

# Using routinely collected regional forest inventory data to conclude that resting habitat for the fisher (*Pekania pennanti*) in California is stable over ~ 20 years



William J. Zielinski<sup>a,\*</sup>, Andrew N. Gray<sup>b</sup>

<sup>a</sup> U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, 1700 Bayview Dr., Arcata, CA 95521, United States

<sup>b</sup> U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Corvallis, OR 97331, United States

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## ABSTRACT

The conservation and management of species-at-risk requires periodically collecting information about their distributions and abundances. A comprehensive monitoring plan should, in addition to monitoring the population itself, also assess the status of habitat elements that are key factors in species survival. Places where animals seek safe and secure places to rest are such key habitat elements. We used previously published models to predict resting habitat for fishers (*Pekania pennanti*) throughout much of their range in California. Unique to this work is that the two models (northwestern California and southern Sierra Nevada) were developed using, as predictors, variables directly from a national plot-based forest inventory program called Forest Inventory and Analysis (FIA). Using these models, relative resting habitat suitability can be estimated at each geographically relevant plot in the FIA system every time the plot is resampled. We applied these predictive models to data collected at 3 time periods over an approximately 20 year period to evaluate the trend in predicted fisher resting habitat. None of the 8 national forests, 4 in the northwestern California region and 4 in the southern Sierra Nevada region – nor either of these 2 regions as a whole – exhibited trends in predicted resting habitat suitability that were significantly increasing or decreasing. Predicted resting habitat suitability tended to be lower on private land than public land, in both regions. As expected, plots that were disturbed by fire exhibited a decrease in resting habitat suitability but, surprisingly, the few plots within harvest unit boundaries had indistinguishable values before and approximately 7 years after the harvest. Using FIA data for future assessments of habitat value will avoid the significant cost incurred when the data need to be collected repeatedly using different data and a field protocol that may vary. We anticipate that the FIA program will continue to be the preeminent plot-based vegetation survey in the United States, and the data to run the resting habitat models will be available every 10 years. Moreover, access to routinely updated plot-based data provides the only way we can envision sampling something as fine-scaled as resting habitat over thousands of square miles of potentially suitable habitat. We hope our example encourages others to parlay the FIA data into a predictive model of fine-scale habitat features that is relevant to other species. Demonstrating the utility of our models should also encourage managers to use the predictions to evaluate the status of fisher habitat in California.

## 1. Introduction

The conservation and management of species of concern requires information about the status and trends of key elements of their habitat. This, however, is a practical and financial challenge when the need is to monitor specific habitat conditions or elements over large and heterogeneous areas. Typically the only feasible approach is to develop a predictive model that relates remotely sensed characteristics of large, coarsely labeled vegetation polygons to general habitat value (e.g. Guisan and Zimmerman, 2000; Nijland et al., 2015; Westover et al.,

2016). More difficult to assess over large areas are changes to the important, but localized, microhabitat elements such as nest sites or resting or roosting sites. These microhabitat features can be essential, but are exceedingly difficult and expensive to assess and monitor over large regions.

Given these constraints, the response to this dilemma is to develop a predictive model that estimates the relative value of habitat elements and track them over time. The traditional approach to doing so, however, has been to collect microhabitat data at used and random locations using a *researcher-derived* protocol for measuring the vegetation

\* Corresponding author.

E-mail address: [bzielinski@fs.fed.us](mailto:bzielinski@fs.fed.us) (W.J. Zielinski).

and topographic covariates to be used as predictors. Examples include a number of papers to which the senior author has contributed (e.g., Slauson et al., 2007; Zielinski et al., 2004a,b). These models generate unique predictors selected by the researcher that help us understand habitat needs but are often measured only once using the researcher's biologically relevant, but often singular, methods. In these cases field data on the model predictors are rarely collected on subsequent occasions, particularly over large regions, making it difficult to use the model to assess the future status of habitat elements as they change. New initiatives, however, have exploited the efficiency of publicly available, routinely resampled, plot-based vegetation databases as the sources for predictors for wildlife habitat models (e.g., Dunk et al., 2004; Huff, 2006; Fearer et al., 2007; Welsh et al., 2006; Dunk and Hawley, 2009; Twedt et al., 2010). The use of vegetation data that are collected regularly over time as part of a reliable government inventory program provides serial remeasurement of the predictors, and serial estimates of microhabitat conditions. Not only will this be more efficient but it may be the only way to assess the status and change in specific habitat elements over large regions.

The fisher (*Pekania pennanti*) is an uncommon carnivorous mammal in the Mustelidae that is of conservation concern in the western United States, as evidenced by multiple petitions for listing under the US Endangered Species Act. Fishers are among a wide variety of species of wildlife that use cavities or chambers in live and dead trees as daily refugia and for reproduction (Zielinski et al., 2004a; Lofroth et al., 2010; Weir et al., 2012; Green, 2017). These resting and denning (reproduction) structures are most typically the largest diameter standing live trees, snags, or logs (conifers and hardwoods) available (Zielinski et al., 2004a; Purcell et al., 2009; Lofroth et al., 2010; Aubry et al., 2013) yet other woody features, such as platforms of branches or mistletoe in tree canopies, can constitute a significant minority of resting locations (Seglund, 1995; Green, 2017). Cavities are typically in large trees and are considered one of critical elements for the maintenance of fisher populations (Paragi et al., 1996; Purcell et al., 2009; Lofroth et al., 2010; Weir et al., 2012; Green, 2017) and these features may take hundreds of years to develop. As a result there is a premium on information about the amount and distribution of resting habitat because it can be degraded at a much higher rate than it develops.

The conservation of fisher populations requires an understanding of their habitat ecology and the development of population and habitat monitoring programs. Native fisher populations occur in California in the southern Sierra Nevada Mountains (hereafter “southern Sierra” or SSN) and in the mountains of northwestern and northcentral California (hereafter “northwestern California” or NWCA) (Lofroth et al., 2010). Monitoring populations directly is a key component of any conservation strategy and a comprehensive program to monitor the status of the fisher population in the southern Sierra has been underway for approximately 15 years (Zielinski et al., 2013; US Forest Service, 2015). An analogous program to monitor the population in northwestern California, however, has not been undertaken. Although this shortcoming should be addressed, even if both regions had population monitoring programs, without a companion habitat monitoring program it is impossible to know whether a change detected in population is caused by a change in habitat condition, or some other factor. Despite this need, no program is in place to monitor changes in fisher habitat quantity or quality in either of the major regions of fisher occurrence in California.

