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Research

Wildfire and Spatial Patterns in Forests in Northwestern Mexico: The United States Wishes It Had Similar Fire Problems

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Erratum: Some stand-level metrics reported in this paper are incorrect because the wrong expansion factor for expanding plot-level measurements was used in the calculations. The attached erratum explains and gives the revised metrics.

ABSTRACT. Knowledge of the ecological effect of wildfire is important to resource managers, especially from forests in which past anthropogenic influences, e.g., fire suppression and timber harvesting, have been limited. Changes to forest structure and regeneration patterns were documented in a relatively unique old-growth Jeffrey pine-mixed conifer forest in northwestern Mexico after a July 2003 wildfire. This forested area has never been harvested and fire suppression did not begin until the 1970s. Fire effects were moderate especially considering that the wildfire occurred at the end of a severe, multi-year (1999-2003) drought. Shrub consumption was an important factor in tree mortality and the dominance of Jeffrey pine increased after fire. The Baja California wildfire enhanced or maintained a patchy forest structure; similar spatial heterogeneity should be included in US forest restoration plans. Most US forest restoration plans include thinning from below to separate tree crowns and attain a narrow range for residual basal area/ha. This essentially produces uniform forest conditions over broad areas that are in strong contrast to the resilient forests in northern Baja California. In addition to producing more spatial heterogeneity in restoration plans of forests that once experienced frequent, low-moderate intensity fire regimes, increased use of US wildfire management options such as wildland fire use as well as appropriate management responses to non-natural ignitions could also be implemented at broader spatial scales to increase the amount of burning in western US forests.

Key Words: *Baja California; forest resistance; forest structure; Jeffrey pine; mixed conifer; ponderosa pine; regeneration; resilience; Sierra San Pedro Martir; spatial heterogeneity.*

INTRODUCTION

Public concern over wildfire has eclipsed that for other forest values in the United States (US), and this trend will probably continue for the next few decades. Wildfire area in the US has exceeded 3,200,000 ha in three of the last four years (NIFC 2007), and annual fire suppression costs are now commonly over $1×10^6$ (Stephens and Ruth 2005, USDA 2006). Nearly 50% of the US Forest Service annual budget is used for fire suppression and this severely affects unrelated forest management, recreation, and research programs.

Fire exclusion and past harvesting have increased surface and ladder fuels, particularly in those US forests that once experienced frequent, low-moderate severity fire regimes, resulting in many forested areas that are vulnerable to uncharacteristically severe fires (Agee and Skinner 2005, Stephens and

Ruth 2005). Further increases in fire behavior and extent may occur from a predicted lengthening of the fire season due to global warming (McKenzie et al. 2004, Westerling et al. 2006).

With the loss of Native American ignitions and suppression of lightning fires many conifer forests in the western US have experienced a 100-yr period without fire, i.e., a fire-free period that is unprecedented over at least the last two millennia (Swetnam 1993). This and past harvesting has resulted in important changes in forests including diminished reproduction of shade-intolerant species and more area dominated by dense, intermediate-aged forest patches. Because of these adverse impacts on many conifer forests, uses of old-growth conifer forests as reference points have become important to investigate ecosystems characteristics such as regeneration. Previous research has estimated the size of regeneration patches in old-

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growth ponderosa pine (*Pinus ponderosa*) and mixed conifer forests in western North America that had been subjected to decades of fire exclusion (Cooper 1960, 1961, West 1969, Bonnicksen and Stone 1981, White 1985, Moore et al. 1993, Fulé et al. 2002). Average patch size varied from 0.07-0.26 ha (Stephens and Fry 2005), but none of these studies investigated how recent fire affected spatial patterns of trees and seedlings, which is the focus of this work.

