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High-severity burned area and proportion exceed historic conditions in Sierra Nevada, California, and adjacent ranges

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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 (AAB) 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 to 2009. Our results show that the total average AAB remained below pre-EAS levels, but that gap is decreasing (i.e., ~14% of pre-EAS for 1984–2009, but 39% for 2010–2020 [including ~150% in 2020]). Although the average AAB has remained low compared with pre-EAS, both the average annual area burned at high severity (AAHS) 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 AAHS exceeded the pre-EAS average for the first time on record. By contrast, the 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.

KEYWORDS
annual area burned, Cascade Range, fire ecology, fire regime, forest, 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 & 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; Miller & Safford, 2020; van Wagtendonk, Sugihara, et al., 2018). Over the past century, however, fire regimes in many western North American forests have departed from their natural range of variation (NRV). These modern changes have been driven largely by anthropogenic factors, for example, 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 & Williams, 2016; Balch et al., 2017; Klimaszewski-Patterson & Mensing, 2016; Parks, Holsinger, Miller, & Parisien, 2018; Stephens et al., 2016).

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; Parks, Holsinger, Panunto, et al., 2018; Safford & Van de Water, 2014; Steel et al., 2015). 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 (Cassell et al., 2019; Hanberry, 2014; Johnstone et al., 2010; Stevens et al., 2019), and such shifts may be exacerbated by climate warming (Halofsky et al., 2020; van Mantgem et al., 2013). In yellow pine (i.e., ponderosa pine and/or Jeffrey pine; *Pinus ponderosa*, *Pinus jeffreyi*) and mixed conifer forests (the above species, plus, among other species, sugar pine [*Pinus 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 (Boisramé et al., 2017; Keeley & Syphard, 2016; Miller, Safford, et al., 2009; Steel et al., 2018; Thorne et al., 2017; Welch et al., 2016), threatened and endangered animal populations (Blomdahl et al., 2019; Jones et al., 2020), plant species diversity (Miller & Safford, 2020; Richter et al., 2019), and ecosystem services (Rakhmatulina et al., 2020; Richter et al., 2019; Wu & Kim, 2013).

To better understand how wildfire patterns in the western United States have been changing, Mallek et al. (2013) compared modern versus historical patterns of annual area burned (AAB) 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, that is, prior to ca. 1850. We also use the 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 the overall AAB 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. ~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 did burn. 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 AAB by wildfires in California has increased considerably since 2009, the last data year considered by Mallek et al. (2013), with 9 of the 10 largest fires in the State's history having 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 (Abney et al., 2019; Coppoletta et al., 2016; Dove et al., 2020; Jones et al., 2016, 2020; Steel et al., 2022; Welch et al., 2016). The Sierra Nevada–Southern Cascades ecoregion (i.e., the study region; Figure 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 11 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 to 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

