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Short-term Effects of Fire and Fire Surrogate Treatments
On Fisher Habitat in the Sierra Nevada

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INTRODUCTION

The National Forest Management Act requires national forests to manage habitat to maintain viable populations of native vertebrates (36 CFR 219.19). This requirement has been recognized in the Sierra Nevada Forest Plan Amendment (USDA 2001, 2004), which emphasizes viability concerns for fisher (*Martes pennanti*) and other species associated with mature forest conditions (e.g., America marten [*M. Americana*], goshawk [*Accipiter gentilis*], and spotted owl [*Strix occidentalis*]). For fisher the concerns are exacerbated due to the species' limited geographic distribution and association with mature forests; the species currently is limited in distribution to <50% of its historical range, occurring only in the Sierra Nevada south of Yosemite National Park (Zielinski *et al*. 1995, 2005), and recent research stresses the importance of large trees and dense canopies to meet daily resting and annual denning needs (Zielinski et al. 2004, Mazzoni 2002, Dark 1997, Seglund 1995). Of equal urgency to maintaining habitat for viable populations of fisher and other wildlife species is the need to manage forest fuels to protect human communities occurring within the Sierra Nevada and to reduce risk of catastrophic fire, ultimately restoring historic fire regimes (USDA 2001, 2004).

The goal of managing habitat for a viable fisher population in the Sierra Nevada conflicts somewhat with fuel management strategies described in the SNFPA. These treatments (USDA 2004) are generally similar to those being experimentally investigated by the ongoing Fire and Fire Surrogate Treatments (FSS) study (Weatherspoon 2000). The conflict arises in part due to the uncertainty surrounding the effects of various fuels management treatments on fisher habitat, and how cumulative

effects of these treatments may affect fisher viability. Only by examining the effects of proposed treatments on fisher habitat quality will this uncertainty be reduced and the potential impacts of treatments on fisher habitat better understood and potentially mitigated.

The Fire and Fire Surrogate (FFS) program provides an unprecedented opportunity to better understand the potential impacts of vegetation treatments on habitat quality for fisher by taking advantage of planned experimental treatments to be applied as part of the FFS study. The FFS study includes 2 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 is one of 10 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 4 primary treatments include a control (no treatment), mechanical harvest (typically including mastication following harvest), mechanical harvest followed by area burn, and fire only treatments (area burn) (Stephens and Moghaddas 2001). The SEKI research is focused on different burning strategies and includes 3 treatments: control, early season burns, and late season burns (Keeley and Knapp 2001). By collecting the same suite of habitat variables that have been used to assess fisher resource selection models (Manly *et al*. 1993) for fisher (Zielinski *et al*. 2004, and 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 is to compare changes in habitat conditions important to fisher at the Blodgett Forest Research Station (BFRS) and Sequoia-Kings Canyon (SEKI) FFS resulting from treatment implementation. Specifically, we will assess change in predicted probability of resource use (as a surrogate for habitat quality) for fishers as well as select variables considered important to fisher and other species associated with old-forest conditions.

STUDY AREAS

Blodgett Forest Research Station (BFRS) is a 1780 ha experimental forest owned and managed by the University of California, Berkeley. BFRS is located along the Georgetown Divide 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. Topography is generally rolling with slope averaging <30%, and elevation ranges from ~1200 – 1500 m above sea level. Fishers historically occurred in this part of the central Sierra Nevada was

historically occupied by fishers (Grinnell *et al*. 1937), but currently appear to be extirpated from the region (Zielinski *et al*. 2005).

The Sequoia-Kings Canyon (SEKI) FFS site occurs in Tulare County within Sequoia National Park in the southern Sierra Nevada. The FFS site at SEKI is located on a NW aspect bench above the Marble Fork of the Kaweah River (Knapp *et al*. 2001). The FFS site occurs at higher elevations than the BFRS site, ranging from 1900 – 2150 m above sea level and is 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 – 50% slope. Fishers historically occurred in the region (Grinnell *et al*. 1937) and are known to currently occupy the area (Zielinski *et al*. 2005).

