

Fisher use of post-fire landscapes; implications for habitat restoration.

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Abstract

Background: The relationship between fishers, fire, and fuel management in the Sierra Nevada region is complex and highly controversial. While wildfire is a natural component of fisher ecology in the western region, it also has the capacity to destroy large swaths of suitable habitat and to fragment tenuous populations, particularly given recent shifts in wildfire scope and severity. Therefore balancing fuel reduction with habitat conservation requires careful consideration of spatial and temporal context, cumulative effects, and management prescriptions. One unresolved question is how and when fishers begin to re-use post-fire landscapes, and whether this information can be applied to post-fire restoration efforts in order to restore habitat connectivity and limit the negative impacts of population fragmentation. To better understand fisher response to post-fire landscapes and provide land managers

with guidance regarding habitat restoration efforts, we investigated fisher behavior in and around two recent Sierra Nevada wildfires. We hypothesized that fisher use of post-fire landscapes would be driven by one of four primary variables: pre-fire habitat quality, fire severity, topography, and/or time since fire. We developed *a priori* hypotheses regarding how fishers would begin to explore post-fire landscapes, and tested these using a combination of fine and coarse-scale analyses.

Results: At fine scales, the strongest relationship we observed included the combination of fire severity and landscape curvature on fisher scat location. At coarser scales, fisher colonization rates reflected avoidance of areas dominated by higher severity fire, and fishers had a higher probability of being found in areas with larger and more contiguous patches of low-severity fire.

Conclusions: Our results suggest that fisher use of post-fire landscapes centers on low severity or unburned islands (i.e. fire refugia / fire skips), and on fine scale topographic features associated with landscape concavity, such as ravines or topographic depressions. Efforts to promote a sustainable low-to mixed-severity fire regime that creates habitat heterogeneity and forest resiliency may provide a buffer against fisher population decline, and restoration efforts that capitalize on connecting fire refugia using micro-topography as a template may facilitate habitat connectivity more rapidly than traditional methods.

Keywords: fisher, *Pekania pennant*, fire, restoration

Background

The relationship between fishers (*Pekania pennanti*), fire, and fuel management in the Sierra Nevada region is complex and highly controversial (Collins et al. 2010, Truex and Zielinski 2013). On one hand,

fire is a natural process that maintains habitat heterogeneity and long-term resiliency, and creates critical fine-scale habitat elements such as snags and coarse woody debris (Logan et al. 2016). In contrast, current fires burning at the upper end or outside the range of natural variability remove large swaths of suitable habitat and consume more fine-scale habitat elements than they create. For example, the 2013 Rim Fire in Yosemite National Park and the Stanislaus National Forest burned over 107,000 acres of potential fisher habitat. And in 2015, the Rough Fire burned large areas of suitable habitat on the Sierra National Forest and adjacent Giant Sequoia National Monument. Examples such as these illuminate why in 2012, the U.S. Fish and Wildlife Service ranked uncharacteristically severe wildfire as one of the highest and most consistent threats to fisher persistence in the western United States (Naney et al. 2012).

Added to this conflict is that fisher habitat in the Sierras is found primarily in a narrow, north-south band along the western slope, bounded by elevation and periodically interrupted by bottlenecks created by large, steep, east-west oriented river canyons. Described as “habitat linkage areas” (Spencer et al. 2015), these bottlenecks restrict movement and genetic flow between larger areas of suitable habitat. Fires that occur in these canyons have the potential to fragment the population, resulting in population-level effects far greater than the direct result of habitat loss (Tucker et al. 2014, Spencer et al. 2015). In 2013, the Aspen Fire burned 22,000 acres on the southern slope of the San Joaquin River drainage, precisely where habitat modelling predicted that fishers were most likely to attempt crossing the drainage. The following year, the French Fire burned 14,000 acres on the northern side of the drainage, directly opposite the Aspen Fire. Taken together, the two fires effectively removed one of the six critical habitat linkage zones identified in a recent conservation strategy (Spencer et al. 2016). In a population estimated to contain less than 300 adult individuals (Spencer et al. 2011), large-scale habitat fragmentation such as this can greatly increase the impact of stochastic events and the risk of local extirpation (Wilcox and Murphy 2011, Fischer and Lindenmayer 2007).

Fuel reduction efforts can reduce wildfire intensity, limit extent, and protect critical structures, yet at the same time these activities come with short-term reductions in habitat quality (Scheller et al. 2011, Truex and Zielinski 2013, Hanson 2013, Sweitzer et al. 2016). Balancing fuel reduction with habitat conservation requires careful consideration of spatial and temporal context, cumulative effects, and management prescriptions (Lindenmayer et al. 2006), yet recent work suggests that the two objectives are not necessarily mutually exclusive (Zielinski et al. 2013, Truex and Zielinski 2013, Sweitzer et al. 2016, Thompson and Purcell 2016). Other authors have suggested that fishers are not negatively impacted by large wildland fires (Hanson 2013), and that greater harm is posed by mechanical thinning (Hanson 2015). One unresolved question is how and when fishers begin to re-use post-fire landscapes, and whether this information can be applied to post-fire restoration efforts in order to restore habitat connectivity and limit the negative impacts of population fragmentation.

To better understand fisher response to post-fire landscapes and provide land managers with guidance regarding habitat restoration efforts, we investigated fisher behavior in and around the French Fire (2014) and Aspen Fire (2013) landscapes for between one to three years following fire. We hypothesized that fisher use of post-fire landscapes would be driven by four primary variables: pre-fire habitat quality, fire severity, topography, and/or time since fire. We developed a suite of hypotheses regarding how fishers would begin to explore post-fire landscapes, and tested these using a combination of fine and coarse-scale analyses. We hypothesized that:

- 1) Fisher use of post-fire landscapes would be driven by a combination of distance within the fire and time-since-fire. Under this scenario, we predict fisher use would begin near the fire perimeter and gradually work inward. Fishers would not show a preference for patches of different fire severity.
- 2) Fishers would be motivated by a memory of pre-fire habitat conditions, and would seek to preferentially use and recolonize areas of high pre-fire habitat suitability.

3) In the absence of suitable vegetative habitat, fishers would rely on topographic variation to provide security while exploring and recolonizing post-fire landscapes. Under this scenario, fisher use would be associated with fine-scale topographic features such as ravines or drainages. Use and occupancy would be unaffected by fire severity.