The US is presently in a fire paradox. We continue to allocate more resources to suppress fires, yet annual area burned continues to rise, particularly in the western US (Stephens 2005, Donovan and Brown 2007, NIFC 2007). Can the US learn from other areas of the world with different fire management practices? To address this question we investigated wildfire effects on structural characteristics in an old-growth Jeffrey pine (*Pinus jeffreyi*)-mixed conifer forest in the Sierra San Pedro Martir (SSPM), northwestern Mexico, and compared them to similar forests in southern California with different management practices. Specifically, our objectives were to 1) determine what changes occurred to forest structure following a wildfire in July 2003, the first wildfire in this forest in over a decade; 2) identify what factors were related to tree mortality; and 3) determine what changes occurred to the spatial patterning of trees and regeneration. Conifer forests in the SSPM are relatively unique in that harvesting has never occurred and limited fire suppression only began in the 1970s (Stephens et al. 2003, Skinner et al. 2008). Thus, these forests serve as a model in studying the ecological effects of disturbances where ecosystem processes remain relatively intact and are in strong contrast to conditions in millions of hectares of similar forests in the western US (Minnich et al. 2000). We hypothesized that this wildfire would maintain or increase spatial heterogeneity in forest structure. Previous work has determined that SSPM forests have high spatial heterogeneity without experiencing a fire for the last 40 yr (Stephens 2004, Stephens and Fry 2005, Stephens and Gill 2005, Stephens et al. 2007) and another SSPM study found large-scale variability in fire damage from the analysis of color photographs after a 1989 fire (Minnich et al. 2000). This project is the first SSPM project to examine how wildfire affects the spatial relationships of seedlings and trees from detailed field-based measurements.

METHODS

Our work occurred in the Sierra San Pedro Martir (SSPM) National Park (31° 37' N, 115° 59' W) located approximately 100 km southeast of Ensenada in northern Baja California, Mexico. The SSPM is the southern terminus of the Peninsular Mountain Range that begins at the boundary between the San Jacinto and San Bernardino Mountains in California; approximately 350 km separates the SSPM from the San Bernardino Mountains. Vegetation is predominately conifer forests and shrublands of the Californian floristic province (Minnich et al. 2000). The flora of SSPM forests is very similar to that found in the Peninsular and Transverse Mountains of southern California and eastern Sierra Nevada (Minnich et al. 1995, Savage 1997, Minnich and Franco 1998, Stephens 2001, Stephens and Gill 2005). The North American Mediterranean climate zone extends into the SSPM although this area does receive more summer precipitation than most of California (Stephens et al. 2003, Evett et al. 2007a). Mean annual precipitation measured at Vallecitos meadow (1989–1992), approximately 7 km southeast of our study area, was 55 cm (Minnich et al. 2000). Soils in our study area are Entisols derived from diorite (Stephens and Gill 2005). From 1999 to 2002 the area experienced a severe drought, similar to that of the forests in southern California (Stephens and Fulé 2005). SSPM forests have low understory grass cover and there is little evidence that grasses were a significant component of these forests for the last several 100 yr (Evett et al. 2007b). Past median fire return intervals in Jeffrey pine dominated forests in the SSPM were shorter than 15 yr for all composite scales and years between fires varied from 1-43 yr on individual trees (Stephens et al. 2003).

To examine wildfire-caused mortality patterns we systematically located a forested watershed (approximately 14 ha, elevation 2600 m; Fig. 1) within a remote July 2003 wildfire, approximately 400 m from the nearest burn perimeter. This area was selected because it was outside the influence of atypical conditions such as those on the steep western escarpment where the fire entered the upper plateau forests via a chaparral headfire. The site was also selected to encompass all aspects and slopes varying from 0-60% to determine if these abiotic factors influenced fire behavior and effects. The 2003 wildfire was extinguished by summer thunderstorms and no summer rainfall proceeded the fire (R. Minnich, *personal communication*,

2008). A permanent weather station was not available in the SSPM; Mount Laguna (elevation 1860 m, approximately 200 km north of study site) is the closest mountain to the SSPM with a permanent weather station (R. Minnich, *personal communication*, 2008), and during the week of the fire temperatures varied from 7°-26 °C, relative humidity from 15-50%, and winds varied 0-3 m/s (upslope in the day, downslope in the evenings).

Forest stand inventory within the watershed was conducted using a grid of 27 circular plots (plot area=0.04 ha) installed 11 mo after the fire (June 2004). Plot centers were systematically located (75 m spacing); the starting point of the grid was randomly placed. Forest measurements followed the methods described in Stephens (2004) and Stephens and Gill (2005). In each plot, all stems at or above 2.5 cm dbh were measured for the following attributes: species, status (live or dead), dbh, and three variables quantifying fire damage. If the stem was dead, a condition rating of 1-3 was assigned based on decay characteristics (Cline et al. 1980) to distinguish between preexisting snags and wildfire-caused mortality. Forest canopy cover was measured using a site tube using a 5×5 grid was overlain each plot at a 5 m spacing. At each point on the grid, the sight tube was used to determine if a tree crown was directly overhead. Percent canopy cover was estimated by the total number of points under canopy divided by the 25 grid points sampled.

