Landscape fuel reduction, forest fire, and biophysical linkages to local habitat use and local persistence of fishers (*Pekania pennanti*) in Sierra Nevada mixed-conifer forests

R.A. Sweitzer [a,*], B.J. Furnas [b], R.H. Barrett [c], K.L. Purcell [d], C.M. Thompson [d]

[a] Great Basin Institute, 16750 Mount Rose Highway, Reno, NV 89511, USA
[b] California Department of Fish and Wildlife, 601 Locust Street, Redding, CA 96001, USA
[c] Department of Environmental Science, Policy, and Management, University of California, Berkeley, CA 94720, USA
[d] USDA Forest Service, Pacific Southwest Research Station, 2081 E. Sierra Avenue, Fresno, CA 93710, USA

ABSTRACT

Fire suppression and logging have contributed to major changes in California's Sierra Nevada forests. Strategically placed landscape treatments (SPLATS) are being used to reduce density of trees, shrubs, and surface fuels to limit wildfire intensity and spread, but may negatively impact fishers (*Pekania pennanti*). We used camera traps to survey for fishers among 1-km² grid cells of forest habitat in the Sierra National Forest, California. We used single-season ($n = 894$ cells) and multi-season ($n = 361$ cells) occupancy modeling to evaluate responses of fishers to fuel reduction in the 5 years prior to camera surveys. We also assessed occupancy in relation to burn history, elevation, and an index of canopy cover. Camera traps detected fishers most often between 1380 m and 1970 m elevation, and fisher occupancy was maximized at 0.80 at 1640 m elevation. Probability of detection was higher after initial fisher detection, in habitats with high canopy cover, and when surveys were done in fall to spring. Fisher occupancy was positively linked to canopy cover, and trended lower among cells with higher levels of managed burning or forest fires within 25 years of surveys, and in cells where 5 years of cumulative disturbance from restorative fuel reduction was higher. Local persistence declined 24% in areas with more restorative fuel reduction (0–100% of a cell), but was not diminished by prior burning, or disturbance from extractive activities (tree removals for commerce or hazard mitigation). Reduced local persistence by fishers in areas with extensive restorative fuel reduction was likely temporary; evidence from other sources intimated that they would resume higher use within a few years of ecological recovery. The trend for lower occupancy in extensively burned forest cells suggested that forest fires reduced but did not eliminate foraging opportunities for fishers. We also found that wildfires increased in frequency in our study area after the 1980s, and recent fires may increasingly impinge on higher elevation forests with higher fisher occupancy. Forest fuel reduction likely imposes a more limited short term cost to fisher habitat use than previously believed, but less is known about the responses of resting or denning fishers to management disturbance. Fuel reduction treatments could be intensified below ∼1450 m elevation, which may reduce spread of fires into higher elevation forests where fishers are more common, and where denning is focused.

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1. Introduction

Detecting the response of plants and animals to anthropogenic disturbance or degradation of habitat is a necessity when resource managers are required to maintain viable populations of sensitive or threatened species (Clark et al., 2013; Nicholson and Van Manen, 2009). Current disturbance and reduction of wildlife habitat in western forests administered by federal or state agencies is primarily from extractive resource management, wildfires, or management activities intended to improve forest health and limit fire intensity (Stephens et al., 2012). Because of decades of fire suppression, managers are correcting the buildup of small trees and surface fuels by activities that disturb habitats used by sensitive or rare forest vertebrates (Tempel et al., 2014; Truex and Zielinski, 2013).

In 2004 the USDA Forest Service adopted a new strategy for managing national forests in the Sierra Nevada in California that

combined revised guidelines for timber harvest with strategically placed landscape treatments (SPLATS) designed to decrease the intensity and spread of wildfires (Finney, 2001). As part of the Sierra Nevada Forest Plan Amendment (SNFPA; USDA Forest Service, 2004), forest fuel reduction projects (forest projects) typically include sequential application of selective harvest of mid-sized trees up to 76 cm DBH (commercial thinning), mastication of understory trees and shrubs (deposits shard of wood around splintered stems of small trees and shrubs; Amacher et al., 2008), and follow-on managed burning to remove woody debris and surface fuels (Stephens et al., 2009). Forest projects significantly reduce density of trees, shrubs, litter depth and surface fuels of multiple sizes, and canopy and herbaceous cover (Amacher et al., 2008; Garner, 2013; Meyer et al., 2007; Stephens and Moghaddas, 2005).

Zielinski et al. (2013a) and others (Noss et al., 2006) have characterized the types of "mechanical fuel reduction" treatments under this approach as emphasizing restorative management (mastication/mowing, precommercial thinning, managed burns) over traditional extractive activities (group selection thinning, clearcuts, hazard tree logging). Plans by the USDA Forest Service to implement mechanical fuel reduction as detailed in the 2004 SNFPA raised concerns that vertebrates known associated with mature forest habitats such as the fisher (*Pekania pennanti*), spotted owl (*Strix occidentalis occidentalis*), and northern flying squirrel (*Glaucomys sabrinus*) might be negatively impacted (http://www.forestsforever.org/archives_resources/e-alerts/Framework2.20.04.html). Basic ecological information on species-specific habitat use may suggest which organisms are most likely to be affected by extractive or restorative fuel reduction (North, 2012), but quantitative data in the context of past and ongoing forest management is needed for balancing risks without exceeding thresholds where short term negative impacts precipitate persistent decline of sensitive species (Lee et al., 2013; Tempel et al., 2014). Advances in modeling are providing insight on how different configurations of SPLATS mediate how fires propagate across forest landscapes (Syphard et al., 2011; Stephens et al., 2012), and SPLATS can be effective for diminishing the intensity of wildfires in Sierran mixed-conifer forests (Moghaddas et al., 2010; Collins et al., 2011). However, there are often conflicts when forest projects overlap habitats known important for rare forest vertebrates, forcing compromises that may reduce treatment effectiveness (Scheller et al., 2011; Tempel et al., 2014; Thompson et al., 2011).

The fisher is an intermediate-sized mammal associated with late-successional, relatively dense canopy, mixed hardwood-conifer forests in western North America (Raley et al., 2012; Schwartz et al., 2013). This forest carnivore has been the focus of conservation concern in this region since first being petitioned for listing under the U.S. Endangered Species Act in 1990. Fishers are considered likely to be negatively affected by commercial thinning and other types of mechanical fuel reduction because loss of mature forests by extensive timber harvest was at least partly responsible for the reduction in their range and abundance in the early to mid-20th century (Lewis et al., 2012; Zielinski, 2013). Also, empirical research has identified that dense, closed canopy, high biomass forest habitats are important for denning and reproduction by female fishers (Zhao et al., 2012), used as preferred resting sites by male and female fishers (Purcell et al., 2009), and may provide refuge from larger predators that kill fishers at high rates (Wengert et al., 2014; Sweitzer et al., in press). Information from several recent studies is improving our understanding of how fishers respond to habitat changes (Garner, 2013; Thompson et al., 2011; Zielinski et al., 2013a), but we lack an adequate understanding of how the cumulative, recent history of extractive and restorative fuel reduction impacts local scale habitat use by the species.

Fishers in the west coast states of the United States currently exist in 2 remnant populations (1 in northern California, 1 in the southern Sierra Nevada, California), and 3 reintroduced populations (1 each in western Washington, southern Oregon, and northeastern California, USA; U.S. Department of Interior Fish and Wildlife Service, 2014a). While there remains uncertainty regarding the overall spatial extent of decline of fishers in California (Zielinski, 2013), the remnant population of fewer than 500 fishers in the southern Sierra Nevada is separated from fishers in northern California by 400 km (Spencer et al., 2011; Zielinski et al., 2005), and there is evidence for long term genetic isolation as well (Knaus et al., 2011). Spencer et al. (2015a) estimated the current geographic range of fishers in the southern Sierra Nevada at 12,860 km^2, of which $\approx$4400 km^2 was ranked as being of moderate to high suitability Sierran mixed-coniferous forest fisher habitat. It is possible that ongoing and planned extractive and restorative fuel reduction will reduce suitability of those habitats for resting, denning, and foraging fishers.

In October 2014 the U.S. Fish and Wildlife Service proposed to list the West Coast Distinct Population Segment of fisher as threatened under the Endangered Species Act (U.S. Department of Interior Fish and Wildlife Service, 2014b) due to evidence for multiple threats to fisher survival from a diversity of mortality factors (Chow, 2009; Gabriel et al., 2012; Keller et al., 2012; Wengert et al., 2014). The species review specifically identified vegetation management (e.g., fuel reduction activities) as one of several concerns for maintaining suitable habitats for fishers in the West Coast population segment (U.S. Department of the Interior Fish and Wildlife Service, 2014a).

Our research was focused in the Sierra National Forest (Sierra NF) near Bass Lake, California (Sweitzer et al., 2015a; Zhao et al., 2012), where our primary objective was to evaluate the effects of forest projects on local scale habitat use by fishers. Additionally, because there has been debate regarding the temporal duration over which forest fires are detrimental to fisher habitat (Hanson, 2010; Thompson et al., 2011), we were interested in how fisher habitat use would vary in areas with managed burning and previous forest fires. Several studies used detections of fishers at camera traps or track plates to evaluate movement dynamics (Popescu et al., 2014), estimate population size (Sweitzer et al., 2015a), and to evaluate potential change in the range of fishers in the overall Sierra Nevada (Zielinski et al., 2013b). We used camera traps to survey for fishers in 1-km^2 grid cells across an 1127 km^2 forest landscape, and evaluated detection records in an occupancy modeling framework (MacKenzie et al., 2006) to assess how they responded to habitat disturbances from logging activities (extractive management activities), hazardous fuels reduction (restorative fuel reduction), and managed burns + forest fire. We used data from all cells surveyed $\geqslant$1 season (single-season) to evaluate fisher distribution and habitat preferences, and cells surveyed $\geqslant$2 seasons (multi-season) to evaluate fisher response to management activities. We hypothesized that (1) fisher occupancy in single-season cells, and local scale colonization and persistence among multi-season cells would be lower among cells with larger areas of disturbance from recent forest projects, (2) that local extinction would be higher in forest patches with larger areas disturbed by either extractive or restorative mechanical fuel reduction, and (3) that forest patches with larger areas of prior managed burning + forest fires would have lower fisher occupancy and reduced persistence of use.

2. Materials and methods

2.1. Study area

The overall study area was 1127 km^2, and encompassed the non-wilderness region of the Bass Lake Ranger District in the Sierra NF, and a relatively small portion of Yosemite NP where camera

trap surveys were completed in winter 2009 (Fig. 1; Sweitzer et al., 2015a). Our camera surveys were almost entirely on public forest or parkland between 920 m and 2700 m elevation, which bracketed the typical lower elevation (~1050 m) and upper elevation (~2100 m) limits for fisher occurrence in the region. Included within the overall study area was a focal study area encompassing 4 watersheds (Sugar Pine, Nelder Creek, White Chief Branch, Rainier Creek) where camera surveys were repeated each year for assessing the effects of forest projects on fisher habitat use. The focal study area was also a region where information from the Sierra NF indicated that 3 different forest projects were anticipated to occur during the study.

