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Post-fire surface fuel dynamics in California forests across three burn severity classes

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Abstract. Forest wildfires consume fuel and are followed by post-fire fuel accumulation. This study examines post-fire surface fuel dynamics over 9 years across a wide range of conditions characteristic of California fires in dry conifer and hardwood forests. We estimated post-fire surface fuel loadings (Mg ha^{-1}) from 191 repeatedly measured United States national inventory plots in dry conifer and hardwood stands of 49 California forest wildfires and identified differences across fire severity classes – low, moderate and high. No significant change in duff load was detected within the first 9 years post-fire across all forest types and fire severities. Litter, 1-h and 10-h fuels exhibited a quadratic trend over time in dry conifer stands, peaking ~6 years after fire, whereas hardwood stands displayed a constant rate of increase in those fuel types. For 100- and 1000-h fuels, the annual rate of change was constant for dry conifer and hardwood stands with differing rates of change across fire severity classes. This study was based on an extensive, spatially balanced sample across burned dry conifer and hardwood forests of California. Therefore, the estimated patterns of fuel accumulation are generally applicable to wildfires within this population.

Additional keywords: fire severity, fuel, post-fire impacts.

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Introduction

Forest wildfires have been increasing in frequency, extent and severity in the western United States (Stephens 2005; Westerling *et al.* 2006; Dennison *et al.* 2014). Historically, dry, pine-dominated forest ecosystems in California were burned in frequent, low-severity fires (Miller *et al.* 2009). Extensive fire suppression over the last century has resulted in surface and canopy fuel load increases (Parsons and DeBenedetti 1979; Knapp *et al.* 2005), which, in turn, has increased the susceptibility of forest ecosystems to high-severity fires (Stephens *et al.* 2009). As large, severe wildfires presently become more common in western North America, post-fire management, such as reforestation and fuel-reduction treatments as tools to reduce susceptibility of reburn, has become a major concern for land managers (Thompson *et al.* 2007; Stevens-Rumann *et al.* 2012; Dunn and Bailey 2015). Therefore, it is important to understand post-fire stand structure and fuel dynamics (i.e. changes in fuel load over time), as they are the main drivers of reburn severity (Coppoletta *et al.* 2016). Yet, information about post-fire fuel dynamics is limited (Agee and Huff 1987; Hall *et al.* 2006) and often restricted to case studies of high-severity fires (Dunn and Bailey 2015), prescribed fires (van Mantgem *et al.* 2016) or one-time post-fire measurements across fire severity classes (Hudec and Peterson 2012).

There are multiple approaches to study post-fire fuel dynamics. Detailed studies involving repeated-measurements of fuel conditions on a site by establishing a network of litterfall traps are considered the most accurate (e.g. Busing *et al.* 1999; Keane 2008). However, this approach is costly, limited in temporal and spatial scale, and inference is limited to the study sites where these networks have been established. Another commonly used approach is the analysis of a chronosequence of selected stands that burned at different times (i.e. a space for time substitution; e.g. Passovoy and Fulé 2006; Jenkins *et al.* 2012; Dunn and Bailey 2015). Chronosequence analyses and similar cross-sectional studies assume that the only relevant difference among stands is the time since fire (Johnson and Miyanishi 2008). Yet, differences in post-fire fuel dynamics among sites may be influenced by other factors, such as differences in landscape and biophysical conditions, fire behaviour, and climatic variation. Because chronosequences are based on selected stands (e.g. pure ponderosa pine stands that burned in high severity fires), analysis results may not reflect the full range of variability occurring across the landscape. Yet another approach is to

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quantify fuel consumption and fire dynamics based on prescribed fires (e.g. Hille and Stephens 2005; Vaillant *et al.* 2009). However, there may be important differences in fire behaviour between prescribed fires and wildfires that may affect consumption rates and post-fire fuel dynamics. Many other studies are based on a single fire event (e.g. Campbell *et al.* 2007; Ritchie *et al.* 2013), which limits the scope of inference. Post-fire fuel dynamics (i.e. changes in post-fire fuel loadings) can also be monitored as part of a regional or national multipurpose inventory and monitoring system that collects relevant information on ground and surface fuels. This approach allows direct observation of a representative cohort of burned plots over time, which is considered the most reliable approach to study temporal changes (Walker *et al.* 2010). An additional advantage of analysing data from large-scale inventory and monitoring systems is that they sample the fire conditions (e.g. fire severity) in proportion in which they occur on the landscape, ensuring that the results can be generalised to the population that the plots represent, provided that the plots were selected from a well-defined population with known probabilities (Eskelson *et al.* 2016).