Analyses of fire-caused tree mortality often include variables quantifying fire damage on trees as surrogates for fire intensity. Studies have shown measures such as crown scorch are correlated with fire intensity (discussed in Kobziar et al. 2006). We collected three tree severity measurements: percent of the crown volume scorched (PCVS), crown scorch height, and bark char height. Trees in plots were resurveyed for status the third year post-fire (2006) to monitor for delayed mortality. Wildfire effects on regeneration were assessed in the forest inventory plots. For each seedling (dbh less than 2.5 cm) encountered, species, status (live or dead), and location relative to shrub and overstory tree crowns were recorded. The location of each tree and seedling relative to plot center was determined using a laser rangefinder. Percent shrub cover (live or dead) in each plot was estimated from three 10 m line transects installed on randomly selected compass directions.

The Winkelmass (Wi) is a metric that describes the spatial arrangement of individuals based on the classification of the angles between the nearest four individuals (Gadow and Hui 2002, Schmidt et al. 2006). Although two angles can describe the separation between neighboring individuals, the smaller angle is always used. The concept is based on the classification of the angles αj ($j = 1,2,3,4$) between the immediate neighbors of the four trees with reference to the plot centers. An immediate neighbor is the next tree following a given clockwise direction. Through simulations from regular, random, and clumped stands (Gadow and Hui 2002) $Wi=0$ indicates individuals are regularly distributed, $Wi=0.5$ indicates a random distribution whereas $Wi=1$ suggests individuals are clumped (see Schmidt et al. 2006 for more information on Wi). To describe live tree and seedling spatial patterns pre-fire (estimated from 2004 observations) and post-fire (2006), the average Wi was calculated for the 27 plots, and its distribution graphically illustrated, depicting the variability and changes as a result of the wildfire (Fig. 2).

We explored relationships between fire-caused tree mortality and site (slope and aspect), stand (tree density, basal area and canopy cover), and average tree and shrub damage variables mentioned above using regression tree analysis, weighted by tree sample size in each plot. Regression trees explain the variation in a numeric response variable by one or more explanatory (categorical and/or numeric) variables (Breiman et al. 1984). The ability of regression tree analysis to handle complex ecological data, e.g., discontinuity, nonlinearity and high-order interactions, and convey relationships clearly gives it an advantage over traditional statistical methods (De'ath and Fabricus 2000). A tree is constructed by repeatedly splitting the data into increasingly homogeneous groups based on the response variable. Each split minimizes the sum of squares within the resulting groups. The number of terminal nodes, or leaves, was determined using the one standard error rule on the cross-validated relative error (Breiman et al. 1984, De'ath and Fabricus 2000). Analysis was conducted with the rpart library in R statistics software (Version 2.4.0, R Development Core Team 2006).

Fig. 1. This photo of an unmanaged Jeffrey pine-mixed conifer forest was taken 3 yr after a 2003 wildfire in northern Sierra San Pedro Martir (SSPM), northwestern Mexico, the first fire in this area in several decades. Observations of average forest stand characteristics, i.e., tree and snag density, basal area, fuel loads, are uncommon in these forests. This spatial heterogeneity is due to recurring disturbances such as fire, which influence landscape patterns.