The study area was centered in the California Wildlife Habitat Relations (CWHR) Sierran mixed-conifer forest habitat type (http://www.dfg.ca.gov/biogeodata/cwhr/wildlife_habitats.asp), and the most common tree species in approximate order of abundance for conifers and then hardwoods were incense cedar (*Calocedrus decurrens*), white fir (*Abies concolor*), ponderosa pine (*Pinus ponderosa*), sugar pine (*Pinus lambertiana*), California black oak (*Quercus kelloggii*), mountain dogwood (*Cornus nuttallii*), white alder (*Alnus rhombifolia*), and quaking aspen (*Populus tremuloides*). Giant sequoia (*Sequoiadendron giganteum*) was present, but restricted to remnant populations in the Nelder Grove Historic Area (Sierra NF), and Mariposa Grove (southern Yosemite NP). Common shrubs and tree-like shrubs in the study area included willow (*Salix* sp.), whiteleaf manzanita (*Arctostaphylos viscida*), greenleaf manzanita (*Arctostaphylos patula*), mountain misery (*Chamaebatia foliolosa*), elderberry (*Sambucus caerulea*), bush chinquapin (*Chrysolepis sempervirens*), mountain whitethorn

(*Ceanothus cordulatus*), Sierra gooseberry (*Ribes roezlii*), and buckthorn (*Rhamnus californica*).

2.2. Historical forest management and changes in forest habitats

Historically, logging in the forests of the Sierra Nevada was extensive from the early 1900s until the 1990s (McKelvey and Johnston, 1992), and policies to suppress wildfires were widely implemented after 1920 (Stephens et al., 2009). Before fire suppression, forests in the Sierra Nevada were characterized by frequent low-intensity fires (often ignited by Native Americans, but then by sheep herders; McKelvey and Johnston, 1992), and relatively infrequent large, high severity fires (Collins et al., 2011). Many fires continue to be suppressed, particularly when they occur within the wildland–urban interface (U.S. Forest Service Sierra Nevada Forest Plan Amendment, 2004). Several studies have reported that large, high intensity wildfires are now more common (Allen et al., 2002; Safford et al., 2012), but others contend that neither fire frequency nor fire severity have increased (Hanson and Odion, 2014; Odion et al., 2014). There is consensus that current forests are not characteristic of pre-European settlement in that they have more numerous, and smaller trees and different species compositions (Dolanc et al., 2013; McIntyre et al., 2015; Scholl and Taylor, 2010).

At our Sierra NF study site prior timber harvest removed most of the large, merchantable conifers by the 1970s (Lutz et al., 2009; D. Smith, Personal communication), and with the exception of remnant stands of giant sequoia in the Nelder Grove Historic Area, and in Mariposa Grove, the composition and age structure
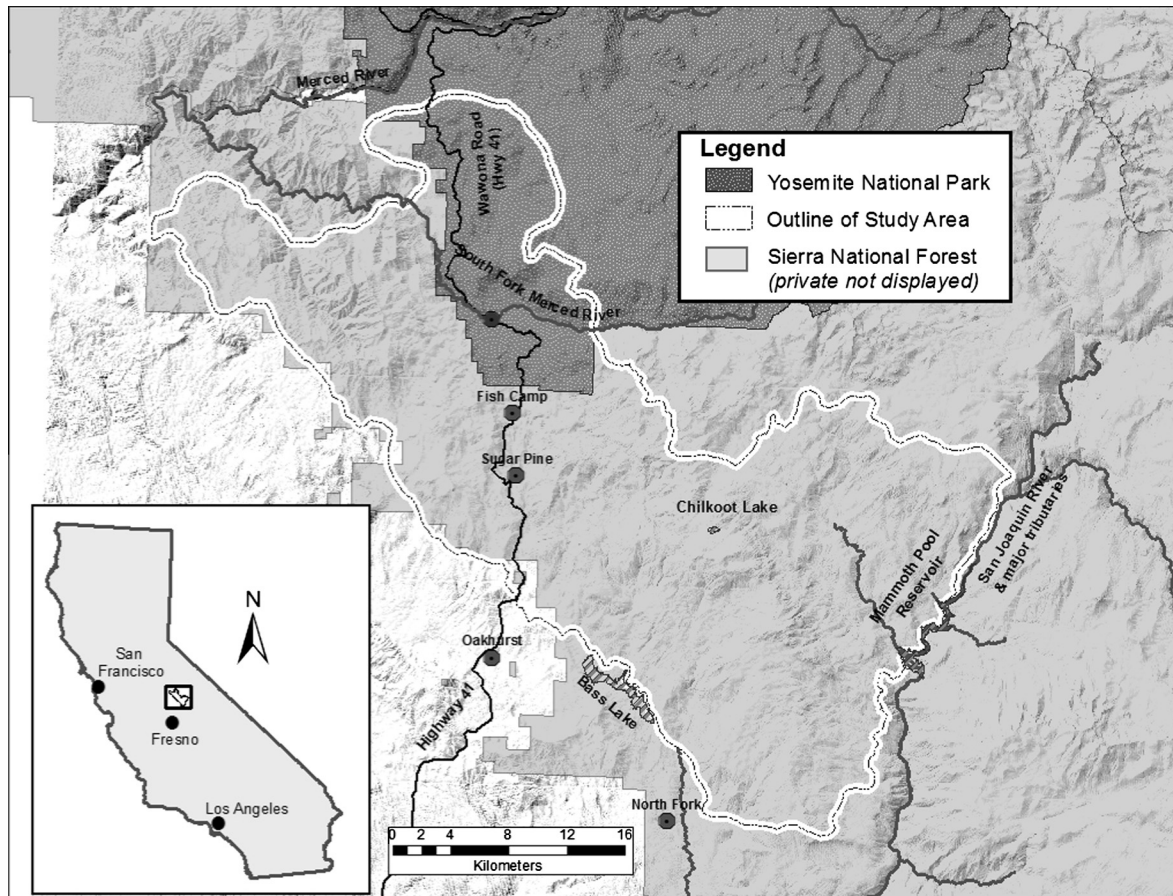

Fig. 1. Study area for research on the camera trap-based occupancy of forest habitats by fishers (*Pekania pennanti*) in the Bass Lake Ranger District, Sierra National Forest, and a portion of southwestern Yosemite National Park.

of the current forest stands are much different than in the early 1900s. Wildlife considerations in our study area were in accordance with the SNFPA (USDA Forest Service, 2004), and included restrictions on commercial thinning, precommercial thinning, and managed burning near known fisher den structures from March 1 to June 30 each year.

2.3. Camera trap surveys

We used a 1-km^2 grid matrix overlain on the study area for planning camera surveys. Motion sensing camera traps (Silent Image Professional, Rapidfire PC85; RECONYX Inc., Holmen, WI) were systematically deployed near the center of 1-km^2 grid cells in the study area at the start of each of 7 camera survey years beginning around October 15 and ending the next year in early October. The research period was from October 2007 to October 2014. Fuel reduction treatments in the Sierra Nevada are not particularly large in size ($\leqslant\sim$25 ha; Garner, 2013; Syphard et al., 2011), and we organized our camera trap surveys at the scale of 1-km^2 cells to capture forest heterogeneity at that spatial scale. A cell size of 1-km^2 was small in relation the average home range size of fishers in our study area (Popescu et al., 2014), which would be problematic if camera trap detections were used to infer abundance from patterns in occupancy. However, our single- and multi-season occupancy analyses were restricted to assessing patterns and changes in local scale (1-km^2 cell) habitat use in relation to varying levels of disturbance from mechanical fuel reduction or wildfires, wherein lack of independence among cells was not a significant concern. We placed camera traps within cells in the matrix by navigating to grid centers with a handheld Global Positioning System unit (Garmin model 60 CSx; Olathe, KS), and placing camera traps at the nearest position including one or more habitat elements known important for fishers (Sweitzer et al., 2015a). Cameras were focused on the base and lower bole of bait trees, upon which we attached baits 1.1–1.5 m up from base, and applied scent lures as attractants. We used small pieces of venison (140–250 g) in a dark colored sock as meat bait for fishers, and 8–10 hard-shell pecans strung onto a length of wire and formed into a small ring as a nut bait for squirrels. Our initial purpose of the nut bait was to index squirrel abundance, but the pecans were also consumed by fishers. The scent lures we used were Hawbaker's Fisher Scent Lure (Fort Loudon, PA), Caven's "Gusto" scent lure (Minnesota Trapline Products, Pennock, MN), and $\sim$4 g of peanut butter smeared on the nut ring. We set the cameras to high trigger sensitivity, 3 pictures per trigger event on a 1 s interval, and no delay for images between trigger events.

Camera traps were typically visited every 8–10 days over 32–40 days in the fall to spring period (October 1–May 31), or 40–50 days in the summer (June 1–September 30) to retrieve images, maintain cameras, and to refresh baits and scent lures. The survey period for cameras deployed in the summer was extended due to indications of lower probability of detection for fishers in the summer than in fall to spring (Popescu et al., 2014). Also, the camera trap protocol varied depending on whether or not the camera station was within the focal study area, or elsewhere. Camera traps placed outside of the focal study area were removed shortly after a fisher detection if it occurred prior to $\sim$35 survey days, unless the unit had been disturbed or was inoperative and required an extension of the survey period. Camera traps that were deployed in summer were removed after 50 survey days unless a fisher was detected beforehand, or if the unit had been inoperative (Sweitzer et al., 2015a). It was possible that camera trap detections were associated with foraging because fishers removed the venison and pecan nut baits when they were still present (Truex and Zielinski, 2013). It was also possible that some detections were fishers that were moving to and from den or rest sites in the vicinity when olfactory cues attracted them to camera baits. For these reasons, we considered detections to represent active fishers (foraging, traveling, etc.).

2.4. Image interpretation and processing

Digital images retrieved from camera traps were reviewed by a technician in Mapview software (www.reconyx.com), wherein all animal images were assigned identifiers including codes for the survey grid and survey session, and the identity of each species detected. Subsequently, the lead author reviewed the image data to identify stations where fishers were detected, providing a quality check to ensure accurate species identification. Image data were used to create a detection history indicating whether a fisher was detected in each 8–10 day survey session. We also recorded notes on camera function, which were used to track both the number of days the camera was deployed (total camera days), and effective camera days (total camera days − nonfunctional camera days) for each grid survey.

2.5. Single year and multiple year camera surveys

We had 2 purposes when using camera traps to survey 1-km^2 cells of habitat for fisher presence, which influenced whether cells were surveyed in a single year (single-season cells) or in 2 or more years (multi-season cells). Cells that were surveyed in 1 year only were typically associated with estimating the overall distribution and habitat use by fishers in the study area. We deployed camera traps in single-season cells independent of planned or prior forest management, and many of these cameras were placed at relatively low or high elevations for understanding patterns in habitat use influenced by elevation. The majority of multi-season surveys were in areas with recent histories of extractive or restorative fuel reduction between 2002 and October 2008, or because forest projects were planned to occur before December 2011. We were not aware of all planned or prior forest projects within the study area when the study was initiated, however, and some multi-season cells were identified and included several years after the first camera year.

All 128 1-km^2 cells that were $\geqslant$50% public lands and encompassed by the 4 watersheds in our focal study area were multi-season survey cells. Most of these cells were repeat surveyed in 7 camera years. Three forest projects within the focal study area were significantly delayed by poor market conditions related to the depressed national economy beginning in 2008. Therefore, and because we had expanded the number of multi-season cells elsewhere in the study area, we assessed responses of fishers to forest disturbance using a combination of single-season and multi-season occupancy analyses. Our *a-priori* hypothesis was that fisher use of multi-season cells would decline if forest projects reduced habitat suitability within them. We therefore placed camera traps within 10 m of the original camera trap position except when forest management or windthrow removed or toppled trees previously used for cameras or baits.

2.6. Basic habitat and biophysical covariates

We developed local, cell-specific, biophysical covariates for use in analytical models of occupancy. We calculated the mean elevation (elev) for each surveyed cell, which was always included in occupancy analyses with its quadratic term (elev^2). This covariate was standardized. Habitat covariates included an index of canopy cover based on the proportion of each cell with CWHR conifer and hardwood tree canopy closure classes M (40–59% canopy closure) or D (60–100% closure) (denMD; http://www.dfg.ca.gov/biogeodata/cwhr/wildlife_habitats.asp). We did not include

Table 1

Classification, descriptions, and source details for forest management and biophysical covariates used in single-season and multi-season models of fisher presence in the Bass Lake Ranger District, Sierra National Forest, and southwestern Yosemite National Park in the period from 2002 to 2013. FACTs is the Forest Service Activity Tracking System, and numeric management codes are provided for each type of activity (FACTs User Guide, 2012).

