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Short-term effects of fuel treatments on fisher habitat in the Sierra Nevada, California

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ABSTRACT

The characteristics of western forests have changed as a result of fire suppression and fuel reduction treatments have become a public land management priority. The effects of these treatments on wildlife habitat, however, have received limited attention. The fisher (*Martes pennanti*) is a species of concern in California and is vulnerable to fuels treatments due to its association with dense forests and use of large and old trees as resting sites. We evaluated the effect of fuels treatments by estimating predicted resting and foraging habitat at two sites in the Sierra Nevada that are part of the national Fire and Fire Surrogate Study. One site included three treatments (mechanical harvest, prescribed fire, and mechanical harvest plus prescribed fire) and the other included early and late-season prescribed fire; both sites included control treatments. We sampled vegetation before and after treatment application to estimate variables that were included in resource selection probability functions. Predicted resting habitat was significantly lower for mechanical plus fire treatments, but the control did not differ from the fire only or the mechanical only treatment. Late, but not early, season burns had significant impact on predicted resting habitat. Reductions in canopy cover affected predicted resting habitat directly. Fisher foraging habitat, unlike resting habitat was unaffected by treatments at either site. Within a stand, a number of management actions can mitigate the potentially negative short-term effects of fuels treatments on fisher habitat. Evaluating the effects of fuels management at the resting site, home range and landscape scales will be necessary to administer a treatment program that can address fuel accumulation while also restoring and maintaining fisher habitat.

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1. Introduction

The policy of suppressing fires in western North America has led to an unnatural accumulation of woody debris, an increase in the density of trees, and a shift in species composition (Skinner and Chang, 1996; Brown et al., 2003; Scholl and Taylor, 2010; Collins et al., 2011). These changes have increased the risk of uncharacteristically severe fires, which threatens the human communities in or near these forests and the wildlife that depend on them. Land managers are interested in reducing forest fuels, propelled by federal legislation designed to accelerate treatment activities, but the zeal to reduce fuels has not always been accompanied by consideration of the effects of fuels treatments on the habitat of species associated with dense forest conditions. When the effects of treatments on wildlife have been evaluated, they have typically involved either small mammals or birds (Stephens et al., 2012), rarely on other taxa such as carnivorous mammals.

The fisher (*Martes pennanti*) is a carnivorous mustelid associated with dense stands of mature conifer and mixed conifer-hardwood forests in the Sierra Nevada (Zielinski et al., 2004; Purcell et al., 2009; Spencer et al., 2011). Fishers select daily resting locations that are often in cavities of large trees and, of particular relevance, fishers select trees for resting that are most often in dense stands with abundant small-to-medium sized trees (Zielinski et al., 2004). These trees, often referred as ‘ladder fuels’ because they can provide vertical continuity of fuels from the forest floor to the overstory, are often the target for fuels reduction treatments (Menning and Stephens, 2007). Further exacerbating the risk to fisher habitat, from both uncharacteristically high severity fire and from fuels treatments, is the fact that fishers occur primarily in the mid-elevation forests in the Sierra (Zielinski et al., 2005) where the risk of fire is exceptionally high (McKelvey and Busse, 1996). The consequences of loss and alteration of fisher habitat is magnified further by the fact that fishers occupy a limited portion of their historical range in the Sierra (Zielinski et al.,

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1995, 2005), one of the reasons why fishers in the Pacific States have been found to be ''warranted but precluded'' for listing under the Endangered Species Act (U.S. Fish and Wildlife Service, 2004).

A conflict arises because it is unknown whether the potential risk of fuels treatments to fisher habitat (especially cumulative spatial and temporal effects) is offset by a commensurate reduction in risk of wildfire to habitat. Scheller et al. (2011) simulated the trade-off between treating stands to reduce their risk of fire and the direct effects of the treatments on fisher habitat and found that the indirect effects of treatments led to an increase, over time, in fisher habitat compared to the untreated condition. However, this work was based on simulations focused on the landscape scale; we still have much to learn about the effects of fuels treatments at the various scales of habitat selection important to fishers.

The fuels management strategies for public forests in the Sierra Nevada are outlined in bioregional forest management plans (Sierra Nevada Forest Plan Amendment [SNFPA]; USDA, 2001, 2004) and in similar, forthcoming forest management plans. These treatments involve mechanical thinning and the application of prescribed fire and are generally similar to those being experimentally investigated by the National Fire and Fire Surrogate Study (FFS) (Schwilk et al., 2009). The FFS program is a cooperative program among federal agencies, universities and private organizations to investigate the relative effects of fire and fire surrogate (mechanical) treatments on forest ecology and fire risk (Schwilk et al., 2009). The FFS provided an opportunity to understand better the potential impacts of vegetation treatments on habitat quality for fishers by taking advantage of planned experimental treatments to be applied as part of the FFS study.

