Is fire severity increasing in the Sierra Nevada, California, USA?

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Abstract. Research in the Sierra Nevada range of California, USA, has provided conflicting results about current trends of high-severity fire. Previous studies have used only a portion of available fire severity data, or considered only a portion of the Sierra Nevada. Our goal was to investigate whether a trend in fire severity is occurring in Sierra Nevada conifer forests currently, using satellite imagery. We analysed all available fire severity data, 1984–2010, over the whole ecoregion and found no trend in proportion, area or patch size of high-severity fire. The rate of high-severity fire has been lower since 1984 than the estimated historical rate. Responses of fire behaviour to climate change and fire suppression may be more complex than assumed. A better understanding of spatiotemporal patterns in fire regimes is needed to predict future fire regimes and their biological effects. Mechanisms underlying the lack of an expected climate- and time since fire-related trend in high-severity fire need to be identified to help calibrate projections of future fire. The effects of climate change on high-severity fire extent may remain small compared with fire suppression. Management could shift from a focus on reducing extent or severity of fire in wildlands to protecting human communities from fire.

Additional keywords: conifer forests, early-successional habitat, fire-dependent biota, high-severity fire.

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Introduction

There is widespread concern about an increase in fire severity in the forests of the western United States (Agee and Skinner 2005; Stephens and Ruth 2005; Littell *et al.* 2009). It has been hypothesised that fire severity is increasing due to decades of fire suppression and fuel accumulation (Stephens and Ruth 2005) and climate change (Westerling *et al.* 2006). Excess fire is a concern for the maintenance of sufficient levels of old-growth forest for different elements of biodiversity that may be fairly fire sensitive and that have lost habitat due to logging. For example, rare wildlife species such as the Pacific fisher (*Martes pennanti*) in the Sierra Nevada range are associated with dense, older forest. Recent modelling of this species' viability has assumed that future fire will burn with ~90% tree mortality in the Sierra Nevada (He and Mladenoff 1999; Spencer *et al.* 2008; Scheller *et al.* 2011; Syphard *et al.* 2011; Thompson *et al.* 2011).

Conversely, there are also concerns that fire suppression and post-fire management has reduced natural early-successional habitat created by fire (Hutto 2006; Odion and Hanson 2006; Swanson *et al.* 2011), including in the Sierra Nevada (Nagel and Taylor 2005; Odion and Hanson 2006, 2008). High-severity fire has key ecological and evolutionary implications. Many species or vital life history stages (e.g. germination) are endemic to conditions created by high-severity fire, and some rare wildlife indicator species, such as the Black-backed Woodpecker (*Picoides arcticus*), depend upon such habitat (Hanson and North 2008; Hutto 2008). Due to the richness of biological legacies (e.g. snags, downed logs, patches of montane chaparral) in natural early-successional habitat created by high-severity fire, abundance and diversity of plant and animal species is high (Lindenmayer and Franklin 2002; Swanson *et al.* 2011). For many species, post-fire habitat occupancy is brief. Therefore, early-successional habitat created by fire must be continuously replenished for these species to be maintained in a landscape (Bond *et al.* 2005; Hutto 2006).

