SELECTION OF DISTURBED HABITAT BY FISHERS (*MARTES PENNANTI*) IN THE SIERRA NATIONAL FOREST

by

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Southern Sierra fisher (*Martes pennanti*) populations currently pose a management challenge for forest managers whose goals of forest fuels reduction often conflict with elements of fisher habitat conservation. My research draws upon Sierra National Forest management records and data collected by the Kings River Fisher Project to investigate the long-term effects of past management actions on fisher habitat. I used location data for 36 individual fishers (27 female, 9 male) to study second-order (home range and core-use area) and third-order habitat selection (resting and foraging sites) of national forest lands treated with management activities between 1992 and 2006. To better understand the possible drivers of fisher selection behavior I used light detection and ranging (LiDAR) data to compare treated and untreated forest structural characteristics.

My findings indicate that fisher home ranges tend to include larger proportions of treated areas than are found on the landscape as a whole. In contrast, when selecting microsites within their home ranges, fishers tend to avoid using sites within 200 meters of a treated area. A possible explanation for the conflicting selection found here between home range and microsite selection is that the treated areas are generally small compared to a fisher's home range, and relatively dispersed on the landscape, allowing fishers to avoid the treated areas while still using the untreated areas surrounding them.

Analyses of the LiDAR data indicate that forest areas that have been treated in the past continue to have reduced canopy cover and three-dimensional complexity, both primary indicators of fisher habitat quality, relative to untreated forest. These impacts do not appear to have rendered the habitat wholly unsuitable, however, and may be offset by increased fire resiliency. My findings suggest fishers may tolerate such fuels reduction treatments provided they focus on the reduction of surface and ladder fuels, and care is taken to maintain both canopy cover and sufficient abundance of forest structures, such as large diameter defective and standing dead trees, most likely to provide suitable rest and den sites.

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INTRODUCTION

Fishers (*Martes pennanti*) are mid-sized arboreal carnivores, of the family Mustelidae, native to North America. While often larger in other parts of their range, female fishers in the southern Sierra Nevada grow to 1.5 to 2.5 kilograms and males to 2.5 to 5 kilograms. Fishers are territorial animals that are nomadic within individual home ranges. Relative to their small body size, their home range requirements are large (Buskirk 1992), and estimates of average home range size in the Sierra vary between 527 hectares (ha) and 998 ha for females, and between 2150 ha and 4096 ha for males (Boroski et al. 2002, Zielinski et al. 2004a). As suggested by sexually dimorphic home range sizes, the much larger male home ranges typically overlap several female home ranges.

Fishers survive primarily in mature coniferous and mixed coniferous/deciduous forests where they rely on dense canopy cover and complex three-dimensional structure created by large woody debris, branch clusters, snags, broken tops, and cavities for foraging, resting, denning, and predator evasion (Allen 1983). Past research has demonstrated a preference for large-diameter, and often defective, conifers and hardwoods along steep slopes and riparian areas (Aubry and Houston 1992, Truex et al. 1998, Mazzoni 2002, Zielinski et al. 2004a, Zielinski et al. 2004b, Zielinski et al. 2006, Purcell et al. 2009). Research suggests that complex three-dimensional forest structure and suitably dense groundcover provides a more accurate characterization of suitable habitat than any specific forest type (Buskirk and Powell 1994).

The current range of fisher in the United States depicts an arc extending in the east down along the Appalachian Mountains south into West Virginia, in a thin band north along the forested border with Canada, and in isolated subpopulations along the mountains of the Pacific coastal states as far south as the southern Sierra Nevada (Allen 1983). A combination of habitat loss due to logging and human encroachment, loss of a primary prey species via porcupine (*Erethizon dorsatum*) extirpation, and the impact of overharvesting for pelts had eliminated their presence by the late 1940s in Washington and pressed the remaining western population into three small isolated sub-populations in southern Oregon and California (Allen 1983, Powell and Zielinski 1994). Within California a northern sub-population extends from the Coast Ranges through the Klamath Mountains, while a southern sub-population inhabits the western slopes of the Sierra Nevada south of Yosemite National Park. The two populations are separated by approximately 420 kilometers, far beyond the possible range of dispersing individuals (Zielinski et al. 1995, Zielinski et al. 2005). Recent research into the genetic differentiation of California fisher populations suggests that the isolation of the southern Sierra population is a product of glacial separation during the last ice age rather than anthropogenic activities (Tucker et al. 2012), though the same genetic evidence also indicates a more recent population bottleneck coinciding with an era of extensive human settlement and resource extraction 1848 - ca.1950 (Beesley 1996). Consequently the isolated southern subpopulation is currently estimated to consist of as few as 160 individuals and likely no more than 360 (Spencer et al. 2008), placing them at imminent risk of extinction (Lamberson et al. 2000). The isolation and vulnerability of the California fisher sub-populations led the U.S. Fish and Wildlife Service to classify West Coast fisher populations as "warranted but precluded by higher priority actions" under the Endangered Species Act (USDI Fish and Wildlife Service 2004), and later this year the California Fish & Game Commission is again scheduled review the status of California fisher populations (California Department of Fish and Wildlife 2013). Recognition of the plight of western fisher populations has led to increased attention from the research community over the past decade; however, the dilemma of developing an effective conservation strategy remains. The difficulty is that the complex, dense three dimensional forest structure that fishers rely on tend to be the very forest conditions that create a high risk of wildfire (Zielinski et al. 1995, Zielinski 2004a, Zielinski et al. 2004b, Truex and Zielinski 2013, Zielinski et al. 2005, Jordan 2007, Spencer et al. 2008, Purcell et al. 2009, Scheller et al. 2011). This leaves forest managers facing a conflict where, on the one hand, aggressive fuels reduction could create a fire-resilient forest, but likely extirpate fishers for lack of suitable habitat; and on the other hand, lack of action could maintain current high quality habitat, only to risk its total loss in the event of a catastrophic fire (North et al. 2009, Figure 1). Past work (Truex and Zielinski 2013) reported that in the short term (within one year) fire and fire surrogate treatments significantly reduced the availability and quality of large-diameter, dense canopy forest associated with preferred fisher resting habitat. Truex and Zielinski (2013) also assert that while fuels reduction actions will likely have a negative impact on fisher habitat, if

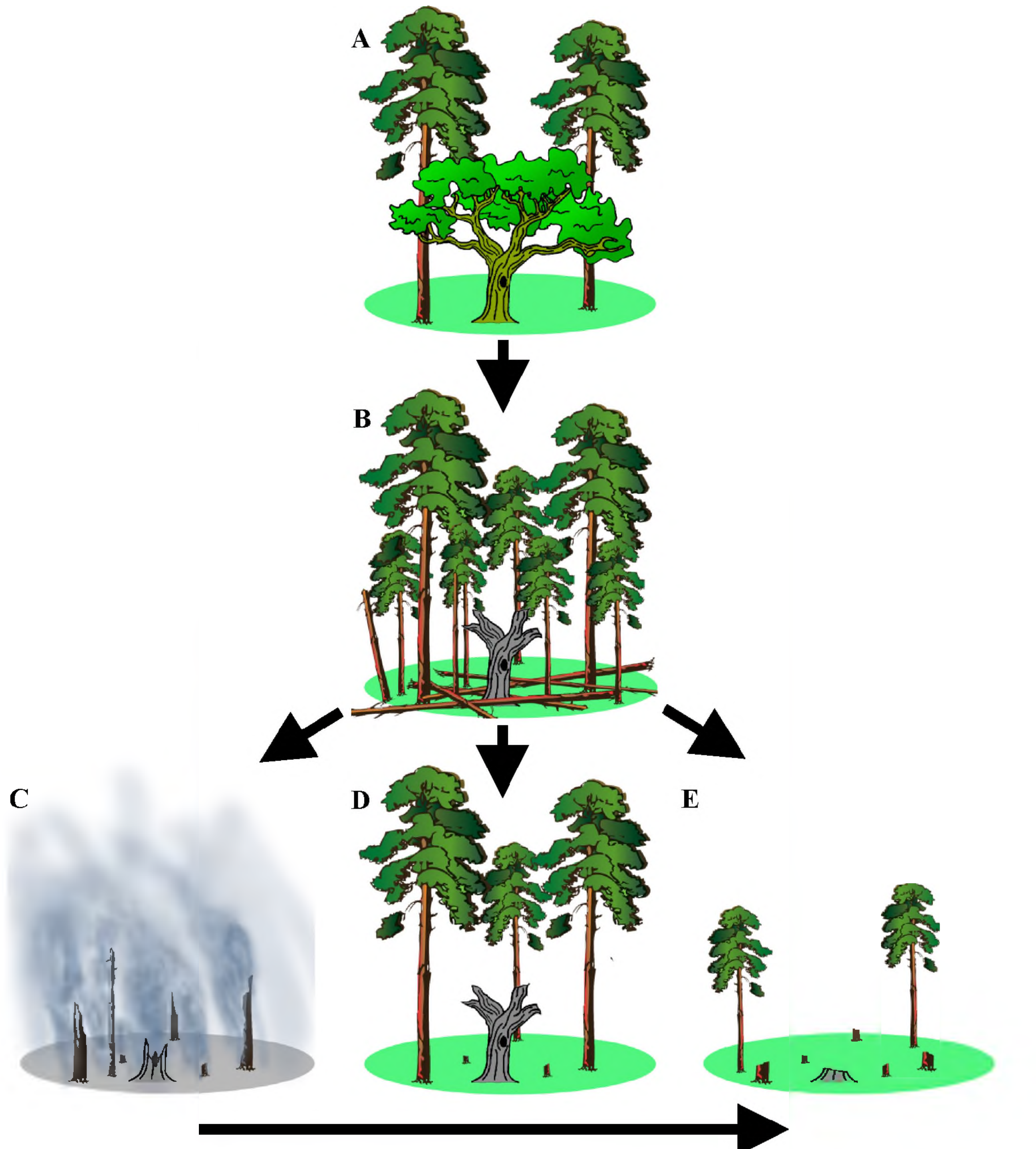

Figure 1. Historically fire resilient fisher habitat (A) has, through fire exclusion, become overburdened with surface and ladder fuels (B), increasing habitat quality, but reducing fire resiliency. Inaction risks loss of habitat through catastrophic fire (C), while overly aggressive action risks rendering the habitat unsuitable (E). Appropriate action (D) must strike a balance between maintaining habitat quality and reducing fuel loading.

properly applied, those impacts will not necessarily render the habitat unsuitable and are likely offset by the beneficial long-term decreases in potential fire severity.

In a similar study, Scheller et al. (2011) combined dynamic forest disturbance, fisher habitat, and fisher metapopulation models to simulate the effects of various fuels treatment and wildfire scenarios. Supporting the findings of Truex and Zielinski (2013), Scheller et al. (2011) assert that while fuels treatments do have direct negative effects on habitat quality, these effects were generally less severe than the negative effects on habitat caused by wildfire in the absence of treatments.

In 1993, with the implementation of the Kings River Sustainable Forest Ecosystems Project (KRSFEP, McCandliss 2002), the Sierra National Forest began to work toward reducing forest fuel loads and restoring fire regimes. The KRSFEP comprised landscape-level small-group selection harvests and prescribed burns aimed at reducing fuel loads while maintaining continuous forest cover and wildlife habitat by retaining older trees, large snags, and large logs (Phillips 2002). In 2002 the KRSFEP evolved into the Kings River Project (KRP), which continued the basic principles of its predecessor, employing small-group selection and uneven aged management to restore the forest to historical conditions of increased heterogeneity and fire resiliency (Rojas et al. 2006). The KRP plan was fraught with disagreement however, with many feeling it was too severe and lacked sufficient protection for wildlife habitat (Lockyer 2006, Barrett 2007, Zielinski 2007) and it was eventually abandoned in 2007 pending litigation.

Efforts were resurrected in 2009 by the Dinkey Landscape Restoration Project

(DLRP). The DLRP is a collaborative effort among forest managers, researchers, and community activists that, guided by geomorphology, fire behavior, and climate prediction models, aspires to produce a forest resilient to future climate conditions while maintaining wildlife habitat (North et al. 2009, Dinkey Collaborative Group 2010). The DLRP proposes to achieve these goals by 2019 primarily through selective tree harvests aimed at restoring forest resilience by emulating historic conditions (Dinkey Collaborative Group 2010).