The FFS study included two sites in California's Sierra Nevada: Blodgett Forest Research Station (BFRS) and its satellite study site in Sequoia-Kings Canyon National Park (SEKI). The BFRS study site was one of 12 main study sites contributing toward long-term research on the effectiveness of various fuel management treatments to restoring fire as an ecosystem process and reducing the risk of catastrophic fires. The four treatments at BFRS included no treatment (control), mechanical harvest, mechanical harvest followed by area burn, and fire only treatments (area burn) (Stephens and Moghaddas, 2005). The SEKI research was focused on different burning strategies and included early and late season burns as well as control units (Knapp et al., 2005). By collecting the same suite of habitat variables that have been used to assess fisher resource selection models for fisher (e.g., Zielinski et al., 2004, and models presented herein) before and after treatment implementation, a quantitative assessment of the short-term impacts of FFS treatments on fisher habitat quality can be made. Additionally, given the general similarities between treatments described in the SNFPA and the FFS treatments, the opportunity will exist to develop a qualitative understanding of potential impacts on fisher habitat resulting from implementation of SNFPA treatments.

Thus, the primary objective of this research was to compare changes in habitat conditions important to fisher at the BFRS and SEKI FFS sites resulting from treatment implementation. Importantly, we did not examine the effects of treatments on fishers themselves (in fact, fishers do not occur on or near the BFRS site; Zielinski et al., 2005); instead we evaluated the effects of treatments on *predicted habitat value*. We assessed change in predicted probability of resource use (as a surrogate for habitat quality) for fishers and we tracked changes in select variables presumed to be important to fishers and other species associated with old-forest conditions. This information will help us understand how we can improve vegetation management to reduce risks of severe wildfire while maintaining habitat value for fishers.

2. Materials and methods

2.1. Fire and Fire Surrogate Study areas

BFRS is a 1780 ha experimental forest owned and managed by the University of California, Berkeley. BFRS is located in the central Sierra Nevada, El Dorado County, California. Common tree species at BRFS are typical of those found in mid-elevation forests of the Sierra Nevada: Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), ponderosa pine (*Pinus ponderosa*), sugar pine (*P. lambertiana*), incense cedar (*Calocedrus decurrens*), California black oak (*Quercus kelloggii*) and tan oak (*Lithocarpus densiflora*). Mixed conifer habitats dominate BFRS, with some ponderosa pine dominated and montane hardwood-conifer also present. Old-growth stands are very limited. Topography is generally rolling with slope averaging <30%, and elevation ranges from ~1200 to 1500 m. Additional details about the BFRS study area can be found in Stephens and Moghaddas (2005). Fishers have been described as historically occurring in this part of the central Sierra Nevada (Grinnell et al., 1937), but currently appear to be extirpated from the region (Zielinski et al., 2005).

The Sequoia-Kings Canyon (SEKI) FFS site occurred in Tulare County within Sequoia National Park in the southern Sierra Nevada and is described in detail in Knapp et al. (2005). The SEKI site occurred at higher elevations than the BFRS site, ranging from 1900 to 2150 m and was dominated by old-growth mixed conifer. White fir was the dominant tree species in the study area, and others present included red fir, ponderosa pine, sugar pine, incense cedar, Pacific dogwood (*Cornus nutalli*) and California black oak. Topography is somewhat steeper at SEKI than BFRS, ranging from 20% to 50% slope. Fishers currently occupy the SEKI region (Zielinski et al., 2005).

Treatment units at each FFS site were identified by Fire and Fire Surrogate Study site managers (Knapp et al., 2005; Stephens and Moghaddas, 2005). BFRS was divided into management compartments ranging in size from ~15 to 30 ha. Twelve compartments (hereafter, treatment units) were randomly selected from all compartments at BFRS, and each was randomly assigned to one of the four treatments. Within each treatment unit, an array of existing permanent plots was complemented with an array of grid points established at 60 m intervals to create the FFS sampling locations, hereafter referred to as plots (Stephens and Moghaddas, 2005). At SEKI treatment units were established based on recent fire history, accessibility, and ease of applying prescribed fire treatments (Knapp et al., 2005). Treatment units ranged in size from 15 to 20 ha and plots were established at 50 m intervals within each treatment unit.

At BFRS, mechanical treatments occurred in two stages which included thinning from below during fall 2001 and mastication of approximately 90% of understory trees between 2 and 25 cm dbh (Stephens and Moghaddas, 2005). Mechanical plus fire units followed the same treatment schedule as mechanical only units but were followed with backing fires from 23 October 2002 to 6 November 2002 (Knapp et al., 2005). Fire only units were burned during the same period, but used strip head-fires. At SEKI, early season burns were conducted 20 and 27 June 2002 and late season burns occurred 28 September and 17 and 28 October 2001 (Knapp et al., 2005).