In the present study we used an extensive set of national inventory plots from California, United States of America (US), which was measured 1 year after a wildfire and remeasured at least once thereafter within the first 9 years following a wildfire. The remeasured inventory plots allowed us to estimate trends of post-fire fuel loadings over 9 years post-fire in two forest types – dry conifers and hardwoods – across three fire severity classes. Our two objectives were to (1) describe post-fire woody surface fuel, litter and duff loadings in California's dry conifer and hardwood forests, and (2) identify differences in post-fire trajectories of woody surface fuel, litter and duff loadings across fires that burned at low, moderate and high severity. Quantifying post-fire fuel dynamics across fire severity gradients provides critical information for forest managers who have to make informed fire management and mitigation decisions (Dunn and Bailey 2015), such as prescribing and implementing fuel reduction treatments to minimise risk of high-severity reburns or predict future fire behaviour.

Methods

Plot selection and fire severity

We used data collected by two programs of the USDA Forest Service: (1) the Forest Inventory and Analysis (FIA) program administered by the Pacific Northwest Research Station, and (2) the Vegetation Mapping and Inventory Program of Region 5 (R5). Each year, a spatially balanced panel of FIA field plots – one FIA plot per 24 km^2 – is visited, resulting in each plot being remeasured every 10 years (Bechtold and Patterson 2005). Plots from the R5 sample are a spatial intensification of the FIA plots within lands managed by the USDA Forest Service (approximately twice the number) in selected forest types that were identified with remote sensing imagery. The plot layout, field work and variables measured were identical for both sets of plots.

One year after the fire seasons of 2002–03, 2006–09 and 2012–13, forested plots in public land that fell within a large fire perimeter were included in the Fire Effects and Recovery Study (FERS), following recommendations by Jain *et al.* (2010). Field measurements at this visit included the standard suite of FIA measurements detailed below (USDA Forest Service 2016) and supplemental metrics to assess fire severity and effects. For each tree in the plot, the proportions of the compacted crown ratio that was unburned (green), scorched (brown) and burned (black) were estimated. Based on these data and the post-fire crown fire severity index (Jain and Graham 2007), we classified each FIA plot into three severity classes: low, or mostly green crowns (at least 75% of the trees of the plot with more than 30% of the crown unburned); high, or mostly brown or black crowns (at least 75% of the trees in the plot with less than 3% of the crown unburned); and moderate, a mixture of green, brown and black crowns (the remaining plots). The threshold of 30% of the crown unburned was chosen by Jain and Graham (2007) because trees with a crown ratio greater than 30% have a high chance of survival and can respond with increased growth after the disturbance.

Then, we selected the FIA and R5 plots that had a fire visit – this is the measurement within 1 year post-fire – and at least one additional post-fire visit thereafter as part of the regular inventory panel assignment ($n = 191$). One dry conifer plot had two plot visits after the initial fire visit. Based on the FIA's forest type definition (USDA Forest Service 2016, appendix E), we classified the plots into two large forest type groups, to which we will refer as dry conifer ($n = 109$) and hardwood ($n = 82$). The dominant forest type within the dry conifer group was the California mixed conifer (83 plots). This is a complex association of intermixed conifer species, typically including ponderosa pine (*Pinus ponderosa* Lawson), sugar pine (*Pinus lambertiana* Douglas), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), red fir (*Abies magnifica* A. Murray) and incense cedar (*Calocedrus decurrens* (Torr.) Florin) (USDA Forest Service 2016). The remaining dry conifer forest types occupied similar biophysical conditions, but one of the component species dominated. They included the Douglas-fir (bigcone Douglas-fir (*Pseudotsuga macrocarpa* (Vasey) Mayr) and Douglas-fir forest types, 5 plots), ponderosa pine (ponderosa, Jeffrey (*Pinus jeffreyi* Balf.) and sugar pine forest types, 15 plots) and the fir–spruce (white fir and red fir forest types, 6 plots) groups. The dominant hardwood group was the western oak forest type group (71 plots). It includes the grey pine (*Pinus sabiniana* Douglas), California live oak (*Quercus agrifolia* Née), canyon live oak (*Quercus chrysolepis* Liebm.), California black oak (*Quercus kelloggii* Newberry) and interior live oak (*Quercus wislizenii* A.DC.) forest types. The remaining hardwood forest type groups were the tanoak/laurel (*Lithocarpus densiflorus* (Hook. & Arn.) Rehder and *Umbellularia californica* (Hook. & Arn.) Nutt. forest types, 10 plots) and Pacific madrone (*Arbutus menziesii* Pursh, 1 plot). To avoid including very different ecological conditions, we excluded the wet, coastal forest types and subalpine and alpine forest types. The 191 plots were observed within the fire boundaries of 49 large fires (Fig. 1).

