Using one year post-fire fire severity assessments to estimate longer-term effects of fire in
conifer forests of northern and eastern California, USA

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Abstract

Fire severity maps derived from immediate or one year post-fire satellite images are commonly used by US federal land managers for making post-fire management decisions and provide fundamental insights into broad scale fire ecology questions. How these maps relate to longer term post-fire conditions in the absence of management actions is an unanswered question, and a key issue for post-fire restoration planning. We characterized field-measured fire effects in forests 5–7 years after fire in severity categories as they were mapped with one year post-fire satellite data. The severity maps we used were produced from a relativized differenced normalized burn ratio (RdNBR) derived from Landsat images and calibrated to three severity metrics: composite burn index (CBI), percent change in basal area (BA), and percent change in tree canopy cover (CC). Relationships of field-measured values with severity generally followed expected patterns. BA of dead trees and shrub cover increased with increasing fire severity. BA of live trees, total tree CC, conifer CC, litter cover, litter depth and duff depth all decreased with increasing severity. When forest type (yellow pine, dry mixed conifer, moist mixed conifer, fir) was added as an interaction term there were differences in the relationships of field-measured values to severity, some of which can be attributed to differences between pine and fir dominated forests. We did not find any difference in live BA between unburned, unchanged, low and moderate severity plots for yellow pine forests; the only significant decrease in live BA was in high severity plots. In contrast, both dry and moist mixed conifer forests showed more or less steadily decreasing live BA with increasing fire severity. For yellow pine dead BA there were no significant differences between unchanged, low and moderate categories, while there was a difference between unchanged and moderate for all other forest types. In the high-severity

categories $\geq 91\%$ of plots had $< 10\%$ conifer CC, and $\geq 82\%$ of plots had $< 10\%$ total tree CC. Comparing our 5–7 year post-fire dataset to a separate dataset measured one year post-fire demonstrates that delayed tree mortality occurs at all levels of severity, and notable snag fall (especially pines) is already underway a half-decade after fire. The effects we measured provide useful information for managers interested in planning for post-fire activities up to 5-7 years after fire, including soil erosion abatement, habitat restoration, fuel and snag management, and tree regeneration management.

Keywords: Basal area, California, canopy cover, composite burn index, fire effects, severity

1. Introduction

Information describing fire effects is needed by land managers for making post-fire management decisions. However, acquiring plot data to assess fire effects is time consuming, expensive, and can only be collected in limited locations. Employing satellite-acquired images calibrated to field-measured variables is a more cost-effective method for extrapolating field measurements over broader scales (Roller and Bergen, 2000). There is a need, therefore, to establish relationships between satellite derived severity maps and detailed fire effects (e.g., tree mortality, surface fuels, etc.) measured on the ground (Morgan et al., 2014).

Severity maps depicting wildland fire effects are commonly produced from satellite images by the US Federal land management agencies (Eidenshink et al., 2007; Parsons et al., 2010; Miller and Quayle, 2015). These data are currently being used by land managers in making post-fire management decisions, for monitoring, and for providing insights into basic broad scale fire ecology questions (e.g., van Wagendonk and Lutz, 2007; Collins et al., 2009; Holden et al., 2009; Dillon et al., 2011; Picotte and Robertson, 2011; Miller et al., 2012a; Mallek et al., 2013;

Cansler and McKenzie, 2014; Steel et al., 2015). Severity maps derived using similar methods have also been used in other fire prone areas of the world (Hall et al., 2008; Kumar et al., 2008; Tanase et al., 2011; Rivera-Huerta et al., 2016).

There have been two major issues with interpreting many commonly produced severity maps. First, the degree of severity depicted on severity maps is often represented in broad categories such as low, moderate, and high without precise definitions of what the categories mean in terms of individual fire effects, e.g., tree mortality. For example, the composite burn index (CBI), a field-based estimate of fire severity, has been widely used to describe fire effects in mapped categories on some severity maps (e.g., Cocke et al., 2005; Miller and Thode, 2007; Cansler and McKenzie, 2012). The CBI protocol includes separate ocular estimates of individual effects to soils, fuels and four vegetation strata, which are typically integrated into a single composite value (Key and Benson, 2006a). Although CBI was designed to correlate to severity, CBI values are not as meaningful for forest and wildlife management as leaf area index, tree basal area or canopy cover (Cade, 1997; Broham and Bryant, 2005; Boer et al., 2008; Miller et al., 2009a; Kolden et al., 2015). Second, some methodologies used for creating severity maps from satellite images require unique calibrations for each fire because of differences in vegetation types and density (Spanner et al., 1990; Miller and Yool, 2002; Key and Benson, 2006b; Kokaly et al., 2007; Miller and Thode, 2007). Some of the most readily available severity maps in the US are produced by the Monitoring Trends in Burn Severity (MTBS) program (www.mtbs.gov). The MTBS maps are derived from satellite images converted to a satellite index [the differenced normalized burn ratio (dNBR)] that is closely associated to the absolute change pre- to post-fire in chlorophyll (Eidenshink et al., 2007; Miller and Thode, 2007). Chlorophyll content is directly tied to vegetation type and density, however, and therefore a

particular change in chlorophyll can represent different severities across fires, and even with the same fire (Miller and Yool, 2002). Consequently, MTBS image analysts, who have no direct knowledge of ground conditions, derive categorical severity maps by choosing satellite index thresholds unique to each fire to delineate severity categories. This methodology leads to uncertainty in the precise definition of the severity categories and an inability to directly compare severity ratings between fires, and can translate into high classification error rates (Kolden et al., 2015). In a study conducted in the Pacific Northwest region of the US, Whittier and Gray (2016) found that MTBS categorical maps reflected lower severity than did tree mortality data on 51% of forest inventory plots.

In an attempt to remedy these issues, Miller and Thode (2007) described a new satellite index, a relativized version of the differenced normalized burn ratio (RdNBR), and for California forests produced calibrations between one year post-fire (“extended assessment”) Landsat data and CBI measured in 741 field plots from 14 wildfires. Miller and Thode claimed that their approach allowed for calibrations to field-measured variables to be applied to future fires without further field validation. Subsequently, Miller et al. (2009a) published calibrations of one year post-fire Landsat data to field-measured percent change in canopy cover (CC) and tree basal area (BA) using field data from 16 additional California fires to the Miller and Thode (2007) dataset, and demonstrated that those calibrations as well as their earlier CBI calibration remained valid for other fires in the same study area. Most recently Miller and Quayle (2015) published calibrations of immediate post-fire images (“initial assessments”) to CBI, percent changes in tree BA and CC for a suite of California fires.