2.2. Fisher habitat use and availability data

From 1993 to 1997, Zielinski et al. (2004) conducted an extensive study of fisher ecology at two locations in California, including a study site in the Tule River watershed of Sequoia National Forest, approximately 50 and 350 km south of the Sequoia-Kings Canyon and Blodgett Forest FFS sites, respectively. The authors used radio telemetry techniques to assess habitat use and selection by resting fishers (Zielinski et al., 2004) and baited track-plate station surveys to assess habitat selection by active or foraging fishers (reported herein).

Field techniques used to investigate habitat associations by resting fishers are reported in detail by Zielinski et al. (2004). The authors used ground-based telemetry surveys to track fishers to resting locations which typically occurred in large trees or logs (Zielinski et al., 2004). Walk-in surveys were conducted when radio-telemetry signals indicated animals were inactive, and efforts were made to track each radio-collared fisher to a resting location at least once per week. Individual fishers that met minimum monitoring criteria were considered 'focal' animals for resource selection analysis. For 19 focal animals, 100% Minimum Convex Polygon home ranges were calculated and 20 random Universal Transverse Mercator (UTM) coordinate locations were generated within each home range. Habitat data were collected at all rest sites, centered on the resting structure (typically a large tree or log). Habitat availability was assessed at each random location by installing a random habitat plot and a structure-centered habitat plot using a modified T-square sampling approach (Besag and Gleaves, 1973). The random habitat plot was installed in a random direction 10–50 m from the random UTM coordinate, and the second plot was then established in a direction away from the original random coordinate and was centered on a large tree or log with characteristics similar to those used by resting fishers in the study area. Thus, Zielinski et al. (2004) collected two sources of habitat available to resting fishers: random points and structure-centered random points.

To assess habitat use and selection by foraging (active) animals within the Tule River study area of Zielinski et al. (2004), we systematically surveyed for fishers from July to November 1995 by deploying track-plate stations using a 2 km grid. Plywood track-plate stations (Ray and Zielinski, 2008) were established at 101 grid locations ranging in elevation from ~600 to 2700 m throughout the area. Track-plate stations were baited with chicken and surveyed for 4 weeks, revisited by field technicians weekly to collect tracks and replace bait. Habitat plots were installed at all track-plate survey locations. This sampling approach simultaneously sampled used and available habitat resources; locations where fishers were detected at least once during the 4-week survey period were considered used and those that failed to detect fisher during a survey were considered available but unused.

The same habitat sampling protocol was used at all sites (fisher rest sites, random points, and track-plate stations) and was described in detail by Zielinski et al. (2004). Briefly, this protocol focused on collecting habitat variables presumed to be important to fishers and included several to describe topography, canopy closure, tree size, and tree abundance (Table 1). Percent slope was measured by averaging the uphill and downhill clinometer recordings from plot center. Water was considered present if visually estimated to be <100 m of plot center. Canopy closure was estimated using a concave spherical densitometer by recording at plot center and at the termini of two perpendicular 25 m transects; the transects were established based on a random azimuth and intersected at plot center. Canopy closure estimates from these five locations were used to calculate average canopy closure. A 20 Basal Area Factor prism was used to estimate variables describing forest composition and structure. For each tree 'in' the prism sweep, diameter at breast height (dbh) was measured, tree species was identified, and trees were assigned a condition class (Maser et al., 1979).

2.3. Developing habitat selection models

The data collected at rest sites, random points, structure-centered random points, and track-plate stations were used to estimate relative habitat value using multivariate resource selection analysis (Manly et al., 2002) to develop models that predict resting and foraging habitat. Fishers in the Sierra Nevada typically rest in trees at sites characterized by late-successional features (e.g., dense canopy, large diameter trees and snags; Zielinski et al., 2004; Purcell et al., 2009). In contrast, fishers can forage in areas lacking these features. Evaluating the effects of fuels treatments on both resting and presumptive foraging habitat represents a balanced view of the effects on fisher habitat.

The resting habitat resource selection functions developed for fishers in the southern Sierra (Zielinski et al., 2004) relied on comparing habitat data collected at rest sites to that collected at structure-centered random points located within fisher home ranges, and thus represented resource-selection conditioned on the presence of a large woody structure at the center of the plot. The sampling grids at the SEKI and BRFS FFS sites, however, were established systematically within treatment units and thus not centered on large woody structures. Accordingly, we considered the FFS sampling points akin to the random habitat availability points of the Zielinski et al. (2004) study. Thus, rather than directly applying the resting habitat resource selection models developed by Zielinski et al. (2004), we developed new resting habitat models by comparing habitat data collected at random points to the same type of habitat data collected at fisher rest sites.