Existing studies have important limitations for describing fire severity trends in the Sierra Nevada. Miller *et al.* (2009) and Miller and Safford (2012) used a regional fire severity dataset that includes many but not all fires occurring partially on national forest lands, and vegetation maps that post-date the fires were used to identify pre-fire vegetation. Miller *et al.* (2012) hypothesised that maps of potential vegetation that post-date the time series may miss some conifer forests that experienced high-severity fire and are reclassified as montane chaparral. The study by Collins *et al.* (2009), which showed no severity trend, was limited by spatial extent, covering a portion of the Sierra Nevada in Yosemite National Park. Hanson *et al.* (2009), Dillon *et al.* (2011) and Miller *et al.* (2012) found no trend, but their study areas overlapped with the Sierra Nevada ecoregion in small areas. Schwind (2008), though spatially broad, included all non-forest vegetation, which may not reflect burning patterns in conifer forest.
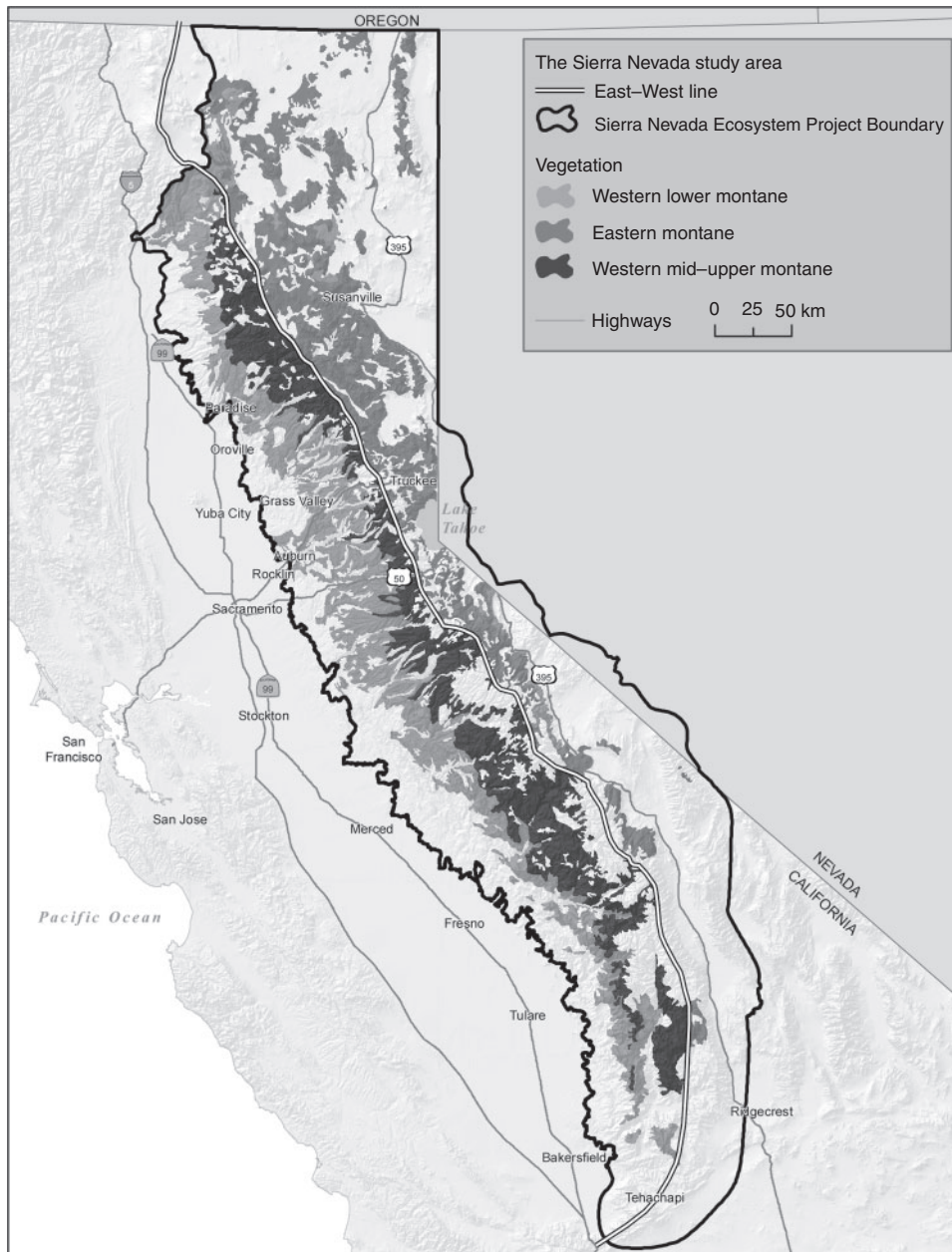
Fig. 1.   The Sierra Nevada and southern Cascades region that has been defined as the Sierra Nevada study area in regional analyses.

Thus, there is a need to comprehensively analyse fire severity data from the forests of the Sierra Nevada to determine whether a trend currently exists with high-severity fire. Herein, using remote sensing data from satellite imagery, we investigate whether trends in fire severity have occurred since 1984 in this region. Specifically, we analyse all available fire severity data to investigate whether trends in proportion, area and patch size of high-severity fire since 1984 exist in three Sierra Nevada forest zones: western lower montane, western mid–upper montane and eastern montane. Our analysis is based upon 24 and 91% additional data relative to the amount in Miller *et al.* (2009) and Miller and Safford (2012) respectively, because we use a

national fire severity dataset that allows more complete coverage of fires, and because we analyse all forested lands.

