Forest Ecology and Management 441 (2019) 20–31

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Status and trends of fire activity in southern California yellow pine and mixed conifer forests

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ARTICLE INFO

Keywords:
Southern California
Fire return interval
Burn severity
Fire size
Natural range of variation
Yellow pine
Mixed conifer

ABSTRACT

Frequent, low to moderate severity fire in mixed conifer and yellow pine forests of California played an integral role in maintaining these ecosystems historically. Fire suppression starting in the early 20th century has led to altered fire regimes that affect forest composition, structure and risk of vegetation type conversion following disturbance. Several studies have found evidence of increasingly large proportions and patches of high severity fire in fire-deprived conifer forests of northern California, but few studies have investigated the impacts of fire suppression on the isolated forests of southern California. In this study, spatial data were used to compare the current fire return interval (FRI) in yellow pine and mixed conifer forests of southern California to historical conditions. Remotely sensed burn severity and fire perimeter data were analyzed to assess changes in burn severity and fire size patterns over the last 32–100 years. Half of the yellow pine and mixed conifer forest in this study has missed multiple burn opportunities and has not experienced fire in the 109-year fire record. The average proportion of conifer forests burned in high severity fire (> 90% tree basal area loss), average high severity patch size, and maximum high severity patch size all increased significantly from 1984 to 2016. The average fire proportion burned at high severity from 2000 to 2016 is 1.5 times higher than predicted for the natural range of variation (NRV). Additionally, the years after 2000 had high severity patches larger than 25 ha, whereas no fires before 2000 had patches this large, indicating a deviation from NRV since the turn of the century. Fire size in conifer forests significantly increased from 1910 to 2016, owed to a substantial increase in the occurrence of large fires (larger than the natural range of variation) after 2000. This analysis indicates that southern California conifer forests are like their northern counterparts in that they have burned very infrequently since the early 1900s, resulting in large and homogenous areas of stand replacing burns. This is likely exacerbated even further by recent fire-conducive weather conditions and extended periods of drought. In southern California, recovery from large, high severity burns is likely to be impeded by the small and disparate nature of mixed conifer forests and limited seed dispersal capabilities of remaining trees. Therefore, preemptive forest treatments and careful fire management is needed to return natural structure and function to these forests.

1. Introduction

Fire has played an integral part in defining and shaping vegetation patterns and composition in California for millions of years, both via natural, and more recently, human-caused ignitions (van Wagtendonk et al., 2018). Before Native American colonization, lightning was the main source of ignition for fire (Minnich, 1988) and fire frequencies in some parts of the state increased following Native American occupation (Anderson, 2006). Millions of hectares of vegetation burned annually in California, consuming 4.5–12% of the state's area (1.8–4.8 million ha per year) (Stephens et al., 2007). The frequency and severity with which these fires burned helped shape the numerous vegetation types that exist in California today. However, effective national fire suppression practices beginning in the early 20th century have severely reduced the annual area burned in California, which averaged only 102,000 ha per year from 1950 to 1999 (Stephens et al., 2007; Safford and Stevens, 2017).

The fire regime in mixed conifer and yellow pine forests has historically been defined by one of the most frequent fire return intervals (FRI) of any vegetation type in the state (Van de Water and Safford,

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https://doi.org/10.1016/j.foreco.2019.01.020
Received 11 September 2018; Received in revised form 14 January 2019; Accepted 17 January 2019
Available online 21 March 2019
0378-1127/

2011), partially due to the high concentration of lightning strikes where these forests exist (Safford and Stevens, 2017). An extensive review of the literature on fire regime prior to 1850 revealed that yellow pine and mixed conifer forests burned an average of every 11–16 years in California (Van de Water and Safford, 2011). Analog systems that have experienced little fire suppression, such as the Sierra de San Pedro Mártir (SSPM) (pre-1970s) in Baja California, Mexico and parts of Yosemite and Sequoia and Kings Canyon national parks managed with a policy of wildland fire use, have maintained recent fire frequencies similar to historical estimates (Caprio and Graber, 2000; Stephens et al., 2003; Collins, 2007).

Most mixed conifer and yellow pine forests in California have been managed under a strict policy of fire suppression for the last century and, as a result, 75% of these forests have not experienced fire since 1908 (Steel et al., 2015). The effects of this altered fire frequency have been well studied in the Sierra Nevada of northern California, where longer intervals between fires are contributing to stand densification and increased fire severity. In the beginning of the 20th century, before significant fire suppression began, mixed conifer forests in the Sierra Nevada were dominated by trees that were large (average DBH > 90 cm), old (250–350 years on average), and sparsely distributed (McKelvey and Johnston, 1992; Collins et al., 2011; Knapp et al., 2013; Safford and Stevens, 2017). However, resampling efforts in the last 35 years have revealed that tree density and canopy cover have increased significantly, owing to a substantial increase in the number of small-diameter trees in these forests (McKelvey and Johnston, 1992; Collins et al., 2011; Knapp et al., 2013). While the loss of large diameter trees in the Sierra Nevada mixed conifer forests has been attributed to past logging practices, the increase in density of smaller diameter trees is most likely due to fire suppression (Knapp et al., 2013). Even Baja California's recent 30 years of fire suppression are starting to reveal signs of densification, which were measured as a 237% increase in seedling density, a 120% increase in live tree density, and a 13% increase in tree basal area since 1998 (Dunbar-Irwin and Safford, 2016).

Stand densification and increased ladder fuels (small and intermediate sized trees that carry fire from the understory to the overstory) in the Sierra Nevada are most likely the primary drivers of the increasing proportion and size of high-severity burns in fires over the last 30 years (Miller et al., 2009b; Miller and Safford, 2012). The proportion of annual area burned at high severity has more than tripled since before Euro-American settlement in the Sierra Nevada, resulting in a deficit of low and moderate severity burns (Mallek et al., 2013). In addition, high severity patches doubled in size between 1984 and 2016 in the Sierra Nevada (Miller et al., 2009b; Steel et al., 2015). In the SSPM, recent fires are still predominantly burning at low severity and have not seen increasing trends over time (Rivera-Huerta et al., 2016). However, there is evidence that fires in the SSPM conifer forests may be becoming more severe and will continue to do so with continued fire suppression (Rivera-Huerta et al., 2016).

The effect of fire suppression on the mixed conifer forests of southern California has been less well-studied than effects on their northern and southern counterparts. Dendrochronology studies in southern California align with historic fire regime estimates from the rest of the state and reveal a historical pre-suppression FRI of less than 14 years in mixed conifer and yellow pine forest types, though it may have been slightly longer (up to 19 years) in the drier forests dominated by Jeffrey pine (McBride and Laven, 1976; McBride and Jacobs, 1980; Keeley, 2006; Skinner et al., 2006). Fire in the mixed conifer forests of southern California has also been severely suppressed over the past century (Safford, 2007), resulting in stand densification and increased numbers of small-diameter, shade-tolerant trees (Savage, 1994; Minnich et al., 1995; Stephenson and Calcarone, 1999; Goforth and Minnich, 2008). However, the effects of fire suppression on fire frequency and burn severity have not been well quantified. Mixed conifer forest makes up a small proportion of the vegetation in southern California, which is mostly dominated by shrublands (e.g. chaparral, sage scrub, desert scrub) (Minnich and Everett, 2001). These conifer forests exist at only the highest elevations of the Transverse and Peninsular Ranges above the chaparral belt and often occur in disparate patches throughout the region (Minnich and Everett, 2001). Therefore, conifer forests are a unique resource for recreation and biodiversity in southern California (Stephenson and Calcarone, 1999; Minnich, 2007). To better understand the susceptibility and resilience of these systems to disturbance, we set out to characterize the status and trends for fire activity across southern California conifer forests.

In this study, we specifically aim to (1) characterize the current FRI in conifer forests across southern California and compare it to pre-Euro-American settlement estimates of fire frequency, (2) analyze changes in burn severity patterns in southern California conifer forests over the last 32 years, with respect to proportion of area burned at high severity and high severity patch size, and (3) evaluate trends in fire size since the early 1900s.