To document the impact of potential forest management actions on the Dinkey Creek area fisher population, the USDA Forest Service Pacific Southwest Research Station initiated the Kings River Fisher Project (KRFP) in 2007. From February 2007 through February 2012 the KRFP collected behavioral and habitat use data on 85 radio-collared fishers through live trapping, radio-telemetry tracking, and scat detection surveys.

My research makes use of data collected by the KRFP to build upon that project's goals, by investigating fisher habitat selection relative to the ongoing effects of 15 years of management activities prior to this study (1992-2006). The objectives of this study were: (1) to quantify the degree and scale of potential second- and third-order (Johnson 1980) avoidance of historically treated forest by fishers in the study area by comparing KRFP radio-telemetry and rest site locations to a management history spatial dataset (FACTS, USDA Forest Service 2012); and (2) to identify lingering structural differences within treated forest relative to untreated forest using light detection and ranging

(LiDAR) data, collected over a portion of the study area in 2010. Taken together, these two analyses provide useful information about the long-term effects of management activities on both the selection and structure of fisher habitat in the Sierra Nevada.

STUDY SITE

The study area consists of approximately 91,000 hectares of high-slope forest in the Upper Kings River watershed in Fresno County, California (37.05° N, 119.19° W, Figure 2). This area is between Shaver Lake and Wishon Reservoir in the southern portion of the Sierra National Forest. It is roughly bordered by Pine Flat Reservoir to the southwest and the Dinkey Creek Ranger Station to the northeast. The landscape lies between 600 and 2000 meters elevation and is dominated by ponderosa pine (*Pinus ponderosa*), sugar pine (*Pinus lambertiana*), white fir (*Abies concolor*), and incense-cedar (*Calocedrus decurrens*), with California black oak (*Quercus kelloggii*) occurring with increasing frequency at lower elevations. Below about 900 meters, the forest transitions into mixed chaparral consisting of greenleaf manzanita (*Arctostaphylos patula*) and whitethorn (*Ceanothus cordulatus*), with scattered California black oak and interior live oak (*Quercus wislizeni*). Large granite outcroppings, boulders, and wet meadows typical of the southern Sierra Nevada occur commonly throughout the entire study area.

Approximately two-thirds of the study area is within the management boundaries of the Sierra National Forest. The U.S.D.A. Forest Service (USFS) manages this forest for recreational use, timber resources, and habitat conservation. Fuels management is accomplished primarily through low-intensity under-burning and selective thinning. Of the remaining third, approximately one half is owned by Southern California Edison, and managed for fuels reduction and forest products, and the other half consists primarily of

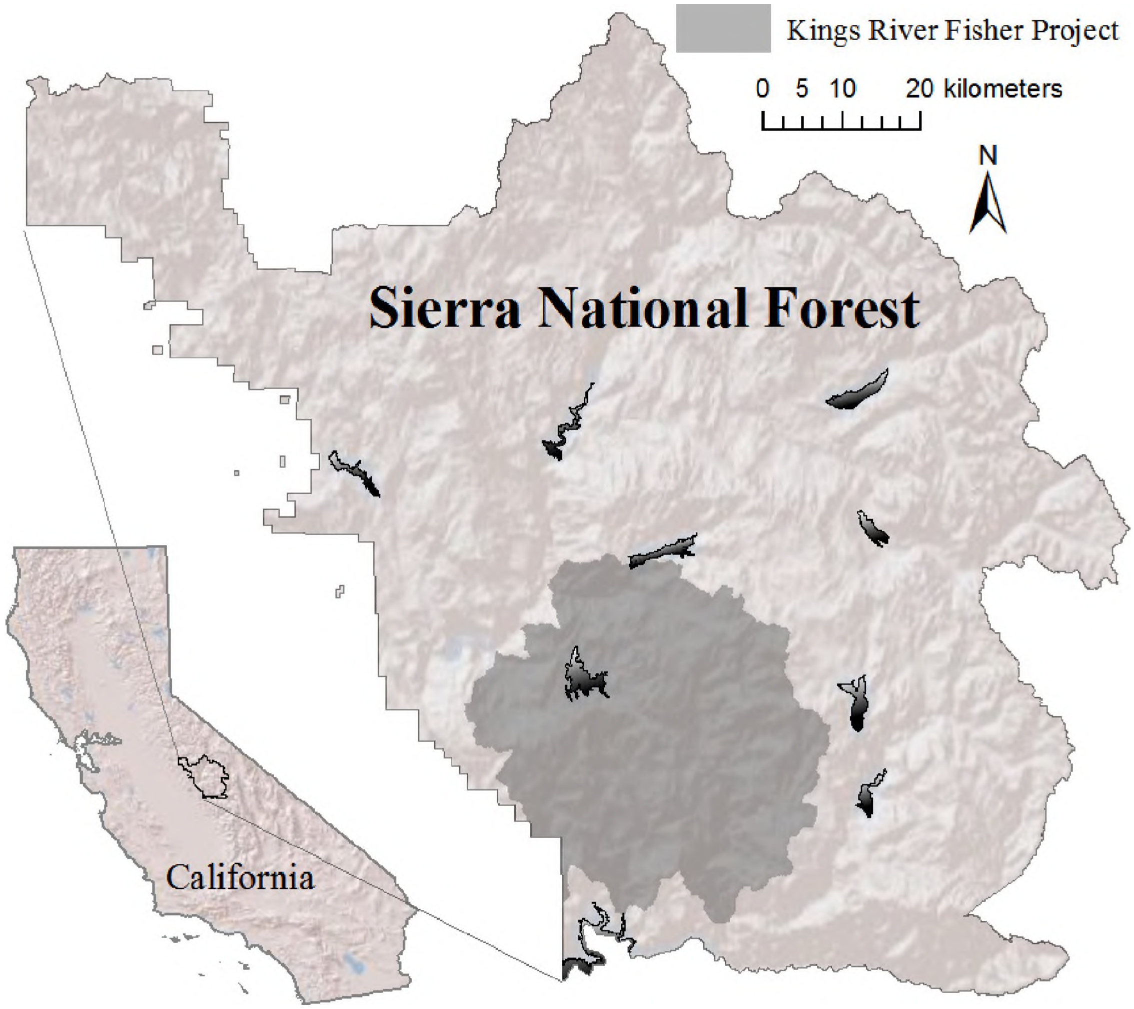

Figure 2. Location of the study site within the Sierra National Forest. The Kings River Fisher Project study area is located within the shaded southwest corner.

homes, camps, summer cabins, and the small community of Shaver Lake.

Historically the area is characterized by a high-frequency / low-intensity fire regime, with an average fire return interval of two to twelve years for low-intensity fires (Phillips 2002) and 50 years for high severity fires (Minnich et al. 2002). Due to widespread fire exclusion, much of the area has not burned in more than 100 years (McCandliss 2002). This exclusion of fire has caused what historically had been open forest dominated by scattered large trees to shift structurally to dense stands of smaller trees with an abundance of surface fuels (McKelvey and Johnston 1992, Purcell et al. 2009).

METHODS

Datum and Projection

All spatial data were compiled, analyzed, projected, and presented using ESRI's ArcGIS 10 (Environmental Systems Research Institute, 2011) software package. All data were projected into the North American Datum of 1927 UTM zone 11 North before processing, to match the projection of the KRFP radio telemetry location data.

Habitat Selection Analysis

To identify areas of forest historically impacted by management activities I used data from the Forest Service Activity Tracking System (FACTS, USDA Forest Service 2012). FACTS is a geospatial database that links polygon layers delineating activity boundaries with attribute data describing the management activity undertaken as well as the time period in which it was applied. Geospatial data exist within FACTS detailing management actions planned and carried out within the Sierra National Forest circa 1900 to the present.

While FACTS contains a detailed history of forest activity, the spatial precision of the data is questionable in some places. Until recently, the polygon areas recorded in FACTS described the total area included in the respective treatment plan without distinction between areas that received the treatment and areas that did not. In this way, portions of planned management actions that were not completed, due to inaccessibility or for other reasons, are indistinguishable from completed portions. In spite of this deficiency, in many areas, as in my study area, FACTS data constitutes the best available record of management actions and continues to inform forest management (Vogelmann et al. 2011) and research efforts (Weisz et al. 2009).

The 15 years preceding the initiation of KRFP (1992 through 2006) encompass an era of forest management characterized by an increasing focus on identifying strategies to reduce fuels loads while maintaining habitat quality. To document the impact this era has had on fisher habitat use I used Structured Query Language (SQL) to query from FACTS all management activities between 01 January 1992 and 31 December 2006 (Figures 3 and 4).

Many fisher locations and portions of home ranges occurred within private land; however, data were not available either for management history or structural composition of forests contain within the embedded private land within the study area. For the purpose of habitat selection characterization, private land portions of fisher home ranges were used as a reference habitat category in the development of selection ratios (see Second-Order Selection and Third-Order Selection sections below). In order to distinguish federally managed forest from privately owned land I obtained land ownership data from the California Department of Forestry and Fire Protection (CAL FIRE 2011, Figure 3).

Previous research (Davis et al. 2007) indicates fisher habitat quality varies with forest type as defined by the California Wildlife Habitat Relationship classifications (CWHR, Mayer and Laudenslayer 1988). To minimize the confounding effect of forest

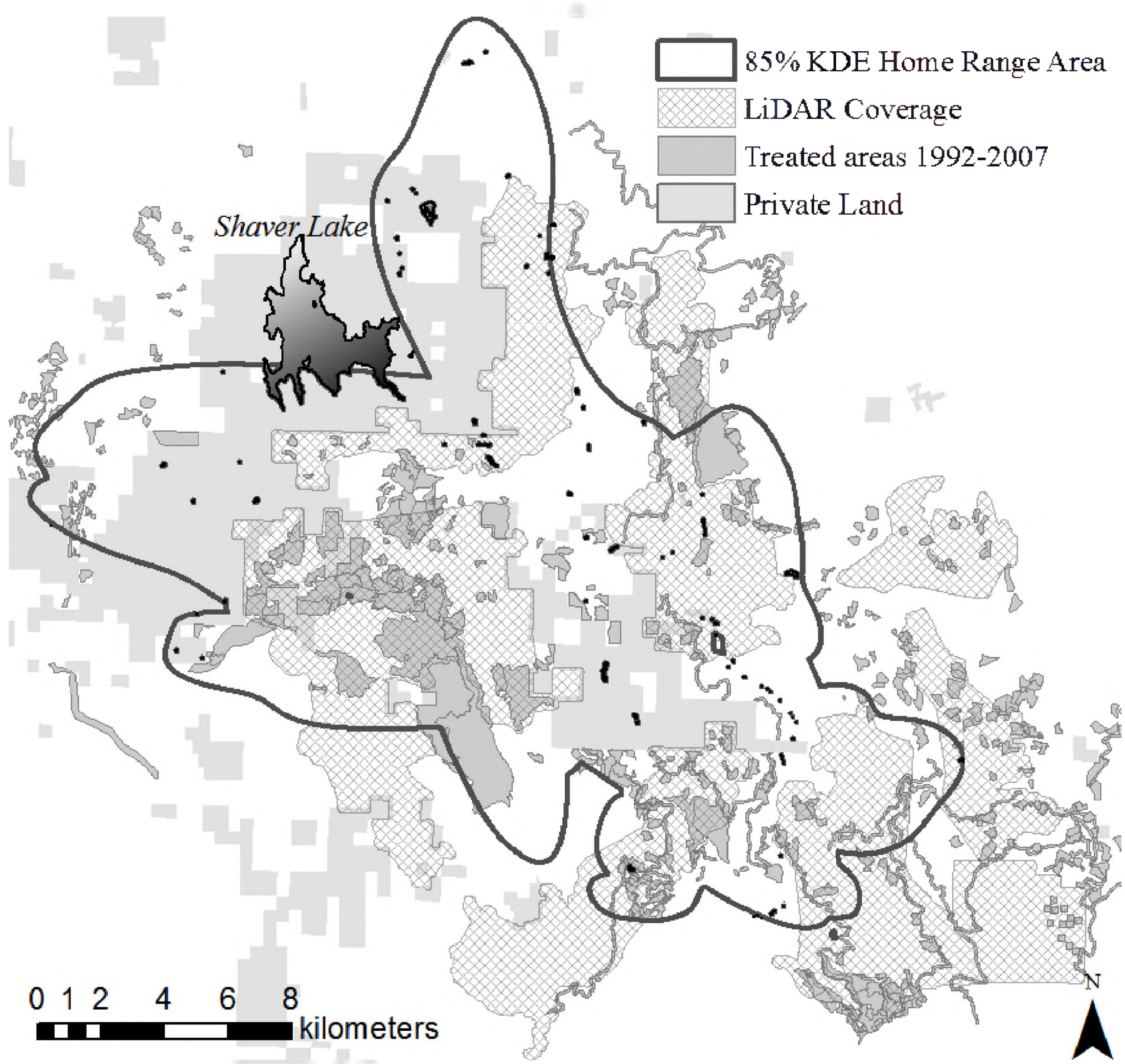

Figure 3. This map illustrates the spatial intersection of the primary datasets. The area within the combined 85% KDE home range estimate in the heavy gray outline. Areas of national forest that received management action between 1992 and 2006 are in solid green. Private land is indicated by the light gray shaded areas. LiDAR coverage areas are indicated by the gray crosshatched polygons.