Materials and methods

Study landscape

To assess high-severity fire trends since 1984, we analysed the entire region covered by the Sierra Nevada Ecosystem Project (SNEP 1996). This area includes all of the Sierra Nevada and southern Cascade mountains within California (Fig. 1). We limited our analyses to forest vegetation of the western lower, mid–upper and eastern montane forests of the study landscape.

These forests were identified by overlaying 1977 vegetation (Cal–Veg, see http://www.fs.fed.us/r5/rsl/projects/classification/zone-map.shtml, accessed 25 January 2013) (Franklin *et al.* 2000) because of the necessity of including all forest present before available fire severity mapping beginning in 1984.

Lower montane forests of the study area are dominated by conifers, including ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), white fir (*Abies concolor* (Gord. and Glend.) Hildebr.), incense-cedar (*Calocedrus decurrens* (Torr.) Florin), sugar pine (*Pinus lambertiana* Dougl.) and mixed and interspersed with the broadleaved deciduous California black oak (*Quercus kelloggii* Newb.) and evergreen canyon live oak (*Quercus chysolepsis* Liebm.) (Barbour *et al.* 2007). We used all forests where the Cal–Veg association was ponderosa pine or mixed conifer-pine (mixed-conifer forest dominated by ponderosa pine) to comprise the western lower montane forest zone.

Mid–upper montane forests are those dominated by white fir, red fir (*Abies magnifica* A. Murr.), Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.), sugar pine and lodgepole pine (*Pinus contorta* Dougl. ex. Loud), which can be mixed with western white pine (*Pinus monticola* Dougl. ex. D. Don.) and mountain hemlock (*Tsuga mertensiana* (Bong.) Carr.) (Barbour *et al.* 2007). This forest zone was identified by selecting areas where the Cal–Veg association was mixed conifer–fir (mixed-conifer forest dominated by white fir), white fir, red fir, Jeffrey pine and lodgepole pine.

Eastern montane forests are comprised of the forest types found in the western mid–upper montane zone, but are located east of the Sierra–Cascade crest and are dominated by Jeffrey pine, occasionally mixed with singleleaf pinyon pine (*Pinus monophylla* Torr. & Frem.), and western juniper (*Juniperus occidentalis* Hook. and subsp. *australis*) east of the crest and in the transition area near the crest (Barbour *et al.* 2007). We identified this forest zone by selecting Jeffrey pine, ponderosa pine, mixed conifer, white fir, red fir and lodgepole pine Cal–Veg associations east of the Sierra–Cascade crest (Fig. 1).

Mid–upper western montane forests and eastern montane forest zones are distinguished from lower western montane forests in that the majority of annual precipitation falls as snow, rather than as rain. The eastern montane forest zone is distinguished from western montane forests by substantially lower annual precipitation on the east side, resulting from the rain shadow created by the Sierra–Cascade crest (Barbour *et al.* 2007).

Identification of forests that burned using vegetation maps

As an initial matter, we assessed the suitability of current post-burn mapping for our analysis. Using the same trend test approach described below in 'Statistical analyses', we assessed whether use of maps of potential conifer forest that post-dated most fires (current mapping from Cal–Veg and Landfire see http://www.fs.fed.us/r5/rsl/projects/classification/zone-map.shtml, accessed 25 January 2013, and http://www.landfire.gov/, accessed 18 January 2013) tend to exclude (i.e. map as non-conifer vegetation) relatively more montane conifer forest experiencing high-severity fire in the earlier years of the time series than in the later years. To test the accuracy of the 1977

Cal–Veg dataset, which pre-dates our time series, we determined the percentage of the 1317 US Forest Service Forest Inventory and Analysis (FIA) plots (http://www.fia.fs.fed.us/tools-data/) in our study area that had forest types matching the three montane forest zones in our study, as described above.