2. Material and methods

2.1. Study area

Our analysis focuses on the four southern California national forests (Cleveland (CNF), San Bernardino (SBNF), Angeles (ANF), and Los Padres (LPNF)) (Fig. 1), which contain 70% of the conifer forest in southern California (defined as Monterey County, and all counties south and inclusive of San Luis Obispo and Kern). This analysis was restricted to U.S. Forest Service land as it contains most of the conifer forest in southern California and this ensured that land management practices were similar across the study area. In addition, the spatial data required for the analyses (described below) were generated using the same methods across these lands. The LPNF was split into two assessment areas – the Monterey District (LPN) in the north, which extends through Big Sur in Monterey County, and the lower Los Padres (LPS), which is mostly contained in San Luis Obispo, Santa Barbara, and Ventura counties. Since these are geographically and environmentally distinct subregions, we chose to classify the LPN and LPS as separate assessment areas.

Vegetation types were classified by pre-Euro-American settlement fire regime (PFR) groups, which lump vegetation types with similar fire regime characteristics together using LANDFIRE Biophysical Settings types (LANDFIRE, 2016) and other relevant literature (Van de Water and Safford, 2011). Van de Water and Safford (2011) performed an extensive literature review of the estimated fire return interval (FRI) for these vegetation types before Euro-American settlement. The data collected from this search were compiled for each PFR group, which was then assigned a mean, median, mean minimum, and mean maximum reference FRI, representing the fire return interval natural range of variation (NRV) (Van de Water and Safford, 2011). The three PFR types referred to as "conifer forest" in this analysis are dry mixed conifer, yellow pine, and moist mixed conifer, which all have similarly low reference FRIs (Table 1; Van de Water and Safford, 2011). Yellow pine forests are dominated by *Pinus ponderosa* and *P. jeffreyi,* while moist mixed conifer is dominated primarily by *Abies concolor* and dry mixed conifer is dominated by *P. ponderosa* and *P. lambertiana*. Yellow pine forests are generally found between 1375 and 2135 m in elevation, whereas mixed conifer forests tend to range higher, from 1675 to 2590 m (Thorne, 1977).

2.2. Fire return interval analysis

To assess the current FRI and how it has deviated from historic (pre-Euro-American settlement) conditions, we used the 2016 California Fire Return Interval Departure (FRID) map, created by the USFS Pacific Southwest Region (Safford et al., 2013). This spatial layer contains information on current and pre-Euro-American settlement FRIs for every major vegetation type in California's national forests, as well as

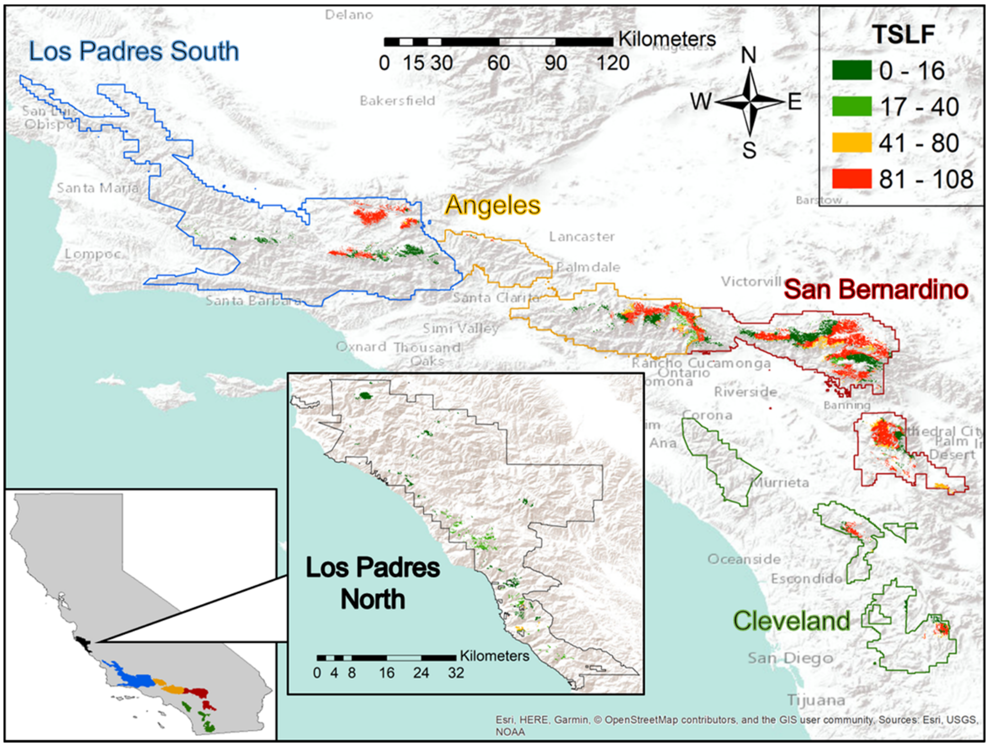

Fig. 1. Map of all yellow pine and mixed conifer forest in the five assessment areas analyzed in this study, broken down into four time since last fire (TSLF) categories (0–16 years, 17–40 years, 41–80 years, 81–108 years). TSLF is the number of years between 2016 and the last recorded fire in each polygon.

the name of and time since last fire (TSLF). Both wildfires and prescribed burns were used in the calculation of the FRID map. Using ArcGIS (ESRI, 2017), we clipped the FRID map to only include areas of "conifer forest" (as defined above) in each of the five assessment areas examined.

The percent area of each PFR type was calculated by assessment area (Table 1) and the mean reference FRI, current FRI, and TSLF for each assessment area were calculated using a weighted mean. TSLF represents the number of years between 2016 (the version year of FRID used in this study) and the last year that fire was recorded in that polygon (Safford et al., 2011). If no fire occurred in the record, TSLF was recorded as 108 years (2016 minus 1908). The current FRI in each polygon is the number of years, inclusive, in the fire record (109 for the 2016 FRID) divided by the number of times that polygon burned, plus one. FRI values found in the literature were averaged to obtain minimum (min RefFRI), mean (mean RefFRI), and maximum (max RefFRI) pre-Euro-American settlement FRIs (Table 1; Safford et al., 2011). The mean reference FRI is consistent across all polygons of the same PFR type and is an estimate of fire frequency prior to the mid-19th century (Table 1). The min and max RefFRI provide lower and upper bounds for the natural fire frequency in each PFR type prior to the mid-19th century.

The range of reference FRI values (from minimum to maximum) can be considered the NRV in FRI for each PFR type, since these values were estimated before significant human impacts were made on the land. Deviations from historical FRI were determined using the max and min refFRI. Polygons with a current FRI within these bounds were considered within the NRV and those with a current FRI outside the bounds were considered departed from the NRV.

2.3. Burn severity

Burn severity was analyzed using the relative differenced normalized burn ratio (RdNBR) between pre- and post-fire Landsat Thematic Mapper imagery (30 × 30 m resolution), calibrated with the Composite Burn Index (CBI) to assess severity to vegetation (Miller and Thode, 2007). These data are available as a spatial layer from the USDA Forest Service fire and fuels monitoring project (VegBurnSeverityBA, 2017). Each polygon is assigned a burn severity category, based on loss of live tree basal area after fire (Miller et al., 2009a). The categories are as follows: 1 (0% loss), 2 (0–10% loss), 3 (10–25% loss), 4 (25–50% loss), 5 (50–75% loss), 6 (75–90% loss) and 7 (> 90% loss). We focus mainly on the last category (> 90% loss), referred to as "high severity" throughout the paper, as these are areas of stand-replacing fire that may experience impediments to recovery. We also combine categories 1 and 2 to represent "low severity" burns that were most likely surface fires (as in Minnich et al., 2000). For many of the fires, the data layer has both an initial assessment (made with imagery acquired immediately after fire containment) and an extended assessment (made with imagery acquired the year after fire). In this analysis, we prioritize

Table 1
Pre-Euro-American settlement fire regime (PFR) vegetation types included in this analysis. Minimum, median, mean and maximum reference fire return interval (refFRI) represent the pre-Euro-American settlement fire return interval (FRI) in years and are derived from Safford et al. (2011). The percent of conifer forest made up by each PFR type in each assessment area is listed.