Fig. 2. Changes in forest characteristics after a wildfire in an unmanaged Jeffrey pine-mixed conifer forest in northern Sierra San Pedro Martir (SSPM), northwestern Mexico (n=27 plots): frequency distribution of Winkelmass (Wi) values for live trees (A) and live seedlings (B) (pre-wildfire, white bars; 1-yr post-wildfire, black bars), live-tree density (C) by size class (pre-wildfire, entire bars; 1-yr post-wildfire gray bars; 3-yr post-wildfire, black bars), and live-seedling density (D) by location relative to shrub and overstory crown cover (pre-wildfire, white bars; 1-yr post-wildfire, black bars).
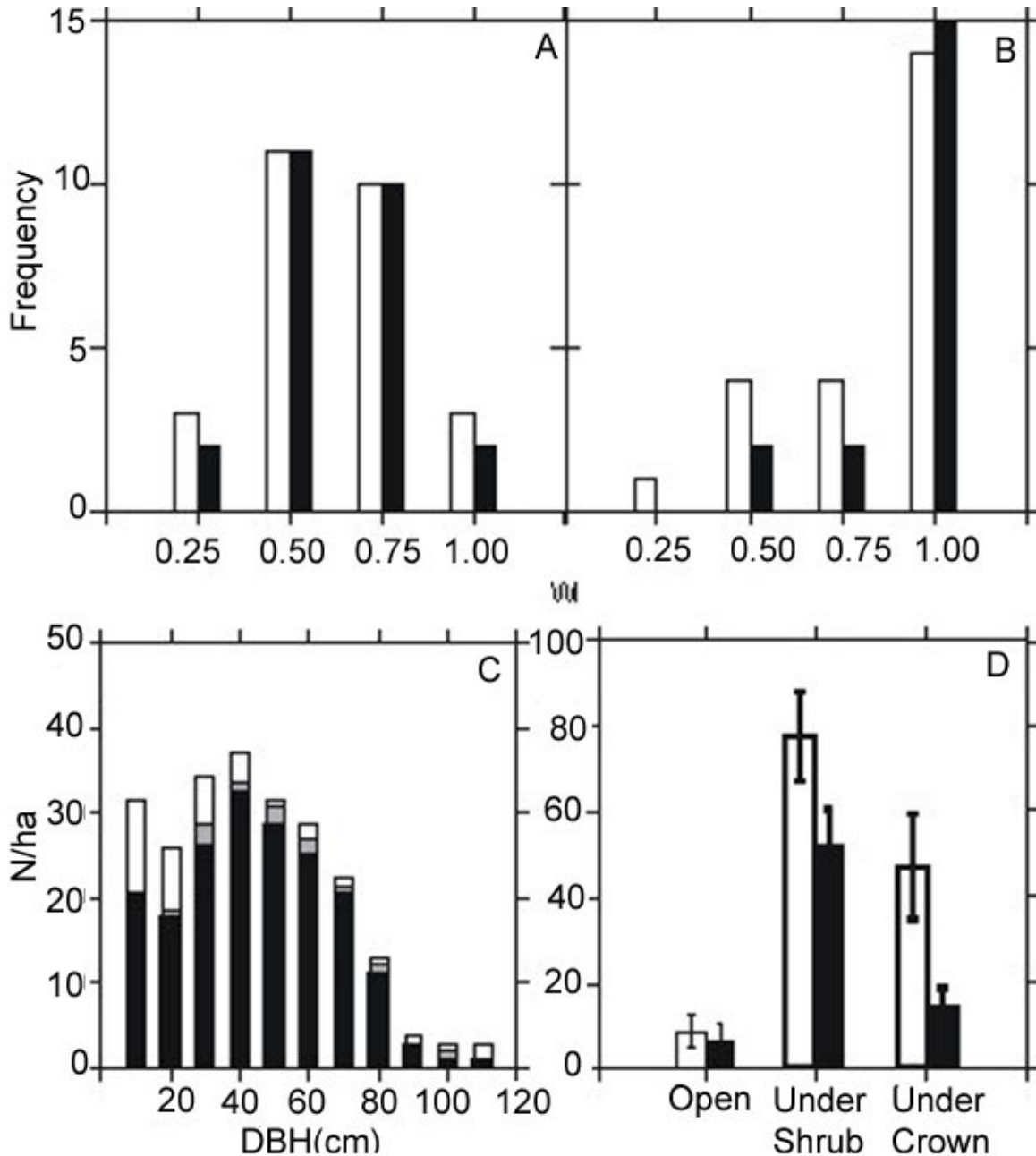
RESULTS

Forest characteristics

The grid of 27 plots inventoried approximately 7.7% of the 14 ha watershed. Evaluation of the tree crowns permitted the distinction between pre-wildfire snags (present before the 2003 wildfire) and fire-caused-mortality. Pre-wildfire plot average tree density was 243.5 trees ha-1 (SE=19.3) and average basal area was 39.9 m²/ha (SE=3.6). Average snag density was 10.2 snags/ha (SE=3.3). A total of 264 trees were measured and Jeffrey pine was the dominant species comprising 88.1% and 80.1% of the plot average live tree density and live basal area, respectively. In 51.9% of the sampled area, Jeffrey pine was the only tree species present. Sugar pine (*Pinus lambertiana*) and white fir (*Abies concolor*) were the next most common species contributing 6.0% and 4.8% of live density and 9.2% and 9.9% of live basal area, respectively. Incense-cedar (*Calocedrus decurrens*) and quaking aspen (*Populus tremuloides*) were minor components. Average tree dbh was 39.0 cm (SE=4.6) and ranged from 2.5 to 108.7 cm. The frequency distribution of Wi values (average=0.62, SE=0.04) pre-fire indicated that live trees in most plots (77.8%) were either randomly distributed or slightly clumped (Fig. 2A).