METHODS

Field Sampling

Plot Selection Within Treatment Units – Treatment units at each FFS site were identified by Fire and Fire Surrogate site managers (Knapp and Keeley 2001, Stephens and Moghaddas 2001). BFRS is divided into management compartments ranging in size from ~15 – 30 ha. Twelve compartments (hereafter, treatment units) were randomly selected from all compartments at BFRS, and each was randomly assigned to one of the 4 treatments; compartments used for the FFS study are hereafter referred to as treatment units. 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

2001). At SEKI treatment units were established based on recent fire history,

accessibility, and ease of applying prescribed fire treatments (Keeley and Knapp 2001).

Treatment units range in size from 15 – 20 ha and plots were established at 50 m

intervals within each treatment unit. We randomly selected 10 plots within each

treatment unit to conduct habitat sampling (Figure 2) before and after treatment

implementation.

Habitat Sampling– We followed habitat sampling protocols used by Zielinski *et al*.

(2004) to assess habitat available to radio-collared fishers at 2 study areas in California.

The authors collected habitat data grouped into 6 variable families: topography,

vegetation cover type, tree abundance, tree size, ground cover, and canopy closure

(Zielinski *et al*. 2004). This approach used a combination of fixed plot and plotless

techniques and was developed to describe habitat conditions in a logistically realistic

manner (i.e., collect all data in <2 hours per site [per obs.]). Variables identified by

Zielinski *et al*. (2004) as important to fishers included several from the following variable

families: 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 within 100 m of

plot center. Canopy closure was estimated using a concave spherical densiometer.

Measurements were recorded 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 5 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*. 1977).

Within each treatment unit at BFRS and SEKI, we collected habitat data at 10 plots

before and after treatment implementation. All habitat sampling occurred during late

spring or summer, and efforts were made to complete post-treatment sampling

approximately 1 year after treatment implementation. In addition to recording dbh,

condition class, and species for each tree in the prism sample, we recorded azimuth

and estimated distance from plot center to each measured tree. This was done to

facilitate post-treatment sampling and minimize influence of observer error on estimated

changes in habitat conditions following treatment implementation. All technicians who

collected habitat data were collected trained by wildlife biologists with extensive

experience using this protocol. The plot marker established by the each study area's

FFS program served as plot center for our sampling.

Estimating Habitat Suitability

The habitat sampling procedures described above were applied by Zielinski *et al*.

(2004) at a collection sites used by or available to fishers in order to assess habitat

selection. Sites sampled by Zielinski et al. (2004, unpublished) included (1) sites used

by for daily resting bouts and annual denning (hereafter, rest sites), (2) random

locations potentially available to fishers for resting purposes (hereafter, random points), and (3) sites established on a 2 km grid within their Southern Sierra Study Area (Sequoia National Forest, Tulare County) to assess habitat selection by fishers at sooted track-plate stations (assumed to reflect habitat selection by fishers engaged in foraging bouts; hereafter, track plate stations). In this report we present habitat selection analysis heretofore unpublished, though collected as part of the study reported by Zielinski *et al*. (2004). We compare rest sites to random locations to describe unconditional resting habitat selection (i.e., selection not contingent on the presence of a large woody structure) as well as habitat selection by foraging fishers. The random locations we use to describe unconditional resting habitat selection are the starting points of the modified T-square sampling approach (Besag and Gleaves 1973) used by Zielinski *et al*. (2004) to establish habitat availability plots.

The comparison of rest sites to available sites and the comparison of track plate stations with detections to those without detections both fall under the general guise of Resource Selection Analysis (Manly et al. 2002) and can be analyzed using logistic regression methods. The details of each comparison differ slightly. In the case of rest sites vs. random sites (hereafter the Resting model), the comparison is made between two samples collected at different times and the sample is therefore considered retrospective. The estimated Resource Selection Function (RSF) assumes the form:

$$W_{(x)} = \exp(\beta_1 x_1 + \beta_2 x_2 + ...\beta_n x_n)$$

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 the RSF was treated as a nuisance parameter and excluded from the logistic model (McCullagh and Nelder 1989).