Resting habitat models were developed by considering a population of *N* units some of which are used (i.e., the resting sites) and others that constitute a random sample of sites that are available, but for which use is unknown (Manly et al., 2002). These resource selection functions assume the form: $W_{(x)} = \exp(\beta_1 x_1 + \beta_2 x_2 + \ldots \beta_n x_n)$, where $W_{(x)}$ is the relative probability of resource use for the given combination of covariates (χ_i) and slopes (β_i) are estimated using maximum likelihood methods. The intercept in this function is treated as a nuisance parameter and excluded from the logistic model (McCullagh and Nelder, 1989).

Table 1
Definitions and acronyms for variables collected at fisher habitat plots on Fire and Fire Surrogate sites in the Sierra Nevada (adapted from Zielinski et al., 2004).

Variable	Acronym	Measurement technique/definition
Percent slope	SLOPE	Clinometer; average of uphill and downhill readings
Presence of water within 100 m	WATER	Visual estimate
Basal area hardwoods	BAHDW	20-Factor prism (m^2/ha)
Average dbh	DBHAVE	Mean dbh (cm) of trees in the prism sample
Average hardwood dbh	DBHAVEH	Mean dbh (cm) of hardwoods in the prism sample
Standard deviation dbh	DBHSTD	Standard deviation of mean dbh (cm) of trees in the prism sample
Maximum dbh	DBHMAX	Maximum dbh (cm) of trees in the prism sample
Presence of large conifer snag	CONSNAG	Presence of $\geqslant 1$ conifer snag >102 cm dbh in the prism sample
Average canopy closure	CANAVE	Mean of densitometer readings at five plot locations

The 'foraging' habitat models were developed by comparing the characteristics at sites where fishers did, and did not, visit baited track-plate stations (Ray and Zielinski, 2008). Because the chicken bait is almost always removed by visiting fishers (82.0% of occasions; W. Zielinski, unpubl. obs.), we assume that visits to stations reflect habitat selected by foraging fishers. We recognize that not all fishers that consume bait may have been actively foraging, but such detection surveys provide better estimates of the habitats visited by active fishers with this same spatial resolution. The foraging models were resource selection *probability* functions, wherein the resource is considered a population of N units that can be assigned unambiguously to the status of either used (stations with detections) or unused (stations without detections) (Manly et al., 2002). Resource selection probability functions have the following form:

$$W_{(x)} = \frac{\exp(B_0 + B_1x_1 + B_2x_2 + \ldots + B_nx_n)}{1 + \exp(B_0 + B_1x_1 + B_2x_2 + \ldots + B_nx_n)}$$

where $W_{(x)}$ is the predicted probability of resource use for the given combination of covariates (χ_i), and slopes (β_i) are estimated using maximum likelihood methods. For the prospective sample, the intercept is an estimable parameter and therefore included in the resource selection probability function.

For both the resting and foraging models, Akaike's Information Criterion (Akaike, 1973) was used to select from a number of candidate predictive models, which were generated in the same manner as described by Zielinski et al. (2004). If a single candidate model for either analysis accounted for >0.90 of the Akaike weight, it was considered the best model; otherwise model averaging (Burnham and Anderson, 2002) was applied to the highest ranking models whose cumulative Akaike weights were >0.90.

2.4. Habitat sampling at fire and fire surrogate sites

We used the habitat sampling protocols of Zielinski et al. (2004; Section 2.2) to quantify the effects of fire and fire surrogate treatments on fisher habitat. We randomly selected 10 plots within each FFS treatment unit and collected habitat data before and after the application of treatments. All habitat sampling occurred between June 2001 and July 2004, primarily during the spring and summer before leaf fall. We conducted post-treatment habitat sampling approximately 1 year after treatments were applied.

2.5. Assessing treatment effects

We used the resource selection models to quantify the effects of treatments on habitat suitability. For each FFS plot sampled we estimated the resting and foraging habitat value prior to, and then again after, the treatments; the difference represented the change in relative habitat suitability due to treatment. We used nested Analysis of Variance (ANOVA) to test the null hypothesis that the change in habitat suitability did not differ among treatment types. Because treatment types varied between the two study areas, we independently tested treatment effects for each site. For the BFRS site, the primary null hypothesis was that there was no difference among the four treatment types (control, mechanical, fire, mechanical plus fire); for the SEKI site we tested three treatment types (control, early burn, late burn). For both study areas, the additive model was: $y_{ijk} = \mu + \alpha_I + \beta_{ij} + \varepsilon_{ijk}$ with $i = 1, 2, 3, 4$ (BFRS treatments), $i = 1, 2, 3$ (SEKI treatments), $j = 1, 2, 3$ (experimental units randomly assigned to treatments), and $k = 1$–10 (plots nested within experimental units). Post-hoc comparisons among treatments, within study area, were evaluated using the Ryan–Einot–Gabriel–Welsch test (SAS Institute, 1990).