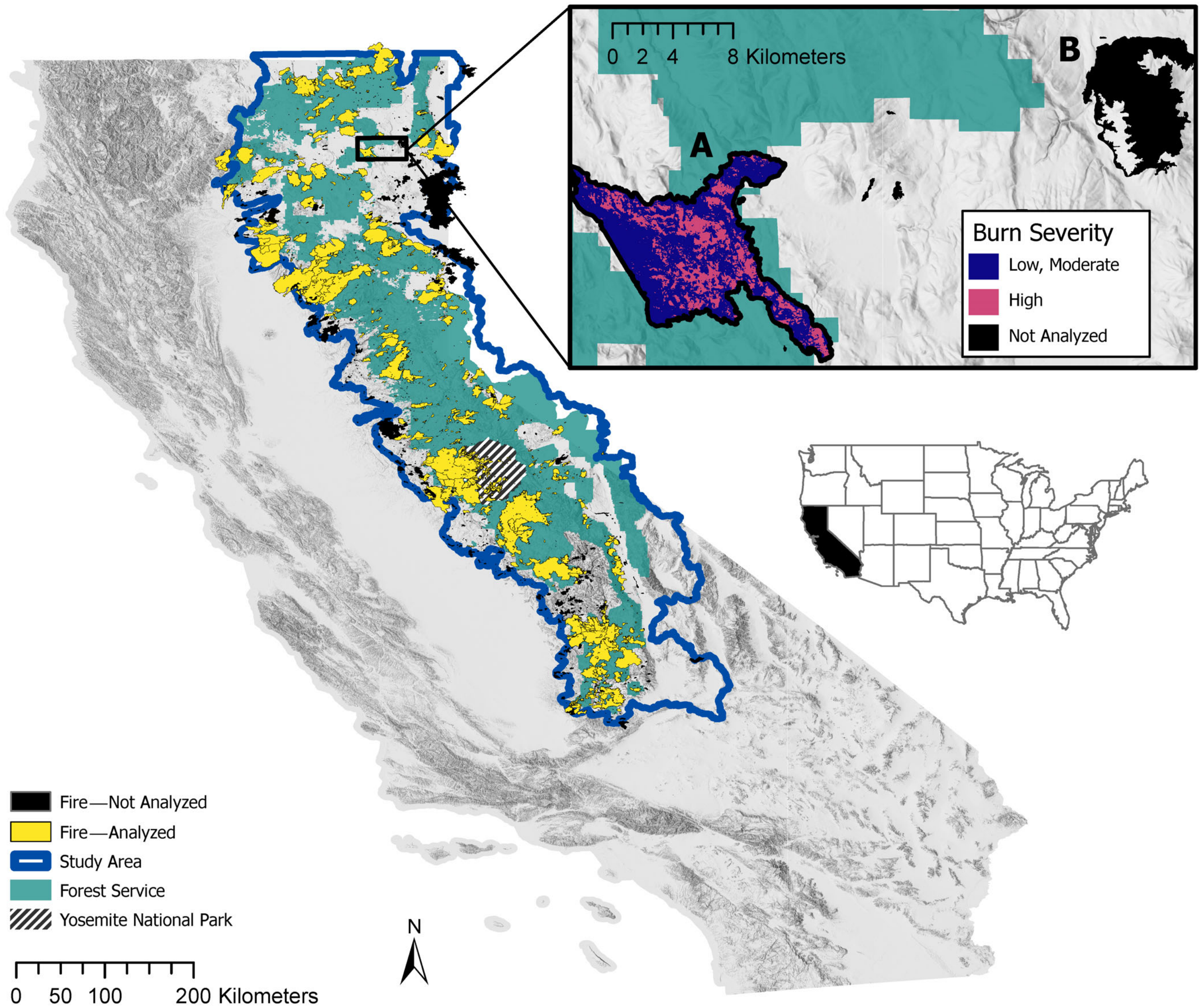

FIGURE 1 The Sierra Nevada–Southern Cascades study region, based on the Sierra Nevada Forest Plan Amendment (USDA, 2004) (see Miller, Safford, et al., 2009 for original map). Yellow polygons indicate wildfires that occurred from 1984 to 2020 and were analyzed for severity following the Mallek et al. (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.

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., Safford & Stevens, 2017; Steel et al., 2015, 2018) and personal observations, we hypothesized that for the study region between 2010 and 2020: (1) AAB would increase relative to the 1984–2009 period, but would still lag behind the average AAB 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 11 National Forests and 4 National Parks (Figure 1). This is the same study area used by Mallek et al. (2013) and Miller, Knapp, 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 and mixed conifer forests at higher elevations (Table 1). Red fir (*A. 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 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 November 1, 2019), which represents modeled potential natural vegetation incorporating climate, soils, topography, and hypothetical pre-EAS fire regimes (Rollins, 2009). BpS types were grouped into presettlement fire regime types defined by

TABLE 1 Forest types considered in this study.

Forest type (code)	Dominant species	Average elevation (m)	Extent (ha)	Burned area (ha) mapped for severity (1984–2020)
Oak woodland (OW)	QUDO, QUWI, and PISA	756	959,252	275,744
Dry mixed conifer (DMC)	PIPO, PILA, CADE, ABCO, and QUKE	1121	737,931	267,624
Moist mixed conifer (MMC)	ABCO, PSME, PILA, CADE, and SEGI	1590	1,372,110	442,759
Yellow pine (YP)	PIJE, PIPO, and QUKE	1714	1,550,530	442,701
Red fir (RF)	ABMA and PIMO	2335	1,026,116	169,204
Lodgepole pine (LP)	PICO	2786	111,178	9640
Subalpine (SA)	PIAL, PIMO, PIFL, PICO, and TSME	3163	264,175	6392

Note: Dominant tree species that characterize each type are listed using the following abbreviations: ABCO, *Abies concolor*; ABMA, *Abies magnifica*; CADE, *Calocedrus decurrens*; PIAL, *Pinus albicaulis*; PICO, *Pinus contorta* ssp. *murrayana*; PIFL, *Pinus flexilis*; PIJE, *Pinus jeffreyi*; PILA, *Pinus lambertiana*; PIMO, *Pinus monticola*; PIPO, *Pinus ponderosa*; PISA, *Pinus sabiniana*; PSME, *Pseudotsuga menziesii*; QUDO, *Quercus douglasii*; QUKE, *Quercus kelloggii*; QUWI, *Quercus wislizenii*; SEGI, *Sequoiadendron giganteum*; TSME, *Tsuga mertensiana*.