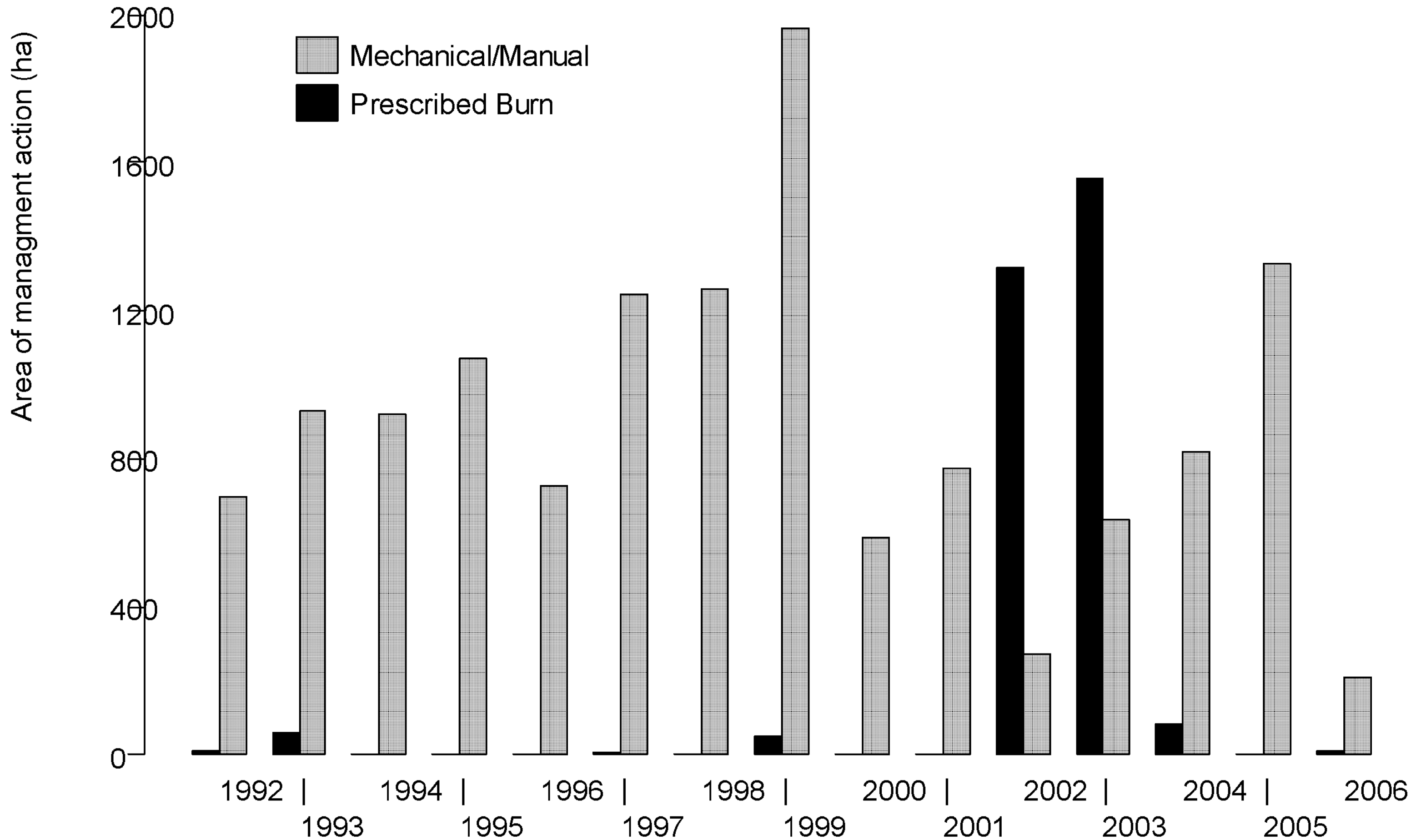

Figure 4. Total area of management activities (hectares) accomplished, from 1992 through 2006 by year and type within the combined area encompassed by the 36 fisher home range estimates considered in this study (85% kernel density estimates).

type on fisher habitat use, I used the USFS Existing Vegetation dataset (tiles EvegTile36A_01_07_24k_v2 and EvegTile36B_01_07_24k_v2; USDA Forest Service, Pacific Southwest Region , Remote Sensing Lab 2012) to identify CWHR forest type composition for each sample and compared selection of treated and untreated forest both in general and within each of the four most abundant forest types.

Telemetry Data

From February, 2007 through February, 2012 the KRFP captured and fitted 85 fishers (46 females, 39 males) with radio-collars in the Dinkey Creek study area. Utilizing traditional land-based radio telemetry triangulation techniques (Millspaugh and Marzluff 2001), technicians tracked these individuals, eventually recording a combined total of 24,540 bearings. Bearings were recorded almost exclusively during daylight hours and included both active and stationary locations, assumed to be representative of foraging/traveling and resting behavior, respectively. In addition, 557 rest sites were identified in which a technician was able to follow a stationary collar signal to its source and subsequently identify the structure being used by the fisher. Use of these data was approved by the Humboldt State University Animal Care and Use Committee (protocol No. 11/12.E.1.A).

I used the maximum likelihood estimation algorithm of the Locate III radio-telemetry triangulation program (Nams 2006) to generate location estimates and error ellipses for all triangulations that included three or more bearings with a minimum separation of at least 20°, as outlined in the KRFP study plan (Thompson and Purcell 2007). I discarded as imprecise all locations with error ellipses greater than 100,000 square meters (Nams 1989). Location estimate accuracy was assessed, in accordance with the KRFP study plan (Thompson and Purcell 2007), as the Euclidian distance between rest sites and coincident triangulation location estimates that preceded the rest site by no more than 90 minutes. Average error distance was 97.1 meters (sd = 89.4 meters). This figure was conservatively rounded up to 100 meters and used for the minimum analysis radius in the third-order habitat selection analysis.

To maximize the independence of each telemetry location (Dunn and Gipson 1977), locations were filtered to one per 24 hour period per animal with preference given to locations with smaller error estimates (McNay et al. 1994), and consecutive locations were filtered to ensure they were a minimum of 100 meters apart, corresponding to the average telemetry location estimate error (Springer 1979).

This process resulted in 3086 validated triangulation locations for 81 of the 85 animals ($\bar{x}$ =39.7 locations minimum = 1 location, maximum = 160 locations). Each of the four omitted fishers did not have a single location that met the filter criteria explained above. When the retained locations were combined with the rest site locations, 36 of the 85 animals (27 females, 9 male) met the minimum of 30 independent locations ($\bar{x}$ =72.8 locations; range = 30 to 155 locations) recommended by Seaman et al. (1999) for estimating home ranges using a kernel density method.

Second Order Selection

Second-order selection refers to selection of the area of a home range within a landscape. Use is defined by the area of each animal's home range, and availability is defined in a similar manner across the landscape used by all included animals (Johnson 1980).

Home range and core-use areas were estimated using the fixed kernel density estimation method (KDE, van Winkle 1975, Worton 1989), which calculates the point density of fisher locations around every cell of a raster surface. The cell values of this raster surface then represent an estimate of the relative amount of time the fisher spent at that point over the duration of the study (van Winkle 1975). I used the KDE function of the Geospatial Modelling Environment (Beyer 2012) software platform to create KDE surfaces for each fisher. As suggested by Seaman et al. (1999), the least-squares cross validation method was used to select bandwidth smoothing parameters. To avoid including bodies of water within the home range estimates, I eliminated them from the each KDE (Knight et al. 2009) using the National Hydrography Dataset (U.S. Geological Survey in cooperation with the U.S. Environmental Protection Agency 2009) high resolution (scale = 1:24,000) Waterbody Theme polygon layer. Isopleth polygons at 50% and 85% of the KDE surfaces were then generated corresponding to core-use (50%) and home range (85%) areas (Dickson and Beier 2002). Finally, each home range was divided into habitat classes based on ownership, forest type, and treatment status.

The question of how to define availability for home range or second-order selection studies can often be difficult and arbitrary (Johnson 1980). As fisher occurrence over the landscape is limited by elevation (Zielinski et al. 1997), I first constrained the extent of available habitat to the minimum (757 meters) and maximum (2555 meters) elevations found within the fisher location data using the National Elevation Dataset (U.S. Geological Survey, EROS Data Center 1999) 10 × 10 meter resolution digital elevation model.

Simply using the proportions of habitat types available on the landscape or within a subjectively defined study area may overestimate available habitat (Potvin et al. 2001). Furthermore, unless habitat types occur in small uniformly distributed areas across the landscape, the proportion occurring within the area of a home range will likely be different than the proportion of occurrence on the landscape (Wilson et al. 1998). Previous studies have avoided these issues by sampling landscape habitat availability using randomly generated home ranges similar in size to, and in equal or greater number than, the observed home ranges (Mladenoff et al 1995, Wilson et al. 1998, Potvin et al. 2001, Katnik and Wielgus 2005). As processing time grows substantially with increasing sample size, I settled on five randomly generated circular home ranges, within the stated elevation constraint, for each fisher with areas corresponding to their respective home range sizes to define second-order availability.

Third-Order Selection

Third-order selection refers to the selection of habitat within an animal's home range (Johnson 1980). Use and availability are defined individually for each fisher as telemetry locations within their home range, and extent of their home range respectively. I calculated availability by generating 100 random points within each fisher's 85% KDE home range. To identify the scale at which selection is occurring (Johnson 1980), each of the random points and telemetry relocations were then buffered at a range of radii between 100 and 1000 meters at 25 meter increments and proportions of each habitat type were then calculated within the buffered zones (Dyer et al. 2001).

Compositional Analysis

The scale of location error for the radio telemetry locations as well as irregularly shaped treatment areas creates uncertainty about the validity of using the ratio of point locations within habitat types as a measure of habitat selection in this study (Nams 1989). In many cases, management activities were conducted in narrow corridors along roads and were not wide enough to encompass telemetry location error. For this reason I used area-based compositional analysis (Aebischer et al. 1993, Manly et al. 2002) to identify selection (Figure 5).