Contemporary high-severity fire

To investigate current high-severity fire trends, we used all 1984–2010 fire severity data, from satellite imagery, available from the US Government's program, Monitoring Trends in Burn Severity (MTBS) (http://www.mtbs.cr.usgs.gov/). These data are based on the Relative delta Normalised Burn Ratio (RdNBR), which adjusts for differing pre-fire vegetation (Miller *et al.* 2009), and compares reflectance from pre-fire vegetation to reflectance at one year post-fire (Miller and Thode 2007). RdNBR data are not gathered for fires <405 ha and, thus are not available. These fires comprised 7.2% of the total area burned, which we determined by comparing the total area of forest with fire severity data in our study area to the total area within all fire perimeters provided by Cal Fire's Fire and Resource Assessment Program, or 'FRAP', which contains a comprehensive database of fire perimeters of all sizes from incident reports (http://www.frap.cdf.ca.gov). To quantify the extent of high-severity fire we used an RdNBR threshold of 641 (Miller and Thode 2007), which translates to ~70% mortality of the pre-fire live basal area in montane forests of the Sierra Nevada, based upon data from US Forest Service field validation plots that assessed basal area mortality at various RdNBR levels (Hanson *et al.* 2010).

Statistical analyses

We used the non-parametric Mann–Kendall trend test (Sokal and Rohlf 1995) to determine whether mapping and fire severity trends existed from 1984–2010. We used this test because the data did not meet the normality assumptions for parametric analyses, the Mann–Kendall test was the most parsimonious test for our univariate hypotheses, and can detect non-linear trends (Önöz and Bayazit 2003). This test is widely used to detect trends over time in environmental data (Önöz and Bayazit 2003; Yue and Hashino 2003; Xiong and Guo 2004), including fire severity trend analyses in conifer forests (Hanson *et al.* 2009). The Mann–Kendall trend test tends to have better statistical power for data that are not normally distributed, including data with high variance in a relatively short time series (Yue *et al.* 2002, figs 7, 8; Önöz and Bayazit 2003), performing 'either as well or better' than parametric regressions for non-normal data (Helsel and Hirsch 2002), such as the data at issue in this study.

Within the Sierra Nevada region as broadly defined by SNEP (1996), we tested for trends in annual high-severity proportion and area in each of the three forest zones (lower western montane, mid-upper western montane and eastern montane), with the year as the independent variable and annual proportion of high-severity fire effects (high-severity fire hectares divided by total fire hectares) or annual hectares of high-severity fire as the dependent variables. We also tested for trends in mean and maximum high-severity patch size across the study area, with the year as the independent variable and annual mean high-severity fire patch size or annual maximum high-severity patch

size as the dependent variables. For the annual mean high-severity patch size, the minimum patch was the pixel (30×30 m), and each group of contiguous high-severity pixels was analysed as a separate patch for the purposes of calculating the annual mean high-severity patch size from 1984–2010. In each analysis, an individual year was the unit of replication. For each time series we tested for autocorrelation using the methods described in Xiong and Guo (2004). In addition, we calculated the rotation for high-severity fire from 1984–2010 in the different forest zones. Fire rotation is the amount of time needed to burn an area equal to the area of interest one time (Heinselman 1973).

Results

Both current Cal–Veg and current Landfire mapping of potential vegetation excluded significantly greater proportions of the areas that were mapped as montane conifer forest by 1977 Cal–Veg, and that experienced high-severity fire, in the earlier years of the time series relative to the later years (Current Cal–Veg: $z = -1.96$, $P = 0.025$; Current Landfire: $z = -2.04$, $P = 0.021$). This effect causes an upward trend in fire severity within montane conifer forest even when such a trend does not actually exist. Because of the trend introduced by mapping that post-dated most fires, we undertook no further analyses using this mapping and relied instead on the pre-burn map: 1977 Cal–Veg. We found that this map was reasonably accurate. The consistency between 1977 Cal–Veg data and FIA data was 88.1, 87.9 and 85.3% for lower western montane, mid–upper western montane and eastern montane zones.