PFR type	Min refFRI	Median refFRI	Mean refFRI	Max refFRI	CNF	SBNF	ANF	LPS	LPN	All forests
Yellow pine	5	7	11	40	71.1%	32.5%	39.9%	52.3%	71.4%	40.0%
Dry mixed conifer	5	9	11	50	6.3%	30.8%	17.3%	26.1%	11.7%	26.0%
Moist mixed conifer	5	12	16	80	22.6%	36.6%	42.7%	21.6%	16.9%	34.0%

Table 2
Summary statistics on conifer forest area, fire return interval (FRI) and time since last fire (TSLF) for all assessment areas, separately and combined. Conifer forest was considered outside of the natural range of variation (NRV) if the current FRI was greater than the max reference FRI assigned to that forest type.

	CNF	SBNF	ANF	LPS	LPN	All assessment areas
Conifer forest area (ha)	6902.2	77140.8	22619.9	22804.1	2538.7	132005.7
Total area (ha)	227358.7	325948.6	285867.3	662240.6	134740.7	1636155.9
% Conifer forest	3.0	23.7	7.9	3.4	1.9	8.1
Ave. mean ref FRI (yrs)	12.1	12.8	13.1	12.1	11.9	12.7
Ave. current FRI (yrs)	73.9	79.6	76.9	78.7	34.2	77.8
Ave. TSLF (yrs)	65.8	70.7	67.7	66.0	14.0	68.0
% Forest in NRV	31.0	28.0	29.9	21.5	79.4	28.3
% Forest outside NRV	69.0	72.0	70.1	78.5	20.6	71.7

extended 1-year assessments to catch delayed conifer mortality that may have occurred after the initial assessments were made (delayed mortality has been documented in Franklin et al., 2006). When the extended assessment was unavailable, the initial assessment data were used (11 out of 80 fires). Burn severity data have been produced for fires larger than 400 ha occurring since 1984 (VegBurnSeverityBA, 2017). We analyzed all fires in the burn severity dataset that burned within the assessment areas and contained any amount of conifer forest (as defined above) (n = 80).

Despite the large geographical area of this analysis, the actual area of conifer forest in southern California is relatively small (8% of analyzed area). Consequently, 7 of the past 33 years had no fires recorded in conifer forest. These missing time points prevented us from performing a time series analysis, which requires equal time steps between data in a series.

Instead, we used fire as the sample unit and the continuous variable of year as the main predictor variable to evaluate the proportion of each fire (in area) that burned at high (> 90% basal area loss) and low (0–10% basal area loss) severity (sum of the total area burned at each severity level, divided by the total area of the fire). The two linear mixed effects models were fit using the package "nmle" (Pinheiro et al., 2016) with the proportion burned at each severity level as the response variable, year as the predictor variable, and assessment area as a random intercept. To satisfy the assumptions of normality and variance equality, the proportion burned at high severity data were log transformed, with a small value (0.1) added to account for zeroes in the dataset. The proportion burned at low severity data met the assumptions of linear regression and were therefore not transformed.

2.4. Patch size analysis

Average and maximum patch size of high severity burns (> 90% basal area loss) were calculated for all fires containing high severity burns (n = 68). Calculations of mean and maximum high severity patch size were achieved by averaging the size of all high severity patches within a fire and identifying the largest high severity patch from each fire, respectively. We defined a patch as a single contiguous area of a distinct fire severity class. Average and maximum high severity patch sizes were used as response variables in separate linear mixed effects models with year as the predictor variable and assessment area as a random intercept. Average and maximum patch size data were log transformed to satisfy assumptions of normality and variance equality.

2.5. Fire size

Total fire size in conifer forest was calculated using the fire perimeter database compiled by the Fire and Resource Assessment Program (FRAP) for the years 1910–2016 (Fire Perimeters, 2017). Only non-prescribed fires larger than 40 total hectares (across all vegetation types) that contained conifer forest and burned within the assessment areas were included in the analysis (n = 397); fires smaller than 40 ha were removed as they tend to be under-reported in the database before 1950 (Miller et al., 2009b). Conifer fire size was calculated for each fire by clipping the fire perimeters included in the analysis to just the area classified as conifer forest (defined above) and calculating the area in ArcGIS (ESRI, 2017). A linear mixed effects model was fit with conifer fire size as the response variable and year as the predictor variable, with assessment area as a random intercept. The fire size data were log-transformed to meet assumptions of normality and variance equality.

3. Results

3.1. Fire return interval analysis

The total conifer area analyzed in this study summed to 132,006 ha, comprising 8% of the total land area in the southern California national forests (Table 2). Yellow pine is the most abundant of the three pre-Euro-American settlement fire regime (PFR) types, followed by moist mixed conifer and dry mixed conifer (Table 1). The weighted average of mean reference (i.e. historical) fire return interval (FRI) across all conifer forest in the five assessment areas is 12.7 years. This represents the relative abundance of the three PFR types – dry mixed conifer, yellow pine, and moist mixed conifer – and their mean reference FRIs (Table 1). However, the current FRI (as calculated from the FRID layer) for these areas is, in most cases, much higher, averaging 77.8 years across all forests, where the average time since last fire (TSLF) is 68.0 years (Table 2, Fig. 1). In addition, the values calculated for FRI and TSLF are almost certainly lower than the actual FRI and TSLF in these forests, since all unburned areas in the record were assumed to have burned in 1908, giving them a default TSLF of 108 years and FRI of 109 years. With respect to fire history, the LPN is not aligned with the more southerly assessment areas. The LPN has an average current FRI of 34.2 years and average TSLF of 14.0 years, which is more closely aligned with its average mean reference FRI of 11.9 years (Table 2). The average current FRI in the LPN equates to approximately two fires burning within the last 109 years, which is six fires less than would have burned if this area was burning at the mean reference FRI. In comparison, the LPS had less than one fire on average throughout the 109 years. The SBNF is the only assessment area with forest that burned 6 or more times, and this area only summed to 40 ha (< 0.05% of SBNF conifer forest) (Fig. 2).

In all the conifer forest analyzed, 49% has not burned since the start of the fire record, 109 years ago. No conifer forests burned more often than the min reference FRI (5 years), meaning southern California forests are not burning more frequently today than before Euro-American settlement. In all assessment areas except the LPN, over 69% of the conifer forest has a current FRI greater than the max reference FRI and is therefore burning less frequently today than before Euro-American settlement. In the LPN, only 21% of conifer forest is outside its natural range for FRI (Table 2).

3.2. Burn severity analysis

Linear mixed effects modeling revealed a significant, positive

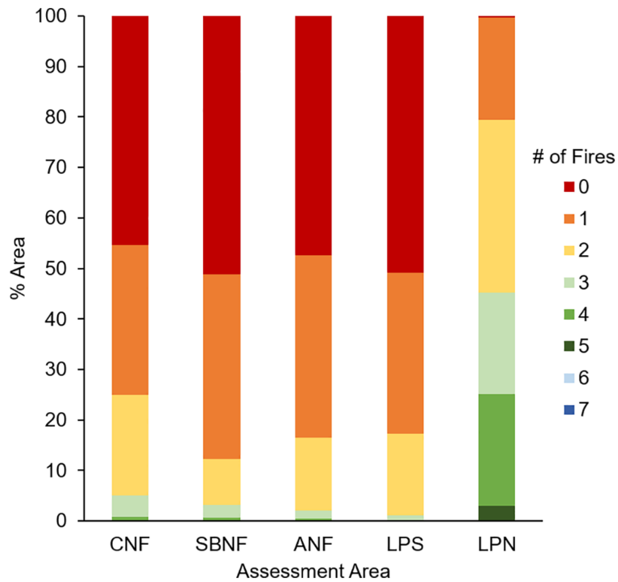

Fig. 2. Percent area of conifer forest in each assessment area burning 0–7 times since 1908. If burning at the mean fire return interval (FRI) predicted for pre-Euro-American settlement mixed conifer and yellow pine forests, most of these areas would have burned 5–9 times during this period.

relationship between proportion burned at high severity and year, accounting for assessment area (Fig. 3A, t = 4.55, df = 74, $p < 0.0001$). Conversely, proportion burned at low severity had a significant, negative correlation with year (Fig. 3B, t = −3.9, df = 74, $p < 0.001$). The variance associated with the random intercept assessment area was extremely low for both models ($\sigma^2 < 1.0E-9$) indicating that assessment area did not have a strong influence on proportion fire area that burned at high or low severity.

