

UNDERSTANDING A FISHER REINTRODUCTION IN NORTHERN CALIFORNIA FROM 2 PERSPECTIVES

Final Report

For the period October 2009 through December 2017

with emphasis on the 2017 research field season

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 2017, 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to learn whether pertinent publications are now available and, if not, that the information in this report has not be superseded or updated.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 40 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 reintroduced fishers and their progeny closely to document their survival, reproduction, dispersal, and home range development through 2017 (year-8). Released fishers experienced high survival (approximately 0.80) during both the initial post-release period (4 months) and for up to 2 years after release. Of the fishers in the 3 release cohorts, we tracked 32 (80%) long enough to document the establishment of home ranges. Males established larger home ranges and travelled farther than females. From November 2009 through December 2017, we capture 220 different, individual fishers (122 F, 98 M). During our annual trapping effort in October-November 2017, we totaled 106 captures of 61 individual fishers (38 F, 23 M), including 27 new fishers (14 F, 13 M), 21 apparent juveniles (12 F, 9 M). We recaptured no fishers that had been reintroduced onto Stirling during our annual trapping effort in 2017 but we did recapture 1 reintroduced male after that effort. Across years, roughly 60% of fishers carried at least some fleas, ticks or eye worms (*Thelazia californiensis*) with fleas generally being the most common. Some fishers live-trapped in Humboldt and western Trinity counties for possible release on to the Stirling Management Area were infected with a previously undescribed trematode. We did not knowingly release any fishers infected with this trematode onto Stirling and, through 2017, we captured no fishers that appeared to be infected with this parasite. We have documented 38 fisher mortalities since 2009. Fisher age had the greatest effect on survival. Twelve mortalities were caused, or possibly caused, by predation, 8 were potentially related to exposure to rodenticides, and 5 were caused by accidents. Fishers reproduced in all years of the study and from each of the 3 reintroduced cohorts. Our best estimates of survival and reproduction are consistent with a stable or growing population on Stirling and, through 2017, reproduction and survival appear not to be limited by logging and other forest management activities. Our population modelling indicates that the fisher population on Stirling is growing but neither short-term population stability nor long-term viability will be demonstrable statistically before year-10 (2020). The population appears unlikely to go extinct in the coming 20 years barring dramatic decreases in survival and reproduction caused by unpredictable events, such as wildfire or an epizootic.

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Introduction

The human-assisted movement of animals dates 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 reintroductions (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). Concern about 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). Nonetheless, critical, hidden effects of reintroductions, such as the effect on a source population of removing prime, reproductive, adult animals (animals with high reproductive value, v_x) for release elsewhere, have received little attention (Armstrong and Seddon 2008, Powell et al. 2012). The effects on a source population 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, especially males.

Because of concern for the status of fishers in California, to understand how fishers (and how 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 (formerly Fish & Game), the US Fish & Wildlife Service, Sierra Pacific Industries (hereafter “Sierra Pacific”) and North Carolina State University 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 Sierra Pacific’s 648 km² Stirling Management Area (hereafter “Stirling”), which is managed for timber production (Figure 1). The Memorandum of Understanding initiating our research states that released fishers and their progeny are to be studied intensively for the 7 years following the year-1 reintroductions, amended to be 8 years.

In a related effort to understand the fisher population in the far northeastern extent of the fisher’s range in California, we began independently in 2006 to use non-invasive methods to estimate population parameters for the fishers living on the managed, forested landscape centered on the Klamath River in northern California and southern Oregon (Figure 1). Combining the non-invasive, genetic surveys conducted in this study area with the research on reintroduced fishers on Stirling provided the opportunity to broaden conservation benefits for fishers (e.g., Seddon et al. 2007, Sarrazin and Barbault 1996), to understand better the dynamics of fisher populations on managed landscapes, and to study a source population for a reintroduction. We moved fishers from this study area to Stirling in the winters of 2009-2010 and 2010-2011. These removals were targeted to lands owned by Timber Products Company, meaning that fishers were removed from managed, industrial timberlands and released on a different but also managed landscape. The removed fishers were targeted to be adult members of the population with high reproductive potential.

Thus, our research on fishers has been done from 2 important and complementary perspectives: 1) from the perspective of how land and forest management affect the abilities of fishers to maintain self-perpetuating populations and 2) from the perspective of basic research and testing hypotheses aimed to increase our



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.

knowledge of fishers, and our knowledge of medium-sized carnivorans in general. The year 2017 was the last and 8th year of this research.

The specific objectives of this research have been

- To estimate annual survival and reproduction of fishers on Stirling from 2010 through 2017.
- To evaluate habitat selection by reintroduced fishers and their offspring and 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 tissue 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 prevalence of and exposure to diseases by reintroduced fishers and their offspring and 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 in 2013-2016.
- To estimate abundance, survival and recruitment, population growth rate and fisher occupancy through 2016 for the source population of fishers on the Eastern Klamath Study Area.
- To estimate the effects on abundance and population growth rate, if any, caused by removing 9 adult fishers (an estimated 17% of the population) from the source 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 objectives directly for year-8, January– December 2017, and cumulatively for all years of the project. We review non-invasive research in the Klamath Region and the reintroduction activities through 2017.

Terms and Definitions

Historically, conservation reintroductions were considered successful or unsuccessful if members of the target species were still present several years after initial releases (Sarrazin and Barbault 1996, Powell et al. 2012). Success has been predicated largely on whether a stable or growing population was established only a few years after initial releases. Although this definition of success is common, it is unsatisfactory because it lacks precision with respect to what defines success for a project and it will often fail to identify the causes and specific times when a reintroduction may have failed. For these reasons, we have included terms and definitions for this reintroduction as well as clarification of the collaborator's perceptions of important time frames and milestones within the general approach and conception of the project. We include this section to reduce errors and confusion within the context of our project and our reported results.

Success

Our collaborative group conceptualizes and defines the *success* of our reintroduction on two separate, but interrelated, standards: success of the *Project*, and success of the *Population* (Facka 2016). Our primary objectives are aimed at understanding if and how fishers use and respond to landscapes managed for timber production. Thus, the success of our *project* is not predicated upon whether a population of fishers is established successfully and becomes persistent on, or near, Stirling in the long-term. Rather, we define *Project Success* as understanding and documenting the mechanisms that dictate fisher population dynamics on Stirling and, more generally, on industrial landscapes that are managed primarily for timber production. If fishers go extinct on Stirling and our research elucidates why and is able to apply the research to general mechanisms, we would consider the project to have been successful.

Alternatively, if fishers persist and expand on and near Stirling, but our research fails to elucidate the reasons for their persistence, then the project could be considered a failure. We view our reintroduction as having inherent scientific value and that such scientific value is a major component of the project's success independent of a persistent fisher population on Stirling.

Our second objective was to document a viable, reintroduced population of fishers on Stirling, *i.e.* a population with a stable or growing, long-term population size. We call this *Population Success*. Consequently, our second measure of success relates specifically to the dynamics of the fisher population on and near Stirling. "Long-term" is a relative term that for populations of mammals implies several to many years and is not some time period that can be evaluated easily on a year-to-year basis. The minimum time needed to estimate population parameters for long-lived mammals is 5-7 years (Anthony et al. 2006, Brongo et al. 2005). Therefore, we suggest that after the population has been established (definition below) that at least 2 metrics be evaluated annually to assess the general trend in population growth rate and to indicate trends regarding the viability and persistence of the population. The first metric is the trend in population size and population growth rate estimated from our annual fall trapping, telemetry and non-invasive surveys. Since the annual variability of these parameters may be high, population growth rate should be estimated over a period of at least 5 years post-establishment. The second metric is the projected long-term population growth rate taken from stochastic simulations (population viability analyses, described below) that are parameterized with estimates of survival and reproduction rates and their variances for fishers on Stirling, and parameterized with predicted habitat changes on Stirling due to projected logging. Estimates of survival and reproduction will be made from trapping, telemetry, non-invasive surveys and other pertinent data. Our cooperative project has not agreed on a definition of *Population Success* but we suggest a working definition: *Population Success* will exist when 1) after year-10, the 5-year estimate of population growth rate is 0 or above and the 95% confidence interval for growth rate does not include 0, and 2) when the population projections show an 80% probability that the population will not go extinct as long as ecological conditions remain the same.

Establishment and Viability

Following Armstrong and Seddon (2008), the Stirling reintroduction has 2 distinct phases: The *Establishment*

Phase during which we establish a fisher population, and the *Viability* Phase after the population has been established. The Establishment Phase is the period of time during and immediately after the actual release of fishers. During this phase, the population is small and at greatest risk of disappearing due to random demographic and environmental events (Allee and Bowen 1932, Armstrong and Seddon 2008, Courchamp et al. 2008). The primary rationale for identifying the Establishment Phase is that random events can cause extinction of very small populations despite adequate habitat (Armstrong and Seddon 2008). We consider the Establishment Phase concluded when the population does not require augmentation to support normal population processes, such as mating, reproduction and recruitment. Having established the population implies neither long-term population viability nor lack of risks for a population. We consider that our fisher population had become established by 2012, by which time no new fishers were released and all population growth was caused by reproduction by fishers on Stirling.

The *Viable* Phase (alternately "persistence phase"; Armstrong and Seddon 2008) begins after the population is established and continues indefinitely thereafter. During this phase, our reintroduced fisher population can demonstrate its own viability. The population does not require augmentation or other intervention from humans to maintain normal population functions. The population is controlled by the ecological conditions on the landscape, such as habitat, prey, predators and climate. The population may go extinct, stabilize, or increase in size and geographic extent during the viability phase and, therefore, the viability phase does not imply that the population is viable, only that it will demonstrate its own ability to be viable or not (Facka 2016). Defining viable is always difficult because viable can only be defined for a given set of conditions, yet ecological conditions always change. Defining viable is particularly difficult for Stirling, where the landscape of the reintroduction changes year-by-year due to logging and other forest management activities. The result is that a population can be defined as viable only in the present and in the past, with a likelihood of being viable in the future. Since Sierra Pacific is committed to monitoring fishers on Stirling until the end of its Candidate Conservation Agreement with Assurances (scheduled for 2026), we suggest that a final determination on viability be made at the conclusion of the CCAA. Interim assessments will document viability when population metrics show *Population Success*.

Years

We define the years of the project as follows.

Year-1: November 2009 – early October 2010. Year-1 began when we commenced live-trapping fishers to be released on Stirling. Live-trapping for moving fishers ceased for year-1 in January 2010, when the last fisher of the year-1 cohort was released. From January through September 2010 we live-trapped occasionally on Stirling to try to catch particular fishers.

Year-2: October 2010 – early October 2011. Year-2 began when we commenced live-trapping fishers to be released on Stirling. Live-trapping for moving fishers for year-2 ceased in February 2011, when the last fisher of the year-2 cohort was released. From February through September 2011, we live-trapped occasionally on Stirling to try to catch particular fishers. In October 2011, we made a small but concerted effort to try to catch as many fishers living on Stirling as possible.

Year-3: October 2011 – December 2012. Year-3 began when we commenced live-trapping fishers to be released on Stirling. Live-trapping for moving fishers for the entire re-introduction ceased in December 2011, when the last fisher of the year-3 cohort was released. From February through September 2012, we live-trapped occasionally on Stirling to try to catch particular fishers. In late October and early November 2012, we made a large effort to trap as many fishers living on Stirling as possible.

Year-4: January – December 2013. We did no live-trapping of fishers outside the Stirling study area and released no new fishers on Stirling. In late October and early November, we made a large effort to live-trap as many fishers living on Stirling as possible.

Year-5: January – December 2014. We released no new fishers on Stirling. In late October and early November, we made a large effort to live-trap as many fishers living on Stirling as possible.

Year-6: January – December 2015. We released no new fishers on Stirling. From February through September, we live-trapped occasionally on Stirling to try to catch particular fishers. In late October and early November, we made a large effort to trap as many fishers living on Stirling as possible and did some live-trapping in areas adjacent to but outside Stirling.

Year-7: January – December 2016. We released no new fishers on Stirling. From February through September, we live-trapped occasionally on Stirling to try to catch particular fishers. In late October and early November, we made a large effort to trap as many

fishers living on Stirling as possible and did some live-trapping in areas adjacent but outside Stirling.

Year-8: January – December 2017. We released no new fishers on Stirling. From February through September, we live-trapped occasionally on Stirling to try to catch particular fishers. In late October and early November, we made a large effort to live-trap as many fishers living on Stirling as possible and did some live-trapping in areas adjacent but outside Stirling. We removed telemetry collars from all fishers we caught wearing them and put no new collars on any fishers.

General Condition, Disease, and Ectoparasites

We conducted detailed physical examinations to assess the health of all fishers that we captured. We collected blood, mucosa and fecal samples to determine 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 were tested for exposure to canine distemper virus, for exposure to *Toxoplasma gondii* (the causative agent of toxoplasmosis), and for infection with canine parvovirus. Since the inception of this project, the vast majority of fishers captured on Stirling have been assessed as being in good health. We have seen no systemic physical abnormalities in either adult or 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 collected genetic information on all fishers released on and born on Stirling that we could have used to assess problems possibly related to inbreeding, had such arisen.

During physical examinations, at least 2 biologists (usually a field biologist and a wildlife veterinarian) 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, having 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. Some fishers in average condition had minor injuries or had missing or highly worn teeth, but none had

conditions that appeared to affect the fishers negatively.

Of 275 captures of fishers on Stirling through December 2017, including recaptures of reintroduced fishers and captures of fishers born-on-site, we graded none as being in poor condition. We graded 30 (11%) as showing below average condition, 145 (53%) as average, and 100 (36%) as above average or excellent. All fishers captured are listed in Appendix 1.

Through 2017, we collected ectoparasites of 4 taxa from fishers. Fleas and ticks were 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 californiensis*) varied so much. The percent of fishers that were 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 2a). Yet, only 20% of the fishers carried 2 different taxa of ectoparasites and fewer than 5% of the fishers carried all 3 taxa. Generally, when fishers did carry ectoparasites, the

infestations were light to moderate in severity. At present, we have no evidence suggests that the processes driving ectoparasite occurrence on fishers on Stirling differ from those acting elsewhere.

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). We captured no fishers infected with these trematodes on Stirling and remain optimistic that we did not transfer the parasite to Stirling.

Population Monitoring on Stirling

From 12 October through 10 November 2017, we conducted a large-scale trapping effort on Stirling to capture as many fishers as possible and to remove all remaining transmitters. We spread our trapping effort across Stirling and adjacent lands focusing on areas where fishers were known to live, on areas where fishers had been detected previously, or on areas we considered

likely to have fishers (Figure 3). To maximize efficiency, we split the study area into 2 sections (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 (27 October - 10 November) for 14 days. Logistical constraints precluded or curtailed trapping in some areas we thought may have had resident fishers. In late-November and December, we did some spot-trapping to catch fishers wearing collars but that we had not trapped during the main trapping period.

We deployed approximately 100 traps each night, totaling 2879 trap days (1472 east, 1407 west). We totaled 106 captures of 61 individual fishers (38F, 23M), yielding 3.7% trap success (number captures per 100 trap days). This capture rate was higher than in all other years of trapping (Table 1). As we experienced in previous years, capture success was greater on the east side (4.1%) than on the west (3.2%). We captured 27 new fishers (14F, 13M), 6 of which were adults (2F, 4M), and we captured 21 juveniles (12F, 9M).

During our main trapping period in 2017, we recaptured no fishers that had been reintroduced onto Stirling in 2009-2011, although we did recapture 1 reintroduced male during spot-trapping. Of fishers born on Stirling, we recaptured 1 female born in 2011, 4 females born in 2012, 4 fishers born in 2013 (3F, 1M), 6 born in

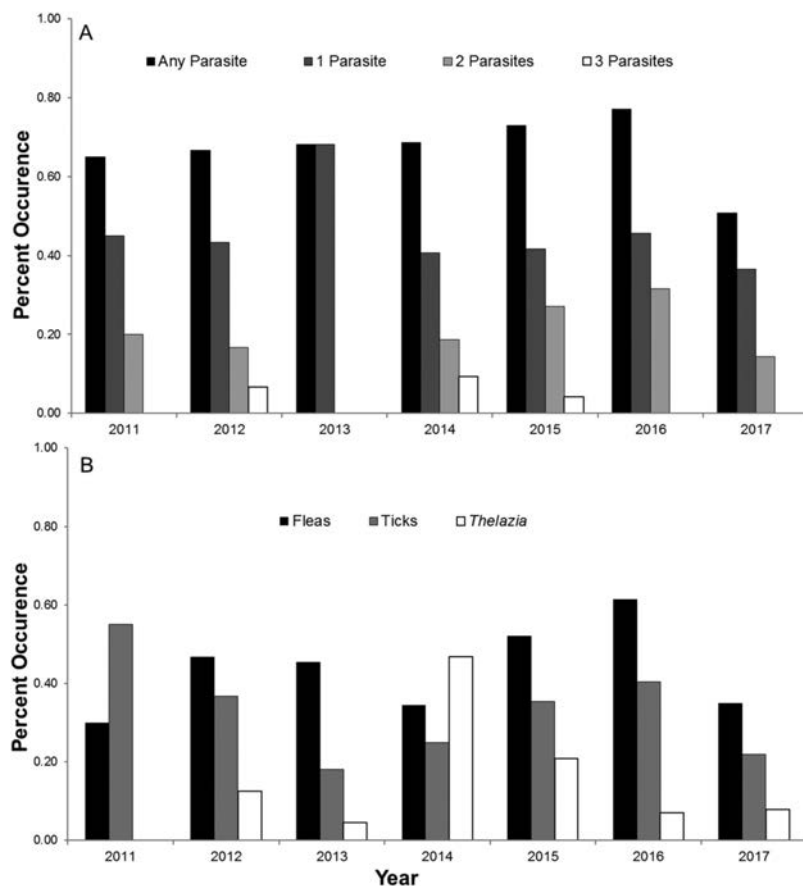


Figure 2. Occurrence of 3 taxa of ectoparasites found on fishers during 2011-2017. Data include only fishers captured on the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California. A. Percentage of fishers on Stirling by year infected with any (at least one parasite of any species), 1, 2, or 3 of the most common ectoparasites by year. B. Occurrence of parasites on fishers on Stirling by year and taxon.

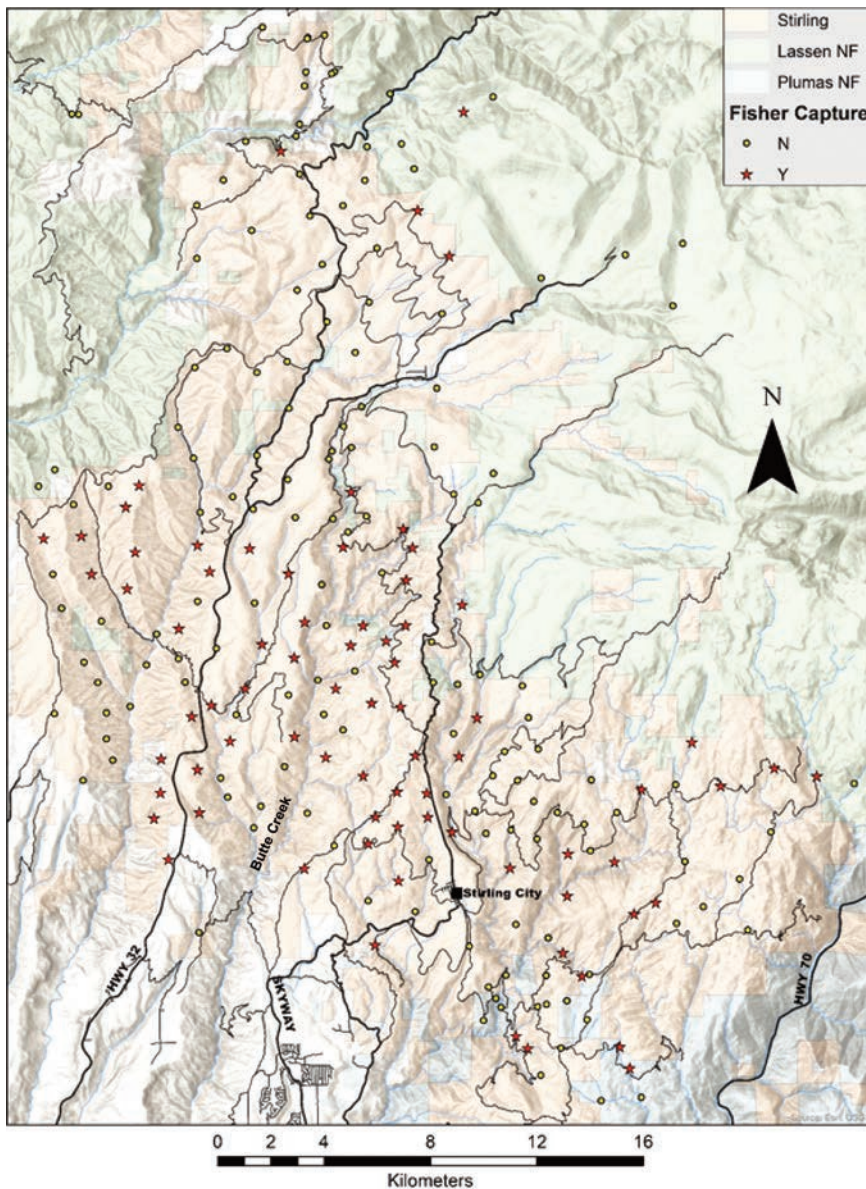


Figure 3. Map of the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade mountains of California (pink shading) and the locations of all traps set during October - November 2017. Yellow dots represent traps that did not capture a fisher and red stars represent traps that captured at least 1 fisher.

Table 1. Total number of trap nights, fisher captures, individuals, females, males, new fishers, and capture percentage during fall trapping of years 2012-2017 on the Stirling district of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California.

Year	Trap Nights	Fisher Captures	# Individuals Captured	Total Females	Total Males	New Fishers Captured	Capture %
2012	2231	43	29	17	12	14	1.93%
2013	2847	33	22	17	5	8	1.16%
2014	2786	53	32	23	9	15	1.90%
2015	2866	84	46	33	13	25	2.93%
2016	2867	93	53	34	19	26	3.24%
2017	2879	106	61	38	23	27	3.68%

2014 (4F, 2M), 13 born in 2015 (8F, 5M), and 8 born in 2016 (6F, 2M).

We removed collars from all fishers captured in 2017. By the end of 2017, 3 fishers wearing working transmitters remained on Stirling (2F, 1M).

We had 147 captures of non-target carnivores for a capture rate of 5.1% (Table 2), higher than capture rates for non-target carnivores in all other years. The capture rates for the east and west sides were similar (5.4% east *vs* 4.8% west). Grey foxes (*Urocyon cinereoargenteus*) and spotted skunks (*Spilogale gracilis*) were the most commonly captured non-target carnivores, accounting for 63% of the total.

After trapping in 2017, the age structure of the known fishers on Stirling emphasized young fishers (Figure 4). Fishers < 2 years old comprised 47% of all fishers known to be alive. Many fishers older than 2 years of age are still in the population but the young age structure suggests healthy reproduction and recruitment. The age distribution in Figure 4 is our best estimate of the true age distribution of the Stirling fisher population but is accurate only to the extent that our trapping results were representative for the population.

After trapping, the minimum known population size for the fishers on Stirling was 68, 43 females and 25 males (total fishers captured + non-captured fishers still wearing functional transmitter collars; Figure 5). For each previous year, we have retrospectively adjusted the

minimum population size when we captured fishers still living that had not been captured or wearing functional telemetry collars in that year (Figure 5). Had we been able to adjust the minimum population size for 2017 as we had in previous years, we are confident that the minimum known population size for 2017 would have exceeded our present calculation (Figure 5). Nonetheless, the female population size appears to have grown little since 2015.

Table 2. Total number of non-target carnivores captured during fall trapping of years 2012-2017 on the Stirling district of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California.

Species	2012	2013	2014	2015	2016	2017
Ringtail (<i>Bassariscus astutus</i>)	23	20	32	22	31	25
Opossum (<i>Didelphis virginianus</i>)	4	4	3	14	19	17
Bobcat (<i>Lynx rufus</i>)	0	1	0	0	0	3
Pacific Marten (<i>Martes caurina</i>)	0	0	0	0	0	7
Striped Skunk (<i>Mephitis mephitis</i>)	0	1	3	1	3	2
Raccoon (<i>Procyon lotor</i>)	3	0	2	1	1	0
Spotted Skunk (<i>Spilogale gracilis</i>)	17	36	44	50	23	44
Grey Fox (<i>Urocyon cinereoargenteus</i>)	6	13	15	54	15	49
Total	53	75	99	142	92	147

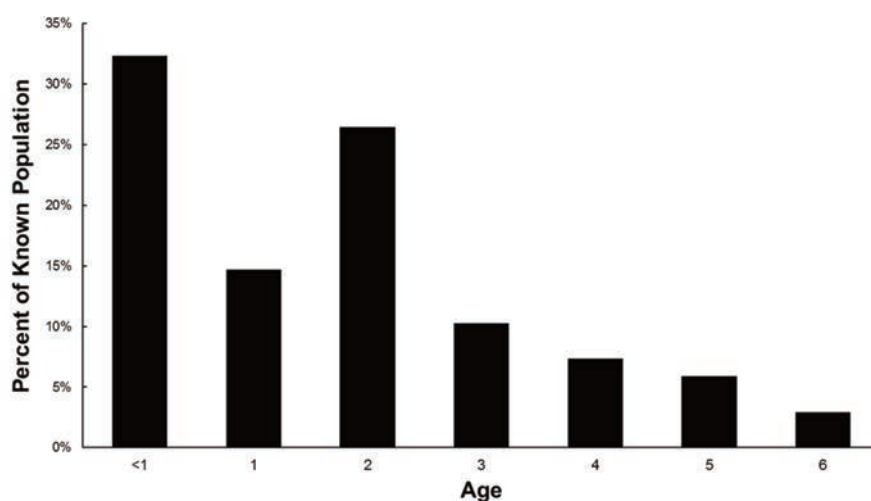


Figure 4. Percent of fishers by age distribution based on cementum annuli estimates (ages for new fishers captured in 2017 were estimated based on body size and development) on the Stirling Management Area of Sierra Pacific Industries in northern Sierra Nevada and southern Cascade Mountains of California.

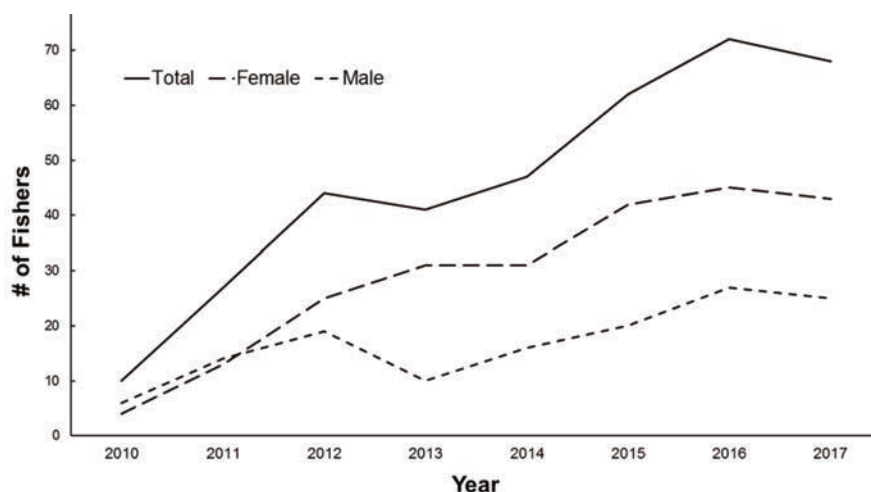


Figure 5. The minimum number of female (line with long dashes), male (line with short dashes) and total (solid line) fishers known to be alive on or near the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade Mountains of California during the autumns of 2010-2017.

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 explore and occasionally settle onto areas off Stirling (Powell et al. 2012). As of 2017, most locations of fishers have occurred within the boundaries of Stirling or very near to it (Figure 6). Similarly, most den locations have occurred on Stirling. Because our research effort concentrated on Stirling (Figure 3), these data are not representative of all fishers in the reintroduced population. We know that some fishers live on adjacent lands. Nonetheless, at least 50 fishers remained on Stirling annually from 2014 on, representing a core population.

Telemetry

In year-1, we implanted female fishers with Telonics IMP-310 very high frequency (VHF) transmitters (Mesa, Arizona) and 4 of 9 (44%) failed prematurely (< 8 months). In year-2 and beyond, we used a mix of Telonics MOD-125 and Holohil Mi-2 collars. In years 2-7, we outfitted young fishers born on Stirling with radio collars only if the fishers had necks that were unlikely to grow substantially (>2 cm) in the future.

We radio-tracked 24 females during the calendar year 2017, 15 until October when we removed collars, and 9 for part of the year before October. The females wearing transmitters maintained home ranges spread widely across Stirling and onto adjacent land.

We had a goal to obtain >150 location estimates for each female fisher per year so that we would have had >30 location estimates per season. With such

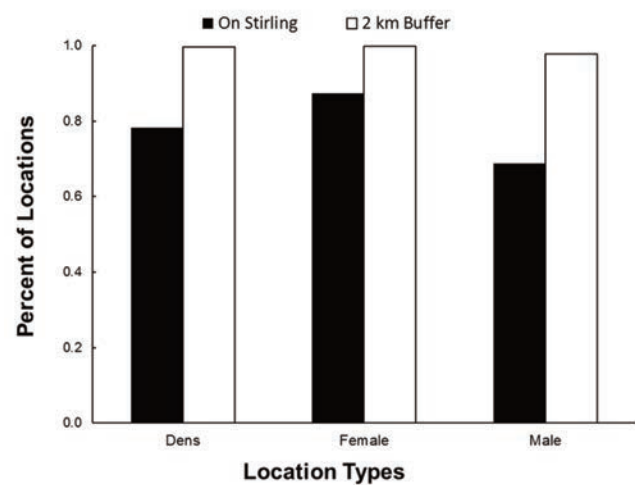


Figure 6. The percentage of all validated locations that have occurred within the boundary for the Stirling Management Area of Sierra Pacific Industries (In Stirling) or within a 2-km buffer of the boundary (2 km Buffer) for fishers’ dens, for female locations, and male locations, 2009-2017, for all translocated and Stirling-born fishers.

a sample, we would have been able to estimate seasonal home ranges for all females. Given the mountainous terrain, limited personnel, weather that limited travel and myriad other conditions that affect VHF telemetry, we rarely achieved this goal. For all females in 2017, we averaged 2.6 ± 0.8 (\pm SD) estimated locations per female per week for the weeks each female was tracked. We averaged 82 triangulations per female in 2017 (Table 3), although for fishers tracked until the fall when collars were removed (N=15), the average was 109 triangulations. For each estimated location, however, almost as many attempted locations did not meet the selective criteria we used when triangulating locations. Sometimes we did not locate females frequently because they moved beyond the perimeter of the area we searched regularly and sometimes females used parts of the study area that blocked their transmitters’ signals, leading to an unknown bias in our estimates of their movements. Female fishers do not travel as widely as do males, however, limiting the effects of bias, if it existed (Powell 1994).

We outfitted adult male fishers with Platform Terminal Transmitter 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 2017, we followed 5 males. Three of the collars on males failed in the spring, while 2 worked until trapping began in the fall. We outfitted one male fisher with a Global Positioning System (GPS) collar in 2017.

Although the batteries in the Argos collars should last over a year, many collars have failed before their projected lifetime. Many failures 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 provided location data that we simply would not have obtained using traditional VHF technology. Several released males made sojourns to places that we did not expect (e.g., Central Valley or north of California Highway 44) and where we would not have searched. Many of these males ultimately returned to the general area of their releases. We would never have documented those long-distance, temporary forays using traditional VHF 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 175 ± 191 Argos locations/male/year across all study years and 137 ± 94 locations/male in 2017 (Table 3). All Argos location estimates are classified into 1 of 6 error classes, some of which are suitable for some analyses but not others. Individual male fishers averaged 36 ± 46 locations/year from the 2 categories with smallest error and 25 ± 27 locations in 2017 (Table 3).

Table 3. Mean numbers (\pm SD, N) of estimated locations per individual fisher per year across all years of study and 2017 organized by location method. Means are for individual fishers who were followed using each particular method. 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
Female	All Years	69 \pm 82, 194	57 \pm 48, 187	7 \pm 6, 138	387 \pm 259, 3	
	2017	93 \pm 43, 24	82 \pm 39, 24	10 \pm 5, 21		
Male	All Years	304 \pm 661, 88	16 \pm 21, 21	2 \pm 1, 8	1232 \pm 1430, 11	175 \pm 191, 73
	2017	734 \pm 1610, 7	37 \pm 50, 2	2 \pm 0, 1	4380 \pm 0, 1	137 \pm 94, 5

Triangulations constitute most of our estimated locations of females and young males. For fishers tracked with VHF telemetry, approximately 88% of all estimated locations were triangulations. Another 4% of VHF locations were estimated from fixed-wing aircraft or a helicopter and 9% were “walk-ins”. Walk-ins included visual observations of fishers and locations of identifiable den or rest trees. Walk-ins also included trapping locations, mortality locations, and locations where fishers dropped collars. Additionally, we have located >250 individual rest locations, >90% of which were in trees, though some fishers rested under rocks, in stumps or in debris piles. Locating rest sites is biased towards finding sites in trees because fishers in trees broadcast strong telemetry signals predominantly in all directions. Fisher resting under rocks, in stumps or in debris piles send signals that are truncated in both strength and direction, making them more difficult to identify and to locate. Location information from cameras at dens and baited stations will also be incorporated into final analyses, but those data have yet to be incorporated into our locations database.

Understanding and estimating error for our triangulations is a critical component of future analyses. We will evaluate triangulation error in 2 ways: 1) calculating triangulation error for test collars in known locations (both moving and stationary; $n = 50$) and 2) comparing triangulations to “walk-in” locations for fishers that were located on the same day (usually within the same hour) in den and rest trees. A preliminary analysis of triangulations of walk-in locations yielded a mean error of 102 ± 132 m (\pm SD). 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 assessed true error rates for Argos locations of each error class by comparing satellite locations to known locations of males held in captivity, of collars that have been dropped (the day they are dropped is known from activity data), or of dead fishers. The mean error for Argos locations estimated across all error classes is 767 ± 1241 m. Our calculated mean error for locations in each error class are consistent with expected error predicted by the Argos service (Sauder et al. 2012; Table 4). Locations in error classes 3 and 2, predicted to have the least error, have mean error of 195 ± 247 m and 458 ± 460 m (\pm SD). Our 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 to eliminate locations that are highly inaccurate).

We attempted modestly to monitor fishers during all times of day and night to ensure that our data are not biased by time of day. VHF transmitters were difficult to locate at night, particularly in the winter when temperature, weather and road conditions hindered access to the study area. Thus, the vast majority of VHF telemetry locations were collected during daylight hours (8 am to 4 pm; Figure 7).

Argos collars were limited to being located only during a 6-hr time block each day so we programmed collars for different male fishers for different time blocks. Therefore, the distribution of locations of fishers wearing Argos collars was relatively even across all times of day. We programmed GPS collars to locate themselves throughout each day, leading to an even distributions of locations.

We collected enough location data to estimate annual home ranges for most fishers. Thirty locations

Table 4. Classes for Argos locations of male fishers, error predicted by Argos services for locations in those classes, our mean observed error, standard deviation, minimum error observed, the maximum error observed and the total number of location estimates for each location class across 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

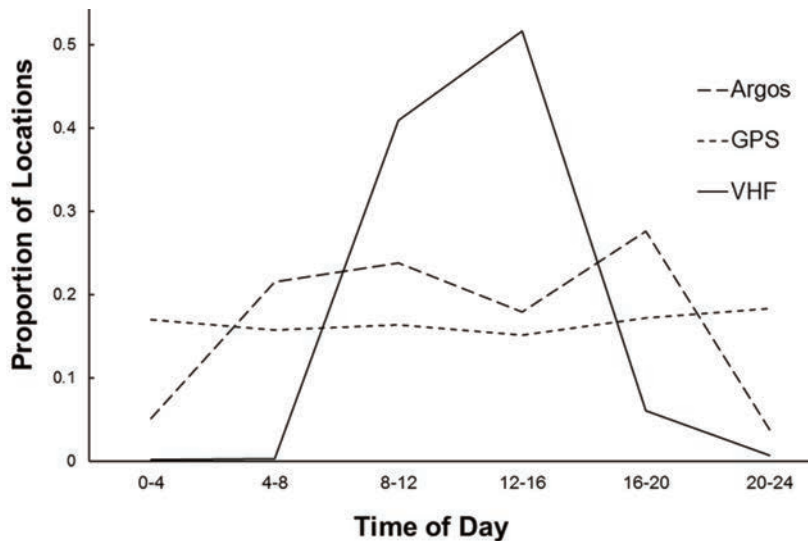


Figure 7. Percent of estimated locations of fishers obtained via Argos, GPS and VHF telemetry at different times of day across all years of study (2009-2017) on the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada and Southern Cascade Mountains of northern California.

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, Noel 1993, Seaman et al. 1999, Seaman and Powell 1996). We have more than 100 estimated locations per year for many fishers.

Home Ranges

We define an animal's home range to be that part of the landscape in which it lives and for which it updates its cognitive map of the landscape (Powell and Mitchell 2012). For this report, 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" 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 statistical error within a distribution of location estimates (Seaman and Powell 1996, Powell 2000). Choosing "h" to minimize such error, 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. Ideally, the choice of "h" should relate directly to aspects of the biology, to characteristics of the technology used to collect data, to management goals for the animals being

studied, or some combination of these. For fishers, different aspects of fisher biology suggest different values for "h". For our fishers, $h=750$ m appears to estimate reasonably well the probability of where researchers were able to find a given fisher using telemetry. The values we obtained for $h=750$ are similar to those reported by Lofroth et al. (2010). Average daily movements of fishers suggest that $h=1500$ m should estimate where a fisher can travel over the coming day. Average distances across distributions of location estimates suggest that $h=1000$ m will estimate the overall range of space a fisher uses but not the fisher's small-scale preferences. Values of "h" tailored to match the estimated error for each location estimate should provide the best estimates of fishers' habitat preferences.

Table 5 shows mean estimates for areas of 95% utilization distributions for 2010-2017 using $h = 750$, 1000 and 1500.

Table 5 shows that males have larger home ranges than do females and that larger values for "h" lead to

Table 5. Mean areas (\pm SD) for 95% fixed kernel utilization distributions (UD) of fishers followed with telemetry for ≥ 6 months on Stirling in 2010-2017 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
	2016	12 \pm 4, 19	25 \pm 2, 3
	2017	13 \pm 4, 19	67 \pm 4, 4
	2010	22 \pm 8, 6	97 \pm 7, 3
	2011	37 \pm 8, 7	143 \pm 24, 3
1000	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
	2016	14 \pm 4, 19	33 \pm 5, 3
	2017	15 \pm 5, 19	90 \pm 55, 4
	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
1500	2014	25 \pm 7, 13	100 \pm 53, 3
	2015	23 \pm 4, 19	88 \pm 27, 3
	2016	19 \pm 5, 19	49 \pm 12, 3
	2017	20 \pm 6, 19	132 \pm 70, 4

larger utilization distributions. Daily tracking of fishers suggested that females established home ranges primarily within Stirling. Some females travelled to adjacent Forest Service or private lands and one traveled north 22 km onto the Lassen Management Area of Sierra Pacific; she died, however, within 3 months of release. Additionally, female fishers have denned in trees on both the Lassen and the 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 the Plumas National Forests.

Male fishers have also established home ranges across most of Stirling. Since males have larger home ranges than were females and disperse more widely, they were located on adjacent lands more often than females. Several males established home ranges off Stirling and up to 40 km from release sites for reintroduced individuals. We stopped tracking these fishers because their home ranges are outside the area that we trapped each year. If male fishers with home ranges beyond Stirling have movements and survival similar to those of males that we tracked on Stirling, untracked males may have a substantial presence on Forest Service lands, private timber lands and Sierra Pacific holdings adjacent to or near Stirling. We did not track most juvenile males born on Stirling and, consequently, we do not know how far away those males dispersed to establish home ranges.

The areas for 95% utilization distributions in Table 5 weight all location estimates equally and, therefore, give insights into where fishers spend time. One can calculate utilization distributions based on currencies other than time. In the 2015 Annual Report, we presented examples of home ranges built using energy as the currency and showing where fishers expended the most energy.

Mortality and Survival

Through December 2017, we confirmed the deaths of 38 fishers (27 F, 10 M, 1 unknown; Table 6). One female slated for release died in captivity in December 2009. During 2010, transmitter failure limited our ability to document mortality yet we still documented the deaths of 3 females. Since 2011, however, we have tracked most females continuously during full years or until death. Trapping in autumn 2017 allowed us to capture fishers whose collars had failed in previous years as well as fishers that had been captured in previous years but had not been collared. We used data from telemetry and trapping to examine patterns and rates of

survival for reintroduced and Stirling born fishers for December 2009 through December 2017.

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

Notable causes of death for fishers found as complete carcasses included drowning in a water tank ($n = 3$), systemic disease (vasculitis, hepatitis, hypertension and pneumonia) of unknown origin ($n = 2$), killed by bobcat (1), anticoagulant rodenticide (1), road kill (1)

Table 6. Fisher mortalities on the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada and the Southern Cascades Mountains. Fishers with birth years before 2011 were captured in northern California and released onto Stirling. Those born 2011 or later were born on Stirling.

ID	PIT5	SEX	Birth Year	1 st Capture	Mortality Date	Mortality Cause
20082	D00B0	F	2004	2009 12 08	2010 07 17	Drowning (water tank), no ars
10054	EB687	F	2005	2009 12 03	2009 12 09	Unknown, in box, brodifacoum detect
20111	93B5A	F	2005	2010 11 02	2014 07 07	Suspect raptor predation
20096	F65B6	F	2007	2010 01 21	2010 06 29	Bobcat predation or scavenge
20117	18FFF	F	2007	2010 11 10	2011 08 25	Unknown
10132	1E003	F	2007	2011 01 23	2015 12 14	Pending
20085	F8B8D	F	2007	2010 01 05	2017 08 27	Predation, rodenticide trace
10079	F6280	F	2008	2009 12 02	2010 06 26	Bobcat predation or scavenge
10131	21FB6	F	2008	2011 01 22	2013 06 20	Traumatic injury head, neck
20178	252FD	F	2008	2011 11 29	2011 12 10	Multiple diseases, brodifacoum 0.17 ppb
20175	1F111	F	2009	2011 11 29	2012 06 02	Bobcat predation
10124	182F4	F	2009	2011 01 08	2012 07 24	Unknown, possible felid
20159	2189C	F	2009	2011 11 04	2012 10 12	Unknown
20155	23955	F	2011	2011 10 26	2014 09 08	Drowned
20148	21392	F	2011	2011 10 22	2016 02 10	Suspect rodenticide, brodifacoum trace
20199	614DF	F	2011	2012 10 30	2016 12 17	Predation
20190	20950	F	2011	2012 01 25	2017 04 05	Suspect disease, multiple symptoms
20192	1EA8D	F	2012	2012 10 24	2015 05 10	Unknown, possible predation/scavenge
20234	51812	F	2013	2014 11 09	2016 04 12	Entrapment, brodifacoum trace
20212	69940	F	2013	2013 10 15	2016 07 26	Unknown
20210	16026	F	2013	2013 10 14	2016 09 21	Herniation, obstructed colon, brodifacoum
20220	35006	F	2014	2014 10 14	2016 05 27	Predation, brodifacoum trace
20266	27E2F	F	2015	2015 10 27	2015 12 08	Unknown
20246	136BC	F	2015	2015 10 13	2016 08 18	Unknown
20274	114C0	F	2015	2016 01 25	2017 05 12	Unknown
20307	6AA7A	F	2016	2016 10 26	2016 11 29	Unknown
20166	1E03E	F		2011 11 11	2012 02 27	Systemic disease, unknown pathogen
20090	596E2	M	2004	2010 01 08	2013 09 18	Possible avian predation or scavenge
10020	18C3E	M	2006	2010 12 09	2011 03 12	Roadkill, brodifacoum, bromadiolone
20093	F0858	M	2006	2010 01 11	2013 06 19	Predation/scavenge
20200	1E613	M	2006	2012 11 06	2014 03 26	Predation/scavenge
20116	18CC8	M	2006	2010 11 10	2015 04 24	Unknown
10060	18308	M	2007	2010 01 07	2012 05 15	Unknown
20150	24101	M	2011	2011 10 23	2013 04 08	Unknown, no rodenticides in muscle
20191	24B09	M	2012	2012 10 22	2014 01 27	Unknown
20211	70926	M	2013	2013 10 14	2016 04 04	Possible predation/scavenge
20218	XXXX2	M	2014	2014 09 08	2014 09 08	Drowned
20217	XXXX1	Unk	2014	2014 07 07	2014 07 07	Suspect raptor predation

(Woods and Wengert, unpublished). Gross and histologic findings suggestive of hypoxia, hyperthermia and suffocation were documented for the fisher that had been a candidate for reintroduction but that died in captivity. The cause of death, however, could not be definitively confirmed (Munson, unpublished). Many carcasses had potential evidence of predation but, due

to carcass degradation and scavenging, many carcasses were not intact and, thus, predation could be documented conclusively. Some fisher carcasses in poor condition yielded bobcat DNA, suggesting that these fishers had been killed or scavenged by bobcats. Predation forensics on other fisher carcasses suggested predation (Wengert, unpublished). In general, the

causes of mortality observed are consistent those found by other studies in California (M. Gabriel and G.M. Wengert, unpublished data).

Anticoagulant rodenticide compounds were documented in liver tissue of fishers. Reintroduced fishers could have been exposed to rodenticides where they had been originally captured or on Stirling after they were released. They and fishers born on Stirling could have been exposed on Stirling. Twelve of the 38 fisher carcasses necropsied (32%) had liver tissue samples tested for exposure to anticoagulant rodenticides and 11 (92%) of these samples tested positive for one or more anticoagulant rodenticide types. Five of the 6 reintroduced fishers tested had been exposed to brodifacoum, a second-generation anticoagulant rodenticide with a long tissue half-life. One reintroduced fisher had been exposed to bromadiolone and another had been exposed to both bromadiolone and to difethialone (both also second-generation anticoagulant rodenticides). All 6 Stirling-born fishers tested had been exposed to a trace amount of brodifacoum. One of these fishers had also been exposed to a trace amount of chlorphacinone, a first-generation anticoagulant rodenticide. Only reintroduced fishers had detectable anticoagulant rodenticide levels above trace amounts, which may suggest a slightly lower risk of exposure on Stirling or may have been caused by most fishers tested that had been born on Stirling were young. Young fishers have low risk of anticoagulant rodenticide accumulation simply because they lived short lives. The overall significance or potential impacts of sub lethal exposure to anticoagulants for fishers and other wildlife is largely unknown but widespread exposure and cases of direct mortality due to anticoagulant toxicity in fishers and other wildlife species has raised significant conservation concerns (Gabriel et. al 2012, 2015). In June 2014, the California Department of Pesticide Regulation restricted the use of second-generation anticoagulant rodenticide products containing brodifacoum, bromadiolone, difenacoum, and difethialone to certified pesticide applicators. Thus, these compounds are no longer available over the counter (California Department of Pesticide Regulation 2014). This regulation change has the potential to reduce non-target wildlife exposure risk from household use, especially in urban and suburban areas, but may not have any impact at reducing use of anticoagulant rodenticides at illegal marijuana cultivation sites, thought to be the most likely source of exposure for fishers (Gabriel 2012, Thompson 2013).

Survival

We analyzed monthly survival for fishers wearing telemetry collars using “known fates” analyses within program MARK (White and Burnham 1999). Known fates analyses account for each time 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 not used to estimate survival for that time. The number of fishers we did not collar has increased throughout the study as the total numbers of fishers captured and population size have increased. We used Akaike’s Information Criterion corrected for small sample size (AICc) to rank hypotheses that could describe best the pattern of mortalities and survival that we documented (Table 7).

Because patterns in survival have not changed throughout the study, we have focused this year on evaluating potential changes in survival through time due to habitat. As in previous reports, we modelled survival first as constant through time, with affects due only to sex and to sex×age class, where age classes were defined as juveniles (<1 year old), yearlings (1 < age <2) and adults (≥ 2 years old). Additionally, we modelled survival as changing by month×year, treating year first as a class variable, then as a continuous variable, and finally with male and female survival potentially differing across study years, again with year as a class variable. We also incorporated a reintroduction effect to evaluate if reintroduced fishers survived differently than fishers born on Stirling. In previous reports, survival was best described by models that incorporated within year effects associated with the seasons of reproduction. This effect was consistent across all years but we have not modelled it here to be able to evaluate metrics associated with habitat.

A critical aspect of our research on Stirling is to understand how habitat and forest management practices affect fisher survival and reproduction. For a population that is growing and has not reached carrying capacity (K), habitat does not prevent population growth but might slow population growth. Thus, our null hypothesis was that habitat within a fisher’s home range does not affect its probability of survival, with the alternate hypothesis that survival increases with increases in habitat quality. To test this hypothesis, we first estimated the utilization distributions of fishers in all years. We defined a year as beginning 1 October (when kits are independent of their mothers; Coulter 1970) and ending 30 September. We estimated all utilization distributions using all locations taken from telemetry, walk ins, captures and Argos collars (error

class was 1, 2 or 3; Table 4). We used a kernel density estimator with a fixed kernel (Silverman's k2) and a smoothing parameter (bandwidth) of 500 m, except for Argos locations, for which we set bandwidth to 500 m or to the error radius associated with the error class (class 1 > 1500 m, class 2 250-1500 m, and class 3 < 250 m), which ever was larger. From our utilization distributions, we generated isopleths that bounded 25, 50, 75, 90, and 95% of the probability of use. We estimated the mean and the standard deviation of the habitat quality occurring in those isopleths for each fisher and year. We used Thomasma's habitat suitability index (Thomasma et al. 1996) to quantify habitat quality because, in previous studies, this model explained fisher habitat selection and use of space better than other habitat models (Facka et al 2016, Appendix 2). Fisher utilization distributions included areas not on Stirling and; therefore, we used the Gradient Nearest Neighbor data set (<https://lemma.forestry.oregonstate.edu/about>) to generate a version of Thomasma's index that extended beyond the boundaries of Stirling. Habitat quality on Stirling also changed by year but the Gradient Nearest Neighbor dataset was static. Consequently, we generated 6 Thomasma index models that covered Stirling only, using forest inventory data provided by Sierra Pacific for each year from 2010 to 2016. We estimated the means and standard deviations of habitat quality from the single Gradient Nearest Neighbor based model for each isopleth and across years for fishers. In contrast, we used the Thomasma habitat model from the most appropriate year to estimate mean and standard deviation of isopleths of the utilization distributions. Because we had 5 isopleths, 2 habitat models and 4 metrics for each model (40 potential covariates to survival) we limited our analysis of habitat quality on survival to only the 50% isopleth for each individual fisher. We also examined mean habitat quality across locations for each fisher and tested for effects from the proportion of a fisher's home range that had been logged within the last 10 years (Habitat Relationships section).

We compared models of survival that incorporated habitat quality to a model with constant survival. Although we know that within year models generally describe survival best, we considered a null model sufficient to test how habitat covariates related to survival. A constant mode of survival provides a monthly estimate of 0.982 (0.97-0.98) but was ranked as the 4th ranked model. This monthly estimate translates to a yearly estimate of survival of 0.813 (0.74-0.86). This point estimate was up slightly from 2016 (0.79). We found that age was the best supported model describing our data (Table 7). Age had a negative relationship to survival rate ($\beta = -0.418 [-0.86 - 0.03]$), indicating that older fishers survived at a lower rate than juveniles. In fact, young fishers seldom died on our study, whereas old fishers must eventually die. We suspect that young fishers that were never captured, some of whom may have moved off Stirling, had higher mortality rates than resident adults. Reintroduction was the model ranked second ($\Delta AICc = 1.73$), indicating that the reintroduced fishers had lower survival (monthly rate = 0.977 [0.96-0.99]) than fishers born on Stirling (0.98 [0.978-0.99]). The model that ranked 3rd

Table 7. Comparison of 17 models of survival used in a known fates analysis in program MARK based on monthly fates of reintroduced fishers and their offspring on the Stirling Management Area of Sierra Pacific Industries, in the northern Sierra Nevada and the southern Cascade Mountains of California, 2009-017. "Habitat (GNN)" refers to Thomasma's index of habitat quality applied within fishers' home ranges calculated using the Gradient Nearest Neighbor data set, which is static for the entire study period. "Habitat (SPI)" refers to Thomasma's index within fishers' home ranges calculated for each year using data supplied by Sierra Pacific Industries. Year was a class variable excepts where notes as continuous.

Model	AICc	$\Delta AICc$	w	Likelihood	K	Deviance
Age	344.12	0.00	0.28	1.00	2	340.11
Translocation	345.86	1.74	0.12	0.42	2	341.85
Age + Habitat (SPI)	345.96	1.84	0.11	0.40	3	339.95
Null	346.08	1.96	0.10	0.38	1	344.07
Habitat (GNN)	346.98	2.86	0.07	0.24	2	342.97
Sex + Reintroduction	347.34	3.22	0.06	0.20	4	339.32
Age+ Reintroduction	347.44	3.32	0.05	0.19	4	339.42
Proportion Logged	347.68	3.56	0.05	0.17	2	343.67
Age + Sex	247.55	3.43	0.04	0.17	4	339.52
Habitat (SPI)	348.00	3.88	0.04	0.14	2	343.99
Sex	348.03	3.92	0.04	0.14	2	344.03
Year (continuous)	348.04	3.92	0.04	0.14	2	344.03
Habitat (SPI)	348.08	3.96	0.04	0.14	2	344.07
Sex + Proportion Logged	350.99	6.87	0.01	0.03	4	342.97
Year	356.00	11.88	0.00	0.00	8	339.92
Reintroduction + Year	358.98	14.86	0.00	0.00	15	328.74
Sex \times Year	362.00	17.88	0.00	0.00	16	329.72
Month \times Year	464.55	120.43	0.00	0.00	97	260.49

Table 8. The number of adult 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 2010-2017 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	2016	2017	Total
Females Tracked	8	9	10	11	7	18	19	20	102
Females Denned	5	7	9	9	6	13	17	17	83
% Denned	63%	78%	90%	82%	86%	72%	89%	85%	81%
Min # Kits	4	12	13	17	8	21	30	32	137
Kits in Fall	1	10	15	15	17	26	27	22	133
Kits Died Den	2	3	3	2	2	1	6	2	21
Litter Size	1 \pm 0	2.0 \pm 0.5	1.9 \pm 0.5	1.9 \pm 0.2	1.6 \pm 0.8	1.9 \pm 0.4	2.0 \pm 0.3	2.0 \pm 0.2	1.9 \pm 0.1
Kits/Female	0.5	1.3	1.3	1.5	1.1	1.2	1.6	1.6	1.3
Natal Dens	5	7	8	9	4	13	16	17	82
Maternal Dens	12	11	16	16	1	30	42	55	183

(Δ AICc = 1.84) incorporated both age and the mean habitat value at the locations where fishers were located. Survival related positively to mean habitat quality (β = 0.38 [-1.48-2.26]) but the slope can not be distinguished from 0 (= no effect).