Compositional analysis consists of calculating the difference of log-transformed habitat type proportions between used and available sample (Aebischer et al., 1993). In the most basic aspect of this study, there are two samples, used (o) and available (π), for each of three habitat categories, treated (t), untreated (u), and private (p), for any one fisher (i) where :

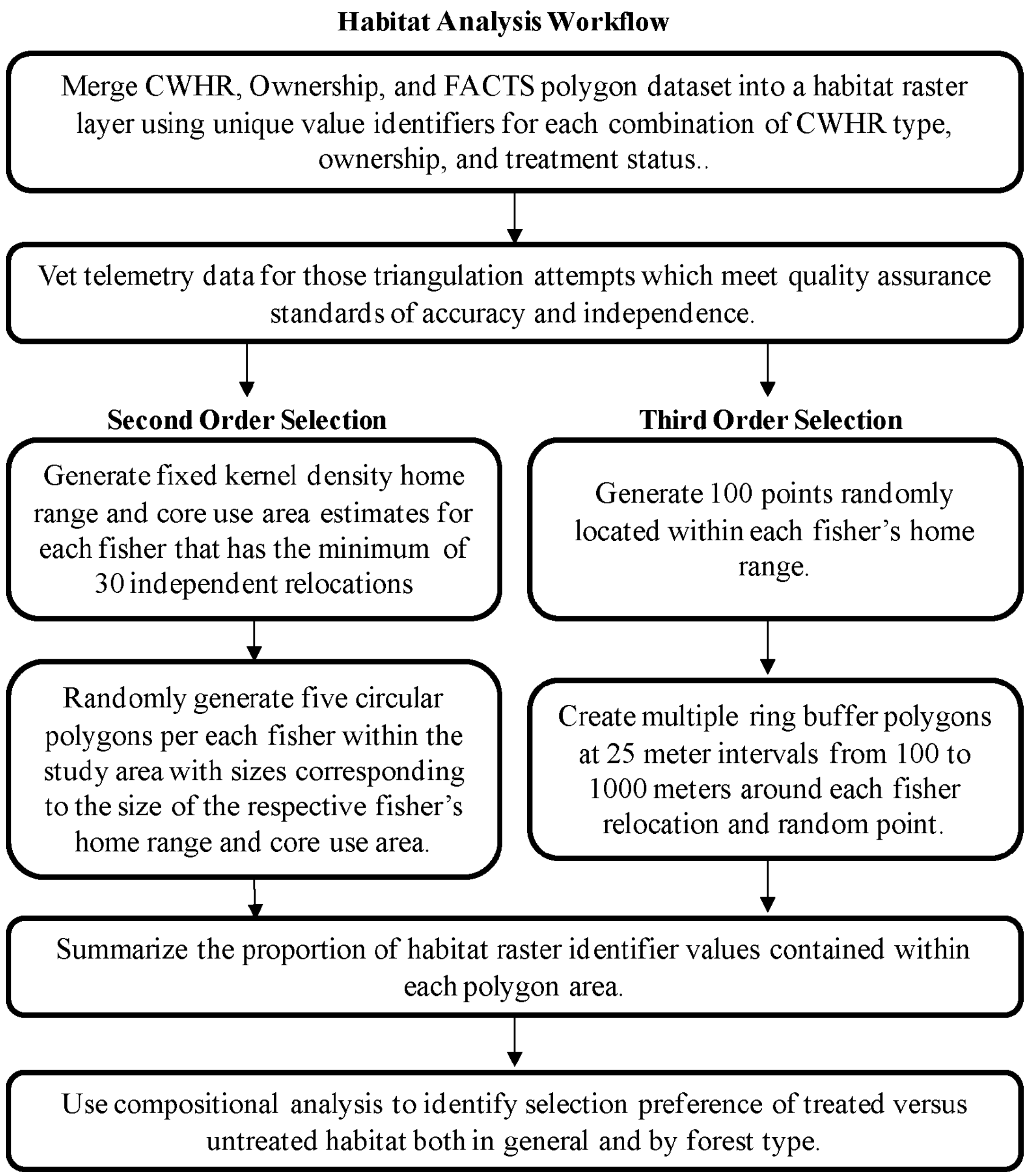

Figure 5. Flow chart describing fisher habitat selection analysis.

$$t_{oi} + u_{oi} + p_{oi} = 1$$

and

$$t_{\pi i} + u_{\pi i} + p_{\pi i} = 1$$

Compositional analysis quantifies the difference in the log-transformed ratio of a given habitat type to a reference category within the used and available portions of the sampled landscape (Mladenoff et al. 1995, Mladenoff et al. 1999, Dyer et al. 2001, Razgour et al. 2011). In this study I used private land as the reference category due to the lack of available management information. To avoid zero division errors in instances where a habitat type did not occur within a sample, the proportion for that type was set to 0.003 (Bingham and Brennan 2004). I then took the difference between the transformed used (o) and available (π) values to produce a test statistic (*d*) for each class for each animal. In this way a *d* score was calculated for treated areas for each fisheras follows:

$$d_{ti} = log(o_{ti}/o_{pi}) - log(\pi_{ti}/\pi_{pi})$$

and for untreated areas:

$$d_{ui} = log(o_{ui}/o_{pi}) - log(\pi_{ui}/\pi_{pi})$$

The *d* values for a given habitat type are then combined into a vector, and a t-test is used to assess significant difference from a null vector (Aebischer et al., 1993, Manly et al. 2002). Positive *d* score values represent selection for untreated areas (and avoidance of treated areas); negative values represent selection for treated areas. Compositional analysis requires an arbitrary reference habitat type for use as a common denominator in the ratio comparisons (Manly et al. 2002). As explained, analysis of private lands was not possible due to unavailable management history information and LiDAR data; for this reason, areas of private forest were utilized as the reference habitat. When analyzing treatment status by forest type, those forest types not being compared were pooled with private land as reference habitat. Not all of the forest type subcategories were available to all animals at the point scale (third-order) analysis. Selection comparisons in which either treated or untreated habitat are unavailable to a fisher provide no information (Pendelton et al. 1998). For this reason analysis among the forest type subcategories at the third-order was limited to those fishers whose corresponding random "availability" sample contained at least 1% by area of each treated and untreated habitat. Zero values of available reference habitat as well as each of the used (treated, untreated, and reference) habitat categories were replaced with 0.003 (Bingham and Brennan 2004).

Characterizing Impacted Habitat With LiDAR

Light detection and ranging (LiDAR) is an active optical remote sensing technology that uses a high resolution laser scanner typically mounted either on a stationary ground or mobile aerial platform to obtain extremely accurate distance measurements. LiDAR has proven effective in the measurement of forest structural characteristics including canopy height, density, basal area (Means et al. 2000). These measurements in turn have been used to generate characterizations of forest type (van Aardt et al. 2008) and successional class (Falkowski et al. 2009).

During October 2010, 23,867 hectares of the Dinkey Creek study area were scanned with LiDAR sensors by Watershed Sciences, Inc. (WSI). The area was divided into six non-contiguous subplots included in the Dinkey Creek Restoration Project (Figure 3). Flying at an altitude between 1100 and 1500 meters, the survey averaged 8.8 pulses per square meter with up to four measurements per pulse. The dual-mounted sensors scanned a 28° downward field of view (±14°from nadir on each side) and a 50% side lap, ensuring all areas were scanned from two angles eliminating areas that may have been shadowed by the first path and ensuring better canopy penetration and understory coverage. Using ground reference stations WSI was able to calibrate the point data to within 0.04 meters in any direction (Watershed Sciences Inc. 2011).

While LiDAR provides highly detailed data, care must be taken to analyze the data at a spatial resolution appropriate to the application. For example, measures of forest canopy density conducted at a 5 × 5 meter resolution tend to be strongly influenced by individual tree crowns and serve more as a test for the presence or absence of trees, whereas a larger resolution might provide a more ecologically relevant result (McGaughey 2012). Considering fisher mobility and home range size as well as the scale of the related data and processing ability, I used FUSION/LDV (McGaughey 2012) GridMetrics, DensityMetrics, and CanopyModel functions to produce a suite of metrics summarizingthe raw point data at a 30 × 30 meter resolution (Table 1). Canopy height, mean canopy height, and canopy cover (Jennings et al. 1996) were calculated due to their

Table 1. LiDAR variables used in comparison of historically treated and untreated forests.

Variable	Units	Description
Maximum canopy height	meters	Maximum height of LiDAR first returns above 2 meters
Mean canopy height	meters	Mean height of LiDAR first returns above 2 meters
Canopy cover	percent	Percentage of LiDAR returns above 2 meters
Canopy relief ratio	none	Canopy height (mean - minimum) / (maximum - minimum)
Canopy surface area ratio	none	Canopy surface area / ground surface area
Upper canopy density	percent	Percentage of all LiDAR returns > 10 meters
Understory density	percent	Percentage of LiDAR returns < 10 meters that are > 2 meters
Groundcover density	percent	Percentage of LiDAR returns < 2 meters that are > 0.5 meter
Inter-quartile range	meters	Difference between height of seventy-fifth and twenty-fifth percentile LiDAR returns
Absolute average distribution	meters	Mean difference in height of each LiDAR point return from mean height
Skewness	none	Degree to which vegetation density is vertically concentrated above or below the mean vegetation height

demonstrated ability to distinguish forest successional stages (Falkowski et al. 2009). Canopy surface area ratio (McGaughey 2012), and canopy relief ratio (Evans et al. 2009) were computed as indices of canopy structural complexity. Absolute average difference from mean, inter-quartile range, and skewness were also included as measures of the three dimensional complexity that has been linked with preferred fisher habitat (Buskirk and Powell 1994).

Three vegetation density strata corresponding to ground cover (height >= 0.5 and < 2.0 meters), mid-story (height >= 2.0 and < 10 meters), and canopy (height >= 10 meters) were included as they correspond well to both the vertical structure used by fishers (Buskirk and Powell 1994) and the ground, ladder, and canopy characteristics typically targeted in fuels reduction treatments (Agee and Skinner 2005). Stratified density is typically defined as the proportion of total LiDAR returns that fall within a minimum and maximum height break (Evans et al. 2009, McGaughey 2012). A LiDAR laser pulse however, does not penetrate through most objects; rather, once it intersects an object, it bounces back and nothing behind that object will be detected. This creates a shadow effect in which lower elevation strata density values are dependent on the strata above. Using the strata defined here as an example, in a case where 40% of the LiDAR pulse returns are from the canopy stratum, the remaining strata are constrained to a total of 60% of the return count. In a case where the canopy stratum accounts for 70% of all returns, the remaining strata are limited to a sum of just 30% of all returns. Here I define stratified density as the proportion of LiDAR returns below the stratum maximum height that are above the stratum minimum height.

A random set of sample points (n = 4325) were generated within the LiDAR covered area with a minimum distance of 100 meters between each. These points were then classified by CWHR forest type and treatment status, and the values of each of the LiDAR variables were collected at each sample point location.

Difference Tests

The value distributions of the LiDAR variables selected tend to be highly positively skewed and/or zero-inflated, violating the basic assumption of normality for parametric testing methods. To avoid the requirement of normality I used the non-parametric Mann–Whitney–Wilcoxon (MWW) U-test (Wilcox 1945, Mann and Whitney 1947) and randomization tests (Manly 2007) to compare LiDAR generated variables within treated areas and untreated areas both in general and within of each of the four forest types (Ponderosa Pine, Sierran Mixed Conifer, Montane Hardwood-Conifer, and Montane Hardwood, Figure 6).