We have previously developed empirical models that predict resting habitat value for fisher populations in both regions using plot data from publicly available and routinely remeasured vegetation inventory data (Zielinski et al., 2006; Zielinski et al., 2010; Zielinski et al., 2012). This was accomplished by integrating models of fisher habitat associations with the Forest Inventory and Analysis (FIA) forest inventory program (Reams et al., 1999; Bechtold and Patterson, 2005; Gray et al., 2012). The FIA program is a nationwide, probability-based sampling scheme designed to inventory and monitor natural resources. The design consists of sample points located in a systematic hexagonal grid (with

centers of hexagons 5.47 km apart for one point per 2400 ha) across all ownerships in the United States with the goal of measuring environmental variables at sample plots every 10 years in the western U.S. (Bechtold and Patterson, 2005). FIA data are most often used to assist in planning forest management activities and monitoring forest composition, structure and disturbance. However, the FIA data are an attractive option for assessing and monitoring wildlife habitat because they are a temporally and spatially reliable source of field-measured habitat data across large regions. Moreover, the FIA program has long-term institutional support for resampling these plots. Although we have developed fisher resting habitat models that are linked to these inventory data, and have demonstrated their utility for monitoring this important habitat across individual national forests (i.e., Zielinski et al., 2010 see Fig. 2 therein), we are unaware of this information being used to inform project plans on national forests or as foundation in updating land management plans. Thus, here we use the most recent three cycles of FIA data available in each region to estimate regional trends in resting habitat suitability over a ~20-year period. In addition to estimating change in fisher resting habitat over a large portion of their range in California, our work should demonstrate the untapped potential of FIA data, which when linked to a predictive habitat model can be used to evaluate how disturbances such as timber harvest and fire affect the relative suitability of wildlife habitat.

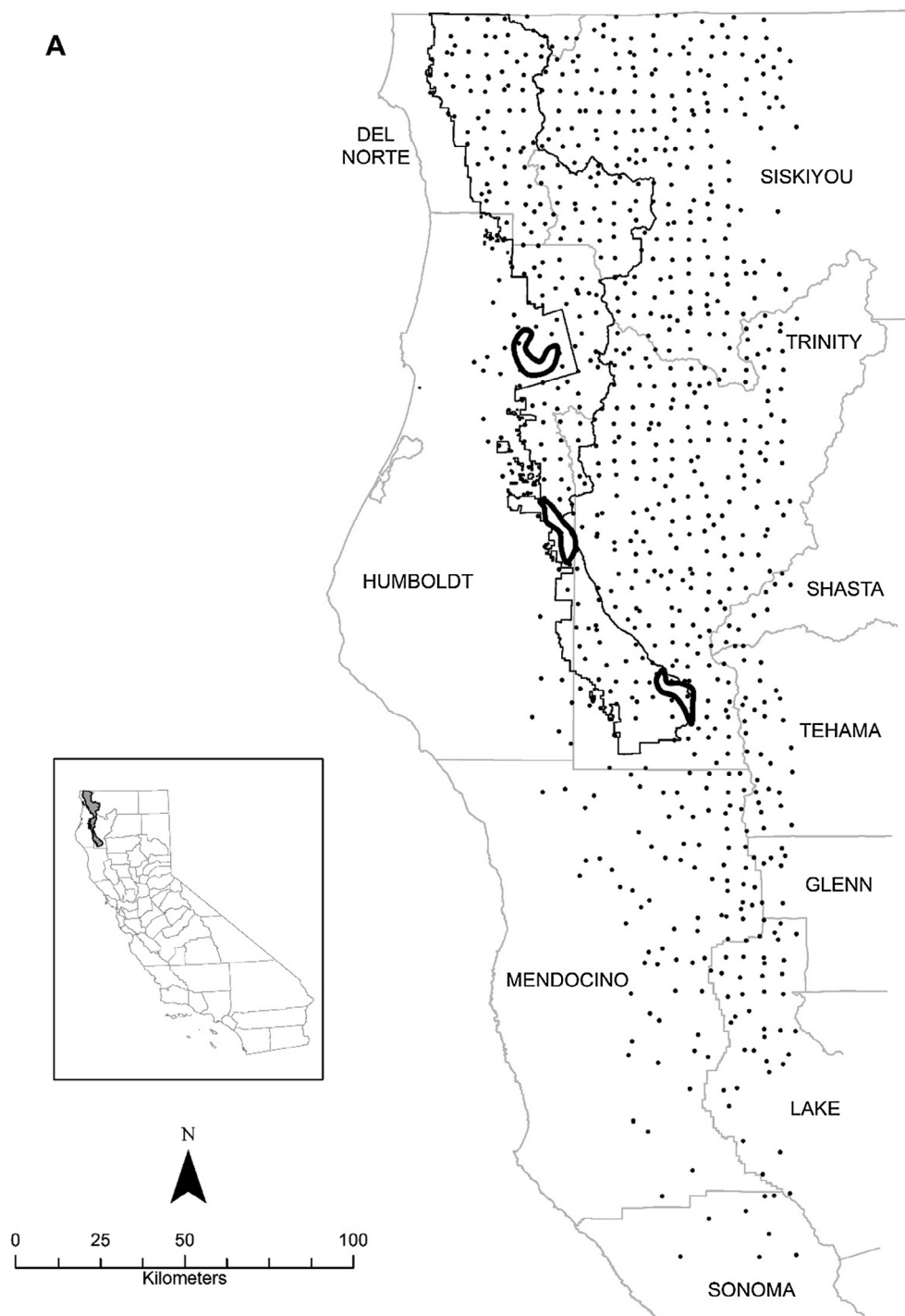
## 2. Methods

### 2.1. The FIA-based fisher resting habitat models

The foundation for this work are two predictive resting habitat models for fishers that use variables from the FIA plot sampling protocol as predictors – one built from field data collected from fishers and their resting locations in the southern Sierra (Zielinski et al., 2006) and one from data collected in northwestern California (Zielinski et al., 2012). Building a predictive habitat model requires a comparison of characteristics at *used* locations (i.e., the sample of resting structures) with those that are *available* in the sample of regularly sampled FIA pots in the general vicinity of the resting structures. The southern Sierra model was built using data from plots centered on 75 randomly selected fisher resting structures as well as similar data from 232 of the nearby and regularly sampled plots in the FIA system, whereas the northwestern California model was built using plots centered on 99 randomly selected fisher resting structures and 883 nearby plots in the FIA grid.