While there are many examples in the literature of satellite index calibrations to CBI (e.g., Hall et al., 2008; Holden et al., 2009; Picotte and Robertson, 2011; Cansler and McKenzie, 2012),

Miller et al. (2009a) and Miller and Quayle (2015) are the only studies to date that report calibrations to percent change in BA and CC that are being used to produce severity maps routinely used by forest managers beyond the scope of an individual study (e.g., Harvey, 2015). There is a need to better describe how these maps relate to important vegetation and ground surface variables, as well as to understand how these maps relate to field-measured variables over longer time frames. Dunn and Bailey (2016) reported tree mortality and structure attributes within severity categories delineated by MTBS categorical severity maps. As far as we are aware, however, no study to date has described field-measured understory characteristics (i.e., herbaceous and shrub cover) across the full range of mapped severity categories (i.e., low, moderate and high). Hudec and Peterson (2012) is the only study that has described field-measured fuel characteristics in mapped categories of BA change. In addition, no study to this point has differentiated these effects between different forest types (e.g., yellow pine, mixed conifer, fir). Finally, there is a major gap in our understanding about how post-fire forest conditions change over time within the different fire severity categories.

Because the satellites most commonly used to make severity maps measure the reflectance from sun light, the satellite images primarily detect changes to the overstory of closed canopy forests, or a combination of understory and overstory in open canopy forests (Spanner et al., 1990; Stenback and Congalton, 1990). Consequently, satellite derived severity data are highly correlated to first order fire effects to overstory trees, i.e., live conifer BA and canopy cover decrease with increasing severity, but dead conifer BA increases (Hudec and Peterson, 2012; Miller and Quayle, 2015; Dunn and Bailey, 2016). Secondary overstory fire effects and effects to understory fuels, herbaceous plants, shrubs and hardwoods, however, are closely associated with interacting individual plant and plant community responses to fire, successional/life histories,

fire intensity and site potential (i.e., biophysical setting) (Agee, 1993; Fites-Kaufman et al., 2006).

It is beyond the scope of this paper to fully describe all possible first and second order effects interactions. However, some generalizations can be made for conifer forests in our study area based upon existing literature. Coarse woody fuels (≥ 1000 hour timelag), litter and duff fuel loadings are all initially reduced by fire (Kilgore, 1971; Campbell et al., 2007; Webster and Halpern, 2010; Vaillant et al., 2013; Welch et al., in press). However, litter, duff, and coarse woody fuels all increase with time since fire at rates that are dependent upon tree species, tree diameter, and degree of tree scorch and/or mortality (Webster and Halpern, 2010; Dunn and Bailey, 2012; Vaillant et al., 2013; Dunn and Bailey, 2015). A greater extent of exposed soil due to a decrease in litter and duff cover in concert with opened tree canopies, especially in severely burned areas where almost all trees are killed, usually leads to an increase in herbaceous and/or shrub cover (Collins et al., 2007; Wayman and North, 2007; Webster and Halpern, 2010; Abella and Springer, 2015; Welch et al., in press). All of the major hardwood species that occur in the conifer vegetation types in our study sprout after fire (Sugihara et al., 2006). Consequently, hardwood cover initially decreases with fire, but usually recovers within a couple years. In areas where competition with conifers is reduced, hardwood cover can become greater than prior to fire (Cocking et al., 2014; Collins and Roller, 2013; Crotteau et al., 2014).

In this manuscript our principal purpose was to develop a better understanding for how field-measured fire effects 5-7 years post-fire in forests of northern and eastern California, USA, relate to fire severity mapped using one year post-fire (so-called “extended assessment”) RdNBR images calibrated to CBI, percent change in tree BA and percent change in tree CC. We hypothesized that across all forest types 5–7 years post-fire dead tree BA, herbaceous cover and

shrub cover would all increase in concert with increasing fire severity. In contrast, we expected that live tree BA, total tree canopy cover, conifer canopy cover, litter cover, and litter and duff depth would all decrease with increasing severity. We hypothesized that shrub height and hardwood cover would be highest in unburned and severely burned areas. Because pine species in our study area are generally more resistant to low- to moderate-intensity fire than firs (Martin, 1963; Ryan and Reinhardt, 1988; Miller, 2000), we also expected that tree mortality would differ by forest type, with the type most dominated by fire resistant pines showing less difference in tree mortality between unchanged, low and moderate severity. Finally, we hypothesized that after the first year, additional conifer mortality would occur due to fire induced stress and bark beetle attack (Youngblood et al., 2009; Hood et al., 2010; van Mantgem et al., 2011), and percent decrease in live tree BA between one year and 5-7 years post-fire would be related to higher severity.

2. Methods

2.1. Study area

Fires used in this study are located within the Sierra Nevada Forest Plan Amendment (SNFPA) planning area (USDA, 2004) and the four National Forests in northwestern California (Fig. 1 and Table 1). The SNFPA planning area, which guides land and resource management on 50,000 km² of National Forest land on eleven National Forests, includes the Sierra Nevada and its foothills and also the Warner Mountains, Modoc Plateau, White Mountains, Inyo Mountains, and portions of the southern Cascades. Climate is Mediterranean-type, with warm, dry summers and cool, wet winters; nearly all precipitation falls between October and April (Minnich, 2007). Forest vegetation is diverse, with different dominant species and high variation in density and

vertical structure. The fires in this study were all located in montane forests comprised mostly of conifers: at lower elevations ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), Douglas fir (*Pseudotsuga menziesii* [Mirb.] Franco) and hardwoods (mostly *Quercus* spp.) dominate; intermediate elevation mixed conifer forests are primarily composed of white fir (*Abies concolor*), incense cedar (*Calocedrus decurrens* [Torr.] Florin), sugar pine (*Pinus lambertiana* Douglas), ponderosa pine, and Douglas fir; and at the higher elevations Jeffrey pine (*Pinus jeffreyi* Balf.), lodgepole pine (*Pinus contorta* ex Loudon), and red fir (*A. magnifica* A. Murray) are the most common trees. Open canopy stands generally have more understory vegetation than closed canopy stands. Fires east of the Sierra Nevada crest and in the Cascade Range were generally located in ponderosa pine and Jeffrey pine forests with an understory of grasses and Great Basin shrub species, such as sagebrush (*Artemisia tridentata* Nutt.) and bitterbrush (*Purshia tridentata* [Pursh] DC). These eastside forests where ponderosa pine and Jeffrey pine dominate are often referred to as “eastside pine” (Smith 1994).

Climate in northwest California is also Mediterranean, but moister on average than the SNFPA area, with higher spatial variability due to proximity to the Pacific Ocean and steep, complex topography (Skinner et al., 2006). While most of the dominant conifer species are shared with the Sierra Nevada and Cascade Ranges, overall conifer diversity is higher with correspondingly greater vegetation diversity and complexity (Barbour et al., 2007). Midstory evergreen and deciduous hardwood trees are often more abundant than in the Sierra Nevada (Sawyer and Thornburgh, 1977).

Across our study area, average area burned per year increased at the end of the 20th century, but much of the study area has not burned since before the beginning of the 20th century (Miller et al., 2009b, 2012b; Safford and Van de Water, 2014; Steel et al., 2015).