FIA surface fuel data collection and data compilation

We used the following classification of woody surface fuels based on piece diameter, which is commonly used in wildland fire science in the US: 1-h (<0.63 cm; twigs), 10-h (0.63–2.54 cm; branches), 100-h (2.54–7.62 cm; large branches) and

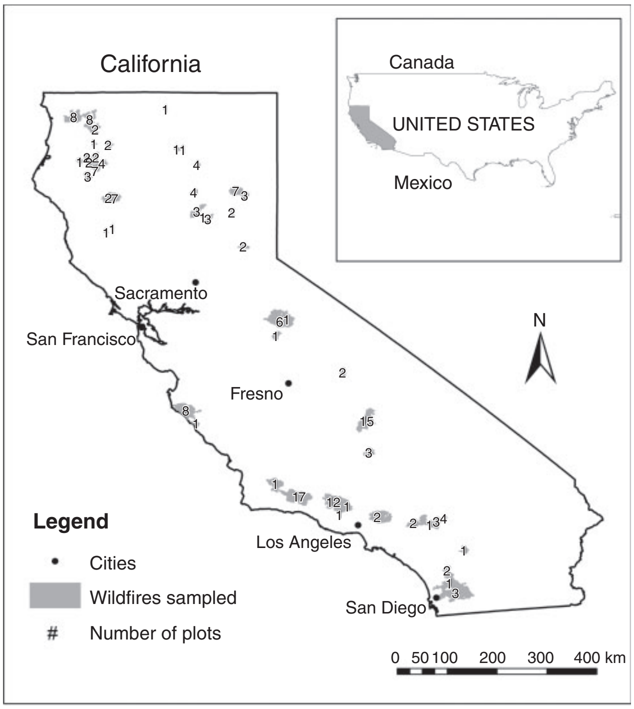

Fig. 1. Map of California fires and fire complexes included in this study. The number of Forest Inventory and Analysis (FIA) plots observed within each fire or fire complex is indicated next to each fire boundary.

1000-h (>7.6 cm; logs) fuels (Lutes *et al.* 2009). The 1-, 10- and 100-h fuels will be referred to as 'fine woody' fuels and 1000-h fuels as 'coarse woody' fuels. Litter is dead, non-woody plant biomass that is freshly fallen, mostly plant foliage. Directly below the litter layer is the duff layer. Duff is partially decomposed, decaying organic material that is no longer identifiable (Keane 2015). These four woody components together with litter and duff provide an optimal description of fuelbeds (Lutes *et al.* 2009). Litter and duff combined will be referred to as ground fuels in the following.

Coarse wood pieces with diameter greater 7.6 cm and length greater than 0.9 m (i.e. 1000-h fuels) were sampled using two 7.32-m radial transects at each of four points in every FIA plot, resulting in a total transect length of 58.5 m for one plot. The species, diameter at the intersection point and degree of decay of each piece were recorded. The 1000-h fuel loads were then calculated based on line-intersect sampling theory (see eqn 4 in table 3.1 in Woodall and Monleon 2008) and the application of species-specific wood density and decay-class reduction factors (Harmon *et al.* 2008, 2011). The 100-h fuels were sampled in four 3.05-m transects per plot (total length: 12.2 m). The 10- and 1-h fuels were sampled in four 1.83-m transects per plot (total length: 7.3 m). The number of pieces that intersected the transect was tallied, and fuel loads estimated following eqn 3.28 in Woodall and Monleon (2008). Litter and duff depth was measured to the closest 0.25 cm in 8 points per plot, at the end of the 1000-h fuel transects. Litter and duff loads were estimated following eqn 3.29 in Woodall and Monleon (2008).

We did not include changes in shrub and herbaceous fuels in our study because we did not have access to adequate inventory information about these fuel types. We also did not have information about salvage logging. However, the FIA plots analysed in this study were in public land and many of them in protected areas, where salvage logging after a fire is limited. Therefore, we assume that the plots in this study were not affected by salvage logging.