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.

TABLE 2 Acronyms related to data on burn area, fire severity, and time periods considered.

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 area burned at high severity
PLMS	Percentage of area burned at low-to-moderate severity
EAS	Euro-American settlement (ca. 1850)
Pre (as subscript)	Refers to pre-EAS, i.e., before ca. 1850

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 11 additional years of wildfire extent and severity for the same region, extending the length of the period analyzed to 37 years from 1984 to 2020. Wildfire perimeters and total 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, Holsinger, Voss, et al. (2018) and Parks et al. (2021). A comparison of 50 randomly selected fires from 1985 to 2017 showed high similarity between the legacy and Earth Engine-derived severity estimates ($R = 0.95$; Appendix S1: 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, Knapp, et al., 2009; Miller & Thode, 2007). 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: Mallek et al., 2013; Miller & Safford, 2012; Miller, Safford, et al., 2009). 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 & 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) 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, Knapp, et al., 2009). For all areas analyzed for severity, total AAB for a forest type was equal to AAHS plus AALMS (Table 2).

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 3), defined as the number of years required to burn an area equal to the forest extent in question (Agee, 1993). We used the presettlement 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 3) to estimate the average AAB_{Pre}. Thus, for an area, A, associated with a pre-EAS fire regime rotation period of Y years, $AAB_{Pre} = A/Y$ (in hectares per year).

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 tab. 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 OW, DMC and MMC, or YP. However, we adjusted the values for RF, LP, and SA forest based on new information (Meyer & North, 2019; Safford & Stevens, 2017; van Wagtendonk, Fites-Kaufman, et al., 2018). These sources yielded AAB_{Pre} and $AAHS_{Pre}$, from which we calculated $AALMS_{Pre}$

TABLE 3 Estimated average pre-Euro-American Settlement fire rotation period in years and percentage burned at high severity (PHS) for the forest types considered in this study.

Forest type	Fire rotation (years)		PHS (%)	Source literature
	Average	Range		
OW	18	12–25	6	Mallek et al. (2013)
DMC	23	11–34	6	Mallek et al. (2013), Safford and Stevens (2017)
MMC	31	15–70	8	Mallek et al. (2013), Safford and Stevens (2017)
YP	22	11–34	5	Mallek et al. (2013), Safford and Stevens (2017)
RF	79	25–163	10	Miller and Safford (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, Fites-Kaufman, et al. (2018)

Note: 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 is given in Table 1.

(AAB − 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 dataset (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. (2013) 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 RF, lodgepole, and SA 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 autocorrelation.

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

The average AAB during the 2010–2020 period, though still well below historical levels (AAB_{Pre}), increased by more than 200% over the 1984–2009 period for all forest types combined (Figure 2). $AAB_{2010–2020}$ was especially impacted by the record-breaking 2020 wildfire season (Table 4; Appendix S1: Figure S2a; 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 PHS increased for all forest types combined between the 1984–2009 and 2010–2020 periods (Table 4; Figure 3). When these two periods are considered together, $PHS_{1984–2020}$ averaged 27%—almost four times

