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High severity burn area and proportion exceed historic conditions in forests of Sierra Nevada and adjacent ranges (USA)

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ABSTRACT

Although fire is a fundamental ecological process in western North American forests, climate warming and accumulating forest fuels due to fire suppression have led to wildfires that burn at high severity across larger fractions of their footprint than were historically typical. These trends have spiked upwards in recent years and are particularly pronounced in the Sierra Nevada-southern Cascades ecoregion of California, USA and neighboring states. We assessed annual area burned and percentage of area burned at high and low-to-moderate severity for seven major forest types in this region from 1984 to 2020. We compared values for this period against estimates for the pre-Euro-American settlement (EAS) period prior to 1850 and against a previous study of trends from 1984-2009. Our results show that total average annual area burned remained below pre-EAS levels, but that gap is decreasing (i.e., c. 14% of pre-EAS for 1984-2009, but 39% for 2010-2020 [including c. 150% in 2020]). Although average annual area burned has remained low compared to pre-EAS, both the average annual *area* burned at high severity and the *percentage* of wildfire area burned at high severity have increased rapidly. The percentage of area burned at high severity – which was already above pre-EAS average for the 1984-2009 period – has continued to rise for five of seven forest types. Notably, between 2010 and 2020, the average annual area burned at high severity exceeded the pre-EAS average for the first time on record. By contrast, percentage of area that burned at low-to-moderate severity decreased, particularly in the lower elevation oak and mixed conifer forest types. These findings underline how forests historically adapted to frequent low-to-moderate severity fire are being reshaped by novel proportions and extents of high severity burning. The shift toward a high severity-dominated fire regime is associated with ecological disruptions, including changes in forest structure, species composition, carbon storage, wildlife habitat, ecosystem services, and resilience. Our results underscore the importance of finding a better balance between the current management focus on fire suppression and one that puts greater emphasis on proactive fuel reduction and increased forest resilience to climate change and ecological disturbance.

KEY WORDS: annual area burned; Cascade Range; fire ecology; fire regime; mixed conifer; natural range of variation (NRV); Sierra Nevada; wildfire; yellow pine.

INTRODUCTION

Fire is a fundamental ecological process that has shaped the forests of western North America for millions of years (Keeley and Safford 2016). The range of forest types found in this region is related to the interactions of multiple factors, including climate, topography, species pool, productivity, and disturbance history. These factors influence and are influenced by the fire regime, which is defined by long-term temporal, spatial and fire intensity patterns of burning that typify an ecosystem and shape its composition, structure, and function (Agee 1993, van Wagtendonk et al. 2018b, Miller and Safford 2020). Over the past century, however, fire regimes in many western North American forests have departed from their natural range of variation. These modern changes have been driven largely by anthropogenic factors – e.g., halting of Native American burning, adoption of fire suppression policies, timber extraction and forest management practices, changing ignition patterns, and climate change – that have altered the way fire interacts with forests (Abatzoglou and Williams 2016, Klimaszewski-Patterson and Mensing 2016, Stephens et al. 2016, Balch et al. 2017, Parks et al. 2018a).

A century of fire exclusion in the western United States has led to changes in fire frequency and burn severity, two key components of the fire regime (Mallek et al. 2013, Safford and Van de Water 2014, Steel et al. 2015, Parks et al. 2018b). The reduction or removal of regular fire has caused significant changes in forest structure, composition, diversity, and function. For example, changes in forest fire regimes have promoted shifts in forest stand density, fuel loading and continuity, and habitat heterogeneity (Johnstone et al. 2010, Hanberry 2014, Cassell et al. 2019, Stevens et al. 2019), and such shifts may be exacerbated by climate warming (van Mantgem et al. 2013, Halofsky et al. 2020). In yellow pine (i.e., ponderosa pine and/or Jeffrey pine; *Pinus ponderosa*, *P. jeffreyi*) and mixed conifer forests (the above species, plus, among other species, sugar pine [*P. lambertiana*], incense cedar [*Calocedrus decurrens*] and white fir [*Abies concolor*]), wildfires have grown in size and are more likely to include larger contiguous patches of high severity burning than fires that burned prior to the application of fire exclusion policies (Steel et al. 2018). Major changes in the yellow pine and mixed conifer fire regimes have negatively impacted forest resilience, tree regeneration, species distributions (Miller et al. 2009b, Keeley and Syphard 2016, Welch et al. 2016, Boisramé et al. 2017, Thorne et al. 2017, Steel et al. 2018), threatened and endangered animal populations (Blomdahl et al. 2019, Jones et al. 2020), plant species diversity (Richter et al. 2019, Miller and Safford 2020), and ecosystem services (Wu and Kim 2013, Richter et al. 2019, Rakhmatulina et al. 2020).

To better understand how wildfire patterns in the Western U.S. have been changing, Mallek et al. (2013) compared modern vs. historical patterns of annual area burned and wildfire severity in the Sierra Nevada-Southern Cascades ecoregion of eastern and northeastern California and neighboring states. For the purposes of this study, "historical" refers to the time before significant Euro-American settlement ("pre-EAS") of the study region, i.e., prior to ~1850. We also use the natural range of variation (NRV) to refer to the forest structure and composition that existed pre-EAS, as defined by Safford and Stevens (2017; NRV as we use it includes the contributions of Native Americans to the fire regime). Mallek et al. (2013) found that while overall annual area burned in the period from 1984 to 2009 was only about 14% of what burned in an average pre-EAS year (Stephens et al. 2007), the percentage of area burning at high severity was much higher in 1984-2009 (29% area-weighted average vs c. 7% pre-EAS). Another important finding of the study was that differences between the 1984-2009 and pre-EAS periods depended on the forest type in question. For example, low- to mid-elevation forest types (i.e., oak woodlands, yellow pine, mixed conifer) were burning much less frequently than under pre-EAS conditions, but at much greater severity when they burned. By contrast, the authors found that higher elevation forest types (i.e., red fir [*Abies magnifica*], lodgepole pine [*Pinus contorta*], subalpine forest), which have longer natural fire return intervals, experienced relatively minor changes in fire frequency, and modern fire severity patterns were not statistically discernable from pre-EAS patterns.

The annual area burned by wildfires in California has increased considerably since 2009 – the last data year considered by Mallek et al. (2013) – and nine of the ten largest fires in the State's history have occurred since then (CalFire 2021b). Over the last decade, severe wildfires have emitted hundreds of millions of Mg of carbon and other pollutants into the atmosphere (CARB 2020) and caused widespread ecological damage to forests, soils, and sensitive animal habitat (Coppoletta et al. 2016, Jones et al. 2016, Welch et al. 2016, Abney et al. 2019, Dove et al. 2020, Jones et al. 2020, Steel et al. In press). The Sierra Nevada-Southern Cascades ecoregion (i.e., the study region; Fig. 1) has experienced similar trends in wildfire as California as a whole, with regional variation driven by complex topography, prominent altitudinal gradients, and geographic clines in the distribution of climates and ecosystems (North et al. 2016, Safford et al. 2021). The recent large, high severity fires in the region, combined with the availability of eleven additional years of fire severity data, led us to revisit and build on the analyses conducted by Mallek et al. (2013).

In this study, we provide an updated assessment of area burned and fire severity patterns for the Sierra Nevada-Southern Cascades region over the 37-year period from 1984-2020. Our goal is to provide the most current and refined assessment possible vis-à-vis changing fire regimes for resource managers who struggle to balance short-term conservation and risk aversion priorities with long-term considerations of ecosystem sustainability under rapid environmental change. Specifically, we evaluate whether previously identified trends in burned area and fire severity (Mallek et al. 2013) continue as before or whether they have slowed or accelerated. Based on recent investigations (e.g., Steel et al. 2015, 2018; Safford and Stevens 2017) and personal observations, we hypothesized that for the study region between 2010 and 2020: (1) annual area burned would increase relative the 1984-2009 period, but would still lag behind average annual area burned during the pre-EAS period; (2) the percentage of wildfire area burned at high severity would increase for all forest types, but proportionally more in low- and middle-elevation forests, where forests have experienced greater departures from historical fire return intervals due to a century of fire exclusion and climate warming; and (3) the average annual area burned at high severity (AAHS) would approach or exceed historical pre-EAS high severity area in low- and middle-elevation forest types, but perhaps not in high-elevation forests, given their longer fire return intervals and relative lack of fire for the last decade.