The U-test is carried out by first ranking the data in ascending order. When ties occur, the average rank is assigned to each value. The U-test statistic is then calculated for each population using:

$$U_1 = n_1 n_2 + \frac{n_1(n_1+1)}{2} - R_1$$

and:

$$U_2 = n_1 n_2 + \frac{n_2(n_2+1)}{2} - R_2$$

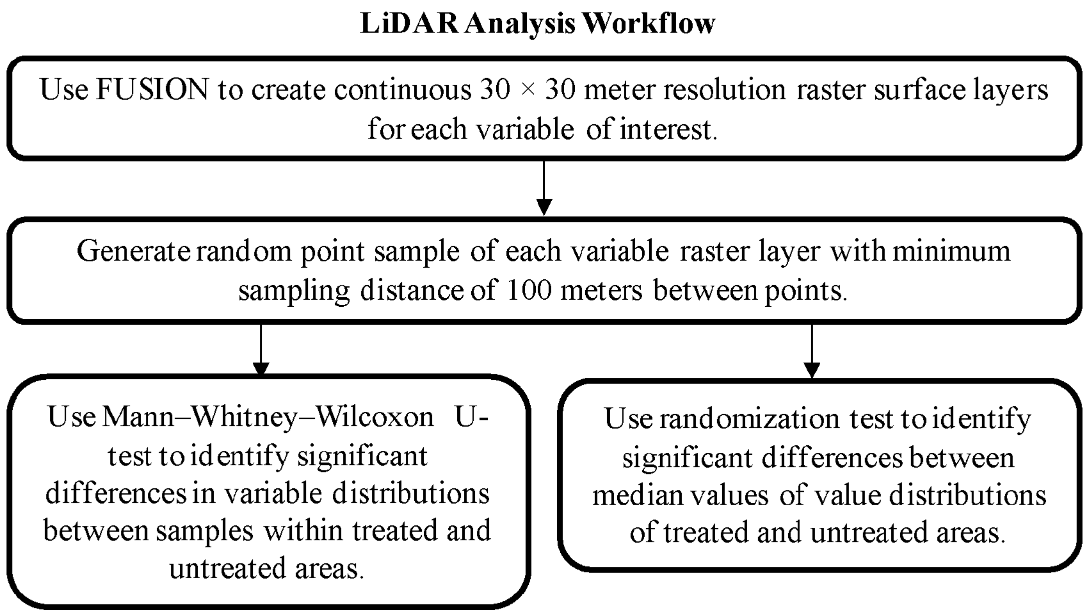

Figure 6. Flowchart describing habitat characterization analysis of LiDAR data.

where n_i is the number of values in the population and R_i is the sum of ranks for the population. These values are then compared to a critical value to test for significance.

Randomization tests (Manly 2007) are conducted by first calculating the difference between sample medians of each variable distribution from treated and untreated areas. The observed difference is then plotted against a distribution of median differences generated by repeatedly randomly assigning each variable to a treatment class. The null hypothesis in this case is that the observed values are likely to have been drawn at random from a single distribution of sample values, and that treatment class assignment is arbitrary. Significance is assessed based on the quantile value of the observed difference within the distribution of randomized differences (Manly 2007). Significance thresholds of $\alpha = 0.05$ for each habitat category set of 11 metrics were adjusted for multiple tests using a Bonferonni adjustment (Holm 1979).

RESULTS

Fisher Habitat Selection

Median fisher KDE home range areas of fishers in the Sierra National Forest were more than four times larger for males (median = 4096 ha, range = 1843 to 12667 ha) than for females (median = 998 ha, range = 551 to 3395 ha). Within the combined fisher home range area (Figure 3), the four most abundant forest type categories are Sierran Mixed Conifer (SMC, 45.7%), Ponderosa Pine (PPN, 18.3%), Montane Hardwood-Conifer (MHC, 11.9%), and Montane Hardwood (MHW 7.4%), which together make up 83.3% of the total area. These same forest types account for 93.4% of the treated forest area within the combined home range area (SMC = 27.1%, PPN = 34.4%, MHC = 18.6%, MHW = 13.3%, Table 2).

Among combined forest types, fishers show a slight preference toward inclusion of treated area within their 85% KDE home range (mean *d*-score difference = -0.33, p = 0.042, Figure 7). The same preference was found within the Sierran Mixed Conifer (mean = -0.51, p = 0.017) and Montane Hardwood (mean = -0.60, p = 0.025) forest type subcategories but was not found among the Ponderosa Pine (mean = -0.03, p = 0.848) or Montane Hardwood-Conifer (mean = -0.13, p = 0.499) forest types in which no significant preference was detected. At the core-use scale (50% KDE), wider

Table 2. Mean and coefficient of variation values for CWHR forest type composition of used and available fisher home range and core-use area samples. Bolded values indicate significant inclusion of treated forest within the respective area and forest type.

		Home Range		Core Use Area	
		Available	Used	Available	Used
	WHR Type	Mean (CV)	Mean (CV)	Mean (CV)	Mean (SD)
Untreated	All	59.4 (0.20)	62.1 (0.26)	58.3 (0.25)	62.4 (0.32)
	PPN	12.4 (0.44)	21.0 (0.58)	11.4 (0.56)	21.7 (0.74)
	SMC	19.4 (0.54)	25.9 (0.80)	26.7 (0.55)	28.9 (0.89)
	MHC	6.2 (0.31)	7.0 (0.56)	5.2 (0.44)	6.1 (0.72)
	MHW	6.1 (0.54)	3.0 (0.70)	3.7 (0.68)	2.2 (0.95)
	Other	15.4 (0.51)	5.2 (0.83)	11.3 (0.66)	3.5 (1.09)
Treated	All	12.2 (0.52)	17.9 (0.50)	16.0 (0.50)	19.3 (0.80)
	PPN	4.8 (0.75)	8.0 (0.80)	5.5 (0.76)	9.4 (1.19)
	SMC	2.4 (0.67)	4.8 (0.77)	4.1 (0.66)	5.3 (0.96)
	MHC	2.4 (0.75)	2.9 (0.93)	3.0 (0.80)	2.9 (1.28)
	MHW	1.6 (0.81)	1.6 (1.31)	2.3 (0.91)	1.3 (1.69)
	Other	0.9 (0.56)	0.7 (0.86)	1.1 (0.82)	0.4 (1.50)
Private	All	28.5 (0.44)	20.0 (0.88)	25.7 (0.51)	18.3 (1.11)
	PPN	2.7 (0.85)	4.1 (1.44)	3.8 (0.87)	4.9 (1.61)
	SMC	16.6 (0.46)	10.6 (1.03)	14.8 (0.60)	8.9 (1.30)
	MHC	4.0 (0.55)	3.1 (1.03)	3.6 (0.69)	2.8 (1.36)
	MHW	1.9 (0.68)	1.1 (1.45)	1.4 (0.86)	1.0 (1.90)
	Other	3.3 (0.52)	1.0 (1.30)	2.1 (0.76)	0.7 (2.00)

PPN = Ponderosa Pine
SMC = Sierran Mixed Conifer
MHC = Montane Hardwood-Conifer
MHW = Montane Hardwood.

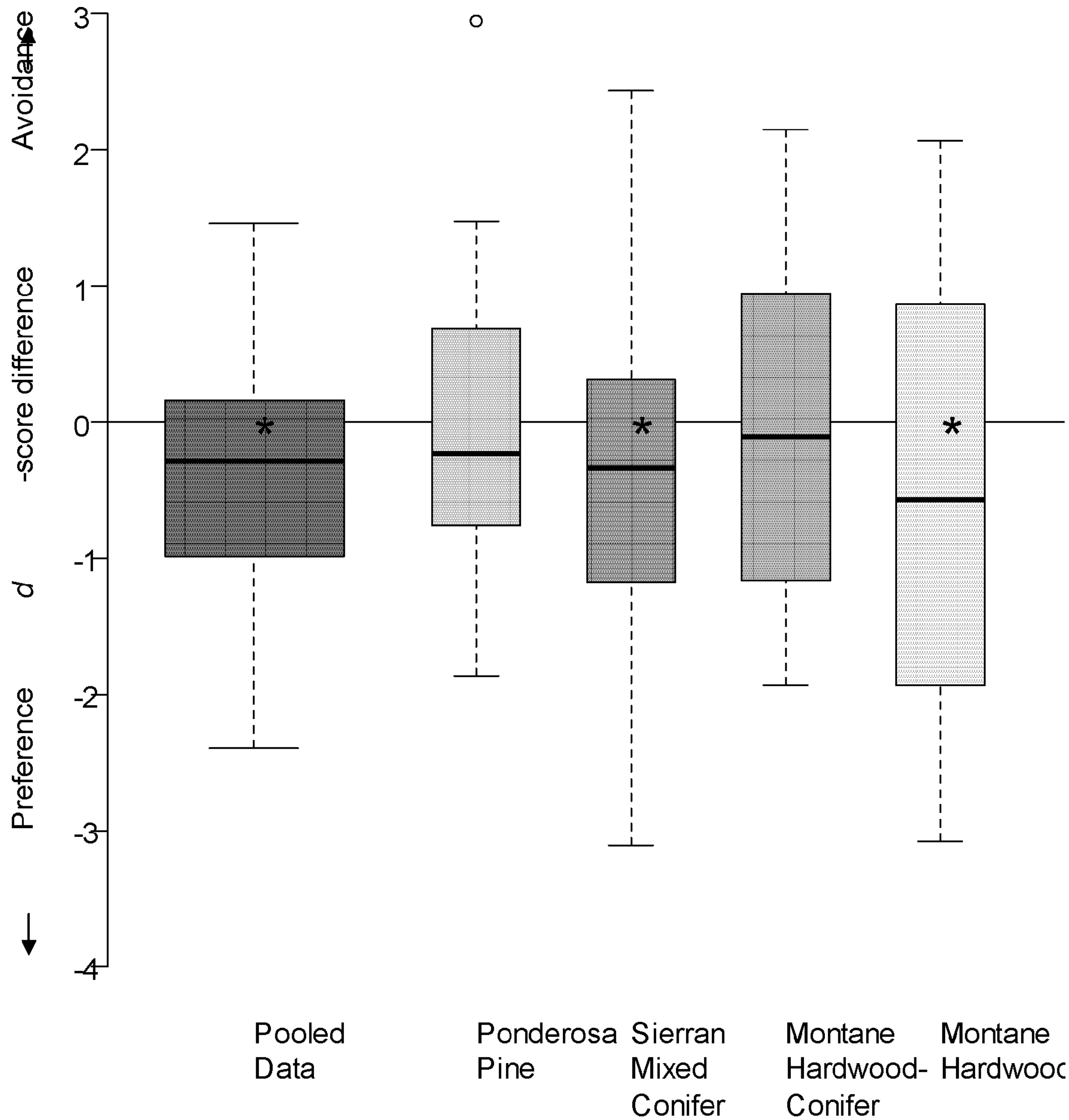

Figure 7. Preference and avoidance of treated forest area by fishers at the home range (85% KDE) scale based on compositional analysis *d*-score distributions. The vertical extent of each colored box represents the 25th (bottom) and 75th (top) percentile value. The bold horizontal line near the center of each box marks the median value, the whiskers mark the largest and smallest non-outlier values, and the dots represent outlier values (more than ±1.5 times the interquartile range). Positive values indicate avoidance of treated forest areas, while negative values indicate preference for treated areas. Asterix (*) indicate significant selection of treated versus untreated areas.

variation and an overall increase of the mean difference in selection scores relative to the home range scale led to a finding of no significant selection both in general and among each of the forest type subcategories (Figure 8).

Third-order selection analysis of the pooled data revealed that treated areas begin to influence fisher selection behavior at a distance of 200 meters (Figure 9), within which preference of untreated forest is significant and increases with proximity to fisher locations (mean *d*-Score at 200 meters = 0.206, p = 0.044). Among the four forest type subcategories, Ponderosa Pine (mean 0.35, p = 0.050, Table 3) and Sierran Mixed Conifer (mean = 0.55, p = 0.019, Table 3) both mirror the general increase in preference toward untreated habitat, though at a slightly closer range, with each significant within a 150 meter proximity of the fisher locations. Neither the Montane Hardwood-Conifer nor the Montane Hardwood (Table 3) forest type subcategories show evidence of third-order selection preference at any distance.