A total of 174 conifer seedlings were measured in 26 of the 27 plots. Estimates of pre-wildfire average density was 161.1 seedlings/ha (SE=29.7), with a range of 0–700 seedlings/ha. Analogous to tree composition, Jeffrey pine was the dominant seedling species comprising 92% of average density. Sugar pine was the next most common contributing 5.2% of density followed by white fir contributing 2.9%. Almost half of the seedlings (47.7%) were found beneath the tree overstory or shrub canopy. Estimated pre-wildfire average *W*i for live seedlings was 0.84 (SE=0.05) indicating a mostly clumped spatial distribution. The frequency distribution of *W*i values indicated that in most plots (66.7%) live seedlings were clumped (Fig. 2B). A few seedlings could have been missed because of complete consumption during the 2003 fire but partial skeletons of even small seedlings were well preserved in the fire area lowering the chances for error. Our estimates of pre-fire seedling density are also similar to a previous study in the Sierra San Pedro Martir (Stephens and Gill 2005).

Fire effects

Plot average fire damage on trees (PCVS, crown scorch height, and bark char height) in the watershed was low overall (Table 1), but some trees received moderate-high damage; 136 trees (54%) showed no evidence of crown scorch while 19 (7.5%) had completely scorched crowns. The fire did not scorch any of the tree crowns in three plots (11.1%).

Eleven months after the wildfire (2004), a total of 39 fire-killed trees (15.5%) were found in 13 of the 27 plots. The regression tree analysis indicates differences in the relative importance of plot-level site, average stand and fire damage variables in explaining percent tree mortality. Plot average bark char height explained the highest proportion (50.6%) of total sum of squares (SS) (Fig. 3). Low average bark char height (<0.88 m at split) corresponded with no tree mortality (12 plots). The opposing group of this split (15 plots, average tree mortality=26.7%) was subsequently split by the second variable, dead shrub percent cover, increasing explained total SS to 58.8%. Seven plots with less than 3.3% dead shrub cover had an average percent tree mortality of 19.5%. The remaining group of eight plots (dead shrub cover >3.3%) had an average tree mortality of 33.8%.

By 2006, 12 additional trees died increasing total cumulative mortality to 21.8%. Insects that were associated with secondary mortality included pine engraver (*Ips pini*), red turpentine beetle (*Dendroctonus valens*), and Jeffrey pine beetle (*Dendroctonus jeffreyi*), although overall insect populations were low based on the number of frass pockets, i.e., mixed sawdust and insect droppings, found on the boles of trees. Mortality increased average snag density to 42.6/ha (SE=9.5). Concomitantly, plot average live tree density and basal area decreased by 24.3% and 21.9%, respectively. Shifts in species composition favored Jeffrey pine, now contributing 90.5% of tree density and 79.3% of basal area. Mortality occurred in all size classes and most (76.5%) were Jeffrey pine. The diameter distribution of live trees shifted toward the middle size classes as a result of tree mortality (Fig. 2C). Half of the smallest (<20 cm dbh) and largest trees (>80 cm dbh) died. Post-wildfire frequency distribution of tree *W*i (average=0.62, SE=0.04) was unchanged compared to the pre-wildfire estimate (Chi-square test with *df*=3; χ2=0.3, *p*=0.957; Fig. 2A).

Table 1. Summary of fire effects after a 2003 wildfire in a Jeffrey pine-mixed conifer forest in the Sierra San Pedro Martir (SSPM), northwestern Mexico (n=27 plots). SE, one standard error of the mean; PCVS, percent crown volume scorched. Data from 2006.

| Forest stand characteristic	Average (SE)	Median	Minimum	Maximum
Slope (%)	36.0 (2.7)	35.0	5.0	60.0
Canopy Cover (%)	28.4 (2.1)	28.0	12.0	52.0
PCVS	18.6 (3.4)	12.5	0	62.3
Scorch height (m)	2.7 (0.5)	1.7	0	10.6
Bark char height (m)	1.5 (0.3)	1.0	0	6.7
Tree mortality (%)	21.8 (5.3)	12.5	0	100
Seedling mortality (%)	45.4 (7.1)	40	0	100
Shrub cover mortality (%)	7.4 (1.9)	3.7	0	32.3

There was a 55.2% reduction in the total number of live seedlings after the fire (Table 1). Average seedling density decreased to 72.2 seedlings/ha (SE=11.7), with a range of 0–475 seedlings/ha. Jeffrey pine was still the most dominant species contributing 89.7% of the total density. The post-wildfire frequency distribution of seedling W_i (average=0.92, SE=0.03) was not significantly different compared to the pre-wildfire estimate (χ^2=2.0, p=0.57; Fig. 2B). There was a significant change in the location of live seedlings with respect to the overstory and shrub canopy. Disproportionately more seedlings beneath (73.5%) canopy (tree crown or shrub) were killed by the wildfire compared to outside (38.5%) the canopy (Chi-square test with df=1; χ^2=20.2, p=0.000; Fig. 2D).