3.3. Patch size analysis

Average high severity patch size ranged from 0.01 to 14.5 ha, with the 2016 Sand Fire having the largest average patch size. A linear mixed effects model with log-transformed average patch size data revealed a significant, positive relationship between year and average high severity patch size, accounting for assessment area (Fig. 4A, t = 5.13, df = 62, $p < 0.0001$). Apart from the Bee 2 Fire in 1996, all fires prior to 2000 had an average high severity patch size of less than one hectare. In contrast, 62% of the fires after 2000 had an average high severity patch size of greater than one hectare.

Maximum high severity patch size per fire ranged from 0.02 to 824.0 ha, with the largest high severity patch occurring in the Butler 2 Fire of 2007. A linear mixed effects model with log-transformed maximum patch size data revealed a significant, positive relationship between year and maximum high severity patch size, while accounting for assessment area (Fig. 4B, t = 4.32, df = 62, $p < 0.001$). Maximum high severity patch size in the years after 2000 reached extremes (824 ha) that were on an order of magnitude larger than the largest high severity patch of fires pre-2000 (25 ha). Before 2000, there were no high severity patches in any fire larger than 25 ha, whereas 41% of the fires after 2000 contained patches larger than 25 ha.

The random intercept, assessment area, had low variance in both the average patch size analysis ($\sigma^2 = 2.46E-09$) and the maximum patch size analysis ($\sigma^2 = 2.89E-08$) indicating it did not have a strong influence on high severity patch size.

3.4. Fire size

A linear mixed effects model of log-transformed conifer fire size data showed a slightly significant positive relationship between year and fire size, while accounting for assessment area (Fig. 5, t = 2.11, df = 391 $p = 0.04$). The variance explained by the random intercept assessment area ($\sigma^2 = 1.3E-08$) was lower than the residual variance ($\sigma^2 = 2.16$), indicating that assessment area did not strongly influence fire size from 1910 to 2016. The percent of fires greater than 456 ha (the maximum predicted for NRV) before 2000 was only 8%, whereas 27% of fires post-2000 were greater than NRV estimates. Eight out of the ten largest conifer fires in the southern California record occurred after the turn of the 21st century. The largest of them was the Day Fire in 2006 (5925 ha burned within conifer forest).

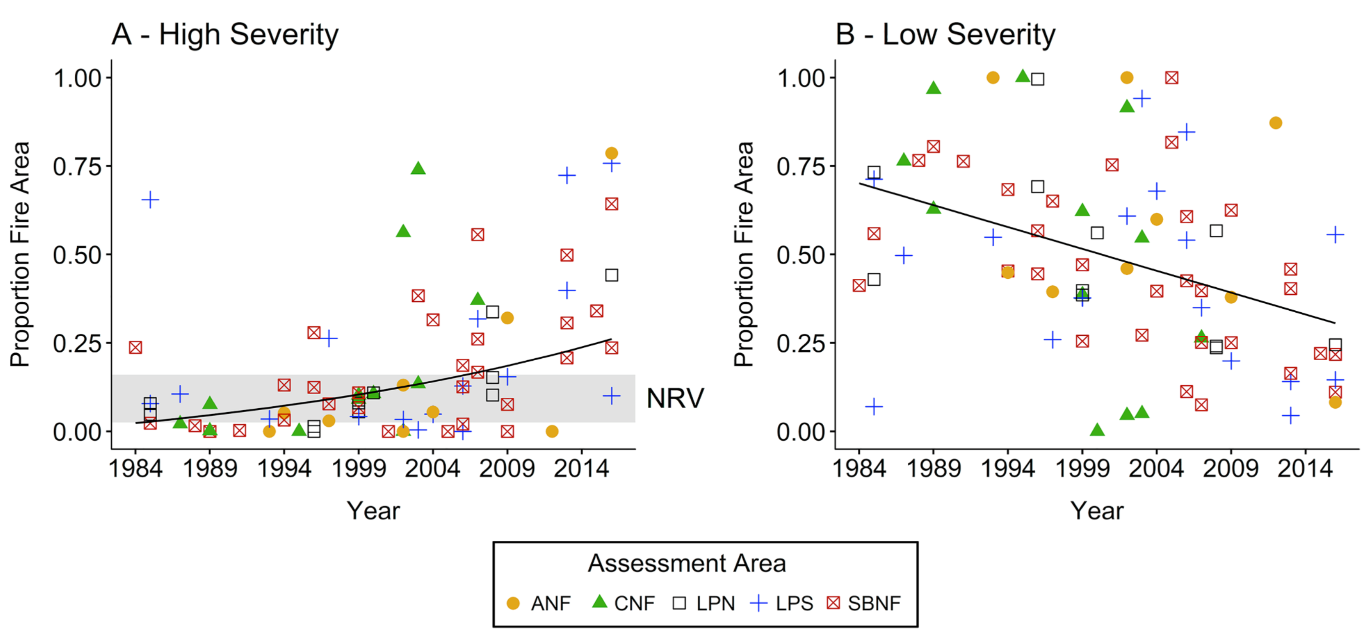

Fig. 3. Proportion fire area burned at – (A) high severity (> 90% basal area loss) and (B) low severity (no change – 10% basal area loss) over time. Lines represent results of the linear regression model fit, back transformed in the case of (A). Fires occurring in different assessment areas are indicated with different symbols and colors. The gray shaded area in (A) represents the estimated high severity proportion for fires in the natural range of variation (NRV).

Box 1

Densification of southern California conifer forests:

To link current fire return interval (FRI) with conifer stand densification, we identified an area of conifer forest in the Los Padres National Forest that did not burn during the entire fire record (current FRI = 109 years) and compared the density and size class distribution of trees in the 1930s to field surveys conducted in 2016. As has been found elsewhere in the state (McKelvey and Johnston, 1992; Minnich et al., 1995; Collins et al., 2011; Knapp et al., 2013), we expected that without fire, conifer stands would be denser and have more ladder fuels in 2016 than they did in the 1930s.

We selected four plots surveyed by the Vegetation Type Mapping Project (VTM) field crews in the 1930s (Kelly et al., 2005, 2008) that were less than 0.8 km from the plots surveyed on Frazier Mountain in 2016 (Fig. 6). As part of the VTM survey, crews tallied all trees (DBH > 10 cm) within a 0.08-hectare area into four diameter at breast height (DBH) categories (Wieslander et al., 1933). In 2016, crews conducted standard Common Stand Exams (See U.S. Forest Service, 2015) within 0.04-hectare area plots, where they measured the DBH of all trees with DBH > 10 cm. There were 41 plots surveyed in 2016, located in forest dominated by *Pinus jeffreyi* with *Abies concolor*.

In the four VTM surveys, *P. jeffreyi* was present in every plot and *A. concolor* was only present in one (25%). Across the plots, the average tree density was 160.6 trees per hectare (Fig. 7). In the 2016 surveys, *P. jeffreyi* was also found in every plot, while *A. concolor* occurred in 27% of the plots. The average tree density in 2016 was 393 trees per hectare, which is a 144% increase and statistically significantly greater than the 1930s average (Welch's *t*-test, $T\,(4.3) = -3.5$, $p < 0.05$). This is predominantly due to a significant increase in the density of trees in the two smallest DBH size categories (10–29.9 cm: Welch's *t*-test, $T\ (3.5) = -2.97$, $p < 0.05$; 30–59.9 cm: Welch's *t*-test, $T\ (10.0) = -3.30$, $p < 0.01$; Fig. 7). Trees 60–89.9 cm and > 90 cm in DBH did not significantly change in density between the two survey years (Kruskal-Wallis, 60–89 cm: $H\ (1) = 0.15$, $p = 0.69$; > 90 cm: $H\ (1) = 0.03$, $p = 0.87$; Fig. 7).