In addition to evaluating the effects of treatments on the multivariate predictive models, we evaluated treatment effects separately for one important variable: average canopy closure. This habitat feature has been universally associated with fisher habitat selection in California (e.g., Carroll et al., 1999; Davis et al., 2007; Zielinski et al., 2004; Purcell et al., 2009) and we presumed FFS treatments would likely have greater impact on canopy closure than other variables considered important to fishers. All statistical analysis was conducted using SAS Statistical Software (SAS Institute, 1990).

3. Results

3.1. Habitat selection models

Resting habitat was predicted by a single model accounting for >0.90 of the Akaike weight:

$$W(x) = \exp(0.0470^{*}\mathrm{CANAVE} + 0.0235^{*}\mathrm{DBHAVEH} + 0.0250^{*}\mathrm{DBHMAX})$$

When comparing resting sites used by fishers to sites randomly located with home ranges, fishers tended to select resting sites that had denser canopy (CANAVE), larger average hardwood diameter (DBHAVEH), and larger maximum tree size (DBHMAX) than sites randomly available within their home range. In contrast to the resting habitat model, no single foraging model demonstrated an overwhelming fit to the data; 10 models combined to account for >0.90 of the Akaike weight. These were averaged to produce the following model:

$$W(x) = \frac{\exp(-7.834 + 0.0724^{*}\mathrm{CANAVE} + 0.0167^{*}\mathrm{DBHAVEH} - 0.0080^{*}\mathrm{DBHMAX} - 0.0155^{*}\mathrm{DBHAVE} - 0.0379^{*}\mathrm{BAHDW} + 0.9581^{*}\mathrm{WATER} + 1.501^{*}\mathrm{CONSNAG} + 0.2387^{*}\mathrm{SLOPE})}{1 + \exp(-7.834 + 0.0724^{*}\mathrm{CANAVE} + 0.0167^{*}\mathrm{DBHAVEH} - 0.0080^{*}\mathrm{DBHMAX} - 0.0155^{*}\mathrm{DBHAVE} - 0.0379^{*}\mathrm{BAHDW} + 0.9581^{*}\mathrm{WATER} + 1.501^{*}\mathrm{CONSNAG} + 0.2387^{*}\mathrm{SLOPE})}$$

The foraging model was more difficult to interpret, but included positive influences of canopy density (CANAVE), hardwood diameter (DBHAVEH), conifer snags (CONSNAG), distance to water (WATER) and steepness of slope (SLOPE). The average dbh of all trees (DBHAVE) and maximum tree size (DBHMAX) contributed negative influences.

3.2. Treatment effects

There were significant negative effects of treatment on predicted resting habitat suitability at both study areas, and highly significant effects on canopy closure (CANAVE) (Fig. 1, Table 2). The influence of canopy reduction on resting habitat suitability likely accounted for the significant treatment effects at both areas. At BRFS, the effects of mechanical and plus fire treatment had a significantly greater impact of predicted resting habitat than the control sites and fire only sites, though the control did not differ from the fire only treatment nor did the mechanical differ from the mechanical plus fire treatments (Fig. 1). At SEKI late season burns, but not early season, had significant negative effects on predicted resting habitat (Fig. 1). Treatment effects on predicted foraging habitat value were not evident at BRFS ($F = 0.93$, $P = 0.468$) but marginally significant at SEKI ($F = 4.66$, $P = 0.060$).