The resting structures that were sampled to create the models were originally located during the course of two studies on the habitat ecology of fishers in the southern Sierra and two studies in northwestern California (Fig. 1) and were used by males and females. In the southern Sierra, the first study was conducted from 1994 to 1996 in the Sequoia National Forest in Tulare County (Zielinski et al., 2004a,b) and the second from 1999 to 2000 in the Sierra National Forest, Fresno County, California (Mazzoni, 2002) (Fig. 1). In northwestern California, the first study had two sub-areas and was conducted from 1993 to 1997 on the Six Rivers and Shasta-Trinity National Forests in Humboldt and Trinity Counties (Zielinski et al., 2004a,b; Fig. 1) and the second was conducted from 1996 – present on the Hoopa Valley Tribal Reservation (Higley and Matthews, 2009). Animals were captured, fitted with radio-transmitter collars and tracked on foot to their resting locations approximately once per week. Details regarding handling and telemetry methods are available elsewhere (Zielinski et al., 2004a,b). None of the resting structures were known to be used for the birth or care of young (i.e., dens).

Vegetation attributes at fisher resting locations were measured using the FIA vegetation sampling protocol (USDA Forest Service, 2007; Christensen et al., 2016) by surveyors that were also contracted to measure the regularly sampled FIA plots. At resting locations the plot was centered on the resting structure. The FIA protocol involves the collection of vegetation data at four or five subplots (see details



**Fig. 1.** The (A) northwestern California and (B) southern Sierra study areas on background that includes county names (in caps) and national forest boundaries (intermediate width irregular lines). Each figure depicts the forested FIA points (small dots) that were within the ecoregion of interest, which is also the area of application for the regionally specific resting habitat model. These are the FIA points that are in the regularly sampled set, and which will continue to be sampled on a 10-year cycle. The thickest lines are the boundaries of the areas where the data on resting sites of radio-marked fishers were collected on each study area. Data on fisher resting locations were collected in these areas to develop the predictive models.

regarding this variation, and its lack of effects on plot estimates, in Zielinski et al., 2010) within a 1.0-ha circular footprint. Within each subplot, a nationally standardized set of attributes are measured or estimated, including live and dead trees, site productivity and topography, stand structure, and disturbance history. In addition, regionally important measurements are taken, including understory vegetation composition, the quantity of downed wood and litter, ground cover, and other physical features (see Christensen et al., 2016; USDA Forest Service, 2007, <http://fia.fs.fed.us> and <https://www.fs.fed.us/pnw/>

[rma/fia-topics/documentation/field-manuals/index.php](http://fia.fs.fed.us/rma/fia-topics/documentation/field-manuals/index.php) for details on national and regional FIA sampling protocols and data availability).

The original FIA-based fisher resting habitat model (southern Sierra; Zielinski et al., 2006) was developed using a regional FIA field protocol that was revised shortly thereafter with the adoption of new national standards. The original protocol was used in the 1990s and included five subplots and variable-radius sampling of trees. The FIA program refers to it as the “Region 5 [California] periodic inventory.” The revised (current) FIA protocol was used to sample fisher resting structures

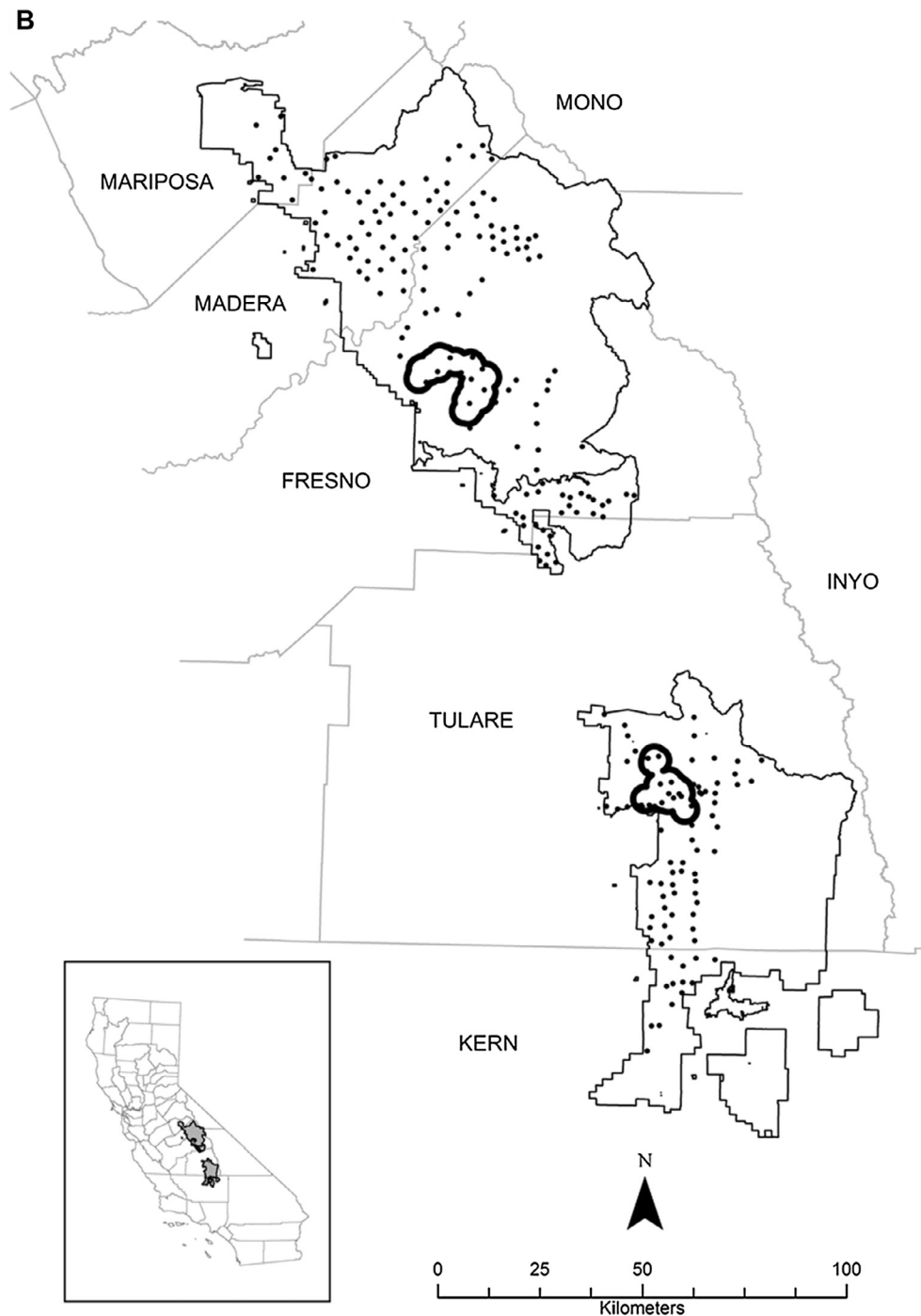


Fig. 1. (continued)

when we developed the northwestern California model (Zielinski et al., 2012) and is referred to as the “FIA annual inventory” and includes four subplots and nested fixed-radius subplot sampling.