Covariates	Brief description	Source, detail	Period	Model variables
Logging[a]	Commercial thin, salvage, sanitation, selection cutting	FACTs codes 4220, 4231, 4232, 4151, 4152, 4310	11 years; 2002–2013	log.5[d]
Logging	Hazard tree removal, Private timber harvest	Sierra National Forest, CalFire shapefiles[b]	3–5 years; 2009–2013	log.5
Fuel management[a]	Mastication, mowing, chipping, fuel breaks	FACTs codes 4580, 1154, 1180	11 years; 2002–2013	hazfuel.5[d]
Fuel management	Tree thinning, Precommercial thinning	FACTs codes 4521, 4542, 1160	11 years; 2002–2013	hazfuel.5
Fuel management	Rearrangement, Compacting, Crushing, Yarding, Piling	FACTs codes 1150, 1152, 1153, 1120	11 years; 2002–2013	hazfuel.5
Managed burning[a]	Underburn, Broadcast, Jackpot, etc.	FACTs codes 1111, 1112, 1113, 4481, 4541	11 years; 2002–2013	burn.1.25[e]
Wildfire	Wildfire/wildland fires in the Bass Lake District, or in southern Yosemite National Park	FACTS codes 1115, 1116, 1117, Sierra National Forest and Yosemite National Park shapefiles	101 years for Sierra NF; 1911–2012, 78 years for Yosemite NP 1930–2008	burn.1.25, burn.26.50, burn.1.50[e]
Biophysical	Elevation	Digital elevation model		elev + elev^2
Biophysical	Canopy closure; M (40–59%), D (60–100%)	CWHR[c]		denMD

[a] All management activities associated with logging were considered "extractive", whereas activities linked to fuel management or managed burning were considered "restorative" (Truex and Zielinski, 2013).

[b] State agency CalFire oversees and monitors private timber harvest (THPs) in California; GIS shapefiles are publicly available from ftp://ftp.fire.ca.gov/forest. We accessed GIS shapefiles on THPs for Madera and Mariposa County in October 2014.

[c] Data from a California Wildlife Habitat Relationships System vegetation layer for the study area (http://www.dfg.ca.gov/biogeodata/cwhr/wildlife_habitats.asp).

[d] Log.5 and hazfuel.5 indicates that we used the cumulative area of each management activity that occurred in a 1-km^2 grid in the 5 years preceding the year of the camera trap survey in model analyses.

[e] Burn.1.25, burn.1.50, and burn.26.50 indicate the cumulative area of managed burning + forest fires that occured in a 1-km^2 grid cell in the 25 years, 50 years, or 26–50 years before the camera survey, respectively.

covariates representing average tree size and slope because of their colinearity with forest cover and elevation.

2.7. Forest management covariates

A diversity of forest management activities occurred on the Sierra NF from 2002 (5 years before the start of our study) through the last camera survey year (2012–2013). Most of the management activities used for forest management covariates were developed from the USDA Forest Service FACTs database (Forest Service Activity Tracking System; http://www.fs.usda.gov/main/r5/landmanagement/gis). FACTs is a tracking system including a geospatial database of forest management activities that occur on national forest service lands in California and elsewhere (USDA Forest Service FACTs User Guide, 2012). Polygon layers included in the FACTS database are associated with attributes detailing management activity codes, and dates for when activities were initiated and completed. There are known uncertainties in FACTS with regards spatial precision, area of treatment polygons, and lack of details on whether a treatment activity was completed for an entire polygon (Garner, 2013). We also know that some entries represent perimeters encompassing smaller subunits treated at the same time as well as some areas unaffected by the management activity (Zielinski et al., 2013a). Nevertheless, FACTS data constitute the best available and consistent record of the annual management activities that occurred on national forest lands in our study area. We reviewed the descriptions of each management activity included in the FACTs User Guide (2012) and identified a subset of 24 activities that were considered as potentially influencing local scale habitat use by fishers related to how each altered forest habitat structure, or whether they represented significant ground-disturbing activities (Table 1; Garner, 2013; Zielinski et al., 2013a). For example, we included forms of harvest that would have direct effects on the basis of their disturbance and alteration of forest structure, but excluded activities that did not meet this criterion, or that rarely occurred (Table 1; Zielinski et al., 2013a).

Four other potentially significant activities or events were not systematically tracked by the FACTS system; hazard tree removals (e.g., hazard tree logging), private timber harvests, historical wildfires, and recent wildfires. Hazard tree logging was the removal of medium and large trees (no DBH restriction) within 91 m of forest roads that were considered likely to fall (USDA Forest Service, 2004). Information on hazard tree logging (road segments along which the activity occurred) in the study area was available for 2009, 2010, and 2011 only. Private timber harvest occasionally occurred on large parcels of private land within or adjacent to the Sierra NF in Madera and Mariposa counties. Harvesting of timber on private lands in California requires preparation of Timber Harvest Plans that are reviewed, approved, and made available to the public by state agency Calfire. The Calfire GIS database (ftp://ftp.fire.ca.gov/forest) was our source for geospatial data on logging activities on private lands within the Sierra National Forest. We obtained GIS layers with spatial information and ignition dates of wildfires that occurred (1) in the Sierra NF portion of the study area from 1911 through 2013 and (2) in the Yosemite NP portion of the study for 1930–2008 (sufficient because no camera surveys were completed by our technicians in Yosemite NP after May 2009). Attribute information included with the various geospatial data were used to assign activities and wildfires to individual camera survey years. For example, if a management activity was identified as completed before October 15, 2009, the disturbance was assigned to camera year 2008–09.

We used ArcGIS 10.2 (ESRI, Redlands, CA) to estimate the area of each 1-km^2 surveyed cell with hazard tree logging, private timber harvest, and wildfires, which were merged with the FACTs information for 2002–2013. After merging, we reviewed the entries, and removed duplicate polygons; those with the same FACTS code with identical shapes and areas but with different years of completion. We also removed any duplicate forest fire records that were included in both the FACTs data and in the Sierra NF wildfire

database. We then used the detailed descriptions of each FACTs activity type to create 3 composite variables for use as covariates for occupancy analyses (Table 1). The first 2 variables were extractive fuel reduction (log.5) and restorative fuel reduction (hazfuels.5), and included the cumulative areas of these activities in each grid in the 5 years immediately preceding each camera trap survey (Table 1). For example, the hazfuels.5 covariate for any cells that were surveyed in camera year 2012–13, was calculated as the sum of the areas of all restorative fuel reduction activities that occurred in those cells during fiscal years 2007–08, 2008–09, 2009–10, 2010–11, and 2011–12, from which we calculated the proportion of the 1-km^2 cells disturbed by the treatment. Because of the coordinated series of extractive and restorative fuel treatments associated with SPLATs, multiple different treatments could be applied on the same forest stand within a 5-year period (Zielinski et al., 2013a). It was therefore possible that the cumulative area of a grid that was treated during a 5-year period could exceed 1-km^2.

The third composite forest management variable was managed burning and forest fires within each 1-km^2 cell (Table 1). Our initial review of the FACTS and Sierra NF and Yosemite NP databases, revealed that managed burning was uncommon during 2002–2013. We therefore combined information on managed burning and the longer time-series of forest fires for developing 3 fire-related covariates (Table 1); managed burns + forest fires within 25 years of a survey (burn.1.25), managed burn + forest fires within 50 years of a survey (burn.1.50), and managed burns + forest fires 26–50 years before a survey (burn.26.50) (Table 1).

Evidence suggests that the frequency, size, and relative severity of wildfires are increasing in the Sierra Nevada (Lydersen et al., 2014; van de Water and Safford, 2011) associated with decreased snowpack in low and mid-elevation forests in the region (Belmecheri et al., 2015). We therefore used information on wildfires to evaluate the number and approximate elevation of forest fires that occurred in the study area from 1911 to 2013. The Sierra NF and Yosemite NP wildfire databases were intersected with a polygon similar to Fig. 1 that was extended to encompass all 1-km^2 cells surveyed from 2007 to 2014. We used ArcGIS to determine the area and centroids for each forest fire polygon, and extracted elevation at the centroids from a 10 m DEM. We used a two-tailed t-test to compare the mean elevation for forest fires ($\geqslant 25$ ha) in the study area for 1911–1956, and 1957–2013. A caveat for this comparison is that centroids for wildfires did not capture variation in elevation burned by individual forest fires.

2.8. Occupancy model analyses

Occupancy represents the proportion of an area on which a species occurs (MacKenzie and Nichols, 2004). If repeat surveys are made over a short enough time when the population can be considered closed, modeling can be used to estimate occupancy while accounting for heterogeneity in detection probability among survey sites (MacKenzie et al., 2006). Addressing detection probability is important because the failure to do so can lead to biases in estimates of occupancy and its associations with covariates. We used single-season models to evaluate the importance of environmental and forest management covariates to fisher occupancy (Slauson et al., 2012), and multi-season models to evaluate how these same covariates were linked to colonization of unoccupied cells by fishers and persistence of occupied cells by fishers (Zielinski et al., 2013b).

For single season analyses, we modeled occupancy (ψ) and detection probability (p) as functions of covariates (x) and parameters (β) where p was defined as the probability of observing fisher during a survey period if it was present.

<u>Single-season Occupancy Model</u>

Detection:	$\text{logit}(p) = \beta_{p0} + \beta_{p1}x_1 + \beta_{p2}x_2 + \cdots$
Occupancy:	$\text{logit}(\psi) = \beta_{\psi 0} + \beta_{\psi 1}x_1 + \beta_{\psi 1}x_2 + \cdots$

For multi-season analyses, we defined colonization (γ) as the probability that a cell unoccupied in year t would be occupied in year $t+1$ and persistence as $1 -$ extinction where extinction (ϕ) was the probability that a cell occupied in year t would be unoccupied in year $t+1$. The model also included components for detection probability and for occupancy in the initial year a site was surveyed.

<u>Multi-season Occupancy Model</u>

Detection:	$\text{logit}(p) = \beta_{p0} + \beta_{p1}x_1 + \beta_{p2}x_2 + \cdots$
1st year	$\text{logit}(\psi_{\text{initial}})$
Occupancy:	$= \beta_{\psi\text{initial}0} + \beta_{\psi\text{initial}1}x_1 + \beta_{\psi\text{initial}2}x_2 + \cdots$
Colonization:	$\text{logit}(\gamma) = \beta_{\gamma 0} + \beta_{\gamma 1}x_1 + \beta_{\gamma 1}x_2 + \cdots$
Extinction:	$\text{logit}(\phi) = \beta_{\phi 0} + \beta_{\phi 1}x_1 + \beta_{\phi 2}x_2 + \cdots$

As noted in Section 2.2, we created a detection history of whether a fisher was detected by a camera trap within each cell during each consecutive survey period after set-up or re-baiting for up to five 8–10 day periods during a survey year. This was repeated for up to 6 consecutive years (e.g., 00101 00000 01110 00010 01101 00000) for every cell. If surveys did not occur during any of the 5 periods and 6 seasons at any of the cells these data were treated as missing data. Models were solved by maximum likelihood estimation via R statistical software (Version 3.0.1, www.r-project.org) using the *unmarked* package (Fiske and Chandler, 2011). Single-season models were fit using the *occu* function and multi-season models were fit using the *colext* function. We followed an information-theoretic approach for comparing models containing different combinations of covariates. We evaluated the top models with AIC weights summing to 0.95 (Burnham and Anderson, 2002). We based decisions on which covariates were important predictors of detection probability, occupancy, colonization, or extinction on the relative AIC weights of the top models and the magnitude and variation of parameter estimates from these models. We chose a best single-season model and a best multi-season model reflecting these decisions, and used these models in the estimation of detection probability, occupancy, colonization, and persistence of fisher in relation to predictive covariates.