Average shrub cover pre-fire was 12.6% (SE=3.3), greenleaf manzanita (*Arctostaphylos patula*) dominated the understory contributing 79% of total cover. After fire shrub cover was 5.3%; SE=2.7 (Table 1). Plot average bark char height was correlated with shrub cover mortality (r=0.47, p=0.014).

DISCUSSION

Fire effects in the 2003 Sierra San Pedro Martir (SSPM) wildfire were moderate, especially considering that the wildfire occurred at the end of a severe, multi-year (1999-2003) drought (Stephens and Fulé 2005). Previous work in similar SSPM forests determined that approximately one new snag/ha was created from the 1999-2003 drought (Stephens 2004). Here we estimated that the 2003 wildfire created approximately 40 snags/ha in Jeffrey pine-mixed conifer forests. In comparable forests in southern California the same multi-year drought killed millions of trees without any impact of fire. Annual precipitation was below half-normal during the drought (1999-2003) and was the lowest ever recorded for the San Bernardino Mountains in 2001–2002 (Eatough-Jones et al. 2004). Decades of fire suppression had increased stand densities and surface fuel loads, past logging practices had also affected these forests. In mixed Jeffrey pine forests in the San Bernardino Mountains in southern California, tree densities increased by 79% from the early 1930s to 1992 (Minnich et al. 1995) and more than doubled at Cuyamaca Mountain from 1932 to the present (Goforth and Minnich 2008). Drought-induced stress led to increased susceptibility of trees to native tree-killing insects similar to those found

Fig. 3. Spatial distribution of fire damage-tree mortality relationship in a Jeffrey pine-mixed conifer forest in Sierra San Pedro Martir (SSPM), northwestern Mexico, 3 yr after a 2003 wildfire. Pie charts represent forest survey plot locations in a 14-ha watershed. Proportion filled (black area) is the average bark char height converted to percent and color identifies plot-level tree mortality.
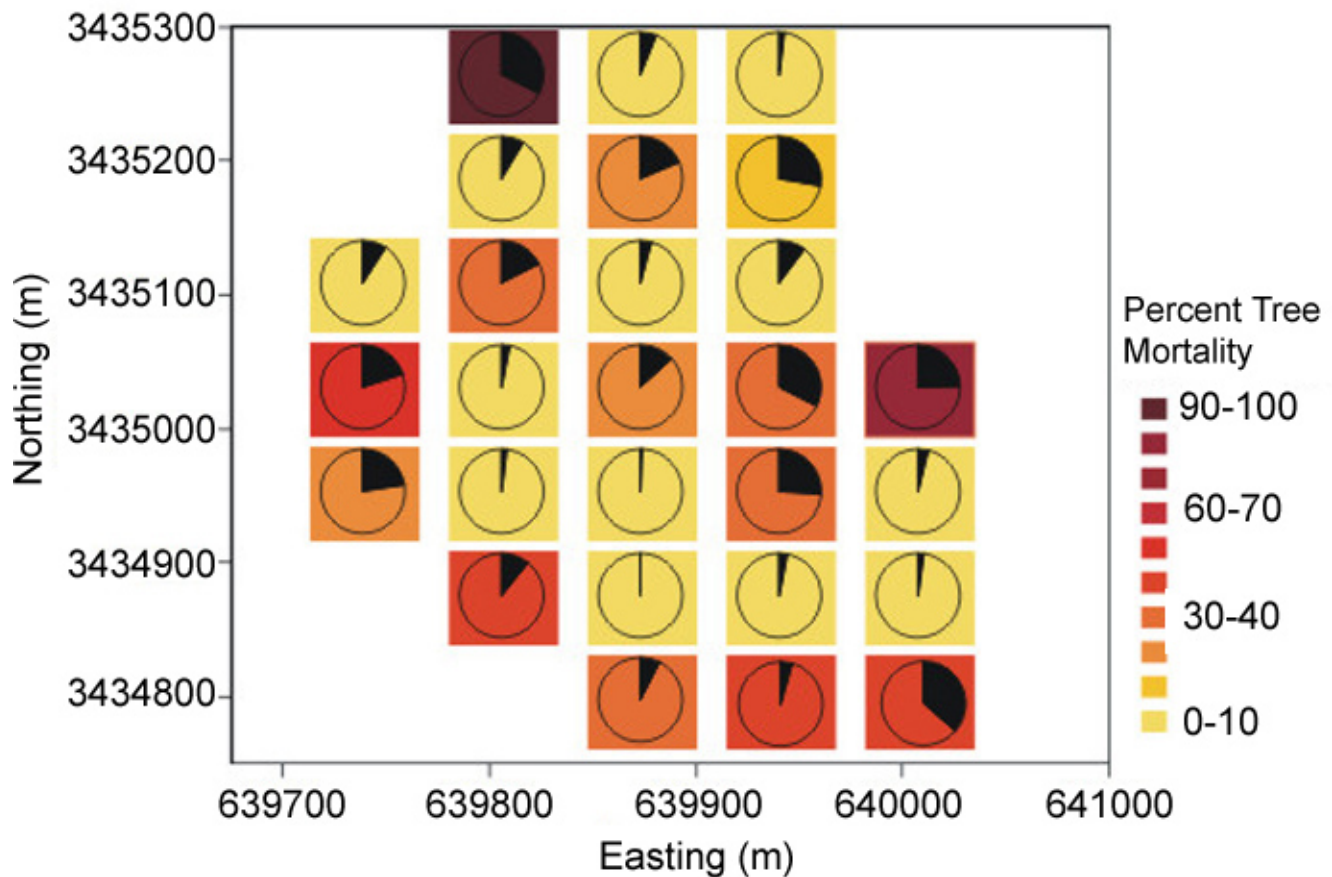
in the SSPM, with the addition of western pine beetle (D*endroctonus brevicomis*) in southern California. Bark beetle induced mortality was approximately 125 trees/ha in southern California forests (Stephens and Fulé 2005), and in some areas forests experienced 30 to 100% mortality (Krofta 1995, Eatough-Jones et al. 2004) without a fire burning a single tree.