No other model that incorporated effects from home range habitat quality or the proportion of home ranges that were recently logged described our data well at all (Table 7). Generally, across the study, survival depends most strongly on sex, age and reproductive status. Our estimates of annual survival varied very little from year to year and models including year as a continuous variable rank very low. Thus, through 2017, logging on Stirling appears not to have affected survival of fishers in the new population. The causes of fishers' deaths were myriad, often resulting from chance (e.g., water tanks; Table 6). The fisher population appeared still to be growing as of December 2017 and not yet to have reached carrying capacity. Consequently, we can not reject our null hypothesis that, through 2017, habitat has not limited growth of the fisher population on Stirling.

Reproduction

Fishers on Stirling have produced kits in all 8 springs since the first releases (Table 8). Our daily telemetry searches for female fishers provided a good knowledge base of their daily movements. We suspected that an adult female (\geq 2 years old) had denned and given birth to kits when we located her via telemetry repeatedly in a localized area, especially in the same tree, on successive occasions in late March and early April. We then verified denning by monitoring a potential natal den via telemetry and remote cameras. Figure 8 shows the locations of dens for all years of this study through 2017.

The mean denning rate for the entire study was 81%

adult females denning per year, with a yearly low of 63% and high of 90% (Table 8). In other studies, females have sometimes aborted or lost litters after they started denning (Matthews et al. 2013a). We never documented the loss of an entire litter except when mothers died while denning. We do not know the minimum age at which fisher kits can survive without their mothers but assumed that all offspring died if their mothers died before 1 September. Over all years of the study, we documented 12 females that died while they were denning (or a mean of 1.5/yr). At a minimum, we estimate that these deaths of mothers represented the deaths of 21 kits (2.6 per year; Table 8). Thus, 75% of denning females had at least 1 kit survive.

Fishers on Stirling denned and gave birth at similar times of the year as fishers elsewhere (Powell 1993). We found natal dens (dens in which females give birth) most often during the final 2 weeks in March or the first week in April (mean = week 13.3; Figure 9A), with the earliest natal den found on 17 March and the last found on 19 April. Because a female must have localized movements before we even looked for a den tree, our identification of natal den trees came a few to several days, maybe even weeks, after a female began restricting movements and, thus, our denning dates are biased late. Figure 9B shows that during the first 2 years of this research (2010, 2011), we found dens later than during the following years. We suspect that this trend reflects our inexperience at finding den trees during early years combined with difficult searching conditions during those early years (e.g., high snow fall, roads blocked by fallen trees). Alternately, all females in 2010 and most in 2011 had recently been released on our study site, possibly causing the females that had been moved to give birth late. One of our early findings from this project was that releasing females after 1 January

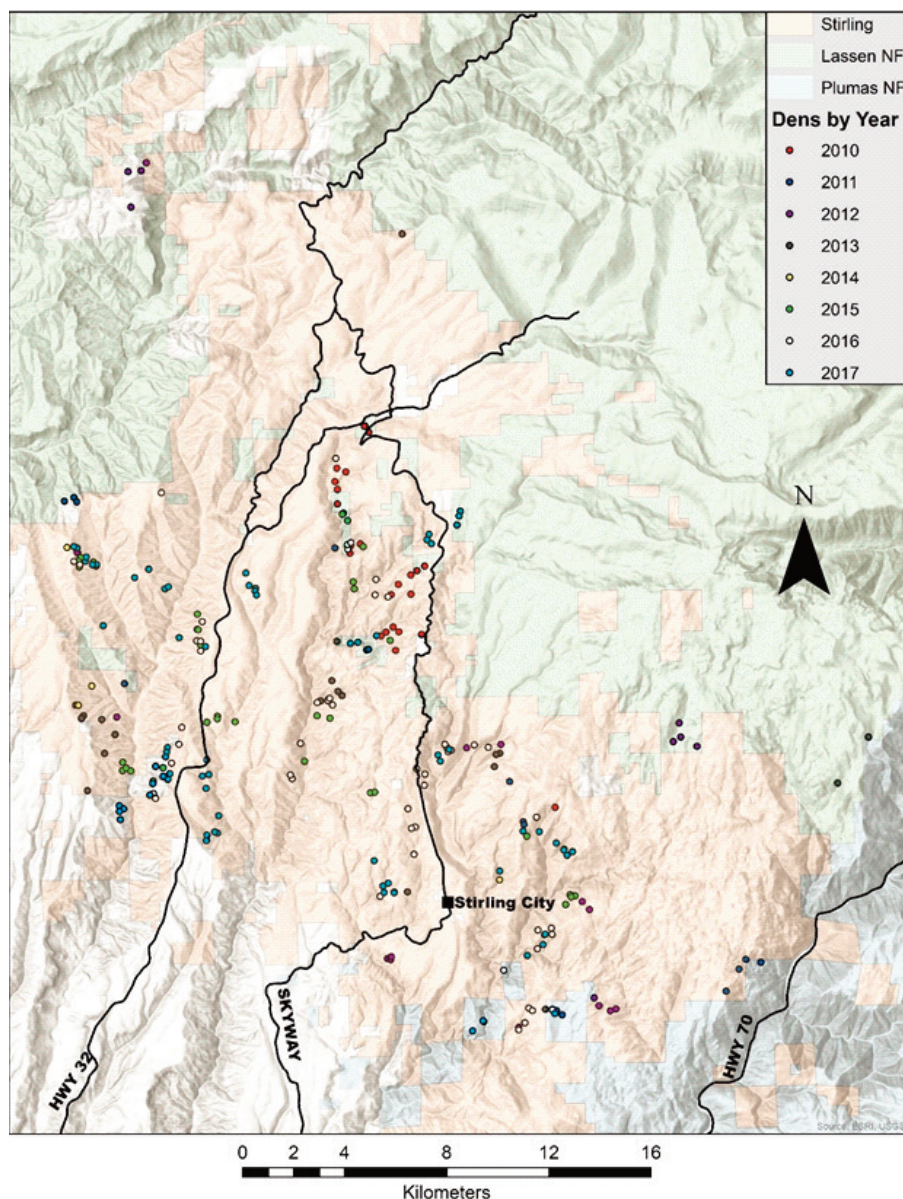


Figure 8. Locations of female fishers' dens located during the springs and summers of 2010-2017 on or near the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade Mountains of northern California.

decreased their probabilities of giving birth during the first spring following release (Facka et al. 2016). Being moved may also have caused females to delay births.

Females moved their kits from natal dens to maternal dens (dens used for raising kits after using natal dens), and to subsequent maternal dens, throughout each spring and summer. The timing of those moves showed no discernable pattern other than that we documented fewer moves after late summer when we monitored dens less intensively (Figure 9A). Some females never moved their kits, which is not shown in Figure 9A. Though we attempted to locate females and their kits throughout the summer, we considered the denning season to have been concluded, effectively, by the end of

June (week 27), after which time most females moved kits often to rest sites that they used while foraging.

Of 20 adult females tracked in 2017, 17 exhibited behavior consistent with denning. Throughout the spring and summer, we documented 32 kits from 16 denning females (2.0 kits per female). Through 2017 and for Table 8, all reproduction metrics have been based only on females we tracked via telemetry. We collected additional data on birth rate each fall by examining the teats of captured females for signs of lactation (Matthews et al. 2013b). In addition to females confirmed via telemetry to have denned and to have had kits, we captured 3 non-collared females that appeared to have raised kits in 2017. For all adult females captured in autumn 2017, 70% had nipple sizes indicative of having lactated earlier in the year and nipple sizes of 43% of adult females not tracked with telemetry indicated lactation (3 of 7). Based on these metrics, we estimate that a minimum of 19 project females gave birth to kits in spring 2017 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 most adult females gave birth on Stirling in 2017.

We documented females denning across Stirling, on other private lands and on national forest lands (Figure 8). Of 266 natal and maternal den trees that we found in 2010-2017, black oaks (*Quercus kelloggii*) were most common for both natal and maternal dens (50%; Table 9). Female fishers used incense cedars (*Calocedrus decurrens*) second most commonly (16%), followed by Douglas firs (*Pseudotsuga menziesii*; 11%). Female fishers used live trees (64 of 82 dens) most often as natal dens but, later in the denning season as kits began to travel with their mothers, used snags, hollow logs and piles of debris more commonly (97 live trees and 87 snags, logs and debris). In 2010-2012, Sierra Pacific committed resources to collect data on vegetative and topographic

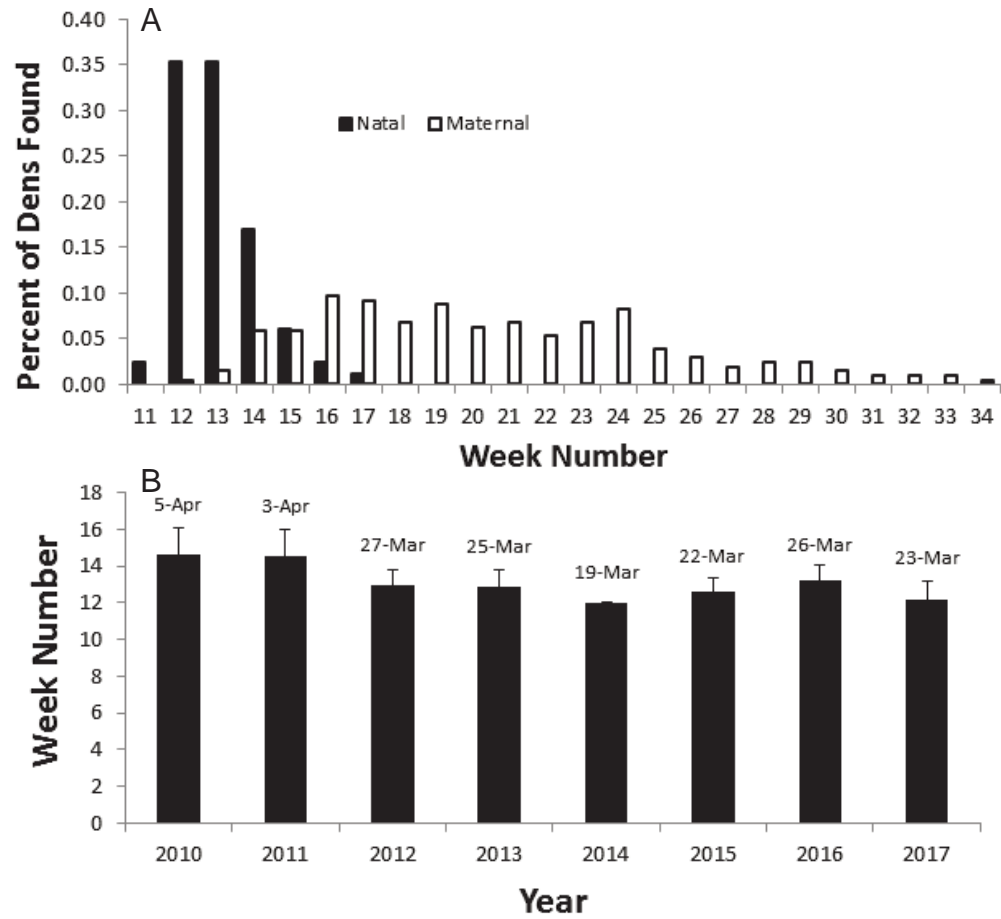


Figure 9. (A) The percent of all natal (black bars) and maternal (open bars) fisher dens found by week in 2010-2017 and (B) The mean week number ± 1 SD (mean date given above bar), by year, that natal dens were found on and near the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade Mountains of California.

characteristics within 90 m of den sites. Future analyses will examine patterns of female denning and movements relative to topography weather, time of year, predators and other factors that might influence female decisions to establish and move dens.

We evaluated variables that could have affected female reproduction during our study. We first tested the hypothesis that reintroduced females produced fewer kits than females born on Stirling. Second, we tested the hypothesis that reproductive output increased with age

Table 9. Numbers of den trees by species for natal and maternal dens from 2010 to 2017, 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>Quescus kelloggii</i>)	41	3	0	67	21	1	133
Douglas Fir (<i>Pseudotsuga menziesii</i>)	4	1	0	7	16	0	28
Incense Cedar (<i>Calocedrus decurrens</i>)	6	7	0	5	25	0	43
Canyon live oak (<i>Quescus chrysolepis</i>)	3	0	0	7	0	0	10
Ponderosa Pine (<i>Pinus ponderosa</i>)	1	1	0	1	1	1	5
Sugar Pine (<i>Pinus lambertiana</i>)	1	1	0	2	2	0	6
Tanoak (<i>Notholithocarpus densiflorus</i>)	6	1	0	5	0	2	14
White Fir (<i>Abies concolor</i>)	2	0	0	3	4	0	9
Unidentified Conifer	0	4	0	0	12	1	17
Total	64	18	0	97	82	5	266

Table 10. Comparison of 10 models with variables hypothesized of affect reproductive output by female fishers on the Stirling Management Area of Sierra Pacific Industries in the northern Sierra Nevada and southern Cascade Mountains of California, 2010-2017.

Model	AICc	ΔAICc	Likelihood	<i>w</i>
Proportion Logged	285.1	0	1.00	0.8
Year	290.8	5.7	0.06	0
Age	292.6	7.6	0.02	0
Habitat Quality	292.7	7.7	0.02	0
Proportion HR + Hardwoods	292.8	7.7	0.02	0
Translocation	293.0	8.0	0.02	0
Age + Year + Age × Year	293.7	8.6	0.01	0
Age + Habitat Quality	294.4	9.3	0.01	0
Translocation + Age	294.5	9.5	0.01	0
HabitatQuality + Translocation	294.8	9.8	0.01	0

of females old enough to give birth. Next, we tested the hypothesis that reproductive output decreased with an increasing proportion of a female's home range that had been logged within the preceding 10 years. We tested the hypothesis that reproductive output increased with the mean habitat quality at a female's locations during the preceding year. Because hardwood trees with cavities are commonly used for denning (Table 9), we tested whether reproductive output increased with the proportion of hardwoods found within a female's home range. Finally, we tested whether reproductive output of females increased through time. We used a generalized linear mixed model using a Poisson distribution and included individual females as random effects. We used the maximum litter size recorded for each female fisher as the dependent variable, assigning a value of 0 to females who did not reproduce.

The model including only the proportion of a home range logged within the previous 10 years was the only model that described our data modestly well (Table 10). The reproductive output of female fishers decreased as the proportions of home ranges recently logged increased ($\beta = -0.89$ [-2.8 - 0.99]). Nonetheless, the 95% confidence intervals for the slope of this relationship bounded zero (as was the case for all other models.) Thus, we accept the null hypothesis that no variables used in the models that we ranked affected reproductive output significantly.

Although recent logging within a female's home range on Stirling appears not to have reduced reproductive output through 2017, our analyses suggest that the potential may exist for the future. As with our analyses of survival, that our population is still growing and has not yet reached carrying capacity limits our ability to state whether logging has the potential to

affect carrying capacity of fishers on Stirling.

Habitat Relationships

For our final habitat analyses we used Thomasma's model to quantify habitat quality for fishers (Thomasma et al. 1994), which we have described in previous reports. We use Sierra Pacific's forest inventory data averaged across forests stands. We previously demonstrated correlations between the habitat quality values of Sierra Pacific's inventory data and the publicly available Gradient Nearest Neighbor data set downloadable at

<http://lemma.forestry.oregonstate.edu/data; Ohmann et al. 2011>). We first tested the hypothesis that the fisher population selected habitats quantified as being of high quality. We compared habitat quality at fishers' telemetry locations to the habitat values at an equal number of locations randomly placed across Stirling. We quantified habitat use vs habitat available by using the modified Ivlev resource selection function, which varies from -1 to 1:

$$\text{Modified Ivlev} = \frac{2 \times (\text{used} - \text{available})}{(1 + \text{used} + \text{available})},$$

where "used" represents the locations where we observed fishers and "available" represents the random locations. Across all years, the mean available habitat quality across Stirling was 0.44 (95% CI = 0.42 - 0.47). The mean habitat value used by fishers was 0.62 (95% CI = 0.60 - 0.65). The resource selection function regressed positively on the predicted habitat quality values from the Thomasma model ($\beta = 0.42$ [0.38 - 0.46], $p < 0.001$, $r^2 = 0.84$). Thus, fishers collectively selected sites that had high habitat quality and avoided areas that had low habitat quality (Figures 10, 11).

We then tested whether individual fishers selected areas quantified as being of high quality within their home ranges. We again used the modified Ivlev resource selection function but, in this analysis, we used the mean habitat value for all locations for a single fisher as the "used" term and the mean habitat value found across the 95% home range as the "available" term. Thus, each fisher in each year had a resource selection value and we used these as independent replicates to evaluate differences in resource selection across sexes and through time.

Habitat quality within female fishers' home ranges

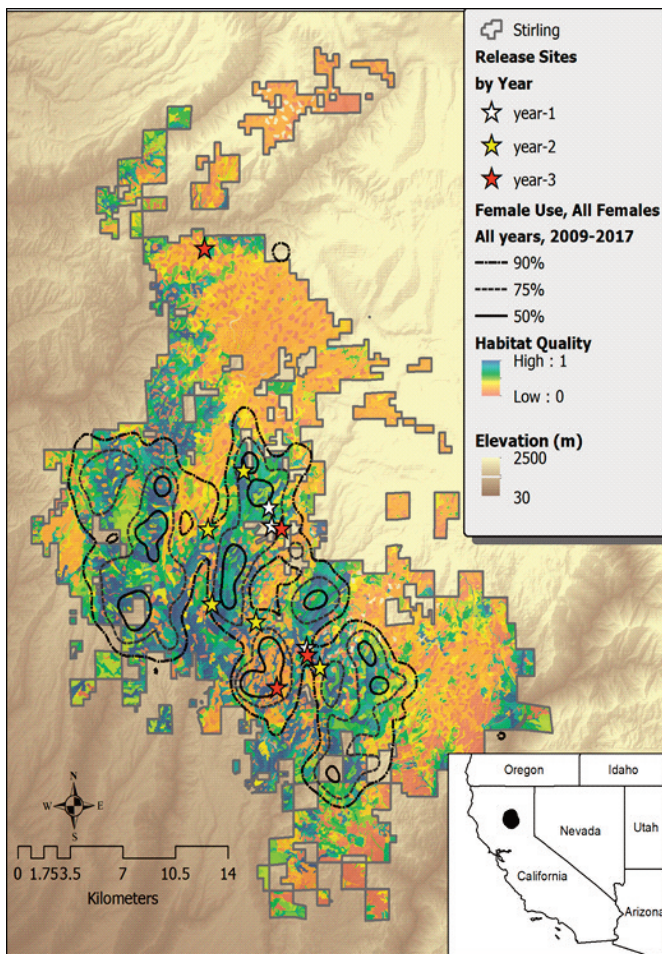


Figure 10. The 90%, 75% and 50% density contour lines (black lines) for all locations of female fishers combined from 2009 through 2017 overlaid on the mean of the habitat quality estimates for 2010 through 2017 using Thomasama's habitat suitability index. Shaded regions outside of the Stirling boundary represent topographic relief and colored stars are the release locations of all translocated fishers.

averaged 0.69 (95% CI = 0.66 - 0.71). Males had slightly lower mean habitat quality values within their home ranges of 0.56 (95% CI = 0.51 - 0.61) and had home ranges with mean habitat qualities of 0.49 (95% CI = 0.45 - 0.53). Habitat selection did not differ between the

sexes ($df = 1$, $\chi^2 = 0.29$, $p = 0.59$) or among years ($df = 7$, $\chi^2 = 5.59$, $p = 0.54$). In all years, females selected habitat that was of higher value than that available (Figure 12). Thus, we have compelling evidence that fishers on Stirling selected habitats with similar characteristics and quality as do fishers elsewhere (Thomasama et al. 1994, Powell 2004). The forest characteristics selected by fishers include relatively mature forests with canopy and shrub layer structuring and with moderate percentages of hardwoods (Figure 13). These results do not suggest that fishers refused to use low quality habitats when no better habitats are available. Additional habitat modeling may show other metrics that help to explain fishers' habitat selection.

Lastly, we tested the hypothesis that, as the study progressed, fishers' home ranges encompassed more areas that had been clearcut within that preceding 10 years. We calculated all areas that were clearcut within the 10 years prior to and including the year for which we estimated the home range of individual fishers and divided this area by the area of the 95% isopleth boundary of the home range. We tested for differences between the sexes and through time (years of our study) using proc genmod of SAS with a gamma distribution.

The percent of home ranges that included recent clearcuts did not differ between females (11%, 95% CI = 10 - 13) and males (14%, 95% CI = 11 - 17; $df = 1$, $\chi^2 = 0.01$, $p = 0.944$). The proportion of home ranges recently logged correlated positively with year ($\beta = 0.106$ [0.0042 - 0.208], $\chi^2 = 4.17$, $p = 0.041$; Figure 13). We found no interactions between sex and year ($df = 1$, $\chi^2 = 0.01$, $p = 0.946$). Fishers in the first few years after the releases established home ranges with only small proportions recently logged but that proportion increased through time. Over time, and as the fisher population grew, the number of clearcut stands increased and some clearcuts were inside established

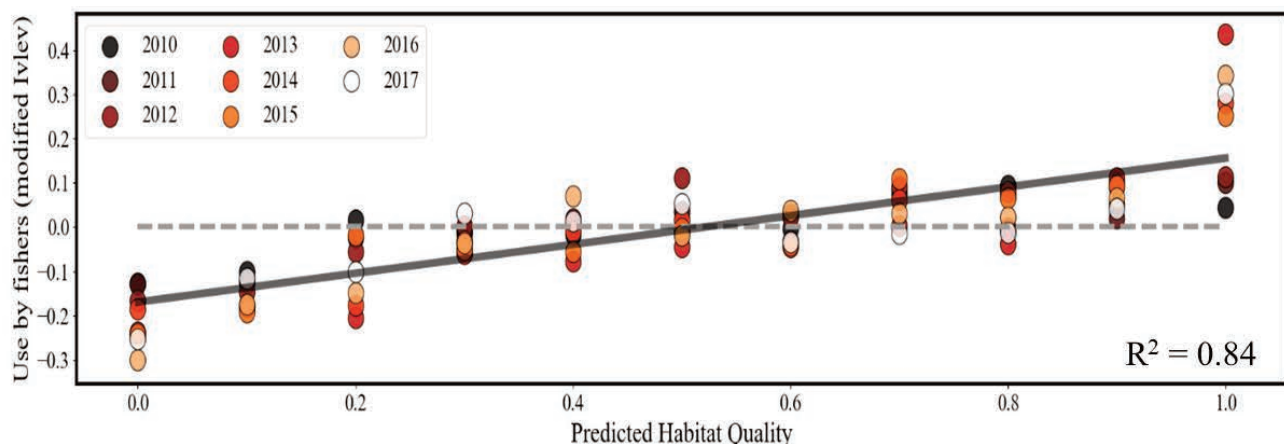


Figure 11. A). Resource selection of fishers across all years (2010-2017) based habitat quality estimates from the Thomasama model.

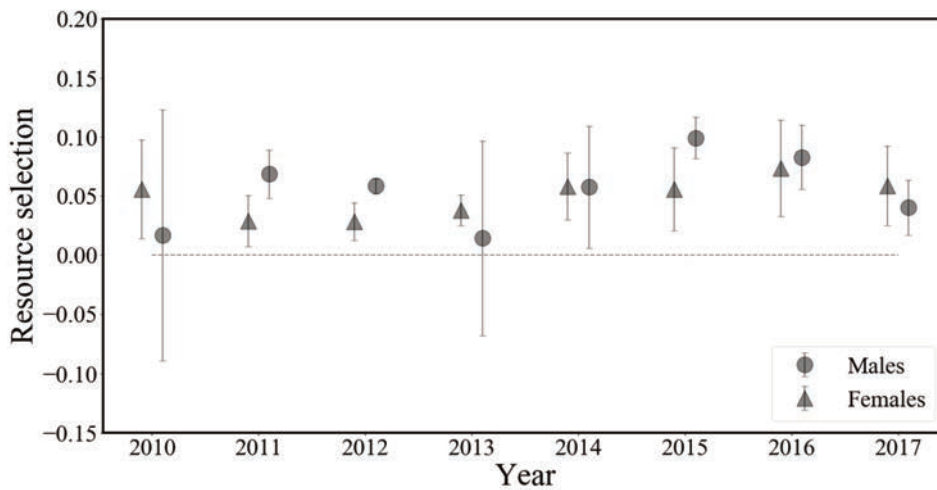


Figure 12. The mean modified Ivlev resource selection value \pm 95% CI for male and female fishers by year for all fishers with at least 20 locations in a year (sufficient to generate a kernel density estimate of home range). For each fisher the resource selection function is calculated as the mean values from all their locations in a year (used) as compared to the mean habitat value found in the 95% boundary of their estimated kernel density estimate from all locations (available) (2009 data are combine with 2010 data for this year only). Estimates of habitat are from SPI forest inventory stand-data and only for fishers.

home ranges of fishers. Although fishers avoided clearcut stands (they rank of low quality in Figure 10), the mean proportion of fishers' home ranges in clearcuts remained below the 25% threshold that has been observed for martens on logged landscapes elsewhere (Payer and Harrison 2003).

Population Viability Analysis

We used a 4-stage Leslie matrix model (Figure 14) and stochastic simulations of the female portion of the reintroduced fisher population on Stirling to explore potential long-term (50 years) outcomes for the population. Our analyses for the Survival, Reproduction

and Habitat section of this Final Report have shown that the fisher population on Stirling appears to have grown through 2017 and appears safe from imminent extinction under present conditions. Consequently, we used simulations to explore the future viability of the population if the environmental conditions remain constant or as projected by Sierra Pacific and to explore the conditions that might cause the population to go extinct. We simulated 2 sets of scenarios to investigate the potential for habitat changes to affect the fisher population:

1) A reintroduced fisher

population living on a landscape similar to Stirling in 2010 and with no changes in habitat quality for 50 years and, consequently, no systematic changes in vital rates (reproduction and survival), carrying capacity (K) or dispersal. We call this our "Baseline" scenario and it represents a landscape managed to maintain constant proportions of forest at different management stages, which vary in their quality of habitat for fishers. We varied this scenario by simulating the effects of a wildfire in year 10. In subsequent years, the area burned succeeds through stages of forest management through year 50. Vital rates, K and dispersal varied in response to forest

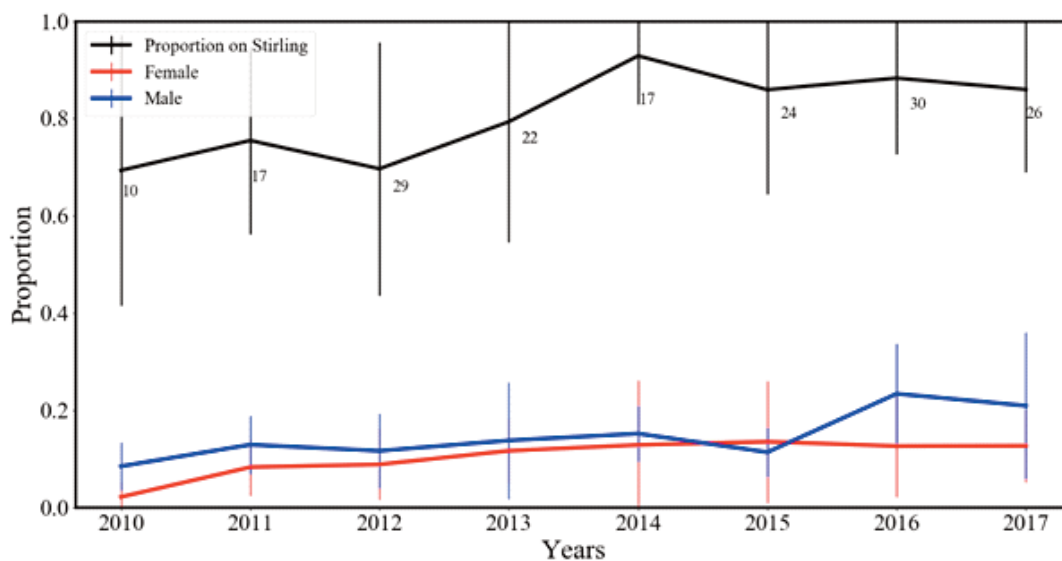


Figure 13. The mean proportion of fisher home ranges that overlapped Stirling (black lines; numbers represent the number of home ranges that overlapped Stirling by at least 50%) and the proportion of a home range that was logged in the prior 10 years, for females (red lines) and males. All error bars are standard deviations.

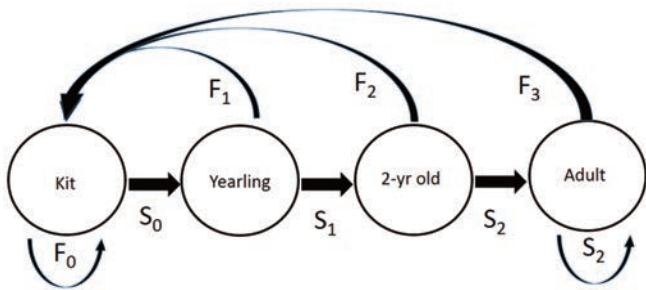


Figure 14. Life-stage model of the fisher life cycle used to construct projection matrices to simulate reintroductions. Each circle represents the discrete life-stage of a female fisher with the kit life-stage consisting of all females ages 0 to <1 year old, the yearling life-stage is females 1 to <2, 2-yr olds ages 2 to <3, and the adult life-stage is all females ages 3 and older. The arrows connecting the circles represents the transition values between the life-stages: F_0 = the expected number of female offspring born to a newborn female (always 0), F_1 = the expected number of female offspring born to a yearling female (also always 0), F_2 = the expected number of female offspring born to a 2-yr old female, and F_3 = the expected number of female offspring born to an adult female, S_0 = probability of surviving from birth until 1st birthday, S_1 = probability of surviving from 1st to 2nd birthday, S_2 = probability of surviving from 2nd to 3rd birthday and annually after 3rd birthday.

changes following the fire. In separate simulations, the fire covered 10% to 60% of the forest in 10% increments.

2) A reintroduced fisher population living on a landscape with forest cover changing as projected by Sierra Pacific for their land, leading to changes in habitat quality with concomitant changes in K , vital rates and dispersal. As with scenario 1, we varied this scenario by simulating the effects of a wildfire of various extents in year 10, with concomitant forest succession and changing vital rates, K and dispersal.

For these scenarios, we started with the vital rates presented in the Survival and Reproduction sections of this Final Report. To calculate K , we first placed a 100x100 m grid across Stirling and calculated the habitat quality (quantified using Thomasma’s index) for each cell in each year, 2010-2017, using stand data supplied by Sierra Pacific. We summed the cell-specific habitat quality values over all cells on Stirling in each year to obtain a total for habitat quality on Stirling in each year. We then placed the outline of the 95% utilization distribution for each female fisher for each year on the habitat quality map for each year and summed the habitat quality values within each outline, giving us a total for the habitat quality for each fisher for each year. Finally, we calculated the mean total of habitat quality for all female fishers for each year and divided that number into the total for Stirling for each year, yielding an estimate of the number of female home ranges that

Stirling could support each year. We truncated that number to the nearest whole female fisher, yielding estimates for K that ranged from slightly over 40 to slightly under 50 female fishers. For scenario 1 with no fire, we used $K = 45$. We note that this estimate of K is just slightly larger than the minimum estimated number of female fishers on Stirling in 2015-2017. For scenario 2 with no fire, K gradually grew from 45 to 57. For other scenarios, we used calculations of K based on the projected habitat for each year.

Sierra Pacific provided us with the projected changes in forest cover for the coming 50 years and we used those projections to change the distribution of habitat quality for our simulated landscape annually. In the projections, forest cover is categorized as 1) early seral, 2) small tree (high canopy) forest, 3) open forest, 4) medium tree (high canopy) forest, and 5) large tree (closed canopy) forest. Forest succession and forest management move forest stands progressively through these 5 stages. In early years, Stirling is roughly 15% early seral, 10% open forest, and 25% each small, medium and large trees. During the 1st 10 years, early seral increases to about 20%, open forest decreases to about 5% with the other stages mostly unchanged. From year 20 on, early seral, small trees and open forest gradually decrease while medium trees increase to about 35% and large trees to about 45%. The mean and maximum habitat qualities for stands of the 5 stages are given in Table 11.

For each year of our simulations, we adjusted survival in a logistic manner based on the size of female fisher population relative to K for that year. We did not reduce reproduction because Bulmer’s (1974, 1975) extensive analyses of fisher’s responses to snowshoe hare population cycles showed that fisher populations responded by changing juvenile and adult survival but not reproduction.

Because Stirling does not exist as an island but is embedded within a forested landscape, we allowed young-of-the-year fishers to disperse from Stirling. As a simulated population for Stirling approached K , young

Table 11. Mean and maximum habitat quality (\pm SD), quantified using Thomasma’s index (0=not habitat for fishers; 1= best habitat for fishers)), for forest stands of different stages. All stages had at least one stand with habitat quality = 0.

Forest Stage	Habitat Quality	Maximum
Early Seral	0.001 \pm 0.007	0.06
Small Tree Forest	0.20 \pm 0.20	0.84
Open Forest	0.02 \pm 0.05	0.27
Medium Tree Forest	0.30 \pm 0.25	0.89
Large Tree Forest	0.27 \pm 0.25	0.96

of the year dispersed in numbers that we calculated using a logistic function.

We simulated the release of fishers over 3 years to match our actual releases, reducing the mean denning rate for females in year-1 and year-3 as documented by Facka et al. (2016).

We generated random deviations from mean values of vital rates based on the documented standard deviations using a random normal distribution, limiting survival values to vary between 0 and 1. In previous annual reports, we have demonstrated the effects of juvenile survival on population dynamics, as have Lewis et al. (2012) and Powell et al. (2012), and we have provided evidence from field data and simulations that juvenile survival on Stirling is ≥ 0.5 and probably ≈ 0.6 . We set juvenile survival at 0.6 for our simulations. We also set the standard deviation around this parameter to be equal to that for adult survival (0.1).

At each time step, we truncated population size to its integer value, *i.e.*, to whole numbers of fishers.

We simulated forest fires covering 10% to 60% of our simulated landscape by reducing the areas of the

small tree to large tree stages proportionately and adding the total amount to the early seral stage. We then moved this area of forest through Sierra Pacific's forest stages, reaching the medium tree stage by the end of 50 years.

For each scenario we replicated 50 years of population change 1000 times. We present mean values (\pm SD) of the mean female population size across all runs for each year, excluding populations after they went extinct. If a simulated population fell below 5 females, we considered the population to have gone extinct and ended the simulation. For all simulations, we calculated the percent of 1000 replicates that went extinct and the year of extinction. We consider the extinction rate for simulated populations to be an index of the probability of extinction for real fisher populations. Thus, if the extinction index goes up or down as simulated conditions changed, we expect that the extinction probability for a real population would go up or down with such changes in the environmental conditions but not necessarily match the extinction index.

Finally, we iteratively increased and decreased by 10% the values of vital rates across all life-stages and the

values of K , denning rate and dispersal rate to estimate the elasticities. Elasticity quantifies the potential for each variable to affect population growth or extinction.

Our simulations produced unsurprising outcomes (Figure 15). In our Baseline Scenario with no forest fire and with unchanging habitat quality, vital rates and K , the mean population size across all runs grows for about 10 years, reaching a little over 40 females, after which population size wobbles just under K .

A wildfire in year 10 causes fisher population decrease and the decrease is greater with greater percentage of the area burned (Figure 15). The population continues to decrease for about 10 years

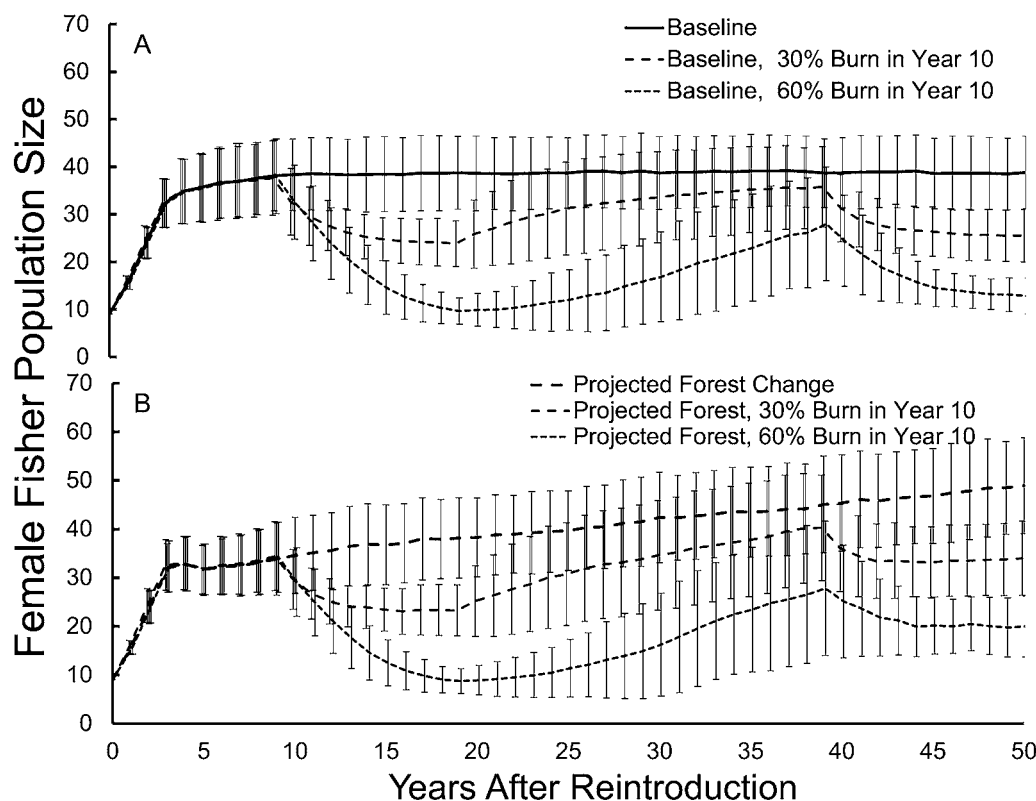


Figure 15. Stochastic simulations of 50 years of population size change for female fishers living under 2 sets of scenarios. The simulations are based on the fisher population reintroduced to Stirling. A. Baseline. Simulation with no changes in habitat quality, vital rates (reproduction and survival) and carrying capacity (K). Also simulations of population change after a wildfire burns 30% or 60% of the simulated landscape in year 10. B. Simulations with habitat quality changing as proposed by Sierra Pacific, leading to changes in K and concomitant changes in vital rates and dispersal. Also simulations of population change after a wildfire burns 30% or 60% of the simulated landscape in year 10.

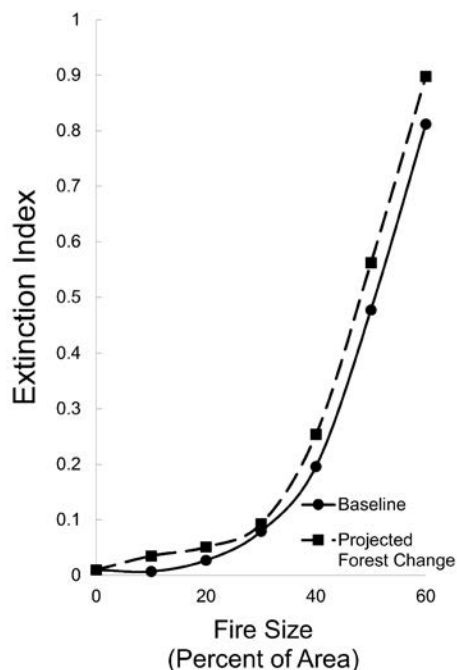


Figure 16. For a simulated population of female fishers, change in the extinction index (proportion of simulation runs that led to extinction) with the extent of a wildfire on a simulated landscape.

after the fire while the simulated landscape has a large area of early seral forest. After 10 years, much of the burned area has succeeded to forest with small trees with a nearly closed canopy and the fisher population increases gradually until the forest succeeds to larger, more open forest, which causes the fisher population to dip again.

The extinction index for the

population is about 0.01 for the Baseline scenario with no wildfire and increases progressively more steeply with increasingly larger burns, and exceeds 0.8 when fire size is 60% of the simulated landscape (Figure 16).

In scenario 2 with no wildfire, simulated fisher population grows rapidly following reintroduction but then dips when the population reaches about 35 female fishers at about 5-6 years. At this time, the population no longer receives supplementation from new releases while the proportion of the forest in early seral stage grows (Figure 15). Thereafter, the population grows gradually to shy of 50 female fishers as more of the forest reaches the stages with large trees.

As with the Baseline scenario, forest fire in year 10 causes fisher population decrease on the landscape managed for forest change and the decrease is greater with greater percentages of the area burned (Figure 15). The population decrease for about 10 years post-fire, after which much of the burned area has succeeded to forest with small trees with a nearly closed canopy.

Thereafter the fisher population increases gradually but to higher population sizes than seen in the simulations without simulated forest change. When the forest develops to larger, more open stages of managed forest, the fisher population dips again.

As with the Baseline scenario, the extinction index for the population is about 0.01 with no wildfire and increases progressively more steeply to nearly 0.9 when fire size is 60% of the simulated area (Figure 16).

These simulations are not predictions for the fisher population on Stirling but explore how the population might respond under various changes in conditions. The extinction index was consistently lower for the simulations with the forest managed for change consistent with the goals of Sierra Pacific for Stirling. The simulated fire, however, occurred at year 10, when the forest had its largest proportion in the early seral stage. Had we simulated fires to occur later when the population was larger, we expect that the extinction index would not be as large.

Our elasticity analyses show that adult survival rates have greater effects on population growth than do litter sizes (Figure 17). Denning rate, however, has an effect similar to the effects of survival during a fisher's first year and mean litter size and juvenile survival.

These elasticity results must be considered within the context of our model-building. First,

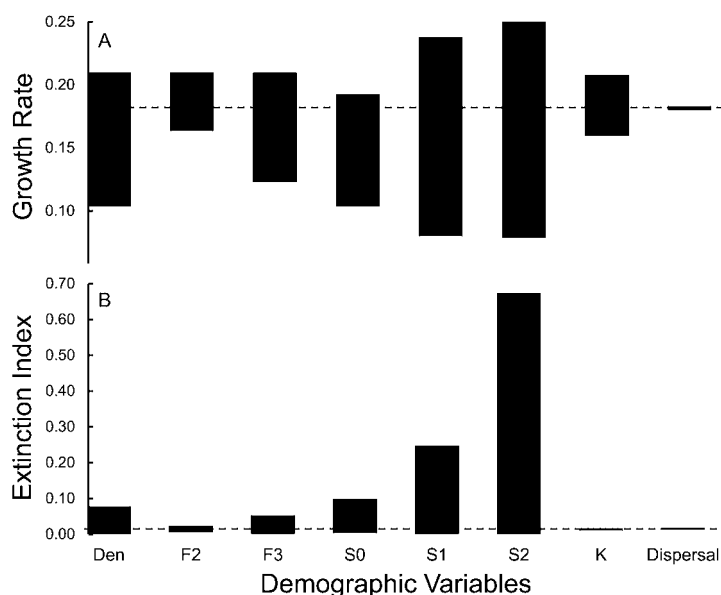


Figure 17. Elasticity shows the effects on population growth rate and on extinction of demographic variables representing life history traits. Baseline values for variables are the estimates for the Stirling fisher population. The horizontal dashed line shows the (A) the rate of population growth during the first 8 years of the reintroduction and (B) the extinction index, calculated as the proportion of 1000 simulations using the baseline values for demographic variables. The bars show the increases and decreases in population growth rate and the extinction index caused by 1-at-a-time 10% increases and decreases of the values of the demographic variables from baseline. Den = mean denning rate. F2 = mean number female offspring born to 2-year old females. F3 = mean number female offspring born to females > 3 years old. S0 = mean annual survival of female kits. S1 = mean annual survival of 1-year old females. S2 = mean annual survival of females > 2-years old. K = carrying capacity. Dispersal = dispersal rate.

to be consistent with the results of Bulmer's (1974, 1975) analyses, we reduced age-specific survival rates as population size increased but not age-specific litter sizes. Thus, the model structure dictates that changes in survival will have greater effects on survival than will changes in reproduction.

Second, we built a stage-specific Leslie matrix and not an age-specific matrix. The consequence is that adult survival is effectively repeated at each time step by as many adult age classes exist within the population. Had we built an age-specific model extending through the 10-year age span for fishers, each older age-specific survival rate would have progressively smaller and smaller effects on population growth, leaving S_0 and S_1 as the age-specific survival rates with the largest effect on population growth.

Nonetheless, juvenile survival in our model has a smaller influence on population growth than survival of yearlings and adults, which contrasts with the models developed by Buskirk et al. (2012) and Lewis et al. (2012). These other models used lower juvenile survival rates, which increased the relative importance of juvenile survival. These contrasts highlight the potential importance of the juvenile cohort and the very real importance of the lack of good estimates for juvenile survival.

Third, we treated litter size as a stage-specific rate, as we did with adult survival. Consequently, the effect of F_3 on population growth is inflated compared to age-specific litter sizes in a age-specific matrix model. Likewise, had our model now allowed different litter sizes for first and subsequent litters, then the importance of litter size would have been inflated.

Using *Vortex* to build an age-specific, spatially-specific model for a fisher reintroduction, Lewis et al (2012) found that litter size and juvenile survival had the greatest effects on reintroduction failure through population extinction. *Vortex* includes some realistic population effects, such as an Allee effect at small population sizes, that we did not incorporate into our model. Consequently, our model may underestimate extinction probabilities. In contrast, however, our model allowed juvenile dispersal beyond the simulated study site, making our simulated population an open population. We did not include recolonization of our simulated study site by fishers from a surrounding population established by dispersal of fishers from our study site. The consequence is that our model may overestimate extinction.

Our estimates of K for female fishers, 45 females at

present, is similar to our estimates of the minimum number of female fishers on Stirling at the end of 2017. If the fisher population on Stirling is, indeed, near carrying capacity, then the 68 fishers of both sexes at the end of 2017 yields a density of about 10 fishers/100 km². This density is somewhat lower than the density estimate for the Hoopa Indian Reservation in 2005 after a steep population decrease, 14 fishers/100 km², and is substantially less than the estimated density in 1998, 52 fishers/100 km² (Matthews et al. 2011).

Food Habits and Prey Population Dynamics

With our 2015 Annual Report, we distributed to all cooperators on this project a report entitled "Fisher (*Pekania pennanti*) prey availability and habitat use on managed timberlands in the northern Sierra Nevada". That report is appended to this Final Report (Appendix 3) and is summarized here.

The relationship between predators and their prey is among the most important but least studied aspects of carnivore ecology and few studies have identified spatial distributions and abundances of prey to understand better carnivore habitat needs. Human-caused disturbance through industrial logging is hypothesized to diminish habitat quality for fishers. Over 3 years (2012, 2014 and 2015), we used live-trapping and remote cameras on 250 locations to identify where prey species were most likely to occur and what habitat attributes best described their presence, relative abundances and diversity on Stirling. We quantified canopy cover, basal area, mean quadratic mean diameter of trees, numbers of logs, and density of roads across 3 scales (microsite, stand, and neighborhood). We detected 44 genera of mammals, birds and herpetofauna and the most widely distributed and detected mammals were deer mice (*Peromyscus* spp.), woodrats (*Neotoma* spp.), chickarees (*Tamiasciurus douglasii*), western grey squirrels (*Sciurus griseus*), California ground squirrels (*Otospermophilus beecheyi*) and northern flying squirrels (*Glaucomys sabrinus*). Average trees size and the number of downed logs drove occupancy, abundance and diversity of these mammals. Deer mice and woodrats, the most abundant mammals, avoided or were indifferent to sites with high mean tree diameter whereas tree squirrels selected such sites. Downed logs are important to fishers.

Non-invasive Sampling of Klamath Fishers

We monitored the fisher population using non-invasive, genetic techniques on the 587 km² Eastern Klamath Study Site (Figure 1), which has been

monitored since 2006. Our analyses of the effects of removing fishers from that population to be reintroduced on Stirling has been published (Green et al. 2018) and is appended as Appendix 4.

The continued research on this site now emphasizes studying the effects of the Beaver Creek and Happy Camp fires of 2014 on fishers and other medium-sized carnivores that are vulnerable to high-severity forest fires. The combination of many years of fire suppression and global climate change is predicted to increase the extent and intensity of wildfires in certain parts of the world, particularly in western North America. Understanding the ecological, conservation and management effects of forest fires on forest obligate species is important. We used data collected from this long-term monitoring program to investigate the effects of the 2 naturally-occurring, mixed-severity forest fires on fishers. We developed a spatial capture-recapture model to estimate the short-term effects of the 2 fires on fisher abundance and distribution using data collected with hair snares 1 year prior to the fires and for 3 years following them. Fisher abundance decreased by approximately 40% following these fires, particularly in the areas of the fire footprints and irrespective of the burn severity. We also estimated that 4% of the northern California and southern Oregon population of fishers has been exposed to forest fires in the last 3 years. Our results suggest that the changing fire dynamics in the western USA will have strong, negative effects on fisher populations. Populations of other forest-obligate species may also fare poorly under these altered fire regimes.

Non-Invasive Population monitoring on Stirling

We conducted non-invasive surveys for fishers on Stirling in 2013 through 2016, expanding the study area and modifying the protocol as needed to deal with the frequency of bear visits to sampling sites. We developed a protocol for monitoring fishers on Stirling and it is appended as Appendix 5.

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Publications Related to Project

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- Powell, R. A., A. N. Facka, M. W. Gabriel, J. H. Gilbert, J. M. Higley, S. D. LaPoint, N. P. McCann, W. Spencer & C. M. Thompson. 2017. The fisher as a model organism. Chapter 11. Pp 299-313. In. Macdonald, D. W., L. Harrington & C. Newman (editors). *Biology and Conservation of Wild Musteloids*. Oxford University Press, London.
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- model predictions with field data. Pgs 93-137 in *Biology and conservation of martens, sables, and fishers: a new synthesis*. K.B. Aubry, W.J. Zielinski, M.G. Raphael, G. Proulx & S.W. Buskirk, editors. Cornell University Press, Ithaca, New York, USA.
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- Swiers, R. C. 2013. Non-invasive genetic sampling and mark-recapture analysis of a fisher (*Martes pennanti*) population in northern California used as a reintroduction source. MS thesis. North Carolina State University, Raleigh.

Papers Presented at Conferences

2009

- Powell, R. A., J. C. Lewis, B. G. Slough, S. M. Brainerd, N. R. Jordan, A. V. Abramov, V. Monakhov, P. A. Zollner & T. Murakami. 2012. Evaluating translocations of martens, sables, and fishers: Testing model predictions with field data. 5th International Martes Symposium, Seattle, Washington.

2010

- Facka, A. N. & R. A. Powell. 2010. Fishers translocated to the northern Sierra Nevada. Annual conference, American Society of Mammalogists.
- Swiers, R. C. & R. A. Powell. 2010. Use of non-invasive genetic data to estimate fisher (*Martes pennanti*) population parameters in the eastern Siskiyou Mountains of California. Annual conference, American Society of Mammalogists.

2011

- Powell, R. A. & A. N. Facka. 2011. Identifying occupied home ranges using movements of translocated fishers (*Martes pennanti*). Annual conference, American Society of Mammalogists.
- Swiers, R. C. & R. A. Powell. 2011. Use of non-invasive genetic data to estimate fisher (*Martes pennanti*) population parameters in the eastern Siskiyou Mountains of California. Annual conference,

Western Section, The Wildlife Society.

Swiers, R. C. & R. A. Powell. 2011. Use of non-invasive genetic data to estimate fisher (*Martes pennanti*) population parameters in the eastern Siskiyou Mountains of California. Annual conference, Society of Mammalogists.

2012

Clifford, D., L. Woods, V. Tkach, E. Hoberg, R. Callas, R. N. Brown, J. M. Higley, K. Haynes & M. W. Gabriel. 2012. Assessing disease risk from a novel parasite infection in Pacific fisher (*Martes pennanti*). Annual conference, Western Section, The Wildlife Society.

Facka, A. N., C. B. Beach, K. P. Smith & R. A. Powell. 2012. The role of predators and temperature in the timing of fisher (*Pekania pennanti*) den movements. Annual Conference, American Society of Mammalogists.

Facka, A. N. & R. A. Powell. 2012. Reintroduction of fishers into the Northern Sierra Nevada of California. Annual conference, Western Section, The Wildlife Society.

Facka, A. N., R. A. Powell & J.C. Lewis. 2012. On determining success in fisher translocations. Fisher Symposium, Annual conference, Western Section, The Wildlife Society.

Powell, R. A. 2012. Using models in ecology and conservation. Fisher Symposium, Annual conference, Western Section, The Wildlife Society.

Powell, R. A. & A. N. Facka. 2012. Identification of occupied home ranges using travel distances, changes in speed and final settlement of translocated fishers (*Martes pennanti*). Annual conference, Western Section, The Wildlife Society.

Powell, R. A. & A. N. Facka. 2012. Identifying occupied home ranges using travel distances, changes in speed and final settlement of translocated fishers (*Martes pennanti*). Annual conference, American Society of Mammalogists.

Swiers, R. C., A. N. Facka, R. Callas, P. Figura, L. Finley, J. S. Yaeger & R. A. Powell. 2012. Two fisher populations in managed forests in northern California. Annual conference, Western Section, The Wildlife Society.

2013

Facka, A. N., J. C. Lewis, P. Happe, K. Jenkins, R. Callas & R. A. Powell. 2013. Effects of timing of release on reproduction and population dynamics for

reintroduced populations of a forest Carnivore. Annual conference, American Society of Mammalogists.

Powell, R. A., S. Ellwood, R. Kays & T. Maran 2013. Stink or swim – Challenges for the study and conservation of small critters that hide, swim or climb and may otherwise make themselves unpleasant. Invited presentation, International Musteloid Conference, Oxford University, United Kingdom.

Powell, R. A., M. W. Gabriel, J. M. Higley, S. LaPoint, N. P. McCann, W. Spencer & C. M. Thompson 2013. The fisher as a model organism. Invited presentation, International Musteloid Conference, Oxford University, United Kingdom.

2014

Facka, A. N. & R. A. Powell. 2014. Identification of occupied home ranges using travel distances, changes in speed and final settlement of translocated fishers (*Pekania pennanti*). Symposium on Animal Movement and the Environment. Raleigh, North Carolina.