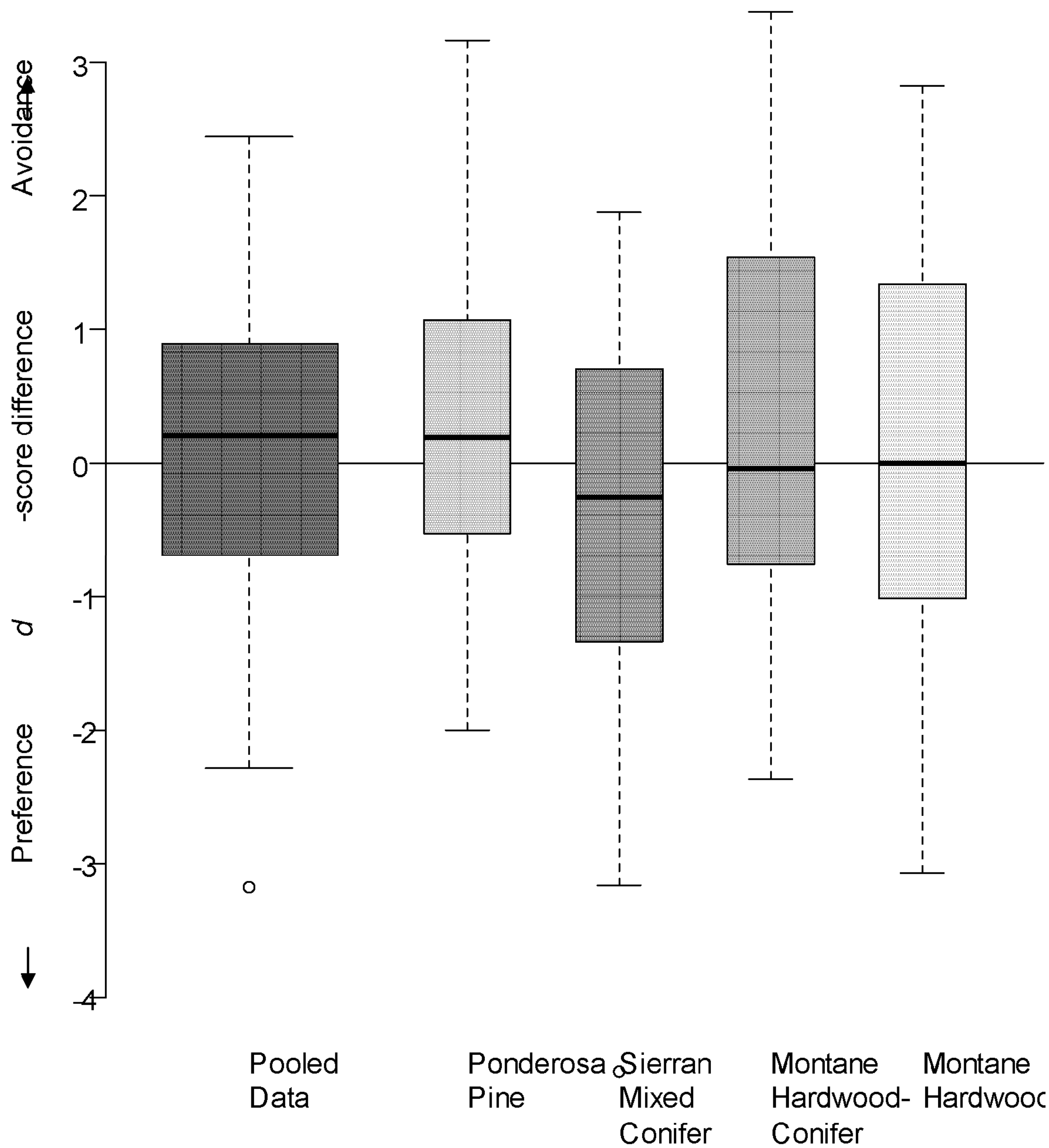

Figure 8. Preference and avoidance of treated forest area by fishers at the core-use (50% KDE) scale based on compositional analysis *d*-score distributions. The vertical extent of each colored box represents the 25th (bottom) and 75th (top) percentile value. The bold horizontal line near the center of each box marks the median value, the whiskers mark the largest and smallest non-outlier values, and the dots represent outlier values (more than ±1.5 times the interquartile range). Positive values indicate avoidance of treated forest areas, while negative values indicate preference for treated areas.

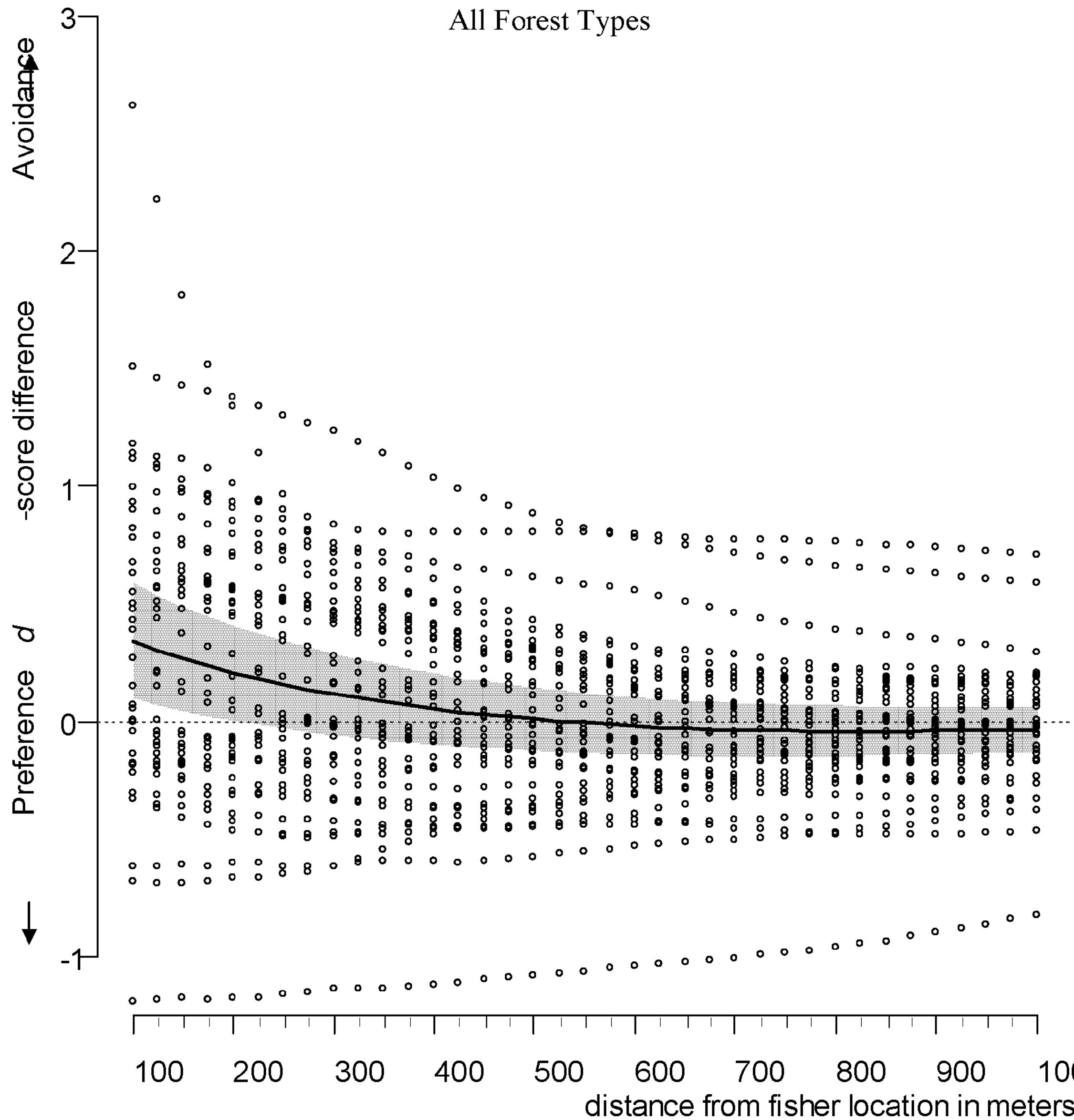

Figure 9. Fisher third-order selection preference of treated versus untreated forest. The black line represents mean difference in Compositional Analysis d-score statistic for fisher selection of untreated versus treated, calculated at distances of 100 to 1000 meters from fisher point location estimates at 25 meter intervals. Preference for untreated forest is indicated by positive values, preference for treated forest is indicated by negative values, and the dotted line at zero marks null selection. The gray zone surrounding the mean line represents the confidence interval surrounding the mean at each distance step. Significant selection is indicated by departure of the gray zone from the red line (here at distances < 200 meters). The points represent the difference values for each individual fisher.

Table 3. Compositional analysis mean *d*-scores and confidence intervals for fisher third-order selection preference of treated versus untreated forest, calculated at distances of 100 to 1000 meters from fisher point location estimates at 25 meter intervals. Positive values represent avoidance of treated areas; negative values represent preference for treated areas. Bolded values indicate statistically significant selection.

Proximity (meters)	Ponderosa Pine	Sierran Mixed Conifer	Montane Hardwood-Conifer	Montane Hardwood
100	**0.389 (0.016 , 0.763)**	**0.683 (0.185 , 1.181)**	0.133 (-0.246 , 0.512)	0.092 (-0.516 , 0.701)
125	**0.360 (0.003 , 0.717)**	**0.631 (0.143 , 1.119)**	0.107 (-0.239 , 0.452)	0.065 (-0.457 , 0.587)
150	**0.350 (0.001 , 0.699)**	**0.554 (0.094 , 1.013)**	0.089 (-0.226 , 0.404)	0.000 (-0.489 , 0.490)
175	0.319 (-0.015 , 0.653)	0.431 (-0.052 , 0.914)	0.084 (-0.195 , 0.364)	-0.051 (-0.527 , 0.426)
200	0.303 (-0.027 , 0.633)	0.414 (-0.048 , 0.877)	0.072 (-0.190 , 0.335)	-0.068 (-0.538 , 0.403)
225	0.289 (-0.023 , 0.601)	0.385 (-0.071 , 0.841)	0.058 (-0.188 , 0.304)	-0.075 (-0.526 , 0.375)
250	0.270 (-0.031 , 0.571)	0.361 (-0.091 , 0.813)	0.030 (-0.211 , 0.271)	-0.084 (-0.521 , 0.352)
275	0.258 (-0.032 , 0.548)	0.335 (-0.108 , 0.777)	-0.020 (-0.257 , 0.216)	-0.095 (-0.524 , 0.334)
300	0.248 (-0.030 , 0.526)	0.317 (-0.121 , 0.755)	-0.035 (-0.270 , 0.200)	-0.123 (-0.515 , 0.268)
325	0.236 (-0.032 , 0.504)	0.295 (-0.135 , 0.726)	-0.050 (-0.282 , 0.182)	-0.124 (-0.501 , 0.254)
350	0.220 (-0.039 , 0.480)	0.283 (-0.142 , 0.709)	-0.063 (-0.292 , 0.166)	-0.141 (-0.501 , 0.220)
375	0.203 (-0.048 , 0.455)	0.271 (-0.148 , 0.691)	-0.069 (-0.295 , 0.156)	-0.156 (-0.515 , 0.202)
400	0.185 (-0.060 , 0.431)	0.251 (-0.160 , 0.663)	-0.075 (-0.295 , 0.145)	-0.145 (-0.496 , 0.206)
425	0.170 (-0.071 , 0.411)	0.236 (-0.169 , 0.641)	-0.082 (-0.296 , 0.131)	-0.144 (-0.485 , 0.197)
450	0.153 (-0.082 , 0.389)	0.228 (-0.170 , 0.626)	-0.089 (-0.298 , 0.119)	-0.144 (-0.474 , 0.186)
475	0.135 (-0.094 , 0.365)	0.220 (-0.173 , 0.612)	-0.091 (-0.297 , 0.114)	-0.154 (-0.469 , 0.161)
500	0.117 (-0.105 , 0.340)	0.208 (-0.179 , 0.594)	-0.095 (-0.298 , 0.107)	-0.165 (-0.469 , 0.138)
525	0.102 (-0.114 , 0.318)	0.197 (-0.184 , 0.577)	-0.095 (-0.296 , 0.105)	-0.170 (-0.468 , 0.128)
550	0.090 (-0.120 , 0.299)	0.183 (-0.191 , 0.557)	-0.095 (-0.294 , 0.105)	-0.174 (-0.466 , 0.118)
575	0.079 (-0.125 , 0.283)	0.168 (-0.197 , 0.533)	-0.095 (-0.293 , 0.103)	-0.171 (-0.459 , 0.118)
600	0.071 (-0.130 , 0.271)	0.152 (-0.204 , 0.509)	-0.096 (-0.291 , 0.100)	-0.163 (-0.447 , 0.121)
625	0.064 (-0.134 , 0.261)	0.141 (-0.201 , 0.484)	-0.095 (-0.288 , 0.097)	-0.156 (-0.438 , 0.127)
650	0.055 (-0.138 , 0.248)	0.134 (-0.195 , 0.463)	-0.095 (-0.285 , 0.095)	-0.146 (-0.429 , 0.137)
675	0.048 (-0.141 , 0.237)	0.130 (-0.187 , 0.447)	-0.091 (-0.277 , 0.095)	-0.132 (-0.393 , 0.129)
700	0.040 (-0.144 , 0.224)	0.128 (-0.180 , 0.437)	-0.089 (-0.272 , 0.094)	-0.124 (-0.384 , 0.137)
725	0.035 (-0.145 , 0.216)	0.124 (-0.168 , 0.416)	-0.085 (-0.265 , 0.095)	-0.119 (-0.378 , 0.141)
750	0.031 (-0.146 , 0.207)	0.126 (-0.161 , 0.412)	-0.082 (-0.259 , 0.095)	-0.110 (-0.369 , 0.149)
775	0.026 (-0.146 , 0.199)	0.127 (-0.155 , 0.410)	-0.079 (-0.253 , 0.094)	-0.099 (-0.357 , 0.159)
800	0.023 (-0.146 , 0.191)	0.127 (-0.150 , 0.404)	-0.074 (-0.245 , 0.097)	-0.071 (-0.315 , 0.173)
825	0.018 (-0.146 , 0.183)	0.125 (-0.147 , 0.397)	-0.069 (-0.237 , 0.099)	-0.056 (-0.301 , 0.189)
850	0.015 (-0.146 , 0.175)	0.121 (-0.146 , 0.389)	-0.060 (-0.226 , 0.105)	-0.043 (-0.290 , 0.204)
875	0.012 (-0.144 , 0.169)	0.118 (-0.137 , 0.374)	-0.053 (-0.216 , 0.111)	-0.030 (-0.279 , 0.218)
900	0.010 (-0.143 , 0.163)	0.114 (-0.137 , 0.364)	-0.044 (-0.206 , 0.117)	-0.019 (-0.268 , 0.230)
925	0.009 (-0.140 , 0.158)	0.107 (-0.137 , 0.352)	-0.039 (-0.198 , 0.121)	-0.010 (-0.260 , 0.240)
950	0.008 (-0.138 , 0.153)	0.102 (-0.137 , 0.341)	-0.033 (-0.190 , 0.123)	0.000 (-0.251 , 0.251)
975	0.007 (-0.135 , 0.149)	0.097 (-0.136 , 0.330)	-0.029 (-0.183 , 0.125)	0.007 (-0.244 , 0.259)
1000	0.006 (-0.132 , 0.144)	0.104 (-0.121 , 0.329)	-0.025 (-0.176 , 0.126)	0.001 (-0.235 , 0.238)