Covariates for potentially explaining detection probability included a dichotomous, 1st order Markov process reflecting whether a fisher was detected in the previous survey period in a season (auto.y; Hines et al., 2010; Slauson et al., 2012), the number of effective camera days in a survey period divided by 10 (camdays), the proportion of CWHR medium and dense canopy closure classes in each grid (denMD), and a dichotomous variable representing whether the survey was conducted in summer (summer) instead of in fall to spring. We fit all 16 combinations of these detection covariates in occupancy-intercept-only single-season models (e.g., $\text{logit}(\Psi) = \beta_{\Psi 0}$, $\text{logit}(p) = \beta_{p0} + \beta_{p1}x_1 + \beta_{p2}x_2 + \ldots$). Covariates deemed important in this step were included in the detection component of all subsequent models. Next, we evaluated the following occupancy covariates: elev + elev^2, denMD, log.5, hazfuels.5, burn.1.25, and burn.26.50. While always including the final detection covariates, we fit all 64 possible combinations of the occupancy covariates in single-season models. We evaluated these models to assess the importance of occupancy covariates and chose a best model for estimation of detection and occupancy parameters.

Due to the smaller sample size of sites available for fitting multi-season models (n = 361), we focused on evaluating the role of forest management and fire covariates log.5, hazfuels.5, and burn.1.50 in explaining annual transitions in occupancy state (colonization and extinction). We replaced burn.1.25 and burn.26.50 with burn.1.50 because of the smaller sample size of survey sites. For the initial occupancy component of the multi-season model, we restricted explanatory covariates to denMD and elev + elev^2 and included each of those covariates in all multi-season models only if they had been identified as important in the single-season models. Similarly, we included the detection covariates selected as important in single-season models in the detection component of all multi-season models. First, we fit models including all 8 combinations of the forest management and fire covariates on the colonization component, and an intercept-only extinction component (e.g., logit(ϕ) = $\beta_{\phi0}$, logit(γ) = $\beta_{\gamma0}$ + $\beta_{\gamma1}x_1$ + $\beta_{\gamma2}x_2$ + . . .). After selecting final covariates for the colonization component, we fit models including all 8 combinations of the forest management and fire covariates on the extinction component while always including the colonization covariates we selected as important. Lastly, we evaluated these models to assess the importance of extinction covariates and chose a best model for parameter estimation.

3. Results

3.1. Camera trap surveys and fisher detections

In the 6 years of our study we deployed camera traps for 1980 different surveys in a total 905 unique 1-km^2 cells. Eleven of those cells were north of the Merced River in Yosemite NP, and were excluded from analyses because fishers are not extant north of the Merced River (Spencer et al., 2015a). A total 894 1-km^2 cells were surveyed in at least one year, and 361 cells were surveyed in ⩾2 years (Fig. 2a). We detected fishers in 448 of the 894 1-km^2 survey cells, and fishers were most frequently detected in the elevation range 1061–2121 m (Fig. 2b).

3.2. Management disturbances and wildfires

Our analyses of FACTS and other extractive and restorative management activities revealed that the estimated area of forest disturbing activities that occurred from 2002 through 2013 in the study area was highest for restorative fuel reduction, moderate for extractive activities, and lowest for managed burning and natural or human-caused wildfires (Table 2). In the 11-year period 2002–03 to 2012–13, an annual average of 2.4% (SD 0.82) of the study area was treated or disturbed by restorative fuel reduction, compared to 1.1% (SD 0.70) from extractive fuel reduction, 0.3% (SD 0.29) from managed burning, and 0.6% (SD 0.85) from managed burning + forest fires (Table 2). Moreover, the combined area disturbed from all types of disturbances from 2002–03 to 2012–13 averaged 45.2 km^2/year, representing an overall annual disturbance in the 1127 km^2 study area of 3.8%/year. Finally, in the 44 years from 1957 to 2001, we estimated that 130.2 km^2 (11.6%) of the overall study area was burned by wildfires.

Five-hundred and three (56%) of the 894 single-season survey cells were treated or disturbed by forest fuel reduction, managed burning, or forest fires in the 11-years from 2002 to 2013 (Table 3). Two-hundred and fifty two (70%) of the 361 multi-season survey cells were treated or disturbed by one or a combination of these activities in the same 11-year period. The most common disturbance in both groups of survey cells was restorative fuel reduction, whereas a more limited number of single-season or multi-season cells were disturbed by managed burn + forest fires (Table 3). The mean disturbance by forest fuel reduction or managed burning

+ forest fires among 894 1-km^2 cells in the single-season analyses ranged from 0.3%/year (managed burn + forest fires) to 1.4%/year (restorative fuel reduction; Table 3). The mean disturbance by forest fuel reduction or managed burning + forest fires among 361 1-km^2 cells in the multi-season analyses ranged from 0.2%/year for managed burn + forest fires to 2.1%/year for restorative fuel reduction (Table 3).

Geospatial data indicated that approximately 24,916 ha of the overall study area was exposed to burning from 130 forest fires (⩾25 ha) from 1911 to 2013, and 23 of them were large fires (⩾250 ha) (Fig. 3a). Temporally, forest fires were more common before the 1940s, uncommon from the 1940s to the mid-1990s, and became more common in the last 15 years (Fig. 3a). The 50 forest fires in the study area from 1957 to 2013 were mostly below 1966 m elevation (72%), and the distribution of forest fires overlapped all but the high elevation areas used by fishers (Fig. 3b). The mean elevation of 50 forest fires that occurred from 1957 to 2013 (1610 m, SD 288) was nearly 200 m higher than the mean elevation of 80 fires that occurred from 1911 to 1956 (1414 m, SD 247; two-tailed t_{128} = −4.09, P < 0.001), which we interpret as an upward trend related to assigning an elevation of each fire from centroids.

3.3. Single-season detection and occupancy

Our single-season model analysis demonstrated high detectability of fishers using camera traps in the fall to spring. Of the sites we surveyed in fall to spring, 97% of surveys provided a detection probability > 0.2 and 91% of sites had a cumulative, site-level detection probability > 0.8 after 4 survey periods. During summer, 11% of surveys provided a detection probability > 0.2 and a mere 1% of sites had a cumulative, site-level detection probability > 0.8 after 4 survey periods. The observed naïve occupancy was 0.41, compared to a modeled estimate of average occupancy across the study area of 0.54 [95% CI: 0.46–0.63].

We deemed all 4 detection probability covariates evaluated in the single-season analysis to be important predictors, because all were included in a single top model with an AIC weight of 1.0 (Table 4). The probability of detecting fisher (if present) increased as the duration of the survey was extended and in forest cells with greater canopy cover, whereas it was lower for summer season surveys than for fall to spring surveys (Fig. 4). Detection probability was also higher in the survey immediately following a survey in which a fisher was previously detected (Fig. 4, 1st order Markov process), indicating an attraction response to baited camera traps.

We concluded that the quadratic effects of elevation (elev + elev^2) and medium to dense canopy cover (denMD) were important predictors of occupancy in the single season models, because they were contained in all top models and their effect sizes were consistently non-zero and differed little in magnitude among those models (Table 4). We identified a strong curvilinear relationship between occupancy and elevation with fisher occupancy estimated as 0.80 [95% CI: 0.74–0.85] at an optimum elevation of 1680 m, compared to occupancies of <0.1 at elevations below 1070 m or above 2300 m (Fig. 5a). Predicted fisher occupancy was higher in forest cells with more areas of moderate and dense canopy cover. For example, occupancy increased by 16% from 0.74 [95% CI: 0.65–0.81] to 0.86 [95% CI: 0.80–0.90] as the proportion of medium and dense canopy closure in cells doubled from 0.50 to 1.0 (Fig. 5b).

We determined that extractive activities in the last 5 years (log.5), and managed burns + forest fires in the 26–50 years preceding surveys (burn.26.50) were of limited importance for local occupancy, because they were not in any of the highest ranked models (with AIC weights summing to 0.5). Moreover, confidence intervals associated with effect sizes for log.5 and burn.26.50
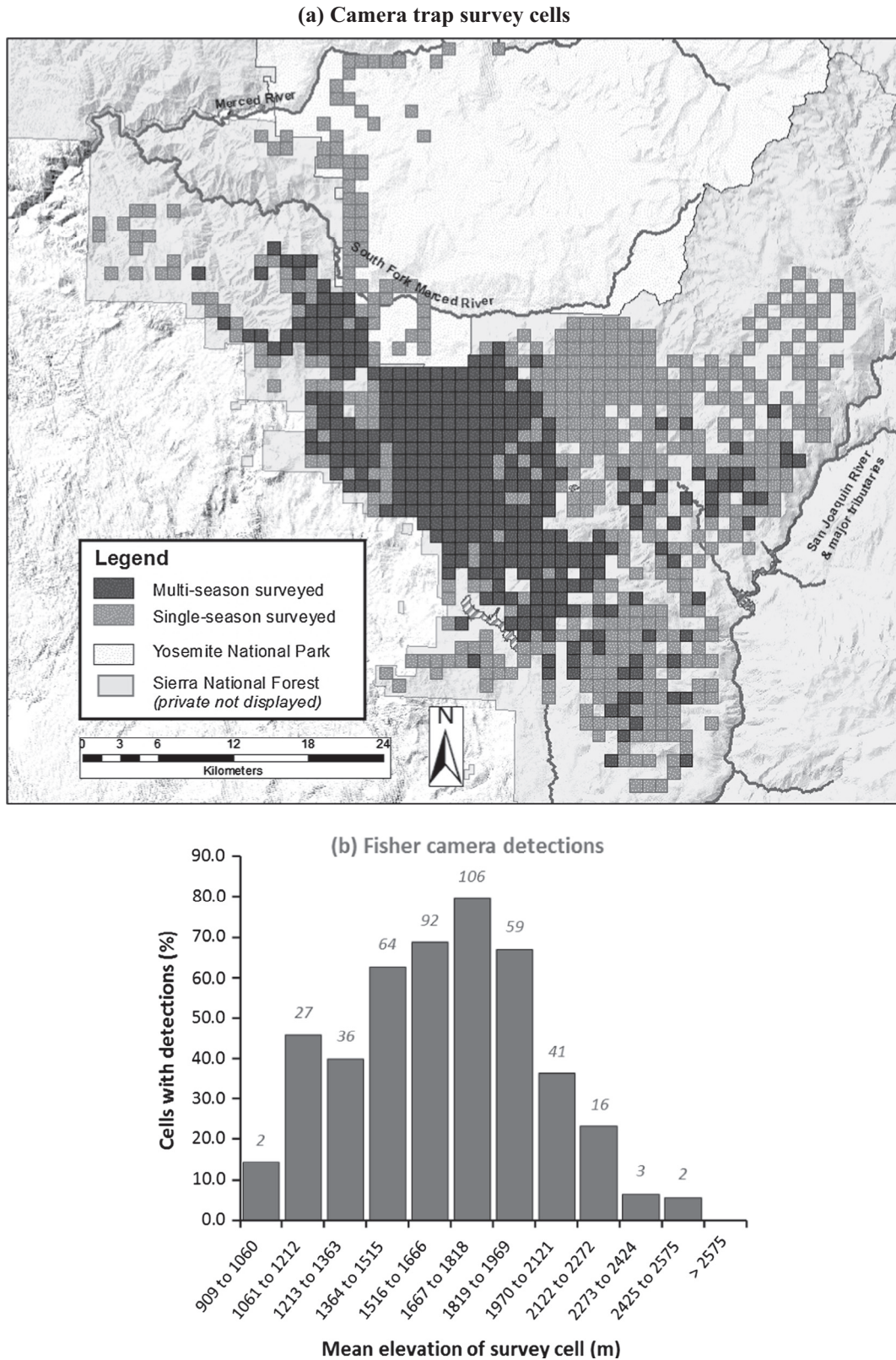
(a) Camera trap survey cells