4) Fishers would rely on unburned and lightly burned portions of the post-fire landscape to provide security while recolonizing burned areas. In this scenario, use and recolonization would focus on areas of lower burn severity. Unburned islands within the fire perimeter would serve as refuges, and use would be focused in their vicinity.

5) A combination of hypotheses 3 and 4, where fisher use of a post-fire landscape is guided by both topography and fire intensity.

Study Area

In the Southern Sierra Fisher Conservation Strategy (Spencer et al. 2016), the section of the San Joaquin River drainage directly below Mammoth Pool Dam is identified as Linkage Area D, a bottleneck of fisher habitat between two larger areas of suitable habitat (Fig. 1). In this area, the mixed conifer forest consists primarily of ponderosa pine (*Pinus ponderosa*), incense cedar (*Calocedrus decurrens*), and white fir (*Abies concolor*), interspersed with California black oak (*Quercus kelloggii*) and manzanita (*Arctostaphylos* spp). The western side of the drainage is relatively steep and rocky, with forested areas patchily distributed among a shrub-dominated landscape. Beyond the river canyon the forest becomes denser and includes additional species such as sugar pine (*Pinus lambertiana*) and mountain dogwood (*Cornus muttallii*). The eastern side of the drainage has a more moderate slope, and the forest cover is more contiguous, though relatively open in structure with a prominent hardwood component.

In July 2013, the Aspen Fire was ignited by lightning and burned 22,800 acres on the eastern side of the San Joaquin River drainage. In July 2014, an untended campfire started the French Fire, which eventually

burned 13,837 acres on the western side of the drainage, directly opposite the Aspen Fire (Fig. 1). The Aspen Fire was generally considered a mixed severity fire, with approximately 18%, 54% and 28% of the burned area burning at low, moderate, and high severity respectively. In contrast, nearly 55% of the French Fire burned at high severity and only 9% qualified as low severity according to Monitoring Trends in Burn Severity (MTBS) data (<http://www.mtbs.gov>).

Methods

Scat detection dog surveys

We conducted three scat detector dog surveys of the French and Aspen Fire landscape, in October 2015, May 2016, and November 2016. To distribute the survey effort consistently across the landscape, we identified twelve 10km² hexagonal sampling units. Hexagonal units included a range of burn severity, including unburned areas outside, but adjacent to, the fire perimeter. For each survey, an individual sampling unit was surveyed twice per survey by different dog/handler teams to minimize the effect of environmental and behavioral variability. Scat detection dog teams were provided by the University of Washington, Center for Conservation Biology's (UWCBC) Conservation Canines program.

Dog/handler teams were assigned a different sampling unit each day, and each team surveyed all units at least one time. Surveys began at a random location along a road. From that point, dogs were allowed to roam freely within the assigned 10 km² unit. Each dog carried a miniature GPS receiver that logged the survey team's location every 60 seconds. Due to the large number of mesocarnivores with overlapping diets in the region, and the risk of field misidentification, all scats collected were sent to the UWCBC genetics lab for species identification. In addition, due to the difficulty in reliably identifying fisher scats in the field, and the fact that rewarding a dog for locating a non-target scat risks teaching the dog to target that species, a rigorous daily training regime was followed. While conducting surveys,

handlers rewarded the dog only when the dog clearly indicated a target (fisher) scat. If the dog appeared hesitant, the sample was collected and tested but the dog was not rewarded. To further insure the dogs remained off of non-target species, additional training took place after each survey, with known fisher and non-target mesocarnivore scats to reinforce fisher as the target species.

To verify species ID, all scat samples were extracted in duplicate using a modified version of Qiagen's DNeasy tissue extraction kit, then amplified in duplicate with Qiagen's Multiplex PCR kit using a restriction fragment length polymorphism marker. Both positive and negative controls were used in amplification for quality control. All PCR products were run on an ABI 3730 Genetic Analyzer, then analyzed using GeneMarker software. The DNA extracts for confirmed fisher scats were further concentrated 2-fold using a Savant SpeedVac DNA Concentrator, and then duplicate extracts were PCR amplified four times using two previously developed and validated sex ID DNA markers targeting the ZFX/ZFY locus on the X- and Y- chromosomes (Statham et al. 2007) and the DBY locus on the Y-chromosome (Tucker et al. 2017). Both markers were amplified together in a 17 μ L multiplex reaction using Qiagen's Multiplex PCR kit. Fragments were separated by size using capillary electrophoresis on an ABI 3730 and then visualized and scored using SoftGenetics' GeneMarker software. Negative controls were used throughout each step of the process, and positive controls of known fisher DNA were amplified along with field samples.

Landscape data acquisition

Data on fire severity were obtained from the Monitoring Trends in Burn Severity (MTBS) program MTBS provides burn severity data in raster format, at a 30 m resolution, categorized into 4 severity classes defined by differences in the normalized burn ratio (dNBR): unburned to low, low, moderate, and high. While there are a number of different approaches to assessing fire severity, MTBS focuses on the impacts of fire on vegetation, particularly overstory (Eidenshink et al. 2007), which is highly relevant to

structure-dependent species such as fishers (Zielinski et al. 2004, Purcell et al. 2009). The spatial configuration of fire severity classes was analyzed using FRAGSTATS (version 4.1; McGarigal and Marks 1995). Spatial metrics calculated included cohesion and Shannon's diversity index at the landscape scale, and both percent of landscape and largest patch index for the four severity classes (Table 1).

Topographic variables were constructed using a 30m DEM and the DEM Surface Tools package for ArcGIS (Jenness Enterprises, 2013). Variables calculated included surface area and general landscape curvature. Surface area, calculated based on the estimated distance between the center of the target cell and the center of each of the 8 surrounding cells, is an indicator of topographic ruggedness and is limited to the estimated surface area of the target cell. Landscape curvature is based on an analysis of the relative position of 9 raster pixels, a target pixel and the 8 surrounding pixels. Negative values indicate a relatively concave arrangement such as a ravine, while positive values indicate a convex arrangement such as a ridge (Jenness 2013).

Animals may recall certain habitat conditions or associations in a post-disturbance landscape (Gautestad 2011, Gautestad et al. 2013), and therefore may exhibit behavior in response to structures that are no longer present on the landscape. Therefore, we included an assessment of pre-fire habitat quality, using a landscape-scale fisher habitat model developed by the Conservation Biology Institute (Spencer et al. 2011). The model is based on a combination of latitude-adjusted elevation, annual precipitation, and above-ground biomass of trees. This model accounted for 53% of the variation in fisher occupancy in a region-wide forest carnivore monitoring program (Spencer et al. 2011).