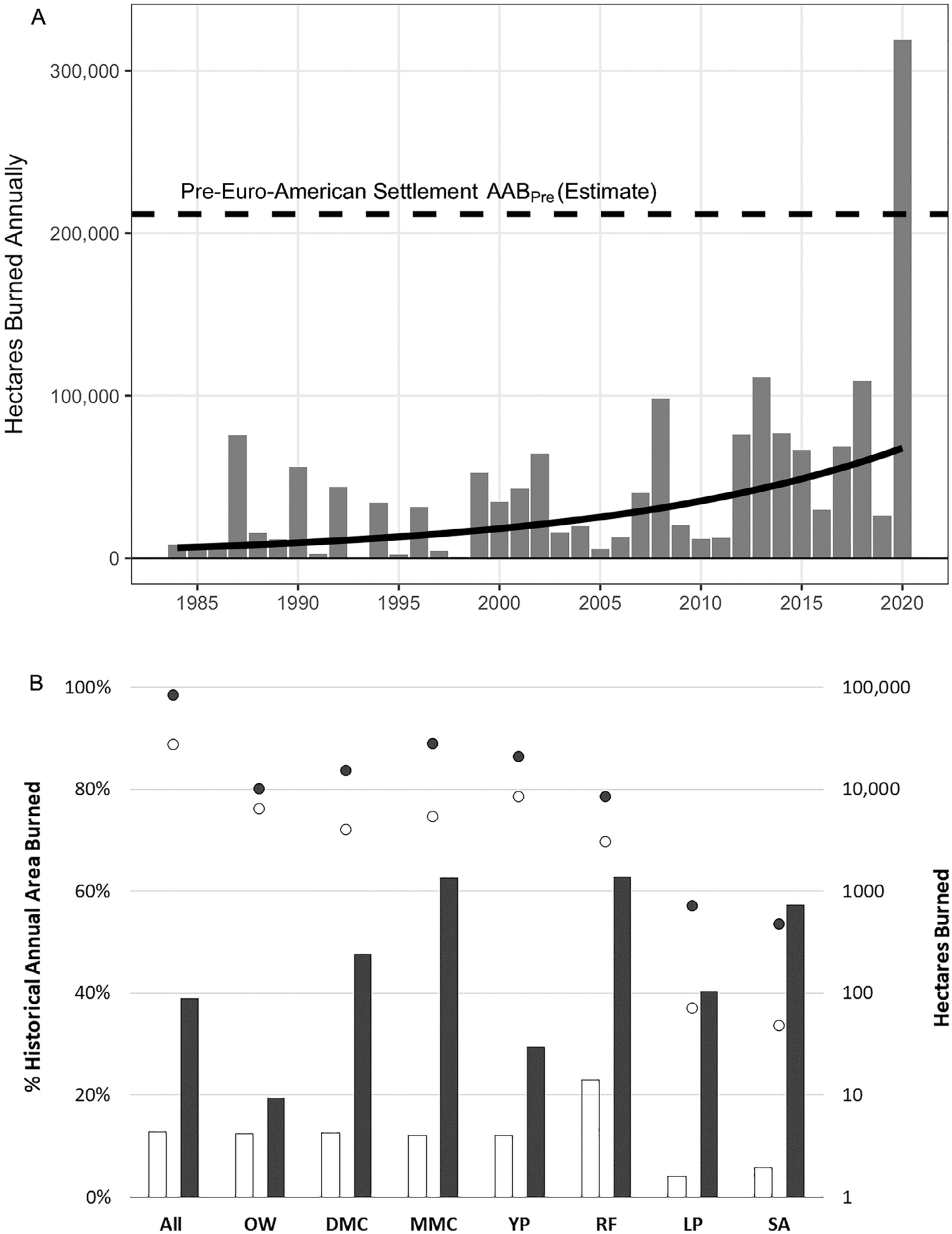

FIGURE 2 (A) Annual area burned (AAB) by wildfire in the Sierra Nevada–Southern Cascades study region for the expanded modern period, 1984–2020, for the seven major forest types considered here (see Table 1 for forest type codes). The dashed line above shows the estimated average AAB across these seven 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).

TABLE 4 Comparison of average annual burned area and percentage burned at different severity classes for the study area by forest type and time period.

	AAB (ha)			PHS (%)			PLMS (%)			AAHS (ha)			AALMS (ha)		
Type	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	7955	30,001	197,819	19,199	52,550
OW	51,168	6387	9972	6	22	32	94	78	68	3275	1421	3189	47,893	4966	6783
DMC	31,461	3947	15,001	6	25	43	94	75	57	1903	986	6411	29,558	2960	8590
MMC	44,076	5328	27,657	8	30	37	92	70	63	3658	1600	10,172	40,418	3728	17,485
YP	69,411	8360	20,485	5	42	39	95	58	61	3349	3511	8066	66,062	4850	12,419
RF	13,132	3014	8258	10	14	24	90	86	76	1313	411	1951	11,819	2603	6307
LP	1758	71	710	24	30	26	76	70	74	422	21	182	1336	49	527
SA	816	47	469	10	12	6	90	88	94	82	6	30	734	42	439

Note: Total annual area burned (AAB) 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.