Habitat Structural Characterization

In the pooled and each of the forest type's habitat structural characterization data there tends to be less canopy cover in treated areas than in untreated areas (Figure 10). Nearly as constant, with the exception of the Ponderosa Pine forest type, is the tendency of treated areas to show positive skewness of the vertical vegetation profile (Figure 11), indicating the vegetative mass is more vertically concentrated below the average vegetation height compared to untreated areas (Antonarakis et al. 2008), likely due to the combination of reduced canopy and released understory regeneration (Reinhardt et al. 2008). Results of the pooled data analysis provide evidence that treated areas tend to have canopies that are lower (maximum canopy height and mean canopy height, Table 4), thinner (canopy and mid-story density, Table 4), and more compact (interquartile range and absolute average deviation, Table 4) with a more uniform maximum height relative to average height (canopy relief ratio, Table 4), though not relative to ground area (canopy surface area ratio, Table 4).

As it comprised the largest portion of the pooled sample (49.2%), results for the Sierran Mixed Conifer subclass tended to correspond with those found for the pooled data, and were significantly different between treated and untreated forest with both tests for all metrics, save ground cover and mid-story density. Results of the LiDAR analysis were not uniform for all forest subgroups. While the treated Montane Hardwood subcategory was also characterized by a decrease in canopy cover and canopy relief ratio

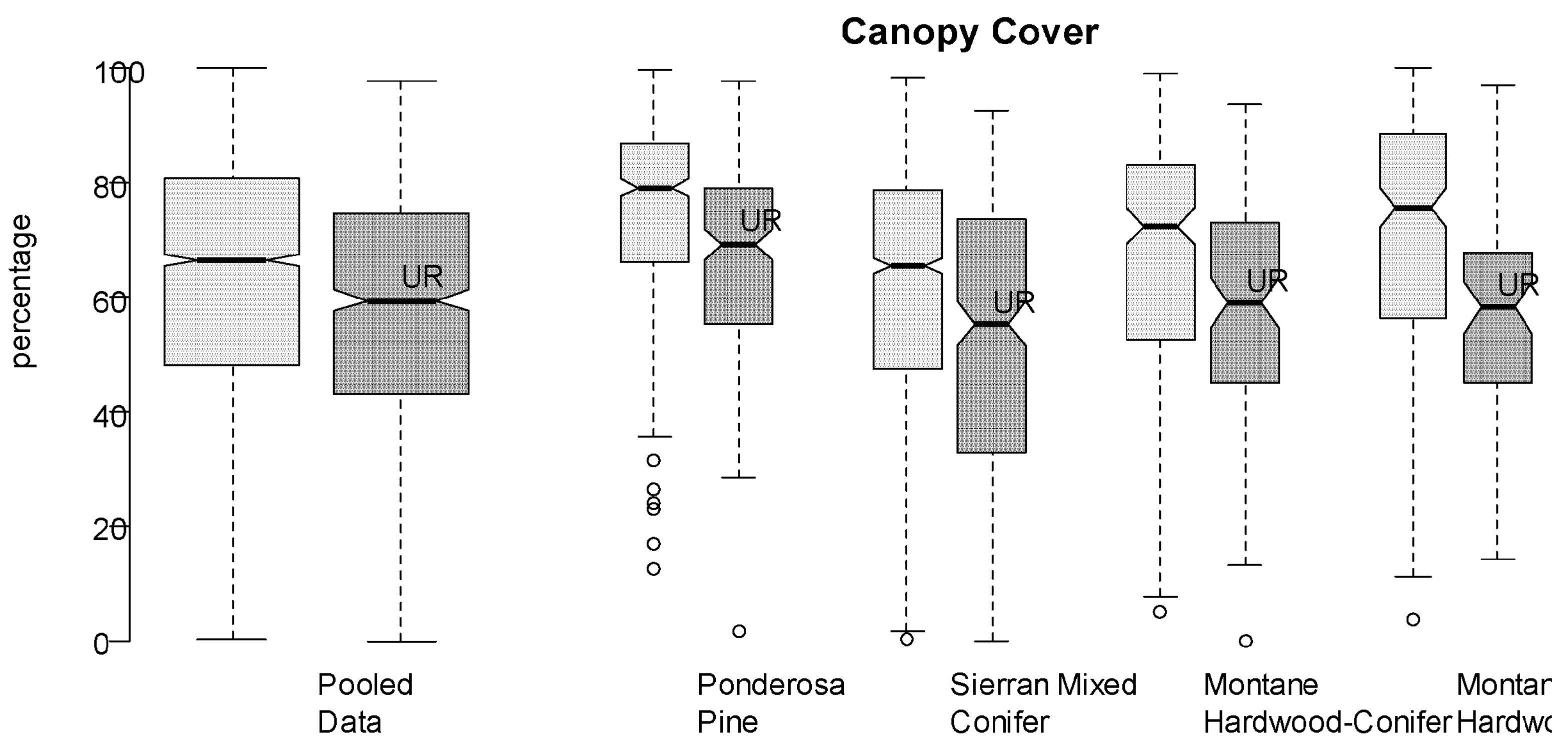

Figure 10. Results of Mann-Whitney-Wilcoxon U-test and randomization test for treated and untreated canopy cover of pooled forest types and each of the four forest type subcategories. Within each habitat category the box on the left represents untreated forest values and the box on the right represents treated forest. The vertical extent of each colored box represents the 25th (bottom) and 75th (top) percentile value. The bold horizontal line near the center of each box marks the median value, the whiskers mark the largest and smallest non-outlier values, and the dots represent outlier values (more than ±1.5 times the interquartile range). The notches within the boxes represent the median plus and minus the interquartile range divided by the squareroot of the sample size, providing rough 95% confidence intervals. Significant differences between untreated and treated value distributions found with the Mann-Whitney U-test are indicated with a “U” in the box representing treated data. Differences found to be significant by the randomization test are similarly indicated with an “R”.

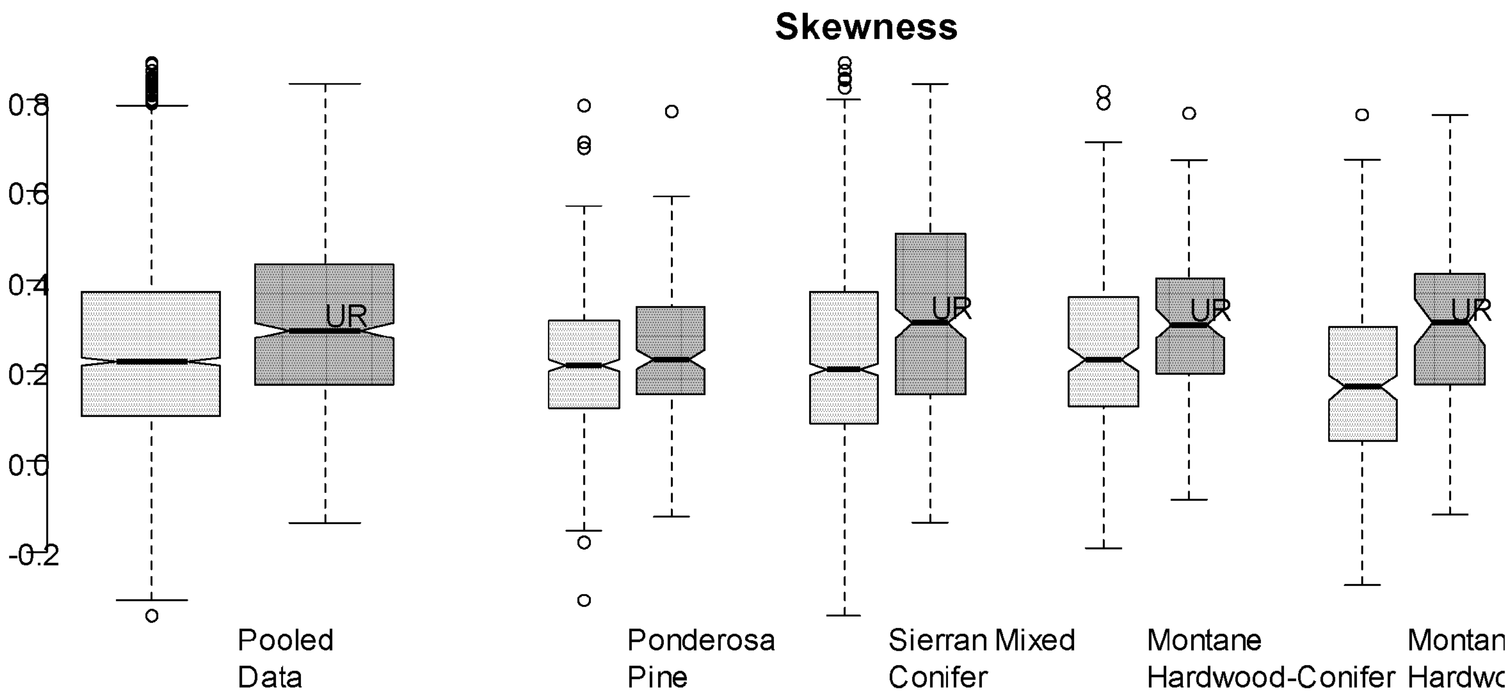

Figure 11. Results of Mann-Whitney-Wilcoxon U-test and randomization test for treated and untreated skewness of the vertical distribution of vegetation for pooled forest types and each of the four forest type subcategories. For this metric, higher values indicate a greater proportion of the vegetation is concentrated nearer to the ground than the overall mean height. Within each habitat category the box on the left represents untreated forest values and the box on the right represents treated forest. The vertical extent of each colored box represents the 25th (bottom) and 75th (top) percentile value. The bold horizontal line near the center of each box marks the median value, the whiskers mark the largest and smallest non-outlier values, and the dots represent outlier values (more than ±1.5 times the interquartile range). The notches within the boxes represent the median plus and minus the interquartile range divided by the squareroot of the sample size, providing rough 95% confidence intervals. Significant differences between untreated and treated value distributions found with the Mann-Whitney U-test are indicated with a "U" in the box representing treated data. Differences found to be significant by the randomization test are similarly indicated with an "R".