Where wildfire did impact mixed Jeffrey pine forests in southern California, mortality was very high. In the Laguna Mountains in the San Diego County, tree mortality was 40-95% after the 2003 Cedar Fire (Franklin et al. 2006). In these forests, several decades of fire suppression has led to forests with high stand densities. In a 1992 survey (Krofta 1995), 43% of these trees were already dead due to drought and bark beetles, which in turn contributed to the high mortality rate after the Cedar Fire (Franklin et al. 2006). In our study, mortality after the wildfire was attributed primarily to fire damage to trees (Figs 1 and 3) from burning shrubs in

contrast to fire behavior in southern California forests that was driven by increased stand densities and high surface fuel loads. Fire weather in southern California during the 2003 wildfires included higher wind speeds and this would have affected fire behavior and effects. High velocity Santa Anna winds drove the 2003 fires to the west and down-slope and most of this burning occurred in chaparral shrublands. After the Santa Ana winds subsided many fires reversed direction and burned upslope and into higher elevation forests.

The 2003 wildfire in the SSPM killed approximately half of the smallest sized trees. Similarly, another SSPM study that analyzed photographs to estimate mortality from two fires in 1989 estimated that 65% of pole-sized trees were killed (Minnich et al. 2000). That the smaller trees would be more vulnerable to death is not surprising since these trees lack the characteristics (thick, insulating bark and adequate crown height) that would enable them to survive surface fires. This wildfire killed half of the largest trees in our stands which is somewhat surprising considering these large trees possess the characteristics such as thick bark and elevated crowns that would increase their chances of survival. It is possible that the severe multi-year drought reduced the vigor of the largest trees (many of them are over 350 years old, Stephens et al. 2003), and when exposed to fire, the combination was fatal for about half of them. The wildfire increased the dominance of Jeffery pine, both in the overstory and in seedlings. This provides evidence that Jeffrey pine dominates these stands at least in part by its ability to survive fire at higher frequency than its other mixed conifer associates.