METHODS

Study Area

The study area comprises approximately 120,000 km^2 of the Sierra Nevada and Southern Cascade Mountain ranges and adjacent forested areas, and includes eleven National Forests and four National Parks (Fig. 1). This is the same study area used by Mallek et al. (2013) and Miller et al. (2009), and is based on the Sierra Nevada ecoregion as defined by the Sierra Nevada Ecosystem Project (SNEP 1996) and the Sierra Nevada Forest Plan Amendment (SNFPA; USDA 2004). The region stretches from Tehachapi Pass at the southern end of the Sierra Nevada to the California-Oregon border in the north, and from the Sierra Nevada Foothills on the eastern edge of California’s Central Valley to the westernmost ranges of the Great Basin, including a strip of the Humboldt-Toiyabe National Forest in western Nevada.

Elevations in the study area range from 300 m above sea level along the western edge to >4000 m along the Sierra Nevada crest. The climate is mostly Mediterranean-type with warm, dry summers and cool, wet winters. Vegetation in the study area is characterized by forests, woodlands, shrublands, and grasslands, although the latter two are not analyzed here as our focus is on forested areas. Oak (*Quercus* spp.) woodlands dominate lower elevations along the western boundary of the study area, transitioning to yellow pine (ponderosa pine [*Pinus ponderosa*] and Jeffrey pine [*P. jeffreyi*]) and mixed conifer forests at higher elevations (Table 1). Red fir (*Abies magnifica*) dominated forests are found above about 1800 m and transition into lodgepole pine (*P. contorta*) and different types of subalpine forest at the highest elevations. Pinyon pine (mostly *P. monophylla*) and juniper (*Juniperus* spp.) woodlands occur at moderate elevations in the north and east of the study area. Yellow pine-dominated forests are also found on the east side of the study area, between about 1500 m and 2500 m elevation (Table 1; North et al. 2016, Safford et al. 2021).

Analyzed forest types and their areas are based on the LANDFIRE Biophysical Settings (BpS) map (www.landfire.gov, v. 105, accessed 11/1/2019), which represents modeled potential natural vegetation incorporating climate, soils, topography and hypothetical pre-EAS (pre-1850) fire regimes (Rollins 2009). BpS types were grouped into pre-settlement fire regime types defined by Van de Water and Safford (2011) using crosswalks in that paper and Mallek et al. (2013). We analyzed the same seven forested pre-EAS fire regimes as Mallek et al. (2013) to facilitate comparisons with that study: oak woodland (OW); dry mixed conifer (DMC); moist mixed conifer (MMC); yellow pine (YP); red fir (RF); lodgepole pine (LP); and subalpine (SA).

While the BpS vegetation delimitations and pre-EAS fire regime estimates are the best-available for this analysis, we nevertheless stress that these parameters are based on a combination of incomplete data and historical reconstructions that necessarily mean that they should be viewed as approximations subject to refinement as new data and analytic methods become available.

Analysis

The data analyzed by Mallek et al. (2013) covered the time period from 1984 to 2009. In this study, we used the most recent burn severity data available to consider eleven additional years of wildfire extent and severity for the same region, extending the length of the period analyzed to the 37-years from 1984 to 2020. Wildfire perimeters and total annual area burned (AAB) were obtained from the most recent version of the California Fire Perimeter database (CalFire 2021a). The primary source of burn severity data for this analysis was the "Vegetation Burn Severity – 1984 to 2017" geospatial data layer (USDA 2018) developed by Region 5 (Pacific Southwest) of the United States Forest Service (henceforth "Forest Service"). For the 2018-2020 fire years, we estimated burn severity using Google Earth Engine following Parks et al. (2018c, 2021). A comparison of 50 randomly selected fires from 1985-2017 showed high similarity between the legacy and Earth Engine-derived severity estimates (R = 0.95; Figure S1). For both datasets, severity data were calculated from Landsat Thematic Mapper imagery using the Relative differenced Normalized Burn Ratio (RdNBR) and were classified into severity levels using previously field-calibrated thresholds (Miller and Thode 2007, Miller et al. 2009a). The dataset includes the entire area of all wildfires ≥80 ha in size that occurred at least partially on Forest Service lands or in Yosemite National Park in the study area, plus an incomplete collection of fires <80 ha (see: Miller et al. 2009b, Miller and Safford 2012, Mallek et al. 2013).

We did not include Lassen or Sequoia-Kings Canyon National Parks because fire severity mapping for fires <400 ha has not been carried out in these landscapes.

We used burn severity data to calculate hectares burned in four fire-severity classes (per Miller and Thode 2007) for each forest type for each year from 1984 to 2020. Like Mallek et al. (2013), we condensed the severity data into two categories: 1) "annual area burned at low-to-moderate severity" (AALMS), a single category that combines classes I ("no change"), II ("low severity" = <25% tree mortality), and III ("moderate severity" = 25 to <95% tree mortality); and 2) "annual area burned at high severity" (AAHS), which represents class IV burned areas that experienced stand-replacing fire, where tree mortality at the time of postfire imagery acquisition was ≥95% (Miller et al. 2009a). For all areas analyzed for severity, total annual area burned (AAB) for a forest type was equal to AAHS plus AALMS.

For the pre-EAS burn data, we used the same numbers and methods used by Mallek et al. (2013), with a few updates to the average fire rotation period based on new science (see below; Table 2), defined as the number of years required to burn an area equal to the forest extent in question (Agee 1993). We used the pre-settlement fire regime types cross-walked from the LANDFIRE BpS map (see above) and divided the total area of each type by its pre-EAS fire rotation period (Table 2) to estimate average AAB_{Pre}. Thus, for an area, A, associated with a pre-EAS fire regime rotation period of Y years, $AAB_{Pre} = A/Y\ ha \cdot yr^{-1}$

Whereas the burn severity class data for the modern period are imagery-based, our estimates of characteristic burn severity for the pre-EAS period were made from historical records, the scientific literature, and models. We started from Table 3 in Mallek et al. (2013) and consulted the literature for updated information. Based on new data summarized in Safford and Stevens (2017), we did not change the Mallek et al. (2013) estimates of characteristic burn severity levels for oak woodland, dry and moist mixed conifer, or yellow pine. However, we adjusted the values for red fir, lodgepole pine, and subalpine forest based on new information (Safford and Stevens 2017, van Wagtendonk et al. 2018a, Meyer and North 2019). These sources yielded AAB_{Pre} and $AAHS_{Pre}$, from which we calculated $AALMS_{Pre}$ (AAB minus AAHS), percentage of area burned at high severity (PHS; = AAHS/AAB), and percentage of area burned at low-to-moderate severity (PLMS; = AALMS/AAB).

As in Mallek et al. (2013), we intersected the SNFPA polygon with the LandFire BpS raster data set (version 105) to define the major vegetation classes. We also added fire severity data for a few fires that burned in the study area during the Mallek et al. time frame but were not analyzed for severity in that study. No areas outside of the study area polygon were analyzed or reported, even if part of a given fire burned inside the boundary. As in Mallek et al. (2013), we included all fires >80 ha that intersected both the study area polygon and the Forest Service or Yosemite National Park jurisdictions, while those that did not were excluded from the severity analysis (Figure 1). Sections of fires that fit these criteria but fell outside of the study area boundary were excluded from the analyses. Our data for the period 1984-2009 are nearly the same as, though not identical to, those used by Mallek et al. (2013) because of subsequent updates to the Forest Service fire severity database and our revised PHS estimates for the pre-EAS period for red fir, lodgepole, and subalpine forests. As in Mallek et. al. (2013), we included all fires >80 ha that intersected both the study area polygon and the Forest Service or Yosemite National Park jurisdictions, while those that did not were excluded from the analysis (Figure 1).

Trend Assessment

We used a Bayesian approach to assess trends in AAB, PHS, and PLMS for the full study region and by forest type across the expanded modern period (1984-2020). For this assessment we fit generalized linear models with year as the fixed effect of interest. Area response variables were log-transformed and modeled using a Gaussian error structure. Proportion burned area models utilized aggregated binomial regression and a logit link function with hectares of AALMS or AAHS constituting "successes" and AAB constituting "trials" for a given year and forest type. In all models we included a first-order temporal auto-regressive term to account for potential temporal auto-correlation.

Models were estimated using Hamiltonian Monte Carlo sampling in Stan via the BRMS package and program R (Bürkner 2017, R_Core_Team 2019, Stan_Development_Team 2019). We specified weakly regularizing priors to prevent model overfitting. Models were run with three chains, each for 3000 samples with a warmup of 1500. Trace-plots and R-hat values were assessed for proper mixing and model convergence.

RESULTS

Average annual area burned (AAB) during the 2010-2020 period – though still well below historic levels (AAB_{Pre}) – increased by more than 200% over the 1984-2009 period for all forest types combined. $AAB_{2010-2020}$ was especially impacted by the record-breaking 2020 wildfire season (Table 3, Fig. 2a, Safford et al. 2022), which contributed significantly to the large overall increases in AAB across the expanded modern period (1984-2020) for all forest types, individually and combined.