Facka, A. N., J. C. Lewis, P. Happe, K. Jenkins, R. Callas & R. A. 2014. Effects of timing of release on reproduction and population dynamics for reintroduced populations of a forest carnivore. 6th International Martes Symposium, Krakow, Poland.

Powell, R. A., M. W. Gabriel, J. M. Higley, S. LaPoint, N. P. McCann, W. Spencer, C. M. Thompson 2013. The fisher as a model organism. 6th International Martes Symposium, Krakow, Poland.

2015

Facka, A. N., R. Callas, D. Clifford, T. Engstrom, L. Finley, S. M. Matthews, K. P. Smith, R. C. Swiers, J. S. Yaeger & R. A. Powell. 2015. Reestablishing fishers on a managed landscape in California. Annual conference, Western Section, The Wildlife Society.

Facka, A.N., D. Clifford, M. Schroeder, R.A. Powell, R. Callas, S.M. Matthews, R.C. Swiers & T. Engstrom. 2015. Distribution and occupancy of potential fisher prey on the Stirling District. Annual conference, Western Section, The Wildlife Society.

Matthews, S. M. & R. C. Swiers. 2015. Synthesis of fisher habitat information in California and southern Oregon. Annual conference, Western Section, The Wildlife Society.

Powell, R. A. 2015. Home is (not) where the estimation is. Annual conference, Western Section, The Wildlife

Society.

Powell, R. A. 2015. Home is (not) where the estimation is. Joint Conference, Minnesota and Wisconsin Chapters, The Wildlife Society.

2016

Matthews, S. M., R. A. Powell, A. N. Facka, T. Engstrom, R. Callas, D. Clifford, L. Finley, D. S. Green, R. C. Swiers, J. M. Higley, K. M. Rennie, C. M. Cummins & R. E. Green. 2016. Fisher translocation and research in northern California: a foundation for fisher conservation measures in Oregon. Annual conference, Oregon Chapter, The Wildlife Society.

Green, D. S., S. M. Matthews, R. C. Swiers, L. L. Finley, R. Callas, S. Farber & R. A. Powell. 2016. Investigating the effects of wildfire on fishers and other mesocarnivores in northern California and southern Oregon. Annual conference, Western Section, The Wildlife Society.

Powell, R. A., R. Callas, S. Farber, L. Finley, D. S. Green, S. M. Matthews & R. C. Swiers. 2016. Using non-invasive DNA techniques to document the dynamics of a fisher population in California. Midwest Furbearer Conference, Ely, Minnesota.

2017

Facka, A. N. & R. A. Powell. 2017. Designing mesocarnivore projects to maximize their scientific and conservation value: A case study with a fisher reintroduction. Invited presentation, Annual conference, The Wildlife Society.

Facka, A. N. & R. A. Powell. 2017. Conspecific competition and resource depression affect movement and home range establishment of translocated fishers (*Pekania pennanti*). 12th International Mammal Conference. Perth, Australia.

Facka, A. N., R. Callas, D. Clifford, T. Engstrom, L. Finley, S. M. Matthews, K. P. Smith, R. C. Swiers, J. S. Yaeger & R. A. Powell. 2017. What was old is new again: Testing fisher habitat models with independent data. Annual conference, Western Section, The Wildlife Society.

Fraik, A. K., A. N. Facka, R. A. Powell. 2017. Generalist small mammals and forest management in northern California. Annual conference, American Society of Mammalogists.

Green, D. S., S. M. Matthews & R. A. Powell. 2017. Investigating the interactions among fishers, gray foxes, and ringtails using dynamic occupancy models. Annual conference, Western Section, The Wildlife

Society.

Green, D. S., S. M. Matthews, R. C. Swiers, L. L. Finley, R. Callas, S. Farber & R. A. Powell. 2017. Using spatial capture-recapture to model fisher population change over time in northern California and southern Oregon. Annual conference, Western Section, The Wildlife Society.

Green, D. S., S. M. Matthews, L. L. Finley & R. A. Powell. 2017. The effects of mixed-severity wildfires on fisher population dynamics. Annual conference, Ecological Society of America.

Green, D. S., S. M. Matthews, L. L. Finley & R. A. Powell. 2017. The effects of mixed-severity wildfires on fisher populations dynamics. Annual conference, The Wildlife Society.

Powell, R. A. & A. N. Facka. 2017. Reintroducing a native predator: Multiple objectives and good science. Invited seminar, Waikato University, Hamilton, New Zealand.

Powell, R. A. & A. N. Facka. 2017. Pictures, predators and prey: Camera data and community dynamics after a fisher reintroduction. 12th International Mammal Congress. Perth, Australia.

2018

Anderson, E. L., T. R. Peltier, D. S. Green, S. M. Matthews & R. A. Powell. 2018. Maximizing the detection of fishers: Investigating the biotic and abiotic factors that affect DNA analyses of *Pekania pennanti* with non-invasive hair snares. 7th International Martes Symposium, Ashland, Wisconsin, USA.

Facka, A. N., D. L. Clifford, K. P. Smith, S. M. Matthews, T. Engstrom, E. Murphy, L. Finley, J. S. Yaeger, P. Figura, R. C. Callas & R. A. Powell. 2018. Reintroduction of a mature forest carnivore onto a landscape managed for timber production. Annual conference, Society of American Foresters.

Facka, A. N., D. L. Clifford, K. P. Smith, S. M. Matthews, T. Engstrom, E. Murphy, L. Finley, J. S. Yaeger, P. Figura, R. Callas & R. A. Powell. 2018. Reintroduction of a mature forest carnivore onto a landscape managed for timber production. 7th International Martes Symposium, Ashland, Wisconsin, USA.

Facka, A. N., S. M. Matthews, B. Johnson, G. W. Watts, C. Dillingham, M. Reno, S. Anderson, P. Figura, T. Rickman & R. A. Powell. 2018. Tracking expansion of fishers after release: A case of cooperative conservation. 7th International Martes Symposium,

- Ashland, Wisconsin, USA.
- Facka, A. N., K. M. Moriarty & S. M. Matthews. 2018. Translocations as road maps for conservation planning: The case of fishers in Oregon. Joint conference, Oregon and Washington Chapters, The Wildlife Society.
- Facka, A. N. & R. A. Powell. 2018. Estimating energy expenditure and total body water content of fishers in California with doubly labeled water. 7th International Martes Symposium. Ashland Wisconsin, USA.
- Green, D. S., S. M. Matthews, L. Finley & R. A. Powell. 2018. Fishers, foxes, fire, and salvage – Updates from the Klamath-Siskiyou Carnivore Project in northern California and southern Oregon. Annual conference, Western Section, The Wildlife Society.
- Green, D. S., S. M. Matthews, L. L. Finley & R. A. Powell. 2018. Mixed severity wildfires have a negative effect on fishers and a positive effect on gray foxes. Annual conference, Western Section, The Wildlife Society.
- Green, D. S., S. M. Matthews & R. A. Powell. 2018. The effects of fuels treatments, wildfire, and post-fire salvage logging on fishers and a carnivore community. 7th International Martes Symposium, Ashland, Wisconsin, USA.
- Green, D. S., S. M. Matthews & R. A. Powell. 2018. The effects of mixed severity wildfires on fisher population dynamics. Annual conference, The Wildlife Society.
- Matthews, S. M. & D. S. Green. 2018. Fisher research update. Oregon Forest Carnivore Working Group, Springfield, Oregon, USA.
- Powell, R. A. 2018. Home range peregrinations. Annual conference, Western Section, The Wildlife Society.
- Powell, R. A. 2018. Reintroducing a native predator: Multiple objectives and good science. Invited seminar, University of Wisconsin, Stevens Point.
- Powell, R. A., D. Clifford, A. N. Facka, D. S. Green, S. M. Matthews, K. Smith, R. Callas, L. Finley & T. Engstrom. 2018. Fisher reintroduction to the northern Sierra. Annual conference, Western Section, The Wildlife Society.
- Powell, R.A. & A.N. Facka. 2018. Estimating home ranges using different currencies and incorporating animal perception their environments. 7th International Martes Symposium, Ashland, Wisconsin.
- Powell, R. A., A. N. Facka, L. Rogers & S. A. Mansfield. 2018. Insights from estimating home ranges using different currencies and incorporating how animals perceive their environments. Invited presentation, Annual conference, US - International Association of Landscapes Ecologists.
- Powell, R. A., A. N. Facka, L. Rogers & S. A. Mansfield. 2018. Estimating home ranges using different currencies and animals' perception about their environments. Annual conference, American Society of Mammalogists.
- Townsend, A., A. N. Facka, S. M. Matthews & M. Szykman Gunther. 2018. Stand occupancy by fisher (*Pekania pennanti*) and tree squirrels: Investigating the influence of masting hardwoods in northern California. 7th International Martes Symposium, Ashland, Wisconsin.
- 2019
- Facka, A. N., K. P. Smith, J. S. Yaeger, E. Murphy, M. Reno, L. Finley, R. Callas, P. Figura, S. M. Matthews & R. A. Powell. Annual conference, Western Section, The Wildlife Society.
- Facka, A. N. & R. A. Powell. 2019. Estimating energy expenditure and total body water content of fishers in California with doubly labeled water. Annual conference, Western Section, The Wildlife Society.
- Morrison, E. E., T. R. Peltier, D. S. Green, S. M. Matthews & R. A. Powell. 2019. Effects of wildfire on the structure of carnivore communities in northern California. Annual conference, Western Section, The Wildlife Society.
- Powell, R. A., S. Barber-Meyers, A. N. Facka, T. Gable, S. A. Mansfield, L. D. Mech, L. Rogers and S. Windells. 2019. Estimating home ranges using different currencies and animals' perception about their environments. Annual conference, Western Section, The Wildlife Society.
- Powell, R. A. 2018. Reintroducing a native predator: Multiple objectives and good science. Invited seminar, Coe College.
- Sirakowski, C. A., D. S. Green & S. M. Matthews. 2019. When the fisher's away, the mice will play: The effects of mixed severity wildfire on small mammal occupancy in northern California and southern Oregon. Annual conference, Western Section, The Wildlife Society.
- Smith, K. P, A. N. Facka & R. A. Powell. 2019. Clarifying Breeding System, Breeding Phenology and Infanticide from a Reintroduced Population of Fishers. Annual conference, Western Section, The Wildlife Society.

Townsend, A. M., A. N. Facka, S. M. Matthews & M. Szykman Gunther. 2019. Stand occupancy by fisher and tree squirrels in northern California: Investigating the influence of masting hardwoods. Annual conference, Western Section, The Wildlife Society.

Appendix 1. Data on fishers (*Pekania pennanti*) captured during research on the reintroduction of fishers onto the Stirling Management Area of Sierra Pacific Industries in the Northern Sierra Nevada, California, December 2009 - January 2017. Birth years marked with an asterisk (*) were judged to be young-of-the-year and, therefore, did not have a premolar pulled to estimate age using cementum annuli.

ID	PIT5	Sex	Birth Year	Cohort	1 st Capture			Last Data		
					Date	Age	Wt (kg)	Date	Data Type	Mortality Cause
10057	1776D	F	2007	Pilot	2009 06 28	2	2.1	2009 06 28	Trap	
10066	F776D	F		Pilot	2009 06 28		2.3	2009 10 18	Trap	
10074	19019	F		Pilot	2009 10 26		2.0	2009 10 26	Trap	
10075	19780	F		Pilot	2009 10 27			2009 10 27	Trap	
10080	199B9	F	2007	Year 1	2009 11 30	2		2011 05 11	Den Visit	
10014	E612B	F	2006	Year 1	2009 12 01	3	1.9	2009 12 03	Trap	
10079	F6280	F	2008	Year 1	2009 12 02	2		2010 06 26	Mortality	Bobcat predation or scavenge
10054	EB687	F	2005	Year 1	2009 12 03	4		2009 12 09	Mortality	Unknown in box, + for Brodifacoum
20082	D00B0	F	2004	Year 1	2009 12 08	6		2010 07 17	Mortality	Drowning (water tank), No ARs
20083	17ECC	F	2007	Year 1	2009 12 10	3		2011 03 8	Collar Drop	
20085	F8B8D	F	2007	Year 1	2010 01 05	10		2017 08 27	Mortality	
10087	17582	F	2007	Year 1	2010 01 07	2		2011 05 21	Den Visit	
20089	19316	F	2005	Year 1	2010 01 08	4		2010 04 07	Flight	
20091	168F2	F	2006	Year 1	2010 01 09	3		2010 05 02	Ground	
20095	F6FF3	F		Year 1	2010 01 12		2.2	2010 01 08	Trap	
20096	F65B6	F	2007	Year 1	2010 01 21	3		2010 06 29	Mortality	Bobcat predation or scavenge
20108	F64A2	F	2006	Year 2	2010 10 29	4	2.2	2010 10 29	Trap	
20111	93B5A	F	2005	Year 2	2010 11 02	9		2014 07 07	Mortality	suspect raptor predation
20113	18CCE	F	2004	Year 2	2010 11 03	6	2.0	2010 11 03	Trap	
20117	18FFF	F	2007	Year 2	2010 11 10	4		2011 08 25	Mortality	Unknown
10109	18871	F	2008	Year 2	2010 12 10	5		2013 07 18	Trap	
10124	182F4	F	2009	Year 2	2011 01 08	3		2012 07 24	Mortality	Unknown, possible felid
20125	2114A	F	2005	Year 2	2011 01 11	5	2.2	2011 01 11	Trap	
20126	22DE2	F	2005	Year 2	2011 01 11	5	2.3	2011 01 11	Trap	
10123	17FD8	F	2009	Year 2	2011 01 11	5		2015 01 07	Collar Drop	
10131	21FB6	F	2008	Year 2	2011 01 22	5		2013 6 20	Mortality	Traumatic injury head, neck
10132	1E003	F	2007	Year 2	2011 01 23	8		2015 12 14	Mortality	Pending
20147	1F955	F	2011	Cohort 2011	2011 10 21	6	2.3	2017 10 25	Trap	
20148	21392	F	2011	Cohort 2011	2011 10 22	4		2016 02 10	Mortality	Pending
20155	23955	F	2011	Cohort 2011	2011 10 26	3		2014 09 08	Mortality	Drowned
20158	23775	F	2008	Year 3	2011 11 04	3		2012 04 17	Flight	
20159	2189C	F	2009	Year 3	2011 11 04	3		2012 10 12	Mortality	Unknown
20160	22DF4	F	2009	Year 3	2011 11 08	2	2.3	2011 11 08	Trap	
20164	219BA	F	2011	Year 3	2011 11 10	0	2.0	2011 11 10	Trap	
20165	21DFE	F	2011	Year 3	2011 11 10	0		2014 03 14	Ground	
20166	1E03E	F		Year 3	2011 11 11			2012 02 27	Mortality	Systemic disease, unknown pathogen
20169	20058	F	2010	Year 3	2011 11 15	1	2.1	2012 11 14	Trap	
20170	1F0AC	F	2011	Year 3	2011 11 15	0	1.7	2011 11 15	Trap	
20173	200AF	F	2009	Year 3	2011 11 17	2	2.1	2011 11 17	Trap	
20177	24A39	F	2008	Year 3	2011 11 29	3	1.9	2011 11 29	Trap	

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ID	PIT5	Sex	Birth Year	Cohort	1 st Capture			Last Data		
					Date	Age	Wt (kg)	Date	Data Type	Mortality Cause
20178	252FD	F	2008	Year 3	2011 11 29	3		2012 12 08	Walkin	Systemic disease, + Brodifacoum (0.17 ppm)
20179	714C2	F	2008	Year 3	2011 11 29	3		2013 12 10	Ground	
20175	1F111	F	2009	Year 3	2011 11 29	3		2012 06 02	Mortality	Bobcat predation
20176	213D9	F	2011	Year 3	2011 11 29	0	1.9	2011 11 29	Trap	
20180	716D4	F	2011	Year 3	2011 11 29	0	1.8	2011 11 29	Trap	
20182	7488E	F	2007	Year 3	2011 11 30	4	2.0	2011 11 30	Trap	
20186	17CB9	F		Year 3	2011 12 02		2.1	2011 12 02	Trap	
20189	209DD	F	2011*	Cohort 2011	2012 01 15	0		2014 03 23	Collar Drop	
20190	20950	F	2011	Cohort 2011	2012 01 25	6		2017 04 05	Mortality	Pending
20192	1EA8D	F	2012	Cohort 2012	2012 10 24	0	2.1	2015 05 10	Mortality	
20192	1EA8D	F	2012	Cohort 2012	2012 10 24	3		2011 11 17	Trap	Pending
20193	6178F	F	2012	Cohort 2012	2012 10 24	0	2.2	2012 12 09	Ground	
20196	36A8B	F	2012	Cohort 2017	2012 10 28	5		2017 10 25	Trap	
20197	242DB	F	2012	Cohort 2012	2012 10 29	5	2.2	2017 10 15	Trap	
20199	614DF	F	2011	Cohort 2012	2012 10 30	5		2016 12 17	Mortality	
20201	3828E	F	2012	Cohort 2017	2012 11 07	5		2017 11 10	Trap	
20202	35978	F	2012	Cohort 2012	2012 11 09	0	1.9	2012 12 19	Ground	
20204	397A3	F	2012	Cohort 2012	2012 11 13	5	2.2	2017 10 12	Trap	
20205	35232	F	2013	Cohort 2016	2013 09 24	3		2016 10 26	Genetic	
20206	11720	F	2013	Cohort 2017	2013 10 01	4		2017 11 08	Trap	
20352	None	F		Cohort 2013	2013 10 09			2012 03 13	Telemetry	
20208	76482	F	2013	Cohort 2013	2013 10 13	4		2017 09 09	Collar Drop	
20209	15852	F	2012	Cohort 2013	2013 10 14	3		2016 03 25	Collar Drop	
20210	16026	F	2013	Cohort 2013	2013 10 14	3		2016 09 21	Mortality	Pending
20212	69940	F	2013	Cohort 2013	2013 10 15	3		2016 07 26	Mortality	Pending
20213	23737	F	2013	Cohort 2013	2013 10 18	1		2015 03 31	Telemetry	
20214	34786	F	2013	Cohort 2013	2013 10 22	0		2013 12 15	Telemetry	
20255	178B2	F	2012	Cohort 2015	2013 10 25	3		2016 09 09	Telemetry	
20219	33860	F	2014	Cohort 2014	2014 10 13	2	2.3	2017 12 12	Telemetry	
20231	36069	F	2014	Cohort 2014	2014 10 13	3	2.5	2017 12 06	Trap	
20220	35006	F	2014	Cohort 2014	2014 10 14	2		2016 05 27	Mortality	Pending
20223	31237	F	2013	Cohort 2014	2014 10 16	1	2.2	2017 10 15	Trap	
20226	36775	F	2014	Cohort 2014	2014 10 19	3	2.6	2017 10 20	Trap	
20233	36752	F	2013	Cohort 2014	2014 10 21	4		2017 12 02	Trap	
20229	36651	F	2014	Cohort 2014	2014 10 26	3	2.5	2017 10 12	Trap	
20230	43476	F	2014*	Cohort 2014	2014 10 26	0	2.2	2014 10 26	Trap	
20234	51812	F	2013	Cohort 2014	2014 11 09	3		2016 04 12	Mortality	Pending
20242	0A3CA	F	2015	Cohort 2015	2015 10 08	2	2.3	2017 10 31	Trap	
20244	0DE63	F	2015	Cohort 2015	2015 10 12	2	2.2	2017 11 10	Trap	
20246	136BC	F	2015	Cohort 2015	2015 10 13	1		2016 08 18	Mortality	
20248	27FB2	F	2015	Cohort 2015	2015 10 13	2	2.4	2017 11 03	Trap	
20271	194F6	F	2015	Cohort 2015	2015 10 14	2	2.3	2017 10 29	Trap	
20254	16FDD	F	2015	Cohort 2015	2015 10 15	0	1.8	2015 10 15	Trap	
20257	17105	F	2015	Cohort 2015	2015 10 19	2	2.3	2017 11 16	Trap	

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ID	PIT5	Sex	Birth Year	Cohort	1 st Capture			Last Data		
					Date	Age	Wt (kg)	Date	Data Type	Mortality Cause
20243	0D8F1	F	2015	Cohort 2015	2015 10 20	0	2.3	2015 11 04	Trap	Pending
20266	27E2F	F	2015	Cohort 2015	2015 10 27	0		2015 12 08	Mortality	
20268	29B29	F	2015	Cohort 2015	2015 10 28	2	2.6	2017 11 04	Trap	
20270	2741A	F	2015	Cohort 2017	2015 10 30	2		2017 11 08	Trap	
20272	19277	F	2015	Cohort 2015	2015 11 01	2	2.4	2017 11 03	Trap	Mortality
20274	114C0	F	2015	Cohort 2015	2016 01 25	1		2017 05 12	Mortality	
20316	6C8CA	F	2016	Cohort 2016	2016 09 15	1	2.1	2017 11 10	Trap	
20314	6CA8E	F	2016	Cohort 2016	2016 10 05	1	2.2	2017 10 19	Trap	
20317	6C8E9	F	2016	Cohort 2016	2016 10 12	1	2.2	2017 11 01	Trap	Trap
20287	6B15E	F	2016	Cohort 2016	2016 10 17	0	1.9	2016 10 17	Trap	
20293	82466	F	2016	Cohort 2016	2016 10 21	1		2017 10 18	Trap	
20294	281C5	F	2016	Cohort 2017	2016 10 21	1		2017 10 22	Trap	
20295	6BA97	F	2016	Cohort 2016	2016 10 21	0		2016 10 30	Trap	Trap
20297	6C654	F	2016	Cohort 2016	2016 10 23	0	1.8	2016 10 23	Trap	
20300	6ACDD	F	2016	Cohort 2016	2016 10 24	1	2.1	2017 10 22	Trap	
20302	6ADBE	F	2016	Cohort 2016	2016 10 24	0	2.0	2016 10 24	Trap	
20307	6AA7A	F	2016	Cohort 2016	2016 10 26	0		2016 11 29	Mortality	Trap
20308	6ADD2	F	2016	Cohort 2016	2016 10 26	0	2.4	2016 10 26	Trap	
20312	6BAF2	F	2014	Cohort 2016	2016 10 29	3	2.4	2017 10 20	Trap	
20310	6ACF9	F	2016	Cohort 2016	2016 10 29	0	2.1	2016 10 29	Trap	
20311	6B153	F	2016	Cohort 2016	2016 10 29	0	2.0	2016 10 29	Trap	Trap
20315	6ACB0	F	2016	Cohort 2016	2016 11 04	0	2.1	2016 11 04	Trap	
20320	189A3	F	2015	Cohort 2016	2016 11 15	1	2.2	2017 12 11	Telemetry	
20323	36045	F	2017	Cohort 2017	2017 10 12	0		2017 10 16	Trap	
20324	6CA71	F	2017	Cohort 2017	2017 10 13	0		2017 10 24	Trap	Trap
20325	C7CE3	F	2017	Cohort 2017	2017 10 13	0	2.3	2017 10 13	Trap	
20328	3DE1F	F	2017	Cohort 2017	2017 10 15	0	1.9	2017 10 15	Trap	
20330	BF4A7	F	2015	Cohort 2017	2017 10 17	2	2.3	2017 10 17	Trap	
20335	6C3E7	F	2013	Cohort 2017	2017 10 20	4	2.3	2017 10 20	Trap	Trap
20336	C8D67	F	2017	Cohort 2017	2017 10 23	0	1.9	2017 10 23	Trap	
20337	6BE1C	F	2017	Cohort 2017	2017 10 24	0	1.9	2017 10 24	Trap	
20338	6B09D	F	2017	Cohort 2017	2017 10 28	0		2017 11 29	Trap	
20341	CAFE8	F	2017	Cohort 2017	2017 10 29	0		2017 12 12	Trap	Trap
20344	3D5F2	F	2017	Cohort 2017	2017 11 03	0		2017 11 10	Trap	
20346	6C494	F	2017	Cohort 2017	2017 11 04	0	1.9	2017 11 04	Trap	
20348	BDD2B	F	2017	Cohort 2017	2017 11 07	0	2.1	2017 11 07	Trap	
20349	6AD19	F	2017	Cohort 2017	2017 11 10	0	2.1	2017 11 10	Trap	Trap
20350	BCC38	F	2017*	Cohort 2017	2017 12 01	0	2.2	2017 12 01	Trap	
10055	17815	M	2004	Pilot	2009 06 15	5	4.3	2009 08 8	Trap	
10068	F61F4	M		Pilot	2009 10 19		3.0	2009 10 19	Trap	
10072	F5DBE	M		Pilot	2009 10 23		3.6	2009 10 23	Trap	Telemetry
20084	181F9	M	2008	Year 1	2010 01 05	4		2012 12 31	Telemetry	
10060	18308	M	2007	Year 1	2010 01 07	5		2012 05 15	Mortality	
20088	172D6	M	2008	Year 1	2010 01 07	1	2.1	2010 01 07	Trap	

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ID	PIT5	Sex	Birth Year	Cohort	1 st Capture			Last Data		
					Date	Age	Wt (kg)	Date	Data Type	Mortality Cause
20090	596E2	M	2004	Year 1	2010 01 08	9		2013 09 18	Mortality	Possible avian predation or scavenge
20092	EA479	M	2009	Year 1	2010 01 10	0	3.2	2010 01 10	Trap	
20093	F0858	M	2006	Year 1	2010 01 11	7		2013 06 19	Telemetry	Unknown
20094	16848	M	2006	Year 1	2010 01 12	3		2010 06 09	Telemetry	
20107	F605B	M	2008	Year 2	2010 10 28	3	4.1	2012 01 10	Telemetry	
20112	178AA	M	2006	Year 2	2010 11 03	4	4.6	2010 11 03	Trap	
20114	58985	M	2007	Year 2	2010 11 05	6	4.4	2015 06 14	Telemetry	
20116	18CC8	M	2006	Year 2	2010 11 10	9		2015 04 24	Mortality	Pending
20118	E99DE	M	2009	Year 2	2010 11 10	1	2.9	2010 12 09	Trap	
10020	18C3E	M	2006	Year 2	2010 12 09	4	5.3	2011 03 12	Mortality	Roadkill, + Brodifacoum Bromadiolone
10062	E862D	M	2009	Year 2	2010 12 15	1		2011 01 08	Trap	
10121	E81E8	M	2010	Year 2	2010 12 15	0	3.7	2010 12 14	Trap	
20122	18AA5	M	2004	Year 2	2011 01 06	7		2013 04 19	Telemetry	
20127	20968	M	2009	Year 2	2011 01 14	1	2.8	2011 01 17	Trap	
20129	1FD6E	M		Year 2	2011 01 17		3.5	2011 01 19	Trap	
20128	22526	M	2006	Year 2	2011 01 20	4		2011 03 16	Telemetry	
20130	2152D	M	2009	Year 2	2011 01 21	1	3.0	2011 01 21	Trap	
20146	24033	M	2011	Cohort 2011	2011 10 19	0	3.6	2012 02 15	Telemetry	
20149	2305B	M	2010	Cohort 2011	2011 10 22	1	4.1	2012 11 26	Flight	
20150	24101	M	2011	Cohort 2011	2011 10 23	2		2013 04 08	Mortality	Unknown, No ARs in muscle
20156	1FE60	M	2011	Cohort 2011	2011 10 27	1	4.2	2012 11 08	None	
20163	F0D5E	M	2010	Year 3	2011 11 09	1	2.7	2011 11 09	Trap	
20161	20461	M	2011	Year 3	2011 11 09	0	2.8	2011 11 09	Trap	
20162	1EC6D	M		Year 3	2011 11 09		3.4	2011 11 09	Trap	
20167	1E14C	M	2009	Year 3	2011 11 11	2		2011 12 03	Telemetry	
20168	1FC73	M	2010	Year 3	2011 11 11	1	3.0	2011 11 11	Trap	
20171	241B8	M	2011	Year 3	2011 11 15	0	2.8	2011 11 15	Trap	
20172	24315	M	2010	Year 3	2011 11 17	2		2012 10 14	Trap	
20174	D7E28	M		Year 3	2011 11 17		3.5	2011 11 17	Trap	
20181	846D5	M	2010	Year 3	2011 11 29	1	3.0	2011 11 29	Trap	
20183	1E10F	M	2011	Year 3	2011 12 01	6	4.6	2017 12 07	Trap	
20185	24366	M	2011	Year 3	2011 12 02	0	3.3	2011 12 02	Trap	
20184	207E0	M		Year 3	2011 12 03		4.0	2011 12 02	Trap	
20187	87375	M	2008	Year 3	2011 12 14	3		2011 12 14	Trap	
20188	1EC04	M	2010	Year 3	2011 12 15	4	4.8	2016 02 28	Telemetry	
20191	24B09	M	2012	Cohort 2012	2012 10 22	1		2014 01 27	Mortality	Unknown
20194	64311	M	2012	Cohort 2012	2012 10 25	0		2013 01 27	Telemetry	
20195	39A7E	M	2012	Cohort 2012	2012 10 25	0		2012 10 28	Trap	
20198	3AD54	M	2012	Cohort 2012	2012 10 30	0	3.0	2012 10 30	Trap	
20200	1E613	M	2011	Cohort 2012	2012 11 06	2		2014 03 26	Telemetry	Unknown
20203	38908	M	2012	Cohort 2012	2012 11 10	0	3.4	2012 11 26	Flight	
20211	70926	M	2013	Cohort 2013	2013 10 14	3		2016 04 04	Mortality	
20218	XXXX2	M	2014	Cohort 2014	09 08 2014	0		2014 09 08	Mortality	drowned
20221	36623	M	2014	Cohort 2014	2014 10 15	0	3.4	2014 10 15	Trap	

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ID	PIT5	Sex	Birth Year	Cohort	1 st Capture			Last Data		
					Date	Age	Wt (kg)	Date	Data Type	Mortality Cause
20222	None	M		Cohort 2014	2014 10 15			2014 10 15	Genetic	
20222	None	M		Cohort 2014	2014 10 15			2014 10 15	Genetic	
20224	34589	M	2014	Cohort 2017	2014 10 17	3		2017 11 10	Trap	
20225	32976	M	2014	Cohort 2014	2014 10 19	3	3.9	2017 10 21	Trap	
20227	33890	M	2014	Cohort 2014	2014 10 21	0		2015 01 21	Trap	
20228	12834	M	2014	Cohort 2014	2014 10 22	0	3.1	2014 10 22	Trap	
20232	36056	M	2014	Cohort 2014	2014 11 01	0		2014 11 06	Trap	
20235	36103	M	2012	Cohort 2014	2015 01 28	3		2015 11 29	Telemetry	
20236	33276	M	2013	Cohort 2014	2015 01 31	3	4.3	2017 04 17	Telemetry	
20273	33850	M	2015	Cohort 2015	2015 09 30	1	4.0	2017 12 28	Telemetry	
20239	32402	M	2015	Cohort 2015	2015 10 12	0	3.0	2015 10 24	Trap	
20240	33785	M	2015	Cohort 2017	2015 10 12	2		2017 10 25	Trap	
20241	35734	M	2015	Cohort 2015	2015 10 12	0	3.1	2017 10 25	Trap	
20247	18F2C	M	2013	Cohort 2017	2015 10 13	4		2017 10 14	Trap	
20245	36226	M	2015	Cohort 2015	2015 10 13	2	4.3	2017 11 07	Trap	
20253	34046	M	2015	Cohort 2015	2015 10 15	1	4.0	2017 4 02	Telemetry	
20259	34509	M	2015	Cohort 2015	2015 10 21	1		2016 10 27	Trap	
20260	34104	M	2015	Cohort 2017	2015 10 23	2		2017 10 14	Trap	
20319	6AD53	M	2014	Cohort 2016	2016 09 17	2	3.6	2016 11 11	Trap	
20275	None	M		Cohort 2016	2016 09 20			2016 9 20	Genetic	
20275	None	M		Cohort 2016	2016 09 20			2016 9 30	Genetic	
20275	None	M		Cohort 2016	2016 09 20			2016 10 21	Genetic	
20275	None	M		Cohort 2016	2016 09 20			2016 10 07	Genetic	
20275	None	M		Cohort 2016	2016 09 20			2016 10 21	Genetic	
20276	None	M		Cohort 2016	2016 09 20			2016 9 20	Genetic	
20277	6C7C8	M	2015	Cohort 2016	2016 10 05	1		2016 10 05	Genetic	
20351	6BCCE	M	2016	Cohort 2016	2016 10 17	1		2017 11 18	Trap	
20290	6C84D	M	2016	Cohort 2016	2016 10 18	0	3.1	2016 10 18	Trap	
20291	6CAC0	M	2016	Cohort 2016	2016 10 19	0		2016 10 24	Trap	
20296	6C340	M	2016	Cohort 2016	2016 10 23	1	3.9	2017 10 13	Trap	
20298	6C7A0	M	2016	Cohort 2016	2016 10 23	0	2.8	2016 10 23	Trap	
20301	6AD0C	M	2016	Cohort 2016	2016 10 24	0	3.1	2016 10 24	Trap	
20303	6ADDA	M	2016	Cohort 2016	2016 10 24	0		2016 12 21	Trap	
20309	6AEFB	M	2016	Cohort 2016	2016 10 24	0	3.4	2016 10 28	Trap	
20321	6B0B5	M	2015	Cohort 2016	2016 12 18	2	3.9	2017 10 17	Trap	
20322	33539	M	2017*	Cohort 2017	2017 10 12	0	3.3	2017 10 12	Trap	
20326	CC7EB	M	2015	Cohort 2017	2017 10 13	2	4.0	2017 10 13	Trap	
20327	6C440	M	2017	Cohort 2017	2017 10 14	0		2017 10 25	Trap	
20332	BDBB9	M	2016	Cohort 2017	2017 10 18	1	3.7	2017 10 18	Trap	
20331	BC063	M	2017	Cohort 2017	2017 10 18	0	3.1	2017 10 18	Trap	
20334	CDF0F	M	2017	Cohort 2017	2017 10 18	0		2017 10 25	Trap	
20333	CDE69	M	2017*	Cohort 2017	2017 10 18	0		2017 10 21	Trap	
20339	6C605	M	2017	Cohort 2017	2017 10 28	0		2017 11 02	Trap	
20340	6C7C7	M	2017	Cohort 2017	2017 10 29	0	3.1	2017 10 29	Trap	

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ID	PIT5	Sex	Birth Year	Cohort	1 st Capture			Last Data		
					Date	Age	Wt (kg)	Date	Data Type	Mortality Cause
20342	6AB6A	M	2017	Cohort 2017	2017 10 31	0	3.5	2017 10 31	Trap	
20343	BDCE6	M	2016	Cohort 2017	2017 11 02	1		2017 12 04	Trap	
20345	6C7FF	M	2017*	Cohort 2017	2017 11 03	0	3.0	2017 11 05	Trap	
20217	XXXX1	Unk	2014	Cohort 2014	07 07 2014	0		2014 07 07	Mortality	suspect raptor predation
	272EF							2015 11 17	Telemetry	
	3282E							2016 02 03		
	9C7FF							2017 11 03	Trap	

FINAL REPORT, 2017, FISHER REINTRODUCTION

APPENDIX 2

Fisher Habitat Selection on Stirling from 2010-2014:

A critical test of our understanding of fisher habitat requirements

Aaron N. Facka

Sean Matthews

Roger A. Powell

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INTRODUCTION:

To understand how fishers (*Pekania pennanti*) respond to intensive forest management 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. A major objective of our project is to evaluate models of habitat for fishers, those already existing and ones we develop. No less than 6 models of habitat have been developed, including those of Allen (1983), the California Wildlife Habitat Relationship model (Airoal 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, 1994, 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. 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, Powell et al. 2015, Thomasma 1991, 1994). Over the decades, models of fisher habitat have employed ever more sophisticated 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 1994, 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 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 elsewhere in California and fishers east of the Rocky Mountains, where steep terrain does not exist, can not be associated with steep terrain.

Testing models with independent data has provided feedback and new insight into why habitat characteristics are important to fishers (Powell 2004, Thomasma 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 (*sensu* Mitchell et al. 2002), 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 (Powell et al. 2015). 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 general spatially and temporally (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 home range 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 than 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 1; Allen 1983). Nevertheless, we included all hardwood trees in our models because Allen's original models consider 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 2).

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 3).

Carroll 1999 – adjusted

The Carroll (1999) model is 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. 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 (Figure 4A). 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 4B). 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 5). 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% canopy). We arbitrarily gave these bins the 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 5).

Eq. 4:

CWHR Habitat Quality =

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

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 5). 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 5).

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 6). 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 took 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 3). Within this MCP we generated 12,591 random points and then 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 were aberrantly distant from all other locations but within a similar time window were eliminated. As with our other analyses we then calculated the habitat value across for the Allen, Carroll (1999), Carroll 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 fisher's 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 & 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 7). 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 2). 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 2; Figure 8a). 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 8a). 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 2). 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 9).

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 3). 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 3). 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 3b). 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 3b). The relationship between predicted habitat quality and both the Ivlev and modified-Ivlev RSF were significantly different from zero ($\beta = 0.35$; Table 2). 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 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 produced 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 (Figure 7). 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 10A). For the Allen model a consistent pattern emerged at every level of analysis – female, year, isopleth, and mean values (Table 3; Figure 10). 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 10). As with our previous analyses there is 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 3). 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.

GPS

The proportion of available habitat within the convex polygons of fishers with GPS collars was high variable (Figure 11). 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 11A) whereas other fishers higher proportions of moderate or moderately high habitat quality (Figure 11B). As a result the relationship between habitat quality and RSFs for individual fishers showed high variability as well (Figure 11). 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 4). 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 12). 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 13). 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 habitat 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 14).

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 had the lowest predicted the highest 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 15). 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 15). 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 4).

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 2; Figure 16). 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 8a). 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 2). 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 17). The pattern of female habitat selection we observed within home ranges was markedly different from that seen at the scale of locations (Table 3). 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 3). 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 18). Similar to the Allen model the results of the Carroll analysis with GPS data were highly variable (Table 4). 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 4). 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 16). 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 17). 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

linearly related to percent hardwood that is used in the Allen model. Nevertheless, the mathematical relationship among these variables to habitat quality is different from in 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 19). 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 15). 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 2). Both the Ivlev and modified-Ivlev RSFs indicate fishers avoided habitat quantified below 0.50, but selected habitat values greater than 0.50 (Figure 20). 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 2).

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 3). 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 21). 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 21). 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 2). 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 (Figure 22). 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 17). 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 3). 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 3). 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 13). 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 23). 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 4). 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 5). Thus, the mean value for TRI in our model was on 0.14. The average for canopy and TRI would rarely approach 1.0 and hence there were few locations with predicted habitat values of 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 & 6). 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 5). 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 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.

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 5). 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 24). 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 5). 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 2; Figure 25). 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 25)

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 26A). Similarly, fishers appeared to show selection for habitat in the highest quality habitats disproportionate to their availability (Figure 26). 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 3). 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 13). Moreover, fishers showed relatively weak selection for the highest CWHR habitat values compared to models such as the Allen, and Carroll index (Figure 13).

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 27). Similar to the Allen model the results of the Carroll analysis with GPS data were highly variable (Table 4). 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 4).

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 roughly predict 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 2 and 4). Nevertheless, in contrast to the Carroll index model and the Allen model the mean selection and avoidance scores are relatively weak (Figure 13). Additionally, the available evidence suggests that fishers avoid areas with moderate habitat quality characterized by trees less than 60 cm (24'') an 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 28). 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 29). 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 2; Figure 30). The Ivlev and modified-Ivlev resource selection functions showed very similar relationships between SPI habitat quality and fisher selection (Figure 30). 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 31). Similarly, male and female selection and avoidance were similar to the pattern observed with the combined data set (Table 2) 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 3). We observed an apparent avoidance of the HF2hv classification at the scale of the home range (Figure 3) 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 32). 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 4). 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 13). 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 32, 33). 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 34) 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 34). 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 2; Figure 35). 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 3). 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 36). 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 36). 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 35). Nevertheless, within home ranges Mean habitat provided poor explanatory power (Table 3). 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 37). Five the 6 models predicted the proportion available habitat as over 60% (Figure 38). 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 2). 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 39).

Territory Opportunity

We evaluated 300 individual hexagons from 2010 to 2014 for overlap with female 50% utilization distributions (Figure 40). 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 3, 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 7). 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 41).

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 42). 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 8). 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 8). 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 8). 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 = $1.8 \pm 1.49-2.34$ 95%CI) or 2 (Odds ratio = $2.37 \pm 1.6-3.5$) 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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Table 1. Tested models with variables and data type used during testing.

Model Family	Model	Data Type	Scale	Variables
Allen	HSI	^a SPI inventory	Stand	Canopy, QMD, Percent Hardwood, Canopy Layers
Allen	HSI	GNN/Lemma	30x30 m	Canopy, QMD, Percent Hardwood, Canopy Layers
Carroll	Logistic Regression	^a SPI inventory	Stand	Canopy, QMD, Percent Conifer, Annual Precip, UTM, UTM ²
Carroll	Index	^a SPI inventory	Stand	Canopy, Terrain Ruggedness Index (TRI)
Carroll	Index	GNN/Lemm	30x30 m	Canopy, Terrain Ruggedness Index (TRI)
SPI	Index	^a SPI inventory	Stand	Canopy, QMD, #trees >22"
CWHR	Expert Opinion	^a SPI inventory	Stand	Canopy, Tree Size Category, Tree Type
Davis	GAM_norou	CWHR/Davis	1 km ²	Topographic Relief, CWHR fisher rating, Annual Precip
Davis	GAM_north	CWHR/Davis	1 km ²	Annual Precip, CWHR fisher rating, Canopy + Tree Size
Davis	GAM_south	CWHR/Davis	1 km ²	Annual Precip, Canopy + Tree Size, elevation
Davis	GAM_state	CWHR/Davis	1 km ²	Topographic Relief, Hardwood Ranking, Annual Precip, Prop. Canopy > 60%
Davis	MAX_nor	CWHR/Davis	2 km ²	Annual precip, Proportion Canopy > 60%, Topographic Relief
Davis	MAX_state	CWHR/Davis	2 km ²	Annual precip, Proportion Canopy > 60%, Topographic Relief

^a Models were constructed for each year from 2010 to 2015.

Table 2. Regression results for data on fisher habitat selection across Stirling (DataType = SPI_inv) and surrounding areas (DataType = GNN).

ModelType	DataType	Model	Dependent	Y_int	β	R ²	Y-pval	β -pval
Allen	SPI_inv	meanHSI	Mean mIV	-0.13	0.14	0.06	0.33	0.50
Allen	GNN	HSI_GNN	tot mIV	-0.20	0.35	0.46	0.04	0.03
Allen	SPI_inv	HSI_Fem	Fem mIV	-0.07	0.11	0.07	0.42	0.45
Allen	SPI_inv	mMHSI	Male mIV	-0.09	0.15	0.17	0.23	0.23
Car index	SPI_inv	mClavg	Mean mIV	-0.37	0.66	0.51	0.03	0.02
Car index	GNN	mCarIg	tot mIV	-0.18	0.30	0.21	0.20	0.18
Car index	SPI_inv	CarI Fem	Fem mIV	-0.19	0.23	0.30	0.10	0.10
Car index	SPI_inv	CarI Male	Male mIV	-0.22	0.45	0.21	0.22	0.18
Carroll	SPI_inv	mC9avg	Mean mIV	-0.21	0.49	0.60	0.04	0.01
Carroll	SPI_inv	fCari	Fem mIV	-0.12	0.23	0.30	0.10	0.10
Carroll	SPI_inv	mCari	Male mIV	-0.22	0.46	0.21	0.22	0.18
CWHR	SPI_inv	mCWHRavg	Mean mIV	-0.08	0.12	0.27	0.11	0.13
CWHR	SPI_inv	fCWHR	Fem mIV	-0.10	0.18	0.29	0.10	0.11
CWHR	SPI_inv	mCWHR	Male mIV	-0.09	0.17	0.37	0.06	0.06
Davis	CWHR/Davis	mMaxst	tot mIV	0.03	-0.03	0.00	0.78	0.85
Davis	CWHR/Davis	mMax_nor	tot mIV	0.04	-0.06	0.02	0.73	0.72
Davis	CWHR/Davis	mGstate	tot mIV	0.02	-0.03	0.04	0.55	0.60
Davis	CWHR/Davis	mGsouth	tot mIV	-0.03	0.00	0.00	0.68	1.00
Davis	CWHR/Davis	mGnorth	tot mIV	0.05	-0.07	0.12	0.28	0.32
Davis	CWHR/Davis	mGnorou	tot mIV	-0.07	0.08	0.02	0.65	0.73
Mean	SPI_inv	mMavg	Mean mIV	-0.09	0.16	0.58	0.02	0.01
SPI	SPI_inv	mSPIavg	Mean mIV	-0.09	0.16	0.49	0.03	0.02
SPI	SPI_inv	mFSPIavg	Fem mIV	-0.10	0.20	0.46	0.05	0.03
SPI	SPI_inv	mMSPI	Male mIV	-0.09	0.19	0.51	0.03	0.02

Table 3. Results of home range analysis where the mean modified-Ivlev resource selection function was regressed on predicted habitat quality by each model for female home ranges across Stirling from 2010 to 2014.

Model	Isopleth	Y-int	β	R ²	Y-pval	β -pval
Allen	0.25	-0.064	0.072	0.071	0.295	0.456
	0.50	-0.038	0.036	0.024	0.471	0.669
	0.75	-0.018	0.008	0.003	0.627	0.891
	0.90	-0.009	-0.002	0.000	0.786	0.969
	0.95	-0.014	0.008	0.003	0.665	0.876
	ALL	-0.026	0.025	0.017	0.475	0.718
Carroll	0.25	-0.015	-0.025	0.023	0.676	0.677
	0.50	-0.001	-0.036	0.096	0.955	0.384
	0.75	-0.038	0.046	0.191	0.100	0.207
	0.90	-0.022	0.022	0.102	0.156	0.368
	0.95	-0.026	0.024	0.152	0.070	0.266
	ALL	-0.020	0.006	0.005	0.245	0.845
Car index ^a	0.25	-0.153	0.234	0.433	0.031	0.038
	0.50	-0.122	0.187	0.446	0.028	0.035
	0.75	-0.117	0.178	0.413	0.036	0.045
	0.90	-0.103	0.161	0.389	0.048	0.054
	0.95	-0.110	0.170	0.440	0.031	0.037
	ALL	-0.103	0.186	0.437	0.033	0.037
CWHR ^b	0.25	-0.098	0.142	0.487	0.016	0.025
	0.50	-0.053	0.074	0.314	0.057	0.092
	0.75	-0.028	0.032	0.170	0.103	0.237
	0.90	-0.019	0.023	0.243	0.058	0.148
	0.95	-0.033	0.043	0.307	0.049	0.096
	ALL	-0.040	0.063	0.378	0.030	0.058
Mean	0.25	-0.067	0.074	0.157	0.110	0.257
	0.50	-0.035	0.023	0.029	0.258	0.635
	0.75	-0.027	0.019	0.063	0.133	0.484
	0.90	-0.020	0.014	0.057	0.155	0.507
	0.95	-0.028	0.022	0.095	0.097	0.386
	ALL	-0.033	0.030	0.120	0.070	0.327
SPI	0.25	-0.064	0.072	0.071	0.295	0.456
	0.50	-0.038	0.036	0.024	0.471	0.669
	0.75	-0.018	0.008	0.003	0.627	0.891
	0.90	-0.009	-0.002	0.000	0.786	0.969
	0.95	-0.014	0.008	0.003	0.665	0.876
	ALL	-0.110	0.135	0.773	0.087	0.121

Table 4. Regression results for mean modified-Ivlev resource selection function for 4 fishers outfitted with GPS collars in fall of 2012, 2013, and 2015 (n=2) on Stirling district in northern California.

Model	Y-int	β	R ²	Y-pval	B-pval
Harvest Year	0.129	-0.0001	0.325	0.122	0.085
Allen	-0.004	-0.020	0.020	0.867	0.696
Carroll	-0.015	0.006	0.001	0.724	0.937
Carroll Index	0.010	-0.046	0.104	0.701	0.363
CWHR	-0.015	0.012	0.053	0.154	0.521
SPI	-0.023	0.033	0.028	0.551	0.646

Table 5. Descriptive statistics for variables tested for correlations to terrain ruggedness index (TRI) using Spearman ranked correlation across Stirling district in northern California. Rho is the estimated relationship between TRI and the variable and the P-val is the probability of observing Rho at random with shaded areas those variables with P-val < 0.05.

Variable	N	Mean	Std Dev	Min	Max	Rho	P-val
TRI	20000	22.357	13.147	0.000	160.00	1.000	..
Percent Conifer	20000	71.844	29.457	0.000	101.00	-0.291	<.0001
Percent Hardwood	20000	24.191	26.077	0.000	100.00	0.374	<.0001
Tree Height (feet)	20000	34.720	15.431	0.000	99.00	-0.061	<.0001
Tree Diversity	20000	0.665	0.209	0.000	1.00	0.146	<.0001
Canopy	20000	67.818	35.870	0.000	100.00	0.163	<.0001
Logs/Acre	20000	1.649	1.501	0.000	11.43	0.014	0.0452
QMD	20000	11.682	4.440	0.000	26.31	-0.013	0.0584
Snags >24"	20000	0.528	0.682	0.000	6.37	-0.005	0.4406
Basal Area	20000	135.028	81.986	0.000	380.72	-0.003	0.6582

Table 6. Number of females per year, the mean numbers of locations per female and the mean utilization distribution area.

Year	NumFemales	Mean Locs	Mean UD (km ²)
2010	6	49±32	17±7
2011	7	44±23	28±20
2012	12	55±28	17±56
2013	13	87±32	15±46
2014	13	77±19	16±27

Table 7. The number of hexagons with 0 or at least 1 female fishers home ranges (at 0.50 isopleth level) by year. For hexagons with at least 1 female fisher home the mean number, standard deviation, maximum and minimum home ranges found with the those hexagons.

Year	0	1	Total	Mean Num	Stdev Num	Max Num	Min Num
2010	57	67	124	1.4	0.61	3	1
2011	39	30	69	1.1	0.31	2	1
2012	52	47	99	1.1	0.28	2	1
2013	56	53	109	1.2	0.60	4	1
2014	64	55	119	1.4	0.71	4	1

Table TerrOppResult. Results from logistic regression analysis and the absence or occurrence of at least 1 female fisher home range at 50% isopleth within hexagon for all fishers located in northern California on Stirling from 2010 to 2014.

Model	SubParam	AIC _c	R ²	Beta	OR	OR-low	OR-high
HexHab	none	682.2	0.09	0.7	1.9	1.6	2.4
HexHab(discrete)	CBT vs TC	668.6	0.13	-0.9	0.3	0.2	0.5
HexHab(discrete)	SC vs TC	668.6	0.13	0.5	1.2	0.7	1.9
Random	none	718.7	0.00	0.0	1.0	1.0	1.0
Year	2010vs2014	717.9	0.01	0.2	1.4	0.8	2.3
Year	2011vs2014	717.9	0.01	-0.2	0.9	0.5	1.6
Year	2012vs2014	717.9	0.01	0.0	1.1	0.6	1.8
Year	2013vs2014	717.9	0.01	0.0	1.1	0.7	1.9
Year-cont	none	719.5	0.00	-0.1	0.9	0.8	1.1
Easting	none	706.6	0.04	0.0	1.0	1.0	1.0
Northing	none	712.5	0.02	0.0	1.0	1.0	1.0
Easting×Northing	none	700.1	0.05	0.0	1.0	1.0	1.0

Figure 1: The relationships between habitat suitability for fishers and percent canopy closure, mean diameter at breast height (DBH) of overstory trees, tree canopy diversity, and percent canopy closure.

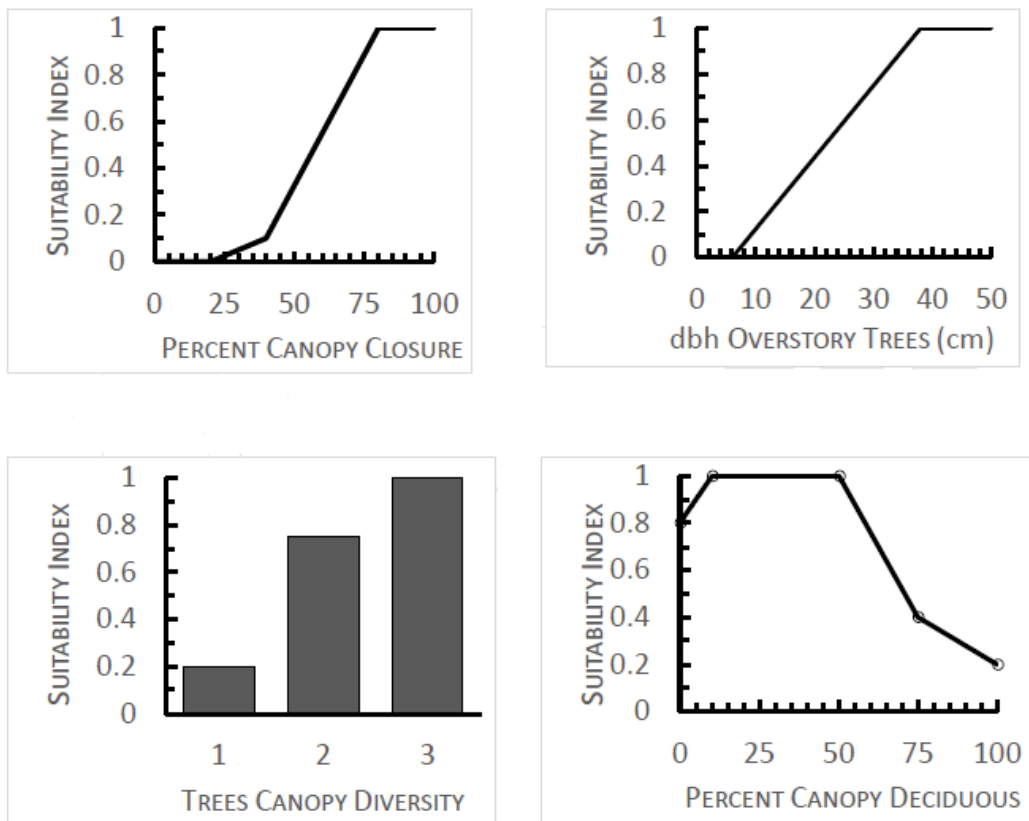


Figure 2. Distribution of habitat across Stirling as predicted by the Allen habitat suitability model.