We analysed a total cumulative post-fire area of 608 815 ha of conifer forests as identified by Cal–Veg 1977 for the period 1984–2010. There was no significant trend over this time in high-severity fire proportion in western lower montane ($z = -0.46$, $P = 0.646$), western mid–upper montane ($z = -0.88$, $P = 0.379$) or eastern montane forests since 1984 ($z = -1.71$, $P = 0.087$) (Fig. 2a–c). There was no trend in annual area of high-severity fire in western lower montane ($z = 0.00$, $P = 1.00$), western mid–upper montane ($z = 1.17$, $P = 0.242$) or eastern montane forests ($z = 0.71$, $P = 0.478$) (Fig. 2a–c). Nor was there any trend in high-severity fire mean annual patch size ($z = -0.625$, $P = 0.529$) or maximum annual patch size ($z = 0.17$, $P = 0.865$) (Fig. 3a–b). None of the time series were autocorrelated.

Rotations for high-severity fire of 461, 893 and 714 years were found in western lower montane, western mid–upper montane and eastern montane forests.

Discussion

Our results represent the first quantitative analysis of high-severity fire in the Sierra Nevada region using the most complete and spatially extensive fire severity dataset (MTBS), and covering all lands. We found no evidence that contemporary high-severity fires have increased in proportion, area or patch size since 1984.

Our findings likely differ from those of Miller *et al.* (2009) and Miller and Safford (2012) because we used vegetation mapping that pre-dated the time series. We found that using vegetation mapping that post-dated the time series introduced a

significant, phantom trend of increasing fire severity because of the effect in excluding high-severity fire early in the time series. This supports the hypothesis of Miller *et al.* (2012) that using vegetation maps that pre-date burn severity data will avoid such error. This is an important methodological consideration for assessing trends in burn severity.

A second reason for differences between our findings and previous findings is that we used the national MTBS fire severity dataset, which is more spatially extensive in its coverage of fires than the regional data (http://www.fs.usda.gov/main/r5/landmanagement/gis, accessed 25 January 2013) used in those analyses, which covers 'most', but not all, fires occurring partially on national forest lands (Miller and Safford 2012), representing a 'subset' of such fires (Miller *et al.* 2009). The MTBS data we used included considerably more total burned area within conifer forests (1984–2010) than the regional dataset (Miller and Safford 2012). Our analysis using MTBS data covered 608 815 ha of burned area, compared with 490 277 ha in Miller *et al.* (2009) and 318 192 ha in Miller and Safford (2012); the relative proportions of federal lands (78%) *v.* private lands (20%) in these studies were nearly identical to the proportions in our analysis (79% federal lands *v.* 20% private). Thus, the relative proportions of national forest lands, other federal lands or private lands would not account for differences in findings.

The MTBS dataset includes, in particular, better coverage of smaller or average fire years in the earlier years of the time series. For example, the smallest fire years in the analysis of high-severity fire area in Miller and Safford (2012) were all in the earlier years of the time series, showing values of 0 to ~70 ha of high-severity fire in ponderosa pine and mixed-conifer forests per year in e.g. 1986, 1991, 1995 and 1998 (corresponding to log-transformed values of 0 to 1.85, or extreme low values, in Miller and Safford 2012). In contrast, the MTBS data in our analysis indicates ~400 to 1000 ha of high-severity fire in these forest types in each of these years, corresponding to log-transformed values of 2.6 to 3.0, or average values, in fig. 3 of Miller and Safford (2012). Similarly, the use of a spatially less-extensive regional data approach can exclude particularly large, severe fires in the earlier years of the time series, such as the Fountain fire of 1992 (over 20 000 ha of high-severity fire in ponderosa pine and mixed-conifer forest), which occurred mostly on private lands, but partially on national forest lands, within the Sierra Nevada Forest Plan Amendment management area, but was not included in the Miller and Safford (2012) analysis. It is because the MTBS dataset has the best coverage of fire areas that it has become the standard system used to evaluate fire severity trends in forests of California and elsewhere in the western USA (Hanson *et al.* 2009; Dillon *et al.* 2011; Miller *et al.* 2012).