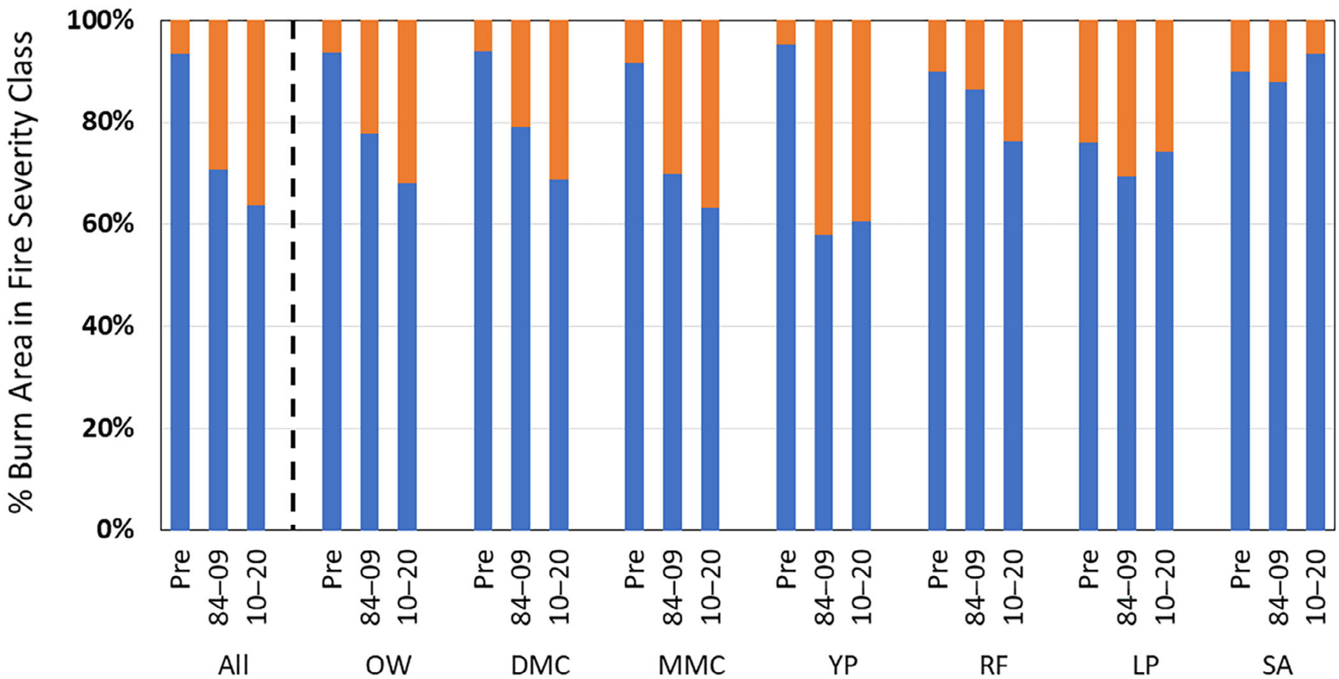

FIGURE 3 Burn severity trends as a percentage of total area burned averaged across years for three time periods: prior to ~1850 (Pre); 1984–2009 (84–09); and 2010–2020 (10–20). Blue bars are percentage burned at low-to-moderate severity; orange bars are percentage burned at high severity. Cumulative 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-Euro-American settlement fire severity due to new information (see Table 3). See *Methods* for details. Forest type codes as in Table 1.

the combined PHS_{Pre} average of 7%. For some forest types, however, PHS did not increase from 1984–2009 to 2010–2020. For YP, for example, PHS was virtually unchanged across the two modern periods (though still much higher than pre-EAS values). $PHS_{2010–2020}$ also decreased for lodgepole and subalpine forests compared with $PHS_{1984–2009}$ (Figure 3; Appendix S1: Figures S2 and S3). By contrast, $PHS_{2010–2020}$ trended noticeably upward for OW, DMC and MMC, and RF forests. The complement of PHS, PLMS, showed a decreasing trend overall from

1984–2009 to 2010–2020, with YP, LP, and SA forests as individual exceptions.

The average AALMS increased since 2009 across all forest types, but remained well below historical ($AALMS_{Pre}$) levels. Notably, for 2010–2020, the average AAHS exceeded pre-EAS levels for the first time on record (Figure 4). These trends are visible for all forest types combined, as well as for the DMC and MMC, YP, and RF forest types separately (Table 4; Figure 4B).