2.2. Plot data

2.2.1. Long term post-fire

The USDA Forest Service's Pacific Southwest Region Ecology Program has collaborated with the University of California-Davis and other partners to establish a network of plots to investigate factors associated with conifer seedling establishment within fires 5–7 years after they occurred (Fig. 1 and Table 1; Welch, 2015; Welch et. al., in press). A sample of 60 m² circular (8.74 m diameter) plots on a grid at an interval of 200 m was established in each fire. Because we were unable to sample the plots prior to the fires, unburned control plots were sampled in a 400 m buffer outside of fire perimeters as a proxy for the pre-fire state. Plot locations were chosen to represent approximately the same range of elevations, slopes, and aspects as the within-fire plots (Welch et al., in press). Variables measured in each plot included stand basal area (BA) of live and dead trees measured with a BA gauge (5, 10 or 20 basal area factor), and ocular estimates of percent cover to the nearest 5% of litter, coarse woody debris (CWD; woody debris >7.6 cm diameter), live understory vegetation and live tree canopy within the plot (Table 2). The modal height of each shrub species was measured at a location that visually appeared representative of the average height. Litter and duff depth were recorded as an average of three locations that visually appeared to be typical in each plot (Welch, et al., in press). We only analyzed plots that were established in the four major conifer forest types: yellow pine (YP), dry mixed conifer (DMC), moist mixed conifer (MMC) and fir (FIR) (see section 2.3). We examined pre- and post-fire aerial photos for all plots to determine whether any management actions that would affect tree counts (e.g., salvage harvest) occurred after fire containment and before plots were established. Plots where post-fire management actions were

visible were discarded. Our final dataset – a subset of the plots analyzed in Welch et al. (in press) – included 952 burned plots and 81 unburned plots (total = 1033).

2.2.2. One year post-fire

The long-term post-fire plot data were not co-located where short-term post-fire effects had previously been assessed on the ground. Therefore to compare long term effects to one year post-fire effects, we used a large dataset where dead and live trees were measured one year post-fire in the same general study area that were previously used to calibrate the satellite data to the three severity metrics (Fig. 1 and Table 1; Miller et al., 2009a). The data include status (live or dead), species and diameter at breast height (DBH). The same plot size was not used on all fires. For fires that occurred in northwest California DBH was measured on each tree >10 cm in a 30 m diameter plot. The numbers of trees <10 cm DBH were counted by species. For fires in the SNFPA, plots were either 60 m or 90 m in diameter depending upon the year sampled. Individual tree status and species of all trees >0 cm dbh were tallied in 10 cm size classes within fixed subplots at each plot location. Number and size of subplots varied, see Miller et al. (2009a) for details. All trees were measured in a few plots of 10m or 15m diameter during the first year. After that, all trees <20cm were measured in four 2.5m diameter subplots and trees $\geq$ 20cm in four 5m diameter subplots. Presence or absence of dead needles as well as bark and wood consumption patterns were used to determine whether trees were killed by the fire or were dead prior to the fire. We only analyzed plots where pre-fire tree CC modeled with the Forest Vegetation Simulator (FVS) was at least 10% (Crookston and Stage, 1999; Dixon, 2002; Miller et al., 2009a), and were in one of the four major conifer forest types: YP, DMC, MMC and FIR (see section 2.3). No plots were salvage logged before the plot data were collected. This dataset totaled 852 plots.

2.3. Vegetation data

To determine whether fire effects differed by forest type, we stratified the plots based upon their pre-fire type in vegetation maps that use the Classification and Assessment with Landsat of Visible Ecological Groupings (CALVEG) vegetation classification system (Keeler-Wolf, 2007; USDA, 2014a). CALVEG contains vegetation groupings of a finer resolution than most classification systems focused on fire regimes (e.g., LANDFIRE biophysical settings or environmental site potential; Comer et al., 2003; Keeler-Wolf, 2007; Rollins, 2009; Van de Water and Safford, 2011). Because the limited number of plots in our study did not permit a robust sample of all CALVEG vegetation types, we used the most recent version of CALVEG prior to each fire to group our plots into four major forest types that closely follow the LANDFIRE system and the pre-settlement fire regime groups in Van de Water and Safford (2011): YP, DMC, MMC and FIR (Tables 3 and 4).

2.4. Severity data

The USDA Forest Service, Pacific Southwest Region maintains a geographical information system (GIS) polygon database of fire severity data containing most large wildfires since 1984 that have occurred at least partially on Forest Service lands in California (USDA, 2014b). All severity data used in this study were mapped using a post-fire image acquired during the summer of the calendar year after fire containment, i.e., an extended assessment (Key, 2006). The severity data were all derived from the relativized differenced normalized burn ratio (RdNBR) to compensate for different pre-fire vegetation conditions and permit inter-fire comparisons of severity (Miller and Thode, 2007). The RdNBR data were converted to units of the composite burn index (CBI), percent change in BA, and CC based upon regression calibrations to plot data

previously reported in Miller and Thode (2007) and Miller et al. (2009a). Following procedures described by Miller and Thode (2007), we applied a 3x3 focal mean to average RdNBR pixel values before applying the regressions and converting the raster data to polygons in order to match the 90 m diameter of the field plots used by Miller and Thode (2007) and reduce the number of single pixel polygons in our database.

The database provides multiple categories for each severity metric. For example, there are seven categories of BA change: 0%, >0% <10%, ≥10% <25%, ≥25% <50%, ≥50% <75%, ≥75% <90% and ≥90%. For this study we combined several of the default categories for BA and CC change into four categories to match the number of default CBI severity categories (Table 5). We also added an unburned category as a control using plots placed outside the fire perimeters (see section 2.2.1 and Table 5).

2.5. Analyses

We used Generalized Linear Models (GLMs) to test for differences in field-measured variables in four ways. Because the RdNBR thresholds for BA and CC are very similar, but differ somewhat from CBI (Table 5), we first compared the field-measured values between the three severity metrics (i.e., severity category was an interaction term) within each of four severity categories (unchanged, low, moderate and high). Unburned plots were not included because they were outside the fire perimeters and were thus identical for each severity metric. Second, for each severity metric we compared differences in field-measured variables between all five severity categories. Third, we again compared differences in field-measured variables between severity categories for each severity metric, but stratified the plots by pre-fire forest type (i.e., forest type was an interaction term). Finally, within each of the four severity categories

we compared live and dead BA between plots measured one year post-fire and plots measured 5–7 years post-fire.

All variables were examined to determine which response probability distribution to use in the GLMs. Litter cover data were arcsine-square root transformed to better fit a normal distribution. A lognormal distribution was used for litter depth and shrub height. An exponential function was used for all other variables. Default link functions were used in all cases; identity for normal and log normal distributions, and log for exponential distributions. *P*-values were adjusted using the Tukey-Kramer method to account for multiple comparisons (Kramer, 1956). All GLM analyses were performed in SAS version 9.4 with the GLIMMIX statement. The SLICEDIFF option was used to test pairwise differences of one factor at a fixed level of another factor. Significant differences between factors are reported at $P < 0.05$.