The average annual percent of area burned at high severity (PHS) increased for all forest types combined between the 1984-2009 and 2010-2020 periods (Table 3, Fig. 3). When these two periods are considered together, $PHS_{1984\text{-}2020}$ averaged 27% - almost four times the combined PHS_{Pre} average of 7%. For some forest types, however, PHS did not increase from 1984-2009 to 2010-2020. For yellow pine, for example, PHS was virtually unchanged across the two modern periods (though still much higher than pre-EAS values). $PHS_{2010\text{-}2020}$ also decreased for lodgepole and sub-alpine forests compared to $PHS_{1984\text{-}2009}$ (Figs. 3, S2, S3). By contrast, $PHS_{2010\text{-}2020}$ trended noticeably upward for the oak woodland, dry and moist mixed-conifer and red fir forests. The complement of PHS, PLMS (percentage of area burned at low-to-moderate severity), showed a decreasing trend overall from 1984-2009 to 2010-2020, with yellow pine, lodgepole pine and subalpine forests as individual exceptions.

The average annual area burned at low-to-moderate severity (AALMS) increased since 2009 across all forest types, but remained well below historical ($AALMS_{Pre}$) levels. Notably, for 2010-2020 the average annual area burned at high severity (AAHS) exceeded pre-EAS levels for the first time on record (Fig. 4). These trends are visible for all forest types combined, as well as for the dry and moist mixed conifer, yellow pine, and red fir forest types separately (Table 3, Fig. 4b).

For the 2010-2020 period, all forest types showed appreciable increases in AAB compared to 1984-2009 (average increase: 410%; range: 56 – 905%, Table 3). AAB increased from 13.6% of AAB_{Pre} during 1984-2009 to 39% of AAB_{Pre} during 2010-2020 (including c. 150% of AAB_{Pre} in 2020 alone; Table 3). For the expanded modern period, 1984-2020, annual area burned averaged 20.6% of AAB_{Pre} across forest types and ranged from 14.6% (oak woodland) to 34.8% (red fir). Thus, despite recent increases, average annual area burned continues to be less than half of AAB_{Pre}, due to an ongoing deficit in low-to-moderate severity fire (Fig. 4a).

A comparison of modeled trends across the 1984-2020 period for burned area and burn severity revealed similarities and differences among forest types (Figs. 5, S3). For example, $AAB_{1984\text{-}2020}$ and $AAHS_{1984\text{-}2020}$ showed positive trends over time across all forest types, though the amount of increase varied in absolute and relative terms. Subalpine, lodgepole pine, and moist mixed conifer – in that order – showed the most robust increases in $AAB_{1984\text{-}2020}$, while dry mixed conifer, moist mixed conifer, and red fir had the strongest positive trends in $AAHS_{1984\text{-}2020}$. For all forest types combined, $PHS_{1984\text{-}2020}$ trended positive for the expanded evaluation period. The results for this trend and $AAHS_{1984\text{-}2020}$ were still positive and significant when the 2020 fire year was excluded. For $PLMS_{1984\text{-}2020}$, in terms of individual forest types, only yellow pine showed a convincingly stable trend; all other forest types showed decreasing trends.

DISCUSSION

Our findings support previous assessments of burned area and severity in California (Miller et al. 2009b, Mallek et al. 2013, Miller and Safford 2012, Steel et al. 2015), but go further in demonstrating that high severity trends have surpassed historical rates and have stepped up markedly since 2009. While part of this jump is due to the record 2020 fire year (Safford et al. 2022), the increases in high severity fire in recent years are remarkable even when 2020 is not considered. The most salient results of our assessment are that: (1) average annual area burned ($AAB_{1984\text{-}2020}$) remains well below pre-EAS averages, although the disparity is decreasing; (2) for the newly evaluated 2010-2020 period, average annual area burned at high severity ($AAHS_{2010\text{-}2020}$) exceeded $AAHS_{Pre}$ for the first time on historical record, particularly in low and middle elevation forest types; and (3) the percentage of area burned at high severity during the expanded modern period ($PHS_{1984\text{-}2020}$) is well above pre-EAS levels and trending upward for six of seven forest types analyzed (Fig. S2). Conversely, the percentage of area burned at low-to-moderate severity ($PLMS_{1984\text{-}2020}$) shows a decreasing trend that adds to an already gaping deficit in the type of burning that is fundamental to the conservation and restoration of most of the Sierra Nevada-Southern Cascades forest base (van Wagtendonk et al. 2018b).

Our data show that the gap between $AAB_{1984\text{-}2020}$ and AAB_{Pre} is closing, due mainly to increases in the area burned at high severity. In California and adjoining western states, forest types such as oak woodland and yellow pine-mixed conifer evolved under fire regimes characterized by frequent, low-to-moderate severity burning (Agee 1993, Van Wagtendonk et al. 2018b, Safford et al. 2021). The dominant tree species in these forests are resistant to fire as adults, with adaptations like thick bark, self-pruning of lower branches, thick cone scales, and highly flammable needle cast that serves to reduce competition from seedlings and saplings when it burns (Safford and Stevens 2017). Most of these species are not adapted to high severity fire, however (Keeley and Safford 2016).

As a result of the relative increases in high severity fire and the concomitant reductions in low-to-moderate severity fire, researchers have documented major ecological impacts on the study region. These changes include: loss of carbon storage; increased plume emissions and decreased air quality; increased erosion; and adverse impacts on soil nutrients, microbial processes and hydrology (Maestrini et al. 2017, Roche et al. 2018, Abney et al. 2019, Dove et al. 2020). Additionally, studies have shown that shifts in burning patterns correlate with failures in conifer regeneration (Welch et al. 2016, Shive et al. 2018), changes in the balance of fire tolerant and fire intolerant species (Stevens et al. 2015, White et al. 2016), negative impacts to overall species diversity and to many plant and animal taxa (Miller et al. 2018, Blomdahl et al. 2019, Dalrymple and Safford 2019, Richter et al. 2019, Steel et al. 2019, Jones et al. 2020, Steel et al. 2021), and vegetation type conversion (Webster and Halpern 2010, Collins et al. 2011, Stevens et al. 2015, Coppoletta et al. 2016, Tepley et al. 2017, Coop et al. 2020, Dove et al. 2020). To reverse these changes and restore the fire regime processes to which the dominant oak, yellow pine and mixed conifer forest types are historically adapted, it will be necessary to substantially increase the area and percentage of forest burned at low-to-moderate severity (Scholl and Taylor 2010, North et al. 2012, Safford and Van de Water 2014). Given the severity trends presented here (and further explored in Safford et al. 2022), wildfire alone appears unlikely to produce the kind of mixed-severity burning that historically characterized these forests. Instead, achieving these goals will likely require increased use of prescribed fire, wildfire managed for resource benefit and/or other types of intentional fuel treatments.

Compared to lower elevation forests, red fir, lodgepole pine and subalpine forests – characterized by patchy, often rocky landscapes, slow rates of growth and fuel accumulation, and colder, shorter fire seasons – have infrequent fires and higher interannual variability in area burned, making trends harder to discern (van Wagtendonk et al. 2018b, Meyer and North 2019). The natural range of variation is also more difficult to define for these forest types because they have longer fire return intervals and historical data are harder to find the further one goes back in time. That said, there were two findings in our results for these forest types that we can interpret. First, while red fir forests experienced a 74% increase in PHS between 1984-2009 and 2010-2020, lodgepole pine and subalpine forests averaged decreases in PHS between these two periods (-16% and -46%, respectively). Second, although average annual area burned in 2010 to 2020 was lower than AAB_{Pre} for all forest types, the deficit decreased markedly in these high elevation forests, including roughly 10-fold increases in average annual area burned for lodgepole pine and subalpine forests over $AAB_{1984\text{-}2009}$. These findings suggest that fire suppression has less of an impact on historical/NRV fire severity and burn patterns at the highest elevations, especially where lodgepole pine and subalpine forests are typically found. We consider the most compelling explanation to be because the lack of fire over the last century represents a smaller departure from the pre-EAS fire return intervals compared to forest types adapted to more frequent fire (Safford et al. 2012b, Safford and Van de Water 2014). Another contributing factor is likely that fire suppression is implemented less in high elevation forests due to reduced access, low density of human assets and fire management policies that are more tolerant of naturally ignited fire for ecological benefit (van Wagtendonk 2007).