The predictive resting habitat models for the 2 regions were developed by comparing environmental characteristics at the FIA-style plots centered on the fisher resting structures in each region with the characteristics of the official, regularly sampled, and grid-based FIA points that were in the same general ecoregional type. For each model we limited the extent of the available set of FIA plots in the regularly sampled grid system to those within the same elevation zone and the same ecosections of the Ecological Unit of California system (Bailey, 1994, 1995) where the fisher resting structures in that area were located (Zielinski et al., 2006, 2012). This assured us that the final model

would be applicable to the general region where the model was developed, reducing sample selection bias (Phillips et al., 2009). We refer to the extent of each model’s inference as the “area of application” (corresponding with the area occupied by the FIA points identified in Fig. 1A and B). We estimated relative fisher resting habitat value using predictors measured from the entire FIA plot, regardless of whether a fraction of the plot was non-forest. However, the great majority of plots (84%) were at least 90% forested.

To model the distinction between the vegetation characteristics at resting structures and the characteristics at the set of nearby FIA inventory plots, we used nonparametric logistic regression (Hastie and Tibshirani, 1987) or maximum entropy models (MaxEnt ver. 3.3.3e, Phillips et al., 2006). For each model (i.e., Zielinski et al., 2006;

**Table 1**

Predictor variables in the top resting habitat model for fishers in each of the 2 regions in California. Original work described in Zielinski et al. (2006) and Zielinski et al. (2012).

| Southern Sierra                                                 | Northwestern California                                     |
|-----------------------------------------------------------------|-------------------------------------------------------------|
| Canopy closure (%)                                              | Canopy closure (%)                                          |
| Basal area of trees < 51 cm DBH ( $\text{m}^2 \text{ha}^{-1}$ ) | Age of dominant conifers (yr)                               |
| Mean DBH of hardwoods (cm)                                      | Basal area ( $\text{m}^2 \text{ha}^{-1}$ )                  |
| DBH of largest conifer snag (cm)                                | Hardwood basal area ( $\text{m}^2 \text{ha}^{-1}$ )         |
| Slope (%)                                                       | Volume of large downed wood ( $\text{m}^3 \text{ha}^{-1}$ ) |
| DBH of largest tree ( $\text{cm}$ ) <sup>a</sup>                |                                                             |

<sup>a</sup> DBH = diameter at breast height.

Zielinski et al., 2012) we evaluated a number of biologically feasible univariate and multivariate candidate models (Burnham and Anderson, 2002). The best model included a different set of predictor variables for each region (Table 1). The values for each of the variables in the best model were derived from the data at each plot and entered into the model to generate an estimate of Resting Habitat Suitability (RHS) that varied from 0 to 1. We used the best predictive model to estimate RHS at each of the regularly sampled FIA plots that fell within the appropriate elevation and ecoregional zones. Estimates of means and variances of RHS were calculated using the ratio of means estimator for a post-stratified sample (Scott et al., 2005), where the strata were built from classified satellite imagery, ownership layers, and climate models (see Christensen et al., 2016). We calculated 95% confidence intervals by multiplying the standard error of the estimate by 2. The mean and 2x the standard error of the predicted value for each time period were used to represent the status of RHS for each region, and each national forest within each region.

We demonstrate the monitoring value of these data by comparing the mean predicted RHS values from plots that were surveyed by the FIA program for each of the 3 occasions they had been sampled. It took several years to sample a single occasion. Thus, the period of time from the first sample of the first occasion to the last sample of the third occasion was approximately 20 years. It was 12 years, however, between the mean measurement year of the first and third occasion. The first sample of vegetation data that the FIA program collected (the R5 inventory) was a partial regular grid inventory with additional pre-stratified points that was designed to support remote sensing classification and mapping and took 3 years to complete (1995–1999; mean measurement year = 1998). The next 2 samples were collected with a new hexagonal sampling frame which selected one previously-installed inventory plot if present in a FIA sample unit, and added a random grid point if prior plots were not present. The design in the western U.S. measures 10% of the plots every year in a spatially- and temporally-balanced design, resulting in 10-year intervals between the collection of data at each plot (i.e., a 10-year inventory cycle; Bechtold and Patterson, 2005). On National Forest lands, panels were doubled in order to install all the annual plots during the second sampling occasion within 5 years (2001–2005; mean measurement year = 2003). For the third sampling occasion on National Forest lands, 10% of the total number of FIA plots were measured in each of 10 consecutive years (2006–2015; mean measurement year = 2010), a sampling regime that is intended to be repeated in perpetuity.

The plots from the first occasion (R5 inventory) which had been selected for the subsequent annual inventory became the plots that were measured 3 times over the period from 1995 to 2015. The initial sample, however, had some irregularities that needed to be addressed. First, the R5 inventory is an incomplete grid which can introduce the potential for bias, and second, the smaller sample of plots that were sampled 3 times can increase sample error. So, to produce the best adjusted estimate for the first sample occasion, we calculated the change in RHS between the first and second occasion for the subset of plots that were sampled 3 times, and subtracted that change from the value calculated with the full sample of annual plots from the second sample.

We evaluated the RHS over time separately for each national forest that included a portion of the area of application in each region. This included the Stanislaus, Sequoia, Sierra and Eldorado National Forests in the southern Sierra region and the Klamath, Mendocino, Shasta-Trinity and Six Rivers National Forests in the northwestern California region. We also pooled data from the 4 national forests in each region to derive separate region-wide estimates. The RHS values were also calculated for private lands in each region. Because FIA plots in California that were on private lands were sampled several years earlier (1991–1994) than those on National Forest (1991–1994) and differed in design we provide estimates for the private lands only for the second and third sampling occasions.

## 2.2. The effect of disturbance on predicted resting site suitability

Given their negative effects on key predictors in each of the models, fire and timber harvest should reduce the RHS, at least in the short term. Thus, evaluating the RHS at FIA plots that are known to be harvested or recently burned should help instill confidence that the model is tracking relative changes in resting suitability. We compared the estimates of resting habitat value at those plots that experienced either harvest, fire, both events or no disturbance between the second plot sample (mean measurement year 2003) and the third plot sample (mean measurement year 2010). These 2 sampling occasions were included because their identical plot layout and measurement protocols provided the most accurate assessment of change; compared to the different protocols used at the first and second occasions. This evaluation was conducted with 495 and 598 plots in our southern Sierra and northwestern California study areas, respectively. We used estimates of disturbance that were based on field crew interpretations of what occurred to each plot between measurements. The stumps of previously measured trees were an unambiguous indication that harvest had affected a plot. Evidence of wildfire and prescribed fire were noted, but we did not attempt to distinguish them. Locations of wild and prescribed fires were well known to field crew members, many of which lived near the areas they surveyed and had local information available to them. This information contributed to their determination that fire had affected the vegetation on a plot.