Forest Service vegetation classification standards specify that “forested” areas must have at least 10% tree CC (Brohman and Bryant, 2005). We therefore produced histograms of live tree and conifer CC measured in plots that were mapped in the highest severity categories to determine the percentage of plots mapped in the highest severity categories that were in a non-forested condition 5-7 years post-fire.

3. Results

When comparing field-measured values between severity metrics we did not find any differences between the BA and CC metrics (Fig. 2). The only differences we found were between CBI and either BA or CC change in three cases: BA live high severity, conifer cover high severity, and duff depth low severity. The results make some sense because the CBI high severity category threshold is higher (Table 5), reflecting more severe effects (BA live and conifer cover are lower for CBI), and the CBI low severity threshold is lower, reflecting less

severe effects (CBI duff depth is higher). The significant differences between severity metrics were small, however (between 22% and 33%), and small standard errors contributed to statistical significance (Fig. 2). Patterns between severity categories were similar for all severity metrics, although there were some differences in significance between categories. We therefore only report the results for the BA metric for all remaining analyses. Results of the GLMs comparing field-measured values between severity categories for CBI and CC are provided in the Supplementary material (Figures S1 and S2).

All GLM's comparing differences in field-measured variables between severity categories were significant at $P < 0.05$ except for CWD. When forest type was added as an interaction term all models including CWD were significant.

3.1. Live and dead tree basal area

Live BA decreased and dead BA increased with increasing severity (Figs. 2 and 3). Live BA in the high severity plots was significantly lower than all other severities, and dead BA in the high severity plots was significantly higher than all other severities. Additionally, dead BA was smallest in unburned plots. However, adjacent severity categories were not always significantly different. Live BA in the unburned and unchanged severity plots were never significantly different.

When forest type was added as an interaction term there were not any statistical differences in YP live BA between unburned, unchanged, low and moderate severity (Fig. 4). In contrast, live BA unburned was different from low and moderate severity for DMC and MMC. For YP dead BA there were no significant differences between unchanged, low and moderate severity, while there were differences between unchanged and moderate for all other forest types. Patterns of BA live and dead for FIR were similar to DMC and MMC except differences between some

severity categories were not significant due to large standard errors, probably mostly due to the small number of FIR plots (Table 4).

We did not find a statistical difference in the unchanged severity category between plots measured 5–7 years post-fire and one year post-fire for either live or dead BA (Fig. 5). When plots were burned at low severity, live BA was significantly lower in the 5–7 year post-fire category, but dead BA was not. When plots were burned at either moderate or high severity, both live and dead BA were significantly less 5–7 years post-fire compared to one year post-fire. The high severity category showed the largest percent decrease between one year and 5–7 years post-fire for live BA (53%), as well as for dead BA (39%).

3.2. Tree live canopy cover

Patterns for total tree cover and conifer cover were similar. Cover generally decreased with increasing severity, and high severity resulted in significantly lower cover than the other severity category except for DMC (Figs. 2-4). Similar to live BA, unburned and unchanged severity were never significantly different.

Hardwood cover was lowest in the moderate severity category, highest in unburned plots, and unchanged plots were similar to high severity (Figs. 2 and 3). Adding forest type as an interaction term revealed that hardwood cover was negligible in FIR regardless of severity as well as in burned YP plots (Fig. 4). Hardwood cover was greatest in unchanged DMC plots and unburned MMC plots.

A high percentage (91–93%) of high-severity plots had less than 10% conifer CC, with only slight variation among the severity metrics (Fig. 6A). When hardwood cover was included, the number of plots with less than 10% tree cover fell to between 82% and 86% (Fig. 6B).

3.3. *Shrubs and herbs*

Shrub cover generally increased with increasing fire severity, except that unchanged was similar to unburned (Figs. 2 and 3). When forest type was added as an interaction term, shrub cover in the high severity plots was significantly greater than low severity in MMC and FIR, but not in YP or DMC (Fig. 4).

Shrubs were generally tallest when plots remained unburned or burned at high severity (Figs 2 and 3), except for the YP type where there was not any difference in shrub height across all severity categories (Fig. 4).

Herbaceous cover was lowest in unburned plots, and highest at moderate severity (Fig. 3). High severity was not significantly different from either unchanged or low.

3.4. *Litter, duff and CWD cover*

Litter cover, litter depth, and duff depth all generally decreased with increasing fire severity (Fig. 2), except in the case of duff depth where moderate severity was greater than low or high severity. When forest type was added as an interaction term, duff depth in unburned and unchanged plots, and litter depth in unburned plots remained significantly greater than high severity for all forest types (Fig. 4). Unburned litter cover was significantly greater than high severity only for DMC and MMC.

CWD was not significantly different between severity categories for any severity metric when plots from all forest types were pooled (results not shown). But, when forest type was added as an interaction term, YP was the only forest type with statistical differences in CWD cover between severity categories, with the unchanged category having the lowest cover (Fig. 4).

4. Discussion

Our study contributes to the growing body of science relating satellite-derived fire severity measures to actual conditions on the ground. In this case, we used data from nearly 1900 field plots to show how conditions 5–7 years after fire in a variety of vegetation and ground cover characteristics (tree basal area and canopy cover, shrub height and cover, CWD cover, litter depth and cover, and herb cover) are related to fire severity. This complements information from one year post-fire calibrations made previously (Miller and Thode, 2007; Miller et al., 2009a; Miller and Quayle, 2015), and provides insight into general trends in these characteristics over the first half decade or so after fire. In the following discussion we highlight some of our major findings, and we focus on a couple of important management implications.

4.1. *Live and dead tree basal area*

We found that general relationships of fire severity with tree mortality 5–7 years after fire followed expected patterns. Dead BA increased, and live BA, total tree cover, and conifer cover decreased with increasing severity (Figs. 2 and 3). However, the relationship between tree mortality and fire severity differed somewhat depending upon forest type. YP forests appeared to exhibit a smaller difference in mortality between the unburned and moderate severity categories compared to DMC and MMC. This may be due to the higher proportion of tree species (*Abies* spp. especially) with thinner bark and lower crown base heights in the DMC and especially the MMC types. The dominant tree species in YP (ponderosa and Jeffrey pine) are generally more resistant to low- to moderate-intensity fire (Martin, 1963; Ryan and Reinhardt, 1988; Miller, 2000; Hood et al., 2007; Stephens and Finney, 2002). Given the lower resistance of fir species to low- to moderate intensity fire, the pattern in live and dead BA and the tree cover variables in the

FIR plots seems odd as we did not find significant differences between unchanged, low and moderate severity in most cases. However, we caution against over-interpretation here as the plot numbers in the FIR type were low, resulting in high standard errors.