Table 4. Median values of investigated LiDAR metrics for untreated and treated forest samples.

	All Forests		Ponderosa Pine		Sierran Mixed Conifer		Montane Hardwood-Conifer		Montane Hardwood	
	Untreated	Treated	Untreated	Treated	Untreated	Treated	Untreated	Treated	Untreated	Treated
Maximum Canopy Height (meters)	31.3	29.1 [R]	28.8	30.6	36.2	29.1 [UR]	25.7	27.9	13.8	25.5 [UR]
Mean Canopy Height (meters)	10.5	9.8 [U]	11.4	11.9	13.3	9.4 [UR]	9.1	9.1	6.4	9.1 [R]
Canopy Relief Ratio	0.26	0.23 [UR]	0.26	0.26	0.28	0.22 [UR]	0.25	0.22	0.28	0.22 [UR]
Canopy Surface Area Ratio	1.0	0.97	0.94	1.05 [UR]	1.15	0.97 [UR]	0.86	0.91	0.46	0.85 [UR]
Canopy Cover (percentage)	66.6	59.5 [UR]	79.2	69.2 [UR]	65.6	55.4 [UR]	72.4	58.9 [UR]	75.7	58.2 [UR]
Ground Cover Density (percentage)	13.3	12.0	22.1	14.0 [UR]	11.3	11.8	20.2	10.9 [UR]	28.4	12.7 [UR]
Mid-Story Density (percentage)	27.5	25.6 [U]	46.5	34.1 [UR]	22.9	20.7	39.9	28.5 [UR]	55.1	24.7 [UR]
Canopy Density (percentage)	37.7	34.2 [UR]	38.7	38.1	45.1	34.0 [UR]	30.4	31.0	11.8	27.1 [UR]
Absolute Average Deviation (meters)	8.3	7.4 [R]	7.4	7.8	9.7	7.7 [UR]	6.4	6.9	3.5	6.6 [UR]
Interquartile Range (meters)	15.0	13.0 [UR]	13.2	14.5	18.6	13.1 [UR]	11.5	12.3	5.9	11.5 [UR]
Skewness	0.22	0.29 [UR]	0.21	0.22	0.20	0.31 [UR]	0.22	0.30 [UR]	0.16	0.31 [UR]

[U] Significant differences between untreated and treated values indentified with Mann-Whitney-Wilcoxon U-test

[R] Significant differences between untreated and treated values indentified with Randomization test relative to untreated, it also had markedly less ground cover (Table 4) and mid-story density, and in contrast to Sierran Mixed Conifer and pooled results, it had a taller, rougher canopy (maximum canopy height, mean canopy height, and canopy surface area ratio), with the vegetative mass more dispersed vertically (absolute average deviation and interquartile range).

Of the four forest type subcategories, Ponderosa Pine and Montane Hardwood-Conifer data displayed the least significant evidence of lingering fuel treatment effects. Treated forests of both types tended to have less canopy cover, and thinner mid-story and ground cover. Though the decreased canopy cover indicates a thinner canopy, as with the Montane Hardwood subcategory, the decrease in the lower vegetation strata, while not as extreme, again potentially influenced the absence of a significant decrease in canopy density. Contrary to Sierran Mixed Conifer forests, treated Ponderosa Pine forest tended to have more variation in maximum canopy height, with a higher canopy surface area ratio.

The Mann-Whitney-Wilcoxon U-tests (Wilcox 1945, Mann and Whitney 1947) and the randomization tests (Manly 2007) revealed similar results in the comparison of treated and untreated forest structure. Differences between the two tests affected findings of significance in only five of 55 instances, and in no case did the two tests produce contradictory results.

DISCUSSION

The shift in fisher habitat selection preference found here, between the largest and smallest scales investigated, offers an indication of the scale at which fuels treatments in the Sierra National Forest effect fisher behavior (Johnson 1980). For pooled forest types, and within Sierran Mixed Conifer and hardwood forest types, there was an unexpected tendency by fishers to include treated areas within their homeranges. When selection within the core-use area (50% KDE) is investigated, this inclusive trend disappears, and as fishers are selecting foraging and resting sites within their home ranges (third-order selection), they tended to avoid treated areas in favor of sites within untreated forest, corroborating previous findings (Truex and Zielinski 2013). Considering the small scale of most treated areas (mean = 16 ha, max = 487 ha) relative to an individual fisher home range size (mean = 2298 ha, min = 551 ha), accompanied with the spatial distribution of treatments over the landscape, it is possible that fishers may tolerate some portion of treated area within their home range, up to a threshold, in order to gain access to the untreated forest surrounding those areas. In this case the treated forest areas may see diminished use, but are included within the home range due to the higher quality habitat surrounding them.

Regardless of the cause behind the observed second-order selection, the significance of third-order selection within 200 meters of a fisher location is a strong indication that there are structural differences between treated and untreated relevant to fishers. Analysis of LiDAR data shows that there is less canopy cover in treated areas across all forest types. Concomitant with this general reduction of cover is the decrease of three-dimensional structural complexity in treated areas of most forest types, represented by the lower understory and ground cover density, as well as a lower distribution of the vegetation in the vertical profile (increased skewness). While direct associations cannot be made in this study between fisher location selection and forest structure, it seems likely that these characteristics, especially canopy cover, that have been repeatedly linked to fisher habitat (Zielinski et al. 2004a, Purcell et al. 2009), could be drivers of the observed third-order selection.

It should be noted that fisher selection was not uniform across all forest type subcategories. Though Montane Hardwood-Conifer and Montane Hardwood forests shared the decreased canopy cover and increased skewness found in treated areas among the pooled data, neither forest type showed significant third-order selection at any distance around fisher locations. Availability of adequate resting structures has been cited as a primary limiting factor of fisher habitat (Zielinski et al. 2004a, Purcell 2009). It may be that in hardwood and Montane Hardwood-Conifer forests, where treatment has actually increased canopy height and vertical distribution, the greater refugia offered by cavity-generating hardwoods helps to mitigate this pressure relative to conifer forests.

On the other hand, in the conifer forests, the harvest of intermediate and large sized conifers (>50 cm dbh) during treatment activities likely decreased the availability of what were already less common structures in those forest types. This possibility warrants further investigation that ideally would employ greater LiDAR coverage, or other structural data, to study direct associations between fisher locations and forest structural features within treated and untreated forest areas.

These findings suggest that the Kings River Sustainable Forest Ecosystems Project, and the management actions that followed, have had a negative net impact on fisher habitat. This impact is most apparent within coniferous forest habitat, where the reduced canopy cover and loss of three-dimensional structure is sufficient to influence fisher selection preference. These findings indicate that the short term negative impacts of management action reported by Truex and Zielinski (2013) continue to have lingering effects on habitat quality beyond the one year term reported in their study. The ultimate conclusion as stated in their report remains relevant: the impacts of management actions have not rendered the habitat unusable and are likely considerably less severe than those of large scale catastrophic fire. This leaves the still-problematic question of how best to reduce fuel loading and increase forest resiliency to fire while maintaining quality wildlife habitat.

Agee and Skinner (2005) describe four general principles forest managers may employ to increase forest resiliency to fire: reducing surface fuel loads; increasing the height to live crowns; maintaining large specimens of fire-hardy species; and reducing canopy density. The successful reduction of ladder fuels (ground cover and mid-story density) seen in the treated Montane Hardwood forests included here has not measurably affected use of the forest type by fisher. This suggests the possibility that fisher may tolerate implementation of the first two principles, provided access to adequate structural refugia, such as plentiful hardwood cavities, is maintained. Such refugia could be protected and provided by a combination of Agee and Skinner's (2005) third principle (maintenance of large, fire-hardy trees), as well as individual identification and conservation of habitat structures that are particularly vulnerable to fire such as snags, and creation of such structures during management actions, either through fire mortality or girdling.

Providing a definition of large, fire-hardy trees essential to fishers is critical in the application of fuel treatment principles. Tree species and stand conditions should be considered on a case-by-case basis, and particularly within conifer forests, care must be taken to maintain adequate resting structure. North et al. (2009) maintain that little forest health benefit is gained by the removal of intermediate-sized conifers (50 to 75 cm). This is well below the average live conifer rest site dbh of 117 cm reported by Zielinski et al. (2004a) or even the smaller value of 94 cm found by Purcell et al. (2009). However both studies report a wide range in these values and Zielinski et al. (2004a) indicate that conifers smaller than 50 cm dbh were on occasion utilized as rest sites. In forests where hardwoods provide adequate refugia, it is possible that removal of some intermediate and larger-diameter conifers may have less of a negative effect on rest site availability, though the data investigated here are insufficient to provide definitive evidence. Management actions that result in harvest of these intermediate-sized trees in conifer forests, on the other hand, are likely truncating the lower end of the distribution of available suitable rest site structures, while reaping diminishing returns of increased fire resiliency.

This study and others have linked dense canopy cover to high quality fisher habitat (Burskirk and Powell 1994, Truex et al. 1998, Mazzoni 2002, Zielinski et al. 2004a, Jordan 2007, Purcell et al. 2009, Truex and Zielinski 2013), putting fisher conservation at direct odds with the fourth principle of fuels treatment: reduction of canopy density. However, research shows that the bulk of the efficacy of fuels reduction actions is accomplished by addressing the first three principles (Agee and Skinner 2005, North et al. 2009), and in the absence of surface fuels, independent active canopy fire can occur only during the most extreme of weather conditions (Agee and Skinner 2005, Stephens and Moghaddas 2005). Additionally reduction of canopy density can increase ground fire intensity by increasing surface wind speed (Graham et al. 2004); increasing exposure of surface fuels to sunlight, facilitating their drying; increasing risk of windthrow (Alexander 1986); and releasing resources (sunlight and water), allowing an increase of ground cover and shrubs fuels (Reinhardt et al. 2008). The paramount importance of sustaining dense canopy cover for quality fisher habitat, in concert with the diminishing returns offered by canopy reduction actions, suggests that forest managers focus on the first three of Agee and Skinner's (2005) principles (ground and ladder fuels reduction, and retention of large, fire-hardy trees), of which fishers appear to me more tolerant, and reserve canopy reduction as a final option for reaching fire resiliency goals.

The loss of the economic offset to management actions provided by harvesting intermediate and larger trees could be partially mitigated by the increased application of prescribed fire. Prescribed fire can be applied at a fraction of the cost per acre of mechanical treatments (McCandliss 2002) and has far less negative impact on soil conditions (North et al. 2009). Properly applied, prescribed fire can reduce surface and ladder fuels while maintaining large downed woody debris and canopy cover (North et al. 2009). Fire opens growing space, provides nutrients, naturally increases heterogeneity, and can create the snags and cavity structures critical to quality fisher habitat (North et al. 2009). Additionally fire is a natural phenomenon that has in part already shaped the evolution of fishers in the area and has been shown to be an effective means of fuels reduction in this region (McCandliss 2002).

The southern Sierra fisher population presents a challenge to forest managers. However fisher conservation and increased fire resiliency of the forest they inhabit are not mutually exclusive goals; rather they are a matter of prioritization, both of fuels reduction goals and fisher habitat needs. By focusing fuels treatments on reduction of ground and ladder fuels and increasing forest heterogeneity, while maintaining areas of dense canopy cover and structures important to fishers, such as large diameter trees and snags, it is possible to restore the ecological role of fire in a sustainable forest suitable for fishers.

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