There may be factors that are mitigating the effects of warming temperatures on fire behaviour. In dry fuels, wind speed is the most important factor in determining fire behaviour (Cruz *et al.* 2004; Cruz and Alexander 2010). Recent research indicates that with climate change, the wind speed probability distribution may be shifting towards slower winds, particularly in mid-latitudes (Pryor and Barthelmie 2010; Pryor and Ledolter 2010). Pryor and her colleagues found that wind speeds appear to be waning in most of the USA, in many locations by more than
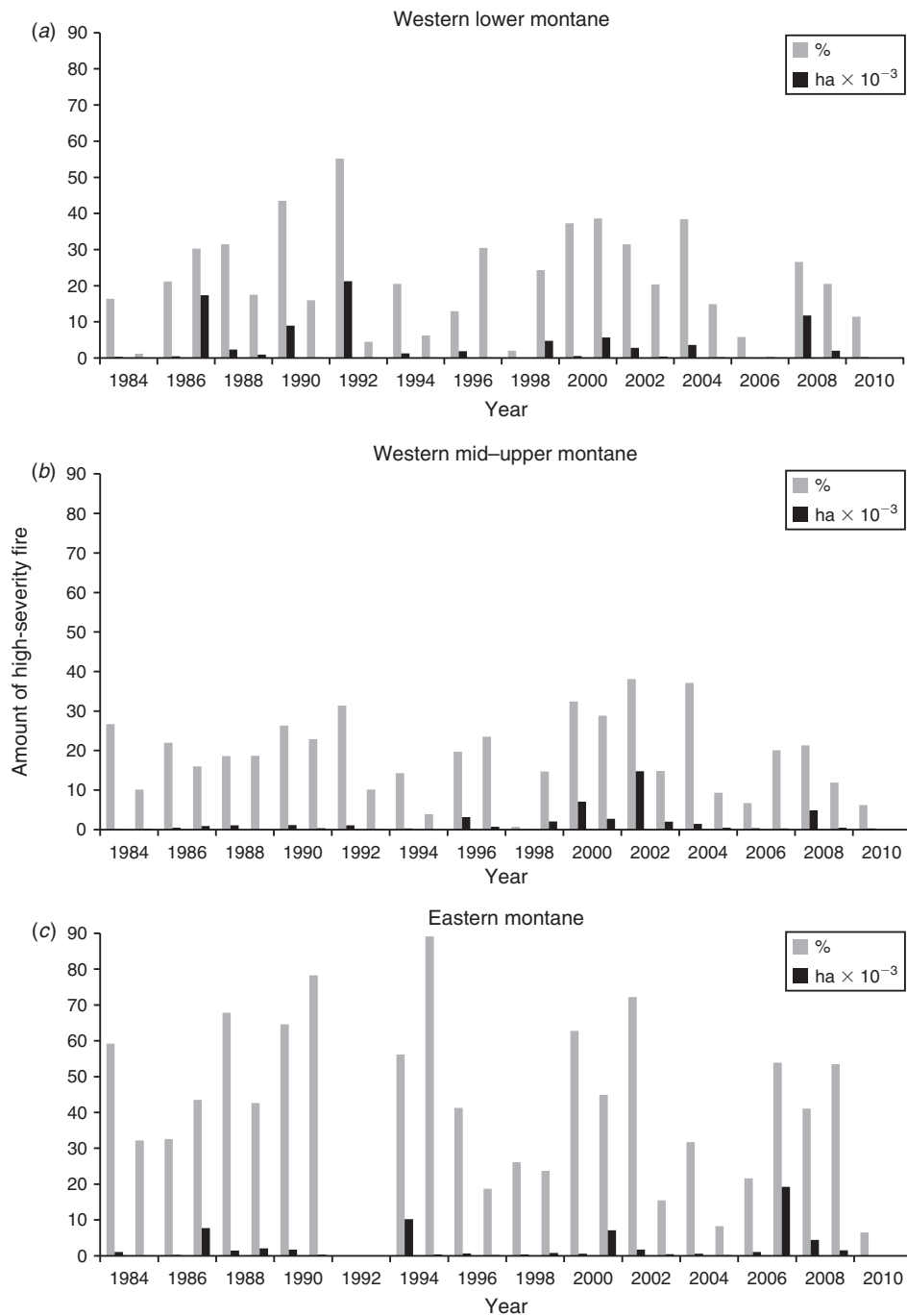
Fig. 2. Percentage of high-severity fire and high-severity fire area from 1984 to 2010 in the montane conifer forests of the Sierra Nevada: (*a*) western lower montane; (*b*) western mid–upper montane and (*c*) eastern montane.