Our results for YP differ somewhat from Kane et al. (2013), who measured a loss of large trees in ponderosa pine forests that had burned at low severity in Yosemite National Park. Kane et al. (2013) were surprised by their findings and speculated that one possible reason for their results was that their study period (1984–2010) spanned an extended drought period (1987–1992) during which a sizeable proportion of large ponderosa pine trees died due to beetle outbreaks, independent of fire (Guarín and Taylor, 2005). A contributing factor may have been a thick layer of litter and duff at the base of large trees that accumulated with decades of fire suppression. Deep litter/duff layers can smolder for a long period of time after being ignited by a fire, girdling the tree and/or causing cambial injury. Girdling alone can cause delayed tree mortality, but it also increases the odds that the tree will succumb to bark beetle attack (Hood, 2010). Ponderosa pine forests in Kane et al.'s study area, however, were rather limited in extent (1089 ha). As a result, localized mortality may have had an undue influence on their results. In contrast, our YP plots were acquired in ten fires over a much larger geographic extent, and our study period was shorter (2001–2012) and did not include a period as dry as the 1987–1992 drought.

Comparing live and dead BA 5–7 years post-fire vs. one year post-fire appears to exhibit delayed mortality of trees that were alive after the first year, as well as a declining persistence of standing fire-killed trees in areas burned at moderate and high severity (Fig. 5). Our results are consistent with previous observations that delayed mortality can continue 6–8 years after a fire (Hood et al., 2010; van Mantgem et al., 2011), and most pine species fall within the first 7–10

years after a fire (Harrington, 1996; Ritchie et al., 2013). Also consistent with other research, our data appear to show that pine snags decay and fall more quickly than fir snags (Russell et al., 2006; Dunn and Bailey, 2012; Ritchie et al., 2013). Comparing across forest types we did not find any difference 5–7 years post-fire in live BA in any severity category (results not shown). However, MMC dead BA 5–7 years post-fire was significantly greater than YP and DMC for low, moderate and high severity (results not shown).

While we would have preferred to have a single set of plots for comparing 5–7 years vs one year post-fire, both of our datasets were robust, consisting of 852 and 952 plots, one year and 5–7 years post-fire respectively (Tables 1 and 5). And, although different field protocols (fixed plot vs. point) were used to estimate BA between the two datasets, the sampling errors were likely similar (Kulow, 1966; Rice et al., 2014).

4.2. Tree live canopy cover

Tree cover and conifer cover do not appear to always mirror live BA in their relationship with fire severity, especially for YP, which has a higher percentage of pine trees compared to DMC, MMC and FIR (Fig. 4; Fites, 1993; Potter, 1994; Smith, 1994). There are a couple possible explanations. First, for YP there were less than ten plots within each of the unburned and unchanged categories (Table 4), and tree cover was not measured in five unchanged plots (Table 2; results not shown). As a result, the standard errors are large and the mean values may not be reliable. However, if the means are dependable then the explanation may be related to the fire resistance traits of yellow pines. Although yellow pines can survive low intensity fire better than other trees, portions of crowns can be killed, decreasing canopy diameter, raising canopy base height and subsequently lowering crown volume (Ryan and Reinhardt, 1988; Stephens and Finney, 2002; Hood et al., 2010). Near-infrared wavelengths that are recorded by the satellite

(and used to create the severity data) are sensitive to the amount of chlorophyll within the total canopy (i.e., canopy volume; Asner, 1998), and differences between mapped severity categories primarily reflect differences in total chlorophyll. Consequently, depending upon tree species and fire intensity, differences in satellite values may not represent actual tree mortality, but rather only a change in total live canopy. Finally, satellite values may also be influenced by changes in the amount of live understory vegetation under open canopies (Spanner et al., 1990; Stenback and Congalton, 1990).

Combining our values of tree canopy cover for DMC and MMC, our unburned and low severity plots (results not shown) are very similar to what Collins et al. (2011) found in mixed conifer forests within Yosemite National Park (52% and 40% vs. 53% and 42%, unburned and low respectively, values from Collins et al. were estimated from Fig. 2, pg. 7). However, our moderate severity tree canopy cover value was a little lower (20% vs. 28%). Most of Collins et al.'s plots were collected longer after fire compared to our plots (11–13 years vs. 5–7 years). Most delayed mortality occurs within the first 6–8 years (Hood et al., 2010; van Mantgem et al., 2011). Therefore, differences in delayed tree mortality were not likely a substantial factor in explaining any differences in our canopy cover values. Our estimates were ocular, however, which can be biased depending upon the observer (Korhonen et al., 2006). In contrast, Collins et al.'s canopy cover values were calculated using the Forest Vegetation Simulator (FVS) and were corrected for canopy overlap (Dixon, 2002). However, when trees are clustered, FVS can overestimate canopy cover (Christopher and Goodburn, 2008). In addition, densely stocked mixed conifer forests burned at moderate severity can have a canopy–gap arrangement where tree clumps formed by remnants of densely spaced trees alternate with gaps consisting of fire killed trees (Kane et al., 2013). Collins et al. did not correct their FVS canopy cover estimates for

any fire-caused decrease in canopy diameter. It is therefore possible that Collins et al. overestimated canopy cover for moderately burned sites.

In severely burned areas, post-fire sprouting of hardwoods can lead to their temporary dominance in stands that were conifer-dominated before fire (Cocking et al., 2014; Collins and Roller, 2013; Crotteau et al., 2014). We found that hardwood cover contributed to 5–7 year post-fire canopy cover in higher severity plots (Fig. 6). In addition, we found that post-fire hardwood cover in high severity plots was significantly higher than conifer cover for DMC and MMC, but not for YP or FIR (Fig. 4; GLM results not shown). Hardwoods are uncommon in upper montane FIR forest types, and our YP plots are heavily weighted towards eastside pine and Jeffrey pine (82% of our 5–7 year YP plots), which support less of a hardwood component than YP forests in northwestern California or the west side of the Sierra Nevada (Potter, 1994; Smith, 1994; Youngblood et al., 2004; Sawyer et al., 2008).

4.3. Shrubs and herbaceous cover

Our results for shrub and herbaceous cover appear to be consistent with most research in mixed conifer forests. Other studies have found that herbaceous cover was greater than or equal to pre-fire levels by the second year after fire (Collins et al., 2007; Wayman and North, 2007; Webster and Halpern, 2010). We found that herbaceous cover was higher in burned areas 5–7 years post-fire compared with unburned.

We found that post-fire shrub cover generally increased with fire severity, especially in the moister, higher elevation MMC and FIR forest types. Other studies that focused on mixed conifer forests have also shown that the amount of shrub cover is positively correlated with severity, and that after 5 to 10 years shrub cover in areas burned at low and moderate severity surpasses pre-fire levels (Webster and Halpern, 2010; Abella and Springer, 2015). Limiting our

plots only to mixed conifer types (DMC and MMC) we also found that shrub cover was significantly higher in areas burned at high severity (means of 12%, 18%, 19% and 32% for unchanged, low, moderate and high severity, respectively; $P < 0.0001$).