## 3. Results

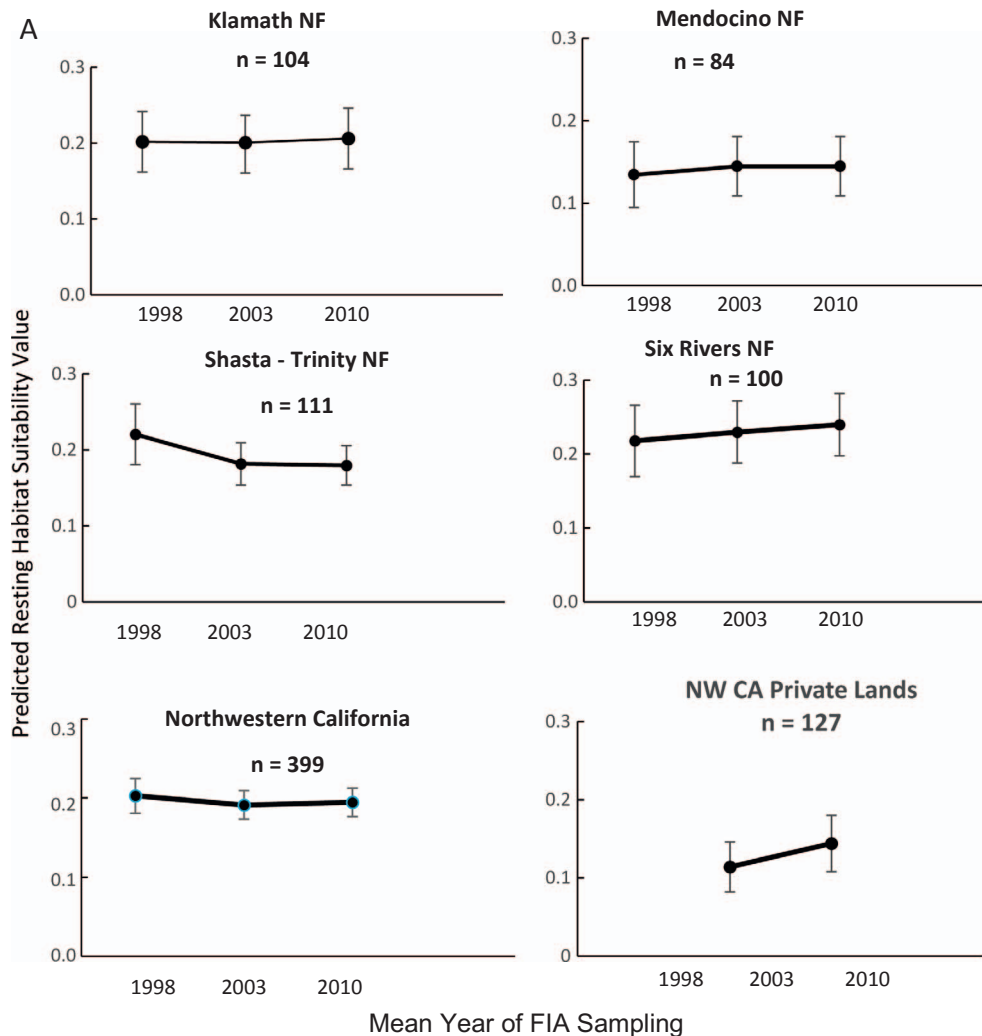
### 3.1. Southern Sierra

There was no change in the estimated RHS values over the 3 sample periods in the southern Sierra study area. Point estimates of RHS hovered around 0.15 on each sampling occasion when the predictions from FIA plots across the southern Sierra region were combined (Fig. 2B). The uniformity of this response was a product of slight increases in RHS over time on the Stanislaus and Sequoia NF and relatively modest decreases in RHS on the Eldorado and the Sierra NF (Fig. 2B). Estimates from plots on the Eldorado and Sierra NFs suggest the beginning of a negative trend in RHS.

### 3.2. Northwestern California

There was also no significant change in the estimated RHS values over the 3 sample periods in the northwestern California study area. RHS was estimated to be about 0.20 on each sampling occasion for the 4-forest region (Fig. 2A). The flat response line was likely because the RHS values on the Klamath and Mendocino NFs were virtually unchanged over time and the Shasta-Trinity and Six Rivers NFs, which had the suggestion of trends in opposite directions, combined to produce a response with near-zero slope (Fig. 2A).





**Fig. 2.** Predicted resting site suitability for 4 national forests in (A) northwestern California and 4 national forests and in (B) the southern Sierra areas of model application. Each also includes pooled data for study area and data for estimates of RHS for 2 sampling occasions on private land. The estimates represent resting habitat predictions for FIA plots that were sampled circa 1998, circa 2003 and circa 2010. Bars represent 2 standard errors above and below the estimate (i.e., approximately 95% confidence interval).

### 3.3. Ownership

Predicted values on private lands were lower, in general, than on federal or state lands (Fig. 2A, B). Predicted values on private lands in the southern Sierra had values between 0.05 and 0.10 whereas private lands in northwestern California were in the 0.125–0.2 range. Only 6 plots fell on private lands in the southern Sierra area which probably accounts for the larger standard errors on estimates from plots on private lands compared to national forests in this region (Fig. 2B).

### 3.4. Effects of disturbance

Plots without reported fire or timber harvest had no apparent change in RHS values from 2003 to 2010 (Fig. 3). However, 179 plots (64 in the southern Sierra and 115 in northwestern California) were disturbed by either harvest or fire resulting in decreased estimates of RHS value (Fig. 3). Too few plots were subjected to fire *and* harvest to include this condition in our analyses ( $n = 2$  and  $n = 3$ ). In the southern Sierra, examination of plot data suggests that recent harvest or fire on 20% of the plots measured on a single national forest, the Eldorado, may have significantly affected the estimates for the southern Sierra region. The conspicuous effects of fire on plots in the southern Sierra is explained largely by decreases in 3 of the 6 predictor variables: basal area of trees < 51 cm dbh, canopy cover, and dbh of the largest

tree (Table 2). The magnitude of change in plots burned in northwestern California region was much less, but the confidence intervals did not overlap (Table 2). On average, harvest had little effect on RHS values, possibly because most of the harvest consisted of partial cutting of understory trees, as indicated by the relatively small changes in most of the predictor variables.