When comparing current burn trends to historical ones, it is important to consider the data accuracy for both time periods. California's fire perimeter dataset is highly accurate after 1950 and the Landsat imagery that makes complete region-wide fire severity mapping possible has been available since 1984 (Miller et al. 2009b). Moreover, the availability of severity atlases and statistical models that relate severity maps to ground-based measurements is constantly expanding. The Forest Service RdNBR-based dataset for California is likely the most trustworthy in the US – it has been extensively ground-validated and calibrated, many smaller fires are included in the dataset, and fire severity classifications use objective thresholds that allow translation of fire effects into biomass loss, permitting comparisons across fires and years (see e.g., Miller and Thode 2007, Safford et al. 2008, Miller et al. 2009a, Miller et al. 2016). Further, the development of partially automated approaches (e.g., using Google Earth Engine) allow for consistent and comprehensive fire severity estimates across broad geographies (Parks et al. 2018, 2019).

In contrast, it is difficult to estimate historical fire severity and rotation periods with high precision because they are (a) variable by nature and (b) based on patchy reconstruction estimates that only get more difficult to piece together the further back one goes in time. We used recent studies (Mallek et al. 2013, Safford and Van de Water 2014) and natural range of variation studies (Safford and Stevens 2017, Meyer and North 2019) to inform our estimates because they represent thoroughly researched, best-available inferences that combine historical data, modern reference sites, current research, and model-based assessments of both the study system in question and adjoining analogous systems. We do not discount the unavoidable imprecision that comes with reconstructing historical fire return intervals and severity patterns across time spans for which data are largely absent. Nevertheless, we believe a more pressing challenge facing future studies may be to determine the likely future range of variability under emerging climatic conditions (Wiens et al. 2012).

MANAGEMENT IMPLICATIONS

Our findings have important implications for fire and forest management, policy, and conservation in and around the study region. First, although it has been widely known for more than 50 years that fire exclusion in western US forests is a major driver of ecosystem and fire regime change, many federal and state agencies persist in suppressing almost all fires (Calkin et al. 2005, Stephens et al. 2016). Wildfire suppression will continue to be necessary to protect human life, property and other important assets, but in fire-adapted landscapes it should be considered as only one of many tools in the management toolkit. Continued focus on reducing burned area – even in ecosystems where the principal ecological missing-link is fire, such as oak woodlands, yellow pine and mixed conifer forests – will not address the urgent need to minimize the ecologically harmful impacts of fire (Stephens et al. 2016, Moreira et al. 2020, Safford et al. 2022).

The disconnect between fire management and resource management was the chief driver of the switch from blanket fire suppression to multipurpose fire management that was made in US federal agencies in the late 1960s and early 1970s (Stephens and Ruth 2005), as well as in the 2009 update to US federal fire management policy that permitted all wildfires to be managed for suppression and/or resource benefit (USDA-USDOI 2009). However, the proportion of the Forest Service budget that goes to wildfire suppression-related activities rose from 16% in 1995 to 52% in 2015 (Stephens et al. 2016), and exceeds 65% today. As North et al. (2015) note, myopic focus on short-term fire management results not from policy constraints but from "entrenched agency disincentives to working with fire." These disincentives relate to nuances of budget allocation, concerns about assets at risk, smoke production, politics, liability, and public perception of all fire as bad (Calkin et al. 2015). Whatever the drivers, as fires grow larger and spread more rapidly, increasingly large portions of the annual budgets of federal resource management agencies are diverted to putting out fires, siphoning already scarce funding from proactive ecosystem management and restoration activities (including fuel reduction) and paradoxically increasing the potential for severe fires in the future as fuels continue to accumulate and the climate continues to warm (Carroll et al. 2007, Calkin et al. 2015, Stephens et al. 2016, Moreira et al. 2020).

The fire-climate modeling literature (e.g., Dettinger et al. 2018, Restaino and Safford 2018) also projects increases in annual area burned that are consistent with our findings. These trends have generated excited headlines that decry a "climate reckoning in fire-stricken California" (NY Times, September 10, 2020) and warn of wildfires in the West "spread[ing] like the plague" (Wall Street Journal, September 8, 2020). However, increasing burned area, the most often cited measure of calamity, is only an ecological concern where annual burning exceeds the NRV, routinely and over the long-term. The 2020 wildfire season was the only year in our study period that came close to being comparable in burned area to the pre-Euro-American settlement average. That said, there *are* ecosystems in California and neighboring states where annual burned area is unsustainably high by ecological standards. These are primarily sagebrush and related ecosystems in the Great Basin and chaparral and sage scrub in central and southern California, where the problem is driven by highly flammable invasive annual grasses, and in chaparral, a surfeit of human ignitions (Safford et al. 2018, Safford et al. 2021). In these places, fire suppression is both ecologically justified and crucial.

For the forest types we analyzed, however, the issue is not too much burning but too much of *the wrong kind* of burning. The tendency of modern forest fires that escape initial attack to burn large areas at high severity is driven by (1) unnaturally high fuel loadings and (2) weather conditions that reflect a steadily warming climate (Abatzoglou and Williams 2016, Keeley and Safford 2016, Parks et al. 2018b, Safford et al. 2021, 2022). For the most part, increased investment in fire suppression is a short-term fix that fails to resolve these issues and, when "successfully" implemented, extends the period of fuel accumulation. While essential for the protection of life and property in the wildland-urban interface, and thus of relevance to any comprehensive solution to wildfire (Schwartz and Syphard 2021), fire suppression of natural ignitions can have an aggravating effect when applied to forest types adapted to frequent fire (Moreira et al. 2020). By contrast, climate change mitigation will be fundamentally important in the long-term, but will not address the immediate need to reduce fuels in erstwhile frequent-fire forest types (e.g., oak woodland, yellow pine, mixed conifer) where fire regime changes and ecologically damaging fires have been most pronounced (Steel et al. 2015). Instead, this objective may be accomplished through strategic expansion of active fuels reduction, enhanced application of prescribed fire, and increased management of wildfires for ecological purposes (i.e., "resource benefits"), alone or in combination (North et al. 2012, North et al. 2015, Stephens et al. 2021).

While by no means the definitive source for setting fire-related management targets, parameters provide forest managers with a useful template for considering burn frequency and severity objectives in the context of historical forest structure and composition. By comparing a contemporary forest to its NRV, managers can assess whether restoration to such standards is (a) appropriate and (b) feasible based on how much a forest resembles or is departed from the conditions under which it presumably functioned before Euro-American settlement (Manley et al. 1995, Landres et al. 1999, Wiens et al. 2012). In the case of yellow pine and mixed conifer forests in our study region, for example, comparisons of contemporary forest stands with NRV reveal forests with tree densities that are 2-4x higher (or more) than before EAS, average tree diameters about half of their historical norms, higher and more continuous canopy cover, and 70-100% increases in surface fuel loadings – changes that suggest modern stands are more fuels-limited than ignition-limited (Safford and Stevens 2017).

Because anthropogenic warming is leading us away from the climatic conditions that characterized the pre-EAS/NRV period, it has been suggested that NRV-based targets should be applied cautiously (Millar et al. 2007). However, Safford et al. (2012a, 2012b) point out that under shifting environmental baselines, NRV conditions retain their value, especially where they are interpreted as management reference points rather than endpoints, and where they are used to better understand the mechanisms of change. Research suggests that future forests in the study region will support lower tree densities and biomass than under current or pre-EAS conditions (Lenihan et al. 2003, Safford and Stevens 2017, Stanke et al. 2021, North et al. 2022). If so, managers could use NRV estimates as a reference point from which to set new targets for forest resilience based on how much current and NRV conditions differ. Substantiation for the value of the NRV in the study area is also found in recent research into the fire responses of key wildlife indicator species (California spotted owl [*Strix occidentalis occidentalis*], Pacific fisher [*Pekania pennanti*], and black-backed woodpecker [*Picoides arcticus*]), whose nesting and foraging behaviors show strong links to pre-EAS ranges of variation in fire severity and high severity patch size (Safford and Stevens 2017, Blomdahl et al. 2019, Stillman et al. 2019, Jones et al. 2020, Kramer et al. 2021). Thus, we see a natural synergy between (a) studies such as this one that provide a multi-decadal perspective on how fire patterns are changing across a cohesive landscape and (b) NRV-type assessments that provide managers and researchers with an ecologically meaningful context in which to consider the implications of those changes and what actions they might implement in response.