We calculated additional spatial variables, such as slope, distance to fire perimeter and distance to closest low-severity patch, using standard tools in ArcGIS v10.3 (ESRI, Redlands CA). Distance to fire perimeter was calculated based on how far (m) within the overall fire footprint a scat was located. Scats outside the fire perimeter received a distance score of 0. Low severity patches were defined as areas >

0.5 ha of contiguous unburned or low severity (MTBS code 1 & 2) within the fire perimeter. While subjective, this threshold allowed us to exclude small slivers and individual raster cells that were unlikely to influence fisher behavior. We examined correlations between variables, and when the correlation between two variables exceeded 0.6 one of the variables was excluded. Time since fire was calculated according to the number of months between the fire and the survey. This varied between the two fires, as the Aspen Fire burned in 2013 while the French Fire burned in 2014.

Fine-scale analysis

To characterize fisher use of post-fire landscapes at a fine spatial scale, we constructed a suite of resource selection functions using the glm package in RStudio (R Core Development Team 2011). The locations of genetically-verified fisher scats defined our set of used locations. Available locations, 25 per used location, were randomly placed within a 100m buffer along the detector dog survey routes in order to constrain the available set to areas actually surveyed. Distance-based metrics were square-root transformed to minimize the effect of skewness. Initially, we created seven univariate random effect models and ranked them using BIC, with fire (French or Aspen) as a random effect. We examined potential interactions between the top-ranked variable and all six remaining variables, then constructed bivariate models using the top-ranked variable and all six remaining variables, and again ranked the models according to BIC scores. This process of adding complexity was continued until no new models outranked the previously highest ranked model.

Mid-scale analysis

Detector dog surveys are most effective when the dogs are allowed to follow air currents and associated scents, but consequently this does not provide a precise characterization of space such as is provided by a fixed array of cameras or defined survey transects. We created this characterization by overlaying a finer grid (1 km² hexagon) over the original sampling unit array, and considering each 1 km² grid cell a

distinct sampling unit (Thompson et al. 2012). Pseudoreplication is a significant concern in the analysis of scat location data, as the deposition of scat by many animals is non-random and can be influenced by the presence of other individuals or territorial boundaries. Based on our experience working with fishers, we assumed that a 1km² resolution was small enough to capture landscape variability, yet large enough that a detection in one cell was unlikely to influence the probability of a detection in an adjacent cell. The detection of a scat within a grid cell identified that cell as occupied vs. unoccupied, additional scats or the presence of a latrine within a cell did not alter this.

We conducted an occupancy analysis using program PRESENCE (Hines 2006) with the 1km² grid cells as the sampling units and three sampling seasons. Because sampling intensity varied between cells, we used the length of the georeferenced tracklog to generate an estimate of survey effort. The length of a tracklog in a particular cell varied according to terrain and forest structure, and not every grid cell was surveyed every season depending on the route a dog chose to follow. Therefore, survey effort (length of the tracklog in a cell) was included as a covariate in the occupancy model associated with probability of detection. Because we were interested in the relative contribution of landscape variables in explaining the behavior of fishers in burned landscapes, we wanted to avoid inadvertently including uninformative parameters (Arnold 2009). We therefore screened variables for potential contribution by creating a set of univariate occupancy models and ranking these models based on AIC values. We also assessed correlations among variables using Pearson's correlation coefficient, and identified pairs of variables with correlations significantly different than zero (S1). Based on this information, we identified a subset of potentially informative variables related to our initial hypotheses, and conducted a structured all-subsets analysis while avoiding correlated pairings (Conner et al. 2018). Based on univariate model rankings and variable correlations, we included 5 variables in the development of multivariate occupancy models: 3 as coefficients on the occupancy estimate (percent low-intensity burn [%land_c2], cohesion of unburned land [cohesion_c1], time since fire [tsf]); 2 as coefficients on the colonization

estimate (percent high-severity burn [%high], largest patch index for low severity fire [LPI_c2]). We also considered the potential for detection probability to vary by survey, resulting in 40 potential models. The probability of extinction was assumed to be constant throughout the study. To evaluate the relative importance of each variable, we again ranked all models that outperformed the null model using AIC, and for each variable summed the weights of the models that included that variable.

Results

Across all three surveys, detector dogs located 157 scats in and around the French and Aspen fire landscapes (Figure 2). Of these, 49 (31%) were genetically confirmed as fisher. An additional 10 scats amplified as multiple species, including fisher and other mesocarnivores such as fox or skunk. This may reflect territorial over-marking, but these locations were excluded from analysis because we were unable to exclude the possibility of intraguild predation or scavenging. Forty-two scats failed to amplify, and the remaining scats included primarily grey fox (*Urocyon cinereoargenteus*) and ringtail (*Bassariscus astutus*).

Of the 49 confirmed fisher scats, 24 were confirmed to sex (14 male, 10 female). An additional 10 samples amplified according to sex (2 males, 8 females), but failed to meet the quality assurance standard for confirmation. The remaining 15 samples failed to amplify. While the sample was too small for statistical analysis, of the 24 scats where sex was confirmed, 9 were located within the burn perimeter. Seven of these were males while only 2 were female. Male scats were located an average of 1077m inside the fire perimeter (n=7), while the two female scats were located an average of 2070m inside the fire perimeter. Male scats were detected fairly evenly across all seasons (4 in fall 2015, 5 in spring 2016, and 4 in fall 2016), while 80% of female fisher scats were detected in spring 2016.

Fine-scale Habitat Selection. Univariate RSF models indicated that of the seven landscape variables considered, fisher scat location was most strongly associated with burn severity, indicating an avoidance of high and moderate severity classes (Table 2). No other models, including interaction terms, were competitive. Bi-variate models indicated that an additive model based on burn severity and landscape curvature outperformed the burn severity model by 11.4 BIC points (Table 2). No tri-variate model outperformed the burn severity + landscape curvature model.