(b) Fisher camera detections

Fig. 2. Distribution of 1-km² cells surveyed for fisher presence in at least 1 camera survey year (n = 894), and in $\geqslant$2 camera survey years (n = 361) (a). Elevations for fisher detections among all individual grid cells that were surveyed in the study area (b). Camera trap surveys were completed from Oct 2007 to Oct 2014 in the Bass Lake Ranger District, Sierra National Forest, and part of southwestern Yosemite National Park.

broadly overlapped zero (Table 4 and Fig. 6). The results for restorative fuel reduction within the last 5 years (hazfuels.5) and managed burns + forest fires in the last 25 years (burn.1.25) were more equivocal. Both of these covariates were in the highest ranked model, and one or both were in 5 of 6 candidate models with support (ΔAIC $\leqslant$ 2.0; Table 4). However, the confidence

Table 2
Estimates of the areas disturbed by mechanical fuel reduction, managed burns, and wildfires in the Bass Lake Ranger District, Sierra National Forest, and southwestern Yosemite National Park.[a] Data on fuel reduction and managed burns were from 15 October 2002 to 14 October 2013, whereas estimates of the area burned by wildfires were from 1957 through 2013.

5 yr period or survey year	Restorative fuel reduction		Extractive fuel reduction		Managed burns + forest fire		Managed burns		Forest fires	
	Area (km²)	Study area (%)	Area (km²)	Study area (%)	Area	Study area (%)	Area	Study area (%)	Area	Study area (%)
1957–1961									36.4	7.3
1962–1966									5.3	1.1
1967–1971									6.1	1.2
1972–1976									3.4	0.7
1977–1981									4.7	0.9
1982–1986									11.5	2.3
1987–1991									41.9	8.4
1992–1996									1.0	0.2
1997–2001									20.1	4.0
Total									130.2	0.3[b]
2002–03	40.4	2.10	11.6	1.0	3.5	0.3	1.9	0.2	1.6	0.1
2003–04	17.9	1.2	2.9	0.3	3.7	0.3	1.1	0.1	2.6	0.2
2004–05	17.3	1.2	7	0.6	4.3	0.4	4.3	0.4	0.1	0.0
2005–06	29.5	2.3	6.8	0.6	2.4	0.2	2.4	0.2	0.1	0.0
2006–07	35.4	3.1	13.1	1.2	5.3	0.5	4.8	0.4	0.5	0.1
2007–08	22	1.4	2.1	0.2	34.0	3.0	11.2	1.0	22.8	2.0
2008–09	35.2	2.6	11.4	1.0	3.8	0.3	3.8	0.3	0.0	0.0
2009–10	35.1	2.4	27	2.4	1.0	0.1	1.0	0.1	0.0	0.0
2010–11	16.1	1.1	13.8	1.2	6.1	0.5	0.5	0.1	5.6	0.5
2011–12	15	1.1	24.3	2.2	0.1	0.1	0.0	0.00	0.1	0.0
2012–13	27.5	2.1	16.1	1.4	5.7	0.5	0.6	0.1	5.1	0.5
Total	291.4	2.4[c]	136.1	1.1[c]	69.9	0.6[c]	31.6	0.3[c]	38.3	0.4[c]

[a] Areas of disturbance were derived from FACTs data, private timber harvest data, and Sierra National Forest and Yosemite National Park databases (Table 1). The overall study area was 1127.5 km² (Fig. 1), which was used to estimate disturbance percentages.
[b] Mean disturbance (%) by wildfires in the study area in 44 years from 1957 to 2001.
[c] Mean disturbance (%) by mechanical fuel reduction, managed burning, and wildfires in the study area in 11 years from 2002 to 2013.

Table 3
Disturbance activities for 2002 through 2013 (11 years), summarized by the total number of 1-km² grid cells impacted. Data are for cells surveyed for fisher occupancy in the Bass Lake District, Sierra National Forest, and southwestern Yosemite National Park.

Analysis, description	No. cells[a]	Cell area disturbed (km²)	Mean annual disturbance (%)[b]	
			Cells with disturbance	All surveyed cells
Single-season (n = 894 cells)				
No management disturbance[c]	391			
Restorative fuel reduction	387	137.0	3.2	1.4
Extractive fuel reduction	234	71.7	2.8	0.7
Managed burn + forest fires	94	28.1	2.7	0.3
Managed burns	56	15.6	2.5	0.2
Forest fires	50	12.5	2.3	0.1
Forest fires 1957–2001	78	25.4	3.0	0.3
Multi-season (n = 361 cells)				
No management disturbance[c]	109			
Restorative fuel reduction	206	83.1	3.7	2.1
Extractive fuel reduction	172	51.9	2.7	1.3
Managed burn + forest fires	28	9.7	3.2	0.2
Managed burns	13	2.5	1.7	0.1
Forest fires	19	7.8	3.7	0.2
Forest fires 1957–2001	19	8.8	4.2	0.2

[a] Included only those cells with ⩾1% disturbance in any single year from 2002 to 2013.
[b] Estimated as cell area disturbed/no. cells/11 × 100.
[c] Many cells were treated for both types of fuel reduction and managed burning, and may have encompassed burning from forest fires. The number of cells with any type of disturbance can be calculated as the number of single-season (n = 894) or multi-season cells (n = 361) minus the number of cells with no management disturbance.

intervals for the parameter estimates for both covariates overlapped zero (Fig. 6). With these qualifications we decided to select the top model including hazfuels.5 and burn1.25 as our best model and considered the estimated effect sizes of negatively-trending associations in those covariates as germane in our overall assessment of forest management and fire on fisher habitat. Occupancy decreased 17% from 0.81 [95% CI: 0.74–0.86] to 0.67 [95% CI: 0.46–0.83] as hazfuels.5 increased from zero to the entire proportion of the 1-km² grid cell (Fig. 5c). Occupancy decreased 24% from

0.81 [95% CI: 0.74–0.86] to 0.61 [95% CI: 0.34–0.83] as burn.1.25 increased from zero to the entire proportion of the 1-km² grid cell (Fig. 5d). The best model also included all 4 detection probability covariates.

3.4. Multi-season occupancy

Naïve initial occupancy among the multi-season cells we surveyed was 0.63, whereas our modeled estimate for average initial
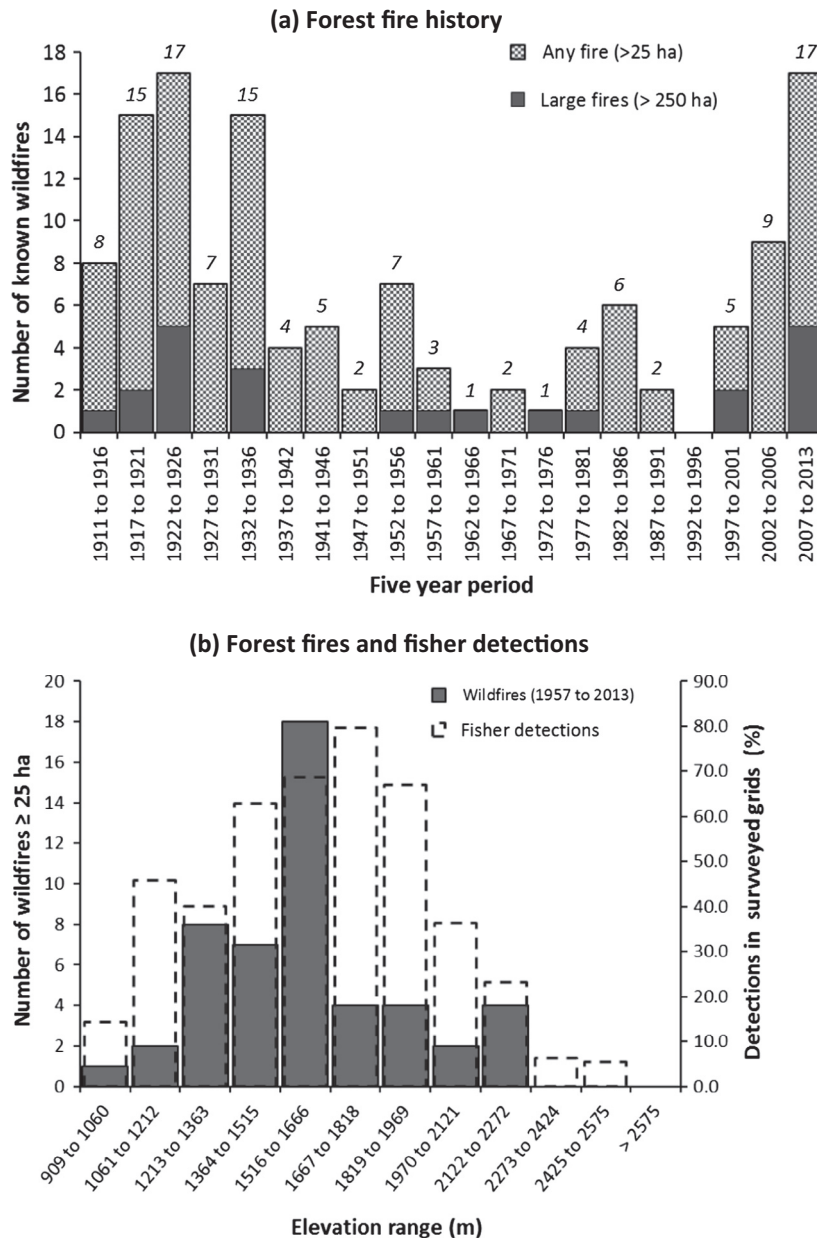
Fig. 3. Number of wildfires in the overall study area from 1911 to 2012 plotted based on fire size (a), and the total number of wildfires that occurred in the study area from 1957 to 2012 by mean elevation of fire centroids, and in relation to detections of fishers (bars with dashed lines) within 151 m elevation bins (b).

occupancy across the study area for multi-season surveys was 0.75 [95% CI: 0.66–0.84]. Mean annual colonization was 0.35 [95% CI: 0.28–0.42], whereas mean annual extinction was 0.14 [95% CI: 0.11–0.18] and the related persistence rate (1-extinction) was 0.86 [95% CI: 0.82–0.89].

The intercept-only null model outperformed all colonization covariate models. For this reason, we concluded that none of the fire and forest management covariates evaluated (log.5, hazfuels.5, and burn.1.50) were important for explaining colonization of unoccupied sites by fisher. On the other hand, extinction was positively associated with recent restorative fuel reduction (hazfuels.5), but not with extractive fuel reduction (log.5), or 50 years of prior burning history (burn.1.50). We based these determinations on following factors. The hazfuels.5 covariate was in the top model and all other models that outperformed the null (intercept-only) model (Table 5). Moreover, parameter estimates for hazfuels.5 in those 4 models were consistently > 0 (Fig. 6), and there was a 24% decline in local fisher persistence (1-extinction; 0.88 [95% CI: 0.84–0.91] to

0.67 [95% CI: 0.49–0.81]) as the proportion of survey grids treated for restorative fuel reduction increased from 0 to 1.0 (Fig. 7). Although the other disturbance covariates log.5 and burn.1.50 were in some of the top models, the confidence intervals of parameter estimates always included zero (Table 5 and Fig. 6). For these reasons, we selected only hazfuels.5 as a covariate on extinction for inclusion in our best model used for parameter estimation, and for assessing how local persistence of habitat use declined with increasing disturbance from restorative fuel reduction. This model also contained an intercept only in the colonization component; denMD and elev + elev^2 in the initial occupancy component; and auto.y, camdays, denMD, and summer in the detection component.