For the comparison of track plate stations with detections and those without detections (hereafter the Foraging model), used and unused resources are sampled simultaneously and the sample is accordingly considered a prospective sample. The Resource Selection Probability Function has the form:

$$W_{(x)} = \frac{\exp(B_0 + B_1 x_1 + B_2 x_2 + ... + B_n x_n)}{1 + \exp(B_0 + B_1 x_1 + B_2 x_2 + ... + B_n x_n)}$$

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 RSFP. For both the Resting and Foraging models, Akaike's Information Criterion (Akaike 1973) was used in the same manner as reported by Zielinski *et al*. (2004) and model averaging methods were used when appropriate. 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 theory (Burnham and Anderson 2002) was applied to the highest ranking models whose cumulative Akaike weights were >0.90. All statistical analysis was conducted using SAS Statistical Software (SAS Institute 1990)

Assessing Effects of FFS Treatments on Fisher Habitat

To assess the effects of Fire and Fire Surrogate treatments on fisher habitat in the Sierra Nevada, we assessed habitat suitability before and after treatment

implementation. We applied the Resource Selection Functions and Resource Selection

Probability Functions described above to estimate the change in habitat suitability due

to treatment implementation. We used nested Analysis of Variance (ANOVA) to test the

general null hypothesis that the change in habitat suitability did not differ among

treatment types. Because treatment types varied between the 2 study areas, we

independently tested treatment effects for each site. For the BFRS site, the primary null

hypothesis to be tested was no difference among the 4 treatment types (control,

mechanical, fire, mechanical plus fire):

$$H_o: \alpha_1 = \alpha_2 = \alpha_3 = \alpha_4$$

The null hypothesis for SEKI was identical, though limited to 3 treatment types (control,

spring burn, fall burn):

$$H_o: \alpha_1 = \alpha_2 = \alpha_3$$

For both study areas, the additive model was:

$$y_{ijk} = \mu + \alpha_l + \beta_{ij} + \varepsilon_{ijk}$$

and 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).

Habitat suitability was calculated and the hypothesis tested for the Resting model RSF using data collected from 19 individual fishers (>500 used and available sites, sampled from 1994-1997). The hypothesis was also tested for the Foraging model RSPF using data collected from 101 track plate stations surveyed in the southern Sierra during 1996. In addition, we tested this null hypothesis for average canopy closure due to its importance to fisher elsewhere in California (e.g., Zielinski *et al*. 2004, Mazzoni 2002, Dark 1997, Seglund 1995)) and our assumption that FFS treatments were likely to have greater impact on canopy closure than other variables considered important to fishers (e.g., maximum tree diameter). For each plot sampled, we estimated the Resting RSF prior to treatment and following treatment; the difference between these values estimated the change in relative habitat suitability due to treatment implementation. The same process was repeated for the Foraging RSPF and other variables of interest.

RESULTS

Development of Resource Selection Functions from Existing Data

Resource selection functions (RSFs) were developed for resting fishers in California using methods identical to those described by Zielinski *et al*. (2004). Applying this approach to compare habitat conditions at sites used by radio-telemetered fishers to those available within their home ranges resulted in a single RSF model accounting for the >0.90 of the Akaike weight in the pool of potential models. Because one model accounted for the majority of the Akaike weight, it was considered the best available model and estimated Resting RSF model took the form:

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

Thus, fishers in California tended to select resting sites that had denser canopy, larger average hardwood diameter, and larger maximum tree size than sites randomly available within their home ranges.

The Foraging RSPF model was developed by comparing habitat characteristics at track plate stations in the Southern Sierra that detected fishers to those that failed to detect fishers. Unlike the case for the Resting RSF model, no individual resource selection model accounted for the majority of the Akaike weight. The maximum Akaike weight for the foraging models was 0.125, and 10 models combined to account for >0.90 of the Akaike weight. Model averaging procedures (sensu Burnham and Anderson, were applied to these 10 models, and the resulting Resource Selection Probability Function assumed the form:

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

The lack of a single dominant foraging model necessarily resulted in the inclusion of numerous predictor variables in the model, though slope estimates for most variables do not differ from 0. These multivariate Resting RSF and Foraging RSPFs were applied

to data collected at each plot, and the difference between the post-treatment and pre-treatment values were used to estimate the change in relative habitat suitability (Resting RSF) and the change in predicted probability of use (Foraging RSPF).