1% per year. Decreases in wind climate may be disproportionately important because of the exponential relationship between wind speed and fire intensity (Byram 1959; Albini and Baughman 1979). However, an analysis of trends in wind climate in the Sierra Nevada region is needed to determine whether the broader trends in the USA apply there as well.

In relatively productive vegetation, fire activity is strongly influenced, inversely, by fuel moisture, which in turn may be controlled by drought and precipitation patterns (Meyn *et al.*

2007; Krawchuk and Moritz 2011). Though precipitation appears to have increased over the past several decades in Sierra Nevada forests (Crimmins *et al.* 2011; North 2012, fig. 3-1), consistent with many global climate model predictions for the Pacific Northwest, USA (Mote and Salathé 2010), precipitation patterns within the shorter time period of our study have been more variable, especially over the latitudinal gradient of the study area. More research is needed to understand how high-severity fire occurrence is affected by the level and seasonality
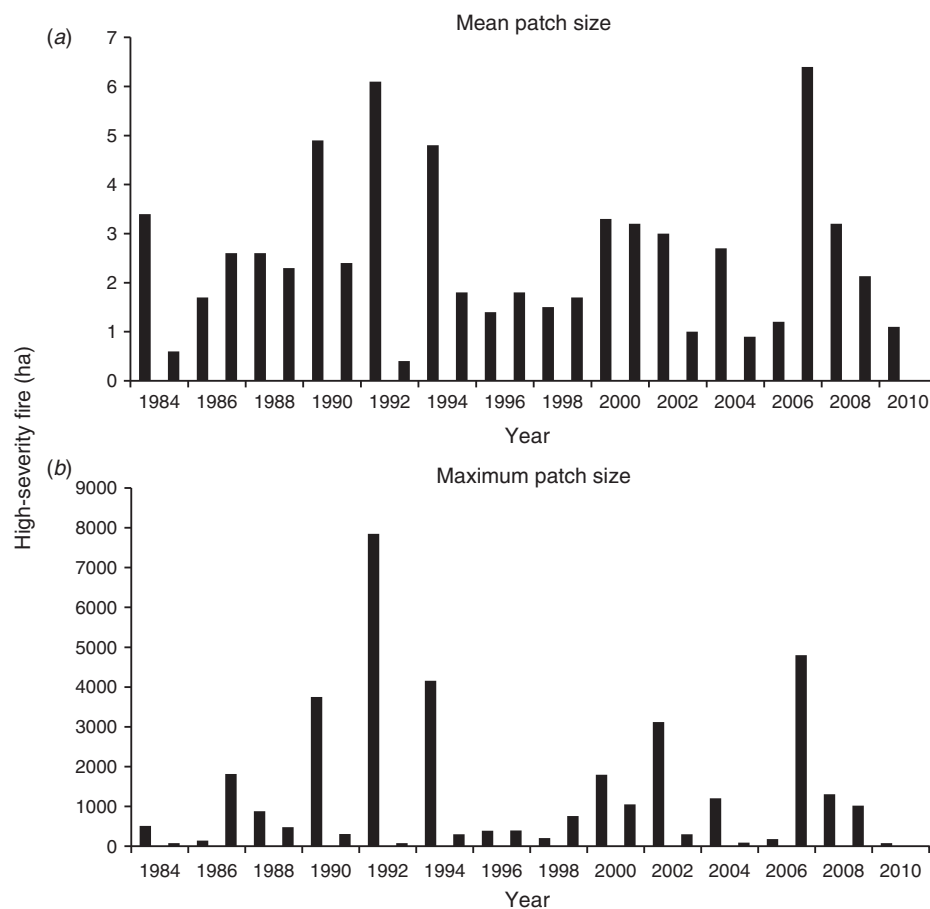
Fig. 3.   Mean (*a*) and maximum (*b*) high-severity fire patch size from 1984 to 2010 in the montane conifer forests of the Sierra Nevada.

of precipitation, and by the influence of precipitation on pyrogenic vegetation (Parisien and Moritz 2009).