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TABLES AND FIGURES

A)

Forest type (code)	Dominant species	Average elevation (m)	Extent (ha)	Burned area mapped for severity (1984-2020)
Oak woodland (OW)	QUDO, QUWI, PISA	756	959,252	275,744
Dry mixed conifer (DMC)	PIPO, PILA, CADE, ABCO, QUKE	1121	737,931	267,624
Moist mixed conifer (MMC)	ABCO, PSME, PILA, CADE, SEGI	1590	1,372,110	442,759
Yellow pine (YP)	PIJE, PIPO, QUKE	1,714	1,550,530	442,701
Red fir (RF)	ABMA, PIMO	2335	1,026,116	169,204
Lodgepole pine (LP)	PICO	2786	111,178	9,640
Subalpine (SA)	PIAL, PIMO, PIFL, PICO, TSME	3163	264,175	6,392

B)

Acronym	Explanation
AAB	Annual area burned (all severity classes)
AAHS	Annual area burned at high severity (class IV)
AALMS	Annual area burned at low-to-moderate severity (classes I-III)
PHS	Percentage of burned area burned at high severity
PLMS	Percentage of burned area burned at low-to-moderate severity
EAS	Euro-American settlement (~1850)
Pre [as subscript]	Refers to Pre-EAS, i.e., before ~1850

Table 1. A) Forest types considered in this study. Dominant tree species that characterize each type are listed using the following abbreviations: ABCO: *Abies concolor*; *ABMA: A. magnifica; CADE: Calocedrus decurrens; PIAL: Pinus albicaulis; PICO: P. contorta* ssp. *murrayana; PIFL: P. flexilis; PIJE: P. jeffreyi; PILA: P. lambertiana; PIMO: P. monticola; PIPO: P. ponderosa; PISA: P. sabiniana; PSME: Pseudotsuga menziesii; QUDO: Quercus douglasii; QUKE: Q. kelloggii; QUWI: Q. wislizenii; SEGI: Sequoiadendron giganteum; TSME: Tsuga*

mertensiana. B) An explanation of the acronyms used for the variables and time periods analyzed.

Forest Type	Fire rotation Average (yrs)	Range	PHS (%)	Source Literature
OW	18	12-25	6	Mallek et al. 2013
DMC	23	11-34	6	Mallek et al. 2013, Safford & Stevens 2017
MMC	31	15-70	8	Mallek et al. 2013, Safford & Stevens 2017
YP	22	11-34	5	Mallek et al. 2013, Safford & Stevens 2017
RF	79	25-163	10	Miller et al. 2012, Mallek et al. 2013, Meyer and North 2019
LP	63	46-80	24	Mallek et al. 2013, Meyer and North 2019
SA	425	75-721	10	Mallek et al. 2013, Meyer and North 2019, van Wagtendonk et al. 2018a

Table 2. Estimated average Pre-Euro-American Settlement fire rotation period in years and percent burned at high severity (PHS) for the forest types considered in this study. Estimates are based on the average values for the range of numbers found in the corresponding published scientific literature sources. Key to forest type codes found in Table 1.

Type	AAB (ha)			PHS			PLMS			AAHS (ha)			AALMS (ha)		
	Pre	1984-2009	2010-2020	Pre	1984-2009	2010-2020	Pre	1984-2009	2010-2020	Pre	1984-2009	2010-2020	Pre	1984-2009	2010-2020
All	211,822	27,154	82,551	7%	29%	36%	93%	71%	64%	14,002	7,955	30,001	197,819	19,199	52,550
OW	51,168	6,387	9,972	6%	22%	32%	94%	78%	68%	3,275	1,421	3,189	47,893	4,966	6,783
DMC	31,461	3,947	15,001	6%	25%	43%	94%	75%	57%	1,903	986	6,411	29,558	2,960	8,590
MMC	44,076	5,328	27,657	8%	30%	37%	92%	70%	63%	3,658	1,600	10,172	40,418	3,728	17,485
YP	69,411	8,360	20,485	5%	42%	39%	95%	58%	61%	3,349	3,511	8,066	66,062	4,850	12,419
RF	13,132	3,014	8,258	10%	14%	24%	90%	86%	76%	1,313	411	1,951	11,819	2,603	6,307
LP	1,758	71	710	24%	30%	26%	76%	70%	74%	422	21	182	1,336	49	527
SA	816	47	469	10%	12%	6%	90%	88%	94%	82	6	30	734	42	439

Table 3. Comparison of average annual burned area and percentage burned at different severity classes for the study area by forest type and time period. Total annual area burned (AAB) in hectares is the sum of annual area burned at high severity (AAHS) and annual area burned at low-to-moderate severity (AALMS) severity. AAHS/AAB is the percentage burned at high severity (PHS) and AALMS/AAB is the percentage burned at low-to-moderate severity (PLMS). Average annual percentage values listed are not weighted by annual burned area. “Pre” refers to the pre-Euro-American Settlement before 1850. Forest type codes as in Table 1.

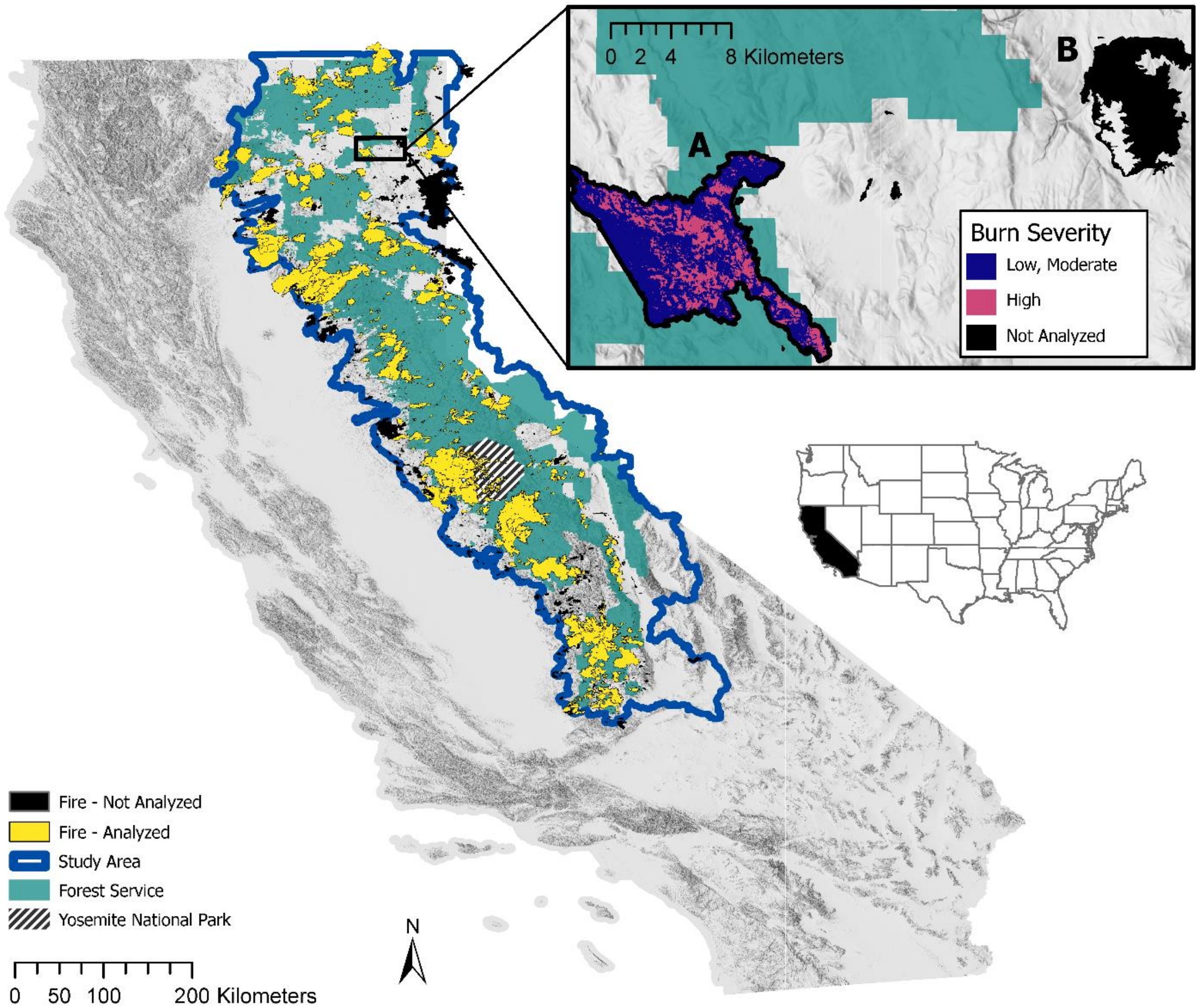
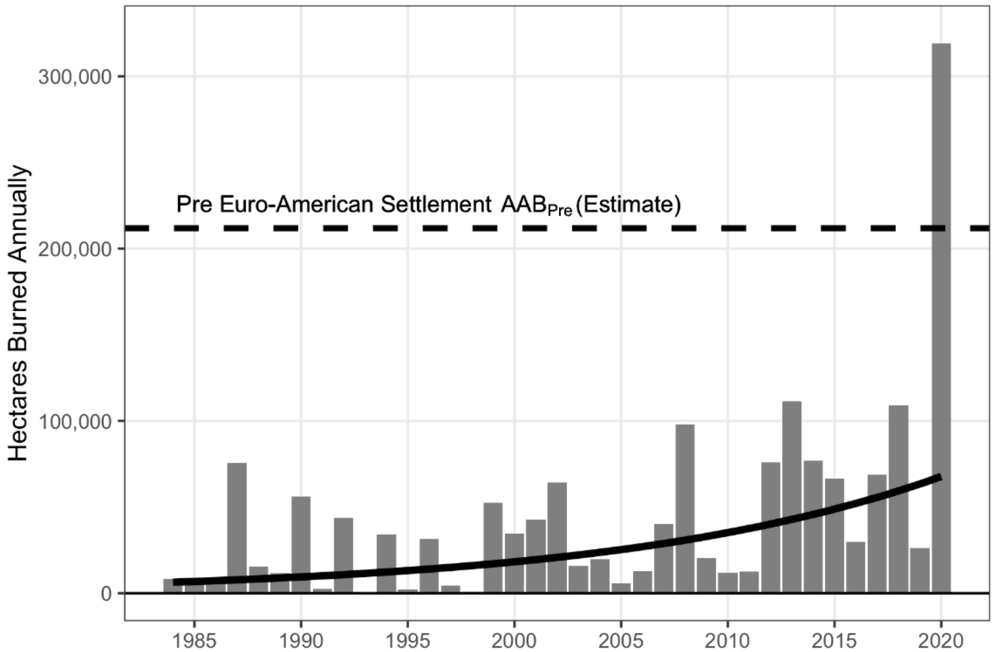