Mid-scale Habitat Selection. At the mid-scale, the highest performing univariate model consisted of a colonization estimate based on the percent of high severity fire within the target cell. This model outperformed the null model by 5.32 AIC points. Adding the percentage of low-intensity fire as a covariate on the occupancy estimate improved the AIC value slightly ($\Delta AIC = -0.19$) (Table 3). The only other competitive multivariate model consisted of the percent high severity fire as a coefficient on colonization and the cohesion of unburned pixels as a coefficient on occupancy ($\Delta AIC = 0.22$). Relaxing the assumption of a consistent probability of detection resulted in the creation of eight additional competitive models (Table 3). In all top models, the consistent feature was a negative association between the percentage of high severity fire within a grid cell and the probability of fisher colonization. Comparable AIC values (<2 AIC points) resulted from a variety of occupancy coefficients: fisher scats were located in cells with higher percentages of low intensity fire, higher cohesion of unburned land, and larger patches of low intensity fire. However the overall top model indicated that the probability of occupancy increased in relation to the time since fire; this model outperformed a null model assuming variable detection probability by 4.16 AIC points.

Discussion

Little published literature exists on how mesocarnivores respond to both wildland and prescribed fire (Wasserman 2015, Fisher and Wilkinson 2005). What does exist focuses on generalist species and suggests a wide variety of responses depending on the physiology and ecological relationships of the focal species. Foxes have been observed more often than expected using burned landscapes (Schutte 2007, Thompson et al. 2008, Borchert 2012), though post-fire declines have also been documented (Cunningham et al. 2006). Coyote use of post-fire landscapes has been observed to increase (Schutte et al. 2014), decrease (Schutte 2007, Borchert 2012), and remain constant (Cunningham et al. 2006). Lindenmayer et al. (2008) observed a mixed response to wildland fire by Australian mesocarnivores; long-nosed bandicoot (*Perameles nasuta*) and common brushtail possum (*Trichosurus vulpecula*) detections increased while detections of common ringtail possums (*Pseudocheirus peregrinus*) decreased. Extensive scavenging of fire-killed large mammals by large carnivores has been documented (Singer et al. 1989, Blanchard and Knight 1990), and it is logical to assume that mesocarnivores may exploit a similar bounty following higher-severity fires.

The five potential hypotheses we postulated each suggest a different combination of landscape variables as primary influences on fisher scat locations. The majority of the fisher scats we located were found around the perimeter of the fires, indicating that fishers remained in the surrounding forest and began to penetrate the burned landscape 2-4 years after the fires occurred. The fact that time since fire emerged in one of the top models indicates that fisher exploration of burned landscapes is a gradual process. Despite the fact that distance to fireline was not a strong predictor of occupancy or colonization, it was correlated with time since fire (Pearson's correlation coefficient 0.625, $R^2 = 0.39$, $P = 0.0006$), indicating that fishers continued to penetrate deeper into the fire interior over time. Ten of 13 scats found within the fire perimeter were detected ≥ 36 months post-fire. In particular, a number of scats were located well within the northern portion of the Aspen Fire, an area that burned at a mixed severity.

At fine scales, the strongest relationship we observed included the combination of fire severity and landscape curvature on fisher scat location. Fisher scats were more often located in areas of lower severity fire and in areas of concave topography such as ravines or canyon bottoms. At coarser scales, fisher colonization rates reflected avoidance of areas dominated by higher severity fire, and fishers had a higher probability of being found in areas with larger and more contiguous patches of low-severity fire. Variables such as pre-fire habitat quality and distance to fireline did not appear to influence fisher scat locations in a noticeable way. Based on the pattern of relationships we observed, fishers appeared to recolonize burned landscape in accordance with our fifth hypothesis, utilizing a combination of fine-scale topography and unburned or lightly burned patches within the fire perimeter. Similar behavior has been reported for black bears (Cunningham et al. 2003) and lynx (Vanbianchi et al. 2017). Cunningham et al. (2003) found that bears relied on unburned islands within the perimeter of large fires for bedding and feeding sites. Similarly, Vanbianchi et al. (2017) reported that lynx in northern Washington used burned areas within 1 year of the Tripod Fire, however they avoided areas of high severity fire and focused their activity within the fire perimeter on unburned or lightly burned islands (fire skips) with residual trees and high canopy cover.

It is worth noting that our evaluation of fisher response to pre-fire habitat quality and distance to fireline are both based on assumptions. We did not attempt to determine individual ID or age for detected fishers, therefore we are assuming that the animals we detected came from the local area and were potentially familiar with the pre-fire landscape. If these animals were juveniles or dispersers from outside the local area, our rejection of the idea that they would respond to pre-fire habitat quality would be invalid. Furthermore, firelines are rarely as distinct on the landscape as they are on a map. They are often better characterized as a patchwork or gradient of fire severity, however our analysis assumed that fishers observed the fire edge as we defined it. Not a likely scenario, but in the absence of better data a necessary assumption.

Acquiring empirical data on the response of a secretive, highly mobile forest carnivore such as fishers to unpredictable events such as wildfire is extremely difficult. To our knowledge, only one other study has attempted this. Hanson (2013) used detector dogs to survey the McNally and Manter fire footprints on the Sequoia National Forest for fisher scat 10 years post-fire. The author reported detecting fisher scats in both post-fire and unburned areas, and that fishers selected areas that consisted of closed canopy, mature/old forest prior to the fire at the same level as similar, unburned areas. Based on this evidence, the author concluded that mixed-severity and high-severity fire did not represent a loss of habitat for fishers, and that fishers may select a combination of burned/unburned areas for foraging and resting, respectively.

Our results differed from Hanson (2013) in several ways. We found no evidence of a relationship between fisher scat detection and pre-fire habitat quality. Fishers on the Aspen/French Fire landscapes appeared to avoid areas of high and moderate fire severity, and instead used islands of unburned forest or low-intensity patches to facilitate movement within the fire perimeter. They also appeared to use fine scale topographic features as corridors in the post-fire landscape, which likely provide some degree of security in an open, post-fire landscape and may have provided refuge for small mammals during the fire. Only one fisher scat was located in an area classified as high intensity fire, and this scat was located in a small ravine under a spanning log. One area of agreement between the two studies may be the idea that fishers' willingness to use a post-fire landscape increases over time. Although in all our surveys the majority of scats were located outside the fire perimeter, the detection of scats deeper within the fire perimeter increased in later surveys. This corresponds with Hanson (2013) report of fisher scats within the McNally and Manter fire perimeters 10+ years post-fire. Similarly, Vanbianchi et al. (2017) reported that lynx activity within the Tripod Fire was primarily limited to within 1 km of the fire perimeter during the first year post-fire. In the Sierra Nevada region, it is common for shrubs to respond strongly and dominate post-fire landscapes that burned and moderate to high severity (Collins and Roller 2013). As

shrubs develop, they provide vegetative cover and likely increase a fisher's willingness to explore the post-fire landscape.