The increase in density of small diameter trees from 1930 to 2016 indicates that the conifer forest surveyed on Frazier Mountain is denser, with a higher proportion of small (and presumably young) trees now than in the 1930s. All the plots in the 2016 survey are in an area that has not burned in the fire record (1908–2016), making it likely that the lack of fire is the predominant cause of densification on Mount Frazier. This compliments similar studies in the San Bernardino Mountains, the San Gabriel Mountains, the mountains of San Diego and the Mount Pinos/Mount Abel area, where there were also significant increases in small diameter trees 60 years after VTM surveys (Minnich et al., 1995; Stephenson and Calcarone, 1999). The current high density of small and medium diameter trees increases the likelihood that the next fire on Frazier Mountain will burn at high severity (Miller et al., 2012b; Safford et al., 2012), potentially eliminating the few remaining large trees.

4. Discussion

Our analysis indicates that the fire regime in yellow pine and mixed conifer forests in southern California has departed from historical conditions. Fire suppression over the last century has lengthened the fire return interval (FRI) such that it is much greater today than the natural range of variation (NRV) for these forest types (Table 3). Over the last 32 years, high severity burn proportions have increased in fires across southern California conifer forests, while low severity burn proportions have decreased. In addition, the average high severity burn patch has increased in size, largely due to the higher frequency of extremely large patches in recent fires. Average conifer fire size showed a modest upward trend over the past century and the average fire size in the 2000s was 300 ha larger than the largest area predicted under NRV (Table 3). Collectively these changes represent a shift in the historical disturbance regime and have the potential to impact forest structure and post-fire recovery trajectories.

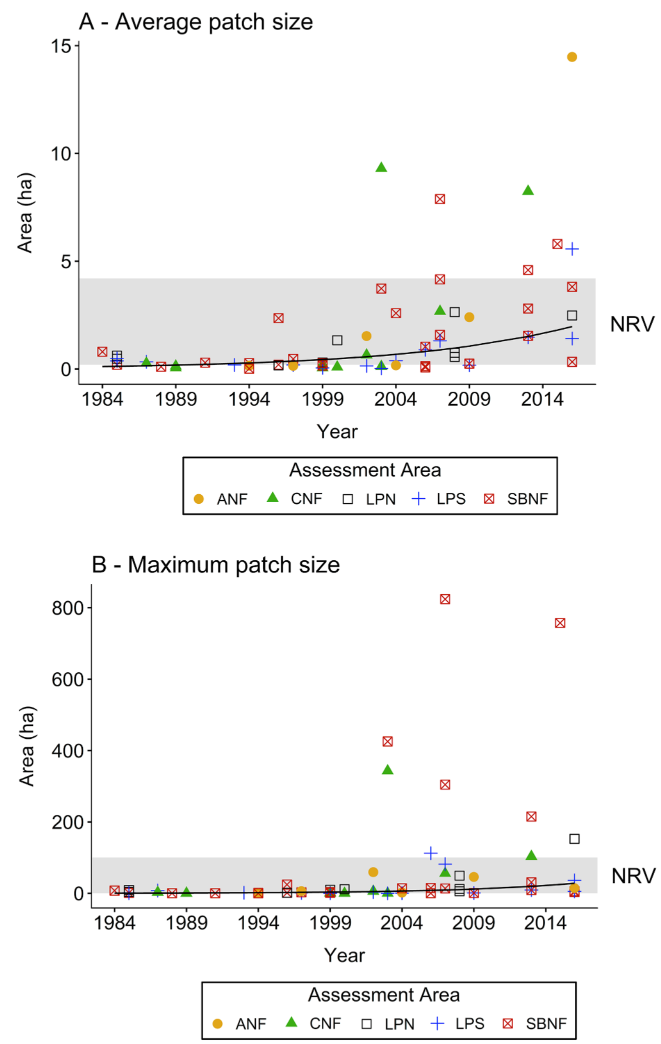

Fig. 4. (A) Average and (B) maximum high severity (> 90% basal area loss) patch size per fire over time. The back transformed regression line is shown for the log-transformed linear model fit of both average and maximum high severity patch size. Assessment area where each fire occurred is indicated by differing colors and shapes. The gray shaded area represents the natural range of variation (NRV) for (A) average and (B) maximum high severity patch size.

4.1. Fire return interval departure

The average current FRI for conifer forests in southern California is 78 years. Nearly half of the conifer forest area has not burned in over a century when historically it would have burned five to nine times in a period of the same length (Table 3). This is consistent with patterns

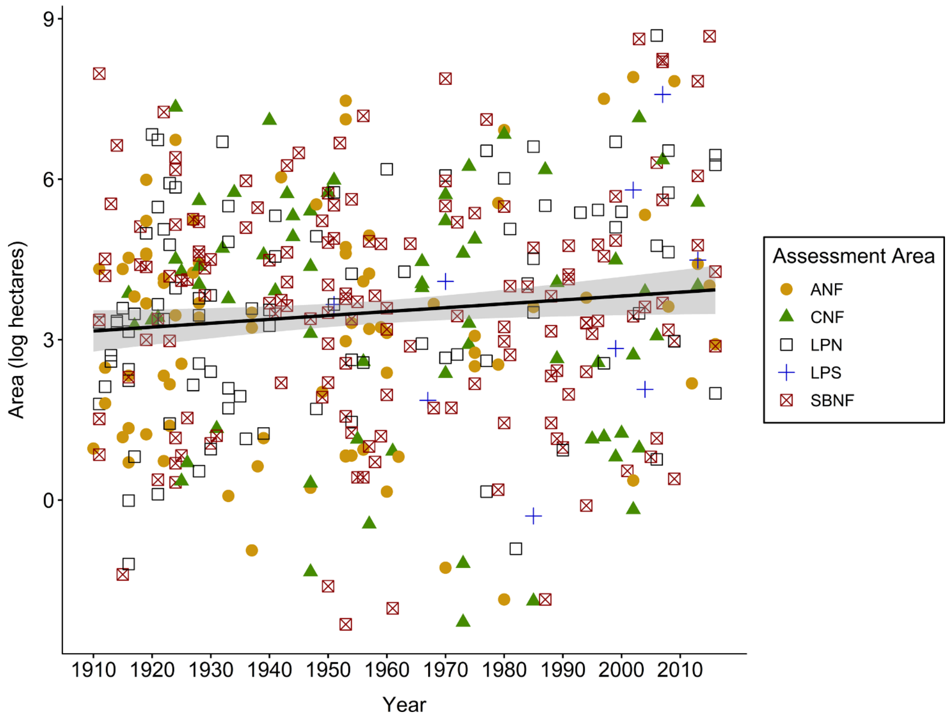

Fig. 5. Trend in conifer fire size in the five assessment areas from 1910 to 2016. All fires greater than 40 ha total in the record and that burned in conifer forest were included. The significant ($p < 0.05$) linear trend line with a 95% confidence interval is shown.

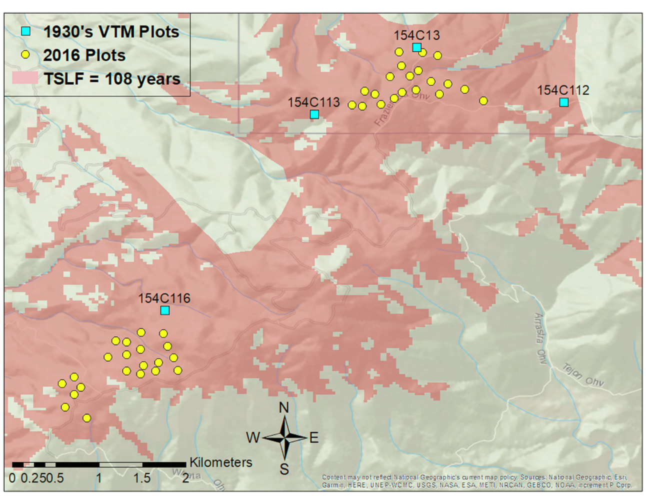

Fig. 6. Map of Frazier Mountain, Ventura County, CA. VTM plots (1930s) are marked by blue squares and 2016 plots are marked by yellow circles. All area shaded in red has not burned since 1908. TSLF = time since last fire.