## 4. Discussion

Our work suggests that fisher resting habitat, as estimated using FIA plot data, has not significantly changed for approximately 20 years in both of the significant regions of the fisher's range in California. This is important information for both regions, but particularly welcome news regarding the southern Sierra population because it is at greater risk due to its isolation (Knaus et al., 2011) and relatively small size (< 500 individuals; Spencer et al., 2011). The population in the larger region that includes our northwestern California study area is almost an order of magnitude larger (~4000 individuals; Furnas et al., 2017). We verified that our model-based approach identified the decreases in RHS expected when the vegetation structure at a site is altered by the most common forms of disturbance in forests: fire. Thus, we believe the method is suitable for assessing the status of resting habitat as well as reflecting the changes in vegetation structure that occur when the predictor variables are altered by disturbance. This is particularly

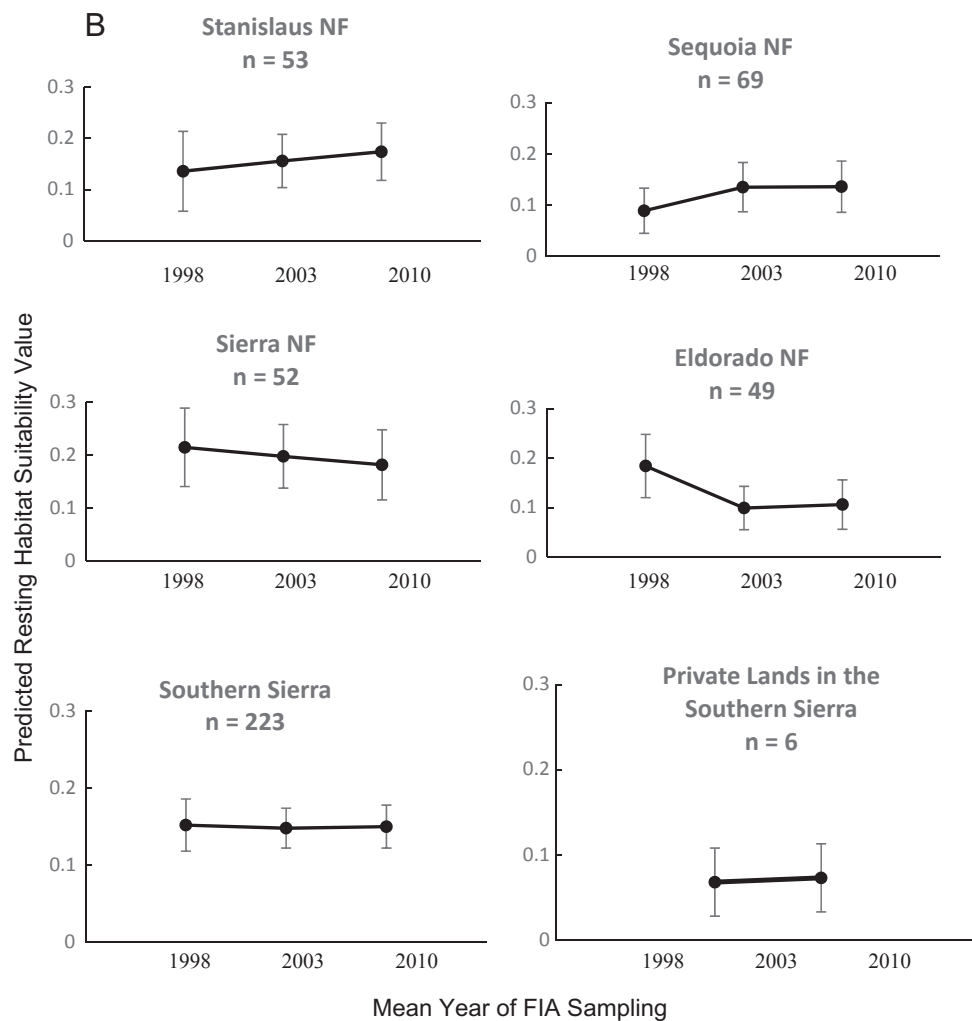


Fig. 2. (continued)

important given that resting structures and the sites immediately surrounding them can be lost quickly but take hundreds of years to develop.

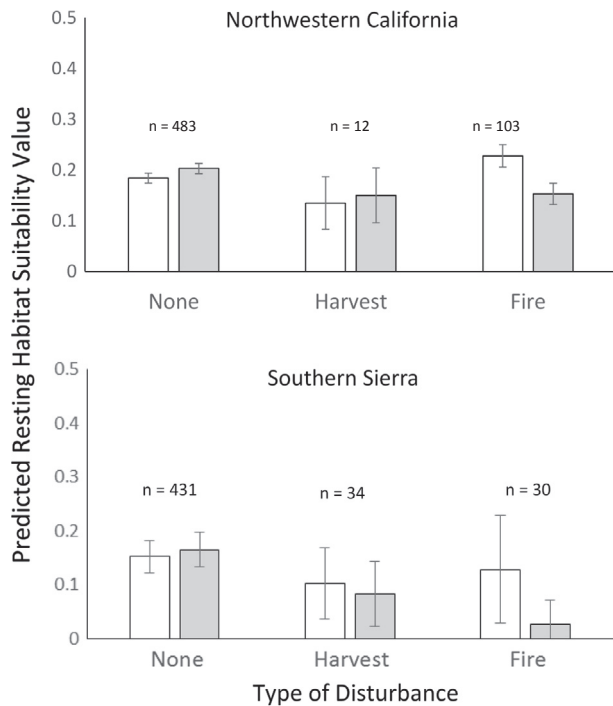
If the FIA program conducts an inventory every 10 years, as expected, the RHS values calculated for a particular national forest can be an important backdrop for planning management activities. For example, based on the information presented here the managers responsible for fisher conservation on the Eldorado and Sierra National Forests, where the data suggests a decline in RHS (Fig. 2), may want to be particularly cautious about proposing activities that may negatively affect resting habitat. Conversely, the direction of trends in RHS values on the Stanislaus and Sequoia National Forests suggest perhaps the beginning of an increase in suitability over time (Fig. 2) and managers on these two forests may have a bit more flexibility. We also anticipate that FIA-based model estimates of RHS will contribute to the evaluation of the effects of planned activities on fishers, which are required of national forests when they revise their land management plans. This information will also be useful to the US Fish and Wildlife Service and California Department of Fish and Wildlife when they conduct periodic status reviews and listing evaluation for fishers.

We noticed a pattern that RHS estimates in the southern Sierra were consistently higher than those in northwestern California (Fig. 2A, B). This applies to the pooled data for all 4 forests in a study area as well as the estimates of RHS on private land in each region. The RHS estimates are consistently about 0.05 higher in northwestern California than in the southern Sierra study area (~0.20 vs ~0.15, respectively; Fig. 2A, B). The reason for this pattern is not immediately apparent but

it may be that the top model for northwestern California has a better fit to the data and, therefore produces higher values of RHS in that region. That we see a similar regional disparity in RHS values on the private lands and the national forests suggests that it may be due to an interaction between habitat selection by fishers and differences in climate or vegetation structure in the two regions. This will need further investigation.