Pseudoreplication is a significant concern when using scats to indicate species presence or activity, as the placement of scats is autocorrelated and the associated clustering can lead to biased results (Thompson et al. 2012). This problem is particularly acute with small sample sizes and with species such as fishers that use latrines and mark territorial boundaries with scat. Instead, when evaluating scat data without individual genetic information on individuals, it is necessary to identify some independent spatial sampling unit appropriate for the species (e.g. a transect, grid cell, etc.) and then to examine occupancy or abundance within that unit (Thompson et al. 2011, Long et al. 2007, Smith et al. 2005). Hanson (2013) relied on a one-tailed chi-square analysis to assess habitat selection without accounting for autocorrelation or multiple comparisons, and in a subsequent publication stated that the absence of statistical significance supported the conclusion of no difference in selection of burned and unburned habitat (Hanson 2015), as opposed to a failure to reject a null hypothesis of differential selection (Anderson et al. 2001).

Issues are further confounded by Hanson's (2013 and 2015) choice of fire severity classes. Hanson (2013) defines low-severity fire as areas with less than 15% basal area mortality (<316 RdNBR), moderate-severity fire as areas with 15% to 50% basal area mortality (316 – 477 RdNBR), and higher-severity fire as areas with greater than 50% basal area mortality (> 477 RdNBR). Management agencies, on the other hand, generally define low severity fire as less than 25% basal area mortality, moderate as 25% to 75% basal area mortality, and high as greater than 75% mortality. Furthermore, to account for small sample sizes, the author combined the 'moderate' and 'higher' severity fire categories into a single broad category, effectively biasing subsequent interpretation toward higher severity fire (Fule et al. 2014). While there is no precise agreed upon threshold of high severity fire, Safford et al. (2008) recommended a 75% basal area mortality threshold, given that the most commonly used thresholds in

the literature range from 70% to 80%. Regardless, the use of common standards and definitions across studies will facilitate comparisons and further our understanding of wildlife response to fire.

Similar to Hanson (2013), we did document both male and female fisher activity within the fire perimeter. The fact that males were detected in all survey efforts, while females were detected overwhelmingly during the spring suggests a behavioral component to females' use of burned landscapes. Our surveys were conducted during the denning season when adult female fishers have dependent young and are tied to a single location. So it is possible that female fishers denning near the fire perimeter have limited foraging options, and may forage in areas they might avoid the rest of the year. The limited female fisher activity within the fire perimeter may suggest that given their full range of foraging options, females largely avoid recently burned landscapes.

Better context for our results may come from a telemetry study of marten response to post-fire series fire in the Alaskan taiga (Paragi et al. 1996). The authors reported that marten abundance and activity was highest within a recent (<10yr) burned landscape, which contained lower canopy cover but higher coarse woody debris than older burn or unburned landscapes. However, they also reported high population turnover and that the vast majority of animals using the burned landscape were juveniles. They reported higher small mammal biomass and diversity within the burned landscape, and expected to find higher use of habitat near the fire perimeter where martens could rest in unburned habitat and forage within the burned area. Instead, they reported very low use of edge habitat. The authors concluded that their results supported the idea that post-fire habitat lacked the conditions necessary for marten reproduction and instead acted as a population sink. Dispersing animals from nearby mature forest were attracted to the post-fire landscape due to unoccupied territories and high prey density, however these animals frequently died before reproducing. Similar dynamics have been observed in other boreal marten populations exposed to both fire (Vernam 1987, Latour 1994) and intensive logging (Chapin et al. 1998).

Martens and fishers display significant overlap in their habitat selection and behavior, so the evidence above suggests that fisher use of post-fire landscapes should be interpreted carefully. Without demographic data, it is difficult to know whether fisher use of post-fire landscapes indicates a resident or transient population or whether species like fisher may experience higher mortality in post-fire landscapes. At the same time, some authors have suggested that a landscape mosaic created and supported by a mixed-severity fire regime could provide a diversity of habitat and prey while protecting against large habitat losses associated with large or uncharacteristically severe disturbances (Kohler and Hornocker 1977).

Our results suggest that fisher use of post-fire landscapes centers on low severity or unburned islands (i.e. fire refugia / fire skips), and on fine scale topographic features associated with landscape concavity, such as ravines or topographic depressions. These results may be linked, as landscape concavity is often associated with water accumulation and therefore may represent landscape patches that are less likely to burn, quicker to recover some form of vegetative cover, and may contain remnant prey populations. Where possible, fishers may use unburned islands to traverse burned landscapes, and in the absence of vegetative cover may use topography as a surrogate. Fisher and Wilkinson (2005) write that “These structural features [fire skips], where examined, were revealed as important predictors of mammalian use of burned and harvested stands for all mammal species groups,” and suggests a “more explicit examination of the influence of live residual trees and their influence on recolonization and persistence of mammal species post disturbance is required.”

Our results differed from the one other study that, to our knowledge, attempted to quantify fisher use of post-fire landscapes in the Sierra Nevada. These differences may have stemmed from the amount of time after the fire that sampling was conducted. Hanson (2013 and 2015) sampled >10 years post-fire, when vegetation recovery may have provided greater cover and structure. In contrast, our sampling occurred in years 2-4 following fire, when only limited vegetation recovery had occurred. Of particular

concern is Paragi et al.'s (1996) conclusion that post-fire habitat may not serve as breeding habitat for martens. Dispersing fishers may be similarly drawn toward landscapes with increased prey availability, however without key habitat elements such as resting and denning structures, post-burn habitat may not support population health, at least in the short term. Efforts to promote a sustainable low- to mixed-severity fire regime that creates habitat heterogeneity and forest resiliency may provide a buffer against population decline. In addition, connecting unburned islands within the fire perimeter and using micro-topography as a template may present opportunities for post-fire restoration to more rapidly promote habitat connectivity.

Declarations

Ethics approval and consent to participate: All animal capture and handling was conducted under the guidelines of the American Society of Mammologists (Sikes et al. 2016); procedures were reviewed and approved by the California Department of Fish and Wildlife (permit #SC-2730) and the Institutional Animal Care and Use Committee, University of California at Davis.