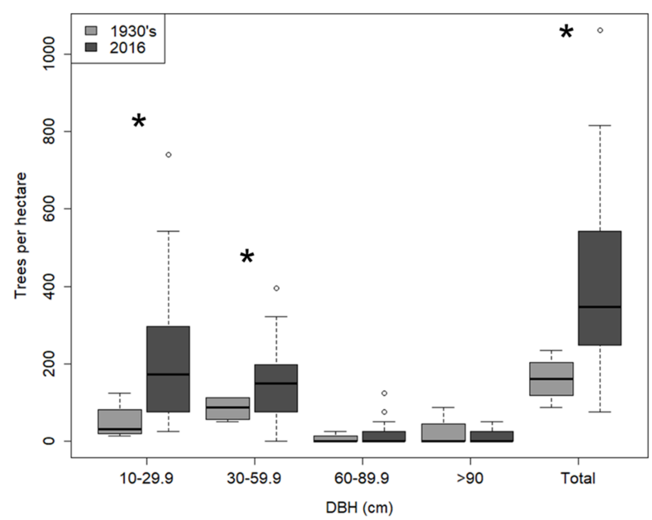

Fig. 7. Average tree density on Frazier Mountain by size class for plots surveyed in the 1930s (light gray) and 2016 (dark gray). Symbols (*) represent a statistically significant ($p < 0.05$) difference between years within each size class. Size classes with no symbol were not significantly different between years. The boxes bound the upper and lower quartiles, with the median being represented by a horizontal line. The hatched bars indicate the lowest and highest observations within 1.5*IQR (inter-quartile range) and the points represent the absolute maximum values in the data.

found for yellow pine and mixed conifer forests in southern California previously (Stephenson and Calcarone, 1999) and for these forest types throughout the state of California, where over 75% have not burned since 1908 (Steel et al., 2015). In contrast, the average FRI for intermediate – large sized fires in the Sierra de San Pedro Mártir (SSPM), where fire suppression did not occur for most of the century, is 9–24 years (Stephens et al., 2003), which is still within the NRV. However, recent fire suppression (since the mid-1970s) in the SSPM lengthened the current time since last fire (TSLF) to 52–53 years (Stephens et al., 2003; Rivera-Huerta et al., 2016), indicating that with the continuation of fire suppression, conifer forests in the SSPM may begin to resemble those in southern and northern California.

The lengthened FRI in southern California conifer forests today seems to be most driven by fire suppression efforts, as suppression of ignitions in conifer forest has been highly successful (Calkin et al., 2005; Stephens and Sugihara, 2006). It is improbable that a lack of ignitions is the source of the reduced fire frequency, as it is thought that modern lightning ignition patterns are similar to those in the pre-Euro-American settlement period (Minnich, 1988) and human ignitions have increased overall in southern California (Keeley, 2006). In contrast to shrublands, which are ignition limited and thus have been burning at higher frequencies than historically, mixed conifer forests have always had abundant ignitions, which are now suppressed before they can burn a significant amount of area (Mallek et al., 2013; Steel et al., 2015;

Table 3
Summary of current fire regime characteristics found for yellow pine and mixed conifer forests in this study compared to the natural range of variation (NRV) estimated from other studies for these forest types. Current fire return interval (FRI) was calculated using all fires in the record (1908–2016), which generally excludes fires < 4 ha after 1950 and < 40 ha before 1950. Current fire severity statistics represent fires > 400 ha in the period 2000–2016 and current fire size statistics represent fires > 40 ha in the period 2000–2016.

	FRI[a,b,c,d]	Mean % high severity[a,e,f]	Mean high severity patch size[a]	Maximum high severity patch size[a]	Fire size[a]
NRV	10–19 yrs	2.5–16%	0.2–4.2 ha	100 ha (rarely larger)	210–456 ha
Current (2000–2016)	78 yrs	24%	2.54 ha	824 ha	761 ha

[a] Safford and Stevens (2017).
[b] McBride and Laven (1976).
[c] McBride and Jacobs (1980).
[d] Skinner et al. (2006).
[e] Minnich et al. (2000).
[f] Rivera-Huerta et al. (2016).

Safford and Stevens, 2017).

The amount that fire frequency has deviated from the NRV may actually be more dramatic than presented in this analysis. First, areas that did not burn over the entire record were assigned an FRI of 109 years, even though the actual FRI was most likely longer. Second, we used a relatively conservative metric (maximum reference FRI) for determining whether the FRIs were within or outside NRV. For example, there were moist mixed conifer forests with a single burn in the 109-year fire history and a current FRI of 54 years that were determined to be within NRV (5–80 years). Even though these areas are still within the maximum reference FRI of 80 years, they are likely to be denser and compositionally different than those that have been burning close to every 16 years. Therefore, forest structure is probably highly variable, even among forests categorized as "within NRV" in this analysis.

The large amount of area that has not burned in over a century is likely to contain denser forests with more ladder fuels, consequently making them more vulnerable to stand-replacing fire in the future. There is evidence of densification in these unburned areas, especially of small-sized trees and shade-intolerant species (Box 1; Minnich et al., 1995; Goforth and Minnich, 2008). Densification increases understory fuels, fuel continuity and water stress in conifer stands (Allen and Breshears, 1998; Guarín and Taylor, 2005; Lydersen et al., 2013), which could result in severe, stand-replacing burns. Indeed, Steel et al. (2015) found TSLF to be a strong predictor of percent area burned at high severity in mixed conifer forests of the South Coast bioregion (where most of this study takes place). This relationship is strong for forest types that are more fuel-limited (such as mixed conifer), but this is not the case for more climate- or ignition-limited systems (such as red fir and redwood forests) (Steel et al., 2015). Interestingly, FRI is not a significant predictor of high severity burn percentage in the South Coast Bioregion (Steel et al., 2015). This discrepancy likely exists because forests with the same average FRI could have drastically different TSLFs, depending on when they burned in the record. For example, a forest with one fire in 1910 would have the same FRI as a forest with one fire in 2010, but the TSLF would be 106 years and 6 years, respectively. Assuming fuels increase with time, TSLF should reflect the amount of fuel loading at a certain point in time more accurately than average FRI. Therefore, while FRI is a useful metric for assessing fire frequency changes over time, TSLF may be more useful in predicting future fire behavior. Area classified in this study as "within NRV" for FRI may still be at risk of a large, high severity burn, especially if the last fire was early in the fire record. These findings indicate that fuel accumulation associated with densification over long fire-free intervals is likely linked to the extent of high severity burn in the next fire.

4.2. Fire severity

Most of the fires in southern California national forests after 2000 burned at well over 10% high severity, with the average from 2000 to 2016 reaching 24% and exceeding NRV estimates (Table 3). The average percent high severity burn over the entire period analyzed (1984–2016) was 18%, which is lower than yellow pine and mixed conifer forests in the Sierra Nevada, which burned at 33% high severity on average from 1984 to 2010 (Rivera-Huerta et al., 2016), though still above the NRV for high severity burn proportion. The percent area per fire burning at low severity decreased over time (Fig. 3), reflecting the continued loss of low severity fire in exchange for more high severity burn area. This is consistent with Mallek et al. (2013), who found a paucity of low and moderate severity burns in the Sierra Nevada compared to pre-settlement, and with a recent state-wide analysis by Steel et al. (2018). The occurrence of fires with large proportions of high severity burn seems to have increased since the turn of the century, as Stephenson and Calcarone (1999) noted that "stand-replacing crown fires" had yet to occur in southern California conifer forests as of their publication in 1999. In the SSPM, which has had little logging and only 30 years of fire suppression, fires from 1984 to 2010 in Jeffrey pine and mixed conifer forest burned an average of 3% at high severity, which is at the lower end of pre-Euro-American settlement estimates (Table 3, Leiberg, 1902; Show and Kotok, 1924; Mallek et al., 2013).

