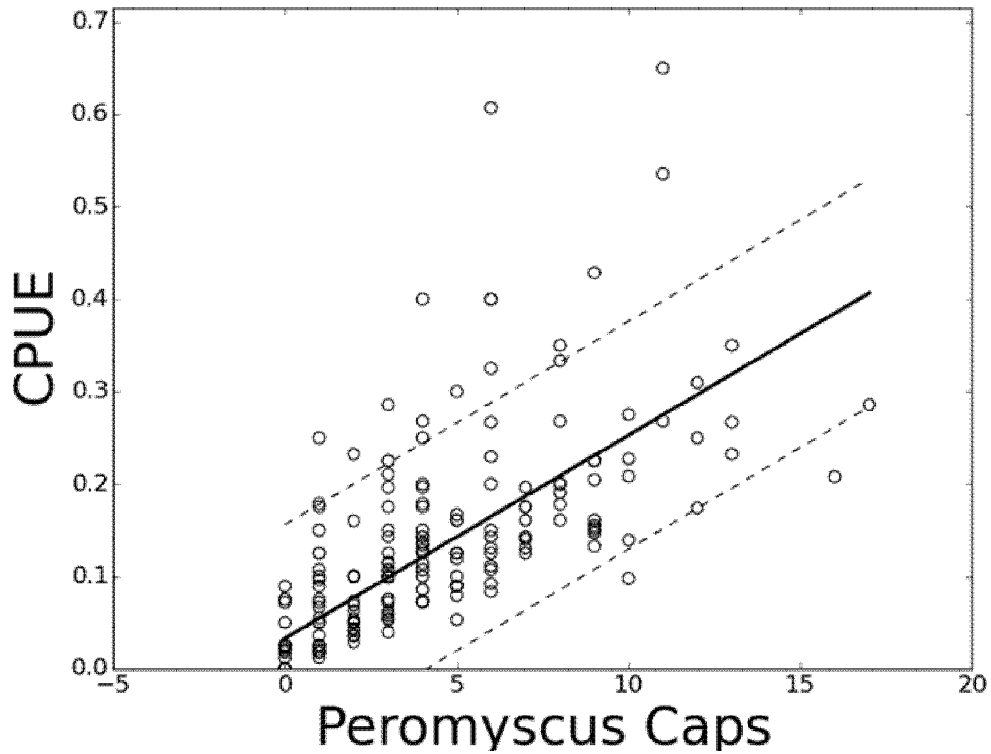
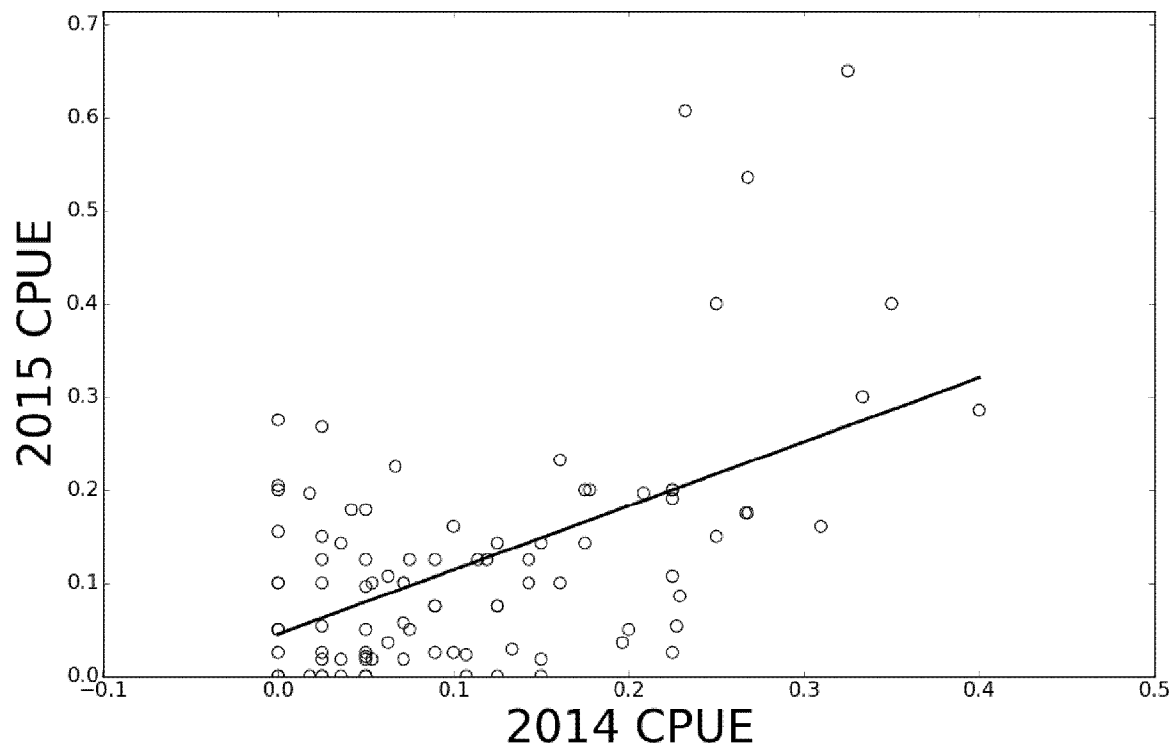
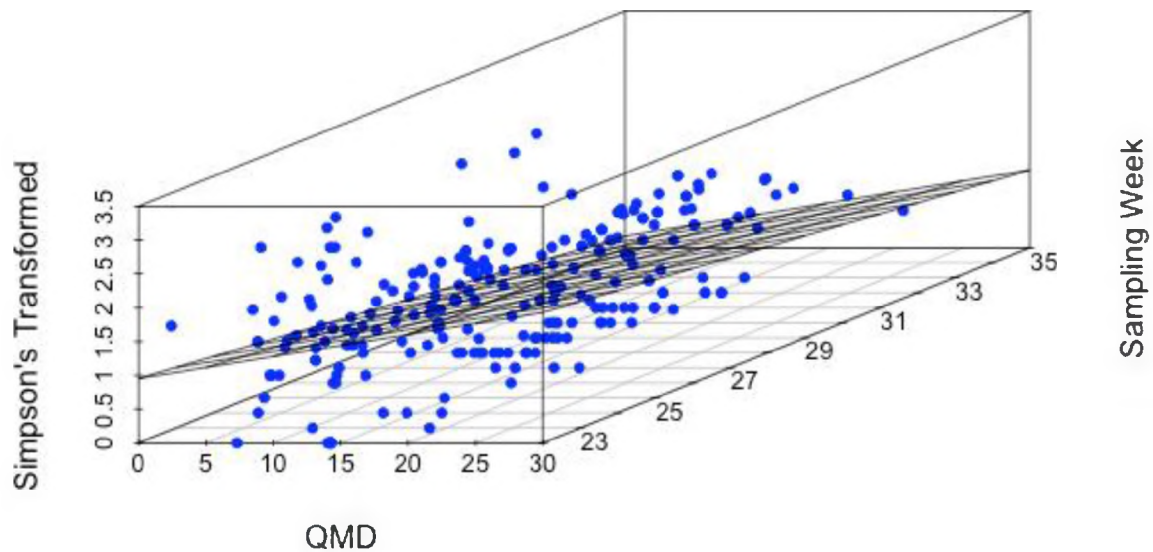


Figure CPUE_REG. Catch per unit effort values between plots sampled in both 2014 and 2015 with linear regression line ($R^2 = 0.28$, $p < 0.0001$, $df = 249$) for fisher prey plots on Stirling in northern California.



Figure_1_Div. Three dimensional scatterplot displaying the relationship between quadratic mean diameter (QMD), Simpson's Transformed reciprocal index of diversity and sampling week. As QMD increased, Simpson's index increased, but as the sampling week increased, there was a slight decrease in diversity. The grid overlay displays the logistic regression model relating Simpson's index to QMD and sampling week.



Figure_2_Div. Three dimensional scatterplot displaying the relationship between tree quadratic mean diameter (QMD), Simpson's Transformed reciprocal index of diversity (high values = high diversity) and sampling year. As both the sampling year and QMD increased, Simpson's index increased. The grid overlay displays the predicted logistic regression values relating Simpson's index to QMD and sampling year.