Our model produces values of *relative* habitat suitability that apply to our study only, and are not directly comparable to values in other studies. Nonetheless, the predicted values of RHS in each of the study areas tend to be low (0.15–0.20). This phenomenon is not uncommon for habitat specialists and for species whose ideal habitat is uncommon (Dunk and Hawley, 2009; Zielinski et al., 2015); conditions that apply to the fisher and its habitat in the Pacific region.

It is of general interest in the field of applied ecology to determine whether a surrogate feature, like fisher resting sites, can predict the status of the population that relies on that feature (Lindenmayer et al., 2014; Pierson et al., 2015). Because resting habitat is a critical resource we hypothesize that if it is stable and sufficiently abundant, that – other things being equal – the *population* of fishers in the same area should also be stable, or at least not decreasing. This prediction is supported by 2 independent sources of fisher population monitoring data in the southern Sierra. The first is a large-scale carnivore monitoring program, based on annual estimates of occupancy at camera and track plate stations, that has been conducted for 14 years throughout our southern Sierra study area. In 2013 the first 8 years of these data were analyzed and published (Zielinski et al., 2013) and the occupancy estimates



**Fig. 3.** Predicted resting site suitability value for the FIA plots on southern Sierra and northwestern California study areas that were either burned by fire, harvested or had no apparent disturbance over the approximately 7 year period from 2003 to 2010. The “fire + harvest” condition occurred too rarely ( $\leq 3$  plots in each region) to include it in this analysis.

revealed no significant positive or negative trend and the population appeared stable. After this work was published 2 additional years were included but this did not change the conclusion that there was no significant trend (J. Baldwin, unpubl. data). Thus, our conclusion that

there has been no change in resting habitat is consistent with the results of the most extensive and longest duration measure of fisher population status in California.

The second source of information on population was a direct assessment of its growth rate from an intensively studied population on the Sierra National Forest (Sweitzer et al., 2015). Occupancy estimation has the shortcoming of being an indirect measure of population status, due to the unknown relationship of occupancy to measures of population growth ( $\lambda$ ) or density. Capture-Mark-Recapture (CMR; Williams et al., 2002) methods are the standard for estimating population growth and the vital rates that affect growth. Ideally, we would have direct population estimates derived from CMR from throughout our study areas, so that we could reconcile the population growth data with the information we have presented about trends in resting habitat. We are fortunate to have, however, the results of a comprehensive mark-recapture study conducted on a portion of our southern Sierra study area (Sweitzer et al., 2015). This study provides a second opportunity to determine if our conclusion that there has been no change in predicted resting habitat is consistent with estimates of population growth. Using radio-marked individuals and camera trapping, the authors estimated that  $\lambda$  for the population was indistinguishable from 0, which agreed also with their conclusion that there was no indication of a positive or negative trend in density. The labor-intensive nature of their work made it necessary to focus it on a part of a single forest (Sierra National Forest). Nonetheless, our estimate of stable resting habitat throughout the southern Sierra study area is consistent with the available estimates of fisher population growth. Of course factors other than resting habitat affect fisher population growth rates (e.g., predation pressure, foraging habitat, disease, poisoning), so stability in the RHS over time doesn’t guarantee that the fisher population will necessarily be spared negative effects nor enjoy positive effects. Something other than resting habitat can be limiting the population. Knowledge of the status of resting habitat is but one of the factors that can affect fisher population status. A stable or increasing trend in resting habitat is a necessary, but not sufficient, condition for determining the health of a fisher population.

**Table 2**

Mean values of the predictors in the selected fisher resting habitat model derived from the set of plots that had either received no disturbance (“none”) or were subjected to timber harvest (“harvest”) or fire or both. The  $\Delta$  represents the change in values for each of these important variables over an approximately 8 year period that occurred between the 2nd and 3rd vegetation sampling occasions (2003; 2011). These two sampling occasions were chosen for this analysis because the FIA sampling protocols were identical.

| Type of Disturbance | Southern Sierra         |                                                              |      |      |                                                     |       |       |                                |      |      |                                                        |      |      |                     |      |       |
|---------------------|-------------------------|--------------------------------------------------------------|------|------|-----------------------------------------------------|-------|-------|--------------------------------|------|------|--------------------------------------------------------|------|------|---------------------|------|-------|
|                     | N                       | BA_SMALL <sup>a</sup> (m <sup>2</sup> /ha <sup>-1</sup> )    |      |      | DBH_MAX <sup>b</sup> (cm)                           |       |       | DBH_MAX_SNAG <sup>c</sup> (cm) |      |      | DBH_HWD <sup>d</sup> (cm)                              |      |      | CC <sup>e</sup> (%) |      |       |
|                     |                         | T2                                                           | T3   | Δ    | T2                                                  | T3    | Δ     | T2                             | T3   | Δ    | T2                                                     | T3   | Δ    | T2                  | T3   | Δ     |
|                     |                         |                                                              |      |      |                                                     |       |       |                                |      |      |                                                        |      |      |                     |      |       |
| None                | 431                     | 14.8                                                         | 15.3 | 0.4  | 98.1                                                | 98.7  | 0.6   | 61.9                           | 63.5 | 1.6  | 10.6                                                   | 9.6  | -1.0 | 47.7                | 49.7 | 2.0   |
| Harvest             | 34                      | 15.9                                                         | 15.0 | -1.0 | 93.3                                                | 99.0  | 5.7   | 61.7                           | 56.8 | -4.9 | 11.8                                                   | 11.4 | -0.3 | 38.2                | 38.1 | -0.2  |
| Fire                | 30                      | 15.2                                                         | 5.6  | -9.6 | 93.7                                                | 68.0  | -25.7 | 47.3                           | 79.2 | 31.9 | 10.5                                                   | 10.8 | 0.3  | 55.1                | 14.7 | -40.4 |
| Fire + Harvest      | 2                       | 13.7                                                         | 9.0  | -4.7 | 123.6                                               | 115.2 | -8.4  | 76.2                           | 80.0 | 3.8  | 20.0                                                   | 18.6 | -1.4 | 55.0                | 31.1 | -23.9 |
| Type of Disturbance | Northwestern California |                                                              |      |      |                                                     |       |       |                                |      |      |                                                        |      |      |                     |      |       |
|                     | N                       | Hardwood BA <sup>f</sup> (m <sup>2</sup> /ha <sup>-1</sup> ) |      |      | BA <sup>g</sup> (m <sup>2</sup> /ha <sup>-1</sup> ) |       |       | AGE <sup>h</sup> (years)       |      |      | LG_WD <sup>i</sup> (m <sup>3</sup> /ha <sup>-1</sup> ) |      |      | CC (%)              |      |       |
|                     |                         | T2                                                           | T3   | Δ    | T2                                                  | T3    | Δ     | T2                             | T3   | Δ    | T2                                                     | T3   | Δ    | T2                  | T3   | Δ     |
|                     |                         |                                                              |      |      |                                                     |       |       |                                |      |      |                                                        |      |      |                     |      |       |
| None                | 483                     | 9.2                                                          | 10.0 | 0.7  | 35.2                                                | 37.2  | 1.9   | 123                            | 122  | -1   | 61.3                                                   | 64.2 | 2.9  | 57.7                | 58.1 | 0.5   |
| Harvest             | 12                      | 9.6                                                          | 6.7  | -2.9 | 42.8                                                | 32.7  | -10.1 | 149                            | 137  | -12  | 73.0                                                   | 68.2 | -4.8 | 58.9                | 36.8 | -22.1 |
| Fire                | 103                     | 6.9                                                          | 6.8  | -0.1 | 26.1                                                | 26.1  | 0     | 84                             | 79   | -5   | 32.4                                                   | 35.9 | 3.5  | 43.2                | 37.5 | -5.7  |
| Fire + Harvest      | 3                       | 9.7                                                          | 13.4 | 3.7  | 38.9                                                | 39.3  | 0.4   | 165                            | 133  | -32  | 38.5                                                   | 36.0 | -2.5 | 54.3                | 32.5 | -21.8 |