Why were the forests in the SSPM able to incorporate drought, insects, and wildfire without producing catastrophic mortality? This and previous research suggests that heterogeneity in spatial patterns of forest structure and fuels are critical for a resilient forest. High variability characterizes all live tree, snag, fuel, and coarse woody debris in the forests of the SSPM without recent fire (Stephens 2004, Stephens and Gill 2005, Stephens et al. 2007). In this research spatial forest structure was random to slightly clumped pre and post wildfire (Fig. 2A). Seedlings were spatially clumped pre-fire and this was maintained after fire (Fig. 2B) and most surviving seedlings were isolated from overstory cover and shrubs where fuel loads would be relatively low.

SSPM forests have high resistance to mortality and this is probably the result of a relatively intact frequent surface fire regime in an old-growth forest. High variability in surface fuel loads would produce equally diverse fire behavior and effects, maintaining high spatial heterogeneity when the forest continues to burn under a frequent fire regime (see Fig. 1). It is likely that similar forests (Jeffrey pine, Jeffrey pine-mixed conifer, dry mixed conifer, dry ponderosa pine (*Pinus ponderosa*)) in the western US also had a high amount of variation and were spatially aggregated because they once experienced comparable disturbance regimes. Spatial variability is a key element in the ability of the forests of the SSPM to resist catastrophic change brought about by drought, insects, and fire and heterogeneity should be included in US forest restoration efforts (Stephens and Fulé 2005). However, spatial variability is uncommon in most US forest restoration practices. The most common forest fuel reduction treatment is a thin-from-below to separate overstory tree crowns and maintain a desired basal area within a limited range (Graham et al. 2004). These practices produce uniform forest conditions over broad areas and are in strong contrast to what is found in the resilient SSPM forests.

How can the fire paradox in the US be ameliorated? In addition to producing more spatial heterogeneity in restoration plans of forests that once experienced frequent, low-moderate intensity fire regimes, higher amounts of burning are needed. Increased use of US wildfire management options such as wildland fire use (WFU) as well as appropriate management responses (AMR) to non-natural ignitions could also be implemented at broader spatial scales to increase the amount of burning in western US forests. WFU is the management of lightning-ignited fires to accomplish specific resource management objectives in pre-defined geographic areas (Stephens and Ruth 2005). In many cases a portion of WFU fires receive suppression actions but many of these fires burn for months and produce positive ecological effects. AMR allows a suppression fire to be managed less intensively and incorporates a full range of responses, from aggressive suppression to managing a fire to minimize suppression costs and increase fire fighter safety.

If fire suppression is maintained in the SSPM, this will inevitably lead to the production of forests with problems similar to many forests in the western US.

Conservation of forests in the SSPM is critical because it is one of the last landscape-scale, old-growth mixed conifer forests in western North America with a relatively intact frequent fire regime. While managers of the SSPM should learn from past US forest management practices, managers in the US wish that they had Jeffrey pine-mixed conifer forests with similar fire problems as those in the SSPM.

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In June 2004, 27 circular 0.1 ha plots were established in a Sierra San Pedro Martír (SSPM) watershed that burned in a 2003 wildfire on a 75m grid with a random starting point, and measured as described in Stephens et al. (2008). In June 2019, we located and re-measured all 27 plots.

We calculated descriptive statistics of tree basal area, tree density, snag density, seedling density, canopy cover, and shrub cover. Based on our re-measurement of the same plots in 2019, we discovered that some stand-level metrics reported in Stephens et al. (2008) *were incorrect*, due to using the wrong expansion factor for expanding plot-level measurements. Therefore, we used the data collected in 2004 and 2006 to re-calculate descriptive statistics for forest characteristics to correct those values. Initially reported and corrected values are compared in Table 1 and included in Murphy et al. (2021).

Table 1: Stand metrics as originally reported in Stephens et al. 2008 ("Reported") and re-calculated using raw data ("Corrected") † denotes that the values were reported only as a percent reduction from pre-fire measures, and therefore, no standard errors were given. Three years post-fire metrics are given to account for delayed mortality.

	Pre-Wildfire		3 Years Post-Wildfire	
	Reported Mean (SE)	Corrected Mean (SE)	Reported Mean (SE)	Corrected Mean (SE)
Basal area ($m^2 ha^{-1}$)	39.9 (3.6)	14.8 (1.3)	31.2 (-)†	11.8 (1.4)
Live tree density (trees ha^{-1})	243.5 (19.3)	93.7 (7.8)	184.3 (-)†	74.4 (7.7)
Snag density (trees ha^{-1})	10.2 (3.3)	4.1 (1.3)	42.6 (9.5)	18.5 (3.5)
Seedling density (trees ha^{-1})	161.1 (29.7)	64.4 (11.9)	72.2 (11.7)	28.9 (4.7)

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