Areas of high shrub cover resulting from conifer forests burning at high severity are of concern where management objectives call for maintaining conifer forest for two reasons. First, natural conifer regeneration is lower in severely burned areas compared with areas that burned at low or moderate severity, and shrub cover is negatively correlated to conifer seedling establishment (Crotteau et al., 2013; Collins and Roller, 2013; Welch et al., in press). Second, because fires in shrub fields are usually stand-replacing, shrub fields can become a self-reinforcing alternative state when fire return intervals are sufficiently short to prohibit conifer forest reestablishment (Coppoletta et al., 2016; Lauvaux et al., 2016).

4.4. Litter, duff and CWD cover

Studies have generally found that litter and duff fuel loadings were reduced by fire and remained lower than in unburned areas for at least 10 years (Kilgore, 1971; Webster and Halpern, 2010; Vaillant et al., 2013). We did not directly measure litter and duff fuel loading. However, depth measurements are correlated to loading (Knapp et al., 2005), and so in our plots litter and duff fuel loadings were almost certainly reduced in comparison with unburned plots.

We found that duff depth for areas burned at moderate severity was greater than areas burned at low or high severity 5-7 years post-fire (Fig. 2). At first glance that may seem counter intuitive because other researchers have found that duff and litter is greatly reduced across all severities (Campbell et al., 2007). But trees in areas that burn at moderate severity are usually scorched to a greater degree than either low severity (few trees are scorched and/or scorch heights are shorter) or high severity (on many trees needles are incinerated). Scorched needles usually drop within

the first couple years and then mostly decompose within 5-7 years post-fire (Stohlgren, 1988; Keifer et al., 2006). Sloughing bark from dead trees will also contribute to the litter layer, especially in severely burned areas that have a larger dead tree component. However, the largest contribution of sloughed bark decomposition to duff would mostly likely occur after 5-7 years post-fire (Dunn and Bailey, 2015).

CWD was not significantly different across fire severity classes in any of our models when forest types were combined. CWD is usually reduced by fire, but then increases with time (Webster and Halpern, 2010; Ritchie et al., 2013; Vaillant et al., 2013). Some studies have found that a large proportion of pine snags that were killed by fire fall or lose large portions of their tops within 7 to 10 years, but fall rate also depends upon tree diameter (Harrington, 1996; Laudenslayer, 1997; Knapp et al., 2005; Dunn and Bailey, 2012). Dead white fir trees fall at a slower rate than pines, however (Raphael and Morrison, 1987; Landram et al., 2002; Dunn and Bailey, 2012). Stephens et al. (2012) found that CWD fuels in mixed conifer forests in our study area were reduced by prescribed fire and remained lower seven years after treatment. Other studies have reported that CWD fuel loads 8–10 years after prescribed fire were similar or higher than prior to being burned in both ponderosa pine and mixed conifer forests (Keifer et al., 2006; Chiono et al., 2012). It therefore seems likely that CWD that had been consumed in our fires had largely been replaced 5–7 years post-fire (except in YP forests) by falling limbs, tops, and trees. It takes multiple low to moderate severity fires, such as were common in these forests before Euro-American settlement, to result in a long-term reduction in CWD (Safford and Stevens, in press).

The fact that we found less CWD in YP forests at unchanged, low and moderate severity compared with high severity may reflect the same pattern that we found in dead and live tree BA

(Fig. 4). As noted above, we also found no difference in live and dead BA between unchanged, low and moderate severity, likely because of thicker, more fire resistant bark of ponderosa and Jeffrey pines and their generally higher crown-base heights (Martin, 1963; Ryan and Reinhardt, 1988). As a result, there were fewer dead trees to create CWD.

4.5. Homogeneity of effects within map categories

The severity maps we used in this study were made using satellite images acquired one year post-fire. Therefore the map categories were primarily influenced by first order overstory effects. However, variability in the 5-7 year post-fire plot data within the mapped categories is associated with both the original first order overstory effects, and additional second order effects that occurred after the first year, which may increase or decrease the variability of effects within each severity category.

Canopy cover and BA are not uniformly distributed throughout mapped high severity patches. Because mature individuals of most of the conifers that comprise the forest types in our study have adaptations to survive low intensity surface fires (e.g., thick insulating bark), crown fire is the typical fire behavior that causes high mortality rates (i.e., high severity) (Skinner and Taylor, 2006; Skinner et al., 2006; van Wagendonk and Fites-Kaufman, 2006; Safford and Stevens, in press), which would seem to homogenize the high severity category. However, there is a transition zone between areas with mostly live trees and mostly dead trees associated with the transition between surface and crown fire. The length of that transition zone is dependent upon factors such as tree density, crown base height, ladder fuels and wind speed (Van Wagner, 1977). Where the high severity patch boundary begins within that transition zone is dependent upon the mortality threshold used to define “high severity.” When a lower high severity threshold is chosen (e.g., 75%) then more live trees will occur inside the “patch” in close

proximity to the patch boundary compared with a higher threshold (e.g., 90%) (Miller and Quayle 2015). At increasing distance away from the patch boundary the proportion of tree mortality usually increases to nearly 100%. The pixel size of the satellite images used to make the maps and the fact that pixels are square also contribute to patch boundary accuracy. Square 30m x 30m pixels rarely align with actual patch edges and therefore contain a mixture of effects integrated over the area of the pixel.

Using the same plot data from some of the fires sampled one year post-fire used in this study, Miller and Quayle (2015) found that >61% of the plots with live trees in the high severity category occurred within 30 m inside high severity polygon boundaries. Our 5–7 year post-fire plots were similarly affected, with 58% of plots with live trees in the BA high severity category occurring within 30 m of high severity polygon boundaries (results not shown). As a result, even though thresholds for the high severity categories may indicate BA change of $\geq 75\%$, that does not mean that BA change is uniformly $\geq 75\%$ within the entire area. Rather, percent change increases for areas ≥ 30 m inside the high severity polygon. The percentage of plots <30m and ≥ 30 m inside the high severity polygon boundaries that had live trees was 60% and 14%, respectively (results not shown). If $\geq 90\%$ BA change were to be used as the high severity threshold, then the percentage of plots ≥ 30 m inside the high severity polygon boundaries with live trees goes down to 9% (results not shown). As a result mean values in the high severity categories should be considered to be conservative estimates for areas ≥ 30 m inside severely burned areas. For example, live BA in the 5–7 year post-fire plots averaged $3.1 \text{ m}^2 \text{ ha}^{-1}$ in the high severity category (Fig. 3), but the averages varied from $7.0 \text{ m}^2 \text{ ha}^{-1}$ for plots <30 m to $1.9 \text{ m}^2 \text{ ha}^{-1}$ for plots ≥ 30 m inside the high severity polygons, respectively (difference significant at $P < 0.001$, results not shown). Comparing two narrower high severity categories of BA change

($\geq 75\%$ < 90% and $\geq 90\%$) highlights the difference in effects close to the edge versus further inside high severity patches (the $\geq 75\%$ < 90% category normally occurs within a narrow 1–2 pixel wide band encircling the $\geq 90\%$ category [Miller and Quayle, 2015]). We found significant differences between the $\geq 75\%$ < 90% and $\geq 90\%$ severity categories for live BA, dead BA, duff depth, litter depth, tree cover and conifer cover, but not shrub height, shrub cover, herb cover, litter cover and hardwood cover (Supplementary material Figure S3).