4. Discussion

Our research advanced prior knowledge of how fishers respond to management disturbances by evaluating how variable intensities of disturbance from different management activities

Table 4

Candidate models for single-season fisher occupancy in the Bass Lake Ranger District, Sierra National Forest, and southwestern Yosemite National Park, California from October 2007 to October 2014. Covariate auto.y was a binary coding variable identifying whether fishers had previously been detected during the preceding 8–10 day survey period between re-baiting visits. Camdays was the number of functional or effective camera days divided by 10, and denMD was the proportion of the 1-km^2 survey cell including California Wildlife Habitat Relationship tree canopy closure class M (40–59%) and D (60–100%) habitat.

Model, covariate	# Parameters	AIC	ΔAIC	AICwt	Cumulative AICwt
Detection[a]					
auto.y + camdays + denMD + summer[b]	6	2388.52	0.00	1.00	1.00
auto.y + denMD + summer	5	2400.80	12.28	0.00	1.00
auto.y + camdays + summer	5	2433.91	45.39	0.00	1.00
auto.y + summer	4	2446.55	58.02	0.00	1.00
camdays + denMD + summer	5	2453.95	65.43	0.00	1.00
denMD + summer	4	2470.94	82.42	0.00	1.00
auto.y + camdays + denMD	5	2489.67	101.15	0.00	1.00
camdays + summer	4	2491.40	102.87	0.00	1.00
summer	4	2506.52	118.00	0.00	1.00
auto.y + camdays	4	2521.58	133.06	0.00	1.00
auto.y + denMD	4	2540.51	151.98	0.00	1.00
camdays + denMD	4	2557.60	169.08	0.00	1.00
auto.y	3	2565.54	177.02	0.00	1.00
camdays	3	2575.55	187.03	0.00	1.00
denMD	3	2605.78	217.26	0.00	1.00
Intercept Only	2	2617.27	228.75	0.00	1.00
Occupancy[c]					
elev + elev^2 + denMD + hazfuels.5 + burn.1.25[a]	11	2226.68	0.00	0.15	0.15
elev + elev^2 + denMD + burn.1.25	10	2227.10	0.41	0.12	0.28
elev + elev^2 + denMD + hazfuels.5	10	2227.28	0.60	0.11	0.39
elev + elev^2 + denMD	9	2227.37	0.69	0.11	0.50
elev + elev^2 + denMD + log.5 + hazfuels.5 + burn.1.25	12	2228.49	1.81	0.06	0.56
elev + elev^2 + denMD + hazfuels.5 + burn.1.25 + burn.26.50	12	2228.56	1.88	0.06	0.62
elev + elev^2 + denMD + burn.1.25 + burn.26.50	11	2229.00	2.31	0.05	0.67
elev + elev^2 + denMD + log.5 + hazfuels.5	11	2229.02	2.34	0.05	0.71
elev + elev^2 + denMD + hazfuels.5 + burn.26.50	11	2229.04	2.36	0.05	0.76
elev + elev^2 + denMD + log.5 + burn.1.25	11	2229.09	2.41	0.05	0.81
elev + elev^2 + denMD + burn.26.50	10	2229.17	2.49	0.04	0.85
elev + elev^2 + denMD + log.5	10	2229.37	2.69	0.04	0.89
elev + elev^2 + denMD + log.5 + hazfuels.5 + burn.1.25 + burn.26.5013	13	2230.37	3.69	0.02	0.91
elev + elev^2 + denMD + log.5 + hazfuels.5 + burn.26.50	12	2230.79	4.11	0.02	0.93
elev + elev^2 + denMD + log.5 + burn.1.25 + burn.26.50	12	2230.99	4.31	0.02	0.95

[a] All 16 candidate models are shown here.
[b] Included in best model.
[c] All detection covariates were included in each and every model. Only the top 15 of the total 64 candidate occupancy component models are shown.

influenced fisher habitat use at a scale appropriate to the size of those disturbances. Overall, fishers preferred forest habitats with relatively high canopy cover, which was in close accordance with multiple prior studies. Canopy cover/canopy density is the single habitat feature that has been universally associated with presence of fishers and indicative of high quality fisher habitat in California (Aubry et al., 2013; Davis et al., 2007; Spencer et al., 2011). We note, however, that the covariate we used in our analyses was an *index* of canopy cover where 0.80 would represent a 1-km^2 cell with 80% of forest stands in either CWHR canopy closure class M (40–59% canopy closure) or D (60–100% closure), or a mix of closure class M and D totaling 80%. We provide this detail to avoid confusion, and to emphasize that a denMD value of 0.80 does not represent 80% canopy cover as measured with a vertical densitometer or some other direct method. Also, assuming our data were representative of areas used by fishers when they were active, the strong positive association between fisher occupancy and the denMD covariate indicated that relatively high canopy closure was important for foraging fishers, which was already known for both resting and denning fishers (Purcell et al., 2009; Zhao et al., 2012).

4.1. Restorative and extractive fuel reduction management and fishers

Concern that initiation of management to reduce fuel levels in Sierra Nevada mixed-conifer forests to correct decades of fire suppression might have negative effects on fishers was only partly supported by results from our study. We found that local scale persistence was negatively associated with restorative fuel reduction, whereas there were indications for a trend for lower single-season occupancy in forest habitats with higher restorative fuel reduction (Tables 4 and 5). We found no evidence that extractive management activities contributed to reduced occupancy or local persistence, which was likely due to the combination of at least 4 related factors. First, the overall extent of extractive logging in our study in the 11 years from 2002 to 2013 appears much lower than historically, and was likely further diminished by poor market conditions for wood products when a severe recession began in 2008. Extractive resource activities that occurred in our study area from 2002 through 2013 (Table 2) was much more limited than the expansive areas that were logged in the Sierra Nevada region during most of the 20th century based on market volume of timber between 1860 and 2000 (McKelvey and Johnston, 1992; page 240). Most biologists concur that extensive past logging in the Sierra Nevada negatively impacted large areas of suitable fisher habitat, and contributed to the decline of the species in California's Sierra Nevada (Lofroth et al., 2010; Raley et al., 2012; Zielinski, 2013). Second, the economy-related delay in the implementation of several planned forest projects in the Sierra NF may have limited our ability to detect an adverse response by fishers to extractive resource management. Third, estimates of annual disturbance from extractive fuel reduction among single-season and multi-season survey cells were equivalent to levels that were tolerated by fishers elsewhere in the Sierra NF. Our disturbance estimates from the FACTS database averaged about 2.8% for extractive activities
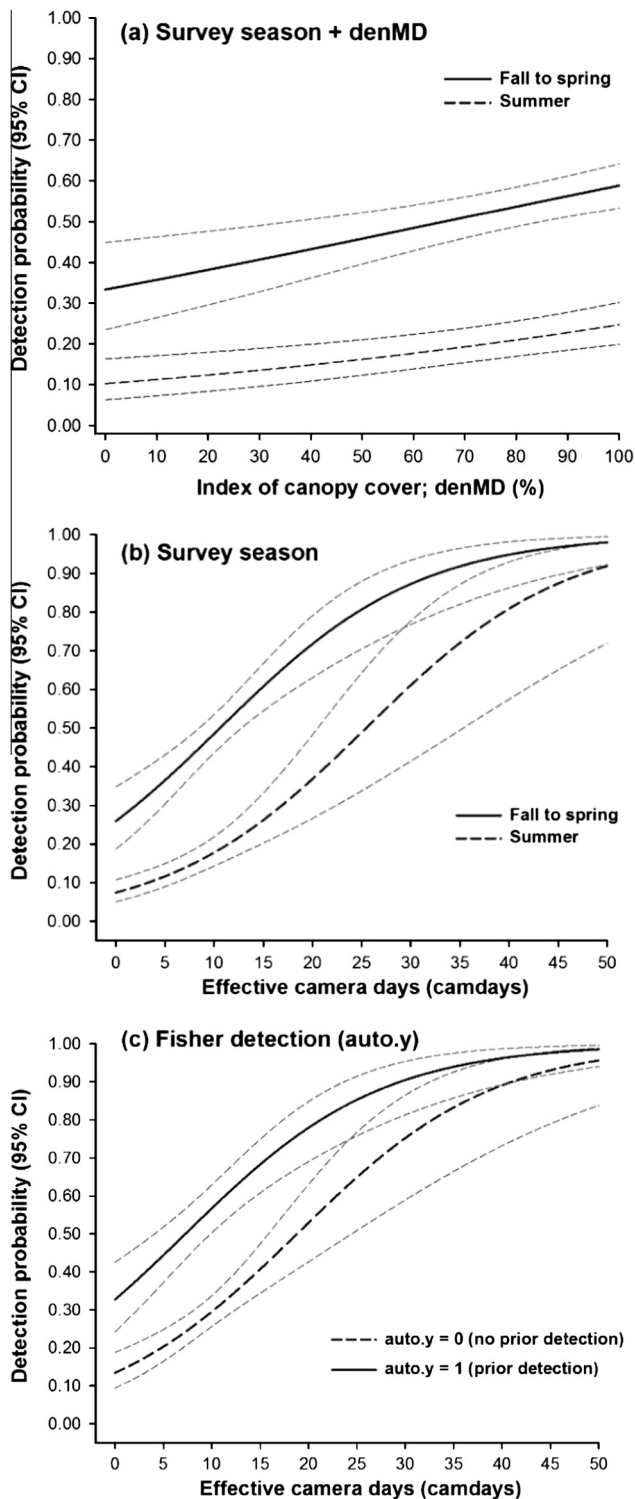
Fig. 4. Probability of detection for fishers (if present) by season in areas with differences in an index of canopy cover (den MD; a), by season and functional camera trap survey days (b), and if fishers had either not been detected (auto.y = 0) or had been detected (auto.y = 1) in the previous survey period between rebaiting visits, by functional camera survey days (c). Covariates in the model included auto. y, camdays, denMD, and summer, which were set to their mean values in the data set when not allowed to vary as shown above.

among surveyed cells (Table 3). Zielinski et al. (2013a) investigated tolerance of fishers to forest management in the High Sierra Ranger District, Sierra NF, and reported that 14 km² cells of forest habitat with high use by fishers typically had 2.6% of the areas disturbed

by management annually, whereas 14 km² cells of forest with low use by fishers averaged 3.5% disturbance/year. Active fishers exploit a wider diversity of forest habitats than resting and denning fishers (Aubry et al., 2013; Raley et al., 2012; Zhao et al., 2012), and the low level of extractive fuel reduction identified in the study area likely did not significantly impinge on their use of these habitats. Truex and Zielinski (2013) developed models of fisher habitat suitability from locations at resting sites, and fisher detections at track or camera traps (considered foraging locations), and evaluated how changes in forests from restorative and extractive fuel reduction altered habitat suitability for fishers. Mechanical fuel reduction induced changes in multiple metrics of forest structure important to resting fishers and reduced resting habitat suitability, but did not significantly reduce foraging habitat suitability. We therefore consider that changes to forest structure introduced by the relatively limited area of extractive fuel reduction in our surveyed cells likely did not significantly impinge on local occupancy or persistence of habitat use by active fishers.