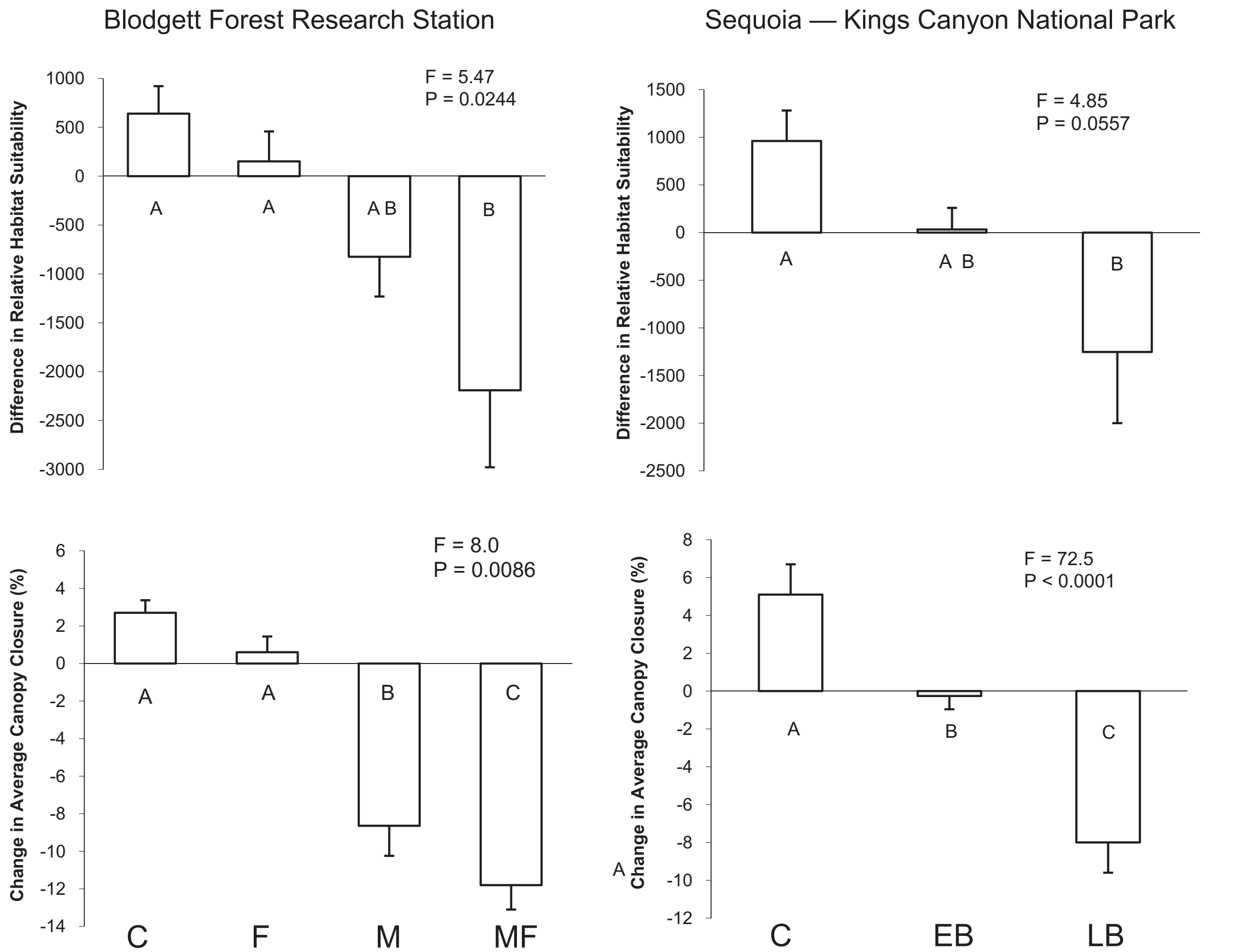

Fig. 1. Treatment means (SE) for changes in relative predicted probability of resting fisher habitat suitability and average canopy closure at Fire and Fire Surrogate Study (FFS) sites at Blodgett Forest Research Station, California and Sequoia-Kings Canyon National Park. For each variable, negative values indicate reduction due to FFS treatments. Treatment types, listed along the bottom of the figure are: C = control, F = fire only, M = mechanical only, MF = mechanical plus fire, EB = early burn and LB = late burn. Statistically significant differences among treatments, within study area, are indicated by different letters immediately under the bars (i.e., A, B, and C).

Table 2
Summary statistics for average (SE) percent canopy closure (CANAVE), average hardwood dbh (cm, DBHAVEH) maximum dbh (cm, DBHMAX), relative habitat suitability for resting fisher (RESTING), and probability of use by foraging fisher (FORAGING) for pre-treatment and post-treatment plots at the Blodgett Forest Research Station (BFRS) and Sequoia-Kings Canyon (SEKI) Fire and Fire Surrogate Study sites in the Sierra Nevada.