There is not a distinct transition in effects between unchanged, low and moderate severity categories because effects are related to different intensities of surface fire, and surface fire with torching of small groups of trees. Variability within and between these categories is often dependent upon the degree of tree canopy cover and integrating overstory and understory effects over the area of a 30m x 30m pixel.

4.6. Applicability to other regions

The field-measured values in our study area may not be not representative of forests throughout the western US, but some of our results may be informative for similar forest types. Hudec and Peterson (2012) is the only study that we are aware of that has reported field-measured variables outside of California by severity category based upon severity maps generated using the California calibrations (i.e., Miller and Quayle, 2015). Hudec and Peterson conducted their study in mid-to-upper elevation mixed conifer forests on the eastside of the Cascades, and reported live tree canopy cover values 2-3 years post-fire that are similar to our 5–7 year post-fire MMC values, except their live tree canopy cover values for the 50–75% BA change severity category were slightly higher (17-25% vs. 12%).

5. Management Implications

5.1. Estimation of longer-term fire effects

The 1976 National Forest Management Act (NFMA) and Forest Service regulations require that productive forest land be restocked after a major stand altering event, such as stand-replacing fire. Five years after fire is an important threshold, because after that burned areas tend to require major extra investments in site preparation before planting, which usually means that neither site prep nor planting are actually undertaken. Our study shows that one year post-fire severity assessments made using field-calibrated RdNBR can be reasonable predictors of general patterns 5–7 years after the fire has been extinguished. Our study provides quantitative estimates of average patterns and trends in tree, vegetation and ground-cover characteristics (see Welch et al., [in press] for estimates of tree regeneration). Managers can use these values to broadly estimate what sorts of conditions are likely to characterize areas that have burned at different severities a half-decade after fire. Such information can be utilized to more efficiently plan and implement, among other things, soil erosion mitigations, habitat restoration, snag density and fuel management, and tree regeneration management.

Our large dataset also complements other (mostly more localized) studies in showing that tree mortality measured directly after fire is only a proportion of the actual mortality that will occur, and that both live and dead tree basal area continue to fall after fire, especially in areas that have burned at high severity.

5.2. Project planning

It is important for managers to consider the scale and geolocation accuracy of satellite derived severity maps when using them for post-fire project planning/analyses. Variability in

plot values for some severity categories is high, leading to a lower confidence in precise estimates at specific locations, especially when the number of severity categories is large (i.e., >4). For example, performing an analysis across all forest types with a greater number of narrower severity categories of BA change (0%, $>0\% <10\%$, $\geq 10\% <25\%$, $\geq 25\% <50\%$, $\geq 50\% <75\%$, $\geq 75\% <90\%$ and $\geq 90\%$) we found no statistical difference between mean field sampled values in the middle severity categories ($\geq 10\% <25\%$, $\geq 25\% <50\%$, and $\geq 50\% <75\%$) for most of the variables that we examined in our study (Supplementary material Figure S3). However, when the number of categories is small (i.e., ≤ 4), the likelihood increases that differences in average field-measured values between severity categories will be statistically significant, especially when the number of plots is high. But, that does not suggest that average values are precise estimates at every pixel location when the variability within severity categories is high. Therefore for project planning, the maps are best used to ascertain broad scale patterns and identify where ground surveys and additional analyses should be conducted to more precisely guide management actions.

6. Conclusions

Fire severity maps are used to describe effects on fuels, vegetation, soils, wildlife habitat and for planning post-fire management actions intended to mitigate potential erosion and/or advance vegetation recovery (i.e., reforestation) (Kuenzi et al., 2008; Parsons et al., 2009; Vogelmann et al., 2011; Miller and Quayle, 2015; Jones et al., 2016). Severity, loosely defined in the fire ecology literature as the effect of fire on an ecosystem (Agee, 1993; Sugihara et al., 2006), however, is dependent upon the resource of interest (e.g., vegetation or soils). In addition, different levels of impacts to a resource (e.g., conifer forest) that define “high” severity may also differ depending upon management objectives. For example, California spotted owl nesting

habitat is associated with forests with at least 70% canopy cover (North et al., 2000; Phillips et al., 2010). Therefore fire effects that reduce canopy cover below 70% may be considered “high severity” from an owl nesting habitat perspective. Consequently “severity” maps need to describe ecological effects in measurement units that are meaningful for making management decisions (Morgan et al., 2014). In this contribution, we characterized ecological effects that managers can use to broadly estimate what sorts of conditions are likely to be associated with different levels of severity for three different severity metrics a half-decade after fire.

Our results generally follow and appear to confirm expected patterns between levels of severity and forest types. Differences in mean field-measured values between severity categories were not always significant, however, for at least two reasons. First, although we had a fairly large number of plots overall, when stratified by forest type there were large standard errors associated with the categories with the fewest number of plots. Second, there are inherent limitations to the moderate resolution (both spatially and spectrally) satellite images that are used to create the severity data. Fire effects to overstory and understory cannot always be reliably differentiated, and because fire can transition from surface to crown fire within the distance of a pixel’s width, effects within a pixel can vary widely. However, the results presented here should be adequate for estimating broad scale fire effects across severity classes that can be expected 5–7 years after fires in our study area where post-fire management actions have not altered conditions. The information may be useful for planning management actions associated with soil erosion abatement, habitat restoration, fuel and snag management, and tree regeneration management.

Acknowledgements

We gratefully acknowledge the work of our field technicians for collecting the 5-7 year post-fire plots data between 2009 and 2012: V. Alla, A. Ayalon, G. Bohlman, C. Carlson, T. Farnum, L. Jerabkhova, A. Jirka, B. Lee, C. Preston, C. Richter, T. Roth, M. Safford, and D. Schmidt. Funding for this study was provided by the USDA Forest Service, Pacific Southwest Region.

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Figure 1. Study area and fires where fire effects were measured 1 and/or 5-7 years post-fire. Fires with names underlined were sampled both 1 and 5-7 years post-fire.

Figure 2. Generalized Linear Model results comparing field-measured values by severity category (unchanged, low, moderate and high) between three severity metrics [percent basal area change (BA), composite burn index (CBI) and percent change in canopy cover (CC)] as mapped with satellite data. Bars represent means and error bars are standard errors. Bars labeled with different letters within the same field variable indicate *P*-values using a Tukey-Kramer adjustment are significant at $P < 0.05$.

Figure 3. Generalized Linear Model results comparing field-measured values in unburned plots and percent change in basal area severity categories as mapped with satellite data. Bars represent means and error bars are standard errors. Bars labeled with different letters within the same field variable indicate *P*-values using a Tukey-Kramer adjustment are significant at $P < 0.05$.