We did, however, find reduced use of forest cells treated with restorative fuel reduction. Our multi-season analyses supported the hypothesis that fishers would reduce use of local patches of forest exposed to proportionally higher levels of restorative fuel reduction (Fig. 7). Annual disturbance from restorative fuel reduction in surveyed cells with those activities was 3.2% (single-season cells) and 3.7% (multi-season cells; Table 3). Those levels of disturbance were comparable to the 3.5% management disturbance/year that Zielinski et al. (2013a) considered sufficient to reduce use by fishers elsewhere in the Sierra NF. Importantly, however, predictions from our single- and multi-season model were that small patches of forest with 100% cumulative 5-year restorative fuel reduction would continue to maintain local fisher occupancy and persistence rates ≈ 0.67 (Figs. 5 and 7). Thus, even at what would be considered a high level of cumulative disturbance, active fishers were not predicted to completely cease using those areas. An example was that in spring 2013, an adult female fisher denned within an area that had been treated for both extractive (2010) and restorative fuel reduction (2011). Moreover, while persistence is not directly comparable to occupancy, a persistent rate of 0.67 would be considered a positive observation with regards maintenance of fisher populations in the southern Sierra Nevada region (Zielinski et al., 2013b).

We did not directly assess how extractive or restorative fuel reduction altered forest habitats in or around our camera survey stations. Instead, we assumed that reduced use of 1-km² cells of forest by fishers with larger areas of restorative fuel reduction treatments was in response to some of the changes in forest habitats previously described in the literature: reduced density of understory trees and shrubs, reduced litter depth, and less surface coarse woody debris (Amacher et al., 2008; Stephens et al., 2009). We note, however, that all studies we reviewed for treatment-induced changes in forest structure included both extractive and restorative fuel reduction activities in the treatments. For example, restorative fuel reduction activities on their own likely would not reduce overstory canopy cover, but would alter mid-story canopy and tree density (Garner, 2013). Below, we consider 3 mechanisms that may explain reduced local habitat use by fishers after disturbance from restorative fuel reduction.

Rotary mastication in our study area was seldom followed by managed burning, and we consider it possible that the presence of residual shards of wood and frayed stems of trees and shrubs discouraged foraging by fishers because it would be difficult for them to rapidly maneuver through such an area when pursuing prey. Amacher et al. (2008) previously reported a negative effect of mastication without follow-on burning on abundance of deer mice, an effect that was not apparent when masticated areas had been burned. Managed burns, or piling and burning, as part of
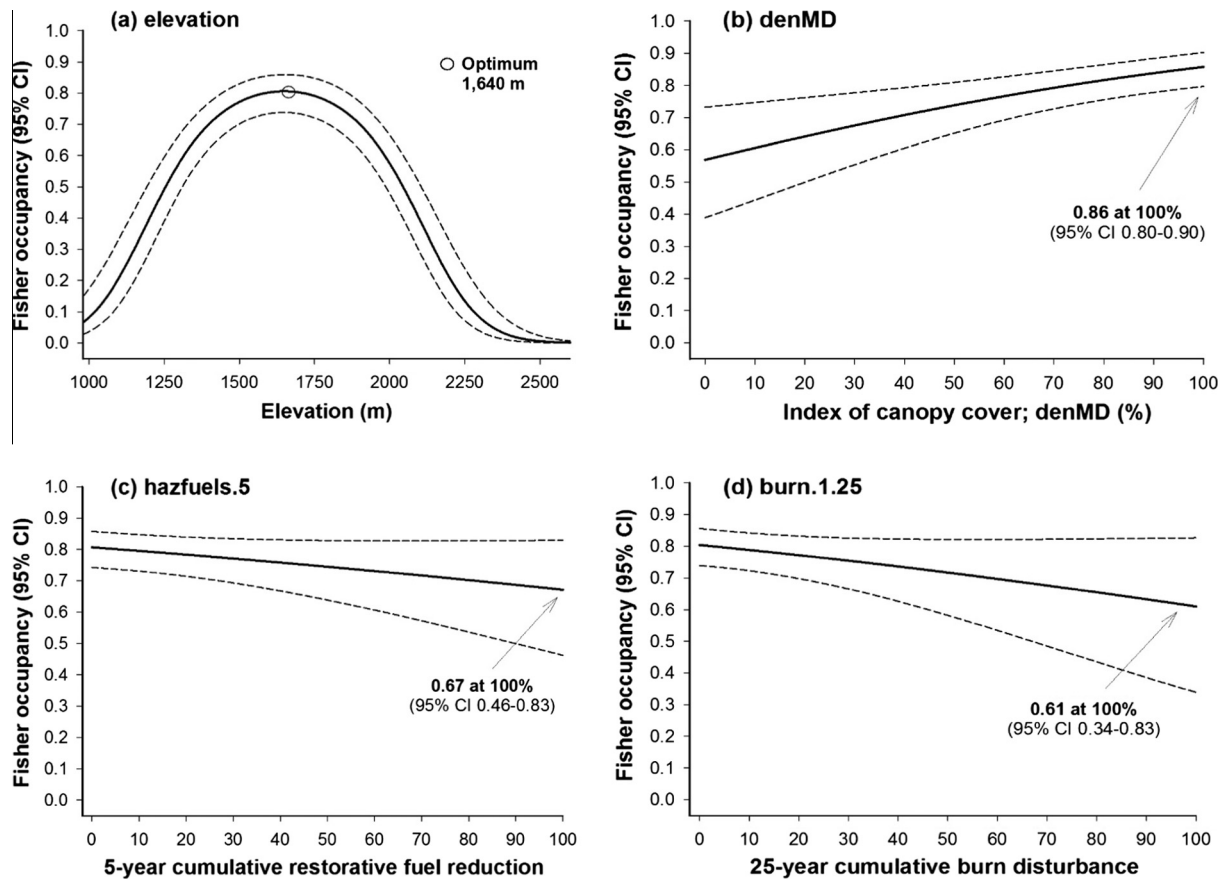
Fig. 5. Estimated relationships for single-season fisher occupancy and elevation (a), an index of canopy cover (denMD; b), cumulative restorative fuel reduction in 5-years before surveys (hazfuels.5; c), and managed burning + wildfires within 25 years of surveys (burn.1.25; d). Covariates in the model included elev, denMD, hazfuels.5, and burn.1.25 which were set to their mean values in the data set when not allowed to vary as shown above.

mechanical fuel reduction typically reduce or eliminate woody debris (Finney, 2001), and we hypothesize that when logistics of staffing or air quality considerations prevent follow-on burning, presence of remnant wood shards and frayed stems discourages use by fishers until natural decomposition returns surface conditions to a more natural state.

Ladder fuels, surface fuels, and thick layers of duff targeted for reduction under SPLAT-based management provide important habitat for squirrels and rodents preyed on by fishers (Zielinski et al., 1999) and other forest carnivores (Kelt et al., 2013). Fisher populations respond to changes in abundance of prey (Bowman et al., 2006; Jensen et al., 2012), and fishers in our study area co-occurred with 6 species of their rodent prey; Douglas squirrel (*Tamiasciurus douglasii*), gray squirrel (*Sciurus griseus*), northern flying squirrel, long-eared chipmunk (*Neotamias quadrimaculatus*), California ground squirrel (*Spermophilus beecheyi*), and dusky-footed woodrat (*Neotoma fuscipes*; see Fig. 2 in Sweitzer and Furnas, submitted for publication). Extractive and restorative management activities may have a negative effect on abundance of northern flying squirrels (Manning et al., 2012; Meyer et al., 2007), but the impacts of those disturbances on the 5 other rodents and deer mice (*Peromyscus maniculatus*) are either unknown, or equivocal (Amacher et al., 2008; Bull and Blumton, 1999; Coppeto et al., 2006; Herbers and Klenner, 2007; Innes et al., 2007; Kelt et al., 2013; Wilson et al., 2008).

We consider it more likely that the 24% decline in local persistence by fishers when cumulative restorative fuel reduction approached 1.0 was associated with fishers shifting to forage in adjacent forest habitat with less disturbance. Our results showing that the colonization rate always exceeded the extinction rate,

suggests that fishers might resume use of treated areas after relatively few years of ecological recovery. The highest predicted extinction was 33% at 100% restorative fuel reduction whereas the colonization rate was 35%, indicating that colonization was strong enough to compensate for local extinction at current levels of restorative fuel reduction in our study area. Analyses by Thompson et al. (2011) applied to a fisher occupied area elsewhere in the Sierra NF suggested that commercial thinning in Sierran mixed-conifer forest did not significantly reduce habitat suitability or "displace" habitat components from reference conditions in home ranges of resident female fishers. We therefore consider it likely that fishers in our study area will resume using forest cells treated for restorative fuel reduction within a few years of extensive disturbance. Additional support for this hypothesis derives from evidence that fishers adjust their space use to avoid disturbed areas within home ranges. Garner (2013) reported that resident fishers included areas treated for fuel reduction in their overall and core home ranges in proportion to availability on the landscape. But at the finer scale of individual locations, resident fishers avoided using areas within ≈200 m of fuel treatments (Garner, 2013). We interpret this result as consistent with ours wherein fishers continued using 1-km² cells of forest with extensive cumulative disturbance by fuel treatments, but at a reduced level compared to areas with less disturbance (Figs. 5 and 7). Finally, our assessment of how fishers responded to forest management was at the scale of 1-km² patches of forest, which was small in relation to resident adult female (≈23 km²) and resident adult male home ranges in our study area (86 km²; Sweitzer et al., 2015b). If a 1-km² patch of habitat within the home range of a resident female fisher was 100% treated for fuel reduction of any type, ≈95% of that
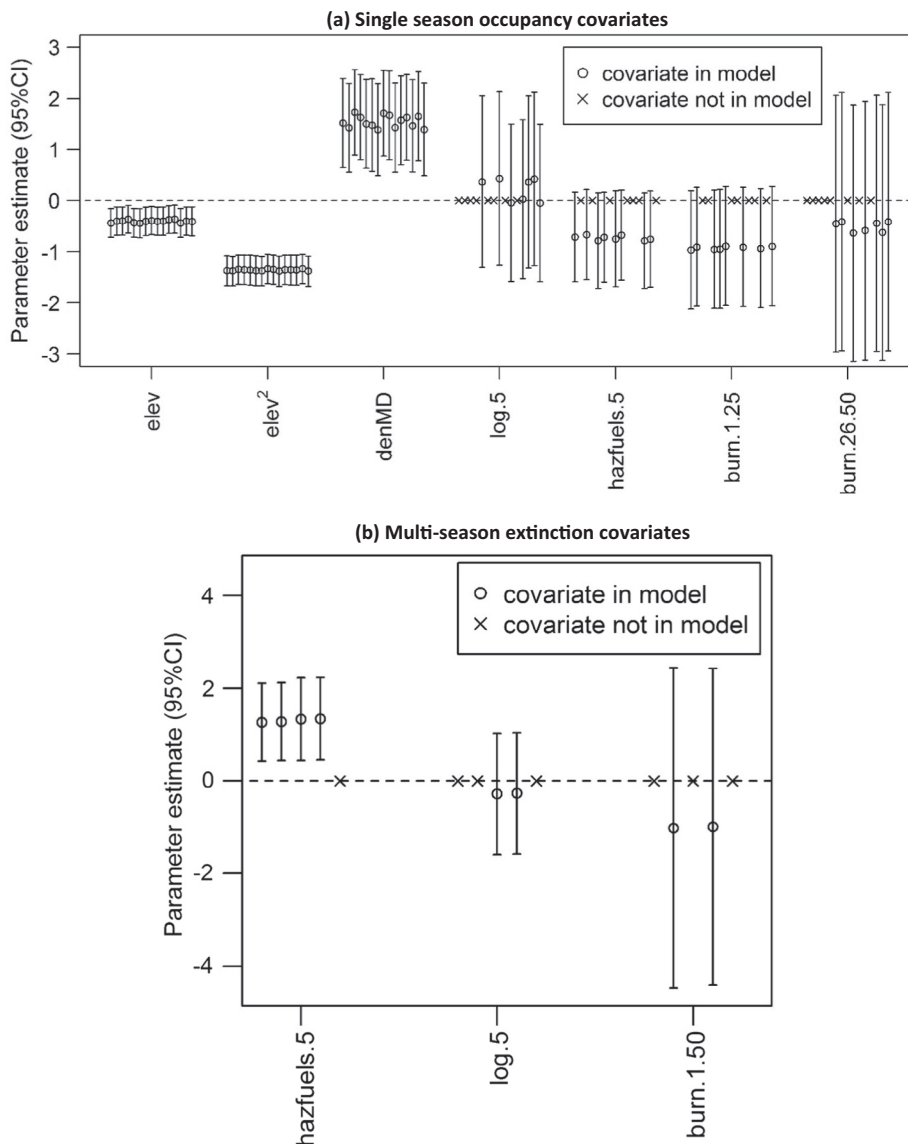
Fig. 6. Parameter estimates of single-season local occupancy covariates in the top 16 single-season models whose weights summed to 0.95 (a), and parameter estimates of multi-season local extinction covariates in the top 5 multi-season models whose weights summed to 0.95 (b). Models are shown from left to right in descending order of model AIC weight. Based on these results and those reported in Tables 4 and 5, we concluded that elev + elev² (elevation), denMD (an index of canopy cover), hazfuels.5 (restorative fuel reduction within 5 years), and burn.1.25 (managed burning + wildfires within 25 years) were important predictors of local occupancy, and that hazfuels.5 was an important predictor of local extinction.