Consent for publication: The authors consent to publication of all details and images contained herein.

Availability of data and material: All datasets used and/or analyzed during the current study are available from the corresponding author on reasonable request.

Competing interests: The authors declare that they have no competing interest.

Funding: Funding for this work was provided by the USDA Forest Service, Region 5 and the Pacific Southwest Research Station. In-kind contribution was provided by the University of Washington's Center for Conservation Biology and Conservation Canines.

Authors' contributions: CT coordinated the research effort, conducted analyses, and was the primary writer of the manuscript. KP acquired funding, assisted with analyses, and was a major contributor to editing the manuscript. RG oversaw the field aspects of the research effort, and assisted with writing and editing the manuscript. HS managed the data collection and contributed to the methodology section of the manuscript.

Acknowledgements: Genetic analyses were conducted by Rebecca Booth at the University of Washington's Center for Conservation Biology. This research effort would also not have been possible without the energy, skill, and dedication of Pips and Winnie, two conservation canines from the UW Conservation Canine Program, as well as their human assistants, Jennifer Hartman, Caleb Stanek, and Suzie Marlow.

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572 Figure 1. Location of the French and Aspen fires on opposite sides of the San Joaquin River in the Sierra
573 National Forest, CA (left). The Aspen fire occurred on the eastern side of the San Joaquin River drainage
574 in 2013, the French fire occurred on the west side in 2014. Upper right – Burn severity based on
575 Monitoring Trends in Burn Severity (MTBS) for the French & Aspen fires. Lower right – landscape
576 curvature for the French/Aspen fire landscape.

577

578 Figure 2. Results of scat detector dog surveys for fisher presence within the French and Aspen fire
579 landscapes on the Sierra National Forest, CA, 2015-2016. Green to red shading indicates fire severity,
580 dark outlines indicate 10km² survey hexagons. Red dots indicate the locations where fisher scats were
581 collected, white dots indicate the random locations chosen for fine-scale RSF analysis.

582

583 Supplemental Data S1: Correlations between landscape-scale variables.

Table 1. Landscape variables used in resource selection (fine-scale) and occupancy (mid-scale) analyses of post-fire fisher scat locations in the Sierra National Forest, CA.

Landscape variable description	Fine-scale analysis	Mid-scale analysis	Variable name
Burn severity (unburned [C1], low [C2], moderate [C3], & high severity [C4])	X		MTBS
Spatial metrics of burn severity patches			
→ Shannon's diversity index		X	SHDI
→ Largest patch index (4 classes)		X	LPI- C1, C2, C3, C4
→ Cohesion (5 classes)		X	Cohesion L, C2, C3, C4
→ Percent landscape (4 classes)		X	%land C1, C2, C3, C4
Surface Area	X		surface area
Landscape curvature	X		curvature
Pre-fire habitat quality	X	X	Land08
Slope	X		slope
Distance to fire perimeter (m)	X	X	Dist to fireline
Distance to low severity or unburned patch (m)	X		Dist to unburned patch
Time since fire (months)		X	tsf

Table 2. Univariate, bivariate and tri-variate resource selection models indicating the fine-scale relationship between fisher locations and landscape metrics on the French and Aspen Fire landscapes in the Sierra National Forest, CA. Δ BIC values are calculated in relation to the best overall performing model overall.

Univariate RSF models		BIC	Δ BIC	<i>P</i> (var1)		
	MTBS	381.9	11.4	< 0.0001		
	Dist to unburned patch	403.5	33.0	< 0.001		
	Curvature	415.2	44.7	< 0.0001		
	Dist to fireline	418.0	47.5	< 0.0001		
	Slope	425.8	55.3	< 0.01		
	Surface area	426.5	56.0	< 0.01		
	Land08	434.5	64.0	0.960		
Interactions		BIC	Δ BIC	<i>P</i> (MTBS)	<i>P</i> (var 2)	<i>P</i> (MTBS*var2)
	MTBS * curvature	377.0		< 0.0001	0.007	0.428
	MTBS * surface area	389.8		0.412	0.032	0.309
	MTBS * slope	390.7		0.155	0.047	0.312
	MTBS * Land08	393.1		0.002	0.352	0.156
	MTBS * dist to fireline	390.6		0.015	0.628	0.348
	MTBS * dist to unburned patch	395.3		< 0.001	0.345	0.321
Bivariate RSF models		BIC	Δ BIC	<i>P</i> (MTBS)	<i>P</i> (var 2)	
	MTBS + curvature	370.5	0.00	< 0.0001	< 0.0001	
	MTBS + surface area	383.9	13.4	< 0.0001	0.014	
	MTBS + dist to fireline	384.4	13.9	< 0.0001	0.032	
	MTBS + slope	384.6	14.1	< 0.0001	0.032	
	MTBS + Land08	388.2	17.7	< 0.0001	0.365	
	MTBS + dist to unburned patch	389.0	18.5	0.0004	0.828	
Tri-variate RSF models		BIC	Δ BIC	<i>P</i> (MTBS)	<i>P</i> (curvature)	<i>P</i> (var 3)
	MTBS + curvature + surface area	373.7	3.2	< 0.0001	< 0.0001	0.03
	MTBS + curvature + slope	373.7	3.2	< 0.0001	< 0.0001	0.044
	MTBS + curvature + Land08	376.4	5.9	< 0.0001	< 0.0001	0.274
	MTBS + curvature + dist to fireline	373.5	3.0	< 0.0001	< 0.0001	0.045
	MTBS + curvature + dist to unburned	377.6	7.1	0.001	< 0.0001	0.768

Table 3. Univariate and bivariate occupancy models indicating the mid-scale relationship between fisher locations and landscape metrics on the French and Aspen Fire landscapes in the Sierra National Forest, CA. Coefficients include occupancy (psi), colonization (gamma), extinction (eps), and detection (p). Only models within 5 AIC points of the top models, either within group or overall, are reported.