Figure 1. The Sierra Nevada-Southern Cascades study region, based on the Sierra Nevada Forest Plan Amendment (USDA 2004) (see Miller et al. 2009b for original map). Yellow polygons indicate wildfires that occurred from 1984 – 2020 and were analyzed for severity following the Mallek et al. (2013) method – see inset for severity detail (A). Black areas (B) are burn perimeters not mapped for severity because they occurred on lands outside US National Forests or National Parks or were less than 80 ha in size.

A)



B)

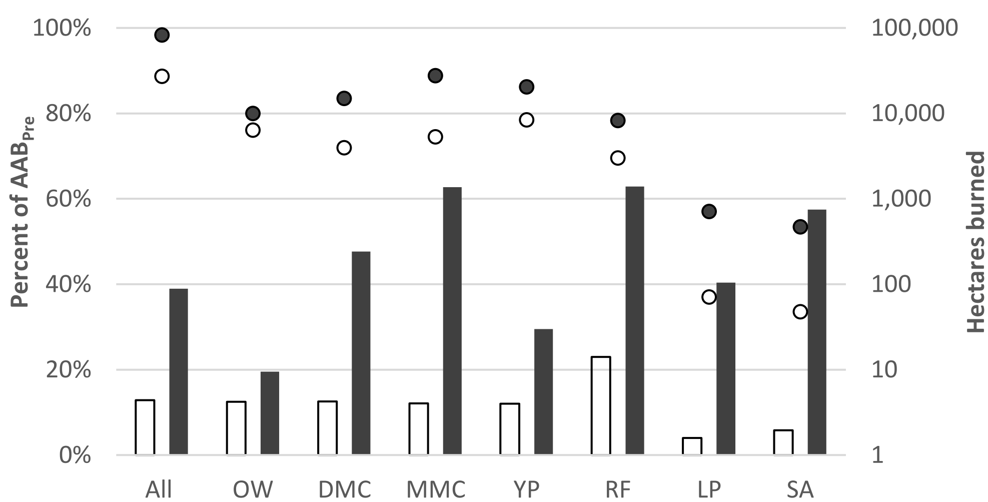


Figure 2. (A) Annual area burned by wildfire (AAB) in the Sierra Nevada-Southern Cascades study region for the expanded modern period, 1984-2020, for the seven major forest types considered here (see Table 1 for forest type codes). The dashed line above shows the estimated average AAB across these seven forest types for the pre-Euro-American period (AAB_{Pre}) based

on previous literature. The solid upsloping trend line shows the fitted linear model from this study with log area as the response variable and time as the predictor variable with an autoregressive term. B) AAB by major forest type for the periods 1984-2009 (white bars) and 2010-2020 (black bars) – as a percentage of AAB_{Pre} (left axis). White and black circles show AAB in hectares for the same two periods, respectively (right axis, note log scale).

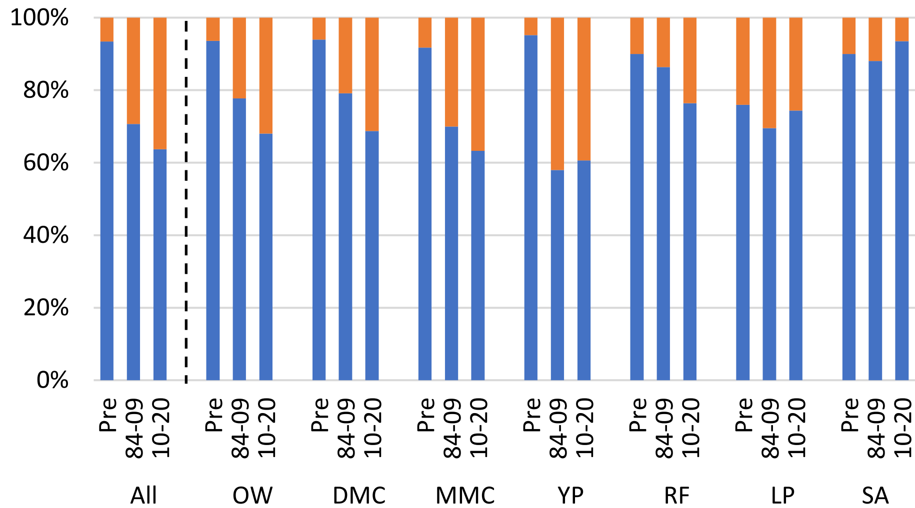
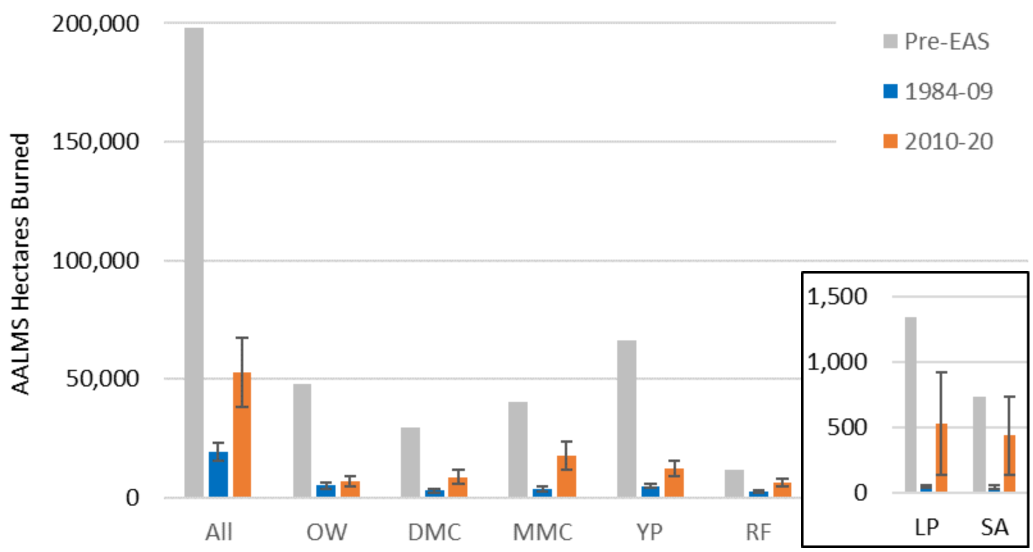


Figure 3. Burn severity trends as a percentage of total area burned averaged across years for three time periods: prior to ~1850 (Pre); 1984 – 2009 (84-09); and 2010 – 2020 (10-20). Blue bars are percentage burned at low-to-moderate severity (PLMS); orange bars are percentage burned at high severity (PHS). Cumulative data for all forest types combined are indicated by "All" and separated with a vertical dashed line. Slight differences between 1984-2009 values and values in Mallek et al. (2013) are due (1) to addition of pre-2010 fires to the burn severity dataset after 2013, and (2) to changes in pre-EAS fire severity due to new information (see Table 3). See Methods for details. Forest type codes as in Table 1.