Figure 4. Generalized Linear Model results comparing field-measured values in unburned plots and percent change in basal area severity categories by forest type as mapped by CALVEG. Bars represent means and error bars are standard errors. Bars labeled with different letters within the same forest type indicate *P*-values using a Tukey-Kramer adjustment are significant at $P < 0.05$. See Table 3 for forest type definitions.

Figure 5. Generalized Linear Model results comparing field-measured live (A) and dead (B) tree basal area (BA) 1 year and 5-7 years post-fire as mapped in severity categories of percent change

in BA in one year post-fire severity maps. Bars represent means and error bars are standard errors. Bars labeled with different letters within the same percent change in BA severity category indicate P-values using a Tukey-Kramer adjustment are significant at $P < 0.05$. Numbers inside bars are percent change between one year and 5-7 years post-fire.

Figure 6. Frequency of field plots by proportion of A) live conifer canopy cover and B) total tree canopy cover classified as high severity.

Table 1. Fires and number of plots used in this study. Plots are where pre-fire tree cover was $\geq 10\%$ and pre-fire forest type was yellow pine, dry mixed conifer, moist mixed conifer, or fir.

Fire year	Fire name	National forest	Number	Number	CALVEG date
			of years post-fire	of burned plots	of unburned plots
2001	Blue	Modoc	1	70	1991
2001	Gap	Tahoe	1	17	1996
2001	Star	Eldorado, Tahoe	1	90	1996
2001	Stream	Plumas	1	41	1996
2002	Birch	Inyo	1	11	1990
2002	Cone	Lassen	1	58	1991-1999
2002	McNally	Sequoia	1	164	1995-1999
2002	Showers	Lake Tahoe Basin	7	17	1996
2003	Albanita/Hooker	Sequoia	1	47	1995-1999
2003	Dexter	Inyo	1	20	1990
2003	Kibbie	Stanisluas	1	32	1995
2003	Mountain Complex	Stanisluas	1	24	1995
2003	Mud	Stanisluas	1	40	1995
2003	Spanish	Mendocino	5,7	138	14 1994-2003
2003	Whit	Stanisluas	1	22	1995
2004	Deep	Sequoia	5	2	1995-2001

2004	Freds	Eldorado	5,7	89	3	1996
2004	Power	Eldorado	1	90		1996
2004	Power	Eldorado	5	127	9	1996
2004	Sims	Shasta-Trinity, Six Rivers	6	87	8	1994-2003
2004	Straylor	Lassen	1	57		1991-1999
2004	Straylor	Lassen	5	41		1991-1999
2005	Harding	Tahoe	5	56	6	1996
2006	Bar	Klamath, Shasta-Trinity	5,7	67	7	1994-2003
2006	Bassetts	Tahoe	5	83	7	1996-2000
2006	Hancock	Klamath	1	16		1994-2004
2006	Ralston	Eldorado, Tahoe	5	46	10	1996-2005
2006	Rush	Klamath	1	21		1994-2004
2006	Somes	Six Rivers	1	16		1994-2004
2006	Titus	Klamath	1	11		1994-2004
2006	Uncles	Klamath	1	5		1994-2004
2007	Antelope	Plumas	5	64	8	1996-2000
2007	Moonlight	Plumas	5	135	9	2000

Table 2. Variables measured in 5-7 year post-fire plots.

Variable	Units	Measurement protocol	Years measured
Live tree basal area (BA)	m ² /ha	Basal area gauge (5, 10 or 20 factor) measured from plot center.	2009-2013
Dead tree BA	m ² /ha	Basal area gauge (5, 10 or 20 factor) measured from plot center.	2009-2013
Litter	%	Ocular estimate of litter cover within the plot.	2009-2013
Litter depth	cm	The average of litter depth at three representative places in the plot.	2009-2013
Duff depth	cm	The average of duff depth at three representative places in the plot.	2009-2013
Coarse woody	%	Ocular cover estimate of woody debris >7.6 cm diameter (≥ 1000 h timelag fuels) within the plot.	2009-2013
Herb	%	Ocular cover estimate of herbs (forbs + grass) within the plot.	2010-2013
Shrub	%	Ocular cover estimate of shrubs within the plot.	2009-2013
Shrub height	cm	Average of most common height of each shrub species within the plot.	2009-2013
Tree cover	%	Ocular estimate of total live tree canopy cover within the plot. Note this	2011-2013

value is not a sum of conifer and
hardwood cover due to canopy
overlap.

Conifer cover	%	Ocular estimate of live conifer canopy cover within the plot.	2010-2013
Hardwood cover	%	Ocular estimate of live hardwood canopy cover within the plot.	2010-2013

Table 3. Pre-fire forest type groupings for field plot pre-fire CALVEG dominant vegetation type.

Forest type	CALVEG dominance type
Yellow pine (YP)	Ponderosa Pine
	Ponderosa Pine - White Fir
	Eastside Pine
	Jeffrey Pine
Dry mixed conifer (DMC)	Douglas-Fir - Ponderosa Pine
	Mixed Conifer - Pine
Moist mixed conifer (MMC)	Pacific Douglas-Fir
	Douglas-Fir - White Fir
	Mixed Conifer - Fir
Fir (FIR)	Red Fir
	White Fir

Table 4. Number of unburned plots and burned plots mapped in extended assessments with severity categories of percent basal area change by forest type.

Forest type	Basal area change	
	severity	# plots
Yellow pine (YP)	Unburned	8
	Unchanged	9
	Low	28
	Moderate	54
	High	92
Dry mixed conifer (DMC)	Unburned	31
	Unchanged	11
	Low	42
	Moderate	71
	High	256
Moist mixed conifer (MMC)	Unburned	32
	Unchanged	23
	Low	37
	Moderate	65
	High	195
Fir (FIR)	Unburned	10
	Unchanged	14
	Low	19

Moderate	10
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High	26
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Table 5. Severity metric categories and number of plots 1 year and 5-7 years post-fire.

Severity category	Basal area change	BA # plots	BA # plots	Composite burn index	CBI # plots	Canopy cover change	CC # plots
	(BA)	1 yr	5-7 yrs	(CBI)	5-7 yrs	(CC)	5-7 yrs
	[% (RdNBR values)]	post-fire	post-fire	[CBI (RdNBR values)]	post-fire	[% (RdNBR values)]	post-fire
Unburned	N/A	0	81	N/A	81	N/A	81
Unchanged	0 (<167)	97	57	<0.1 (<69)	33	0 (<161)	70
Low	>0 <25 (≥ 167 <370)	220	126	≥ 0.1 <1.25 (≥ 69 <316)	119	>0 <25 (≥ 161 <367)	131
Moderate	≥ 25 <75 (≥ 370 <574)	194	200	≥ 1.25 <2.25 (≥ 316 <641)	299	≥ 25 <75 (≥ 367 <572)	203
High	≥ 75 (≥ 574)	341	569	≥ 2.25 (≥ 641)	501	≥ 75 (≥ 572)	548
Total		852	1033		1033		1033

Figure 1

Figure 2

Figure 3

Figure 4

Figure 4 continued.

Figure 5

Figure 6