For the 2010–2020 period, all forest types showed appreciable increases in AAB compared with 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 ca. 150% of AAB_{Pre} in 2020 alone; Table 4). For the expanded modern period, 1984–2020, AAB averaged 20.6% of AAB_{Pre} across forest types and ranged from 14.6% (OW) to 34.8% (RF). Thus, despite recent increases, the average AAB continues to be less than half of AAB_{Pre}, due to an ongoing deficit in low-to-moderate severity fire (Figure 4A).

A comparison of modeled trends across the 1984–2020 period for burned area and burn severity revealed similarities and differences among forest types (Figure 5; Appendix S1: Figure S3). For example, $AAB_{1984–2020}$ and $AAHS_{1984–2020}$ showed positive trends over time across all forest types, though the amount of increase varied in absolute and relative terms. SA, LP, and MMC—in that order—showed the most robust increases in $AAB_{1984–2020}$, while DMC, MMC, and RF had the strongest positive trends in $AAHS_{1984–2020}$. For all forest types combined, $PHS_{1984–2020}$ trended positive for the expanded evaluation period. The results for this trend and $AAHS_{1984–2020}$ were still positive and significant when the 2020 fire year was excluded. For $PLMS_{1984–2020}$, in terms of individual forest types, only YP 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 (Mallek et al., 2013; Miller & Safford, 2012; Miller, Safford, et al., 2009; 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) the average annual area burned ($AAB_{1984–2020}$) remains well below pre-EAS averages, although the disparity is decreasing; (2) for the newly evaluated 2010–2020 period, the average $AAHS_{2010–2020}$ exceeded $AAHS_{Pre}$ for the first time on historical record, particularly in low- and middle-elevation forest types; and (3) the PHS during the expanded modern period ($PHS_{1984–2020}$) is well above pre-EAS levels and trending upward for six of seven forest types analyzed (Appendix S1: Figure S2). Conversely, $PLMS_{1984–2020}$ 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, Sugihara, et al., 2018).

Our data show that the gap between $AAB_{1984–2020}$ 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; Safford et al., 2021; van Wagtendonk, Sugihara, et al., 2018). 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 & Stevens, 2017). Most of these species are not adapted to high-severity fire, however (Keeley & Safford, 2016).

As a result of the increases in high-severity fire and the concomitant reductions in the percentage of area burned at low-to-moderate severity, 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 (Abney et al., 2019; Dove et al., 2020; Maestrini et al., 2017; Roche et al., 2018). Additionally, studies have shown that shifts in burning patterns correlate with failures in conifer regeneration (Shive et al., 2018; Welch et al., 2016), 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 (Blomdahl et al., 2019; Dalrymple & Safford, 2019; Jones et al., 2020; Miller et al., 2018; Richter et al., 2019; Steel et al., 2019, 2021), and vegetation type conversion (Collins et al., 2011; Coop et al., 2020; Coppoletta et al., 2016; Dove et al., 2020; Stevens et al., 2015; Tepley et al., 2017; Webster & Halpern, 2010). 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 (North et al., 2012; Safford &