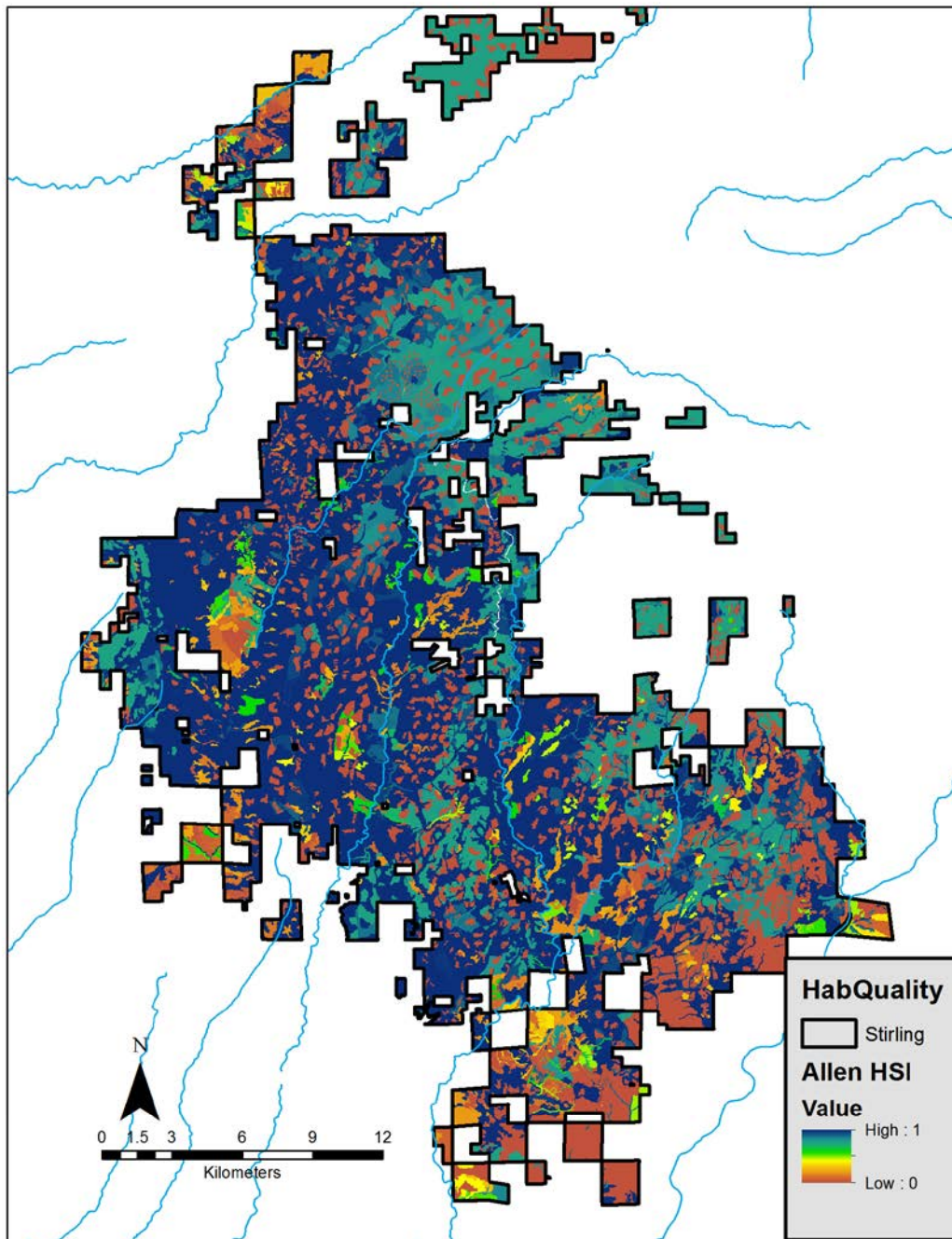


Figure 3. Distribution of habitat across Stirling and surrounding area as predicted by the Allen habitat suitability model (HSI) with underlying vegetation data taken from Gradient Nearest Neighbor (GNN) data (Lemma 2012).

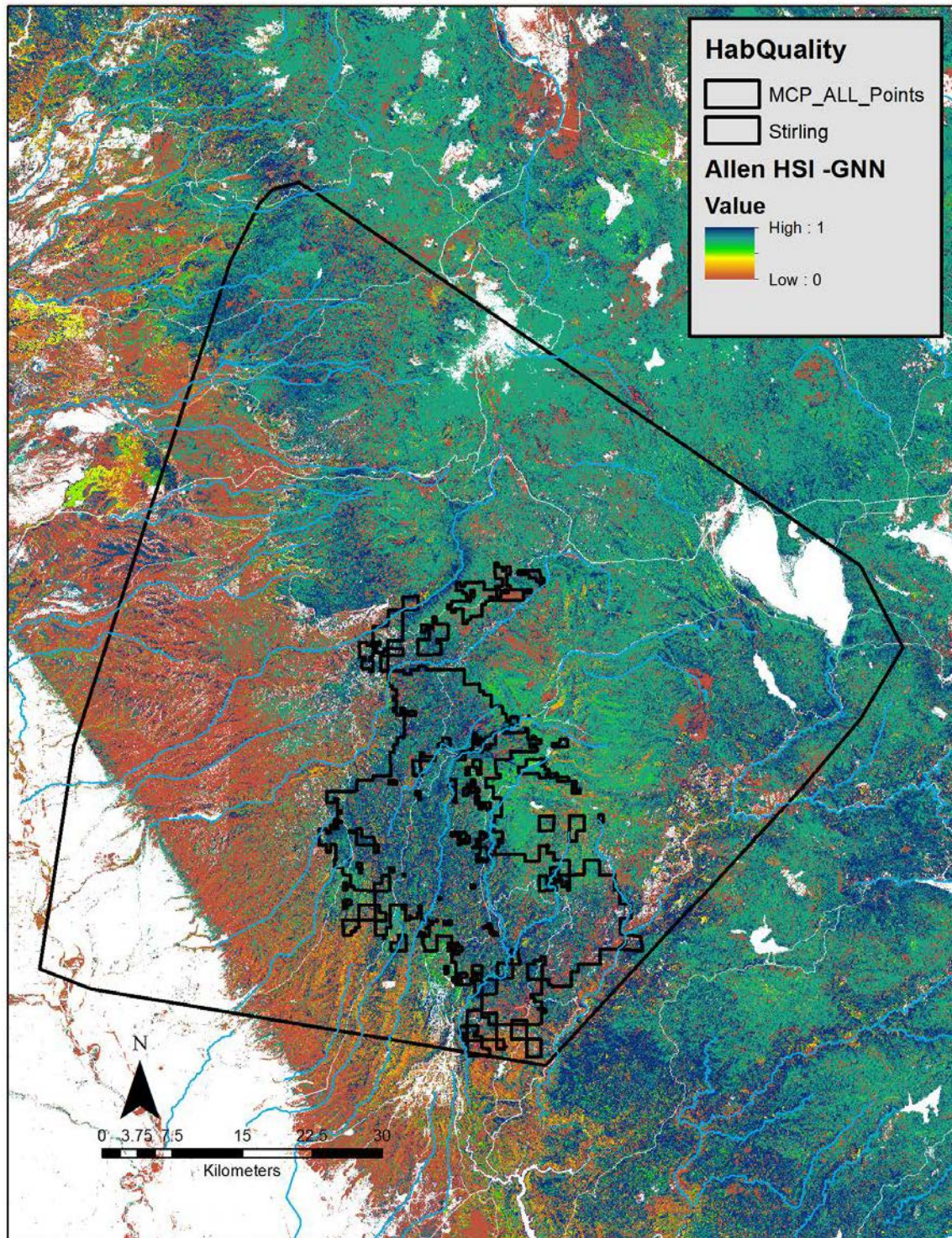


Figure 4. A) Distribution of habitat across Stirling implemented using parameters in logistic regression from Carroll et al. 2009. B) Distribution of habitat across Stirling from Carroll et al. 2009 after correction of stands with canopy < 10 and habitat quality > 0.90.

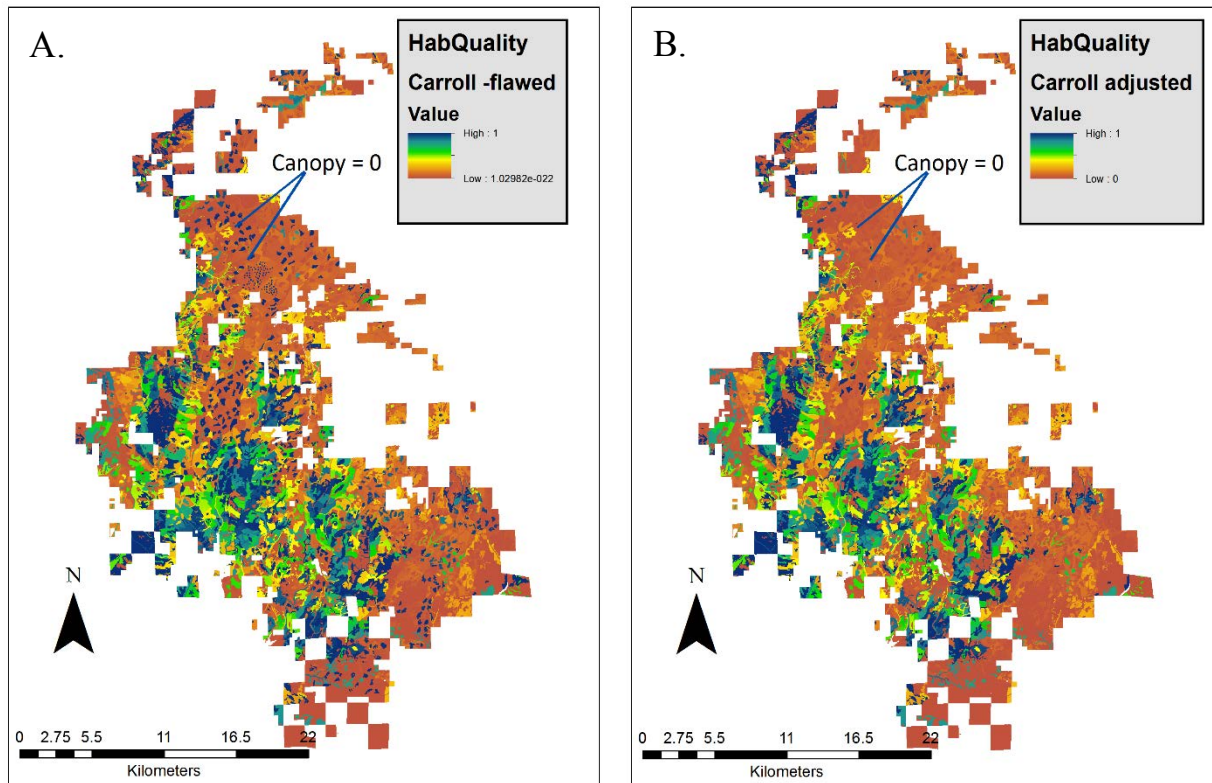


Figure 5. Screen capture from California Wildlife Habitat Relationships (CWHR) program 9.0 with habitat suitability information fishers in Sierran Mixed Conifer forests.

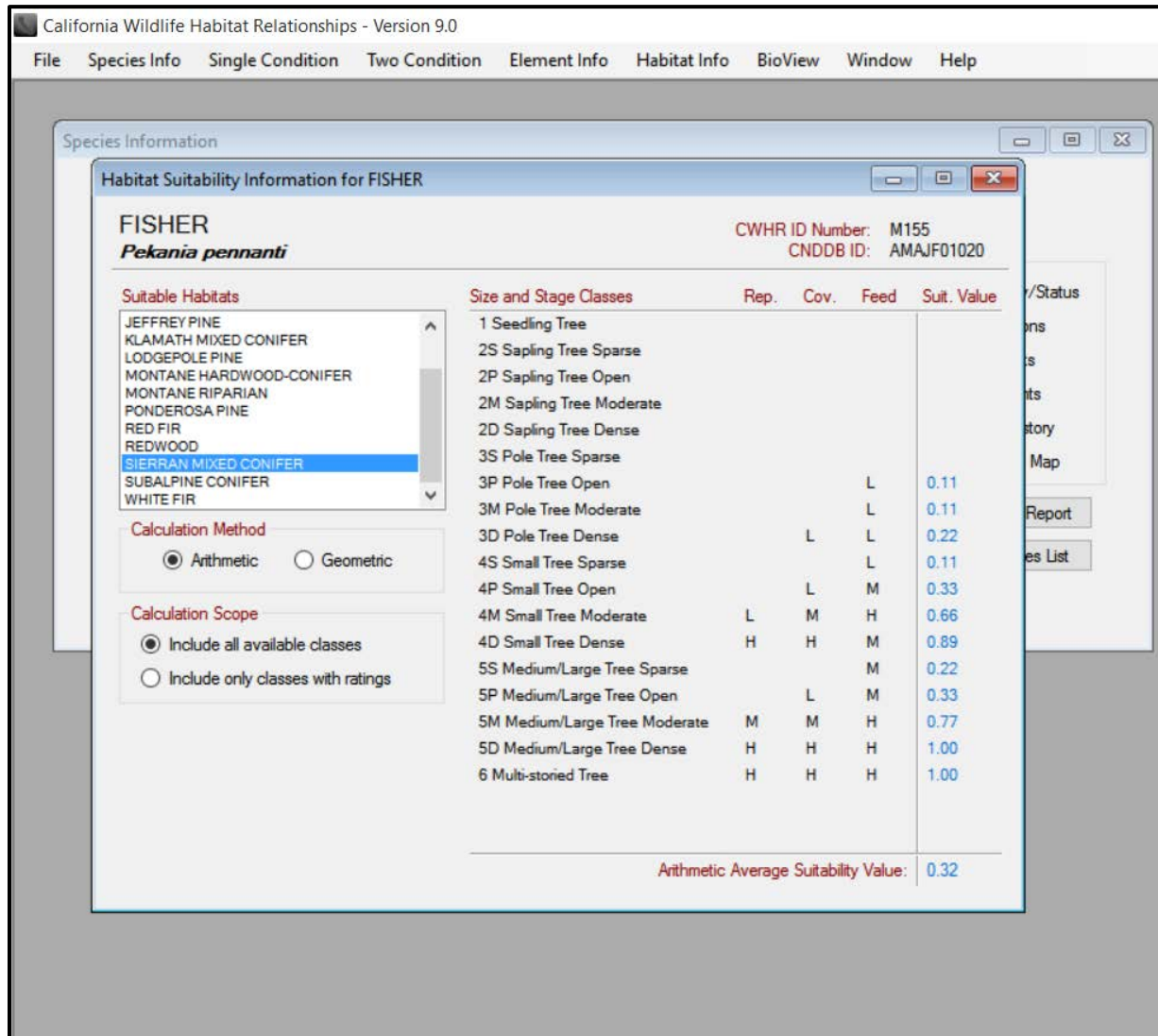


Figure 6. The number of locations for female and male fishers by year on and near Stirling study area in northern California.

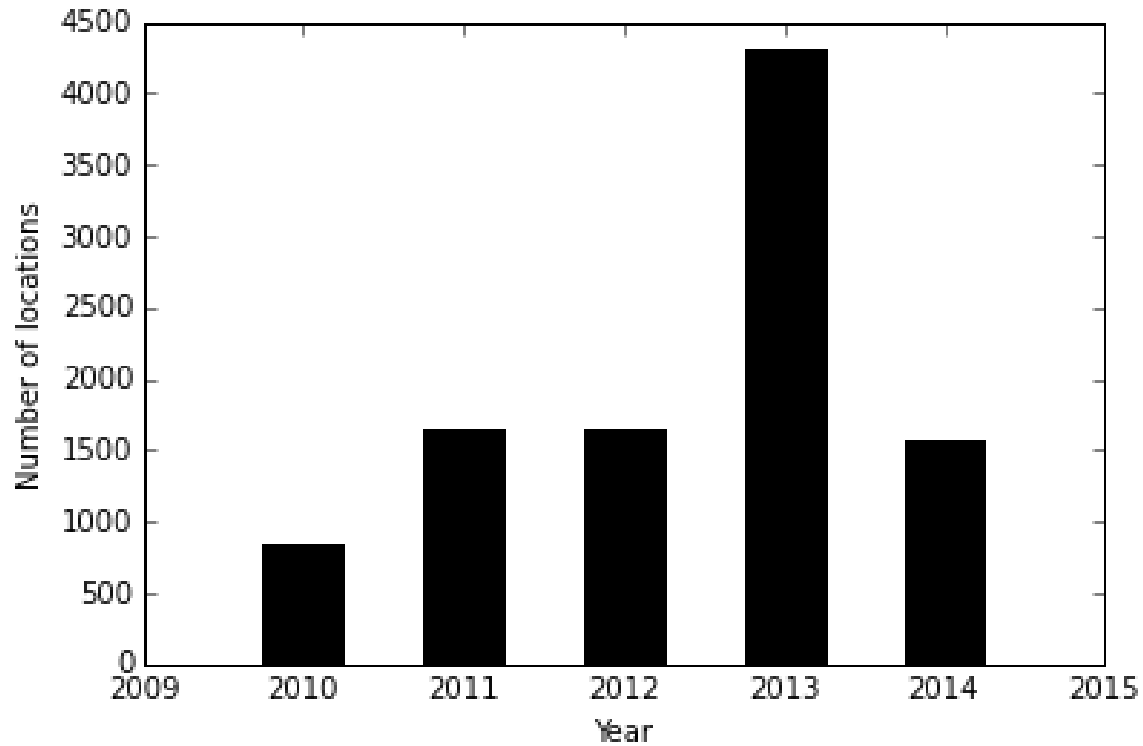


Figure 7. Distribution of available habitat as predicted by the Allen habitat suitability model for the Stirling district in northern California in study year 2010.

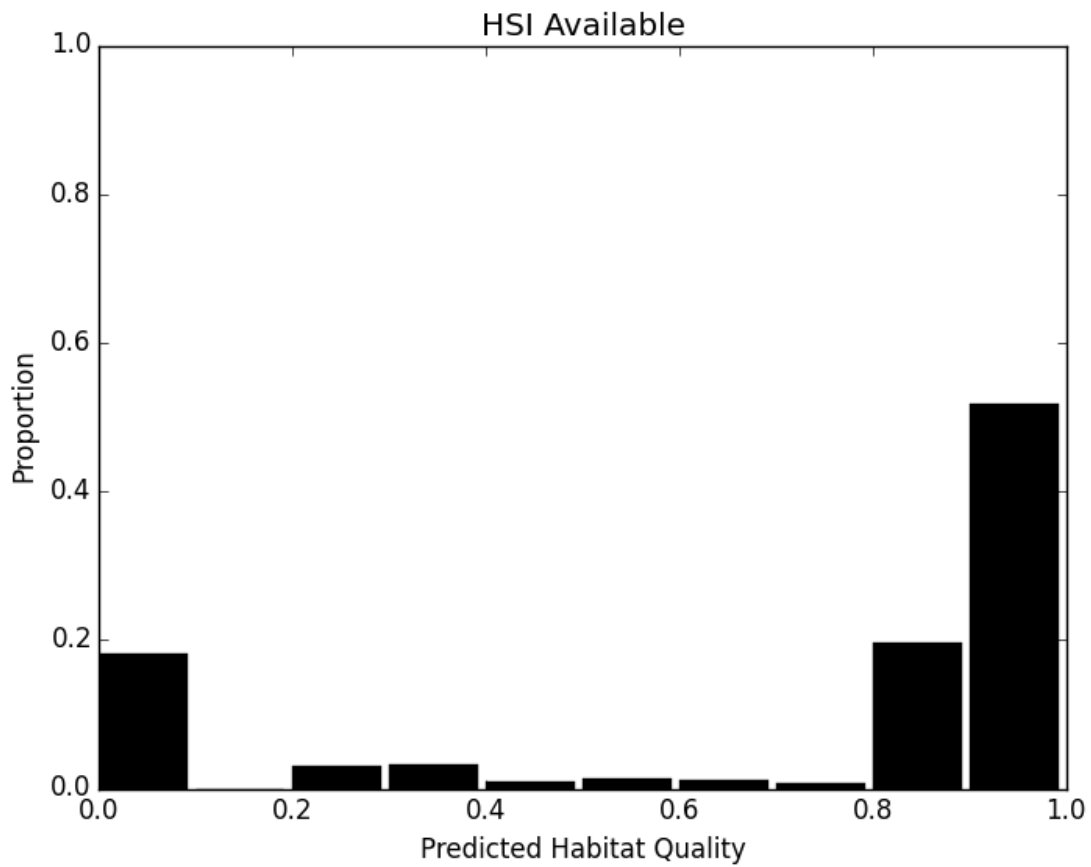


Figure 8. a Top) Mean Ivlev and modified-Ivlev resource selection functions for all fisher locations on Stirling using Allen's model and b Bottom) Mean Ivlev and modified-Ivlev resource selection functions for all fisher locations on Stirling and surrounding areas from 2010 to 2014 in northern California.

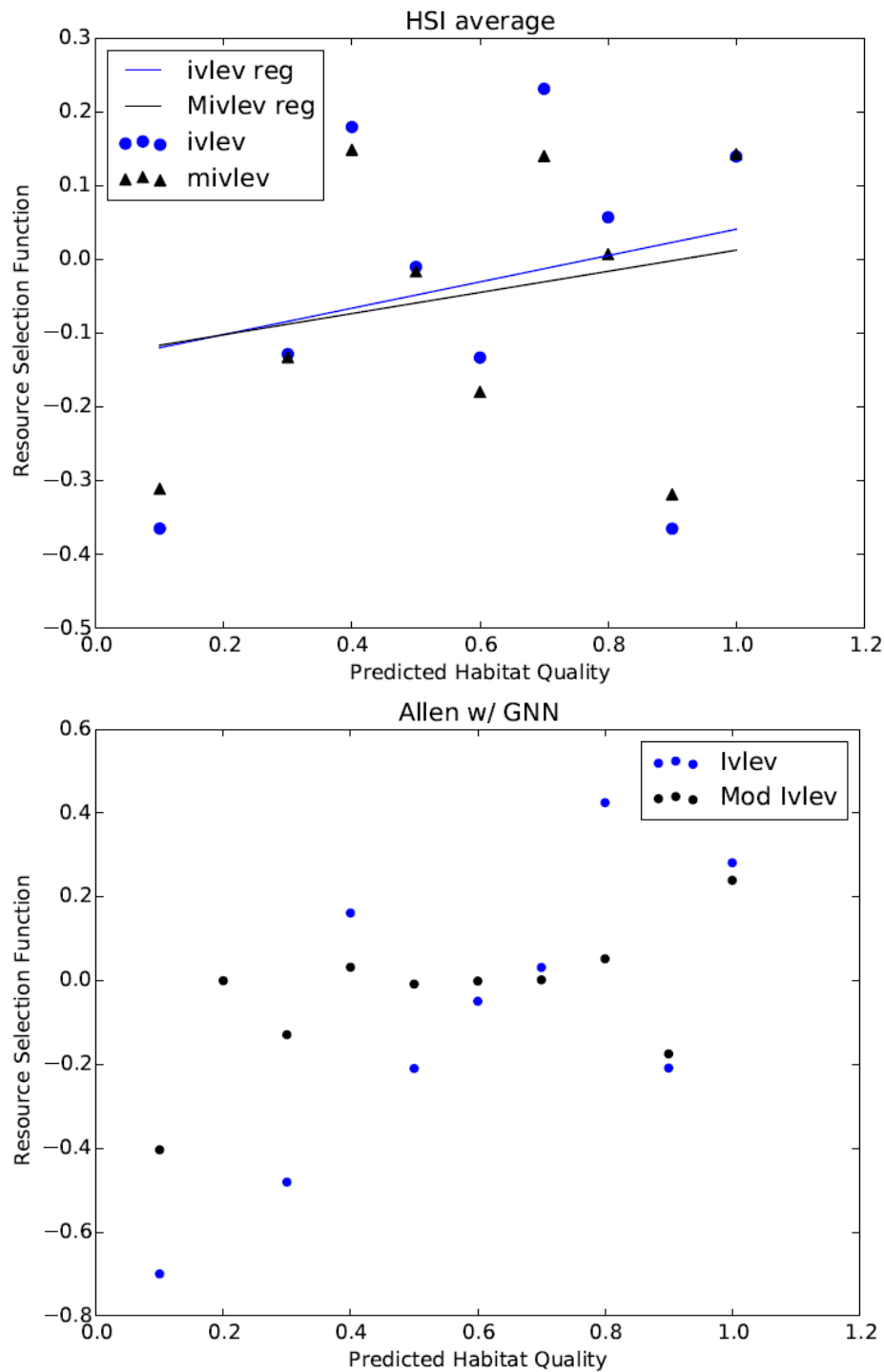


Figure 9. Mean modified-Ivlev resource selection function for male and female fisher locations on Stirling management area in Northern Sierra from 2010 to 2014.

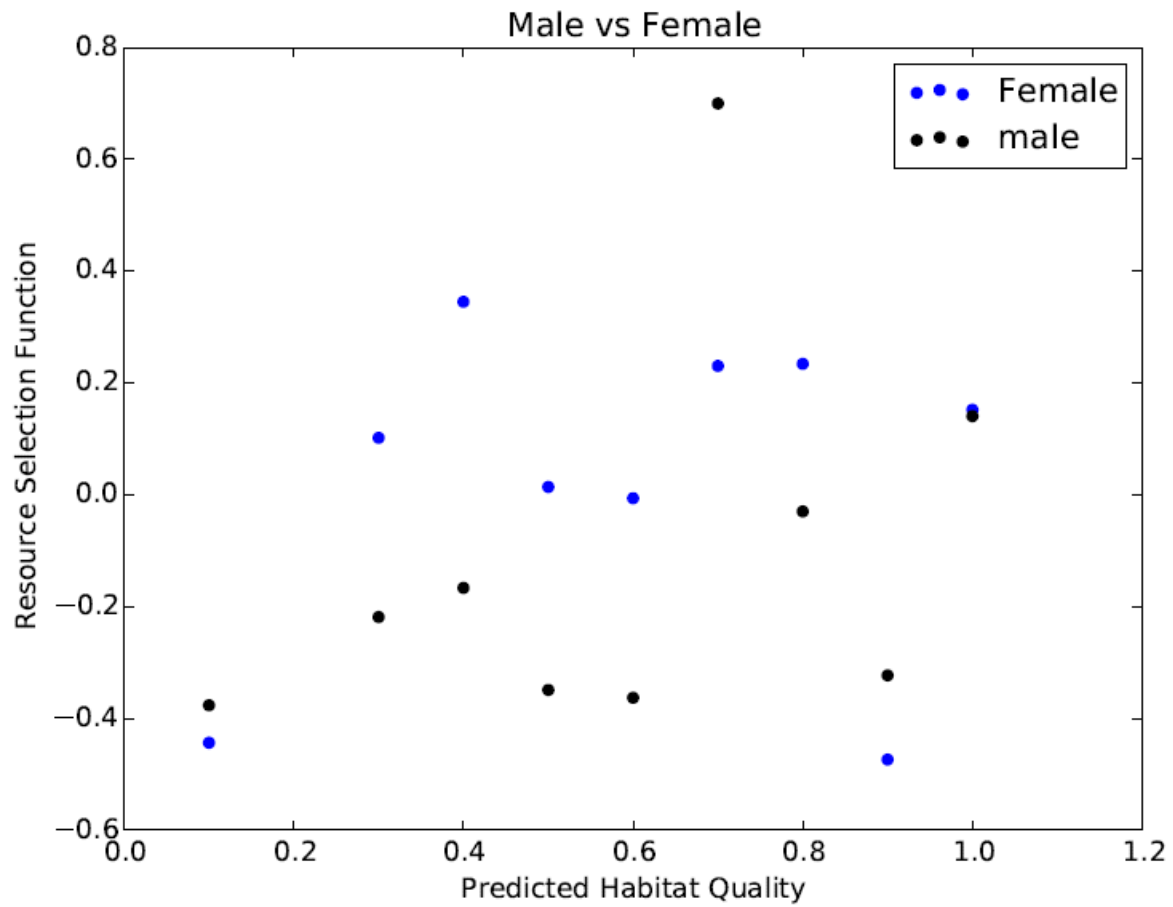


Fig. 10. A. Mean modified-Ivlev resource selections female habitat selection for Allen habitat model for all female within the 50% isopleth of their home ranges for years 2010 to 2013 (error bars = 1 SD). B. Mean modified-Ivlev resource selection function for all females and years within 50% isopleth of their home ranges for years 2010 to 2014 on Stirling district in northern California.

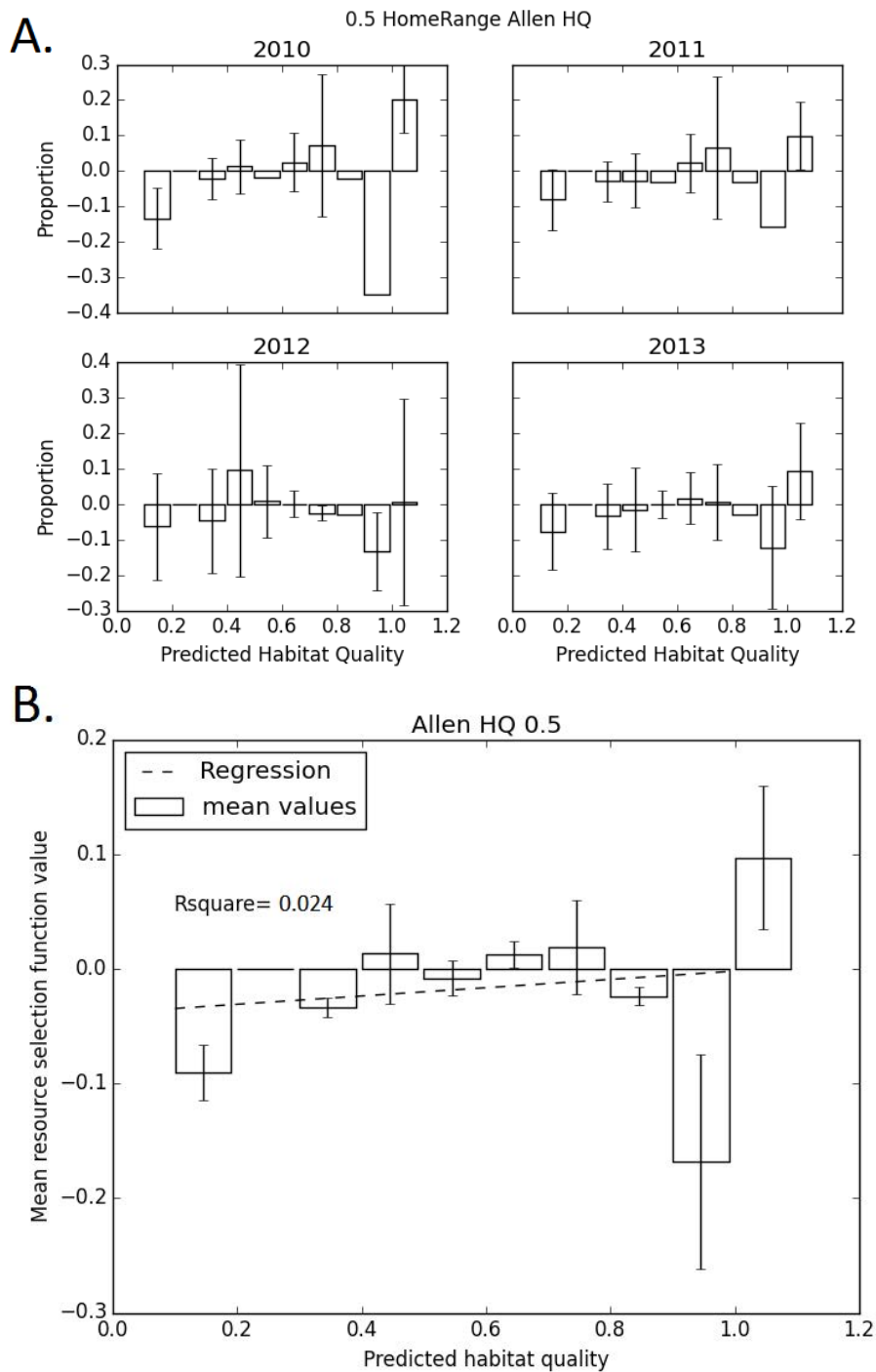


Fig 11. Proportion of available and used habitat as predicted by the Allen habitat suitability model and resultant modified-Ivlev and Ivlev resource selection functions for A. data from a female fisher who wore a GPS collar 5 days in fall of 2015 and B) data from a second female fisher who wore a GPS collar for 6 days in fall of 2015, both data sets collected in northern California on the Stirling district owned by SPI.

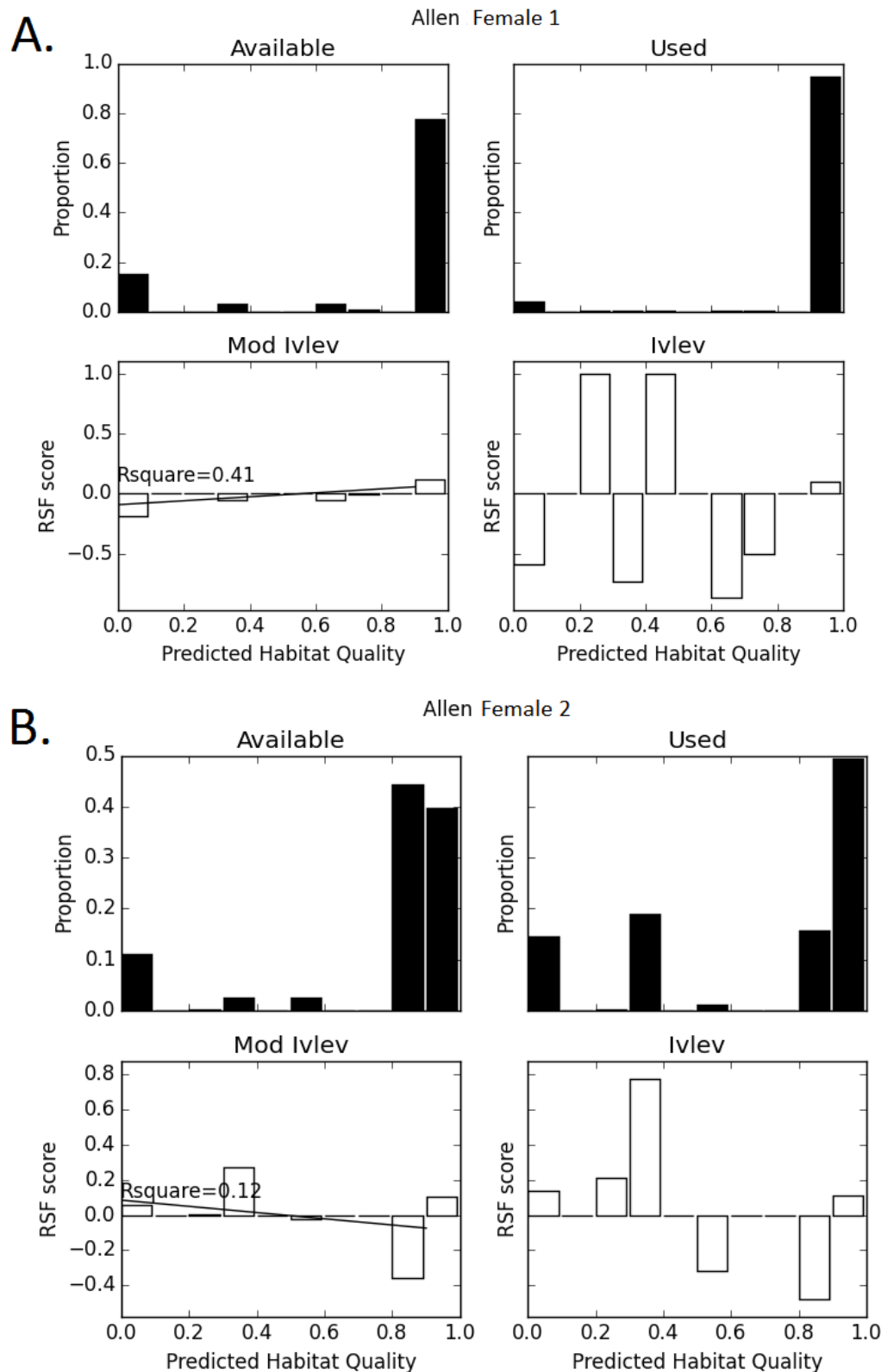


Figure 12. Estimated GPS locations (black dots) for a female fisher wearing an active collar for 6 days with collared polygons representing the year an area was logged. Areas with no color have not been clear-cut logged after 1950.

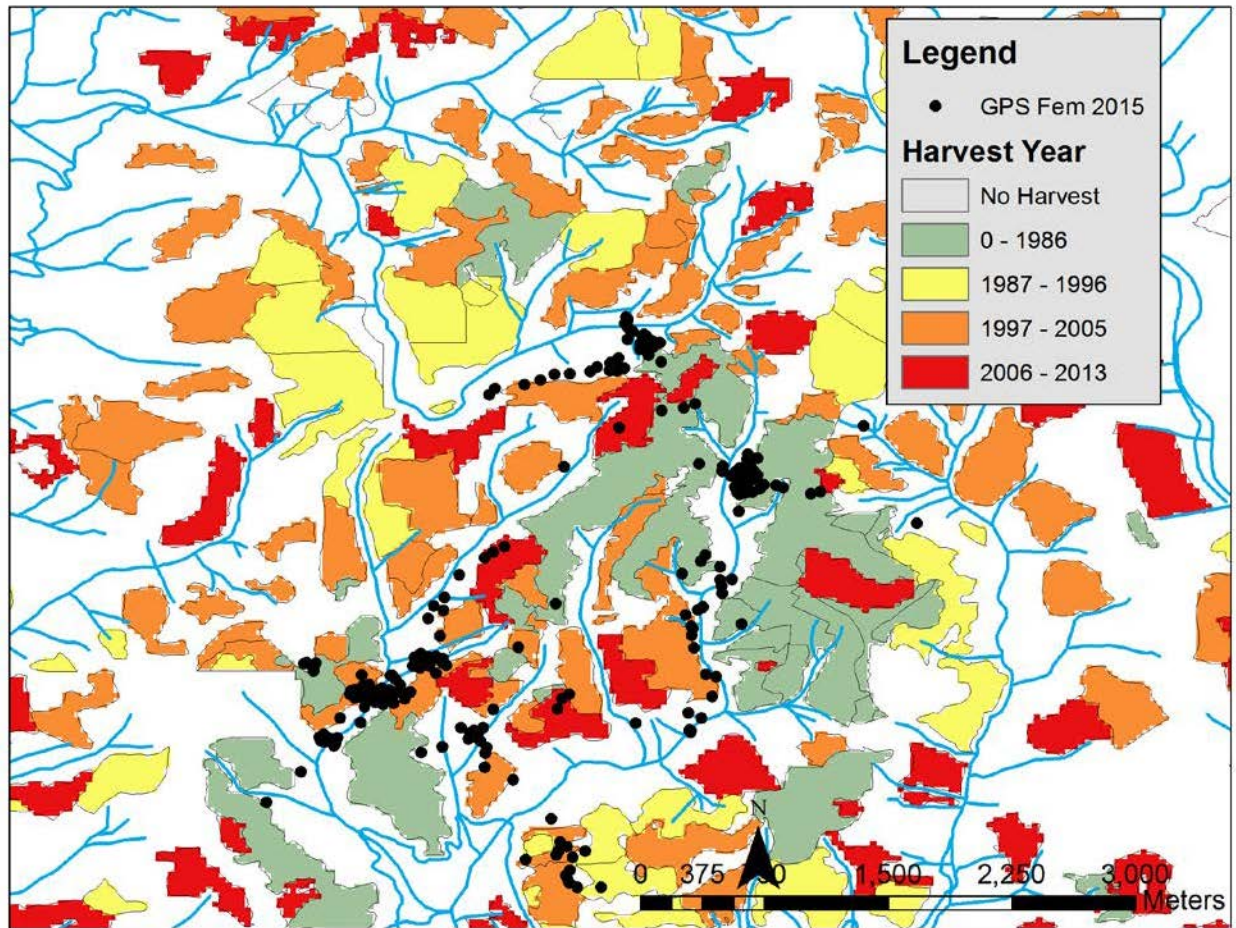


Figure 13. Mean modified-Ivlev resource selection functions for female fishers within their home ranges for year 2010 to 2014 for the Allen habitat suitability model (upper left), Carroll et al. 1999 fisher detection model (upper middle), Carroll index model (upper right), California Wildlife Habitat Relationships model (lower left), SPI habitat form model (lower middle) and a composite (mean) model from the others presented (lower right).

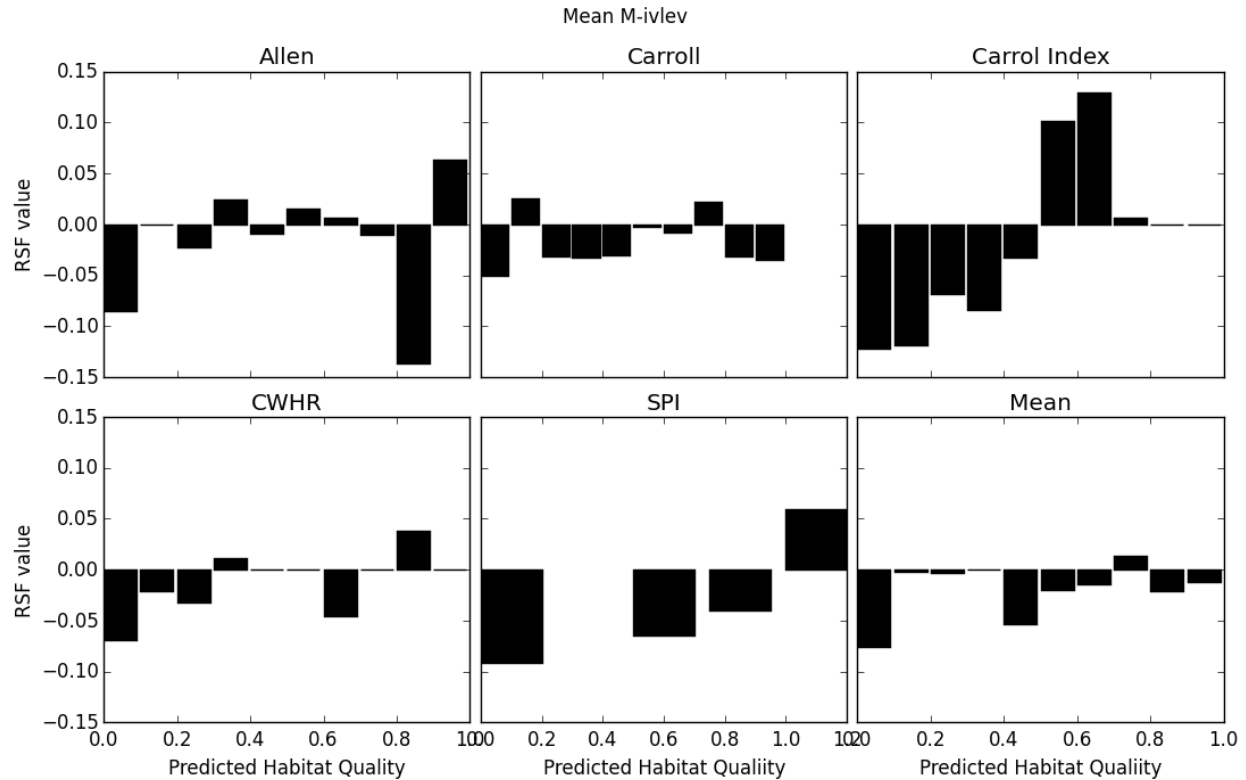


Figure 14. Distribution of fisher habitat as predicted by the Allen habitat suitability model classified into three discrete value ranges (<0.79, 0.80-0.90, and >0.90) for Stirling management district in northern California with A) no fisher locations and B) all fisher locations from 2010-2014 (red dots) overlaid on habitat distribution map.

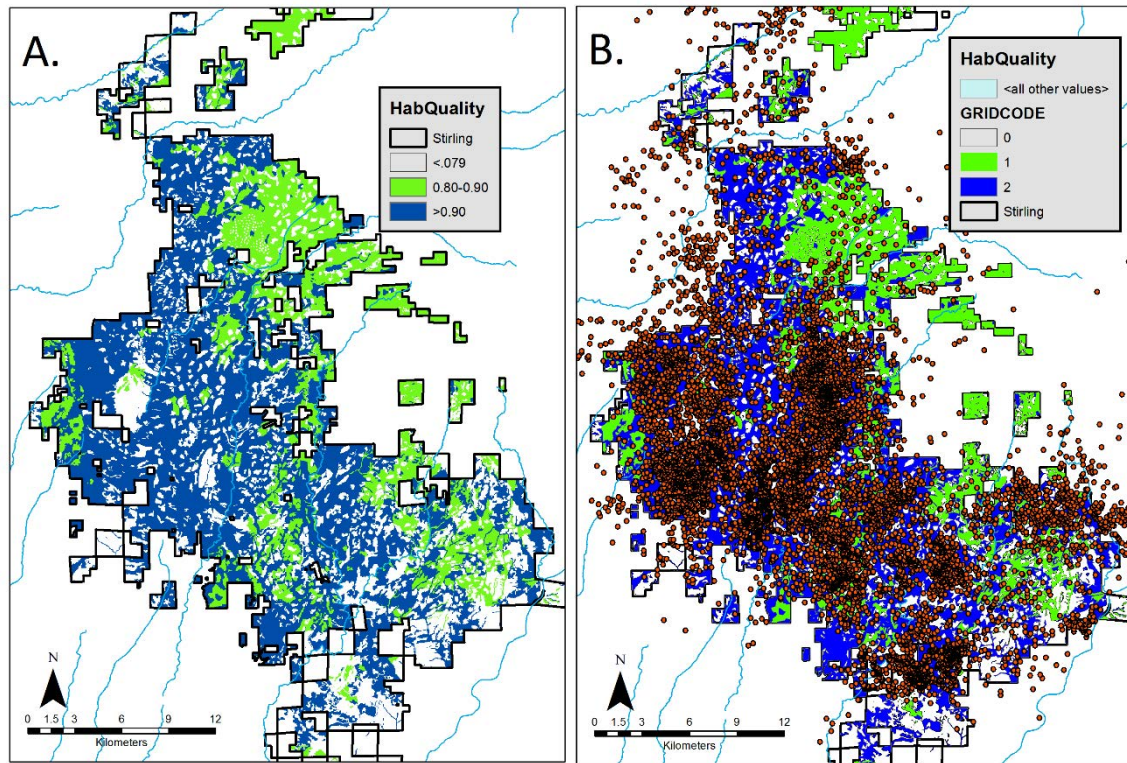


Figure 15. Distribution of available habitat as predicted by the Carroll et al. 1999 habitat suitability model for the Stirling district in northern California in study year 2010.

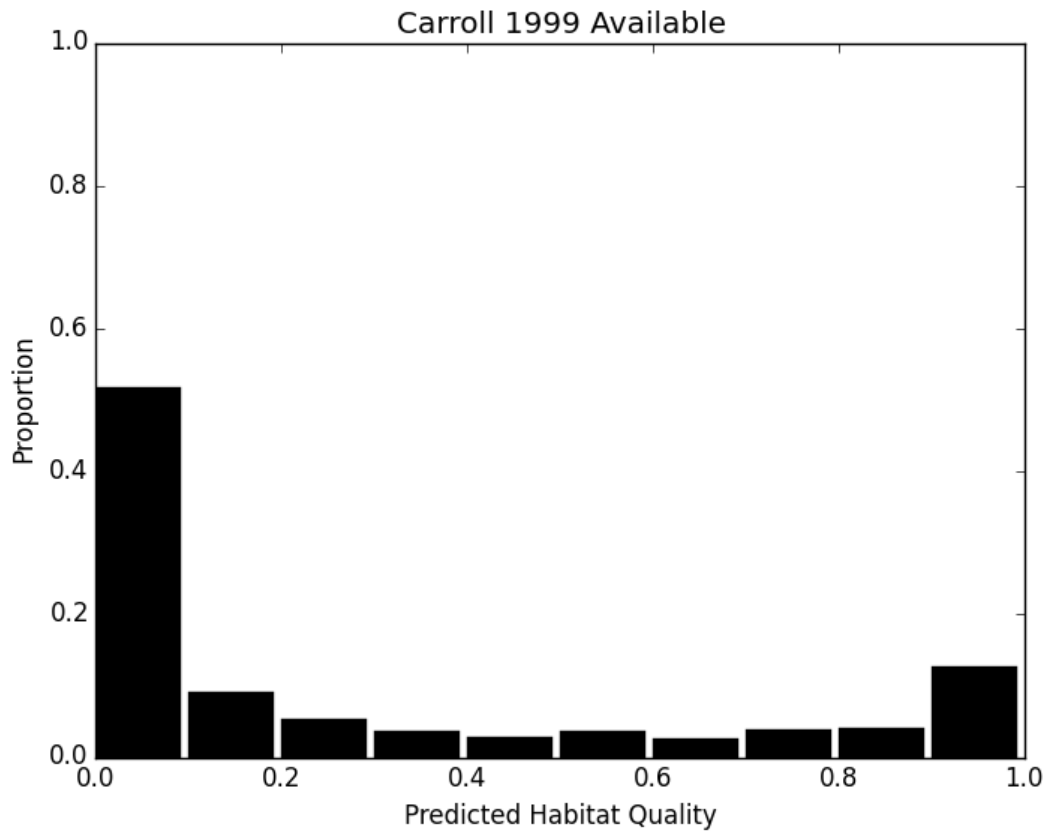


Figure 16. Mean modified-Ivlev resource selection functions for all fisher locations on Stirling from 2010 to 2014 in northern California.

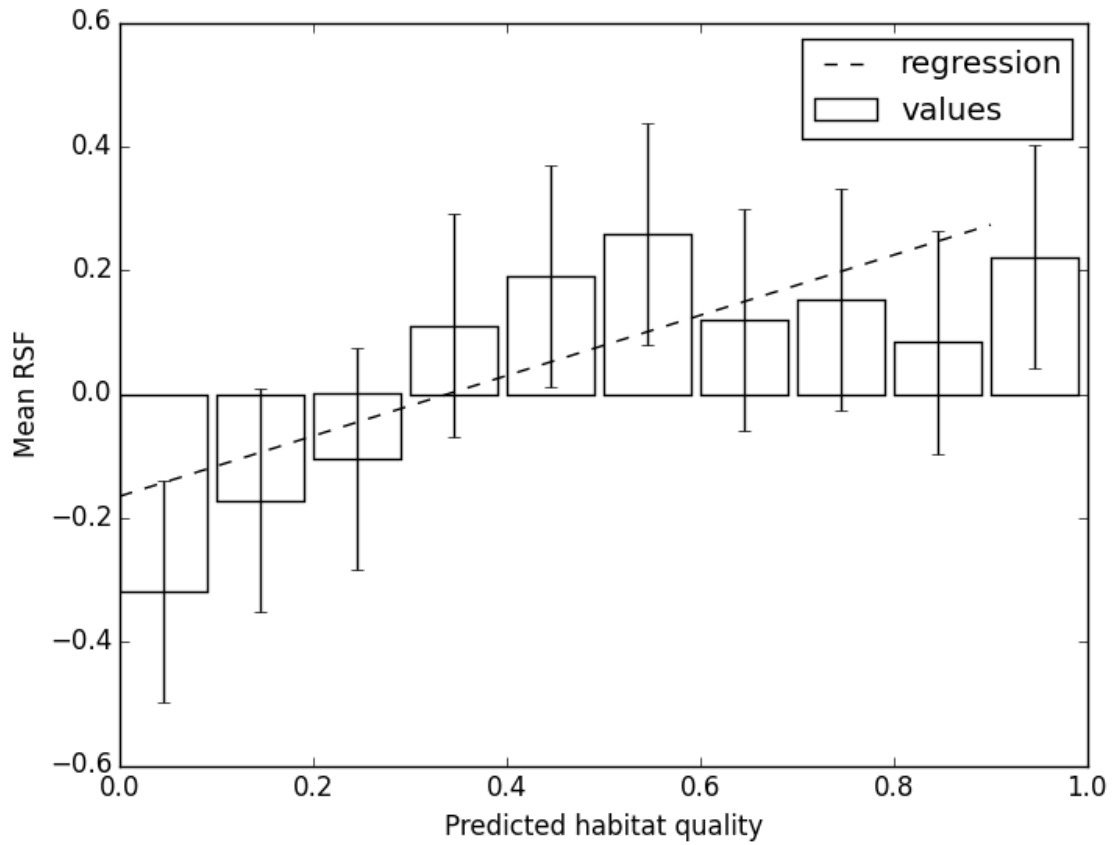


Figure 17. Mean modified-Ivlev resource selection functions for female habitat selection as predicted by the Carroll et al. 1999 model for all female within the 50% isopleth of their home ranges for years 2010 to 2013 (error bars = 1 SD). B. Mean modified-Ivlev resource selection function for all females and years within 50% isopleth of their home ranges for years 2010 to 2014 on Stirling district in northern California.

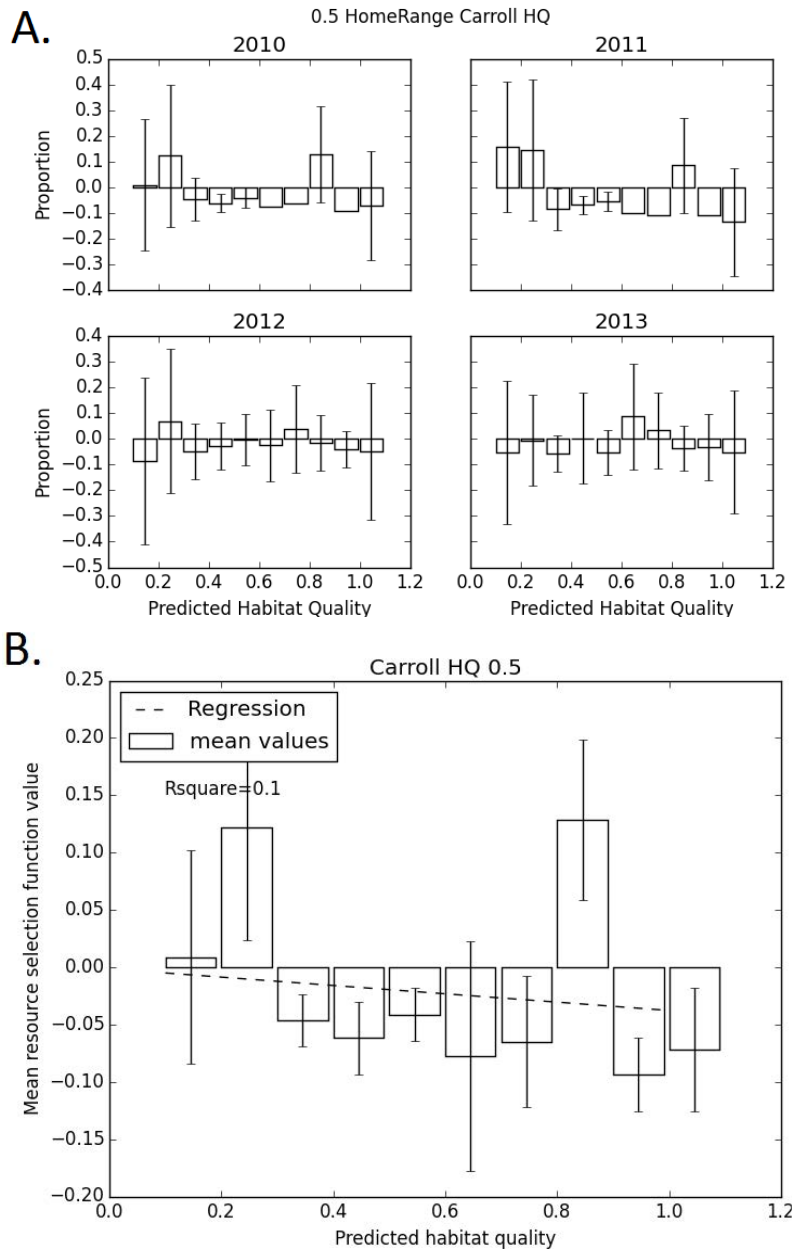


Figure 18. Proportion of available and used habitat as predicted by the Carroll et al. 1999 fisher habitat model and the resultant modified-Ivlev and Ivlev resource selection functions for A. data from a female fisher who wore a GPS collared 5 days in fall of 2015 and B) data from a second female fisher who wore a GPS collar for 6 days in fall of 2015, both data sets collected in northern California on the Stirling district owned by SPI.