Although this study does not investigate the relationship between TSLF and burn severity directly (as in Steel et al., 2015), the average percent area burned at high severity in fires during 2016 was 49%, which is around what Steel et al. (2015) predicted for a TSLF of 75 years. The increase in high severity burn proportion over time seems to be most driven by the fires after 2000, indicating that either (1) fuel accumulation reached a threshold after 2000 that made fires more severe and harder to suppress than before, or (2) climate conditions became more conducive to fire ignition and spread and fuel loads were high enough to encourage crown fires. Since mixed conifer forests are naturally fuel-limited systems, it is unlikely that fire-conducive climate conditions alone would cause these forests to burn at higher severities. This is exemplified by the 2003 fire season, when mixed Jefferey pine forests in the SSPM suffered moderate fire severity effects, whereas mixed Jefferey pine forests in southern California suffered extremely high mortality, even though both were preceded by a severe multi-year drought. The high stand density of southern California forests and different fire weather (increased wind speeds) likely contributed to the discrepancy in burn effects (Stephens et al., 2008).

Forest structure, the biophysical environment and fire weather are other important factors influencing burn severity (van Wagtendonk et al., 2012; Lydersen et al., 2014; Kane et al., 2015). For a fire to spread into the canopy of a conifer forest, there usually needs to be sufficient ladder and surface fuels present to facilitate the upward spread of fire. However, Lydersen et al. (2014) found that in the Rim Fire of 2013 (in the Sierra Nevada), plume-dominated burning (resulting in extreme fire weather) was associated with high severity burns, even in areas with a restored fuel structure and FRI, indicating that assumed constraints on fire severity may be overwhelmed by burn conditions. In addition, larger-scale climate conditions (such as drought) can alter the effect that biophysical factors normally have on fire severity (Kane et al., 2015). In southern California, föhn-type winds (also known as Santa Ana and sundowner winds) are known to increase fire size (Jin et al., 2014), which could lead to more high severity burn area (Lutz et al., 2009; Miller et al., 2009b; Miller and Safford, 2012). Though the fire weather conditions are unknown for the fires in this analysis, they likely played a role in driving severity patterns, especially in forests with high stand density.

Collins et al. (2009) investigated the relationship between climate and TSLF for fires in the mixed conifer forests of Yosemite's Illilouette Basin and found that the probability of an area re-burning was more dependent on fire weather when the TSLF exceeded nine years – which is presumably when fuels ceased to be limiting. TSLF has also been found to be an important predictor of burn severity in other systems (van Wagtendonk et al., 2012; Lydersen et al., 2014; Kane et al., 2015; Steel et al., 2015). In southern California, where the average TSLF is 68 years in conifer forest, fire weather and long-term climate trends are likely to influence fire severity. From 2013 to 2016, the South Coast Bioregion (where most of the current study takes place) experienced above average temperatures and below average precipitation (Fig. A1), which coincided with a large number of fires burning with a proportion high severity greater than the NRV (Fig. 3A), and an average high severity burn proportion of 45% during that period. Conversely, weather patterns in the 2010–2012 time period had markedly cooler summer temperatures and more precipitation than other years after 2000. For instance, 2010 had the third lowest average summer temperature and the fourth highest annual precipitation in the 32 years analyzed and summer temperatures remained relatively cool for the proceeding two years (Fig. A1). These conditions coincided with dampened fire activity, such that only one fire burned during 2010–2012 and this fire lacked high severity burn (Fig. 3A). These examples reinforce the increasing importance of climate in fuel rich conifer forests.

4.3. High severity patch size

The large increase in the proportion of fires burning at stand-replacing severity in recent years is reflecting that either (1) recent fires have a greater number of "flare-ups" – small patches of stand-replacing burn scattered throughout the fire – or (2) the size of stand-replacing patches in recent fires has increased substantially. Based on the results of the patch size analysis, scenario 2 best explains what is happening in fires of southern California's conifer forests today.

In comparison to the NRV for high severity patch size in conifer forest, high severity patches today are much larger, and probably more frequent than they were pre-Euro-American settlement (Table 3). This is reflected in the increase in average and maximum patch sizes of stand-replacing burns since 1984. According to Safford and Stevens (2017), the NRV for high severity patch size in conifer forest of the Sierra Nevada was usually less than 5 ha, and rarely greater than 100 ha. This is true of the southern California conifer forest fires in years 1984–2002. However, in most years after 2002, high severity patches were generally larger and larger patches more frequent than the NRV. Similar patterns have been found in the Sierra Nevada, where high severity patches doubled in average size from the period 1984–1993 (2.8 ha) to the period 1995–2004 (5.3 ha) (Miller et al., 2009b). Despite the short duration for fire suppression in the SSPM, the conifer forests there have also experienced an increase in average size of high severity patches from 1984 to 2010 (Rivera-Huerta et al., 2016).

The increase in large patches of stand-replacing burns post-2002 may indicate that forests burning in recent years are much more homogenous in terms of fuel loads, allowing high severity fires to burn large, continuous patches as opposed to flaring up in small areas of dense forest. For example, one high severity patch may make up as much as 78% of the total burn area (Sand Fire, 2016). This distinction is important in evaluating recovery trajectories in high severity burn areas, as some small high severity patches are needed for conifer regeneration, but large stand-replacing patches could impede or greatly delay forest recovery (Keeley, 2006; Collins et al., 2017). Conifer regeneration, especially of species in the genus *Pinus*, can be severely reduced in large, high severity burn areas due to limited seed dispersal distance, lack of seed source trees, seed predation, and competition with shrubs (Goforth and Minnich, 2008; Zwolak et al., 2010; Collins and Roller, 2013; Crotteau et al., 2013; Chambers et al., 2016). Regeneration densities are likely to experience decline as stand-replacing patches get larger, and the distance to the nearest seed tree increases, especially for species with limited dispersal abilities (Bonnet et al., 2005; Donato et al., 2009; Chambers et al., 2016). This could ultimately lead to a major shift in species composition or type conversion to a different vegetation type. Large, stand-replacing burn patches also disrupt habitat connectivity and heterogeneity, reducing available habitat for species dependent on mixed conifer forests (Kalies et al., 2010; Fontaine and Kennedy, 2012). Even if these large patches of forest eventually recover, there will be many years in which they are not functioning as healthy mixed conifer habitat. This is especially threatening to the conifer forests of southern California, which already exist in small and fragmented sky islands.

The northern district of the Los Padres National Forest (LPN) provides an interesting comparison to the other assessment areas, as only 522 of the 2539 ha of conifer forest has a current FRI greater than historic, making most of the conifer forest within the NRV. The discrepancy between FRI in the LPN and other assessment areas may be owed to the relative remoteness of the forest, lack of an extensive road network for firefighting, and steep topography that can impede suppression efforts. Also, just over 80% of the LPN is protected by the Ventana and Silver Peak Wilderness Areas (while all other forests have 13–37% wilderness), which can restrict the scope of fire suppression activities. These challenges to containment may have encouraged wildfires to burn more frequently within the LPN. However, high severity burn patterns in the LPN did not appear to significantly differ from the other assessment areas. This reiterates the notion that FRI, while useful for comparing current fire frequency to that of the past, may not be an accurate predictor of future fire behavior. In addition, the discrepancy between FRI and fire severity patterns in the LPN could be an indication that climate is becoming more influential in driving fire severity. When evaluating FRI more carefully for the LPN, we found less than one percent of conifer forests on the LPN burned prior to 1970, and that fire activity picked up substantially in the mid-1970s with a few large fires (> 30,000 ha) occurring in recent years (Marble Cone (1977), Gorda-Rat (1985), Kirk (1999), Basin-Indians Complex (2008), and Soberanes (2016)). The occurrence of several recent fires would drive the FRI down, but fuel buildup since the start of fire suppression and changes in climate could still result in these fires having large portions of high severity burn.

4.4. Fire size

The modest increase in conifer fire size over the 32 year period analyzed and increased occurrence of large fires (> 456 ha) after 2000 suggests that unusually large fires are becoming more common. The average pre-Euro-American settlement mixed conifer fire size in the Sierra Nevada was 210 ha and contemporary references range from 221 ha (SSPM) to 456 ha (Illilouette Creek, Yosemite National Park) (Safford and Stevens, 2017). On U.S. Forest Service land in southern California, 27% of the fires after 2000 exceed the highest of these estimates by 72–5469 ha (Table 3). Extremely large fires could be attributed to a higher continuity of fuels in conifer forests in recent years, due to densification, or to recent weather patterns that increase vegetation flammability (drought) and allow rapid fire spread (föhn-type winds). While it is unclear how many of the fires in this analysis correspond to strong wind events, Jin et al. (2014) report a spike in the average fire size associated with Santa Ana driven fires and a slight increase in the size of non-Santa Ana driven fires after 2000. In both of these cases, the median fire size has remained fairly stable through time suggesting that small fires continue to dominate, and large fires, likely corresponding to Santa Ana wind events, have become increasingly common (Jin et al., 2014).