Study area	Sampling effort	Treatment	CANAVE	DBHAVEH	DBHMAX	RESTING	FORAGING
BFRS	Pre-treatment	Control	88.9 (1.6)	19.3 (5.3)	82.2 (3.4)	1309.8 (329.1)	0.18 (0.02)
		Fire	88.9 (1.3)	20.6 (5.4)	83.2 (4.3)	2001.5 (830.7)	0.17 (0.03)
		Mechanical	90.8 (1.3)	41.1 (7.1)	99.8 (3.4)	3504.4 (668.8)	0.17 (0.02)
		Mechanical + fire	90.6 (1.1)	40.0 (6.9)	94.7 (3.5)	4496.6 (1596.6)	0.12 (0.01)
	Post-treatment	Control	91.6 (1.3)	26.6 (6.2)	88.1 (3.1)	1949.2 (400.3)	0.16 (0.02)
		Fire	89.5 (0.9)	28.2 (6.3)	86.5 (2.7)	2153.0 (659.2)	0.15 (0.02)
		Mechanical	82.2 (1.4)	31.9 (7.4)	97.9 (3.1)	2679.4 (763.1)	0.09 (0.02)
		Mechanical + fire	78.8 (1.3)	31.2 (7.1)	93.5 (3.5)	2306.3 (861.6)	0.05 (0.01)
SEKI	Pre-treatment	Control	83.9 (1.3)	1.3 (1.3)	153.8 (5.1)	3909.7 (906.0)	0.14 (0.02)
		Early burn	91.1 (1.1)	3.0 (1.9)	144.2 (4.9)	3700.0 (465.3)	0.17 (0.02)
		Late burn	86.4 (1.6)	2.0 (2.0)	163.4 (5.1)	5787.4 (1304.2)	0.16 (0.02)
	Post-treatment	Control	89.0 (1.2)	1.0 (1.0)	154.7 (5.2)	4870.8 (1123.0)	0.17 (0.02)
		Early burn	90.9 (1.4)	1.7 (1.3)	143.6 (5.0)	3712.7 (513.1)	0.17 (0.02)
		Late burn	78.4 (2.4)	2.0 (2.0)	164.5 (5.7)	4534.7 (968.2)	0.09 (0.02)

Habitat suitability for fishers at SEKI was somewhat higher than at BFRS prior to treatment, though predicted resting and foraging values were relatively low for both sites (Table 2). At SEKI, hardwoods, an important predictor in both models, were less common and of small diameter ($\bar{x} = 2.1$ cm dbh, SE = 1.00) though maximum tree diameter was large ($\bar{x} = 153.8$ cm dbh, SE = 3.00). Hardwoods were generally larger at BFRS ($\bar{x} = 30.2$ cm dbh, SE = 3.21), but DBHMAX was smaller than at SEKI ($\bar{x} = 90.0$ cm dbh, SE = 1.94).

4. Discussion

Mechanical plus fire treatments and late-season prescribed fire had significant short-term impacts on predicted fisher resting habitat conditions, as well as on canopy closure, a key habitat element for fisher in California. Although the treatments that included mechanical methods had greater short-term reduction on estimated fisher resting habitat suitability than prescribed fire alone, these effects were mitigated by the fact that mechanical treatments could avoid effects on individual trees of high value to fishers because they thinned only trees <25 cm dbh. Furthermore, even the use of fire could be controlled somewhat by raking debris from the base of particular trees that were viewed as important to protect and retain after the treatment. Thus, it appears that if care is taken to apply treatments with the goal of protecting large hardwoods and conifers, the potential reduction in predicted habitat quality may be reduced. The biggest effect of treatments on predicted resting habitat, however, was the reduction in canopy closure; a result confirmed at BFRS by another study (Stephens and Moghaddas, 2005). Canopy density is an important predictor of fisher habitat at a variety of scales and all treatments reduced canopy. Mechanical treatments reduced canopy density much more than did fire (see CANAVE, Table 2), presumably because, unlike mechanical thinning, fire killed but did not remove trees. Fire also may have delayed effects on loss of habitat suitability, compared with mechanical methods, which immediately affect the value of variables directly related to habitat suitability. We assessed the effects of prescribed fire 1 year after its application, perhaps before some of its effects on variables important to predicted resting habitat were realized. For example, canopy of some conifer trees in the understory had not been completely reduced because many dead trees still retained needles a year after treatment. Importantly, however, regardless of the method by which canopy is reduced, it can recover more quickly than the loss of large live and dead trees, and thus effects on canopy would be expected to be short term in nature.

The short-term effects of FFS treatments on fisher foraging habitat were not statistically significant at either site. This is likely because the complex RSFP foraging model developed to predict foraging habitat suitability included several variables that were either not affected by the FFS treatments or were relatively rare at each site. The foraging model includes slope and availability of surface water, neither of which is affected by vegetation management. Foraging habitat is also much less likely to be limiting to fishers than resting habitat because it can often be fulfilled at locations that do not have mature forest elements and because the fisher diet is quite diverse in the Sierra Nevada (Zielinski et al., 1999). However, the statistically ambiguous effect of treatment on foraging conditions at SEKI (i.e., $P = 0.060$) may have biological meaning. It is possible that habitat conditions associated with sites where fishers visit track stations (where they are assumed to be foraging), such as average canopy, basal area of hardwoods and conifer snags for example, may be diminished by prescribed fire in the short term, particularly in national park and park-like settings in the southern Sierra Nevada.