Application of Resource Selection Models at FFS Sites

Habitat sampling occurred at FFS sites from June 2001 – July 2004. All pre-treatment sampling was completed within 1 year preceding treatment implementation. Post-treatment sampling was post-treatment sampling was generally completed 1 year after treatment implementation (Table 2). All habitat sampling occurred during the spring and summer and prior to leaf fall.

Habitat suitability for fishers at SEKI was somewhat higher than at BRFS prior to treatment, though RSF and RSPF values were relatively low for both sites (Table 3) compared to sites studied by Zielinski *et al*. (2004) where relative habitat suitability averaged 10198.6 (n = 452, SE = 998.4) at rest sites and 8768.3 (n = 385, SE = 4496.1). Predicted probability of use estimated for the Foraging RSPF was considerably higher at the southern Sierra study area of Zielinski *et al*. (2004) for sites detecting fisher ($\bar{x}$ = 0.29, n = 21, SE = 0.03) than at either SEKI or BRFS, but similar to sites that did not detect fisher ($\bar{x}$ = 0.14, n = 78, SE = 0.02). At SEKI, hardwoods were rare 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).

Nested ANOVA for the Resting RSF model indicate significant treatment effects at both study areas and highly significant effects on CANAVE (Figure 3, Table 3). The influence of canopy reduction on habitat suitability likely accounted for the significant treatment effects at both areas. At Blodgett, the effects of mechanical and mechanical plus fire treatments on resting habitat suitability differed significantly from the control sites, though the control did not differ from the fire only treatment (Figure 3). Treatment effects were not evident at Blodgett for the Foraging RSPF (F = 0.93, P = 0.4684) and marginally significant at SEKI (F = 4.66, P = 0.0600).

DISCUSSION

Our research indicates that Fire and Fire Surrogate (FFS) treatments have significant short-term impacts on fisher resting habitat quality, as well as canopy closure which is generally considered a key habitat element for fisher in California. At the Blodgett Forest Research Station FFS site, both mechanical treatments (mechanical and mechanical plus fire) significantly reduced fisher resting habitat suitability and average canopy closure. At the Sequoia-Kings Canyon FFS site, the late season burn treatment had a significant impact on fisher habitat suitability as well as canopy closure. The short-term effects of FFS treatments on fisher foraging habitat were generally not significant. This is likely because the complex RSFP 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 RSFP includes SLOPE and

WATER, neither of which is affected by vegetation management. Foraging habitat is also much less likely to be limiting to fishers than resting habitat, primarily because it can often be fulfilled at locations that do not have mature forest elements and because the fisher diet appears to be quite diverse in the Sierra Nevada (Zielinski et al. 1999).

The fact that mean habitat suitability, as estimated using the resource selection functions, was greater at SEKI than Blodgett may be due to the fact that the SEKI site was much closer than Blodgett to the location where the selection functions were developed (i.e., Sequoia National Forest) and within the area of the Sierra Nevada currently occupied by fishers. Moreover, the national park site, unlike Blodgett, has not been managed extensively for forest products and currently appears to have greater capability of providing suitable habitat for resting fisher. 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 the treatments that included mechanical methods (MECH and MECH/FIRE) had greater short-term reduction on estimated fisher resting habitat suitability than prescribed fire at BFRS, these effects were mitigated by the fact that mechanical treatments could target or avoid individual trees. Hardwoods and all large trees and snags are important predictors of fisher habitat use and the effects on these habitat elements could more easily be avoided using mechanical means of treatment. 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 habitat quality may be mitigated. The biggest effect of treatments, however, was the reduction in canopy closure. Canopy density is an important predictor of fisher habitat at a variety of scales (e.g., Carroll et al. 1999, Zielinski et al. 2004; Mazzoni 2002) and all treatments reduced canopy. However, canopy cover can recover more quickly than the loss of large live and dead trees so these effects would be expected to be short term in nature. Remeasurements of the treatment units in 5 or 10 years will provide important information on the recovery rate of canopy closure in treated stands.