A)


AALMS Hectares Burned
200,000
150,000
100,000
50,000
0
All
OW
DMC
MMC
YP
RF
Pre-EAS
1984-09
2010-20
1,500
1,000
500
0
LP
SA

B)

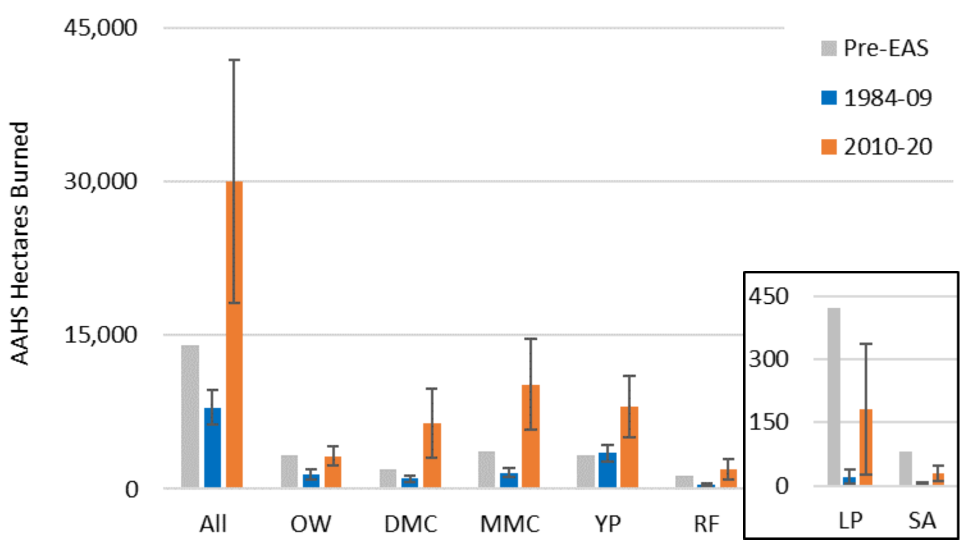

AAHS Hectares Burned
45,000
30,000
15,000
0
All
OW
DMC
MMC
YP
RF
Pre-EAS
1984-09
2010-20
450
300
150
0
LP
SA

Figure 4. Comparison of average annual area burned (hectares) in the Sierra Nevada-Southern Cascades study region by forest type for A) low-to-moderate severity fire (AALMS) and B) high severity fire (AAHS). Gray bars are estimates for pre-Euro-American settlement (“Pre”); blue bars are for the period 1984-2009; and orange bars are for the period 2010-2020. Forest type codes as in Table 1. Error bars based on standard error of the mean.

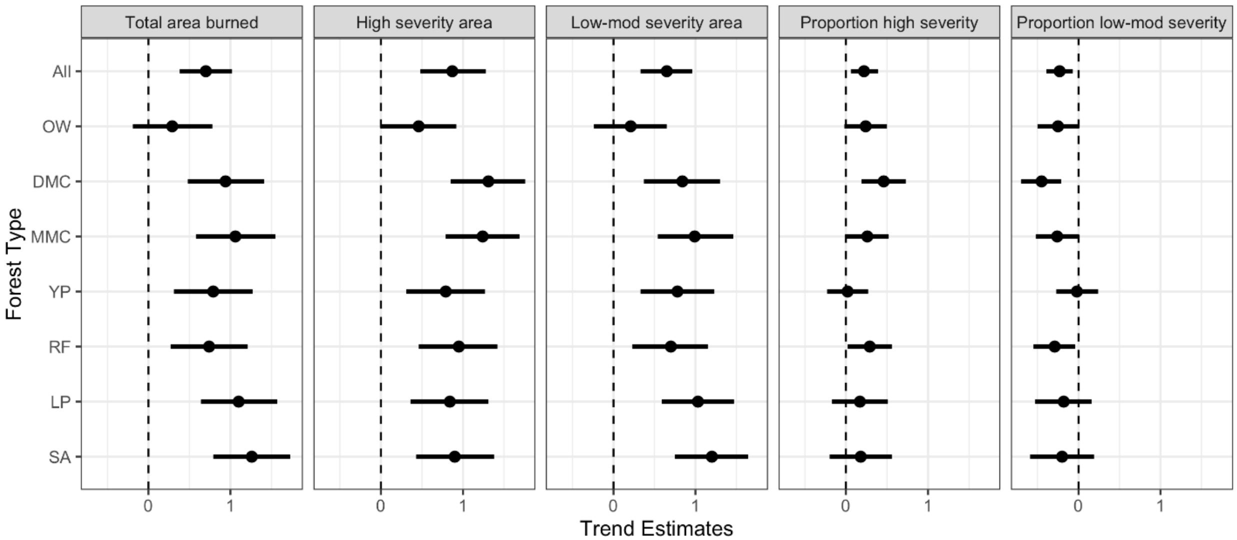


Figure 5. Standardized trend estimates by forest type for wildfire burned area and severity in the Sierra Nevada-Southern Cascades from 1984 to 2020. Trends were derived using generalized linear models and are for total annual area burned (AAB) and its high severity (AAHS) and low-to-moderate severity (AALMS) components, together with estimates in trends for percentage of area burned at high (PHS) and low-to-moderate severity (PLMS). Estimates to the right and left of the dashed lines indicate increasing and decreasing trends with time, respectively. Forest type codes as in Table 1.

SUPPORTING INFORMATION

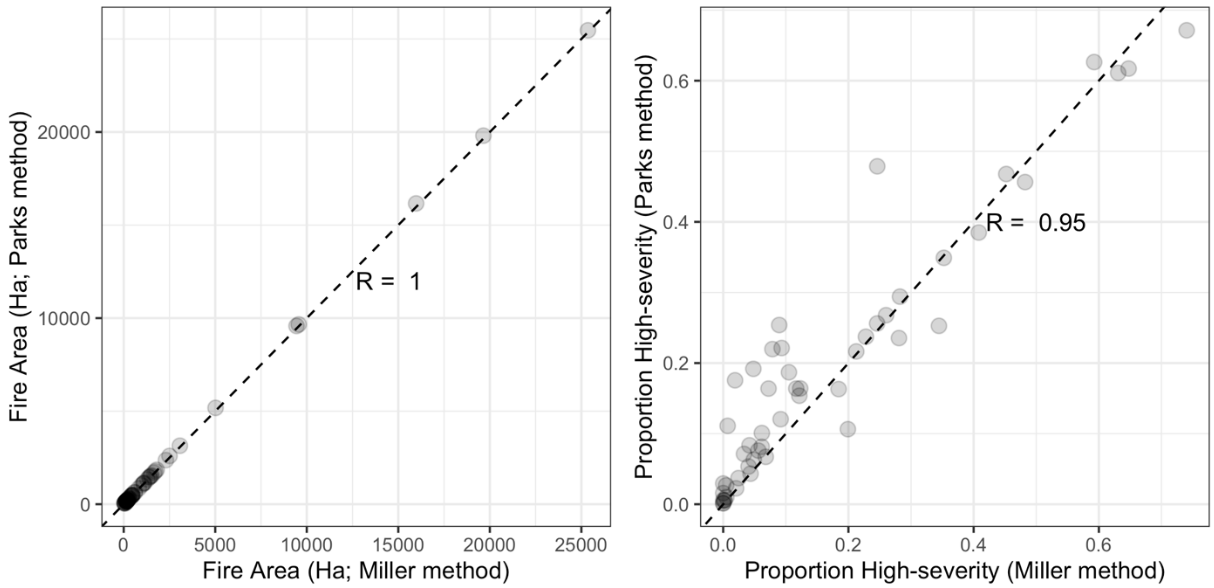
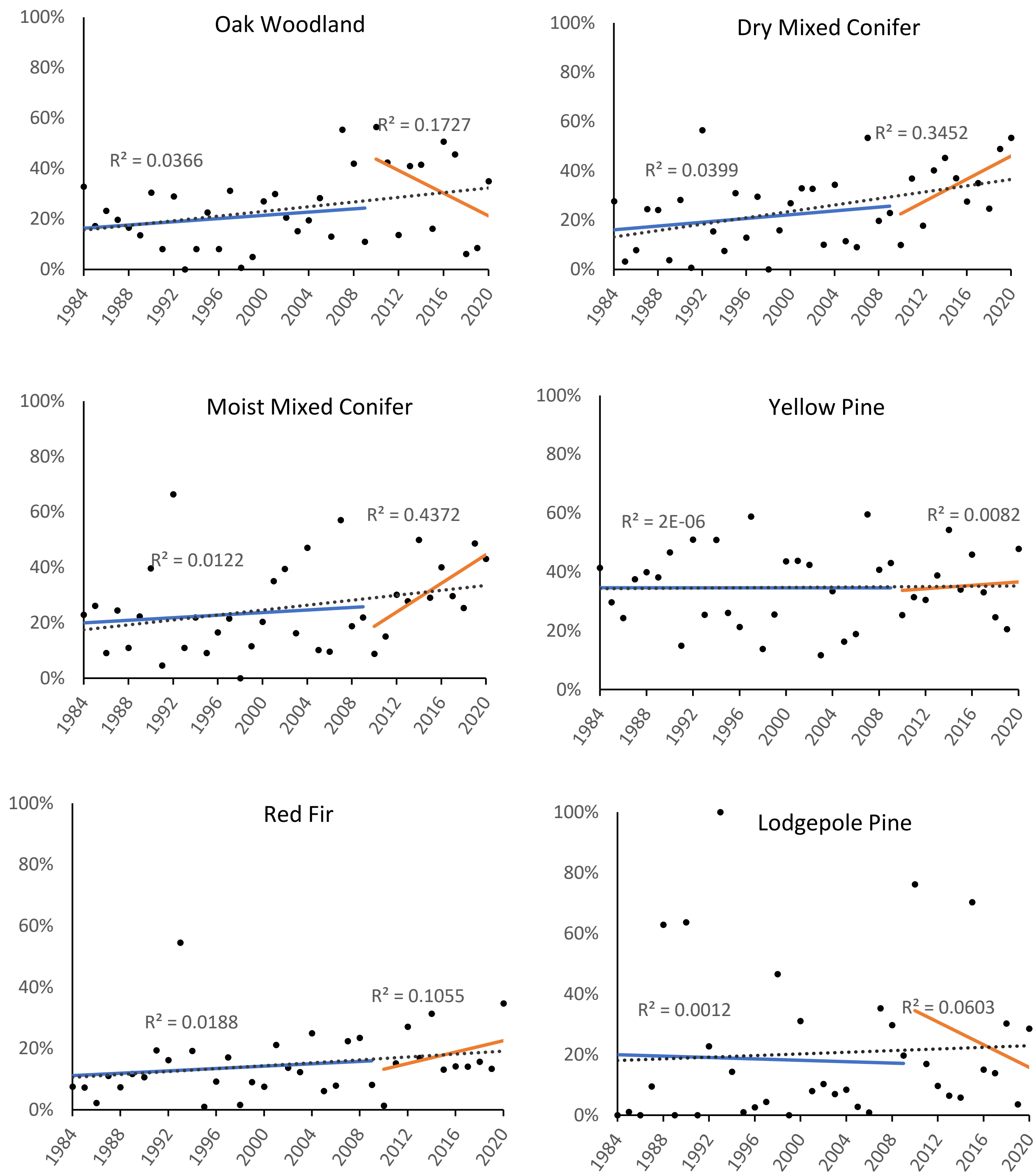