animal's home range could remain undisturbed and suitable for foraging and other activities, contingent on SPLATs continuing to be dispersed on the forest landscape rather than locally concentrated (Graham et al., 1999; Moghaddas et al., 2010).

4.2. Managed burns, wildfires, fisher habitat use

Use of forest fuel reduction treatments to correct for past fire suppression has been partly based on the premise that short term impacts to uncommon or rare vertebrates like fishers or spotted owls will be less consequential than longer term damage to their habitats after large, catastrophic wildfires (Scheller et al., 2011; Tempel et al., 2014; Thompson et al., 2011). Results from our single-season analyses provided limited support for the premise that managed burns + forest fires within the last 25 years are detrimental to fishers, at least in terms of local scale habitat use for active fishers.

When local cells of forest were 100% disturbed by burning within 25 years, those areas were predicted to maintain ∼61% occupancy (Fig. 5d), and our multi-season analysis did not identify an important negative relationship between managed burns + forest fires and local colonization or local persistence of fishers (Table 5 and Fig. 6). Caveats include that the multi-season analysis had low ability to detect effects because only 28 of the 361 surveyed cells encompassed any burning disturbance (Table 3), and that our burn covariates did not consider localized variation in fire intensity, which may influence use of burned forest by fishers (Hanson, 2013). Nevertheless, in 2 successive years in our study (2012, 2013), an adult female fisher successfully denned within a patch of forest burned by a low-severity fire in summer 2008.

Others have found similar recovery of fisher habitat following fire. Hanson (2013) evaluated habitat use by fishers in the Kern Plateau of the southern Sierra Nevada where 2 large, mixed-severity fires occurred 12 years before scat transects provided an index of use. Hanson (2013) reported that 37% (n = 29) of 78 total

Table 5

Candidate model for multi-season occupancy evaluations of local extinction and colonization for camera trap surveys for fishers in the Bass Lake Ranger District, Sierra National Forest, California from October 2007 to October 2014.

Model, covariate[a]	# Parameters	AIC	ΔAIC	AICwt	Cumulative AICwt
Colonization					
intercept only[b]	11	5067.79	0.00	0.33	0.33
hazfuels.5	12	5068.99	1.21	0.18	0.50
log.5	12	5069.43	1.64	0.14	0.65
burn.1.50	12	5069.67	1.88	0.13	0.77
hazfuels.5 + log.5	13	5070.82	3.04	0.07	0.85
hazfuels.5 + burn.1.50	13	5070.88	3.09	0.07	0.92
burn.1.50 + log.5	13	5071.29	3.51	0.06	0.97
hazfuels.5 + burn.1.50 + log.5	14	5072.71	4.92	0.03	1.00
Extinction					
hazfuels.5[b]	12	5062.23	0.00	0.46	0.46
hazfuels.5 + burn.1.50	13	5063.75	1.52	0.21	0.67
hazfuels.5 + log5	13	5064.04	1.81	0.18	0.86
hazfuels.5 + burn.1.50 + log.5	14	5065.59	3.36	0.09	0.94
Intercept Only	11	5067.79	5.56	0.03	0.97
burn.1.50	12	5069.50	7.27	0.01	0.98
log.5	12	5069.53	7.30	0.01	0.99
burn.1.50 + log.5	13	5071.22	8.99	0.01	1.00

[a] All colonization and extinction models included auto.y, camdays, denMD, and summer in the detection component, and elev + elev^2, and denMD in the initial occupancy component.

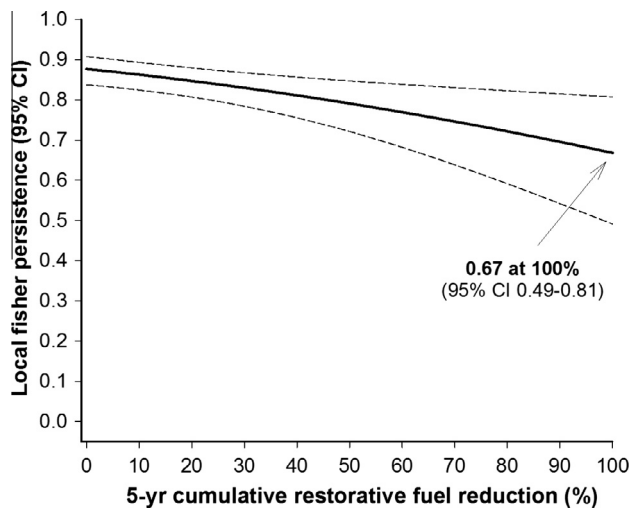[b] Included in the best model.

Fig. 7. Multi-season occupancy model relationship between local fisher persistence (1 − extinction) and the cumulative disturbance from restorative fuel reduction in the 5-years immediately preceding camera surveys in 1-km^2 cells of forest habitat (hazfuels.5). The model was fit assuming average values of elev, denMD, log.5 and burn.1.50 from the data set.

fisher scats were deposited in wildfire-burned forests, indicating that fishers did not completely avoid forests disturbed by fire. Also consistent with our results, Thompson et al. (2011) combined forest growth models and landscape trajectory analysis and predicted that fisher habitat damaged by wildfires would recover within 30 years. Finally, 5–10 years of succession in forests disturbed by wildfire produces habitat conditions suitable for rodents and reptiles consumed by fishers (Swanson et al., 2010), suggesting that

burned forests will support prey for fishers long before they recover to pre-wildfire conditions.

Indications that forest fires in our study area have become more common since the early 1990s supported assertions that the frequency of wildfires in the Sierra Nevada is increasing (Safford and Van de Water, 2013). During our 8-year study 3 large wildfires occurred immediately to the south and north of the study site that reduced availability of suitable fisher habitat in the region (Spencer et al., 2015b). One of these, the 2013 Rim Fire (104,000 ha in size), produced more moderate and high-severity burn effects than many previous fires in the region (Kane et al., 2015). Our data suggested that forest fires in our study area tended to occur at elevations slightly lower than those most commonly occupied by fishers (Figs. 3b and 5a), but a potentially problematic observation was that wildfires that occurred there after the mid-1950s may have burned at a higher mean elevation than those that occurred earlier in the century. These observations are meaningful because climate change, reduced snowpack, and recurring drought will exacerbate the trend of more large, high-severity wildfires in the Sierra Nevada (Flannigan et al., 2000; Kane et al., 2015; Lydersen et al., 2014; Safford and Van de Water, 2013), and it is likely that wildfires will increasingly burn within occupied fisher habitats in the southern Sierra Nevada. Taken as a whole, our research did not identify a consistent negative effect of burning on fisher habitat use, but additional research is needed before concluding that forest fires are not damaging to foraging and denning habitats used by the fishers in the southern Sierra Nevada.

4.3. Probability of detection and design considerations for camera surveys

Our study focused on the potential effects of forest management disturbance and wildfires on fisher occupancy, but several findings are relevant for survey methods for fishers in forest ecosystems. The importance of prior detection of a fisher on detection probability meant that fishers were likely to return after initial discovery of a bait reward, thereby introducing heterogeneity in resight rates that must be accounted for when camera trap detections are used to estimate fisher abundance (Sweitzer et al., 2015a). The importance of effective camera days and season of survey indicated that longer duration surveys were important for reliably identifying fisher use of local patches of habitat, particularly in summer (Fig. 4). For surveys without a fisher detection, the estimated minimum number of camera days needed to achieve 0.90 probability of detection for a survey in fall to spring was 35 days, compared to 50 days for a survey in the summer. Others have noted that detection probability for fishers appears lower in summer (Popescu et al., 2014; Slauson et al., 2012). Our results provide quantitative insight on that observation, and camera surveys should be modified accordingly to assure reliable insight on fisher habitat use from camera traps.

4.4. Known limitations

Camera trap surveys in our study were extensive in scope for cells that were sampled in at least one year, but more limited for those with a recent history of extractive or restorative management that were resurveyed in multiple years. Also, although our analyses identified previously undocumented change in local habitat use by fishers in relation to SPLAT-based forest management, the results do not inform on how changes in occupancy relate to survival or reproduction of fishers. We also note that the relatively minor reductions in local patch use associated with restorative fuel reduction and recent burning may not indicate biologically significant impacts to fishers, particularly if the duration of locally diminished levels of habitat use is years and not decades. Finally,

our results were correlative in nature, and may have been more informative if the analyses were from a before–after control–impact design as originally intended (Popescu et al., 2012).

5. Conclusions

This study identified modest effects of mechanical fuel reduction on occupancy and local scale persistence of fishers, suggesting that management activities intended to reduce risk of frequent, high-severity wildland fires are having limited short term effects on this rare vertebrate. Considering the range of information we report on fisher occupancy, detections of fishers over a 1600 m range in elevation, observations of several female fishers denning in areas with mechanical fuel reduction or prior burning, and the history of wildfires in the study area in relation to the elevation range of fishers, SPLAT-based fuel reduction could be applied at higher rates in relatively lower elevation forest (900–1450 m elevation). More SPLATS below 1450 m could help reduce spread of wildfires into higher elevation forests (Collins et al., 2011) where fishers are more abundant (Figs. 3b and 5a), and where many denning structures are located (Zhao et al., 2012). Coincidentally, the elevation range 900–1450 m in our study area encompasses most of the wildland–urban interface where continuing fire suppression will be necessary to reduce loss of private homes and property (Hanson and Odion, 2014; USDA Forest Service, 2004). Zielinski et al. (2013a) suggested that mechanical fuel reduction can be conducive to continued fisher presence in Sierra Nevada forests if important structural components for fishers (large DBH trees and snags; black oak as well as conifers) are identified and retained. Building on this suggestion, we recommend that management prescriptions for minimizing habitat disturbance around known fisher denning structures (USDA Forest Service, 2004) should be extended to areas identified as suitable for reproduction by models of fisher denning habitat (Spencer et al., 2015b). Doing so will help assure that the relatively minor changes in fisher occupancy from mechanical fuel reduction will not impinge on reproducing fishers already challenged by exposure to other environmental risks that reduce survival and restrict population growth (Sweitzer et al., in press).

Acknowledgements

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