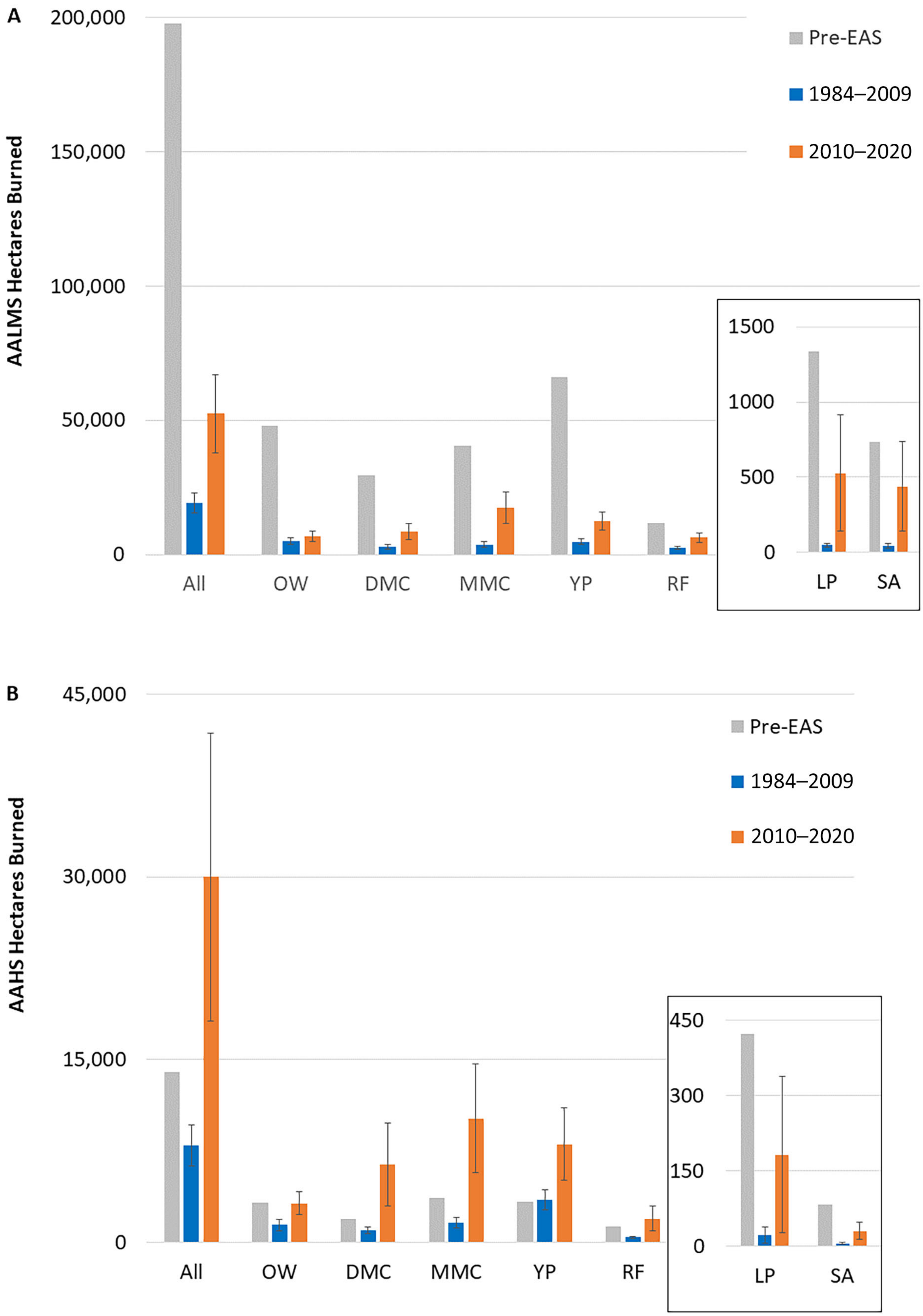

FIGURE 4 Comparison of the average annual area burned in the Sierra Nevada–Southern Cascades study region by forest type for (A) low-to-moderate severity fire (AALMS) and (B) high-severity fire (AAHS). Gray bars are estimates for pre-Euro-American settlement (pre-EAS); blue bars are for the period 1984–2009;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