In addition to climate, the effects of fire suppression have been assumed to have caused a trend of increasingly severe fire (Miller *et al.* 2009). However, Odion and Hanson (2006, 2008) and van Wagtendonk *et al.* (2012) found that fire severity in Sierra Nevada forests did not increase significantly as a function of a metric (Condition Class) for the magnitude of fire suppression. In fact, in the more extensive fires studied, fire severity tended to decrease with very long fire intervals (Odion and Hanson 2006, 2008). This is supported by recent research on flammability trends in Sierran forests by Schwilk and Caprio (2011). In addition, in the dry forest types in the interior northwest California, which range from open to closed canopy, Miller *et al.* (2012) found modest amounts of high-severity fire that did not differ in long-unburned forests relative to more recently burned forests and, in moister closed forests of this region, fire severity decreased with time since fire, consistent with earlier research (Odion *et al.* 2004, 2010; Miller *et al.* 2012).

Because fire severity has been mostly low and moderate (Fig. 2*a*–*c*), and fires relatively infrequent due to fire suppression, the rotation for high-severity fire since 1984 at the bioregional scale of the Sierra Nevada was a relatively long 645 years. This is longer than rotations before the influence of

fire suppression based on available research that allows calculations of historic rotations. Bekker and Taylor (2001), in a remote unmanaged area of mixed-conifer and upper montane forest in the southern Cascades of California, within our study area, found that 50–60% of these forests experienced high-severity fire over a 76-year period before effective fire suppression. Baker (2012), using US Government field plot data from the mid-late 1800s, found a high-severity fire rotation of 435 years in dry mixed-conifer forests of the eastern Cascades of Oregon, adjacent to our study area. Minnich *et al.* (2000) studied fire severity patterns in mixed-conifer forests of northern Baja California, Mexico within an area that had not been logged or subjected to fire suppression. In these forests, Minnich *et al.* (2000) found a natural high-severity fire rotation of 300 years. In a modelling study reconstructing historic fire patterns, Stephens *et al.* (2007) estimated a high-severity fire rate, before 1850, of 5% every 12 to 20 years for ponderosa pine and mixed-conifer forests of the Sierra Nevada (rotation of 240 to 400 years), and shorter rotations for upper montane fir forests. In another study (Collins and Stephens 2010), an average of 15% high-severity fire was found in reference mixed-conifer forests with overall fire frequencies that were similar to those used in Stephens *et al.* (2007), suggesting similar, or slightly shorter, high-severity fire rotations relative to those modelled in Stephens *et al.* (2007).

Conclusion

Our results highlight the need to use vegetation mapping from before high-severity fire occurs to identify patterns of such fire. Using such a map, we found that the occurrence of high-severity fire does not appear to be increasing in recent decades. High-severity fire remains a significant but minor proportion of overall fire effects that has been reduced by fire suppression. This suggests a need for thoughtful assessment of fire management priorities, which often cite problems of increasing fire severity and amounts that are assumed to be excessive compared with historical amounts (Stephens and Ruth 2005; Spencer *et al.* 2008; Scheller *et al.* 2011; Syphard *et al.* 2011; Thompson *et al.* 2011). For example, management is often aimed at improving fire suppression effectiveness and reducing fire behaviour in forested landscapes (Collins *et al.* 2010; Moghaddas *et al.* 2010) *v.* focussing immediately adjacent to human communities (Schoennagel *et al.* 2009; Schoennagel and Nelson 2011).

Finally, our results suggest that predictions that there will be ample, or excessive, high-severity fire, and plenty of suitable habitat for biota dependent on natural early-successional conditions created by fire (Stephens and Ruth 2005), or a diversity of vegetation age classes created by high-severity fire, may be incorrect. Post-fire early-successional habitat appears to be an important conservation priority, but it is not protected as such (Hanson and North 2008; Hutto 2006, 2008). Moreover, many natural forests have been replaced by even-aged, single species tree plantations or modified by silvicultural activities so that they will not function as post-burn habitat for some species (Hutto 2008). Increased recognition of the relative scarcity and ecological importance of burned forest habitats can improve protection of β diversity (Lindenmayer and Franklin 2002; Lindenmayer *et al.* 2004; Swanson *et al.* 2011).

Acknowledgements

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