Although our results suggest that the short-term effects of treatments on fisher habitat suitability are 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, relatively modest 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, so it will be important to monitor the change in predicted habitat value as the stands respond to the treatments.

Second, 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. 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. For example, treatment of 10 percent of the stands in a home-range sized area over a 30 year period, using the methods evaluated here, may have much less effect than treating 50% of the stands in the same area over 10 years. A cumulative effects analysis, where the impacts of likely treatments on predicted fisher habitat value at the stand level are integrated into an analysis of larger spatial (landscape) and temporal (decades) scope would be one logical next step for this research.

Finally, although our results demonstrate significant effects of fire and fire surrogate treatments on fisher habitat, it is important to recognize that the reduction in estimated habitat quality does not necessarily mean the habitat has become unsuitable for resting or foraging fishers. It is possible these habitats have not lost functional suitability to fishers, but without complementary studies examining response of individual fishers and / or fisher populations to vegetation management it is impossible to determine the potential impacts on fisher populations. In the absence of such studies, the conservative assumption should be that such habitat manipulations result in short-term reductions in habitat quality. It is likely safe to assume, however, that the relative impacts of vegetation management projects designed to reduce fuel loads and mimic the natural role of fire are considerably less than large-scale catastrophic fires.

Land managers faced with balancing the challenges of maintaining habitat for fisher, protecting mountain communities, 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 associated with most vegetation management projects, 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. Whenever possible, early burns should be timed to follow the fisher denning period (mid-March through mid-May) to minimize the likelihood of disturbing denning female fishers. 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). Third, whenever possible, managers should plan vegetation management activities in a manner that disperses treatments over space and time to minimize impact on individual fishers. Lastly, managers must be willing to commit to long-term monitoring efforts to better understand the impacts of vegetation management activities on fisher and other wildlife. Monitoring should include both a habitat component, such as the approach described herein, as well as a population monitoring component. Only with such a commitment can we begin to better address the uncertainties inherent in complex land management issues and make decisions less shrouded in uncertainty.

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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).