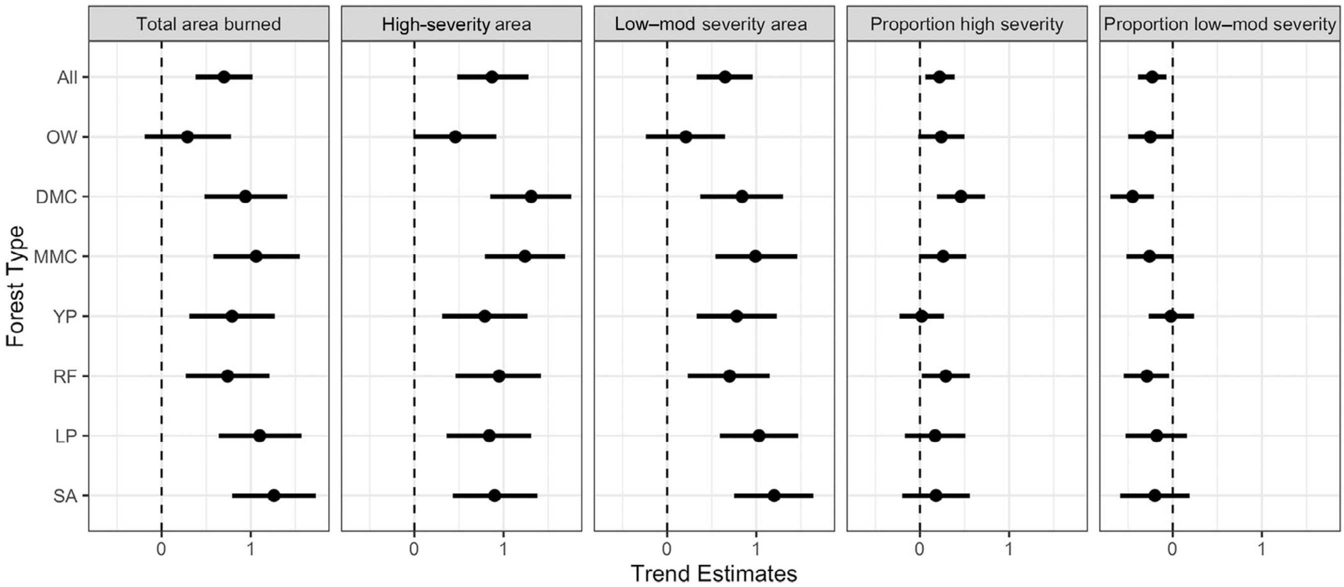

FIGURE 5 Standardized trend estimates by forest type for wildfire burned area and severity in the Sierra Nevada–Southern Cascades from 1984 to 2020. Trends were derived using generalized linear models and are for total annual area burned and its high severity and low-to-moderate severity components, together with estimates in trends for percentage of area burned at high and low-to-moderate severity. 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.

Van de Water, 2014; Scholl & Taylor, 2010). 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 with lower elevation forest types, RF, LP, and SA 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 (Meyer & North, 2019; van Wagtendonk, Sugihara, et al., 2018). The NRV 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 RF forests experienced a 74% increase in PHS between 1984–2009 and 2010–2020, LP and SA forests averaged decreases in PHS between these two periods (−16% and −46%, respectively). Second, although the average AAB in 2010–2020 was lower than AAB_{Pre} for all forest types, the deficit decreased markedly in these high-elevation forests, including roughly 10-fold increases in the average AAB for LP and SA forests over $AAB_{1984–2009}$. 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 LP and SA 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 with forest types adapted to more frequent fire (Safford, North, & Meyer, 2012; Safford & 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, Safford, et al., 2009). 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 United States: 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 &

Thode, 2007; Miller, Knapp, et al., 2009; Miller et al., 2016; Safford et al., 2008). Further, the development of partially automated approaches (e.g., using Google Earth Engine) allows for consistent and comprehensive fire severity estimates across broad geographies (Parks, Holsinger, Miller, & Parisien, 2018).

In contrast, it is difficult to estimate historical fire severity and rotation periods with high precision because they are (1) variable by nature and (2) 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 & Van de Water, 2014) and NRV studies (Meyer & North, 2019; Safford & Stevens, 2017) 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 OW, YP and mixed conifer forests, will not address the urgent need to minimize the ecologically harmful impacts of fire (Moreira et al., 2020; Safford et al., 2022; Stephens et al., 2016).

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 & 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 (Calkin et al., 2015; Carroll et al., 2007; Moreira et al., 2020; Stephens et al., 2016).

The fire–climate modeling literature (e.g., Dettinger et al., 2018; Restaino & Safford, 2018) also projects increases in AAB 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-EAS 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, 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 & Williams, 2016; Keeley &

Safford, 2016; Parks, Holsinger, Panunto, et al., 2018; 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 & 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., OW, YP, 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, 2015; Stephens et al., 2021).

While by no means the definitive source for setting fire-related management targets, NRV 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 (1) appropriate and (2) feasible based on how much a forest resembles or is departed from the conditions under which it presumably functioned before EAS (Landres et al., 1999; Manley et al., 1995; Wiens et al., 2012). In the case of YP and mixed conifer forests in our study region, for example, comparisons of contemporary forest stands with NRV reveal forests with tree densities that are 2–4× 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 ignition-limited than fuel-limited (Safford & 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, Hayward, et al. (2012) and Safford, North, and Meyer (2012) 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; North et al., 2022; Safford & Stevens, 2017; Stanke et al., 2021). 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 (Blomdahl et al., 2019; Jones et al., 2020; Kramer et al., 2021; Safford & Stevens, 2017; Stillman et al., 2019). Thus, we see a natural synergy between (1) studies such as this one that provide a multidecadal perspective on how fire patterns are changing across a cohesive landscape and (2) 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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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data (Williams, 2022) are available from Dryad: https://doi.org/10.25338/B8TP97.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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