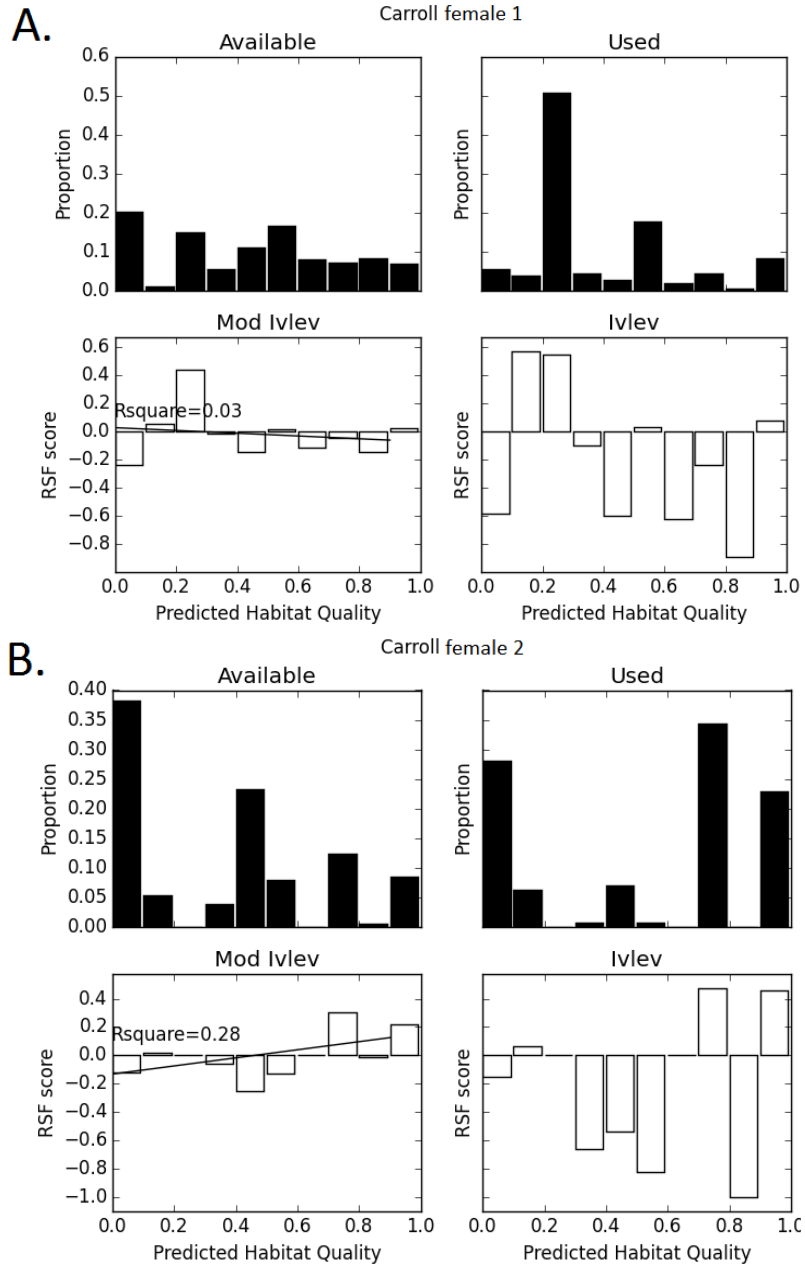


Figure 19. Distribution of available habitat as predicted by the Carroll index model for fisher habitat for the Stirling district in northern California in study year 2010.

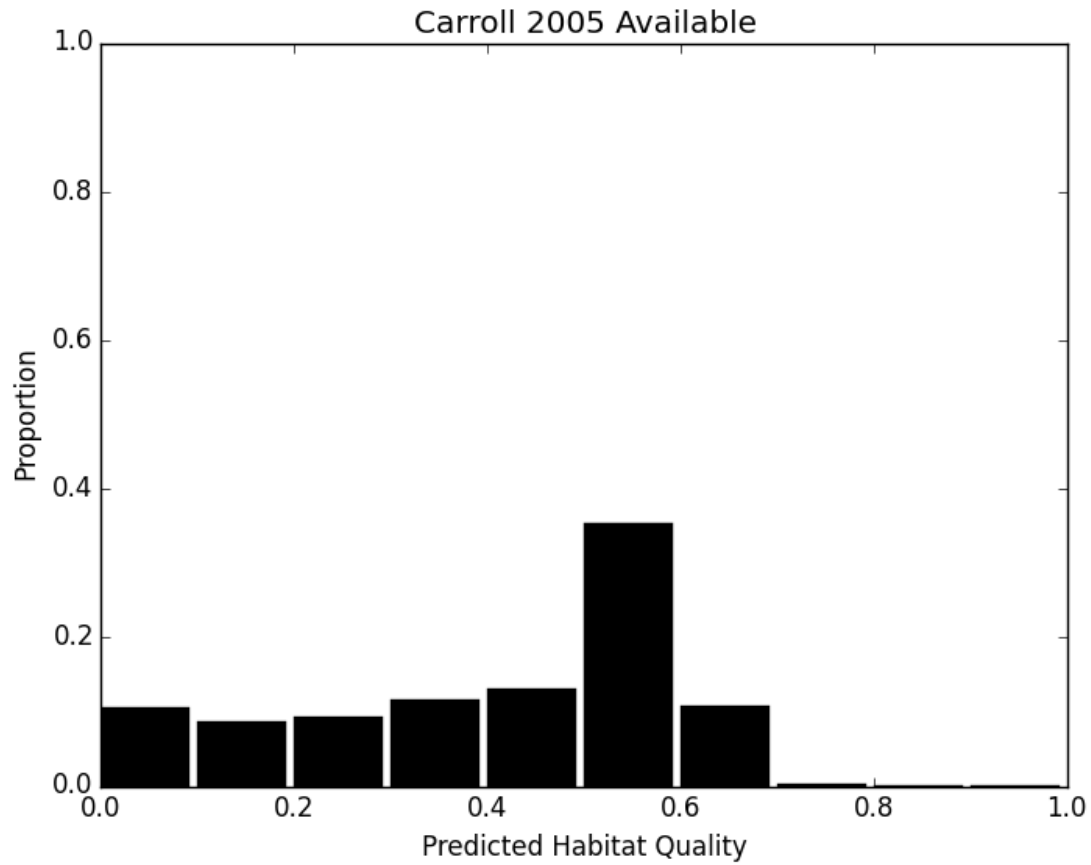


Figure 20. Resource selection functions based on the Carroll Index fisher habitat model for Top) Mean Ivlev and modified-Ivlev resource selection functions for all fisher locations on Stirling and Bottom) Mean Ivlev and modified-Ivlev resource selection functions for all fisher locations on Stirling and surrounding areas from 2010 to 2014 in northern California.

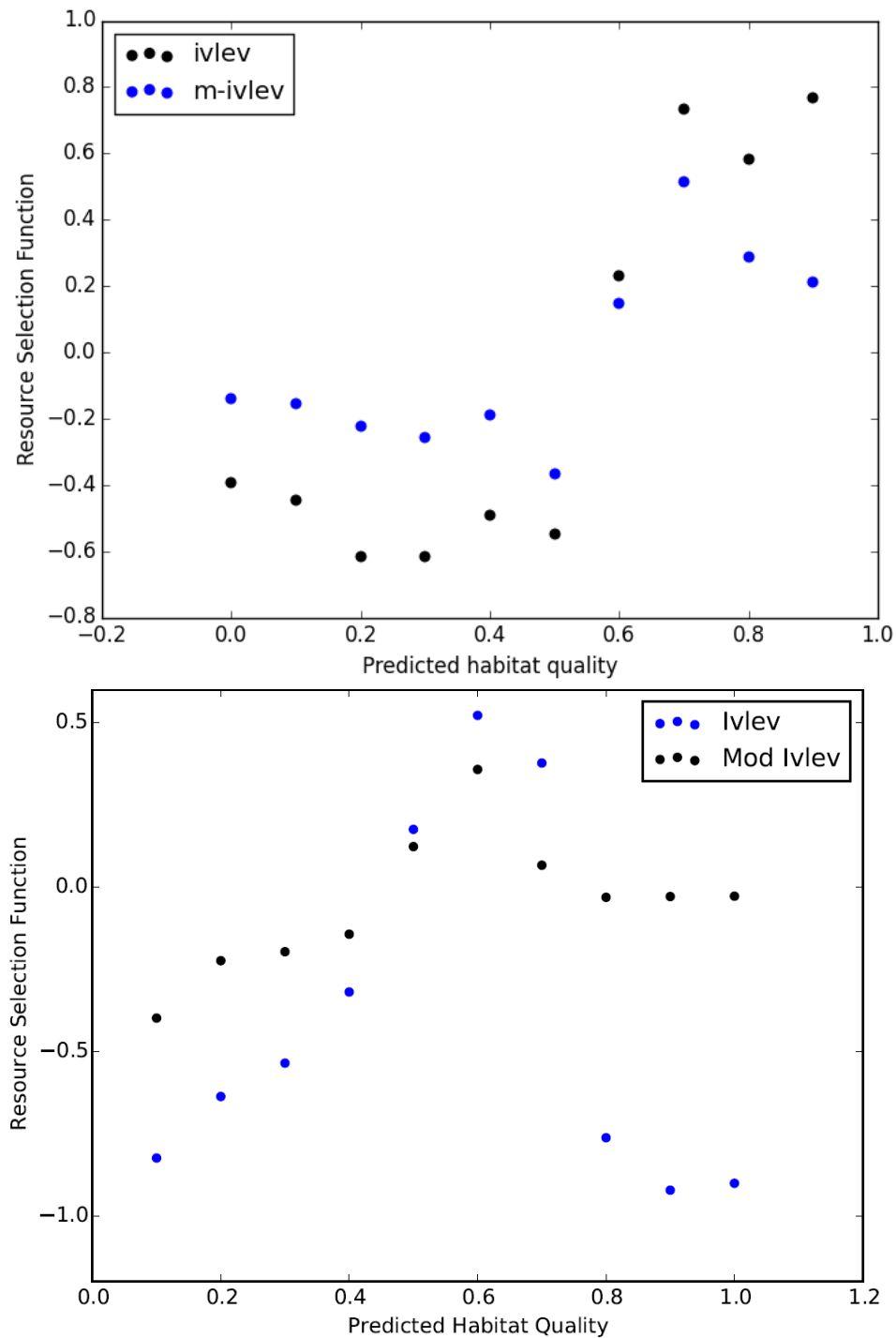


Figure 21. A. Mean modified-Ivlev resource selections female habitat selection for the Carroll index habitat model for all female within the 50% isopleth of their home ranges for years 2010 to 2013 (error bars = 1 SD). B. Mean modified-Ivlev resource selection function for all females and years within 50% isopleth of their home ranges for years 2010 to 2014 on Stirling district in northern California.

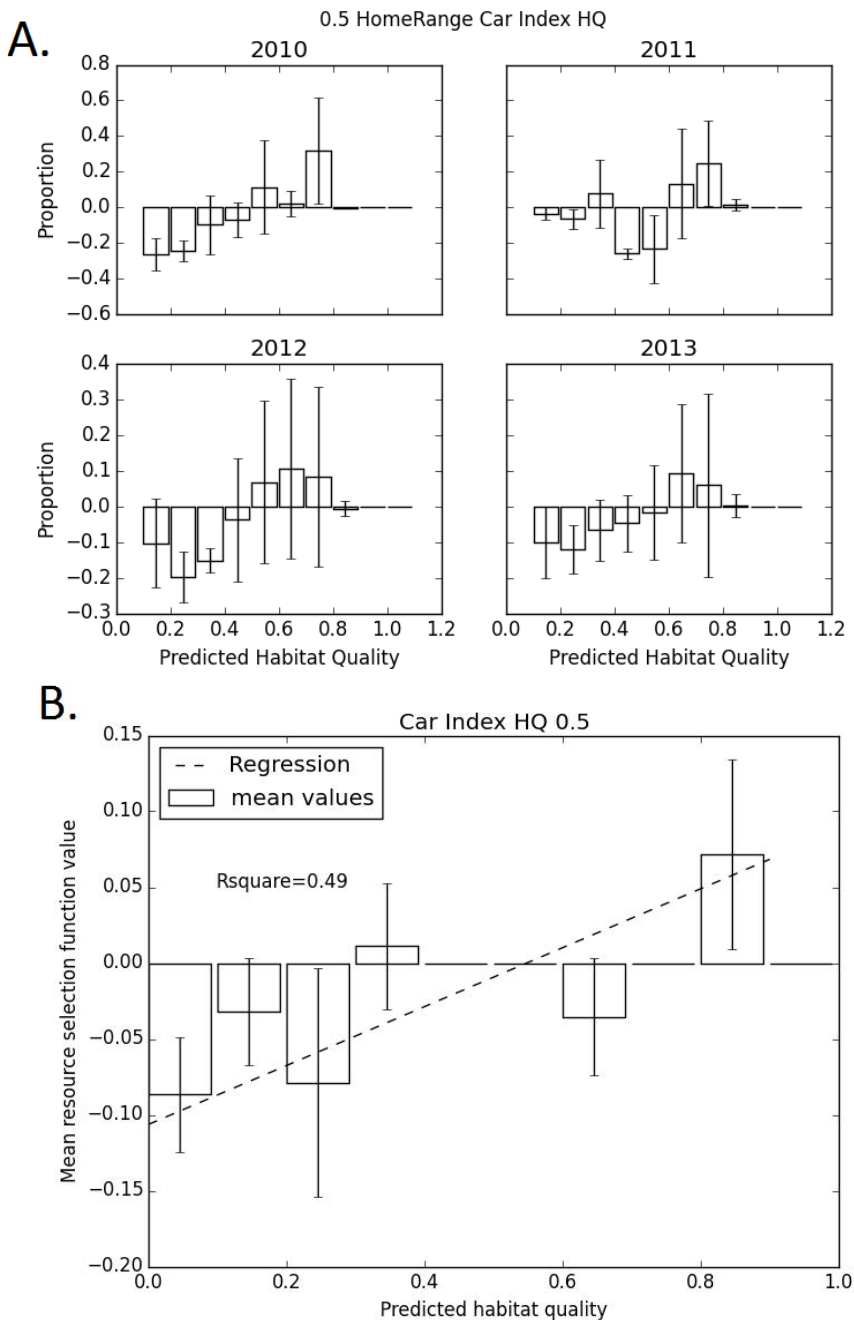


Figure 22. Distribution of mean available habitats as predicted by the Carroll Index fisher habitat model across years \pm SD for years from 2010 to 2014 on Stirling in northern California.

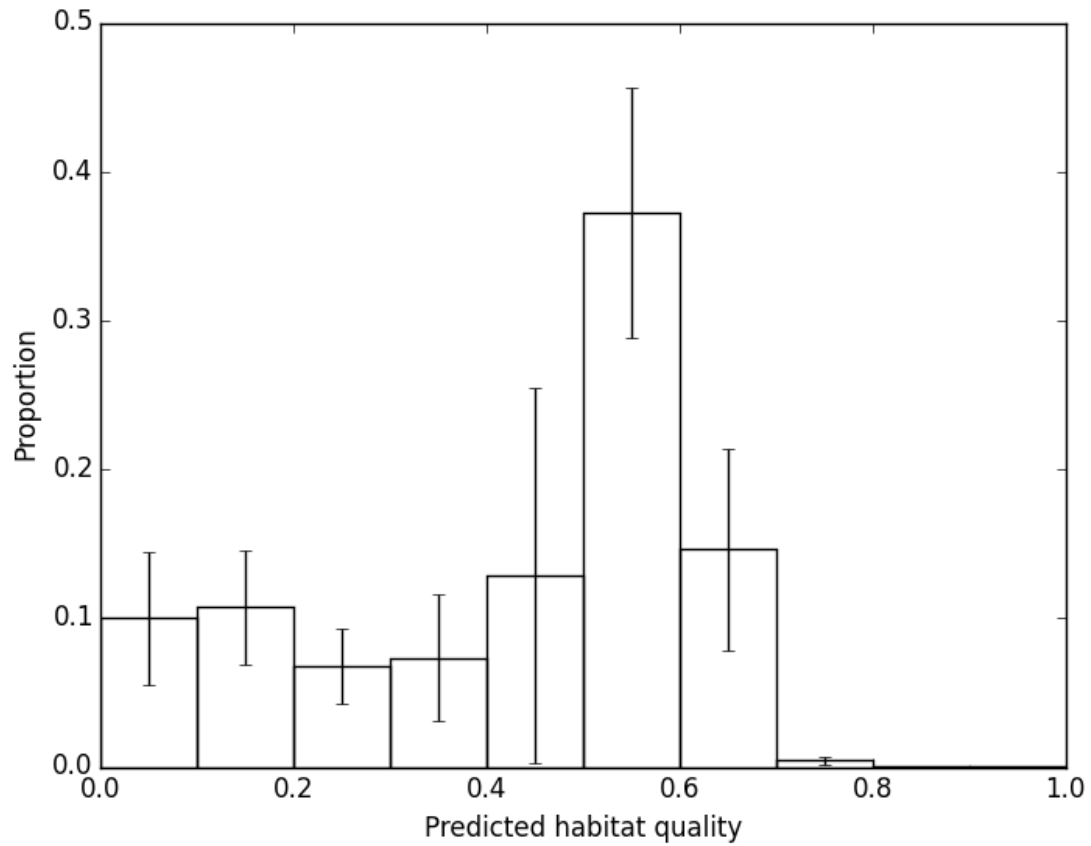


Figure 23. Proportion of available and used habitat as predicted by the Carrol index fisher habitat model and resultant modified-Ivlev and Ivlev resource selection functions for A. data from a female fisher who wore a GPS collar 5 days in fall of 2015 and B) data from a second female fisher who wore a GPS collar for 6 days in fall of 2015, both data sets collected in northern California on the Stirling district owned by SPI.

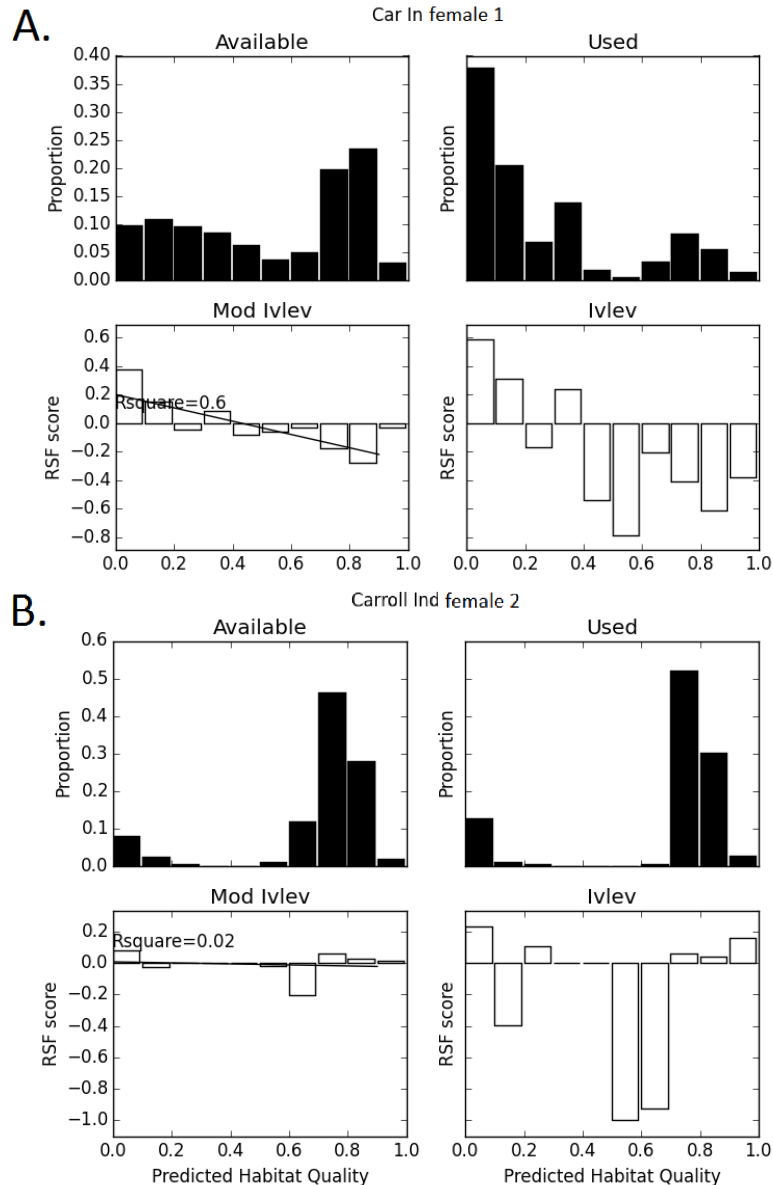


Figure 24. Distribution of available habitat as predicted by the CWHR habitat suitability model for the Stirling district in northern California in study year 2010.

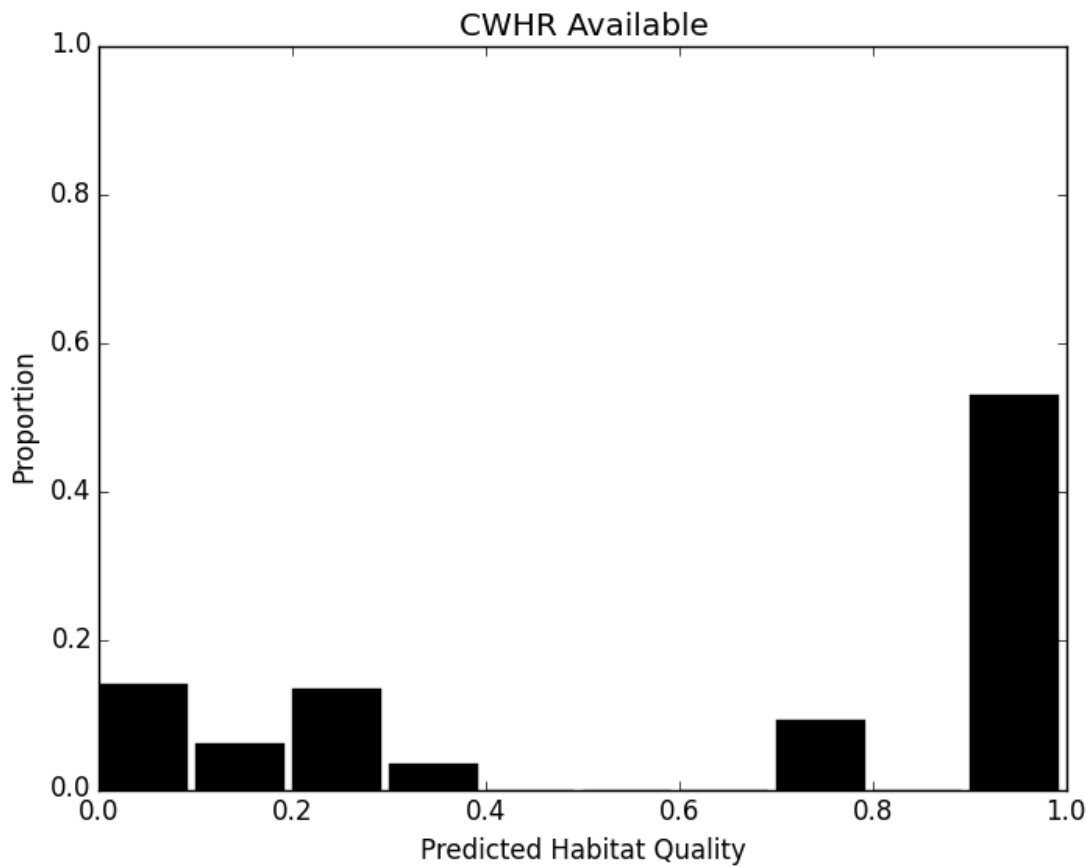


Figure 25. The average Ivlev and modified-Ivlev resource selection functions based on the CWHR fisher habitat model values for all fisher locations on Stirling and surrounding areas from 2010 to 2014 in northern California

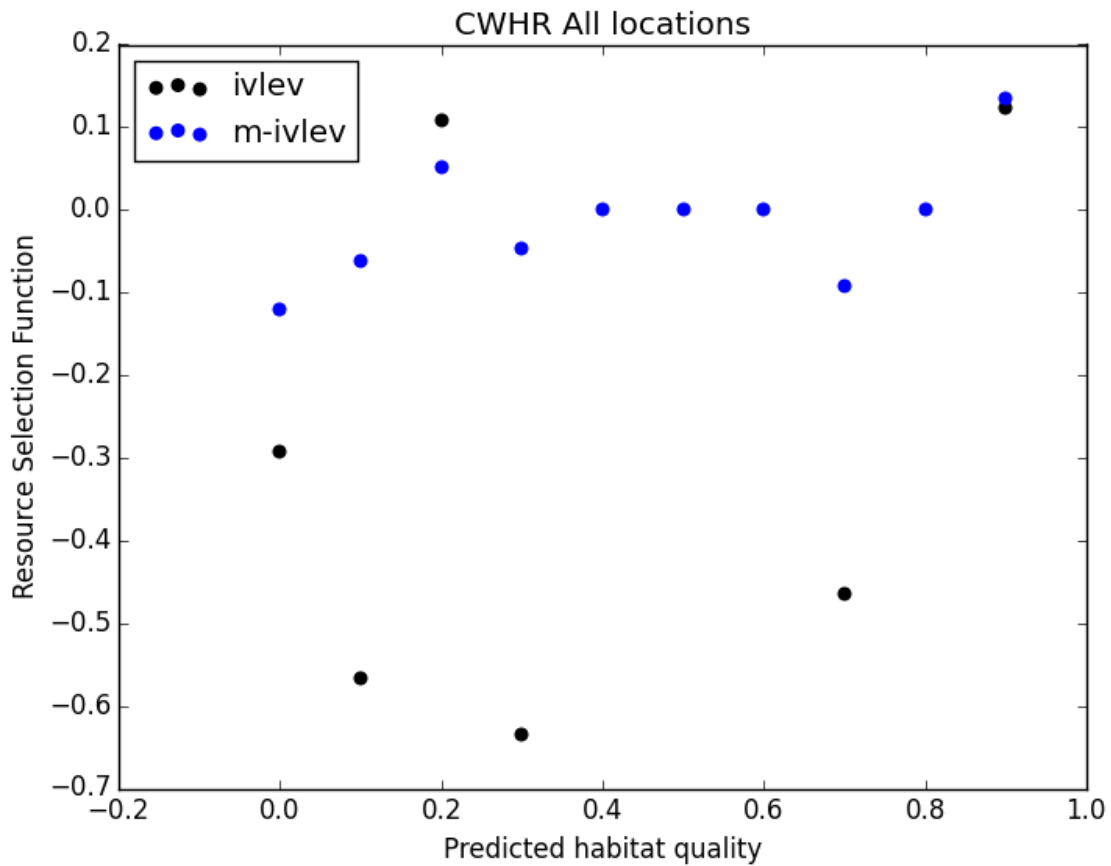


Figure 26. Mean modified-Ivlev resource selections female habitat selection for CWHR fisher habitat model for all female fishers within the 50% isopleth of their home ranges for years 2010 to 2013 (error bars = 1 SD). B. Mean modified-Ivlev resource selection function for all females and years within 50% isopleth of their home ranges for years 2010 to 2014 on Stirling district in northern California.

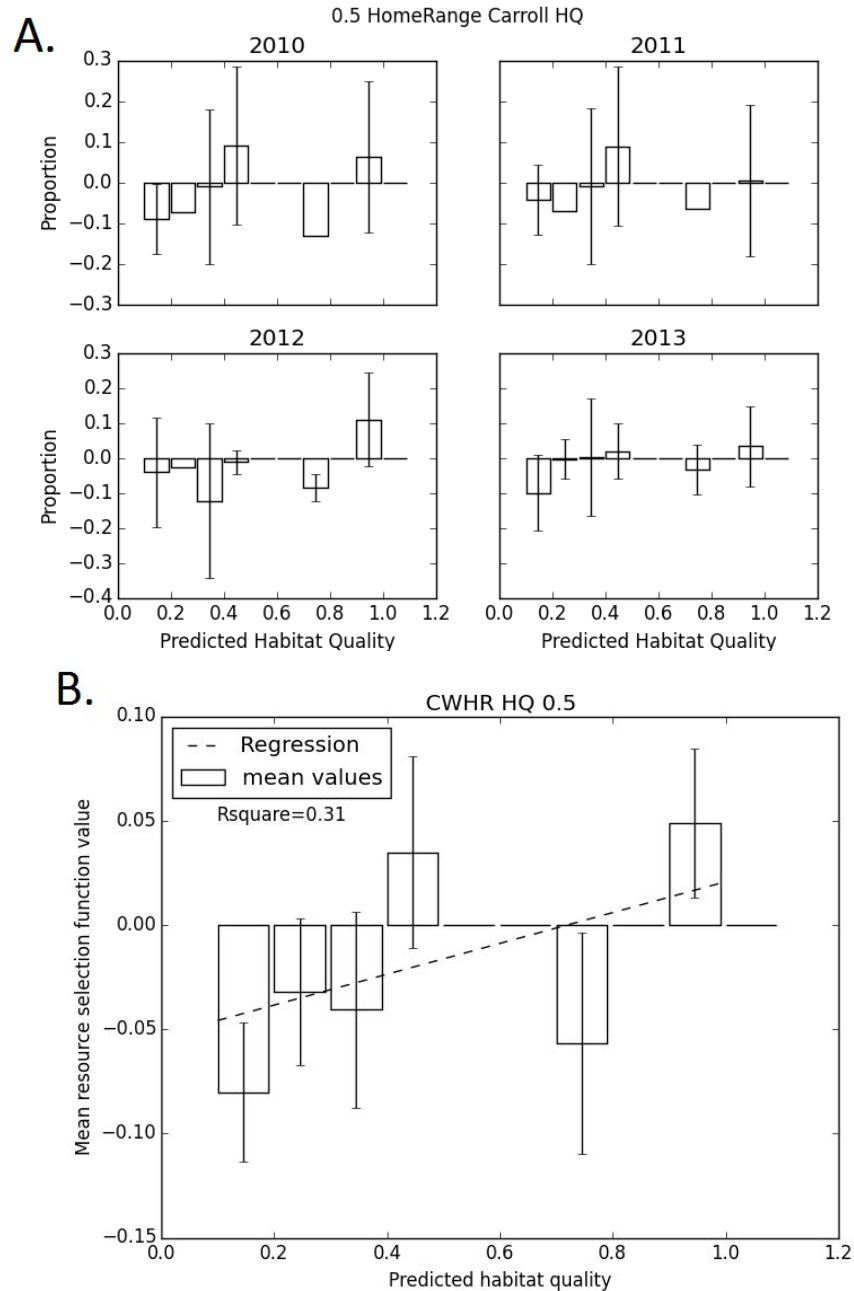


Figure 27. Proportion of available and used habitat as predicted by the CWHR fisher habitat suitability model and resultant modified-Ivlev and Ivlev resource selection functions for A. data from a female fisher who wore a GPS collar 5 days in fall of 2015 and B) data from a second female fisher who wore a GPS collar for 6 days in fall of 2015, both data sets collected in northern California on the Stirling district owned by SPI.

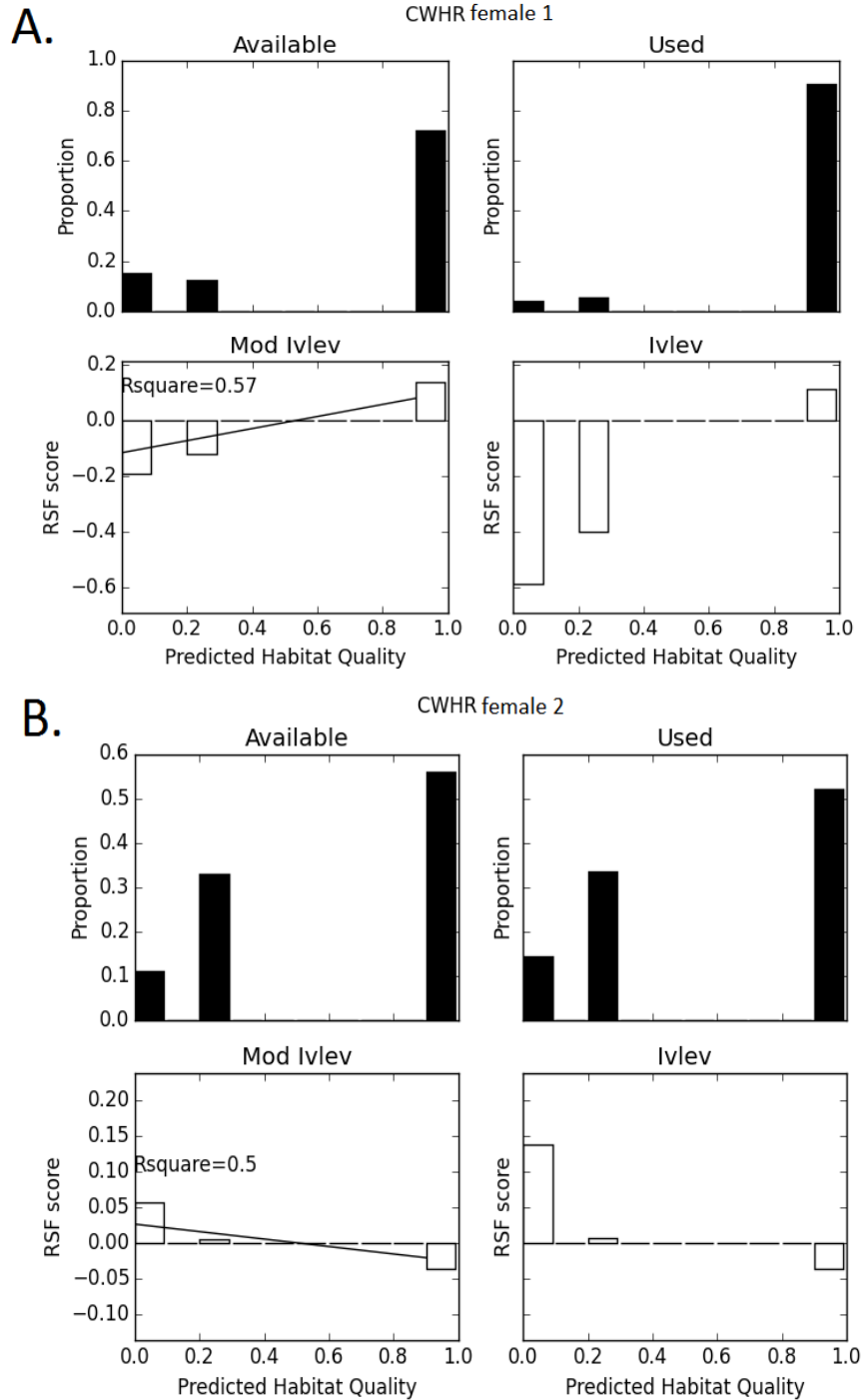


Figure 28. Areas on Stirling showing overlap between the Allen model (values 0.8-0.9 and the CWHR model (values 0.6-0.7) in green (other areas in black) in 2011 with the 50% isopleths for females in 2011, 2012, and 2013.

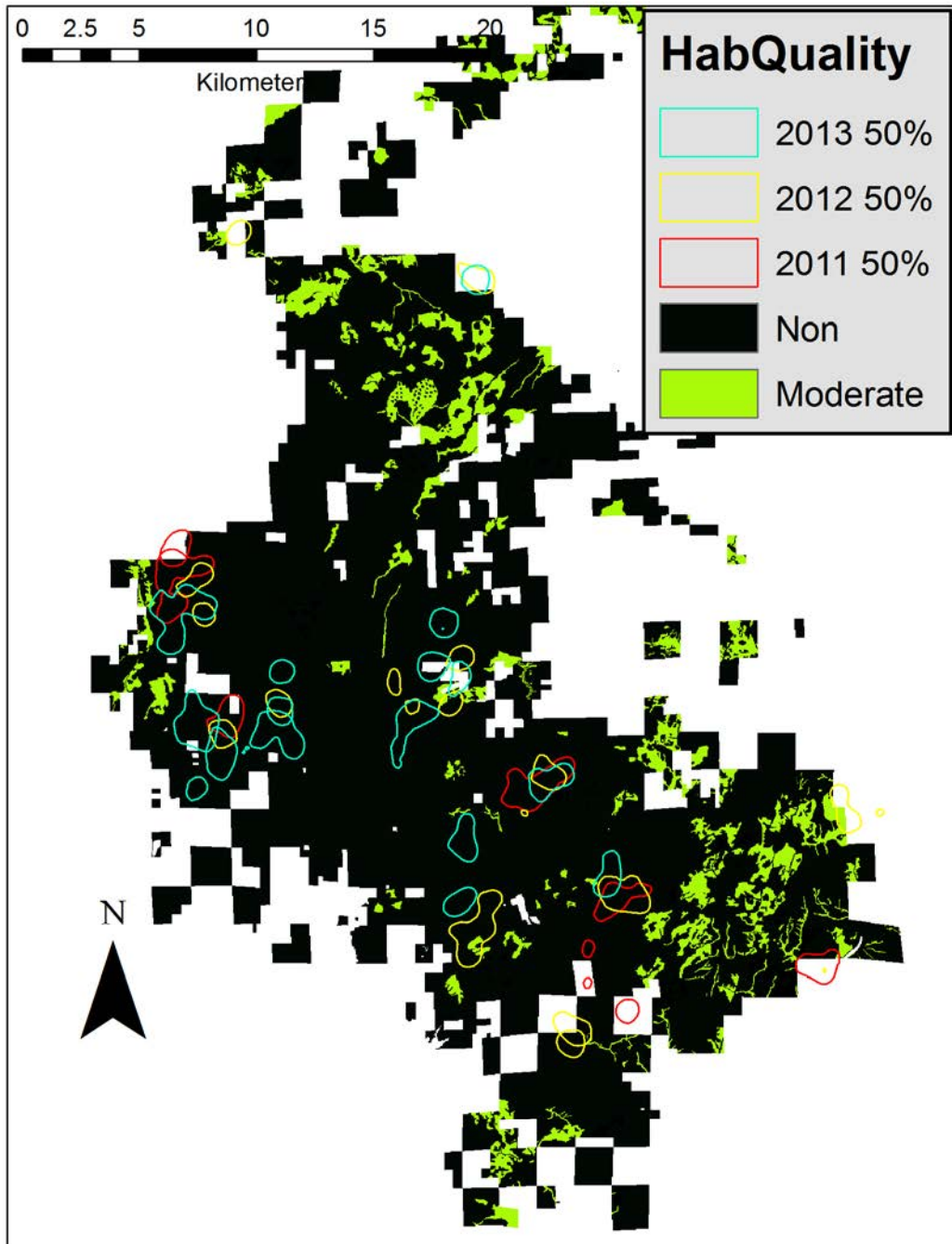


Figure 29. Distribution of available habitat as predicted by the SPI habitat form model for fishers on the Stirling district in northern California in study year 2010.

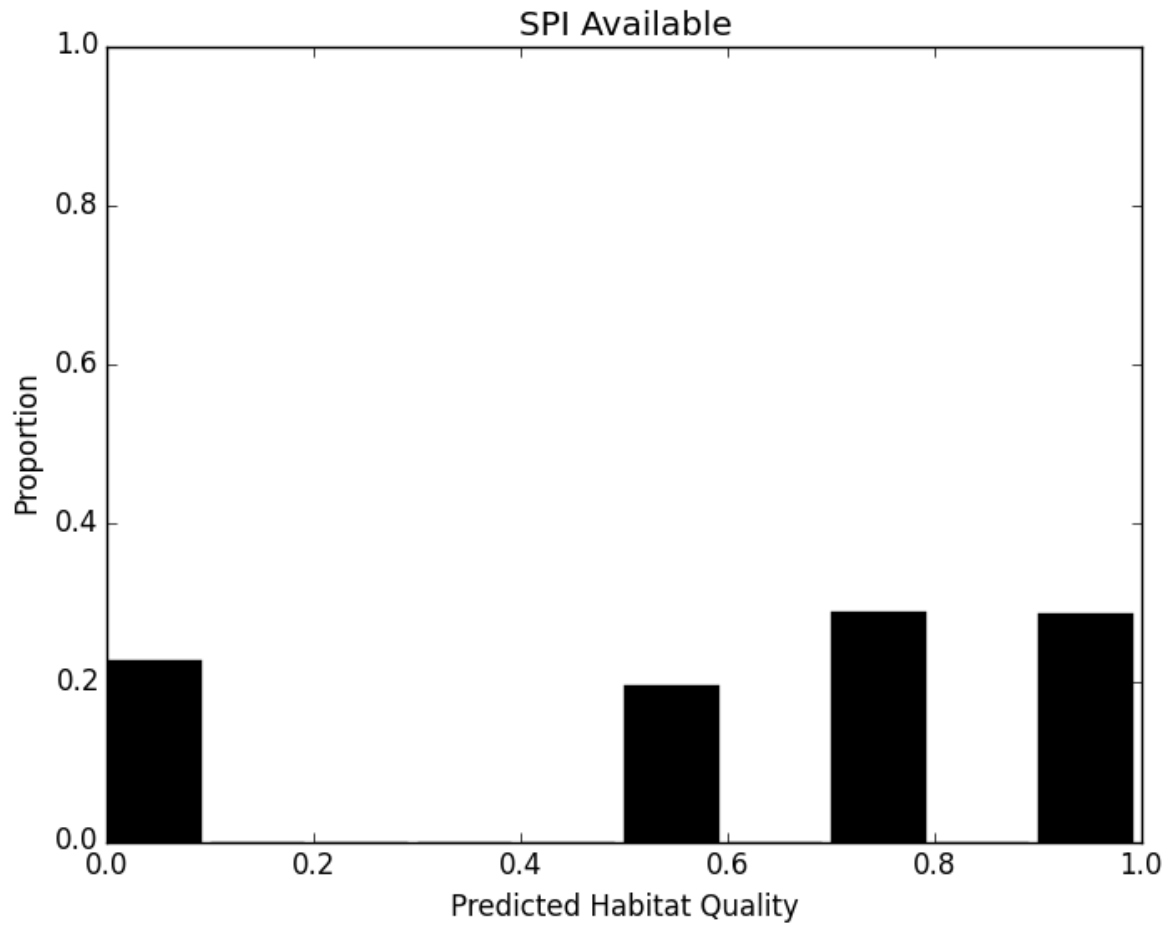


Figure 30. The average Ivlev and modified-Ivlev resource selection functions based on the SPI habitat form model values for all fisher locations on Stirling and surrounding areas from 2010 to 2014 in northern California.

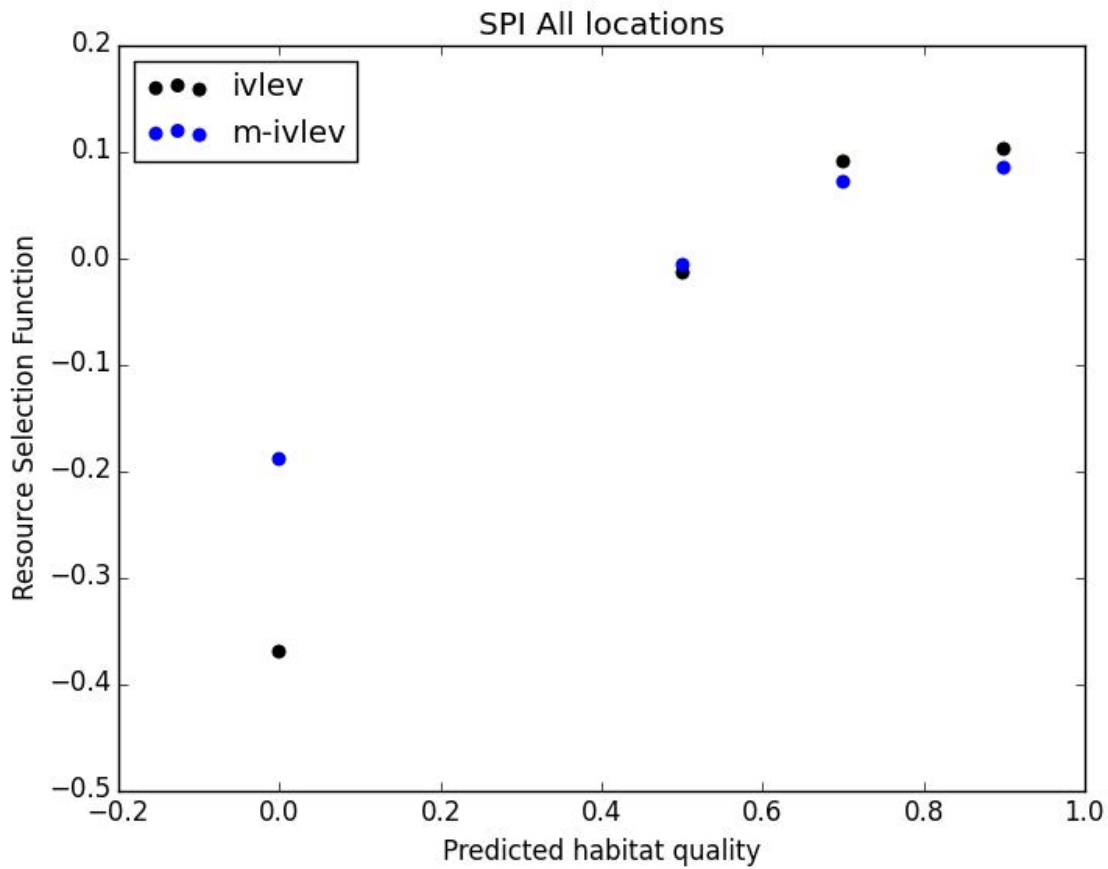


Figure 31.

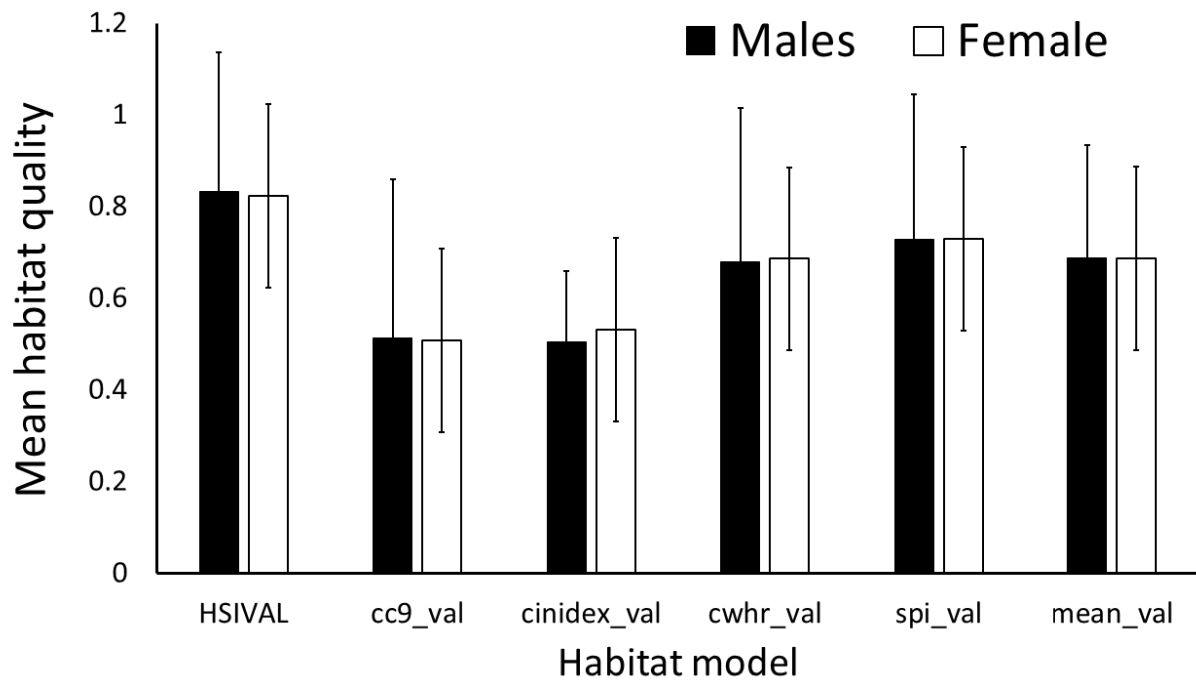


Figure 32. Proportion of available and used habitat as predicted by the Mean habitat suitability model and resultant modified-Ivlev and Ivlev resource selection functions for A. data from a female fisher who wore a GPS collar 5 days in fall of 2015 and B) data from a second female fisher who wore a GPS collar for 6 days in fall of 2015, both data sets collected in northern California on the Stirling district owned by SPI.

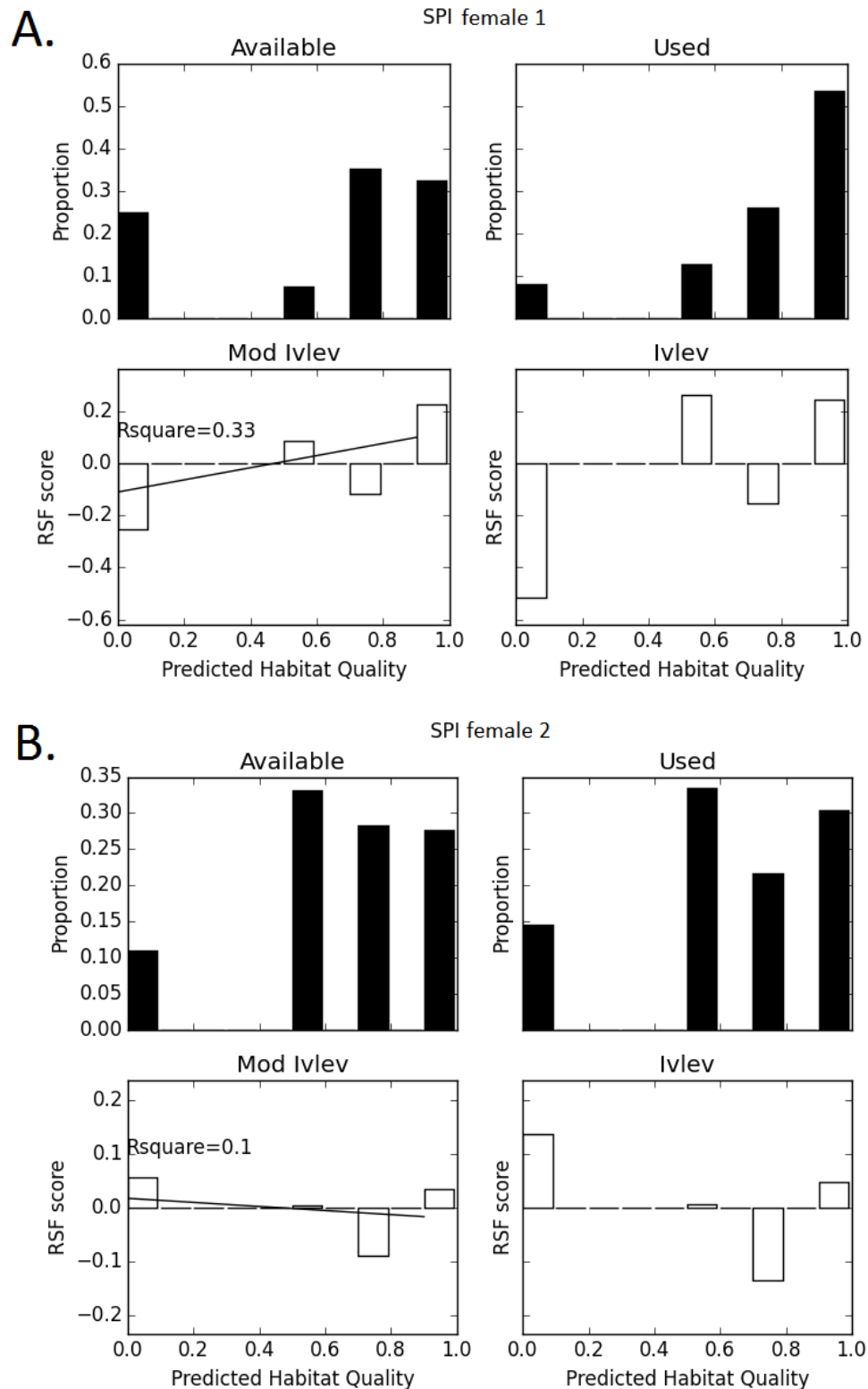


Figure 33. A. Mean modified-Ivlev resource selections female habitat selection for Mean habitat for all female within the 50% isopleth of their home ranges for years 2010 to 2013 (error bars = 1 SD). B. Mean modified-Ivlev resource selection function for all females and years within 50% isopleth of their home ranges for years 2010 to 2014 on Stirling district in northern California.