Univariate occupancy models		AIC	w/in group ΔAIC	Overall ΔAIC
	psi(), gamma(%high), eps(), p(effort)	157.39	0.00	1.72
	psi(), gamma(LPI_c2), eps(), p(effort)	160.26	2.87	4.59
	psi(), gamma(%land_c2), eps(), p(effort)	160.45	3.06	4.78
	psi(), gamma(cohesion_c2), eps(), p(effort)	160.56	3.17	4.89
	psi(), gamma(), eps(), p(effort) [NULL]	162.71	5.32	7.04
Multivariate occupancy models				
	psi(%land_c2), gamma(%high), eps(), p(effort)	157.20	0.00	1.53
	psi(cohesion_c1), gamma(%high), eps(), p(effort)	157.42	0.22	1.75
	psi(%land_c2+ cohesion_c1), gamma(%high), eps(), p(effort)	157.94	0.74	2.27
	psi(tsf), gamma(%high), eps(), p(effort)	158.07	0.87	2.40
	psi(%land_c2 + tsf), gamma(%high), eps(), p(effort)	158.64	1.44	2.97
	psi(cohesion_c1 + tsf), gamma(%high), eps(), p(effort)	160.20	3.00	4.53
	psi(%land_c2), gamma(LPI_c2), eps(), p(effort)	160.27	3.07	4.60
	psi(%land_c2+ cohesion_c1), gamma(LPI_c2), eps(), p(effort)	160.49	3.29	4.82
	psi(tsf), gamma(LPI_c2), eps(), p(effort)	160.96	3.76	5.29
	psi(cohesion_c1 + tsf), gamma(LPI_c2), eps(), p(effort)	161.54	4.34	5.87
Variable detection probability				
	psi(tsf), gamma(%high), eps(), p(effort*survey)	155.67	0.00	0.00
	psi(%land_c2), gamma(%high), eps(), p(effort*survey)	155.80	0.13	0.13
	psi(cohesion_c1), gamma(%high), eps(), p(effort*survey)	155.80	0.13	0.13
	psi(lpi_c2), gamma(%high), eps(), p(effort*survey)	156.23	0.56	0.56
	psi(tsf + %land_c2), gamma(%high), eps(), p(effort*survey)	156.81	1.14	1.14
	psi(), gamma(%high), eps(), p(effort*survey)	157.08	1.41	1.41
	psi(cohesion_c1), gamma (%high), eps(), p(effort * survey)	157.42	1.75	1.75
	psi(cohesion_c1 + %land_c2), gamma(%high), eps(), p(effort* survey)	157.63	1.96	1.96
	psi(), gamma(), eps(), p(effort*survey) [NULL]	159.83	4.16	4.16
	psi(cohesion_c1 + tsf), gamma(%high), eps(), p(effort * survey)	159.98	4.31	4.31

DRAFT

Figure 1 DRAFT

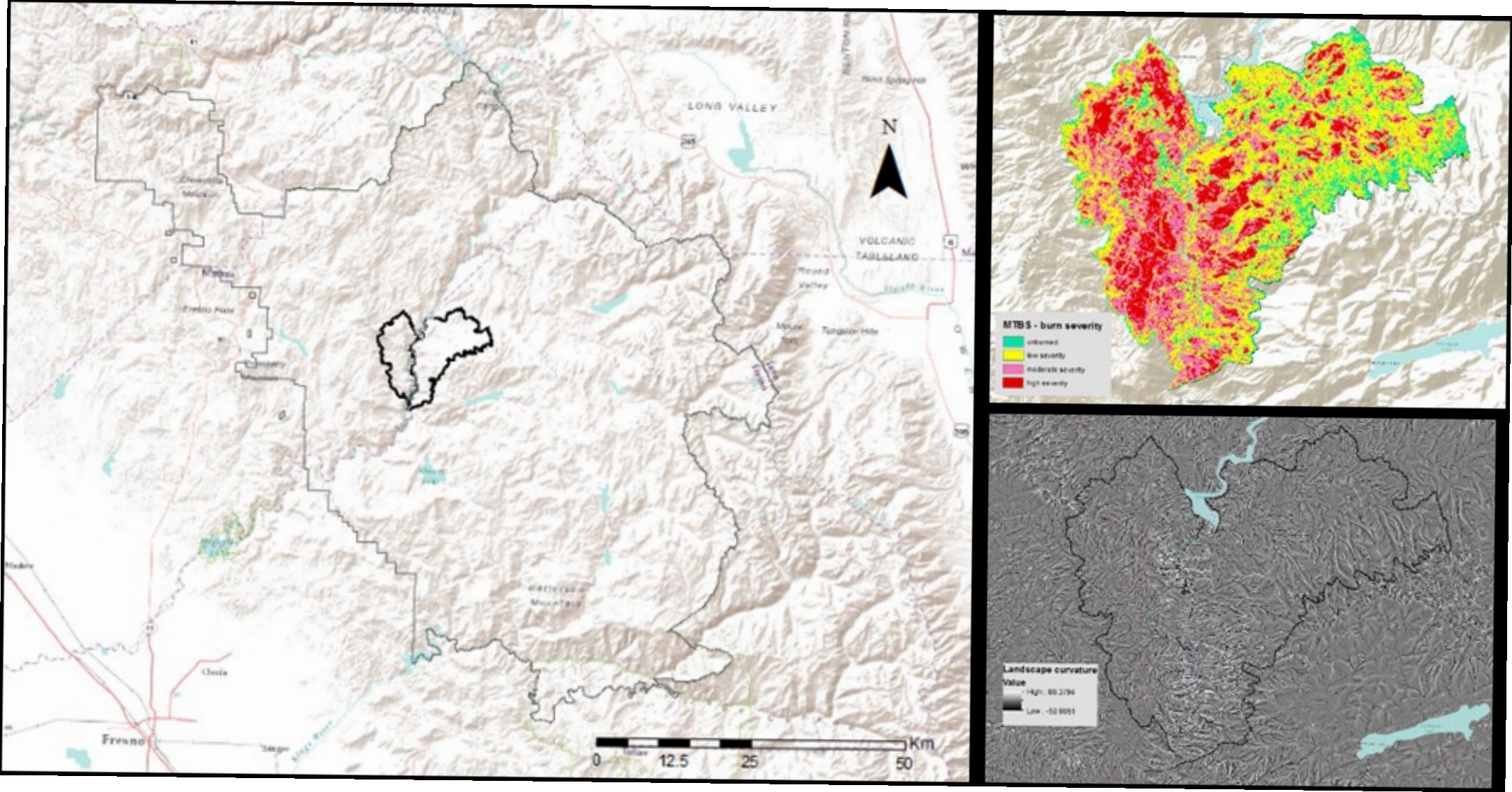
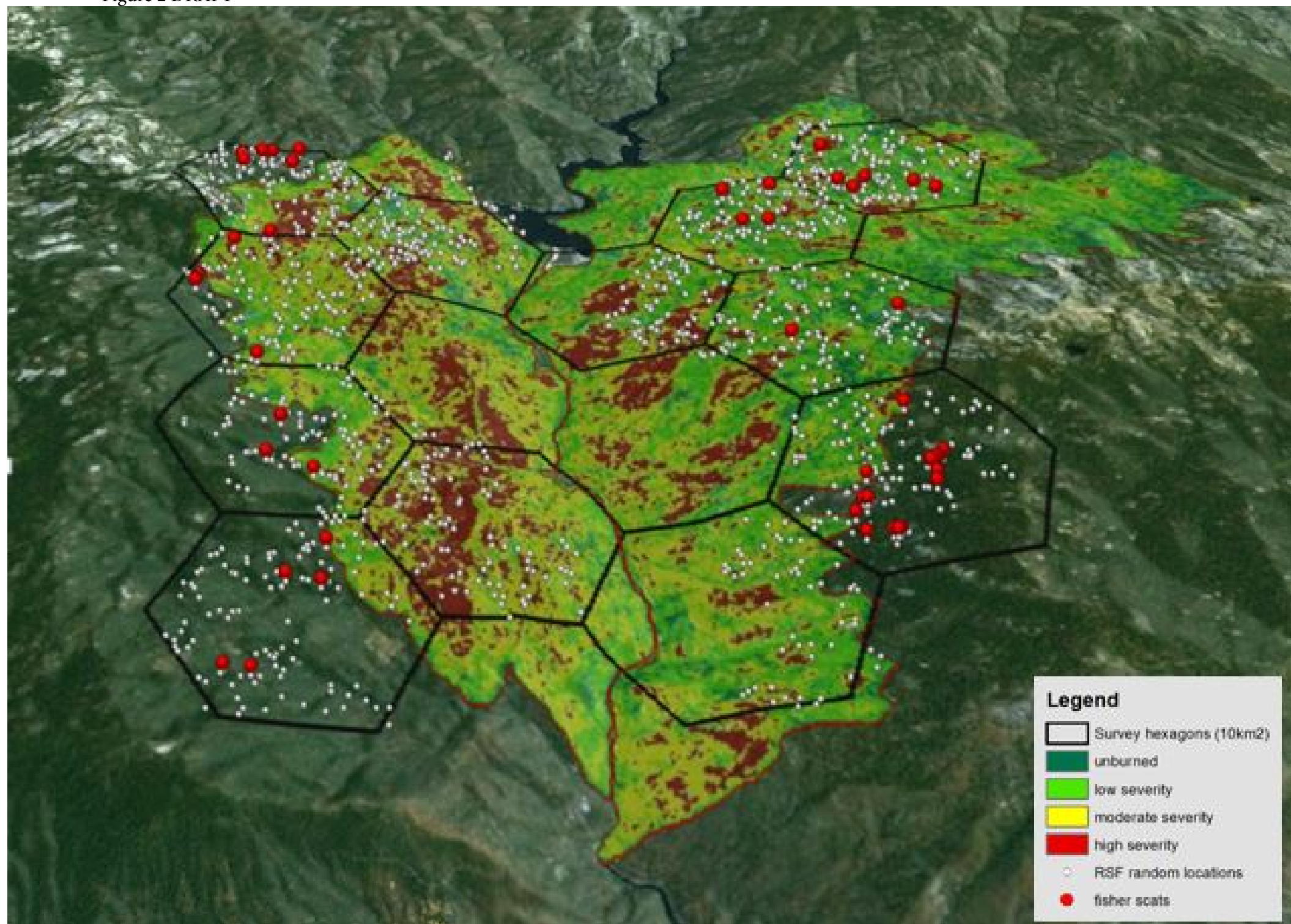