<sup>a</sup> Basal area of trees < 51 cm diameter at breast height (DBH).

<sup>b</sup> DBH of largest tree.

<sup>c</sup> DBH of largest conifer snag.

<sup>d</sup> DBH of hardwoods.

<sup>e</sup> Canopy closure.

<sup>f</sup> Basal area of hardwoods.

<sup>g</sup> Basal area.

<sup>h</sup> Age in years.

<sup>i</sup> Volume of large downed wood.



Examining the effects of disturbances suggests that fire had the most obvious and negative effect on RHS. We did not distinguish between types of fires (wild vs prescribed) in our analysis. This shortcoming should be remedied in future work because different types of fires have different ecological effects; some are prescribed to achieve restorative goals while others are uncontrolled wildfires. If a plot is affected by a severe wildfire, subsequent measurement would probably decrease the values for virtually all the predictor variables in each of the top models (Table 2), at least in the short term. This may explain why plots that were burned had substantial decreases in RHS. Sweitzer et al. (2016) also found that occupancy of sample units by fishers trended lower among those units that had been burned by either prescribed burning or wildfire. Nonetheless, the sum of their research did not identify a consistent negative effect of fire on fisher habitat use. Truex and Zielinski (2013) found that predicted resting habitat was significantly lower for a combination treatment of mechanical thinning plus fire but the controls didn't differ from the fire-only or the mechanical-only treatment. The lack of significant effects of fire was probably because the fire treatments in that study were from prescribed fire only.

Contrary to the substantial effect of fire on RHS, the values before and after harvest were indistinguishable from the values for the plots that had no disturbance. Future analyses will likely benefit from the larger sample sizes which, in turn, will allow us to characterize whether type of harvest, silvicultural prescription or ecological goal have different effects on RHS. That there was no effect of harvest on RHS was surprising but agrees with the results of Sweitzer et al. (2016) who also found no evidence that extractive management activities (i.e., timber harvest) contributed to reduced occupancy or local persistence of fishers. The lack of effect of harvest on RHS is most likely because few of the plots were affected by clearcutting, or other treatments that remove overstory, which would have driven the values of most of the variables very low. Instead, most of the plots where "harvest" occurred were actually understory thinnings that targeted the smaller trees in the stands, a practice commonly used to increase resilience to fire and accelerate the development of large trees and structural heterogeneity (e.g. North et al., 2009). These activities have become more common on public lands than commercial timber harvest (Christensen et al., 2016). Yet it is somewhat surprising that even light harvest would not have a more conspicuous effect because: (1) restorative fuel treatments had a negative effect on occupancy elsewhere (Sweitzer et al., 2016) and (2) the values for a number of predictors in the resting habitat models should certainly decrease on these plots after thinning (i.e., canopy closure, basal area of small trees, basal area, hardwood basal area).

Decreases in the values of these predictors may have been partially offset by the necessary post-harvest increase in one of variables: mean dbh of hardwoods (in southern Sierra model). We should be cautious, however, about our interpretations here because relatively few plots were subjected to harvest ( $n = 34$  and 12 plots in the southern Sierra and northwestern California regions, respectively) in the relatively short time period, and because others have found that fishers reduced their use of areas treated with restorative fuel reduction (Sweitzer et al., 2016). It will be interesting to see how collecting more data on plots that were harvested, using FIA inventory data collected in the future, will affect these tentative conclusions.

We also anticipate value in applying the fisher resting habitat model to assess the impact that the recent historic drought in California has had on forest habitat. The drought of 2012–2015, and induced water stress, led to significant mortality of small and large trees (Asner et al., 2016; Young et al., 2017). The effects of this mortality on forest structure and composition will be monitored by the USDA Forest Service over time using the FIA plot information. This information could also be used to predict values of habitat suitability via the fisher FIA-based model. The result would be an assessment of the: (1) effects of drought-induced tree mortality on resting habitat and (2) effects of how implementing management actions in drought-affected landscapes change the status of fisher resting habitat.

One of the shortcomings of our FIA plot-based approach is that the sample sizes are relatively small, especially for individual forests (e.g.  $n = 49$  in Eldorado NF) and for private lands (e.g.,  $n = 6$ ) in the southern Sierra region. This is the nature of the design of the FIA plot density and the area defined by the elevation and ecoregion criteria. However, what smaller samples sacrifice in resolution or comprehension is gained in efficiency, consistency and logistics because the FIA program appears to currently be reliably funded, enjoys popular support among foresters and planners, and produces high-quality and consistent data that are used for a diversity of monitoring applications, including carbon in moss and lichen ground layers, fire planning, changes in late-successional forest, and invasive plants (Gray, 2009; Jovan et al., 2015; Davis et al., 2015; Fried et al., 2017). Thus, the opportunity exists to develop and compare model estimates over time for multiple species of wildlife and plants.

Our approach, using model-based methods to monitor predicted suitability differs philosophically from the typical approach, whereby a researcher convinces the forest management and planning officials to adapt the researcher's predictive model and to measure the predictor variables originally identified by the research. Instead, we yield to the greater potential of using publically available vegetation data that are regularly collected for inventory purposes, at no additional cost to wildlife biologists, to estimate changes in specific habitat characteristics. Our work demonstrates the effectiveness of this approach in respect to elements of resting habitat for the fisher in California. We encourage others to exploit the FIA plot data for other species for which on-the-ground plot data are essential for reliable estimates of key habitat features.

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