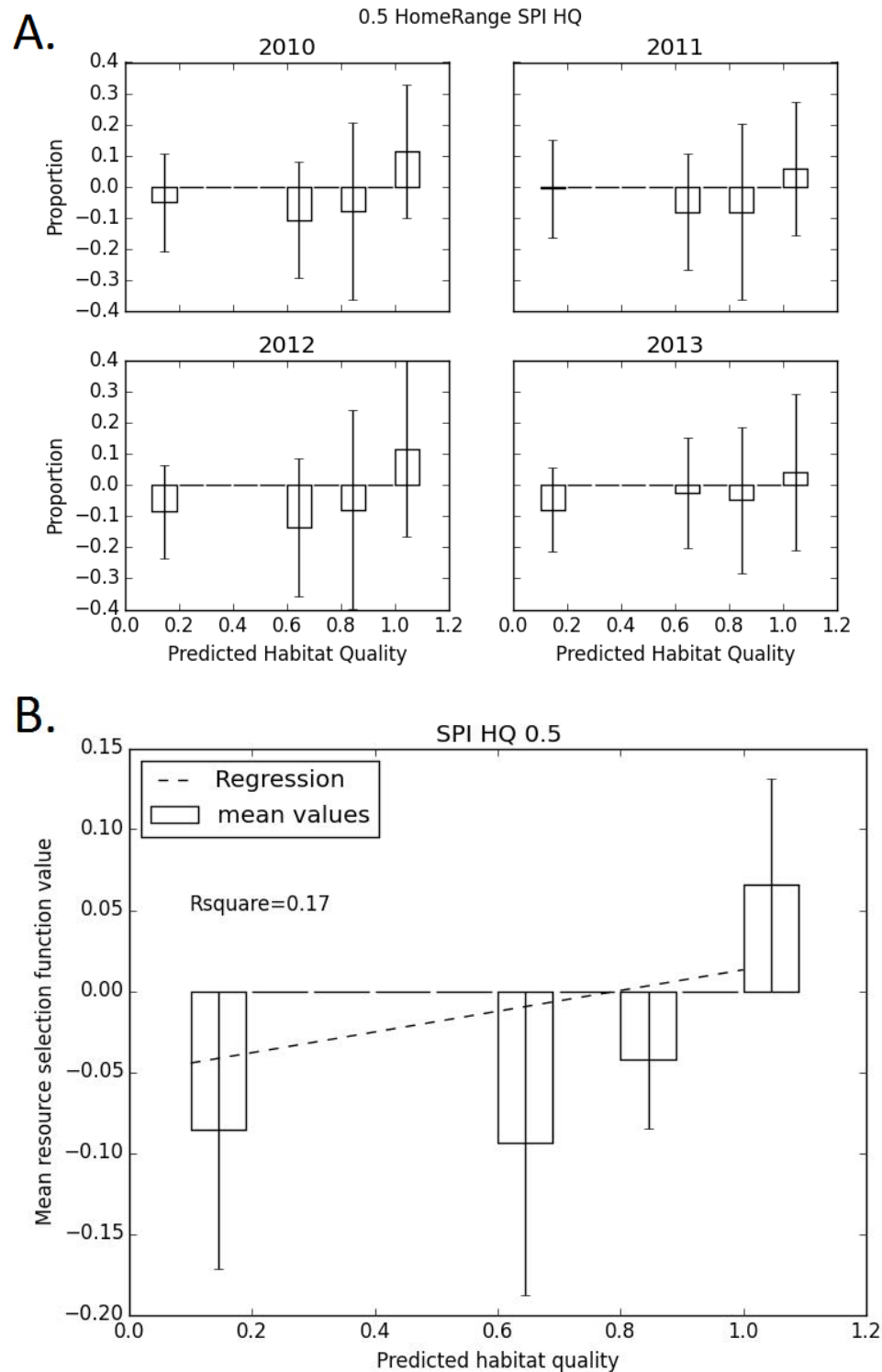


Figure 34. Distribution of available habitat as predicted by the Mean habitat model for the Stirling district in northern California in study year 2010.

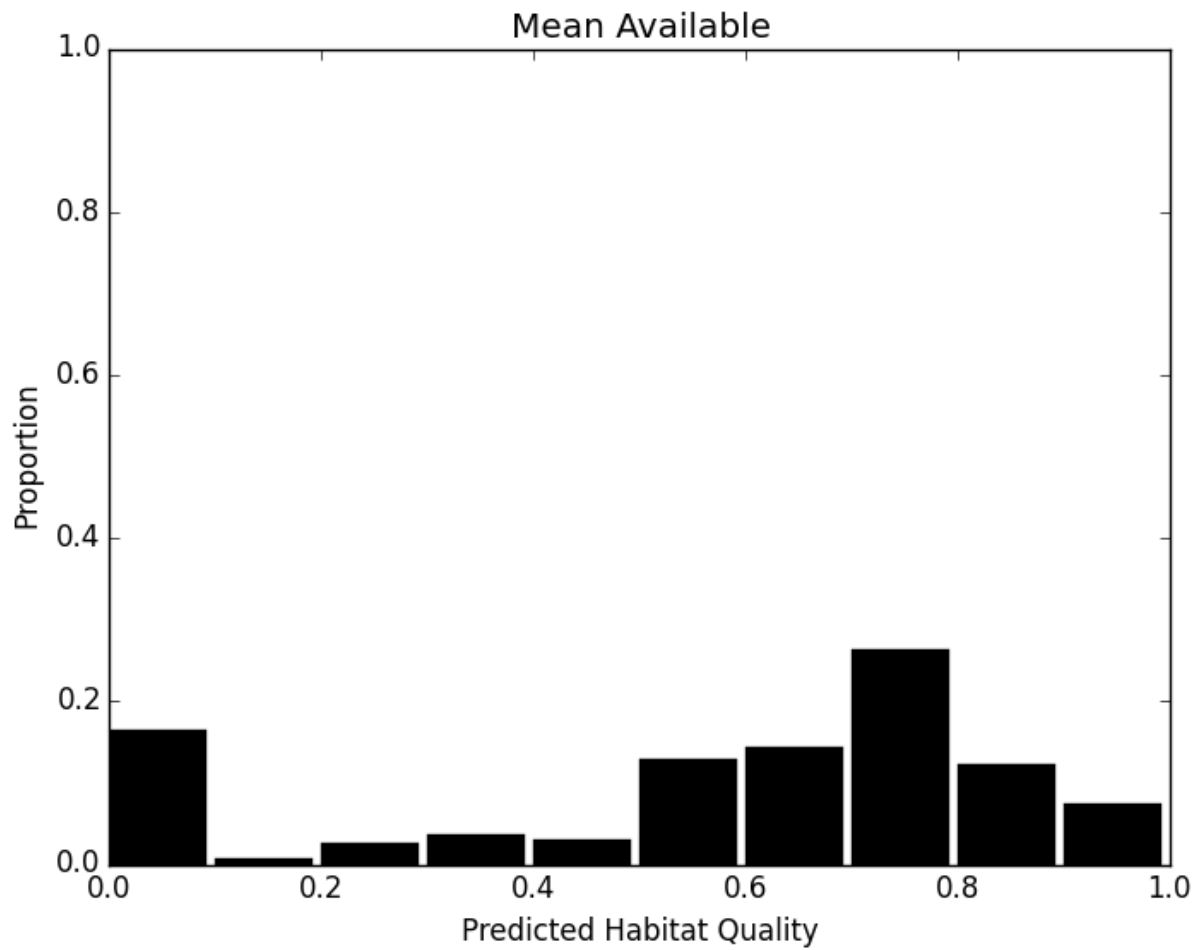


Figure 35. The average Ivlev and modified-Ivlev resource selection functions based on the Mean habitat model values for all fisher locations on Stirling and surrounding areas from 2010 to 2014 in northern California.

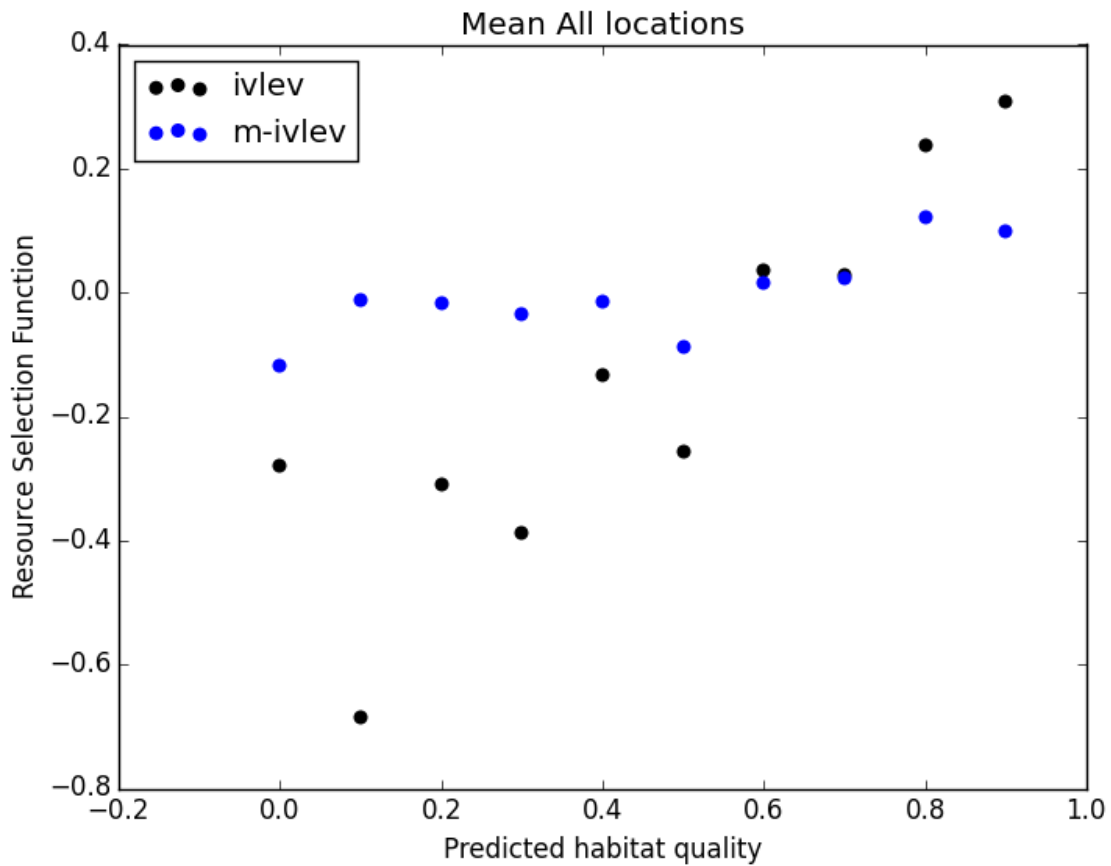


Figure 36. Plot of habitat values from the mean of 6 fisher habitat models (Allen, Carroll et al., Carroll Index, CWHR, and SPI) against the standard deviation from those same models

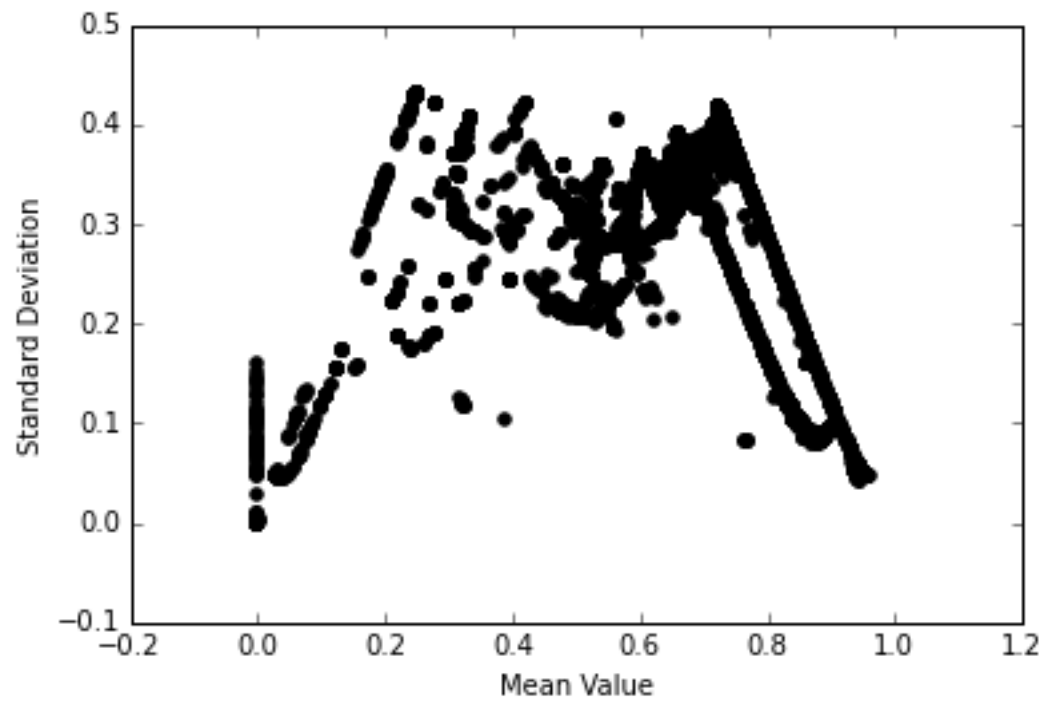


Figure 37. Distribution of available habitat as predicted by 6 different fisher habitat models of Davis et al. 2007 for study area in northern California.

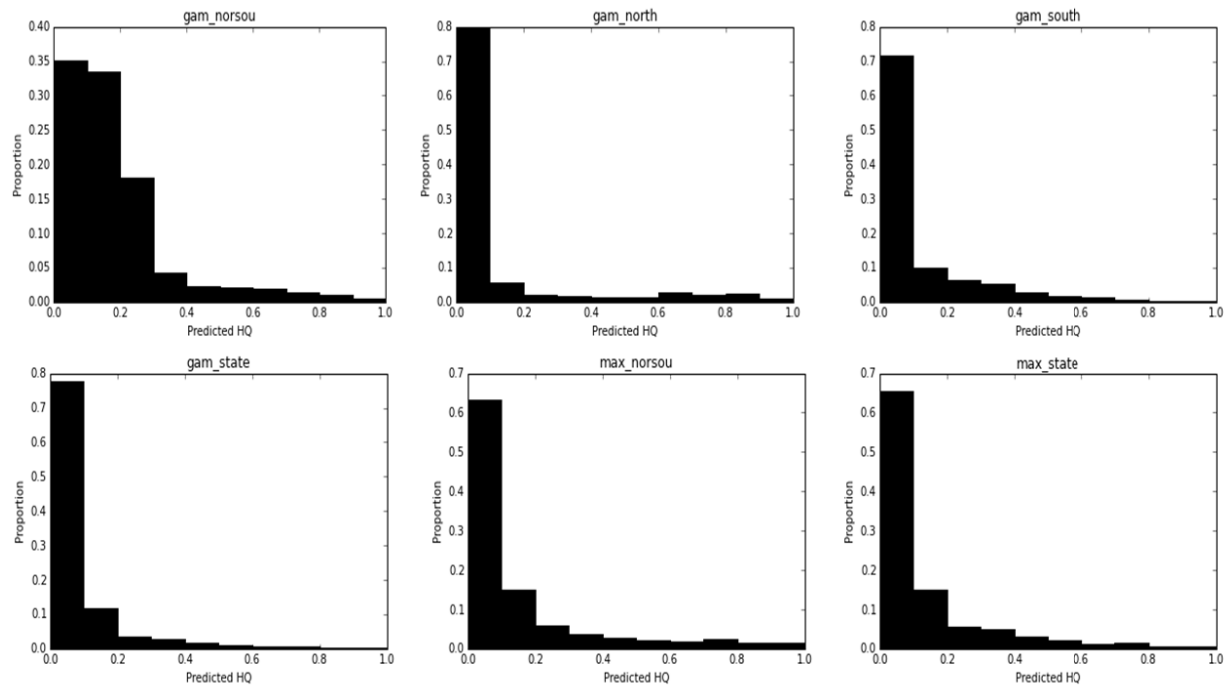


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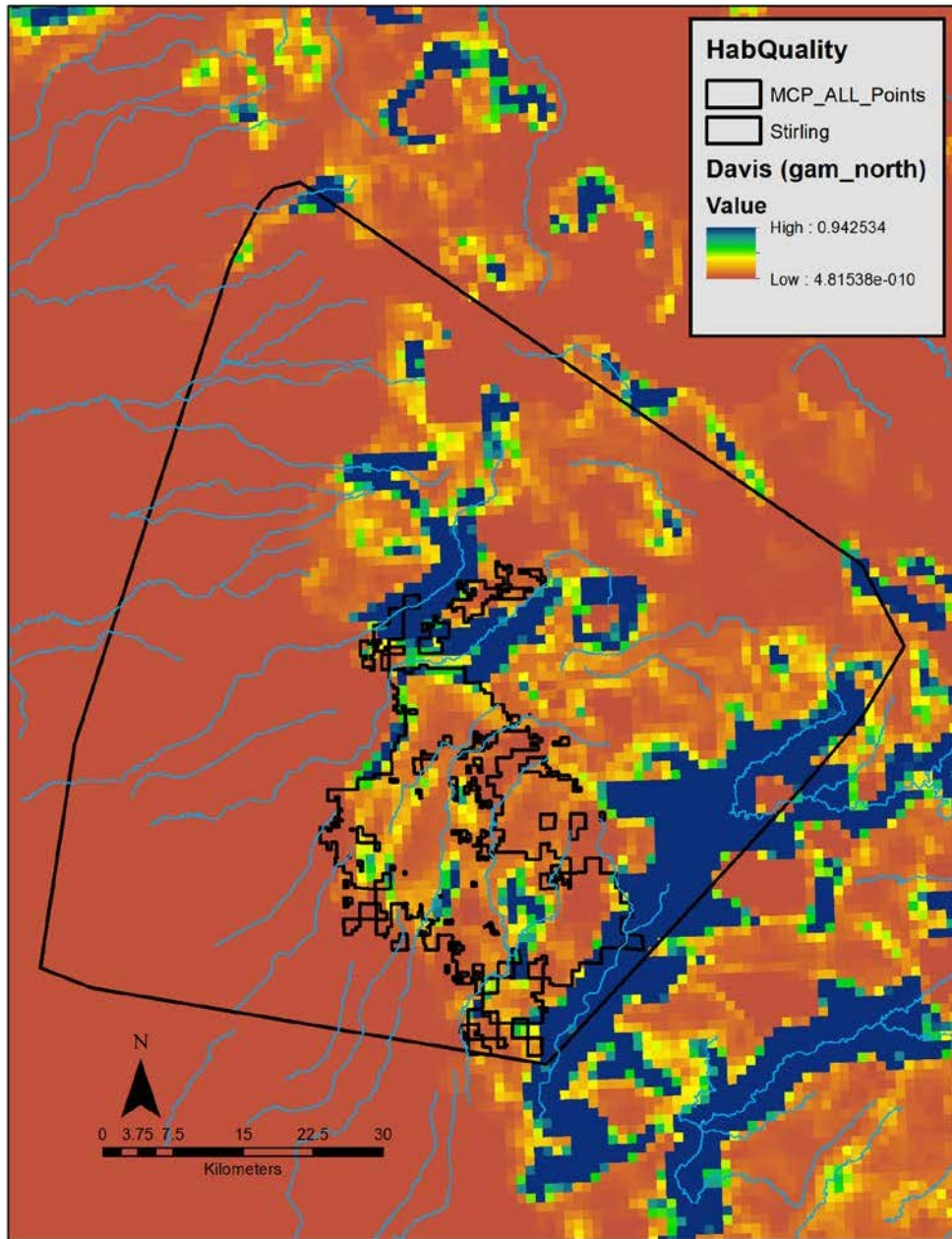


Figure 39. Mean Ivlev and modified-Ivlev resource selection functions all fisher locations on Stirling and surrounding area in Northern Sierra from 2010 to 2014.

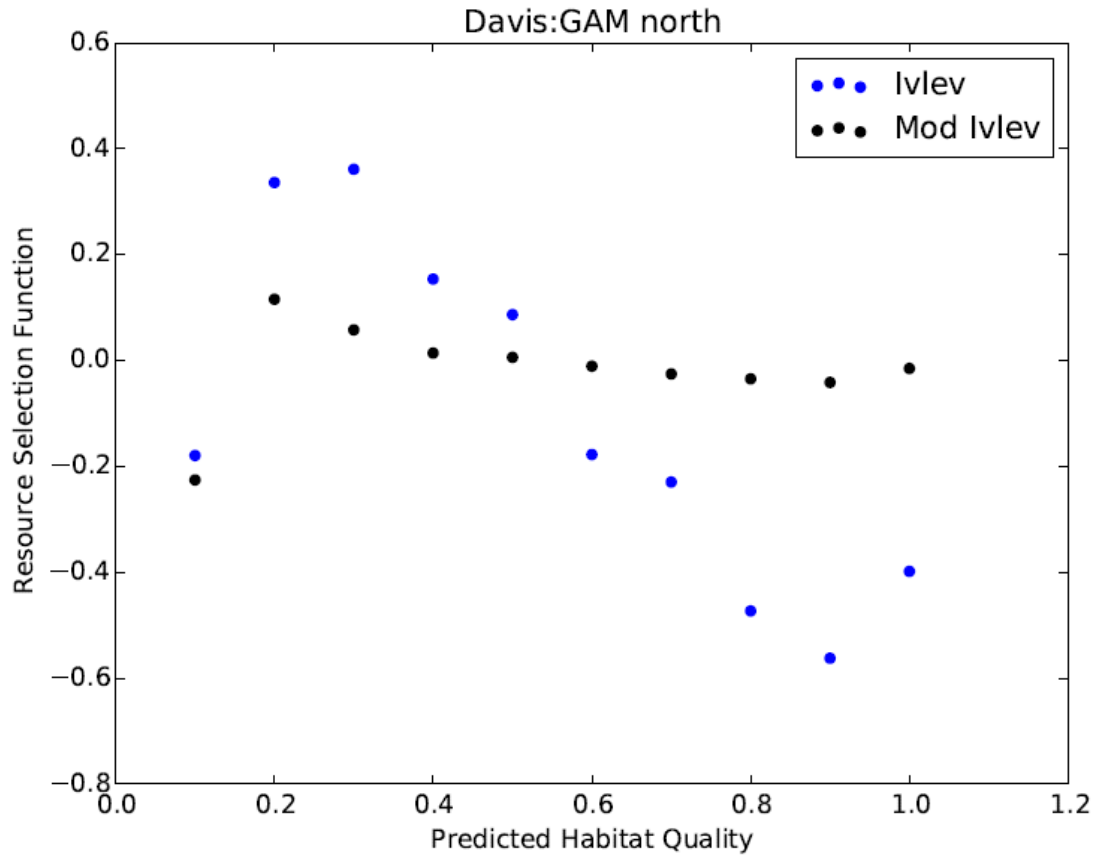


Figure 40. Spatial distribution of hexagon categorized as low habitat value (CBT, yellow hexagons), moderate value (SC, green hexagons), and high habitat value (TC, blue hexagons). Hexagons with line fills have 1 (horizontal fill), 2 (diagonal fill), 3 (dotted diagonal fill) and 4 (with vertical dashed fill – a single cell) female 50% isopleth utilization distributions within them for fishers in northern California from 2010 to 2014.

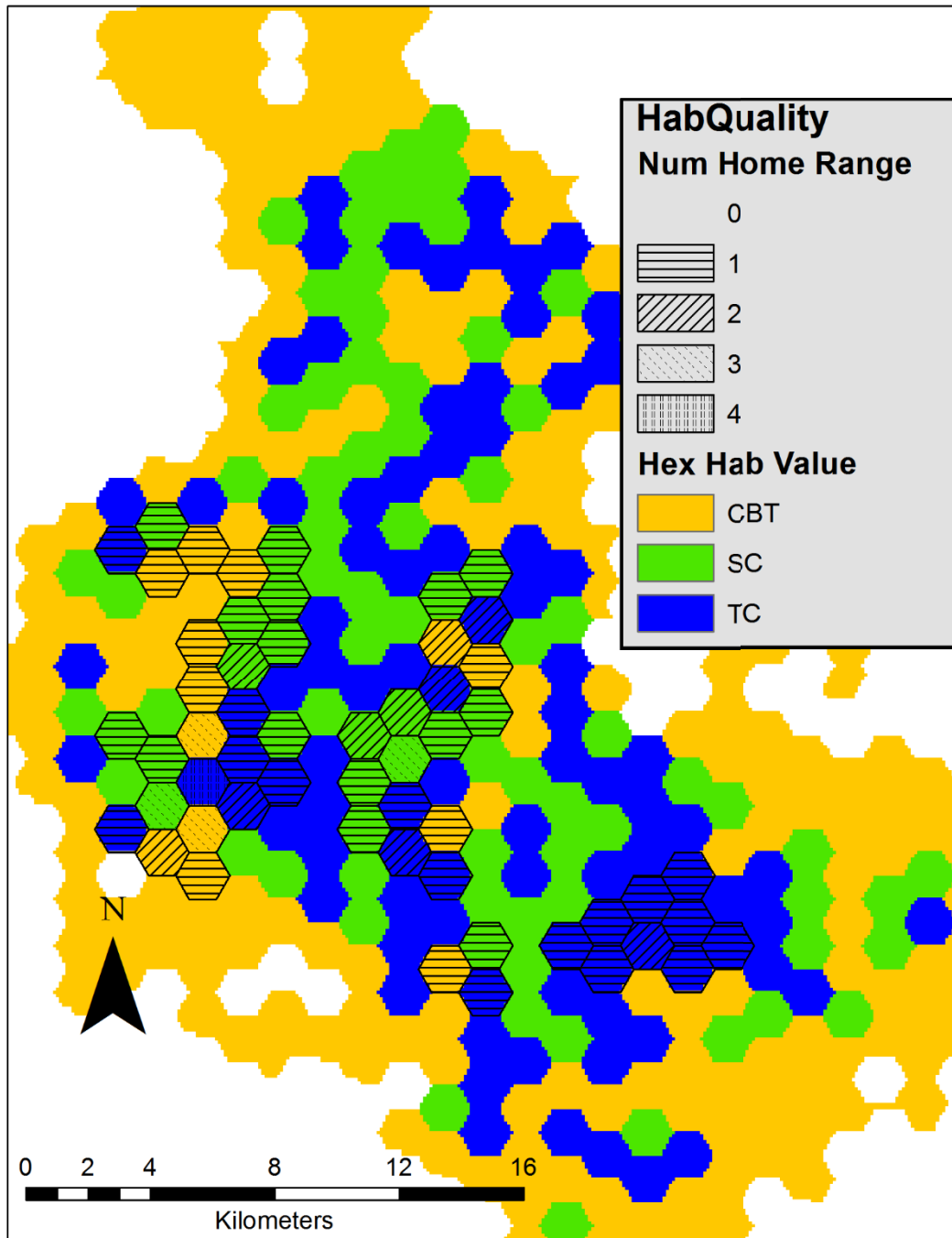


Figure 41. The mean value of hexagons that had 1, 2, 3 or 4 female fisher home ranges within them from 2010 to 2014.

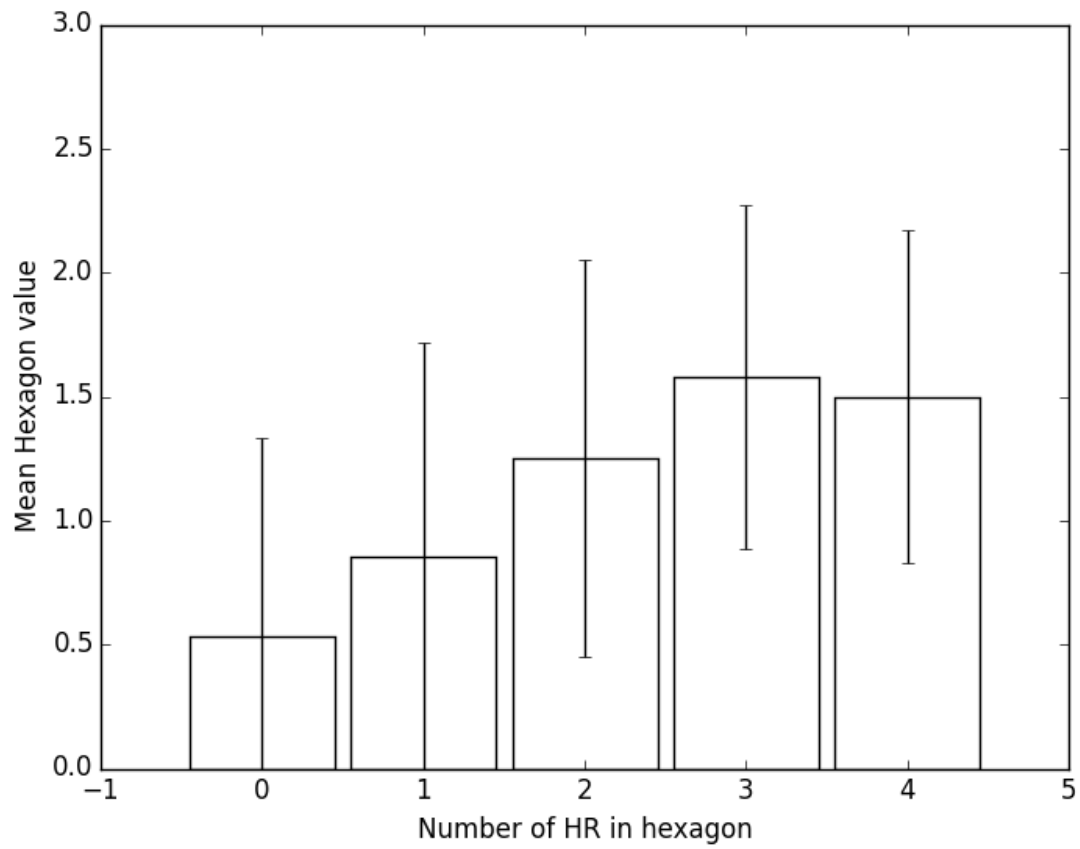


Figure 42. Distribution of the proportion of female fisher utilization distributions that occur in hexagons categorized as low habitat value (Currently Below Threshold [CBT]), moderate value (Support Core [SC]), and high habitat value (Territory Core [TC]) for fishers in northern California from 2010 to 2014.

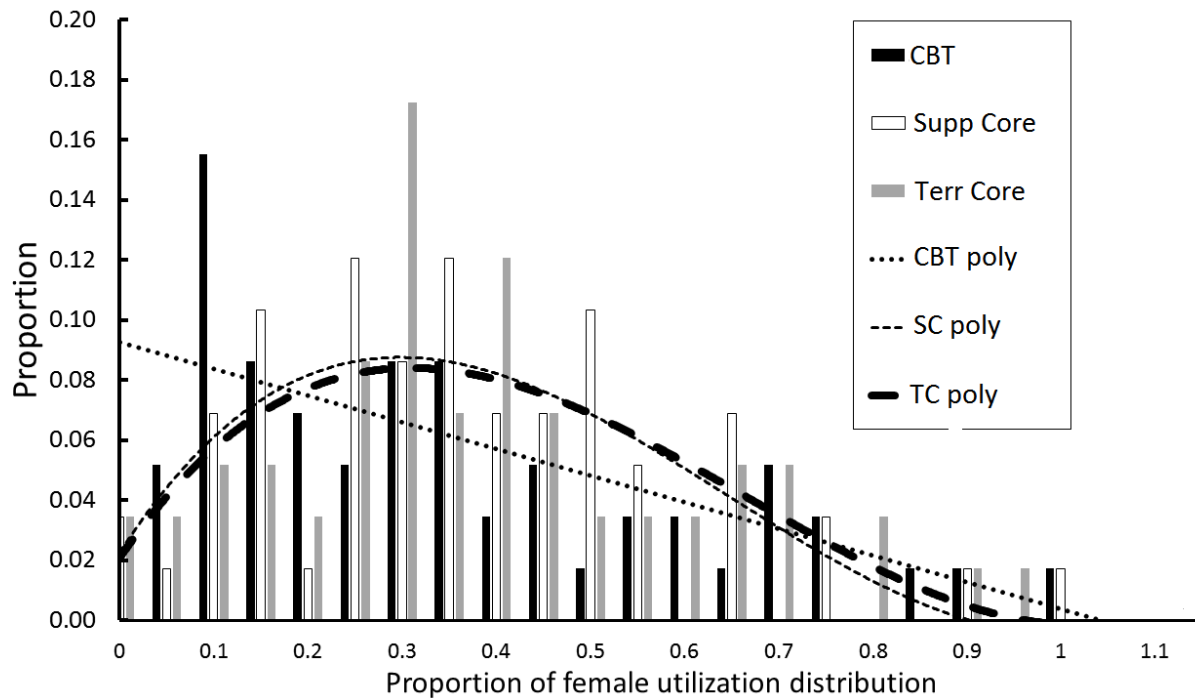


Figure AllenMean. Mean values for canopy closure, percent hardwoods, and quadratic mean diameter at breast height for all overstory trees (QMD) of the Allen habitat suitability index at 3 discrete habitat quality ranges (<0.80, 0.80-0.90, and >0.90). Error bars equal 1 standard deviation.

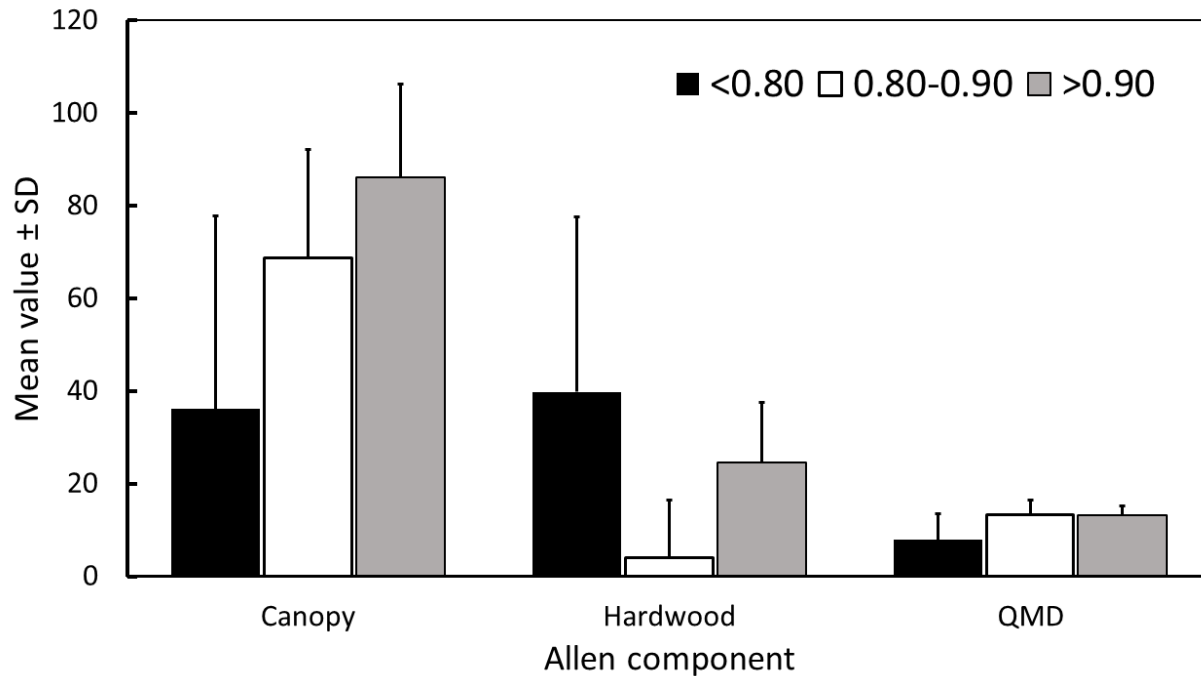
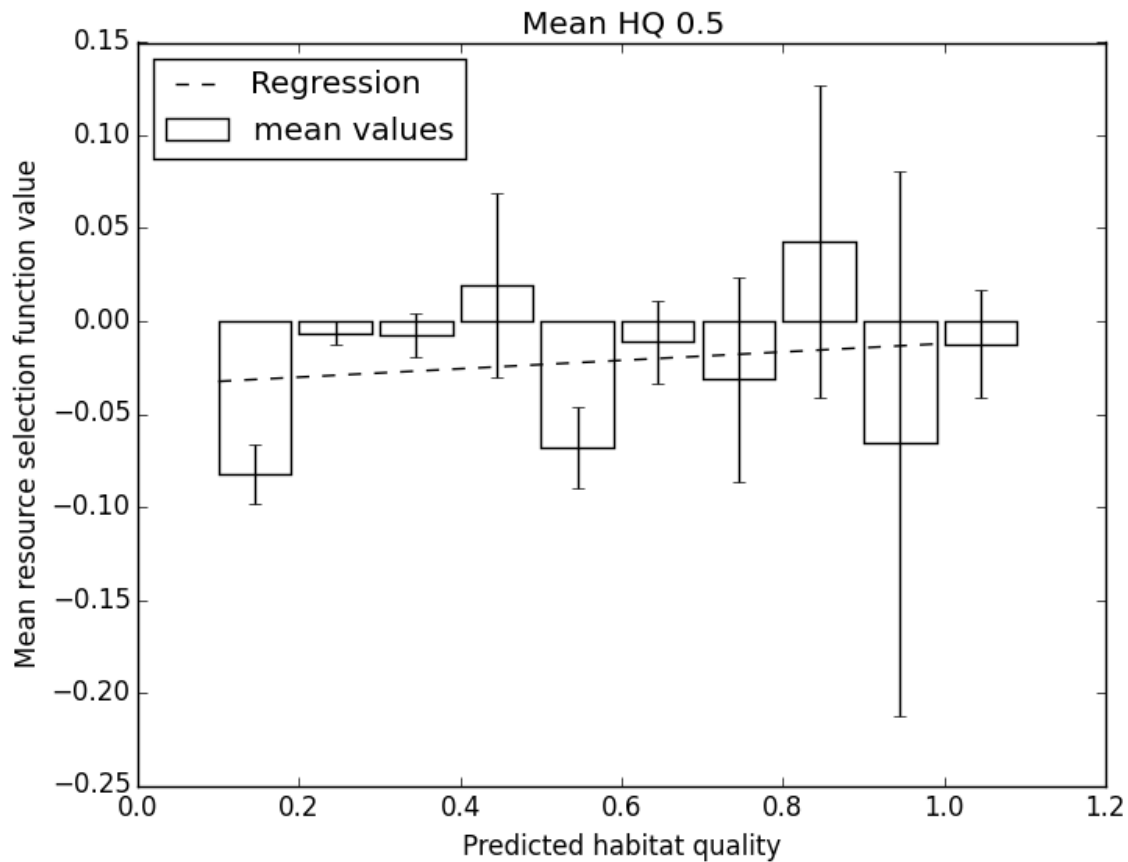


Figure Mean_by_year. The mean Modified-Ivlev resource selection function value (bars) at the 50% utilization distribution, and the estimated regression line (dashed line) for all female fishers on Stirling from 2010 to 2014.



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 $\Delta AICc$ 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 ($\Delta AICc \leq 4$) 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 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 Simpson's 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*, *Tamiascurus*, 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% only when 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 ($\Delta AICc > 10$) 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 ($\Delta AICc = 1.53$) 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 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 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 $\beta = -2.34$), 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_x). 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_x). 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 <50m away receiving a 1 and >50m 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 Plot (30x30m)	Slope	Percentage grade (steepness) at the sampled plot
	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 were 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 below 0.05 for plots (n= 250) location on Stirling district in northern California.

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

Table 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	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]) $\Psi(QMD[St])$	1086.1	2.33	0.04	0.31	5
	p(TBA[N]+Num_logs[MS]) $\Psi(QMD[N])$	1086.2	2.45	0.04	0.29	5
	p(TBA[N]+Num_logs[MS]) $\Psi(QMD[N]+SPA[St])$	1086.4	2.68	0.03	0.26	6
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(QMD[N]+SPA[St])$	1086.5	2.70	0.03	0.26	7
	p(TBA[N]+Num_logs[MS]+mQMD[St]) $\Psi(QMD[St])$	1087.1	3.31	0.02	0.19	6
	p(TBA[N]+Num_logs[MS]) $\Psi(Year + QMD[St])$	1087.1	3.33	0.02	0.19	6
	p(TBA[N]+Num_logs[MS]+pHard[MS]) $\Psi(QMD[St])$	1087.1	3.36	0.02	0.19	6
	p(TBA[N]+Num_logs[MS]+NumHard[MS]) $\Psi(Dist2Road)$	1087.4	3.61	0.02	0.16	6

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	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]) $\Psi(\text{QMD[St]})$	1087.8	4.00	0.02	0.14	6
Tamiasciurus	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+QMD[MS]+LPA[N]})$	526.4	0.00	0.32	1.00	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+QMD[MS]})$	528.0	1.56	0.15	0.46	6
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+QMD[MS]+Canopy[St]})$	528.8	2.34	0.10	0.31	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+QMD[MS]+SPA[N]})$	529.3	2.86	0.08	0.24	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+BA[St]+LPA[N]})$	529.3	2.89	0.07	0.24	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+QMD[St]+LPA[N]})$	529.5	3.11	0.07	0.21	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+Canopy[MS]})$	529.6	3.17	0.07	0.20	6
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+QMD[MS]+RoadDensity})$	530.0	3.57	0.05	0.17	7
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]})$	530.0	3.58	0.05	0.17	5
	p(LPA[N]+LPA[St]) $\Psi(\text{pCon[N]+pCon[N]}^2)$	532.1	5.67	0.02	0.06	6
Sciurus	p(LPA[N]) $\Psi(\text{QMD[MS]+LPA[N]+pHard[St]})$	184.6	0.00	0.34	1.00	6
	p(LPA[N]) $\Psi(\text{QMD[MS]+LPA[N]+BACon[MS]})$	186.1	1.47	0.17	0.48	6
	p(LPA[N]) $\Psi(\text{QMD[MS]+LPA[N]+pCon})$	186.2	1.54	0.16	0.46	6
	p(LPA[N]) $\Psi(\text{QMD[MS]+LPA[N]+BAHard[MS]})$	186.8	2.16	0.12	0.34	6
	p(LPA[N]) $\Psi(\text{QMD[MS]+LPA[N]+BAHard[MS]})$	186.8	2.16	0.12	0.34	6
	p(LPA[N]) $\Psi(\text{QMD[MS]+LPA[N]})$	188.9	4.26	0.04	0.12	5
Otosperm.	p(g \times SPA[St]) $\Psi(\text{BACon[MS]+MBA[N]})$	332.1	0.00	0.25	1.00	21
	p(g \times SPA[St]) $\Psi(\text{BACon[MS]+Bacon[MS]}^2)$	332.2	0.08	0.24	0.96	22
	p(g \times SPA[St]) $\Psi(\text{BACon[MS]})$	334.2	2.07	0.09	0.36	20
	p(g \times SPA[St]) $\Psi(\text{BACon[MS]+BA[St]})$	334.6	2.46	0.07	0.29	21
	p(g \times SPA[St]) $\Psi(\text{QMD[St]})$	335.2	3.09	0.05	0.21	20
	p(g \times SPA[St]) $\Psi(\text{BACon[MS]+QMD[N]})$	335.9	3.73	0.04	0.15	21
	p(g \times SPA[St]) $\Psi(\text{BACon[MS]+Shrubs})$	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	$CI_{AvgLower}$	$CI_{AvgUpper}$
DistH20	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.

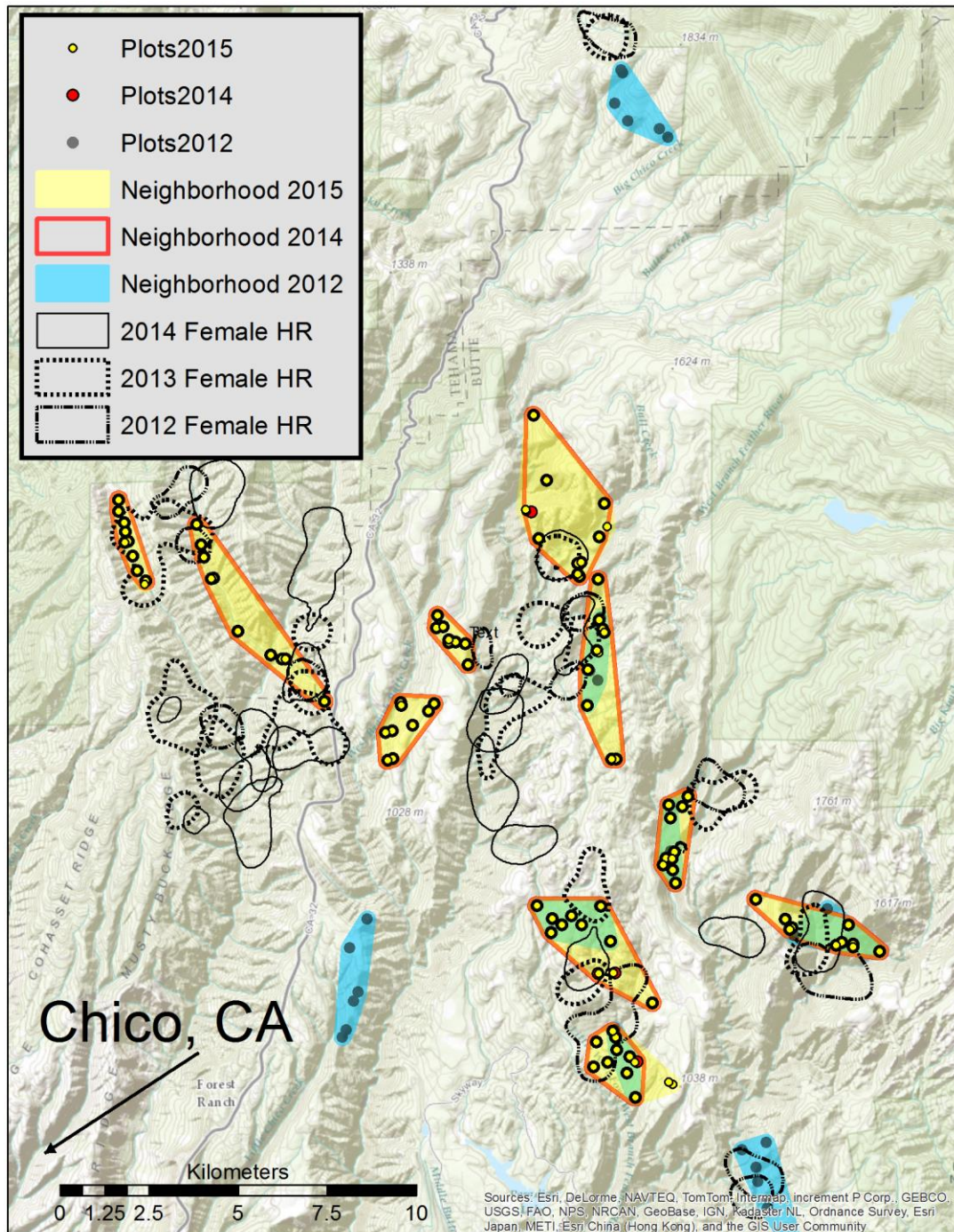


Figure Trap_Bias. Naïve occupancy estimates for 9 genera of mammal using both trapping and camera data (blue bars) and camera data exclusively (gray bars) at 98 fisher prey sampling plots during 2014 on Stirling in northern California.

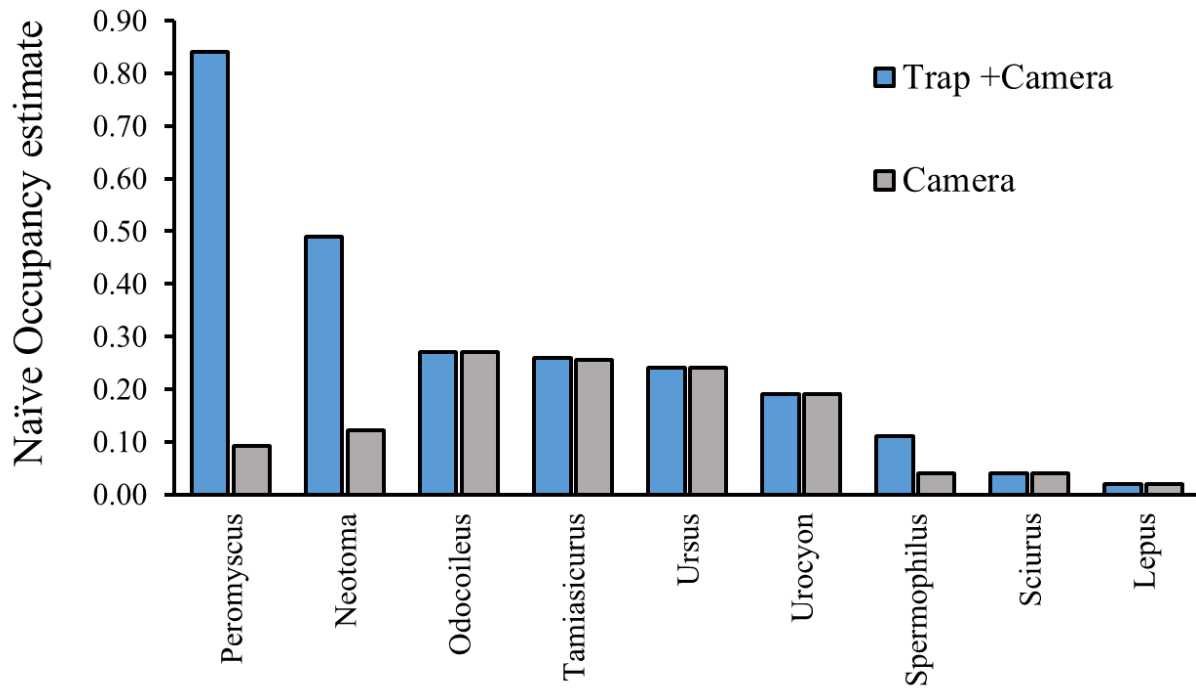


Figure WoodratOcc. The predicted relationship of quadratic mean diameter at breast height of trees at the stand-level to the probability that a plot was occupied by woodrats for 250 fisher prey sampling plots in northern California during 2012, 2014, and 2015.

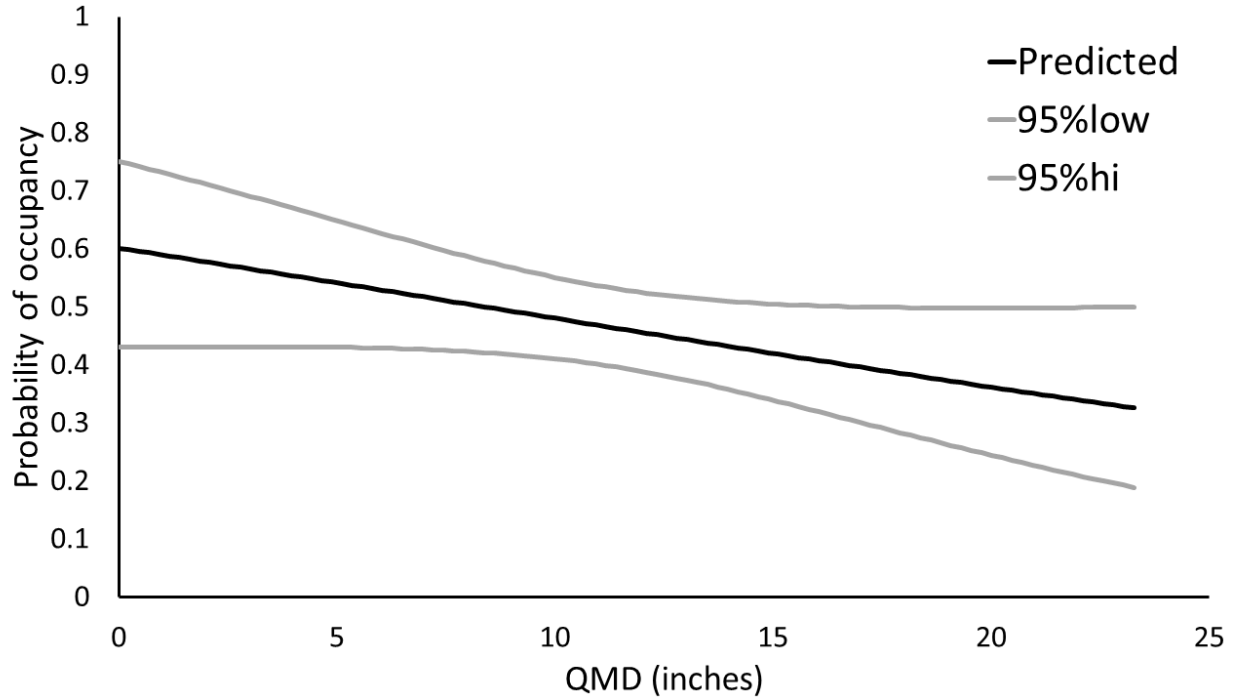


Figure Chickaree. A. Estimated relationship between the percent of trees that were coniferous within a neighborhood to the probability of chickaree occupancy and B. Estimated relationship between quadratic mean diameter of trees at a plot to the probability of chickaree occupancy for 250 fisher prey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

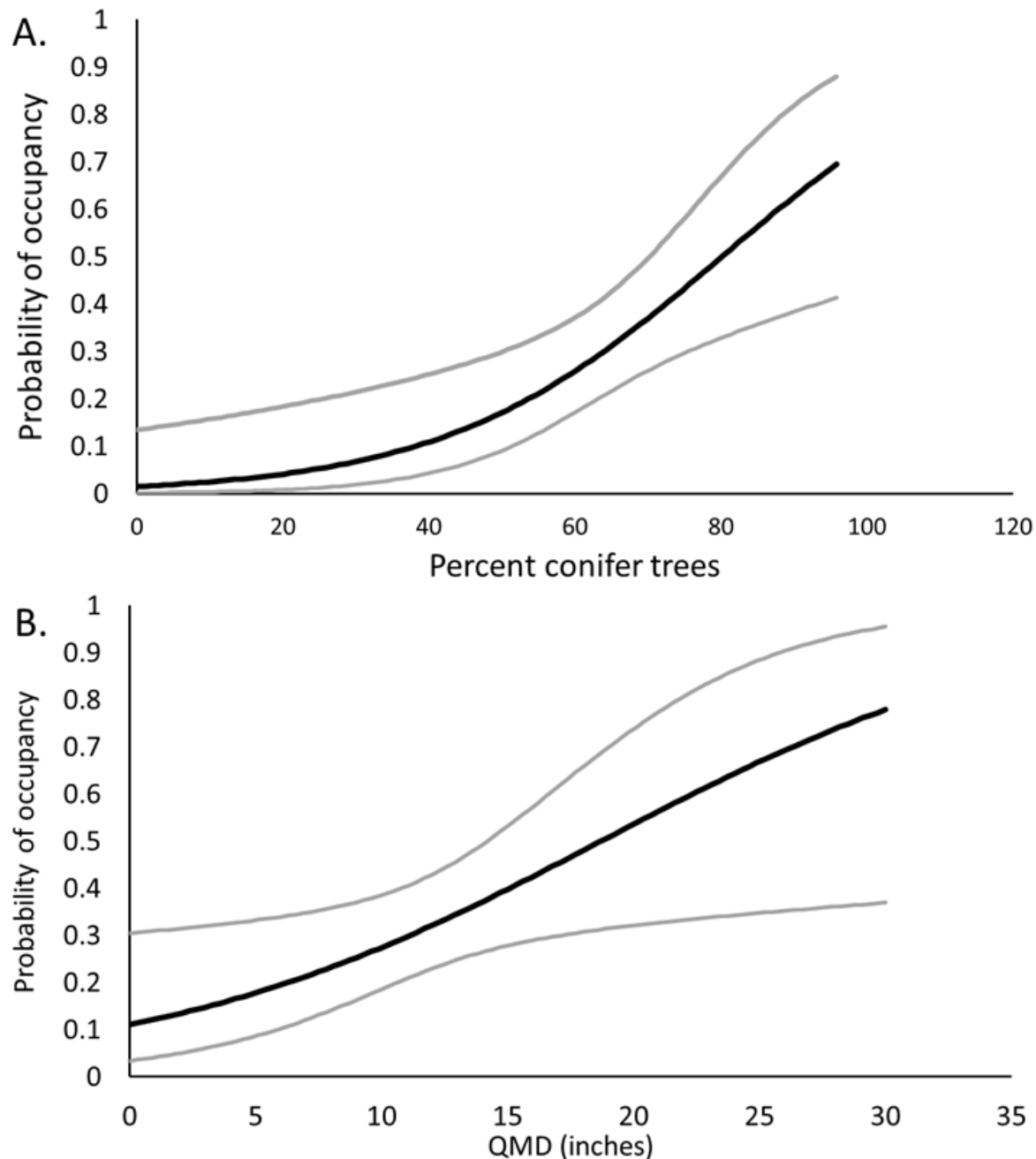


Figure Gray_Detection. The estimated probability of detecting gray squirrels as a function of the mean number of logs per acre within a neighborhood changes for 250 fisher prey sampling plots in northern California in 2012, 2014, and 2015.