The fact that mean habitat suitability, as estimated using the resource selection functions, was greater at SEKI than BFRS may be due to the fact that the SEKI site was much closer than BFRS to the location where the data were collected to develop the selection functions (i.e., Sequoia National Forest) and within the area of the Sierra Nevada currently occupied by fishers. Moreover, the national park site, unlike BFRS, has not been managed extensively for forest products and currently appears to have greater capability of providing suitable habitat for resting fishers. Predicted values at the SEKI location were surprisingly low, despite its protected status, largely because of the reduced hardwood component which appears to be an important element of fisher resting habitat.

Although our results suggest that the short-term effects of treatments on fisher habitat suitability are statistically significant, the effects are relatively modest. These results must be interpreted in the context of at least three additional factors. First, the study areas used in this research had relatively low predicted habitat value for fishers *prior to* treatment. Thus, although the decrease in predicted resting and foraging habitat value attributed to the treatments was small, even moderate reductions in habitat value at sites that are already of relatively low predicted value may have disproportionately greater impact on habitat recovery. The short-term negative effects of treatments, however, may be mitigated by the beneficial effects of the treatments on subsequent stand development and future fire severity (Scheller et al., 2011), so it will be important to monitor future changes in predicted habitat value as the stands respond to the treatments.

Second, we do not assess direct effects on fishers, only on the *predicted effects* of vegetation change on a multivariate resource selection function. This function was developed from a fisher population studied elsewhere in the southern Sierra and, although the functions represent the model that best fit the data, there is no certainty that fishers are responding directly to variation in the predictor variables – these may be merely correlated with features of direct importance to fishers. Thus, the models may be an imperfect abstraction of fisher habitat relations as applied to our two study areas. They are, however, the only option available and they make sense in terms of what is known about fisher habitat ecology.

Third, we addressed only the effects of treatments on individual stands, not on the watershed or landscape scales that we know to be important to wide-ranging predators such as the fisher (Carroll et al., 1999; Davis et al., 2007; Thompson et al., 2011; Spencer et al., 2011). A comprehensive analysis would include also the effects of the spatial and temporal distribution of fuels treatments on home-range sized areas that have biological meaning to individual fishers and to the maintenance of their populations. The specific tradeoff between the benefits of size of treatment area on fire protection and the reduction in predicted fisher habitat are, however, unknown and not intuitive. Scheller et al. (2011) used spatial modeling to simulate the stochastic and interacting effects of wildfires and fuels management on fisher habitat and population size. Their simulations suggested that the direct, negative effects of fuel treatments on fisher habitat were generally smaller than the indirect, positive effects of fuel treatments, because fuels treatments reduced the probability of large wildfires that can damage and fragment habitat over larger areas. However, there was considerable uncertainty in their projections due to stochastic spatial and temporal wildfire dynamics and fisher population dynamics. Direct measurements of the effects of treatments on habitat, as reported here, are an important adjunct to studies that address via simulation the effects at much larger scales.

5. Conclusions

Land managers faced with balancing the challenges of maintaining habitat for fisher and reducing the threat of catastrophic fire can take relatively simple steps to mitigate the effects of vegetation management projects on fisher habitat. First, to mitigate the anticipated reduction of canopy closure managers can plan actions that will maintain other habitat elements important to fisher (e.g., presence of large diameter hardwoods). Second, if conditions permit, early season burns appear to be preferable to late season burns in terms of the short-term impacts on fisher habitat, but should occur after the fisher denning period (mid-March through mid-May). If conditions necessitate burning earlier than mid-May, efforts should be made to avoid treating areas that have high density of structures likely to be used by females for denning (Zielinski et al., 2004; Purcell et al., 2009; Zhao et al., 2012). Whenever possible, managers should plan vegetation management activities in a manner that disperses treatments over space and time to minimize impact on individual fishers. We recognize, however, that this goal can contradict the guidelines for installing effective fuels treatments (i.e., Finney, 2001). Lastly, managers must be willing to commit to long-term monitoring efforts to better understand the impacts of vegetation management activities on fishers and other species of wildlife. Monitoring should include both a habitat component, such as the approach described herein using predictive models, as well as a population monitoring component (e.g., Zielinski et al., 2013). Only with such a commitment can we begin to better address the uncertainties inherent in balancing the tradeoff between treating stands to reduce their vulnerability to loss from crown fire and the reductions in fisher habitat suitability that occur when such treatments occur.

Finally, the research also demonstrates that empirical models have important utility for evaluating the effects of treatments, even when – as in this case – the species of interest does not reside in the treatment area and if it had, would not occur at high enough densities to rigorously test the effects of a treatment on its use of habitat. In short, predictive models make quantitative evaluations of treatment effects possible when direct experimentation is not possible.

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