Total burn area and high severity patch size are probably not mutually exclusive, as the largest patches in recent fires are dominated by the highest burn severity (> 90% basal area loss). Therefore, it may be that the higher severity of recent fires is allowing these fires to grow larger due to the rapid and uncontrollable spread of high severity crown fires, as opposed to low severity ground fires that are more easily and quickly suppressed. However, the weak positive trend in conifer fire size over time signifies that small fires are still quite frequent as well.

4.5. Management implications

In southern California, conifer forests exist in small stands only at the highest elevations (> 1500 m) of the Transverse and Peninsular Ranges, separated from other stands by foothills and valleys of chaparral, oak woodland, and grassland (Minnich and Everett, 2001). Large patches of stand-replacing fire in these sky islands may render the forests unrecoverable if no seed sources are left in the nearby landscape. If forests do recover, high severity burn areas may facilitate recovery of even denser forests or could alter the recovery trajectory to a different vegetation type altogether (Savage and Mast, 2005; Goforth and Minnich, 2008; White and Long, 2018). Several species, such as the California spotted owl (*Strix occidentalis occ.*), Northern goshawk (*Accipiter gentilis*) and White-headed woodpecker (*Picoides albolarvatus gravirostris*), depend on these conifer islands for habitat (Gutiérrez and Pritchard, 1990; Stephenson and Calcarone, 1999). Increased frequency and expansiveness of high severity fire could negatively impact important wildlife populations and reduce and fragment the conifer forests they depend on. There is also a suite of ecosystem services that could be negatively impacted by high severity fire in conifer forests including nutrient cycling, carbon sequestration, and recreation value (Englin et al., 2001; Schimel and Braswell, 2005; Campbell et al., 2007; North and Hurteau, 2011; Gonzalez et al., 2015).

For forests that have burned in large high severity patches, reforestation efforts should be targeted in areas that are least likely to naturally regenerate (far from living seed sources), that contained conifer forest prior to significant fire suppression, and that are still climatically suitable for mixed conifer and yellow pine species (White and Long, 2018; North et al., 2019). Strategies such as planting in clumped patterns and in microrefugia from drought and fire have been proposed in order to increase seedling success and reduce the likelihood of future widespread tree mortality due to fire (North et al., 2019). However, an increasing amount of the U.S. Forest Service budget is being used for fire suppression and thus less area can be planted each year (North et al., 2019). Therefore, fuel management in the nearby unburned forests that will reintroduce a low severity, frequent fire regime is also needed (White and Long, 2018).

Areas that have been managed to include fire on the landscape at a more natural frequency, such as Wildland Fire Use (WFU) areas, have provided examples of how maintaining a natural fire frequency can help prevent unnaturally large and severe forest fires (Collins, 2007; Miller et al., 2012a; Steel et al., 2018). Specifically, Collins et al. (2009) found no increase in the percent area burned at high severity from 1974 to 2004 in mixed conifer forests of Ililouette Creek Basin (WFU management area), whereas the larger Sierra Nevada region (which includes Ililouette Creek Basin) did show an increase in percent area burned at high severity for mixed conifer forests from 1984 to 2004 (Miller et al., 2009b). In addition, Collins (2007) found that the FRI in Ililouette Creek Basin returned to pre-suppression levels (6.8 years) once WFU policies were established after a long fire-free period (1881–1972). These findings, along with the positive correlation between TSLF and probability of reburning, indicate that WFU management may lessen the increasing occurrence of stand-replacing crown fires by allowing recent fires to constrain the spatial extent and severity of future fires (Collins et al., 2009). It is often deemed impractical to establish WFU areas in mixed conifer forests within southern California due to their close proximity to urban areas and overlap with chaparral ecosystems, which are currently experiencing too much fire (Safford and Van de Water, 2014). Yet there are areas within the southern California National Forests where WFU could be implemented as part of forest restoration and community and watershed protection. It would be wise to identify these locations in advance of wildfires.

Fuel treatments in mixed conifer forest primarily focus on restoring forest stand structure to one resembling pre-suppression stands (see Box 1) either via stand thinning, prescribed burning, or both. Fuel treatments that target removal of surface and ladder fuels (usually through thinning and burning) have proven effective in reducing fire severity and tree mortality in mixed conifer forests throughout California (Franklin et al., 2006; Safford et al., 2012). Although prescribed burns and thinning treatments provide some benefit when used separately, they are generally more effective when used in combination (North et al., 2007; Knapp et al., 2017). Prescribed burns in an untreated forest with ladder fuels could result in a higher severity fire than desirable and mastication without burning leaves large amounts of fuel on the ground to carry the next fire (Stephens and Moghaddas, 2005; Innes et al., 2006; Schwilk et al., 2009; Safford et al., 2012). However, there are many constraints to both management strategies including smoke regulations, wilderness designated areas, cost, personnel shortages and other logistics (Quinn-Davidson and Varner, 2012; North et al., 2015). It is also uncertain whether these fuel treatments can restore additional aspects of conifer forests such as species composition and ecological function (North et al., 2007; Van Mantgem et al., 2011), though success has been documented (Fontaine and Kennedy, 2012; Knapp et al., 2017). Large stand-replacing fires remain one of the major threats to mixed conifer forests in southern California, especially as fire activity is expected to increase with future changes in climate (Westerling, 2016). Therefore, focusing on overcoming fuel management constraints in areas that are likely to be most vulnerable to high severity impacts may be the best strategy for preventing future degradation and fragmentation of these valuable habitats.

Declaration of interest

None.

Acknowledgements

Many thanks to Hugh Safford and the lab of Carla D'Antonio for their guidance on early versions of the manuscript. This manuscript benefitted from the valuable input of anonymous reviewers. We also acknowledge Paris Krause, Michael DeNicola and Ryan Fass for their help in the field. Funding for this work was provided by the USDA Forest Service, southern California province.

Appendix A

See Fig. A1.

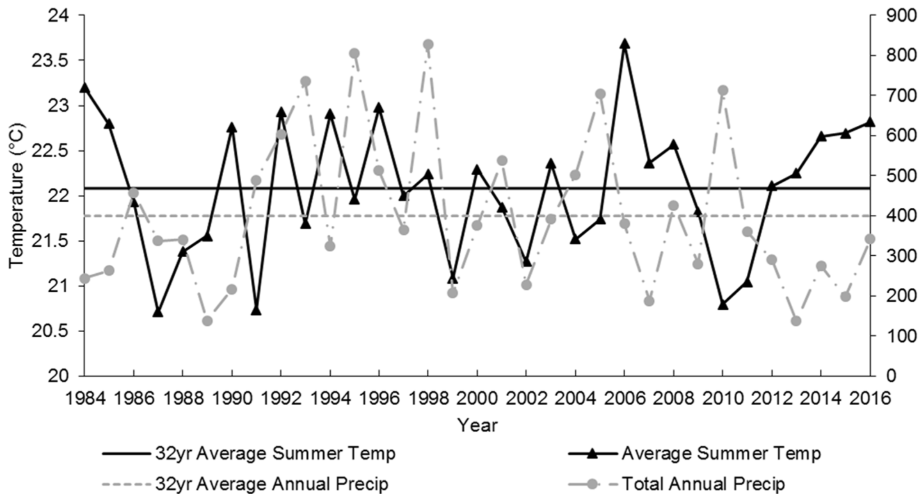

Fig. A1. Average summer (June, July, August) temperature (°C) and total yearly precipitation (mm) for each year 1984–2016 and averaged across the 32-year period. Climate data presented is for the South Coast Bioregion and was obtained from California Climate Tracker (https://wrcc.dri.edu/monitor/cal-mon/).

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