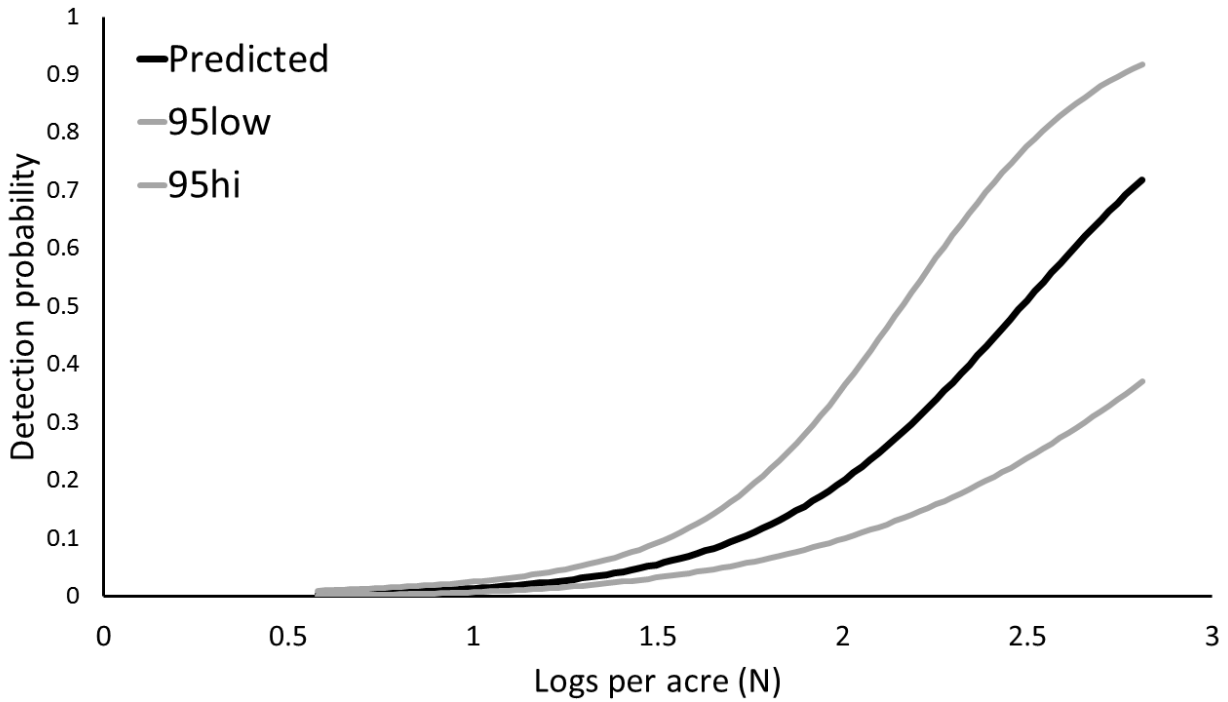


Figure SPBEE_Detec. Estimated detection rates of California ground squirrel (*Otospermophilus beecheyi*) across all neighborhoods on Stirling in northern California during 2012, 2014, and 2015.

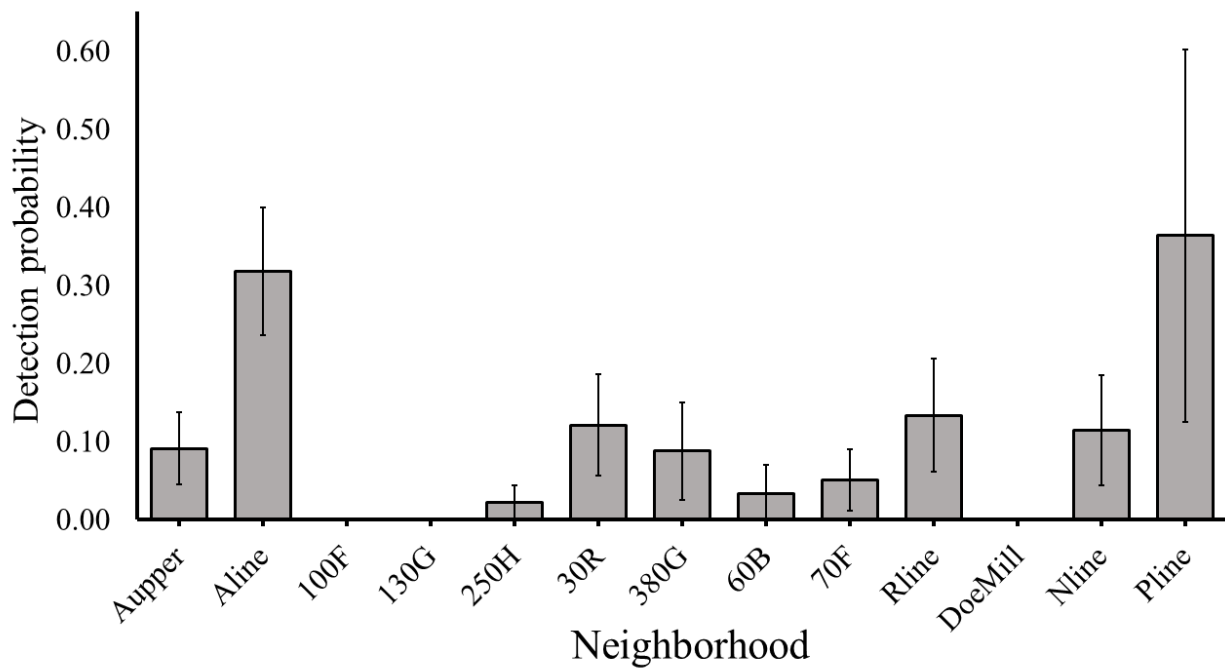


Figure SPBEE_OCC. The estimated probability of detecting California ground squirrels as a function of the basal area of conifers at a plot for 250 fisher prey sampling plots in northern California in 2012, 2014, and 2015.

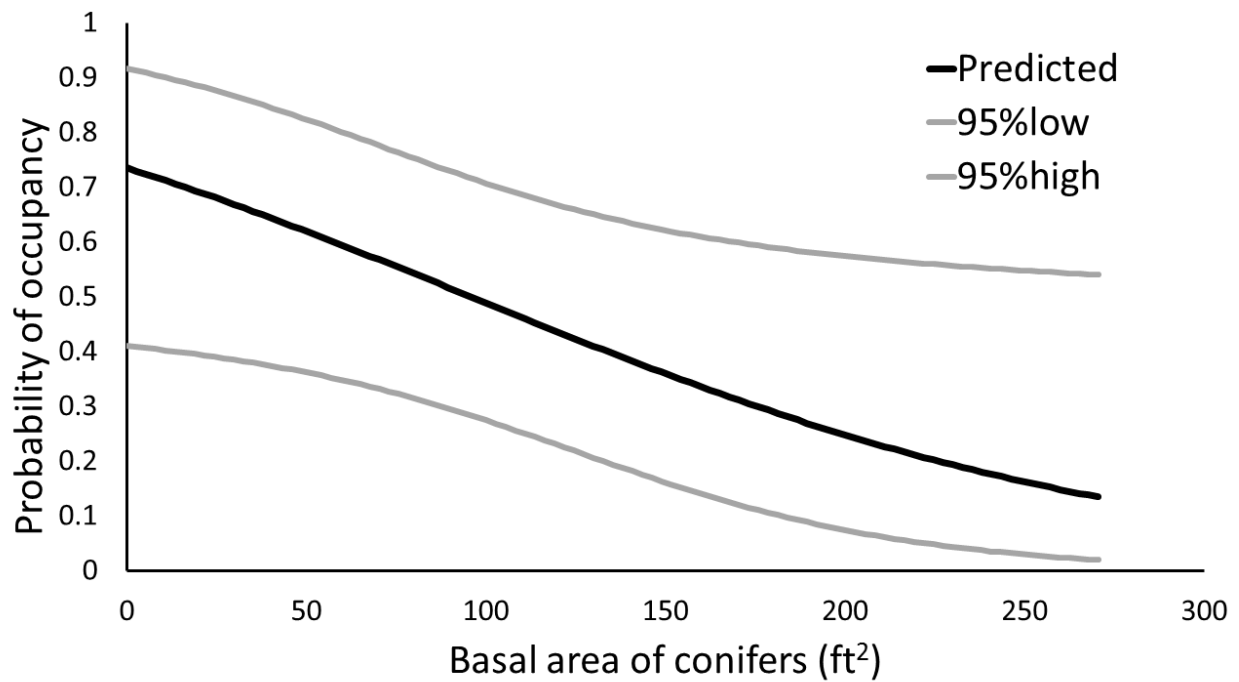


Figure VT12. The total value of a locations for 6 vegetation metrics (canopy cover > 10%, QMD > 12 cm, mean basal area > 15m², logs per acre > 1, snags per acre > 1, and percent hardwoods > 20%) above a specific threshold on Stirling in northern California.

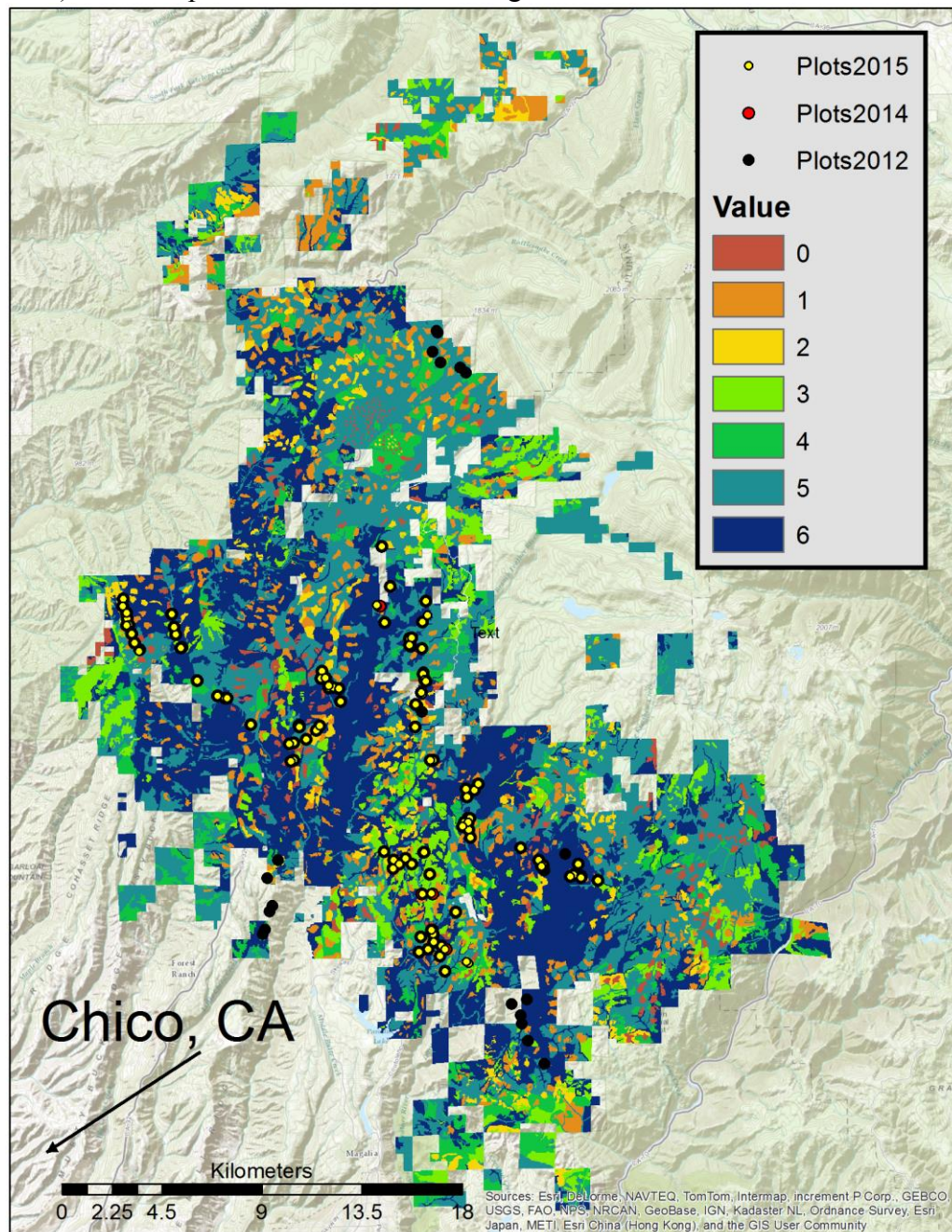
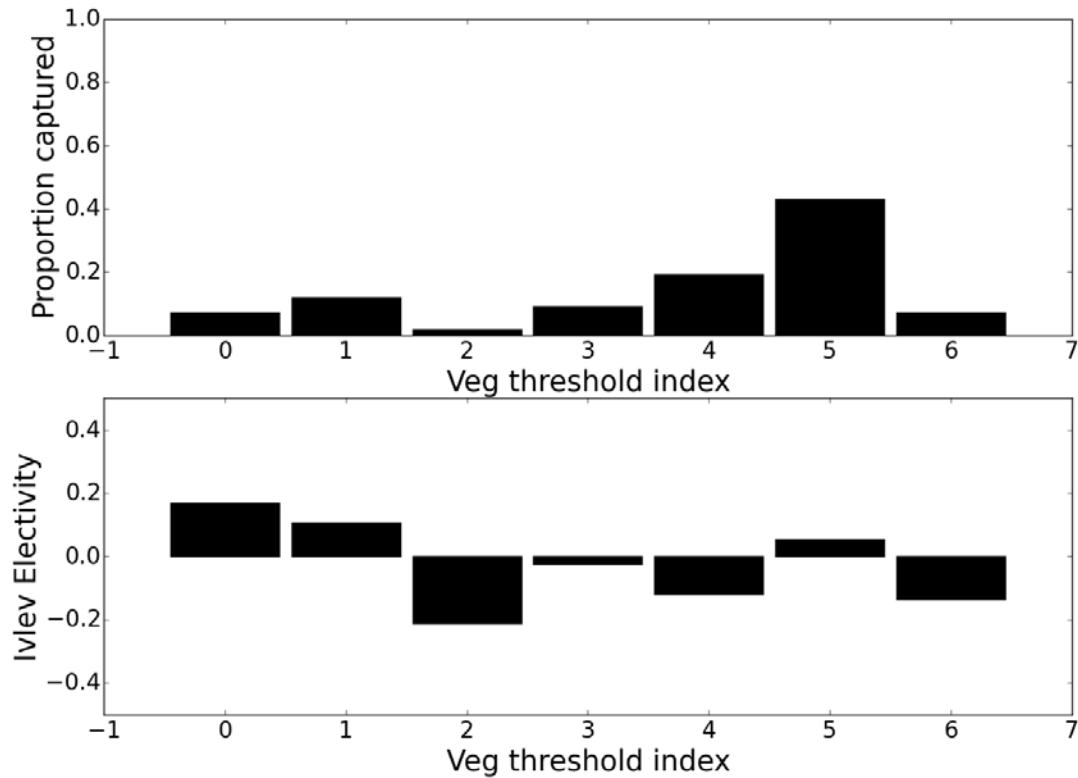


Figure VTI_Wood. Top) The proportion of detections for all woodrats (*Neotoma* spp) at plots meeting 0-6 threshold metrics for canopy cover ($>10\%$), quadratic mean diameter (>12.7 cm), basal area (>15.2 m²), mean logs per acre (>1), mean snags per acre (>2), and percent hardwood trees ($>20\%$) and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



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.

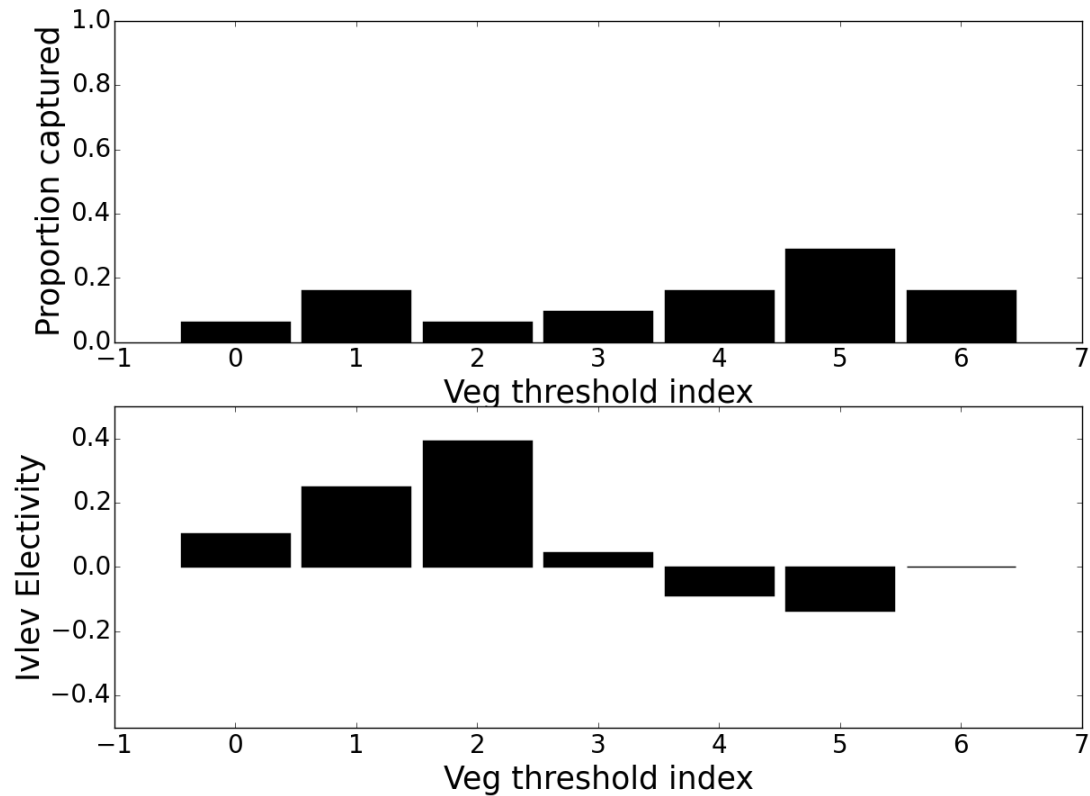


Figure TreeSquirrel. Top) The proportion of detections for all tree squirrels (*Sciurus griseus*, *Tamiasciurus douglasii*, and *Glaucomys sabrinus*) at plots meeting 0-6 threshold metrics for canopy cover ($>10\%$), quadratic mean diameter (>12.7 cm), basal area (>15.2 m²), mean logs per acre (>1), 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.

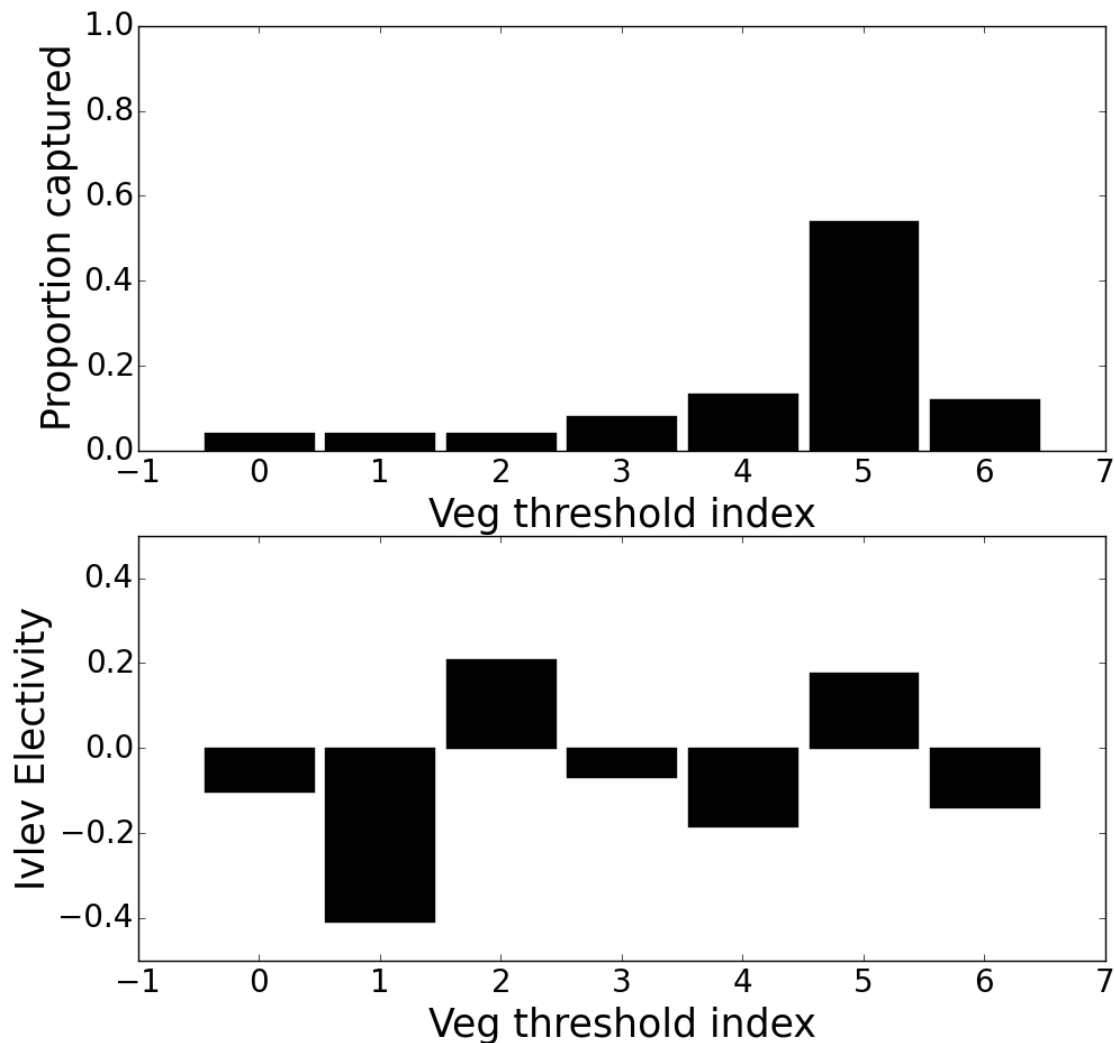


Figure CPUE. Mean catcher per unit effort (CPUE) for all fisher prey species captured or recaptured at all plots found within a neighborhood for years 2012, 2014, and 2015 on Stirling district in northern California.

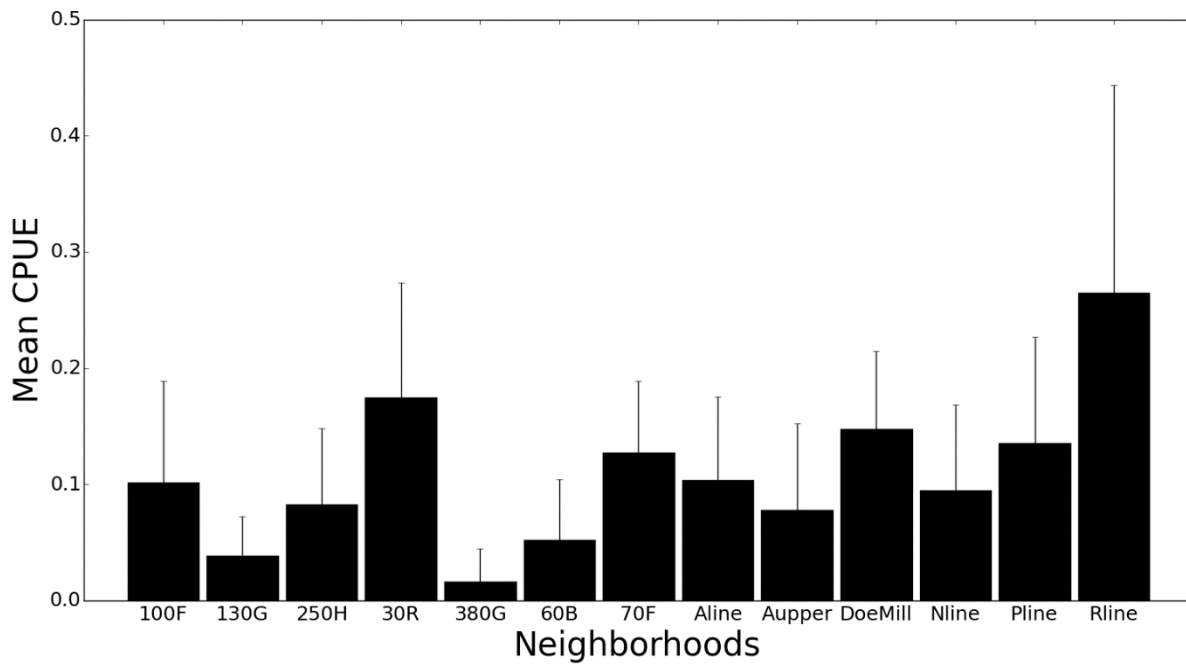


Figure CPUEx. Top) Regression catch per unit effort (CPUE) on the number of deer mice (*Peromyscus* spp) and Bottom) Regression of CPUE on the catch per unit effort rates excluding all deer mice captures for 250 fisher preys sampling plots in 2012, 2014, and 2015 on Stirling in northern California. Solid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

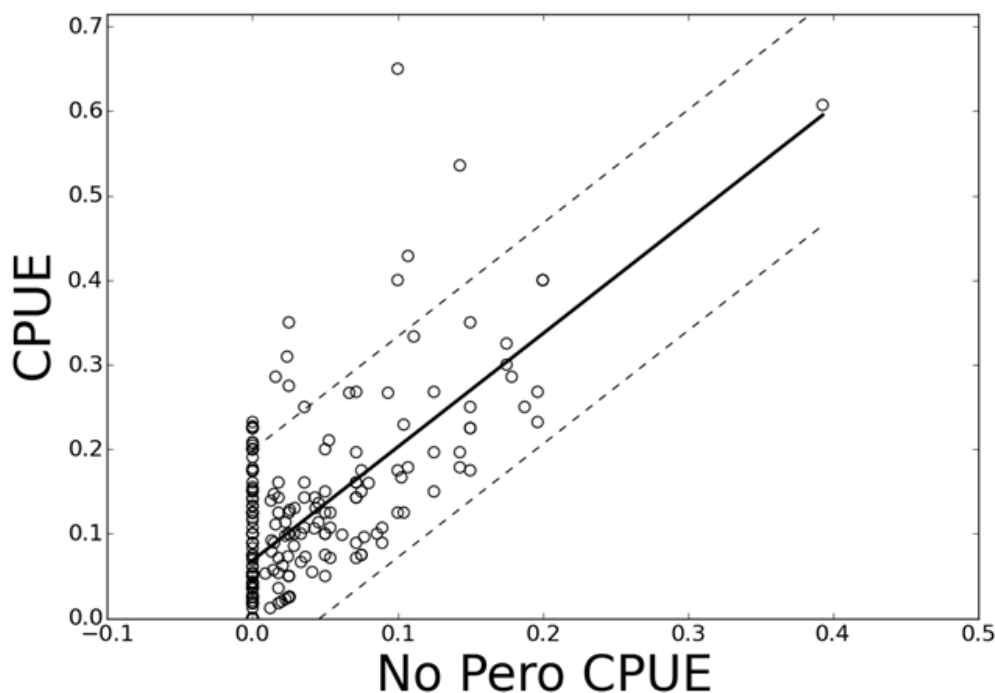
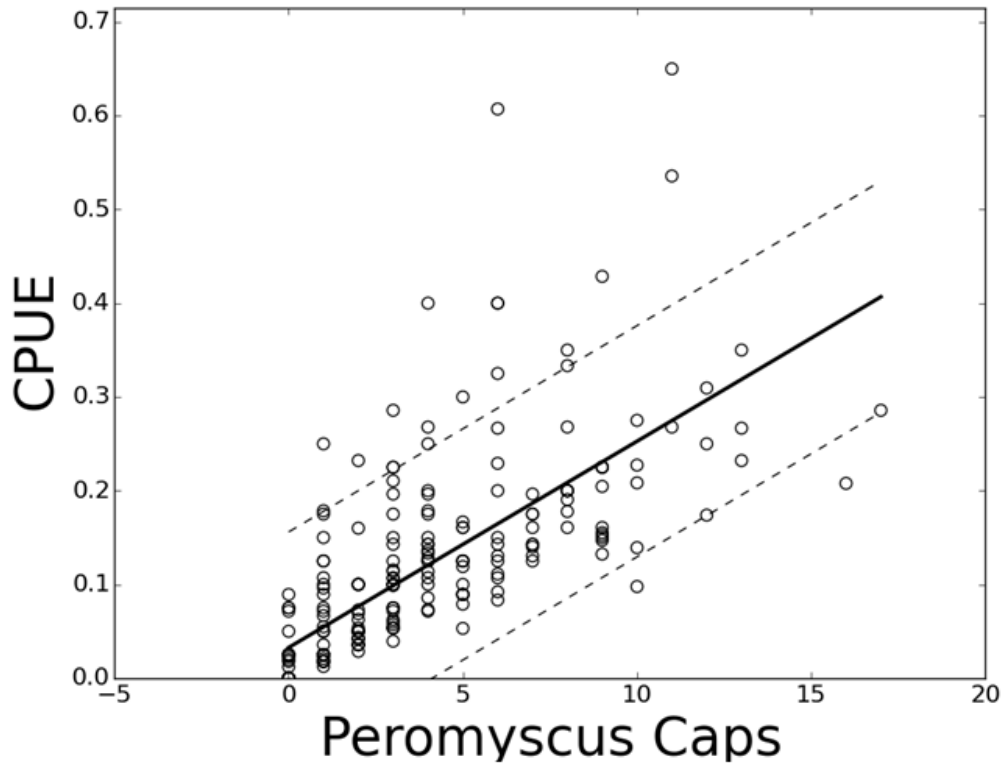
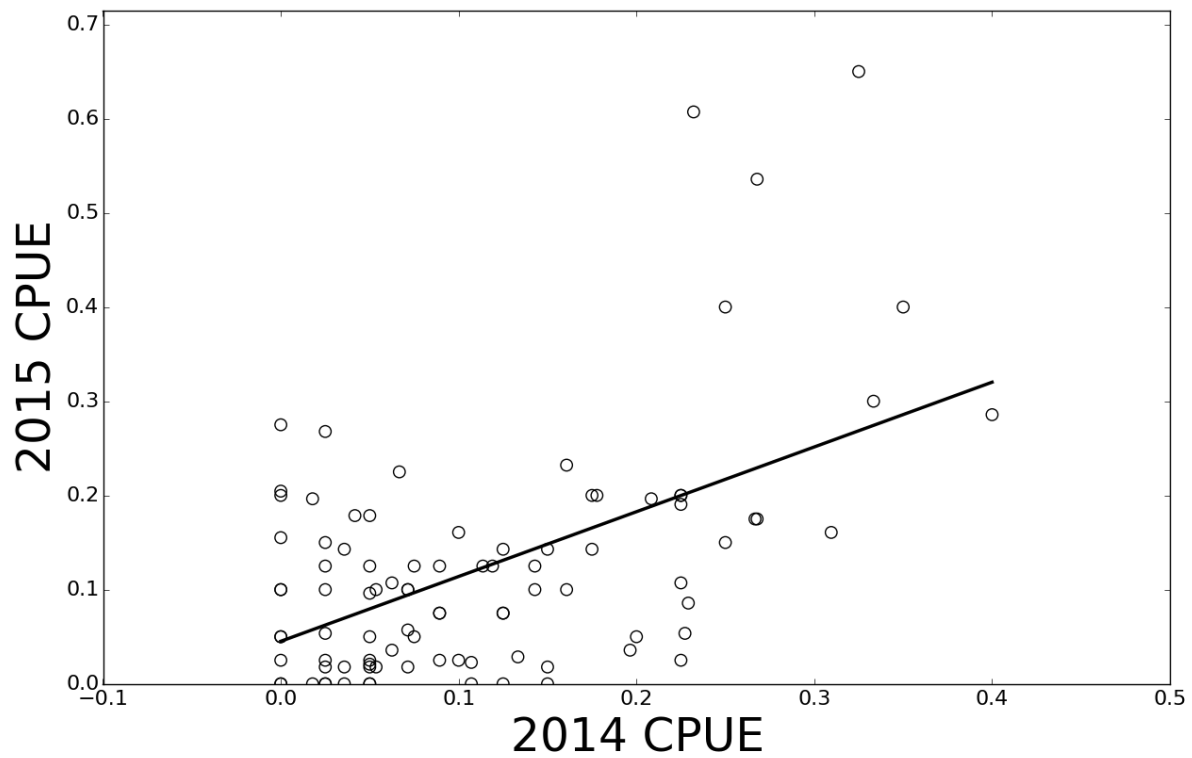
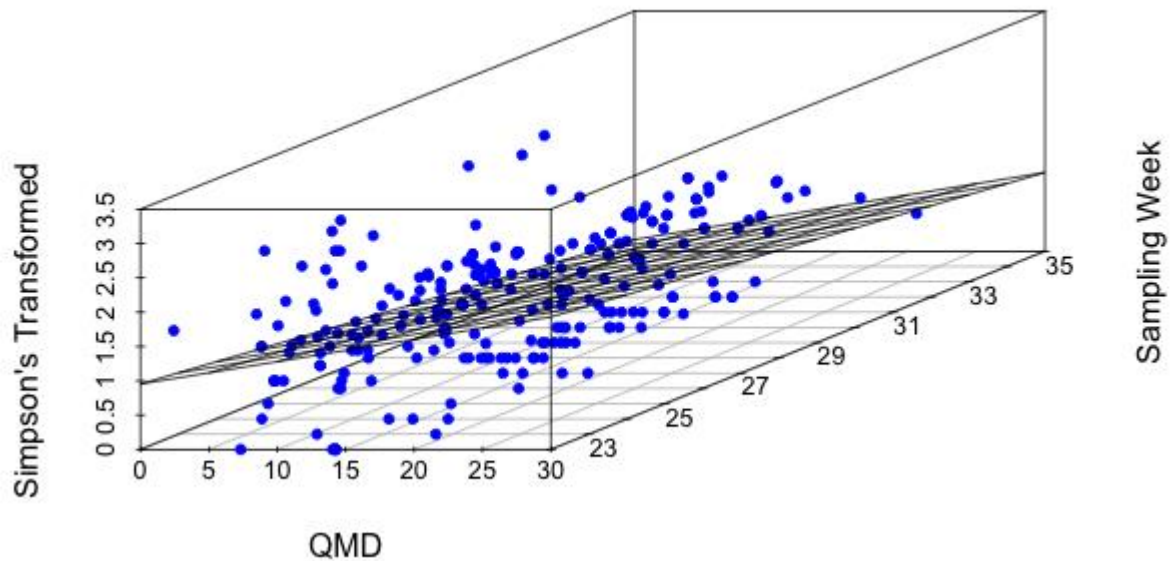


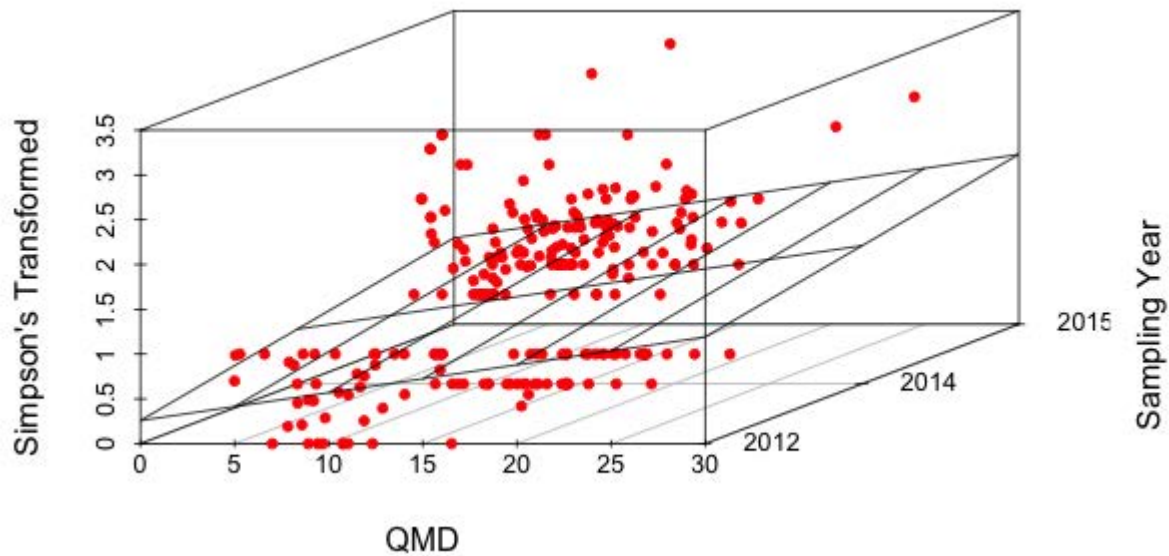
Figure CPUE_REG. Catch per unit effort values between plots sampled in both 2014 and 2015 with linear regression line ($R^2 = 0.28$, $p < 0.0001$, $df = 249$) for fisher prey plots on Stirling in northern California.



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.



RESEARCH ARTICLE

Dynamic occupancy modelling reveals a hierarchy of competition among fishers, grey foxes and ringtails

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Abstract

1. Determining how species coexist is critical for understanding functional diversity, niche partitioning and interspecific interactions. Identifying the direct and indirect interactions among sympatric carnivores that enable their coexistence is particularly important to elucidate because they are integral for maintaining ecosystem function.
2. We studied the effects of removing nine fishers (*Pekania pennanti*) on their population dynamics and used this perturbation to elucidate the interspecific interactions among fishers, grey foxes (*Urocyon cinereoargenteus*) and ringtails (*Bassariscus astutus*). Grey foxes (family: Canidae) are likely to compete with fishers due to their similar body sizes and dietary overlap, and ringtails (family: Procyonidae), like fishers, are semi-arboreal species of conservation concern. We used spatial capture–recapture to investigate fisher population numbers and dynamic occupancy models that incorporated interspecific interactions to investigate the effects members of these species had on the colonization and persistence of each other's site occupancy.
3. The fisher population showed no change in density for up to 3 years following the removals of fishers for translocations. In contrast, fisher site occupancy decreased in the years immediately following the translocations. During this same time period, site occupancy by grey foxes increased and remained elevated through the end of the study.
4. We found a complicated hierarchy among fishers, foxes and ringtails. Fishers affected grey fox site persistence negatively but had a positive effect on their colonization. Foxes had a positive effect on ringtail site colonization. Thus, fishers were the dominant small carnivore where present and negatively affected foxes directly and ringtails indirectly.
5. Coexistence among the small carnivores we studied appears to reflect dynamic spatial partitioning. Conservation and management efforts should investigate how intraguild interactions may influence the recolonization of carnivores to previously occupied landscapes.

KEYWORDS

Bassariscus astutus, dynamic occupancy model, Jolly-Seber, *Pekania pennanti*, source community, spatial capture–recapture, translocation, *Urocyon cinereoargenteus*

1 | INTRODUCTION

Carnivores are integral members of ecological communities. The top-down pressures they exhibit on sympatric species structure ecosystems and their absences can trigger trophic cascades and effect changes in ecosystem integrity (Estes et al., 2011; Ripple et al., 2014; Ritchie & Johnson, 2009). The interactions among carnivore species can also be important drivers of carnivore community dynamics. Competitive, antagonistic behaviours among large carnivores can result in kleptoparasitism (Périquet, Fritz, & Revilla, 2015), spatial or temporal segregation (Dröge, Creel, Becker, & M'soka, 2017) and, at its most extreme, intraguild predation (Creel, 2001; Palomares & Caro, 1999). Subordinate species have adapted to occupy environments with their dominant competitors by partitioning space, time and resources to pre-empt and mediate conflicts (e.g. Bischof, Ali, Kabir, Hameed, & Nawaz, 2014; Kamler, Ballard, Gilliland, Lemons, & Mote, 2003; Swanson et al., 2014). While a large body of research has accumulated on the many mechanisms for large-carnivore coexistence, interspecific interactions governing coexistence among small carnivores are less well-studied.

Interspecific interactions within small-carnivore guilds may differ from those observed within large-carnivore guilds. Many small-carnivore guilds include numerous generalist species of similar body sizes whose members overlap in range, diet and habitat requirements (Lesmeister, Nielsen, Schaubert, & Hellgren, 2015; McDonald, Nielsen, Oyana, & Sun, 2008; Remonti et al., 2012). In addition to competing with carnivores of the same guild, smaller carnivores are also subject to top-down pressures from larger, dominant carnivores, and live in a "Landscape of Fear" (Laundré, Hernández, & Ripple, 2010). The structure and dynamics of these small-carnivore guilds have also changed over time due to decreases in the numbers of apex carnivores (i.e. "mesopredator release"; Estes et al., 2011; Prugh et al., 2009; Ripple et al., 2014) and changes in habitat (e.g. Ritchie & Johnson, 2009). The effects on species remaining in ecosystems following such mesopredator releases can include broad-scale decreases of small mammal and songbird populations (e.g. Crooks & Soulé, 1999; Johnson, Isaac, & Fisher, 2007), increases in disease prevalence (e.g. Levi, Kilpatrick, Mangel, & Wilmers, 2012) and economic consequences in developing nations (e.g. Taylor, Ryan, Brashares, & Johnson, 2016). Limited research has examined how changing population numbers of smaller carnivores affect the dynamics of small-carnivore guilds. Investigating the effects that changing numbers of one small carnivore on the populations of other members of its guild is important to understand niche partitioning and coexistence in complex environments. A North American species that is a member of several small-carnivore guilds and that is also a species of conservation concern in the western portions of its range is the fisher (*Pekania* [formerly *Martes*] *pennanti*)

Fishers are small forest-dwelling carnivores whose range decreased substantially after the mid-1800s (Aubry & Lewis, 2003; Krohn, 2012; Powell, 1993). Fishers are members of diverse small-carnivore guilds that can share environments with up to seven other terrestrial mammalian carnivores of their size or slightly smaller (e.g. grey foxes [*Urocyon cinereoargenteus*], spotted and striped skunks

[*Spilogale gracilis*, *Mephitis mephitis*], ringtails [*Bassariscus astutus*], martens [*Martes americana*, *Martes caurina*], raccoons [*Procyon lotor*] and opossums [*Didelphis virginiana*). Although previous research has investigated range overlap and co-occurrence of some of these species (Campbell, 2004; Gompper, Lesmeister, Ray, Malcolm, & Kays, 2016; Sweitzer & Furnas, 2016), the strength and directionality of interspecific interactions influencing abundances and distributions are not well understood. Fishers are also a species of conservation concern in the western United States, as aside from successful translocations, this species has not recolonized many previously occupied habitats (Lewis, Powell, & Zielinski, 2012). Studying their interactions with other small carnivores may lend additional insight into any mechanisms that may be influencing recolonization.

Here, we investigated the intraguild dynamics among fishers, grey foxes and ringtails over 8 years using a planned translocation of fishers as an experimental removal. There is strong potential for competitive interactions within this small-carnivore guild because of their similar body sizes, and overlapping functional niches, diets and habitat requirements (Simberloff & Dayan, 1991). Very limited evidence exists showing that carnivores of this size engage in intraguild predation and, as such, spatial and temporal partitioning appears to play a large role in reducing their ecological overlap in space and time (de Satgé, Teichman, & Cristescu, 2017). We hypothesized that a hierarchy within small-carnivore guilds influences the distribution of its members on the landscape. We tested three predictions of this hypothesis, that (1) there would be a dominant carnivore within this small-carnivore guild that would limit the presence of subordinate carnivores, (2) subordinate carnivores would increase in their occurrence when the dominant carnivore is removed from the environment, and (3) there would be indirect effects of the dominant carnivore on other members of the guild that may mediate their occurrence.

2 | MATERIALS AND METHODS

2.1 | Study site

We monitored small carnivores in a portion of the Klamath-Siskiyou Ecoregion in northern California and southern Oregon, USA (henceforth, "Klamath"; Figure 1). Land cover was predominantly conifer forest (53%), along with mixed conifer/broadleaf (21%), broadleaf forest (5%) and open areas (11%; determined using Gradient Nearest Neighbour structure maps; Ohmann, Gregory, Henderson, & Roberts, 2011). Elevation ranged from 472 to 2269 m.

We conducted non-invasive surveys annually between 2006 and 2013 from mid-September to early December. The fisher population in Klamath served as one of several sources for a translocation of fishers to the northern Sierra Nevada and southern Cascade Mountains in winter 2009 and 2010 after our surveys in those years (Facka et al., 2016). A total of nine fishers (4F, 1M in 2009 and 3F, 1M in 2010) were removed from Klamath. We used the removal of these individuals with high reproductive value (Facka et al., 2016; Lewis et al., 2012; Powell et al., 2012) as an experimental manipulation to evaluate changes in the distribution of sympatric small carnivores.

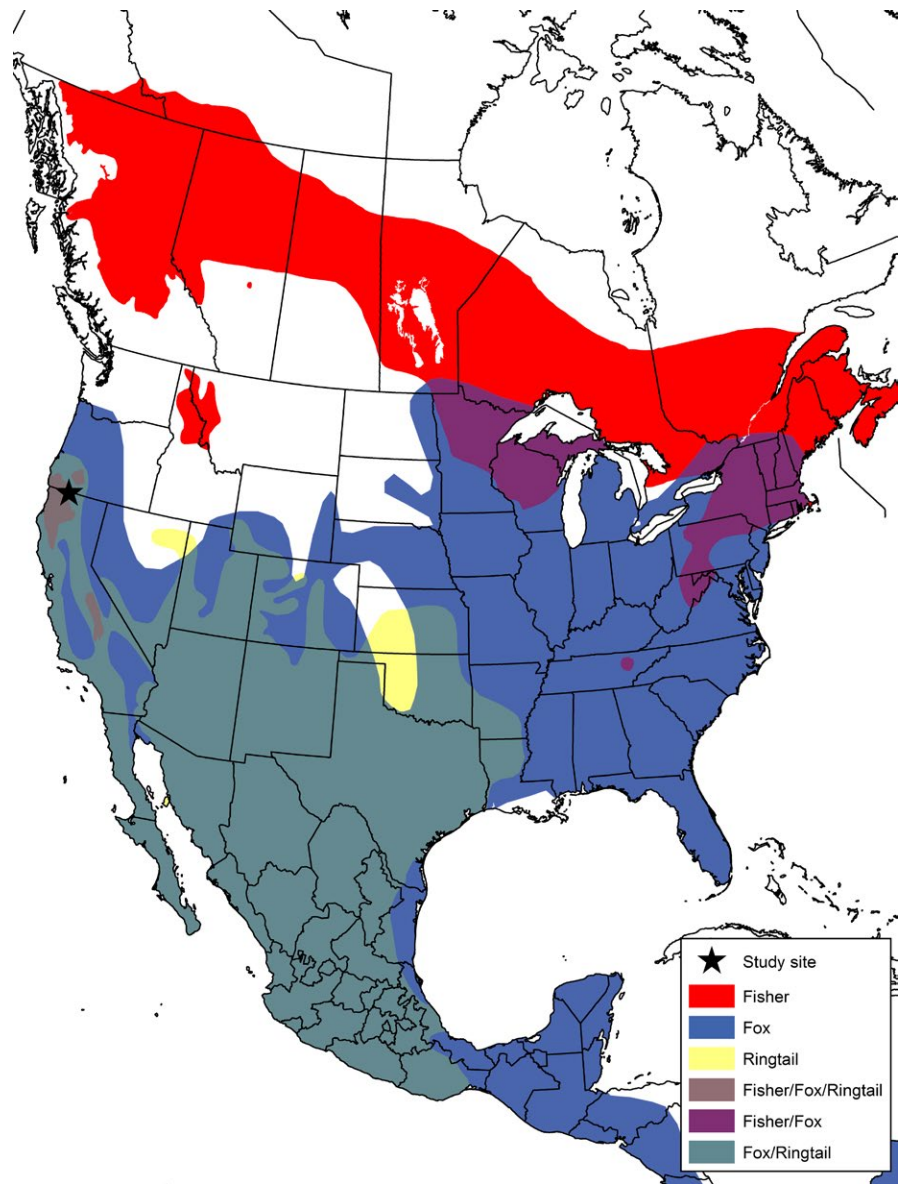


FIGURE 1 The location of our study site and the spatial overlap of fishers, grey foxes and ringtails in North America (Helgen & Reid, 2016; Reid et al., 2016; Roemer et al., 2016)

2.2 | Non-invasive sampling techniques

We surveyed 78 sites in Klamath for 4 to 6 weeks in 2006 and 100 sites in 2007 through 2013, using the same locations each year. We spaced sites 1.3 ± 0.4 km ($M \pm SD$) apart. Each sampling site consisted of a box made of corrugated plastic (Zielinski, Schlexer, Pilgrim, & Schwartz, 2006) baited with a raw chicken leg and a can of wet cat-food. The back of the box was closed with $\frac{1}{2}$ inch (1.3 cm) galvanized hardware cloth and the front was partially obstructed with three wooden slats (Zielinski et al., 2006). We fixed a strip of non-poisonous glueboard (Catchmaster 72MB, USA) to the underside of the bottom wooden slat, so that mammals coming into the box were likely to leave hair with follicles, hereafter “hair samples,” attached to the glue strip to be used for genetic analyses. Boxes were open and functioning for 2708 ± 439 ($M \pm SD$) sampling days per year. Henceforth, we refer to our study area in Klamath as the minimum convex polygon encompassing all of our sampling locations (465 km²)

Starting in 2007, we also fixed to the bottom of the sampling box a metal track plate covered with contact paper and dispersed printer toner to capture track impressions from visiting animals (Belant, 2003; Zielinski & Truex, 1995). By collecting hair samples and track plates from our sampling units, we were able to investigate trends of the fisher population via individual identifications and trends in the occupancy of members of the small-carnivore guild. Units were checked weekly and every hair sample was analysed for the presence of fisher DNA, and all track plates were analysed for the presence of tracks by fishers, grey foxes and ringtails.

We focused on these three species because fishers and foxes are most likely to be the dominant small carnivores in this environment based on their body sizes. Furthermore, previous research has indicated distributional overlap of these three species (Figure 1; Helgen & Reid, 2016; Reid, Schipper, & Timm, 2016; Roemer, Cypher, & List, 2016), but the interactions that govern their partitioning at finer spatial scales are unknown.

2.3 | Individual identification of fishers with genetics

DNA was extracted from hair samples and genotyped using the multi-tube approach recommended for non-invasive samples (Tucker, Schwartz, Truex, Wisely, & Allendorf, 2014). All samples were amplified twice at each locus, and some were amplified a third time if the initial amplifications resulted in a lack of consensus scores (Schwartz & Monfort, 2008). Samples were discarded as being of insufficient quality for genetic analyses if these three scores did not prove to be consistent. We used DROPOUT to screen for any potential errors in genotyping for samples of sufficient DNA quality (McKelvey & Schwartz, 2004; Schwartz, Cushman, McKelvey, Hayden, & Engkjer, 2006). Samples identified to contain putative errors were re-amplified an additional three times.

2.4 | Spatial Jolly-Seber open population model

We evaluated the effects of removing fishers on population dynamics in Klamath with a spatial Jolly-Seber open population model (Jolly, 1965; Seber, 1965). Traditional sampling for these models results in data from secondary sampling periods (e.g. weeks) across >1 primary sampling periods (e.g. years), often referred to as the “robust design” (Pollock, 1982). Recently developed Jolly-Seber models fit with spatial capture-recapture integrate the movements of individuals directly into their formulation to provide spatially explicit estimates of population density (Gardner, Reppucci, Lucherini, & Royle, 2010; Royle, Chandler, Sollmann, & Gardner, 2014). Spatial Jolly-Seber population models have the capacity to improve the precision of parameter estimates (e.g. Ergon & Gardner, 2014; Whittington & Sawaya, 2015).

In these models, the spatially explicit sampled locations of each individual i are assumed to be a function of the distance to its latent activity centre s_i (Efford, 2004; Royle & Young, 2008). Locations of each s_i are then calculated by estimating individual-specific encounter probabilities and the distribution of activity centres in the defined state-space (S). The number of estimated s_i of individuals observed or unobserved in the state-space equates to the population size. We defined S as a continuous state-space bounded by the outside of a rectangular 9-km buffer around our sampling units, greater than three times the size of the estimated movement parameter (σ ; Royle et al., 2014). Initial models were run using a discrete state-space to investigate if elevation or canopy cover affected the distribution of fishers in Klamath, but these covariates were not significant (95% credible intervals overlapped 0); we reparametrized the model using a continuous state-space.