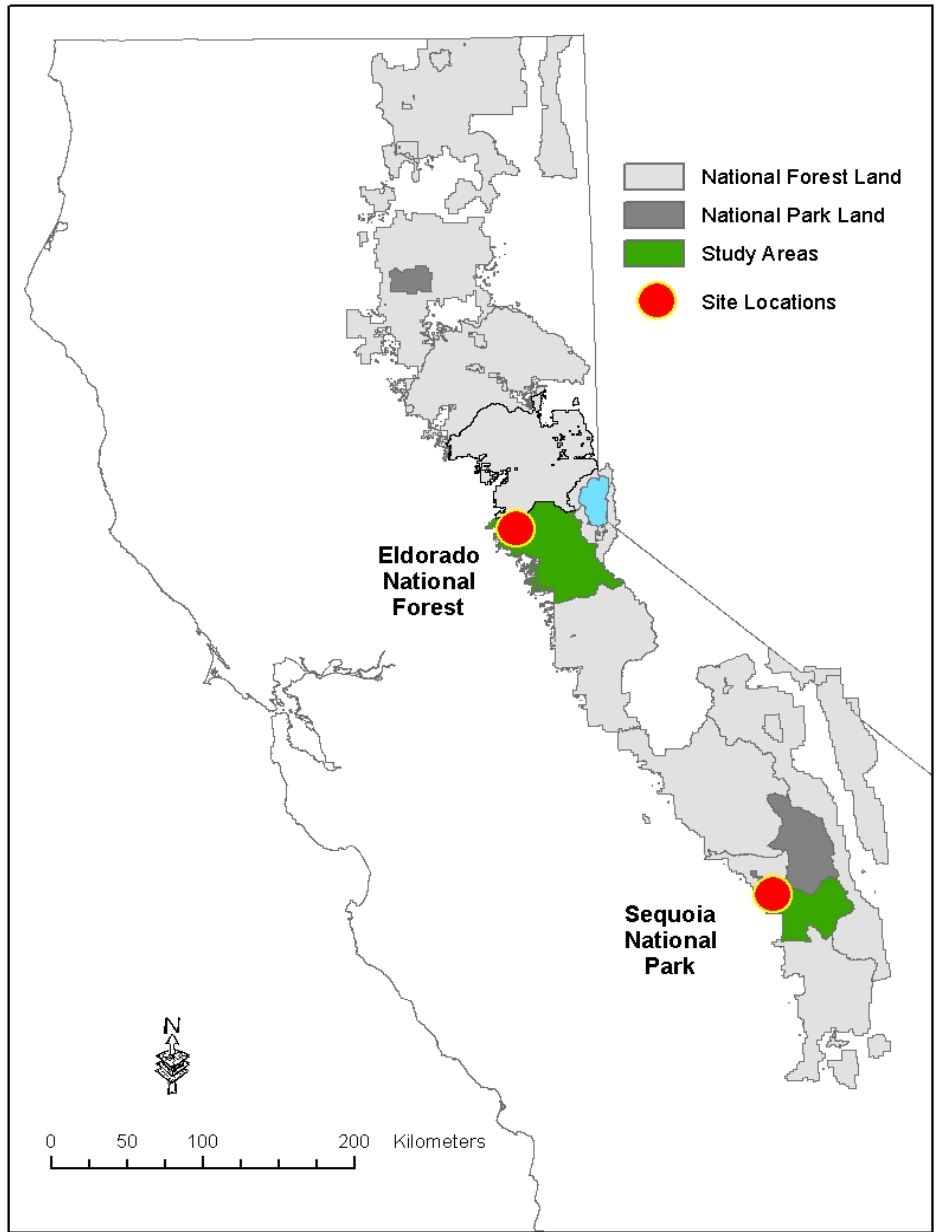| Variable	Acronym	Measurement technique / definition
Percent Slope	SLOPE	Clinometer; average of uphill and downhill readings
Presence of water within 100 m	WATER	Visual estimate
Basal area hardwoods	BAHDW	20-factor prism; m^2/ha
Average dbh	DBHAVE	Mean dbh (cm) of trees in the prism sample
Average hardwood 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 ≥1 conifer snag >102 cm dbh in the prism sample
Average canopy closure	CANAVE	Mean of densitometer readings at 5 plot locations

Table 2. Approximate fisher habitat sampling and treatment implementation schedule for Fire and Fire Surrogate sites at Blodgett Forest Research Station (BFRS) (El Dorado County, California) and Sequoia-Kings Canyon National Park (SEKI) (Tulare County, California).

		Habitat Sampling		
FFS Site	Treatment	Pre-treatment	Post-treatment	Treatment Implementation
BFRS	Mechanical	June 2001	September 2002	August 2001
	Mechanical + fire	June 2001	September 2002	August 2001
	Fire	August 2002	September 2003	September 2002
	Control	August 2002	September 2003	n/a
SEKI	Early burn	August 2001	August 2003	June 2002
	Late burn	August 2001	August 2002	September 2002
	Control	August 2002	September 2003	n/a

Table 3. Summary statistics for average 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 (6.2)	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)	4496.62 (1596.6)	0.12 (0.01)
	Post-treatment	Control	91.6 (1.3)	26.6 (6.2)	88.1 (3.1)	1949.2 (329.1)	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)	5787.4 (1304.2)	0.16 (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 (968.2)	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)

Legend:

- National Forest Land
- National Park Land
- Study Areas
- Site Locations

Eldorado National Forest

Sequoia National Park
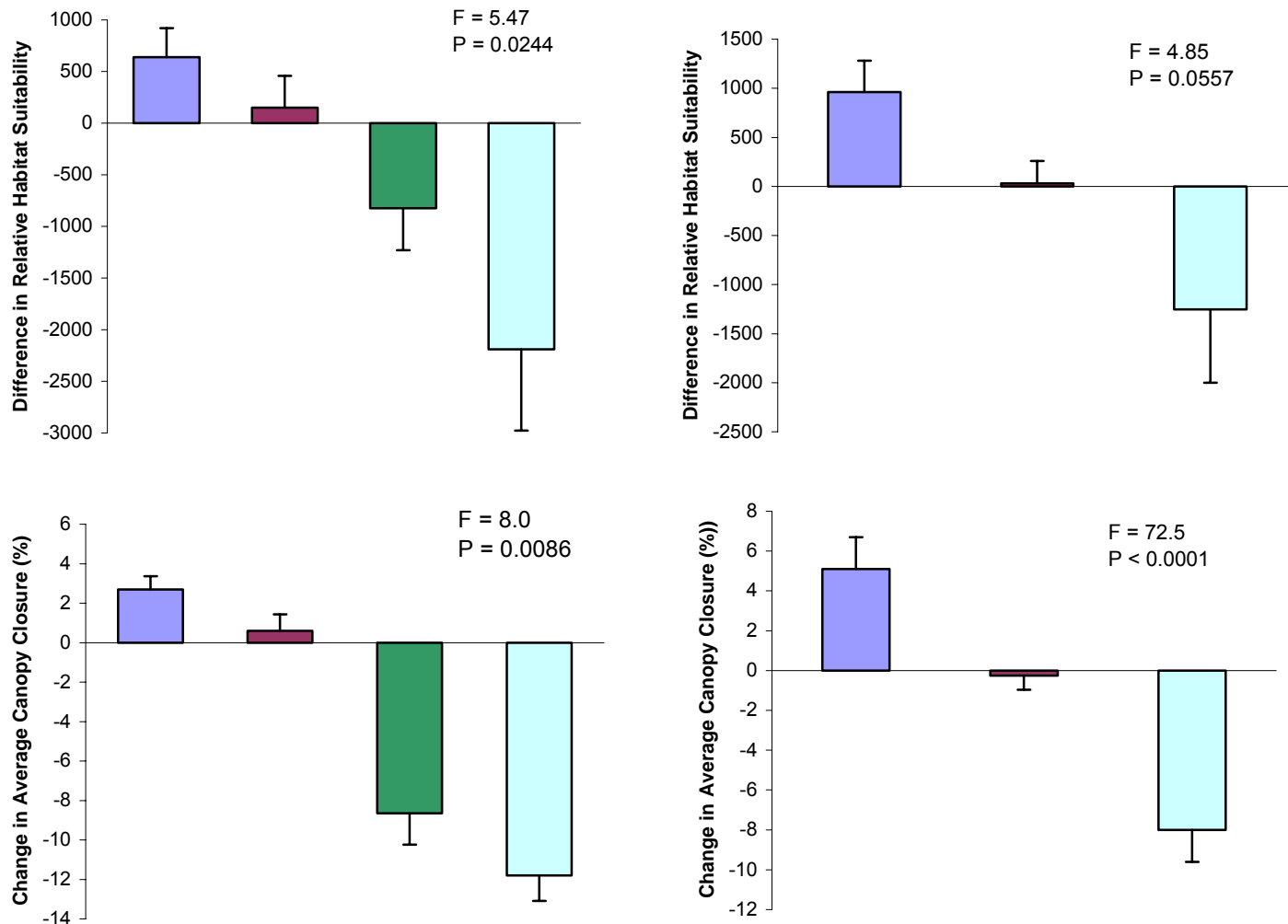
N

| 0 | 50 | 100 | | 200 | Kilometers |

Figure 2. Treatment means (SE) for relative predicted probability of resting fisher habitat suitability and average canopy closure at Fire and Fire Surrogate (FFS) study areas at (A) Sequoia-Kings Canyon National Park and (B) Blodgett Forest Research Station, California. For each variable, negative values indicate reduction due to FFS treatments. Treatment types are: C = control, F = Fire Only, M = mechanical only, MF = Mechanical plus fire, EB = early burn and LB = late burn.