Figure 2 DRAFT



Supplemental APP1 DRAFT

	COHESION_C1	COHESION_C2	COHESION_C3	COHESION_C4	COHESION_L	Dist to fireline	Land08	LPI_C1	LPI_C2	LPI_C3	LPI_C4	%LAND_C1	%LAND_C2	%LAND_C3	%LAND_C4	SHDI_L	%high
COHESION_C1	1																
COHESION_C2	0.75	1															
COHESION_C3	0.36	0.74	1														
COHESION_C4	0.06	0.46	0.79	1													
COHESION_L	-0.56	-0.74	-0.71	-0.61	1												
Dist to fireline	0.09	0.4	0.55	0.6	-0.43	1											
Land08	-0.17	-0.15	-0.06	0.07	0.09	-0.03	1										
LPI_C1	0.57	0.28	-0.09	-0.23	-0.16	-0.12	-0.24	1									
LPI_C2	0.42	0.59	0.3	0.16	-0.27	0.18	0.07	0.21	1								
LPI_C3	-0.18	0.14	0.59	0.57	-0.16	0.39	-0.12	-0.28	-0.27	1							
LPI_C4	-0.22	0.01	0.34	0.59	-0.21	0.36	-0.03	-0.25	-0.32	0.3	1						
%LAND_C1	0.66	0.45	0.04	-0.17	-0.26	-0.03	-0.2	0.86	0.54	-0.35	-0.35	1					
%LAND_C2	0.47	0.67	0.43	0.27	-0.4	0.25	0.06	0.25	0.96	-0.12	-0.3	0.56	1				
%LAND_C3	-0.1	0.31	0.74	0.73	-0.38	0.55	-0.11	-0.31	-0.06	0.92	0.36	-0.31	0.09	1			
%LAND_C4	-0.25	0.05	0.45	0.7	-0.31	0.44	0.01	-0.3	-0.33	0.46	0.94	-0.39	-0.27	0.53	1		
SHDI_L	0.61	0.84	0.83	0.7	-0.87	0.44	-0.1	0.23	0.44	0.32	0.18	0.35	0.56	0.51	0.27	1	
%high	-0.21	0.05	0.49	0.62	-0.28	0.43	-0.17	-0.27	-0.41	0.66	0.77	-0.37	-0.31	0.68	0.85	0.29	1
%high+mod	-0.15	0.24	0.67	0.82	-0.45	0.62	-0.06	-0.3	-0.15	0.71	0.73	-0.32	-0.03	0.81	0.84	0.47	0.9