We fit our models using data augmentation methods (Royle & Dorazio, 2008; Royle & Young, 2008). We introduced 550 all-zero encounter histories (nz) to our population of observed individuals (n) to prevent any truncation of the number of fishers with activity centres in S . We also introduced the partially latent variable z_{it} indicating population membership for observed or unobserved individual i in year t . Following Gardner et al. (2010), $z_{i1} \sim \text{Bernoulli}(\Psi_{\text{pop}})$ for year $t = 1$ and $z_{it} \sim \text{Bernoulli}(\mu_{it})$ for all years $t > 1$, where:

$$\mu_{it} = (z_{it-1} \times \Phi_{t-1}) + (A_{it-1}\gamma_t)$$

Here, the probability that an individual is estimated to be in the population in years $t > 1$ is a function of survival (Φ_{t-1}) from the previous year if already present (z_{it-1}) or of the probability that an individual is recruited (γ_t) into the population if not already present (A_{it-1}). In the latter part of this equation, we used the term A_{it-1} to ensure that individuals can only be recruited into the population once such that:

$$A_{it-1} = 1 - \text{step}(a_{it-1})$$

$$a_{it} = \text{sum}(z_{1:t}, t)$$

We defined a positive detection y of individual i in year t at sampling site j in week w as the Bernoulli-distributed random variable: $y_{itjw} \sim \text{Bernoulli}(p_{itjw}g_{itj}z_{it}m_{tjw})$. In this equation, p_{itjw} was the probability of detecting individual i in year t at site j in week w . The detection function g_{itj} described how the encounter rate of individual i decreased as a function of the distance between its activity centre in that year (s_{it}) and the location of sampling site j . The previously described latent variable z_{it} indicated population membership and m_{tjw} was a binary variable indicating whether or not in year t sampling box j was deployed in week w .

We hypothesized that there would be sex-specific detection probabilities and an increased probability of visitation following an initial detection as has been shown in previous studies (Linden, Fuller, Royle, & Hare, 2017; Popescu, de Valpine, & Sweitzer, 2014; Sweitzer, Furnas, Barrett, Purcell, & Thompson, 2016). We also hypothesized that time of year may influence detection, and predicted that individuals would be more likely to visit sampling locations closer to winter due to decreased food availability. We modelled the logit-linear mean encounter rate (p_{itjw}) as:

$$\text{logit}(p_{itjw}) = \alpha_0 + \alpha_1 \times \text{sex}_i + \alpha_2 \times \text{previousdetection}_{itjw} + \alpha_3 + \text{week}_w$$

where the mean encounter rate was a function of an intercept (α_0), a sex effect (α_1), an effect of a binary variable indicating whether or not the individual had visited this site in a previous week of that year (α_2), and a week-specific effect to estimate the effect of seasonality (α_3). Initial models varied α_0 by year, but there were no significant differences among α_0 in this parametrization (95% credible intervals overlapped each other) and this parametrization was ultimately dropped.

We modelled the detection function (g_{itj}) from a Gaussian encounter probability such that:

$$g_{itj} = e^{(-d_{itj}^2 / 2\sigma_k^2)}$$

where d_{itj} is the Euclidean distance between the sampling units where an individual was located and the location of its activity centre that year (s_{it}), and σ_k is the SD of a bivariate normal distribution reflecting space-use, also called the “movement parameter”, and was calculated independently for each sex (k). Sex was estimated as the Bernoulli-distributed random variable: $\text{Sex}_i \sim \text{Bernoulli}(\Psi_{\text{sex}})$. Data were collected during the fisher dispersal season (Arthur, Paragi, & Krohn, 1993; Matthews et al., 2013), but the movement parameter is robust to violations of stationary movement around the latent activity centres (Royle, Fuller, & Sutherland, 2016). We

modelled activity centres independently each year for every individual as a homogeneous Poisson point process in S (Royle et al., 2014). Population size and density were calculated as a function of the number of estimated activity centres located within the Klamath study area.

2.5 | Assessment of fisher translocations

We performed an analysis of the spatially explicit fisher densities to investigate how the removal of individuals for translocation affected the distribution of fishers in Klamath. We did this post hoc because we (1) wanted to know if the spatial capture–recapture model would identify a decrease in fisher numbers around the removal sites without censoring individuals beforehand, and (2) we were surprised that the overall fisher density did not change in our study area while a known quantity of individuals were removed for translocations. To investigate if the number of fishers declined in areas around the removal sites, we overlaid our study area with a 1×1 -km grid and calculated the estimated number of fisher activity centres per km^2 in this grid. We used a generalized linear model for this analysis such that:

$$\Delta \frac{\text{fisher}}{\text{km}^2} = \beta_0 + \beta_1 \times \text{distance} + \beta_2 \times \text{age} + \beta_3 \times \text{sex} + \beta_4 \times \text{distance} \times \text{sex}$$

where the change in the estimated fisher density between the year before (2009) and after the first removal (2010) per km^2 was modelled to vary by an intercept (β_0), the distance between the grid cell centre and the nearest removal site (β_1), the age of the nearest individual removed (β_2), a binary variable indicating the sex of the individual removed closest to this grid cell centre (β_3), and an interaction between distance and sex of the individual removed (β_4). We included age, sex and an interaction between distance and sex to test the hypothesis that removals of females and older individuals would have a larger effect on fisher density. Only one adult male was removed in 2009, so we were unable to investigate an interaction between age and sex. We performed this analysis on the difference between 2009 and 2010 to isolate this perturbation from any other lag-effects that may be present following the second year of removals. We also only included grid cells within 6 km of a removal site for this analysis because this was twice as large as the radius of a male home range and was likely to capture any variation stemming from the removals.

2.6 | Dynamic multi-species occupancy model with interspecific interactions

Estimating the occupancy of species while accounting for imperfect detection is a technique regularly used in ecology (MacKenzie et al., 2002). Many single season multi-species occupancy models have been developed to investigate spatial-temporal overlap and interspecific interactions among species (e.g. Cusack et al., 2017; MacKenzie, Bailey, & Nichols, 2004; Richmond, Hines, &

Beissinger, 2010; Rota et al., 2016). These single season models, however, do not explicitly test the direct effects of the occupancy of one species on the occupancy of another species at the same site in the future. Dynamic occupancy models account for changes in occupancy over time as a function of the probability of site colonization and persistence (MacKenzie, Nichols, Hines, Knutson, & Franklin, 2003). We developed a novel implementation of a dynamic multi-species occupancy model to investigate carnivore guild dynamics in Klamath following the parameterization of Royle and Kéry (2007). In this parameterization, a detection y of species s in year t at site j in week w (y_{stjw}) was modelled as the Bernoulli-distributed random variable:

$$y_{stjw} \sim \text{Bernoulli}(z_{stj} p_{sw} m_{tjw})$$

where z_{stj} indicates the occupancy of species s in year t at site j , p_{sw} is the probability of detecting species s in week w and m_{tjw} is a binary variable indicating whether or not in year t a sampling box was deployed at sampling site j in week w . Similar to the spatial capture–recapture model, we hypothesized that time of year influenced detection, and we predicted that individuals would be more likely to visit sampling locations closer to winter due to decreased food availability and an increased probability of a previous visit. We modelled the logit-linear mean probability of detection p_{sw} as:

$$\text{logit}(p_{sw}) = \alpha_{0s} + \alpha_{1s} \times \text{week}_w$$

where p_{sw} is a function of a species-specific intercept (α_{0s}) and a species-specific effect of the week of sampling on detection coded as a time-series variable (α_{1s}). We were unable to disentangle a behavioural effect of a repeat visitation vs. a seasonal effect in the occupancy analysis. Thus, the effect of week investigates both of these hypotheses.

In year $t = 1$ (2007), occupancy was modelled as a latent random variable. In years $t > 1$ (2008–2013), we modelled site occupancy as:

$$z_{sjt} \sim (z_{sj,t-1} \times \Phi_{sjt}) + (1 - z_{sj,t-1} \times \gamma_{sjt})$$

where Φ_{sjt} is the probability of a site occupied in the previous year remaining occupied in the following year (i.e. persistence) and γ_{sjt} is the probability of a site previously unoccupied becoming occupied in the following year (i.e. colonization). We did not include any habitat covariates in these analyses because we were primarily interested in the interspecific interactions among these species.

We modelled persistence and colonization as a function of the estimated occupancy of one or two other species to incorporate interspecific interactions within the small carnivore guild in one model. Interspecific interactions among carnivores are often determined by body size (Polis, Myers, & Holt, 1989). Fishers and grey foxes are similar in size (2–6 kgs), but ringtails are smaller (0.7–1.5 kgs). We developed hierarchical models in which the occupancy of foxes could influence the persistence and colonization of fishers, fishers could influence the persistence and colonization of foxes and both fishers and foxes could influence the persistence and colonization of ringtails, such that:

$$\begin{aligned}
\text{fisher persistence: } \text{logit}(\Phi_{\text{fisher},jt}) &= \beta_{0\text{fisher}} + \beta_{1\text{fisher}} \times Z_{\text{fox},jt-1} \\
\text{fisher colonization: } \text{logit}(\gamma_{\text{fisher},jt}) &= \delta_{0\text{fisher}} + \delta_{1\text{fisher}} \times Z_{\text{fox},jt-1} \\
\text{fisher persistence: } \text{logit}(\Phi_{\text{fox},jt}) &= \beta_{0\text{fox}} + \beta_{1\text{fox}} \times Z_{\text{fisher},jt-1} \\
\text{fisher colonization: } \text{logit}(\gamma_{\text{fox},jt}) &= \delta_{0\text{fox}} + \delta_{1\text{fox}} \times Z_{\text{fisher},jt-1} \\
\text{fisher persistence: } \text{logit}(\Phi_{\text{ringtail},jt}) &= \beta_{0\text{ringtail}} + \beta_{1\text{ringtail}} \times Z_{\text{fisher},jt-1} + \beta_{2\text{ringtail}} \times Z_{\text{fox},jt-1} \\
\text{fisher colonization: } \text{logit}(\gamma_{\text{ringtail},jt}) &= \delta_{0\text{ringtail}} + \delta_{1\text{ringtail}} \times Z_{\text{fisher},jt-1} + \delta_{2\text{ringtail}} \times Z_{\text{fox},jt-1}
\end{aligned}$$

where Φ_{ijt} and γ_{ijt} are functions of a species-specific intercept (β_0) and effects of the estimated occupancy of one or two other species at the same site in the previous year on site persistence (β_1 , β_2) and colonization (δ_1 , δ_2). Full model code is provided in Appendix S1.

2.7 | Model fitting and assessment

We fit our models using a Bayesian approach and the Markov-chain Monte Carlo (MCMC) methods of JAGS (Plummer, 2003) with the jagsUI package (Kellner, 2016) in R v. 3.2.3 (R Core Team, 2016). We used uninformative prior distributions for all parameters. Spatial capture-recapture parameter estimates were calculated from 4,000 MCMC samples, taken from four chains run for 3,000 iterations, thinned by three, following an adaptation and burn-in period of 1,000 iterations. The assessment of the fisher translocation on density and occupancy analyses parameters were calculated from 40,000 MCMC samples, taken from four chains run for 30,000 iterations, thinned by three, following a burn-in and adaptation period of 10,000 iterations. We assessed model convergence by examining trace plots and \hat{R} values for parameter estimates (Gelman & Hill, 2007; Gelman et al., 2014). Descriptive statistics are presented as yearly $M \pm SD$ and posterior parameter estimates are presented as medians and 95% credible intervals (CIs). Parameters for which the 95% CI did not overlap zero were considered to be statistically significant.

3 | RESULTS

We collected ($M \pm SD$) 226 ± 28 hair samples that were submitted for genetic analyses each year. Of these samples, 212 ± 20 (91 ± 6%) were of sufficient quality to identify species identity from 2007 to 2013. Hair samples identified not to be fisher in 2006 were not further analysed to species. For samples identified to be fisher, $84 \pm 8\%$ annually were of sufficient quality to determine sex and individual identification. The number of unique individuals sampled each year showed little variation; we identified 139 unique fishers from 2006 to 2013, with 27 ± 3 individual fishers detected annually. We genetically identified more males in 5 years, more females in 2 years, and an equal number in 1 year (number males:females annually = 1.2 ± 0.4). Fishers were detected at multiple boxes each year (number survey units that individuals visited = 1.7 ± 1.2) and re-detected across years (number of individuals sampled that were present in previous years = 16.6 ± 2.8). Between 2007 and 2013, we annually collected 210 ± 30 track plates, with fisher identified on 73 ± 16 (35 ± 8%), grey fox on 49 ± 18 (24 ± 10%) and ringtail on 47 ± 13 (23 ± 6%) of them.

The spatial capture-recapture model failed to detect any changes in the fisher population in Klamath stemming from the removals in 2009 and 2010 (Figure 2). Estimated locations of fisher activity centres, however, varied each year with a larger decrease in the number of activity centres where fishers were removed (2010–2011; Figure 3, Table 1). Regions of Klamath located closer to the sites of fisher removals had a larger decrease in density from 2009 to 2010 (Table 1). Grid cells located near female removals had a smaller change in density than did grid cells located near male removals, and the interaction between distance and sex was significant (Table 1, Appendix S2). The removal of older animals also led to a larger decrease in localized density (Table 1). Female fishers had a lower probability of detection than male fishers (median [lower 95% CI, upper 95% CI] on the logit scale: $\alpha_0 = -1.69$

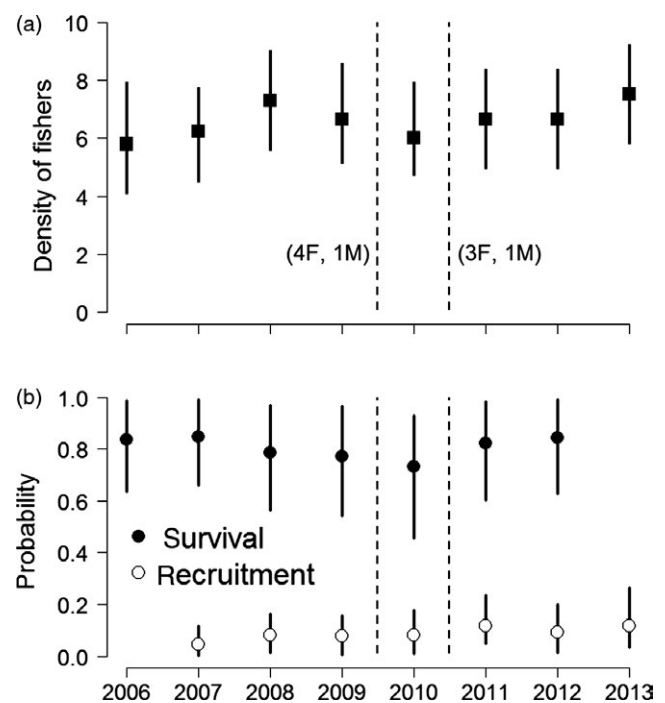


FIGURE 2 The modelled median and 95% credible intervals of (a) fisher density, and (b) apparent survival and recruitment in Klamath, estimated by a spatial Jolly-Seber open population model. Density is reported as the number of fishers per 100 km² within the convex hull encompassing our sampling sites. Apparent survival indicates the probability of an individual surviving to the following year and apparent recruitment indicates the probability of an individual being recruited into the population in that year. The dashed lines indicate the timing of removals along with the quantity and sexes of fishers removed. Density, survival and recruitment estimates were not significantly different among years (95% credible intervals overlapped)

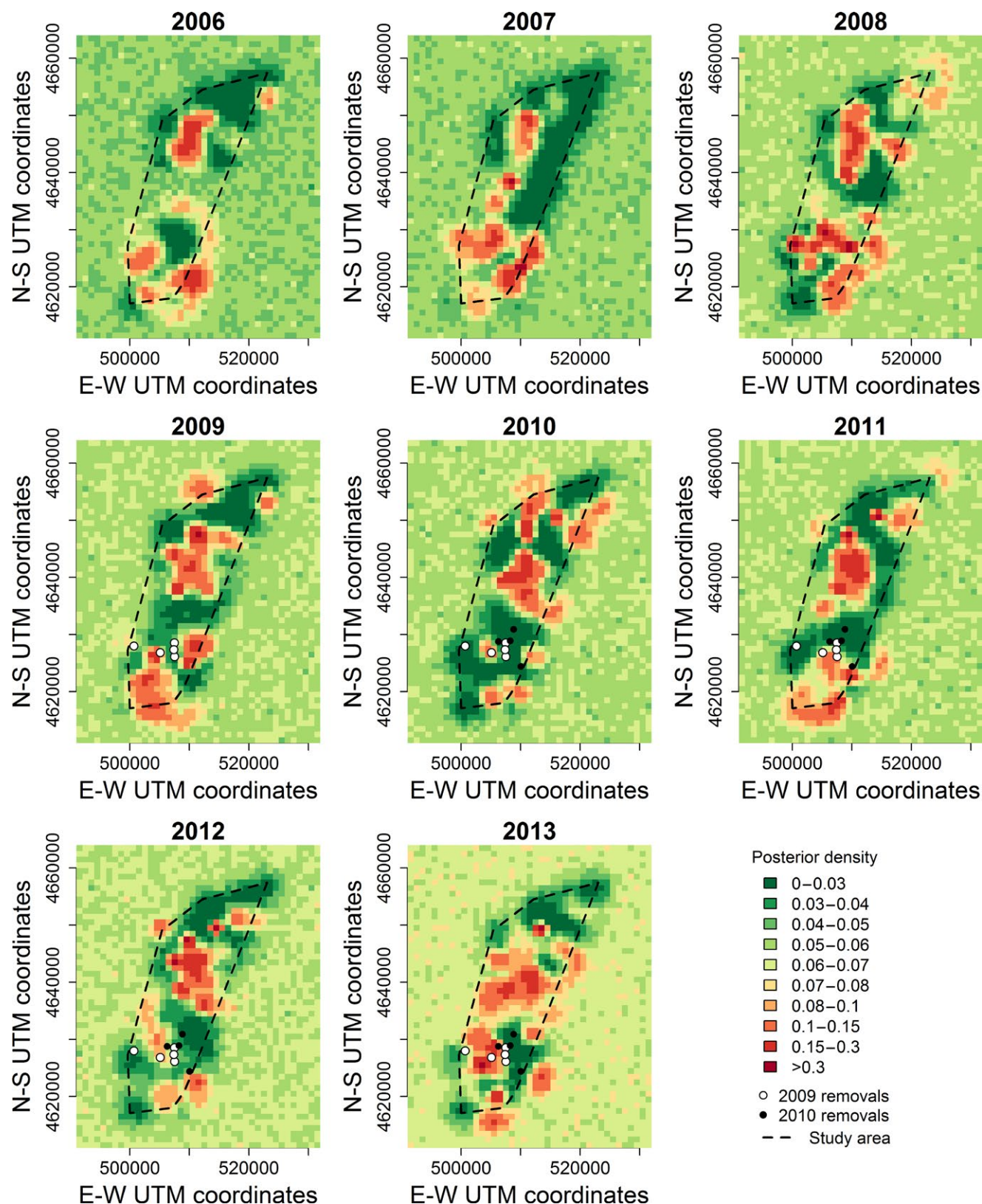


FIGURE 3 Modelled locations of estimated activity centres of fishers in Klamath between 2006 and 2013, as determined with a spatial Jolly-Seber open population model. Warmer colours indicate a larger quantity of fisher activity centres in the 1 × 1-km² grid cell in that year. The locations of live-capture sites used to remove fishers in the winters of 2009 and 2010 are noted with circles, colour coded by year of removal. The convex hull encompassing our sampling sites is indicated with a dashed line

TABLE 1 Results from the analysis investigating the relationship between the change in fisher density following removals for translocation in 2009. All parameters had a statistically significant effect on the change in fisher density from 2009 to 2010 (95% Credible intervals did not overlap 0)

Parameter	M	SD	Credible interval		
			2.5	50	97.5
Intercept	-0.035	0.006	-0.047	-0.035	-0.023
Distance	0.031	0.005	0.02	0.031	0.042
Age	-0.01	0.003	-0.015	-0.010	-0.004
Sex (F)	0.032	0.007	0.019	0.032	0.045
Distance × Sex (F)	-0.023	0.006	-0.035	-0.023	-0.011

$[-2.05, -1.35]$, $\alpha_1 = -1.02 [-1.5, -0.56]$). Detection of fishers increased significantly after an initial detection ($\alpha_2 = 2.35 [1.97, 2.74]$), but did not increase throughout the season ($\alpha_3 = -0.02 [-0.07, 0.02]$). Estimates of σ for male and female fishers did not significantly differ (1.85 km [1.69, 2.04] and 1.64 km [1.35, 2.04], respectively). The sex ratio of fishers in Klamath was estimated to be female biased ($\Psi_{\text{sex}} = 0.61 [0.52, 0.69]$). Full model results can be found in Appendix S3.

Occupancy of fishers, foxes and ringtails varied over the course of this study (Figure 4). The proportion of sites estimated occupied by fishers significantly decreased following the first year of removals in 2009 and remained lower than the estimated occupancy prior to the removals through the end of the current study. The proportion of sites occupied by foxes significantly increased during the 3 years following the initial removal of fishers in 2009 (95% credible intervals did not overlap), and remained elevated through 2012 (Figure 4). The proportion of sites occupied by ringtails showed no direct relationship to the timing of the removals of fishers. Fisher occupancy had a significant negative effect on the

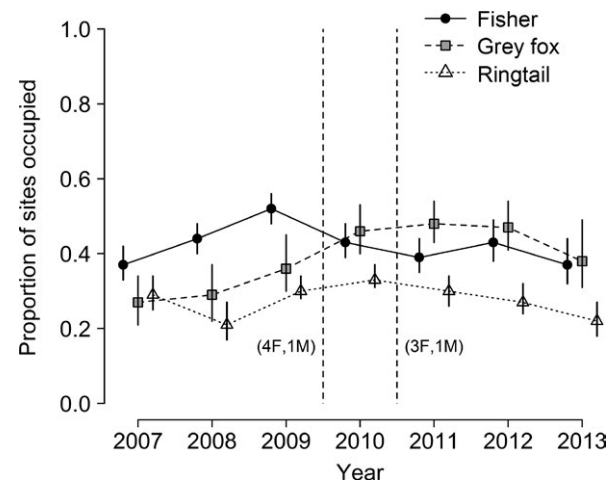


FIGURE 4 The modelled median and 95% credible interval of the proportion of sites occupied by fishers, grey foxes and ringtails from 2007 through 2013 in Klamath. The dashed, vertical lines indicate the timing of removals along with the quantity and sexes of fishers removed

persistence of site occupancy by foxes (β_1 on the logit scale = $-2.58 [-8.4, -0.48]$), but a significant positive effect on their colonization ($\delta_1 = 1.02 [0.14, 2.17]$). Fox occupancy had a significant positive effect on the colonization of sites by ringtails ($\delta_2 = 1.96 [0.72, 4.03]$). No other interspecific interactions were significant (Table 2). Detection of fishers differed by week but detection of the other species did not (Figure 5, Table 2). Full occupancy results can be found in Appendix S4.

4 | DISCUSSION

Our results indicate how dominant small carnivores may elicit spatial turnover in other members within the small-carnivore guild. In the

TABLE 2 Results on the logit scale from the dynamic multi-species occupancy model fit with interspecific interactions. Interspecific interactions were modelled as the effect of a species' site occupancy on the persistence or colonization of another species' occupancy at the same site in the following year. Values for the interspecific interactions are displayed as the effect of the species in the left panel on those in the top panel. All estimates are presented as medians and [95% credible intervals] on the logit scale. Significant parameter effects (95% CI did not overlap zero), not including intercepts, are indicated in bold

	Fisher	Fox	Ringtail
Observation process			
α_0	-1.08 [-1.39, -0.78]	-0.86 [-1.24, -0.48]	-0.86 [-1.27, -0.46]
α_1 (week)	0.11 [0.07, 0.15]	-0.01 [-0.07, 0.05]	0.05 [0, 0.11]
Persistence			
β_0	0.82 [0.33, 1.41]	2.39 [1.28, 9.1]	0.9 [0.09, 1.87]
β_1 - Fisher	-	-1.85 [-8.4, -0.48]	-0.03 [-1, 0.96]
β_2 - Fox	-0.47 [-1.31, 0.34]	-	-0.22 [-1.35, 0.8]
Colonization			
δ_0	-1.08 [-1.47, -0.7]	-2.16 [-3.21, -1.57]	-2.86 [-4.83, -2.08]
δ_1 - Fisher	-	0.97 [0.14, 2.17]	-0.2 [-1.1, 0.66]
δ_2 - Fox	0.1 [-0.57, 0.77]	-	1.83 [0.72, 4.03]

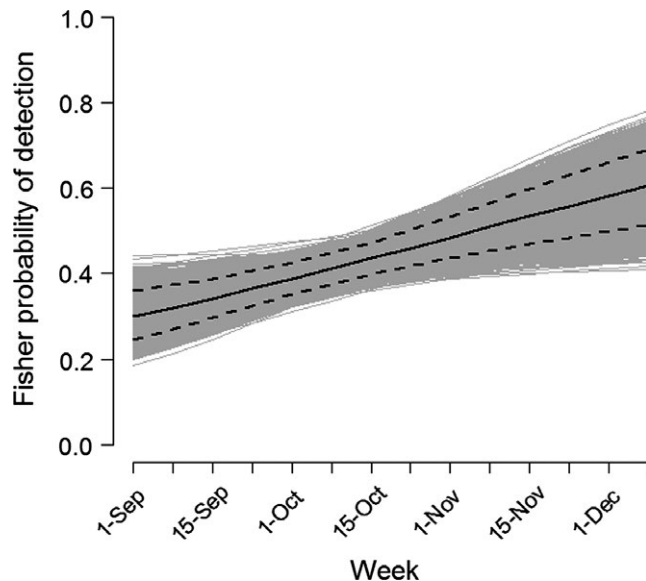


FIGURE 5 The significant effect of sampling week on the probability of detection for fishers, estimated using dynamic occupancy modelling (95% credible interval did not overlap zero). Grey lines indicate the estimated trend from a single MCMC iteration, the solid black line is the median posterior parameter effect, and the dashed lines are the 95% credible interval around this effect

current research, fishers were the dominant predator in the small-predator guild that we monitored, and their presence or absence effected changes in the distribution of grey foxes. Sympatric small carnivores may reduce ecological overlaps through spatial, temporal and niche partitioning (de Satgé et al., 2017). Our findings add to these by showing that subordinate small carnivores may also expand to locations where the numbers of their dominant competitors decline.

Removing nine fishers over 2 years for translocation elsewhere, an approximate 20% of the estimated population in each of the 2 years, had no noticeable effect on the overall density of fishers in Klamath up to 3 years post-removal. These removals did, however, appear to have had an effect on the distribution of the small-carnivore guild. Fishers occupied fewer, and foxes occupied more, sites following the removals. Furthermore, the presence of fishers had a negative effect on sites remaining occupied by foxes but a positive effect on fox site colonization. Thus, the presence of fishers appears to increase turnover in the areas used by foxes, and supports our hypothesis that a hierarchy exists within small-carnivore guilds that affects their distribution on the landscape. A negative relationship between the presence of fishers and the presence of grey foxes in another region of the Pacific States has previously been identified (Campbell, 2004). Our research indicates that this spatial partitioning may also be a function of interspecific competition between fishers and grey foxes.

While we only removed a small percentage of the total fisher population in Klamath, these removals were likely in addition to the natural declines of fishers in Klamath due to death and emigration. Nearly 40% of the fishers we genetically identified each year had not been genotyped previously, and this percentage remained unchanged over time. The relatively stable fisher density over time compared to the

change in occupancy over time, however, is interesting and should be further investigated.

Linden et al. (2017) reported a strong relationship between density and occupancy over broad spatial scales for fisher but our study, similar to that of Matthews, Higley, Yaeger, and Fuller (2011), provides evidence that this may not always be the case when looking at changes over time or at finer spatial scales. We hypothesize that this discrepancy in the current study is because the removals decreased intraspecific competition among fishers. The stable fisher density over time but a change in occupancy could have then resulted from increases in female reproduction and juvenile survival, lack of juvenile dispersal, or increases in the number of fishers immigrating into Klamath; none of these predictions are mutually exclusive. Our results support an increase in female reproduction, an increase in juvenile survival or decreased dispersal based on the locations of estimated activity centres of fishers. If the fisher density in Klamath was unchanged due to an increase in colonizing individuals moving into territories vacated by the removals, then we would predict the spatially explicit densities in Klamath to be static following the removals. There are noticeable gaps in the distribution of fisher activity centres around the locations where animals were removed, especially in 2010 (Figure 3), and our analysis indicated that areas closer to the removals exhibited a greater change in fisher density than areas further away. Both male and female fishers exhibit intrasexual territoriality (Powell, 1993, 1994), making it unlikely that new fishers dispersing into Klamath would colonize regions where fishers were already present. Thus, we surmise the consistent annual densities of fishers we observed was either due to the removals positively affecting reproduction, or that offspring born in Klamath experienced an increase in survival or a decrease in dispersal. The lack of precision with our spatial Jolly-Seber open population model (Figure 2b) prevented us from elucidating the predictions of these hypotheses, but the variation in both apparent survival and recruitment did increase following the initial year of fisher removals (Figure 2b).

The estimated occupancy of ringtails over time appeared to be unaffected by the removals of fishers, but the occupancy of foxes did have a positive effect on the colonization of sites by ringtails. This result, combined with the negative effect of fishers on fox persistence, indicates a possible indirect negative effect of fishers on ringtails. Although ringtails are listed as a sensitive species in Oregon (Oregon Biodiversity Information Center, 2016) and fully protected in California (California Department of Fish and Wildlife, Natural Diversity Database, 2017), very limited, if any, long-term monitoring of ringtail populations has been undertaken. No research to date has identified any interspecific interactions that may affect their distribution or population dynamics. Our results indicate that the turnover fishers elicit on sites occupied by foxes may ultimately have a negative effect on ringtails.

The links between decreasing apex carnivore numbers and subsequent increases in the numbers of mesopredators are well documented (e.g. Johnson et al., 2007; Ritchie & Johnson, 2009), but very limited research has investigated how sympatric mesopredator populations respond to changing numbers of other mesopredators. Our

research indicates that the documented world-wide increases in the number of mesopredators (Prugh et al., 2009) may not only be explained by mesopredator releases but that changing populations of competing mesopredators should also be included in competing hypotheses. For example, declines in the number of fishers, regardless of changing numbers of apex carnivores, may be a source for increased numbers or distribution of grey foxes.

Understanding what currently limits fisher recolonization and expansion into historically occupied habitats is a priority for long-term conservation of fishers in the Pacific States (Naney et al., 2012). Predation by sympatric, larger carnivores (e.g. bobcats, mountain lions) has been identified as one of the greatest natural sources of mortality for fishers (Gabriel et al., 2015; Wengert et al., 2014). Our results suggest that interspecific interactions within the mesopredator guild is a previously unexplored hypothesis that requires attention. Historical and contemporary distributions of fishers and grey foxes overlap in the Pacific States (Figure 1) and individuals of both species are generalist carnivores. While fishers appear limited by habitat (Lofroth et al., 2010), foxes occupy a wide diversity of habitats (Cypher, 2003). We posit that densities of grey foxes are high in places historically, but not presently, occupied by fishers due to the wide distribution of acceptable habitats and lack of fishers. Recolonization of these areas by fishers now, after 100 years of absence, could be hampered by interspecific competition with foxes or predation by larger carnivores (LaPoint, Belant, & Kays, 2015).

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AUTHORS' CONTRIBUTIONS

J.Y., R.C., S.F., M.S., and R.P. conceived the sampling design and field methods; R.S., J.Y., S.F., R.C., and D.G. collected data; D.G. and S.M. analysed the data and led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

DATA ACCESSIBILITY

All data are available from the Dryad Digital Repository <https://doi.org/10.5061/dryad.qb8m4> (Green et al., 2018).

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SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

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APPENDIX 5

Noninvasive Sampling Protocol for Fishers on the Stirling Management Area of Sierra Pacific Industries

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Noninvasive Survey Protocol for Fishers

Reintroductions are conservation measures intended to restore extirpated populations to parts of the former range of a species and can also offer opportunities to test hypotheses addressing habitat conditions and population establishment (IUCN/SSC 2013). Sufficient post-release monitoring, however, is rare for wildlife reintroductions and is increasingly being recognized as essential to measure success in reestablishing a self-sustaining population. For example, monitoring can identify factors that may affect population viability and species recovery (Seddon et al. 2007). Understanding the processes that affect reintroduction success through sufficient post-release monitoring are important because reintroductions, particularly for mammalian carnivores, are expensive, time consuming, and can be controversial (Griffith et al. 1989, Packer et al. 2013).

An effort to reintroduce fishers (*Pekania pennanti*) to an unoccupied portion of their former range in northern California, coupled with a need to better understand fisher population viability in forests managed for timber production, fostered a cooperative venture among the U.S. Fish and Wildlife Service (USFWS), the California Department of Fish and Game (now California Department of Fish and Wildlife), Sierra Pacific Industries (SPI), and North Carolina State University. Project partners agreed to reintroduce fishers to the Stirling Management Area (hereafter “Stirling”) owned and managed for timber production by SPI in the northern Sierra Nevada and southern Cascade Mountains (Callas and Figura 2008).

The Stirling reintroduction involved the capture of 40 fishers (26 females, 14 males) from widely dispersed sites in northern California that were radio-collared and released onto Stirling between 2009 and 2011 (Facka et al. 2016). All reintroduced fishers and many of their progeny were studied intensively using radio telemetry to understand the dynamics of the population and

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to evaluate stakeholder metrics of project success through 2017 (Facka et al. In Prep). A second phase will involve monitoring the reintroduced population over time and will assess the efficacy of the conservation measures outlined in the Candidate Conservation Agreement with Assurances (CCAA) between SPI and the USFWS (USDI Fish and Wildlife Service 2016).

Many reintroductions to date have failed to identify *a priori* latent states of interest and develop, empirically evaluate, and implement long-term monitoring programs to measure them (Fischer and Lindenmayer 2000). For example, site occupancy and population abundance can be latent states of interest when monitoring extant carnivore populations, including fishers (Long and Zielinski 2008, Boitani et al. 2012). Occupancy, in practice, is often used as a surrogate for abundance when monitoring population trends over time (Mackenzie and Nichols 2004). This assumption that there is a positive and linear relationship between occupancy and abundance (MacKenzie and Royle 2005), however, is an area of active research (Efford and Dawson 2012, Hayes and Monfils 2015, Steenweg et al. 2018) and has called this predicted relationship into question. For fishers, Linden et al. (2017) showed high correspondence between site occupancy and abundance at broad spatial scales (e.g., wildlife management units in New York), but Green et al. (2018) and Matthews et al. (2011) have reported discrepancies in occupancy-abundance relationships for fishers at finer spatial scales (e.g., study sites the size of Stirling).

We conducted pilot surveys in Stirling between 2013 and 2016 (Matthews and Green In prep) to evaluate noninvasive population monitoring methods in Stirling. We developed a dynamic occupancy model and a spatial capture-recapture model to identify the strengths and weaknesses of each technique, and to help identify the best way to monitor this reintroduced fisher population over time. Neither our dynamic occupancy model nor our spatial capture-

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recapture model predicted a significant trend over time in fisher occupancy or abundance. The trends that they did exhibit, however, appear to be different. While the estimated occupancy may have increased slightly from 2013 to 2016, the estimated number of female fishers on Stirling appeared to show no such trend.

We recommend the spatially-explicit, annual estimation of population abundance and population change as the appropriate latent-state objectives for monitoring fishers on Stirling. Occupancy through time is unlikely to serve as a reliable surrogate for population size and, thus, will require additional information and cost to validate with other methods. We strongly advocate that non-invasive genetic surveys that identify individual fishers and robustly evaluate changes in population size as the most cost-effective means for monitoring the population on Stirling. Below, we outline a specific protocol with steps to collect sufficient data, making every effort to minimize expense. The data and results produced by this protocol will provide robust, spatially-explicit estimates of the population size of fishers on Stirling through time. Additionally, the protocol provides a means to assess both annual survival and recruitment which can then be related to estimates of habitat quality and measured changes in habitat.

This project represents the first reintroduction of fishers in California and the successful establishment of a population in Stirling (Facka et al. 2016, Powell et al. 2016; Facka et al. In Prep). Evaluating the persistence of the population will require monitoring the population size and trends over time.

SURVEY PROTOCOL

The goal of the recommended survey protocol herein is to cost-effectively estimate the annual population size and to evaluate trends in the fisher population in Stirling. Our

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recommendations are based on the non-invasive survey design used since 2006 to study fisher population sizes and density accurately on the Klamath National Forest (hereafter Klamath) and neighboring privately-owned timberlands (Green et al. 2018). This population served as a source for the translocation of fishers (7 females, 2 males) to Stirling (Powell et al. 2016, Green et al. 2018). We piloted the Klamath survey protocol in Stirling and made iterative, annual adjustments between 2013 and 2016 to determine the most effective way to monitor the Stirling population of fishers (Matthews and Green In prep). The following outlines our recommendations for a cost-effective, annual survey protocol 1) to estimate fisher population size, apparent survival, and apparent reproduction; 2) to monitor fisher population trend; and 3) to estimate the co-occurrence of other mesocarnivore and prey species. We recommend implementing non-invasive survey techniques to achieve survey goals. The survey design uses non-invasive, genetic-based techniques with sooted track plates to individually identify fishers and identify the presence of co-occurring species. Genetic techniques provide individual fisher identification to necessary to estimate fisher population size, apparent survival and reproduction, and trends over time (Gardner et al. 2010, Royle et al. 2014). Sooted track plates provide the benefit of documenting the co-occurrence and the opportunity to model occupancy of other mesocarnivores and potential prey species (e.g., Green et al. 2018).

Survey area and stations

The recommended 330 km² survey area is located in the northern portion of Stirling and overlaps the 2009-2011 translocation release sites, the core of the 2009-2017 Stirling fisher telemetry study area, and the 2013-2016 non-invasive pilot study area (Figure 1). The survey area also overlaps where the highest concentrations of female fishers have lived from 2009 to

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2017 and also the highest predicted habitat quality on Stirling (Facka et al. In Prep). We recommend surveying 145 stations, spaced 1.1 ± 0.3 km ($M \pm SD$) apart (Figure 1). Spacing of survey stations was based on mean female fisher home range sizes estimated from radio-telemetry data (Powell et al. 2016) and Stirling non-invasive survey pilot data (2013-2016; Matthews and Green In prep). Each station is surveyed with a corrugated plastic box that encloses a sooted track plate and a non-poisonous glue board as described below and in Green et al. (2018).

Survey timeline

Stations should be set by the third week in September for a six-week sampling period. Stations should be checked weekly (i.e., 6 checks each year) following the steps outlined below. Surveying the recommended 145 stations (Appendix 1-Stations) will require 1 field-crew leader and 2 technicians for a total of 9 weeks: 1 week of equipment preparation and training, 1 week to set stations, 6 weeks to check stations, and 1 week to demobilize. Year 1 will require an additional 2 weeks of time for each position for purchasing, for construction of sampling boxes, and for organizing supplies.

Assembling materials to build sampling boxes

The non-invasive sampling boxes are made from a 32 inch x 44 inch rectangular piece of corrugated plastic (AKA: Coroplast, Polypropylene fluted sheets; Figure 2). The corrugations (also referred to as flutes) should run parallel to the length of the sheet (i.e. parallel with the 44-inch side) to provide better strength to the boxes when they are built. Constructing the boxes is done by scoring the sheets 4 times across the length of the sheets, creating 4 panels that are 10.25 inches (26 cm) in width, and one that is 3 inches (7.5 cm). Six rectangular holes are then cut out

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of two of the panels to provide places for 3 stakes (1-inch x 2-inches x 18-inches) to slide through.

Fold the sheet of corrugated plastic into an elongated box so that the 3-inch flap is located on the bottom of the box, outside of the tunnel (Figure 3). Attach 2 large (2 inch) binder clips, one at the front and one at back of the box, to hold the flap in place on the bottom of the box. A list of field supplies and vendor links can be found in Appendix 2-Budget. Supplies that are needed to complete construction of the boxes include:

- 3 wooden stakes per box (1 x 2 x 18" boards)
- 14-inch x 14-inch (36 x 36 cm) piece of galvanized wire mesh (hardware cloth, 1/2 inch)
- 4 zip ties
- 30-inch x 7.7-inch metal track plate
- glue strip
- cut piece of white contact/shelf paper
- bait (1 can of cat food, 1 chicken drumstick)
- latex gloves

Selection of microsites in the field

Drive to most logical take-off point for the designated GPS site location. Navigate to the GPS location and select a microsite that is representative of the prevailing habitat within 0.5 ha (1.2 acres; ca. 40 m or 43 yards) around the GPS location. Sites can be relocated closer to a drivable road to increase travel efficiency. Sites should be far enough off the road to be obscured from view from the road. Microsites should be on as level ground as possible. If level ground is not available, place the opening of the boxes lower than the back of the boxes. This will help deter rain, and animals tend to prefer walking up into a box than down into a box. Deploy boxes next to logs where possible and secure the wire mesh backs with material (e.g., rocks) to

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encourage animals to use the fronts of boxes. Make an effort to conceal each box with bark, duff, and other material. Complete the Site Info Form (Appendix 3-Forms).

Assembling boxes in the field

1. Fold the corrugated plastic into a cube.
2. Attach the two binder clips to hold the flap in place (one in the front and one in the back).
3. Fold the chicken wire mesh around the opening of the box on the side without the holes for the stakes. Attach the mesh with 4 zip ties (one per side) to completely cover the rear opening. Fold down the excess chicken wire to prevent entry by animals.
4. Slide 2 wooden stakes into the top two positions of the box.
5. Attach contact paper to the track plate with duct tape, leaving 4-6 inches in the back for the bait. Spread printer toner on the aluminum plate in front of the contact paper with a makeup brush.
6. Place the bait on the back of the track plate and carefully slide the track plate with the bait into the box. Adjust the front binder clip to hold the track plate in place. Make sure to remove the cover sheet from the contact paper before leaving the box.
7. Attach a glue strip to the underside of 1 stake with 3 thumbtacks and a hammer, and carefully slide this into the lowest slot. Make sure to remove the clear plastic cover of the glue strip before leaving the box.

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Checking stations

Return to each sampling box once per week for 6 continuous weeks. Upon arrival, check the box for any sign of visitation. If the box was visited by any animal (i.e. if there are **ANY** tracks of **ANY** critters on the track plates, even those that are not from fishers), then remove the track plate, label the back of it with 1) Station ID, 2) Date, and 3) Your initials.

Slide the track plate into a sheet protector. Check the glue strip to see if there are any hairs attached. If there are visible hairs (or if the box was visited by any critters as visible from the track plates), put gloves on and carefully remove the bottom stake from the box. Place a clear piece of plastic over the glue strip (the clear cover that the glue strip came with), remove the tacks attaching the glue strip to the wooden stake with a screwdriver, snip off the pointy ends of the tacks, fold the glue strip in half (back to back—keeping the folded hair sections away from each other), and place the glue strip in a 50 ml centrifuge tube with ~1 inch of desiccant. Complete the Site Check Form (Appendix 3-Forms). Replace bait as needed—at least every other week.

Determining if a box is “functional” on the datasheet: If it is impossible for an animal to leave either a track plate (e.g. the track plate was pulled from the box) or a genetic sample (e.g. the glue strip was torn in half and eaten) when you come back to check on the box, then the box is *not functional*. Both sampling devices need to be available for the box to have been considered “functional”. Make sure to leave detailed comments on the datasheet as to why the box was not functional for any weeks that you select “Not functional”.

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Sample handling, inventory, and shipping

If collecting a sample for genetic analysis, use gloves to place one of the plastic covers back over the glue strip, and then fold the glue strip in half—keep the sticky sides **AWAY** from each other. Place this glue strip into a 50-ml centrifuge tube filled with 1 inch of desiccant. Label the tube with 1) Sample ID, 2) Station ID, 3) Date, 4) Time, and 5) Your initials.

The Sample ID is the two-digit year, a dash, the Station ID, a dash, and the ordered sample number collected at that station in the current year. For example, sample 17-10-2 was a sample collected in 2017 at station 10 and was the second sample collected at station 10 in 2017. It is important not to confuse sampling week and sample number. For example, sample 17-10-1 (the first sample collected at station 10 in 2017) could have been collected in week 2 and sample 17-10-2 could have been collected in week 5.

It is imperative that samples are handled only with gloves, kept out of the sun, and maintained in a climate controlled room when back from the field. When shipping the samples, make sure that all centrifuge caps are tightly sealed, and place the samples into a gallon plastic zip lock bag. Ship samples via overnight shipping once per week to the National Genomics Center for Wildlife and Fish Conservation, Attn: Kristy Pilgrim, 800 E. Beckwith, Missoula, MT 59801. Include with samples a detailed inventory of samples shipped (Table 1). Email inventory to Kristy Pilgrim (kpilgrim@fs.fed.us) when shipped.

Genetic analyses

Hair samples are shipped to and genetic analyses are conducted by the National Genomics Center for Wildlife and Fish Conservation (USDA Forest Service, Rocky Mountain Research Station, Missoula, Montana, USA). The lab extracts DNA, identifies all samples with

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sufficient DNA to species, and identifies all samples identified as fisher with sufficient DNA to individual using techniques described in Green et al. (2018). Results of genetic identification of hair samples to species are usually available in December and identification of individual fishers are usually available in April.

Statistical analyses

Fisher population size, vital rates, and trends are estimated using a spatial Jolly-Seber open population model (Gardner et al. 2010, Royle et al. 2014, Green et al. 2018). Occupancy of co-occurring species are estimated using dynamic, multi-species occupancy models to monitor species occupancy over time and to investigate specific hypotheses related to their occurrences (Royle and Kéry 2007, Green et al. 2018).

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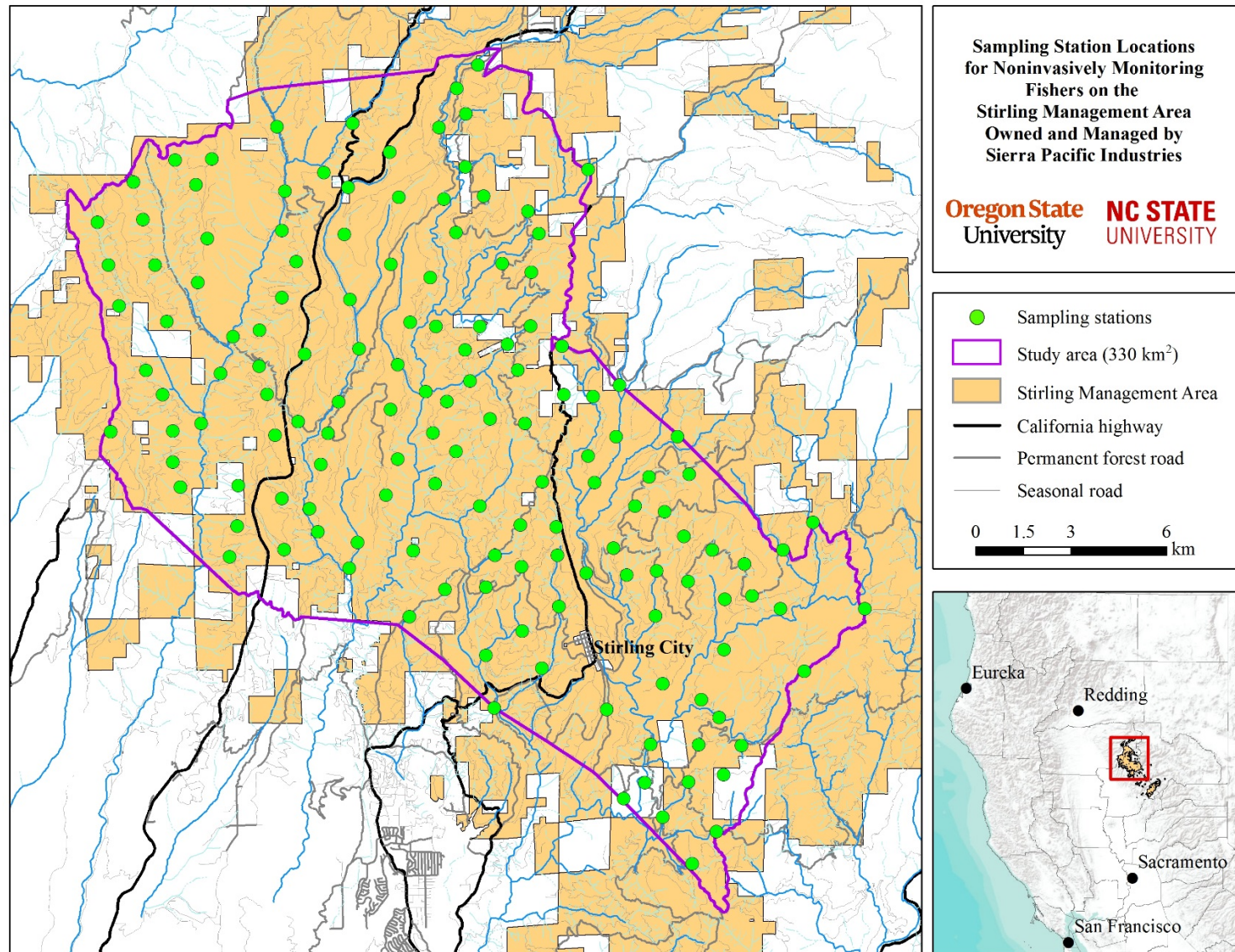
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Table 1. Example of detailed inventory of samples included with shipped samples and emailed to Kristy Pilgrim. Batch equals “1” for the first set of samples shipped for the year from the project site, “2” for the second set, etc.

Sample	Date Collected	Time Collected	Initials	Station	Batch	Comment
15-08B-2	10/15/2015	1017	DSG	08B	3	
15-09B-2	10/15/2015	1030	ELA	09B	3	
15-08A-2	10/15/2015	1054	DSG	08A	3	
15-02B-1	10/15/2015	1122	ELA	02B	3	
15-05A-2	10/15/2015	1200	ELA	05A	3	
15-01B-2	10/15/2015	1255	ELA	01B	3	
15-03A-2	10/15/2015	1355	ELA	03A	3	
15-09A-2	10/15/2015	1500	ELA	09A	3	
15-17B-1	10/19/2015	829	DSG	17B	3	no glue strip, hair sample stuck to thumb tack
15-40A-3	10/19/2015	838	ELA	40A	3	
15-39A-3	10/19/2015	900	ELA	39A	3	
15-17A-2	10/19/2015	917	DSG	17A	3	
15-39B-2	10/19/2015	925	ELA	39B	3	

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Figure 1. Sampling station locations to noninvasively monitor fishers on the Stirling Management Area.

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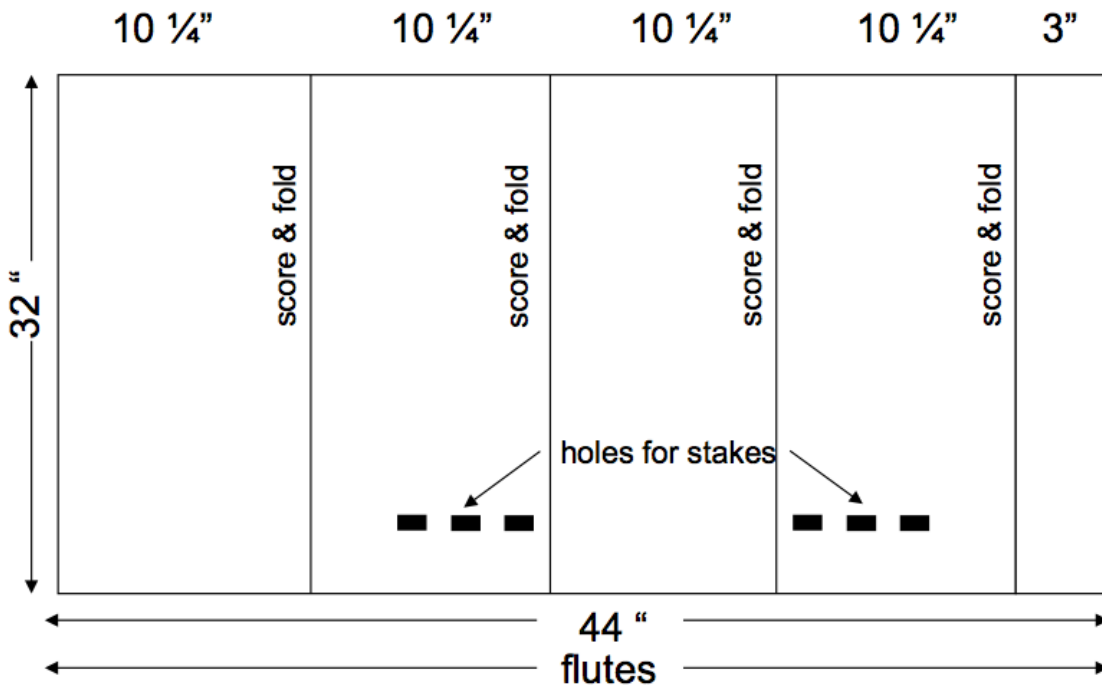


Figure 2. A rectangular piece of corrugated plastic (AKA: Coroplast, Polypropylene fluted sheets) measuring 32 inches x 44 inches showing scoring and hole placement. The corrugations (AKA: flutes) should run parallel to the length of the sheet (i.e. parallel with the 44-inch side).

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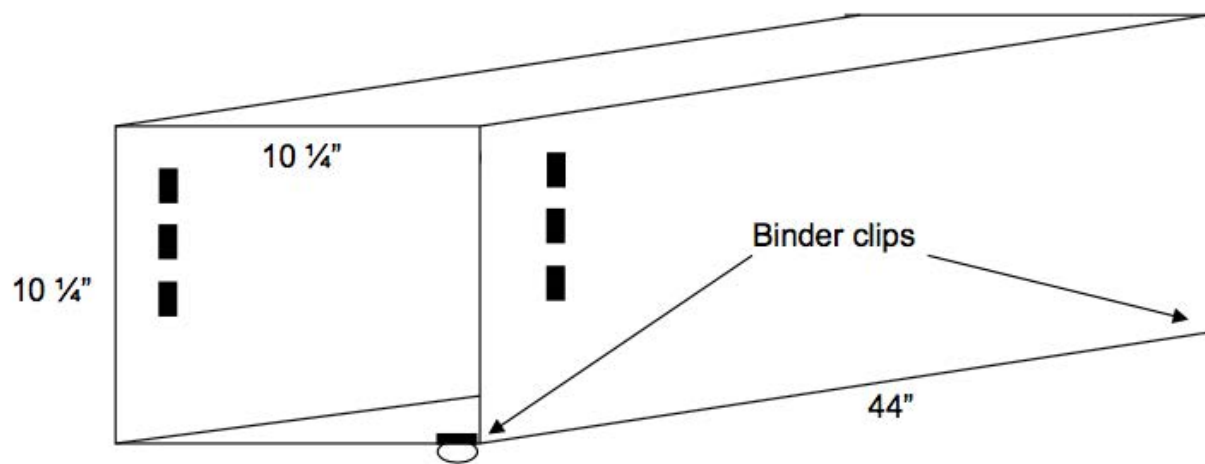


Figure 3. Corrugated plastic sheet folded into a track-plate box.