Figure S1. A comparison of burned area and percent high severity for 50 randomly selected 1985-2017 wildfires calculated using the legacy ("Miller method") and Google Earth Engine-derived ("Parks Method") datasets.


Oak Woodland
$R^2 = 0.0366$
$R^2 = 0.1727$
Dry Mixed Conifer
$R^2 = 0.0399$
$R^2 = 0.3452$
Moist Mixed Conifer
$R^2 = 0.0122$
$R^2 = 0.4372$
Yellow Pine
$R^2 = 2E-06$
$R^2 = 0.0082$
Red Fir
$R^2 = 0.0188$
$R^2 = 0.1055$
Lodgepole Pine
$R^2 = 0.0012$
$R^2 = 0.0603$
0%
20%
40%
60%
80%
100%
1984
1988
1992
1996
2000
2004
2008
2012
2016
2020

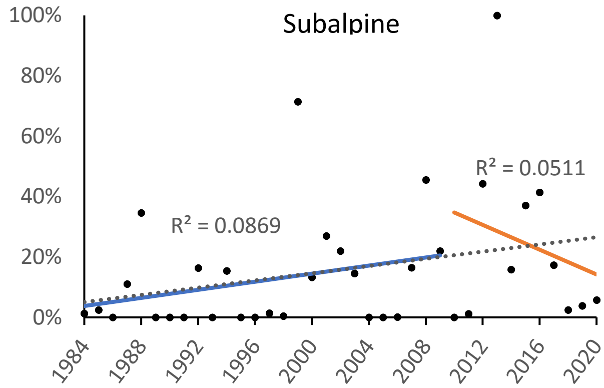


Figure S2. Temporal trends of high severity (PHS) burned area as a percentage of total area burned by wildfire for different forest types of the Sierra Nevada-Southern Cascades region. Linear regression trendlines for the period 1984 to 2009 are shown in blue, while those for the period 2010 to 2020 are shown in orange. The combined period trendline is shown by dotted black lines.

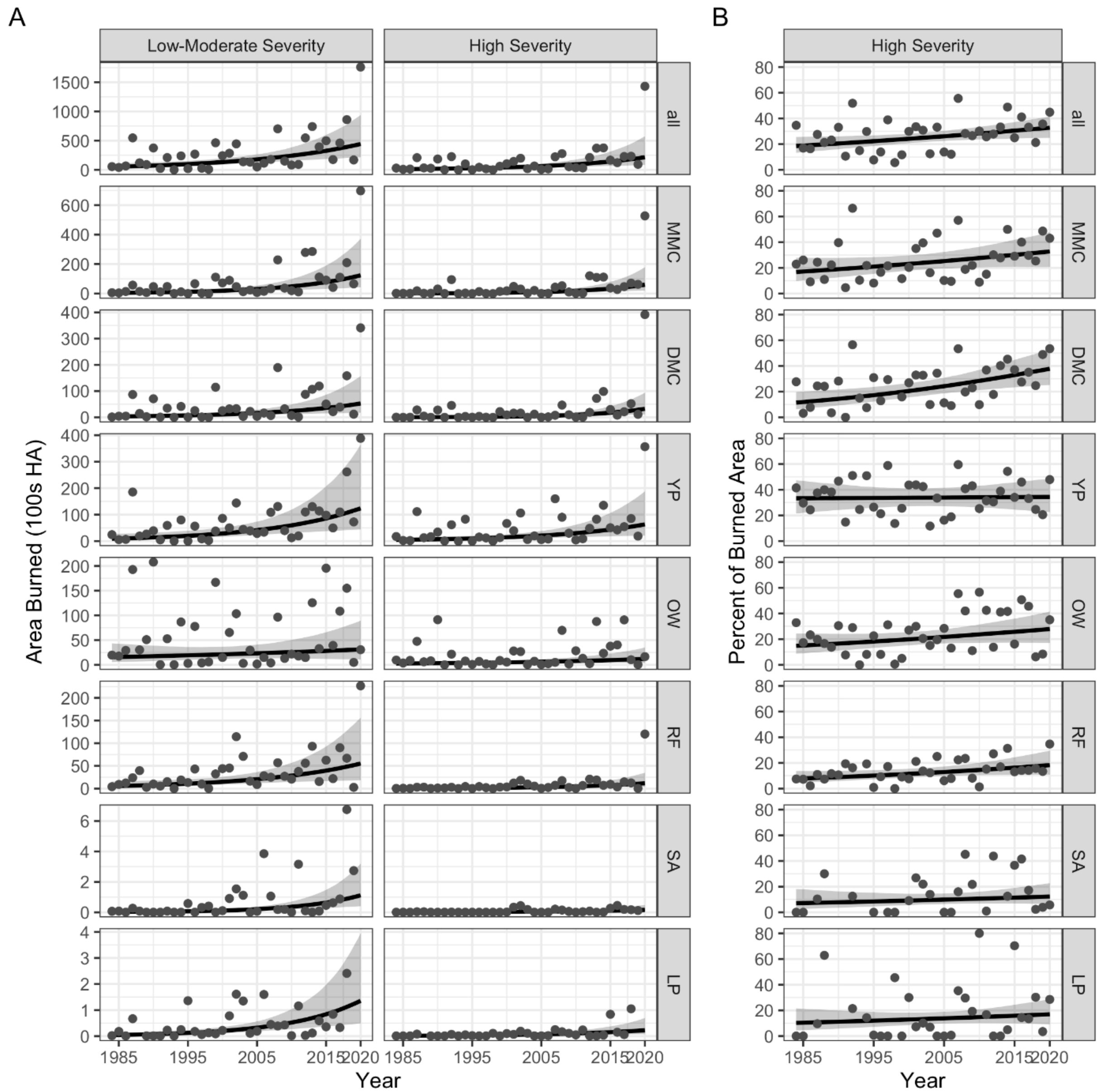


Figure S3. Generalized linear model (GLM) trend lines and credibility intervals (gray area) by forest type for percent of area burned annually by high severity wildfire (PHS) in the Sierra Nevada-Southern Cascades study region for the extended modern period (1984-2017) considered in this study. Forest type codes as in Table 1.