Statistical data analysis

The response variables for this study were woody surface fuel (fine and coarse), litter and duff loads. These fuel loads were positive, very skewed and often zero – especially in high-severity fires. Thus, we modelled these fuel loads (Mg ha^{-1}) as a function of time since fire (TSF) using multiplicative models by fitting Poisson pseudo-maximum likelihood models (Santos Silva and Tenreyro 2006) implemented in SAS PROC GLIMMIX (SAS Institute Inc., Cary, NC, USA). A random plot effect was included in the model to account for the repeated-measurements on plots. A random fire effect to account for multiple plots being measured within a fire was not significant in the model and therefore not included. We fit separate models for the hardwood and dry conifer forest types. In addition to TSF, we included TSF^2 to assess whether the rate of change of the fuel loads was constant over time, and the categorical variable fire severity – low, moderate, high. To test if post-fire fuel load trajectories varied with fire severity, we included two-way interactions between fire severity and TSF and TSF^2, which then allowed estimating separate trajectories for each severity class. The full model (TSF, TSF^2, fire severity, two-way interactions) was initially fit for each response variable. Starting with the interactions, terms that were not statistically significant at the 0.05 level were dropped until only significant terms remained in the model. Terms that were part of a significant interaction but not significant themselves were still kept in the model. A robust estimator of the covariance was used to take into account the heteroscedasticity in the model (option 'empirical' in SAS PROC GLIMMIX) following Santos Silva and Tenreyro (2006). The models are described in more detail in the Supplementary material.

Results

For the dry conifer group there were 48, 41 and 20 plots that burned at low, moderate and high severity respectively. For the hardwood group, 35, 21 and 26 plots burned at low, moderate and high severity respectively. The average proportion of trees with green crowns (>30% of the crown unburned) per plot was 91.8, 47.4 and 2.3% for the low-, moderate- and high-severity fires respectively. The average unburned crown ratio for trees in plots that burned with low, moderate and high severity was 82.1, 39.2 and 2.0% respectively. These results suggest that the classification successfully separated plots according to fire severity, as measured by the effect of fire in tree crowns.

The distributions of the fuel loads were right-skewed, suggesting many light fuel loads and few heavy fuel loads. This can be seen by differences between sample mean and median fuel loads (Table 1). The differences between mean and median fuel loads were especially pronounced for 1000-h fuels and duff, suggesting that distributions for these two fuel types are highly skewed, with many plots with relatively low loads and a few

Table 1. Mean and median fuel loads (Mg ha^{-1}) 1 year post-fire by forest type and fire severity
Both mean and median fuel loads are presented for low, moderate and high fire severity for hardwood and dry-conifer stands because of the skewed distributions. Standard deviations of the means are provided in parentheses

Forest type	Fire severity	Sample size		Fuel loads one year post-fire (Mg ha^{-1}) Duff	Litter	1-h fuels	10-h fuels	100-h fuels	1000-h fuels
Hardwoods and dry conifers combined	Low	83	Mean (s.d.)	3.06 (5.08)	5.93 (4.32)	0.14 (0.15)	0.82 (0.72)	1.87 (2.02)	10.81 (18.02)
			Median	1.31	4.77	0.09	0.64	1.19	2.38
	Moderate	62	Mean (s.d.)	1.91 (3.15)	5.18 (4.71)	0.10 (0.10)	0.63 (0.59)	1.37 (1.45)	7.44 (10.88)
			Median	0.44	3.41	0.06	0.48	1.01	1.76
	High	46	Mean (s.d.)	0.19 (0.52)	1.98 (2.01)	0.06 (0.08)	0.35 (0.35)	0.54 (0.96)	3.55 (10.16)
			Median	0	1.25	0.3	0.25	0	0
Hardwoods	Low	35	Mean (s.d.)	3.46 (4.29)	8.51 (7.06)	0.212 (0.198)	0.75 (0.81)	1.51 (2.00)	6.93 (13.57)
			Median	2.19	6.37	0.154	0.59	0.96	2.05
	Moderate	21	Mean (s.d.)	0.52 (0.87)	6.03 (5.75)	0.161 (0.125)	0.74 (0.67)	1.34 (1.66)	5.25 (8.90)
			Median	0	3.39	0.132	0.59	0.96	1.55
	High	26	Mean (s.d.)	0.16 (0.42)	1.23 (1.54)	0.058 (0.081)	0.24 (0.28)	0.27 (0.53)	1.16 (2.53)
			Median	0	0.91	0.022	0.14	0	0
Dry conifers	Low	48	Mean (s.d.)	2.76 (5.62)	4.68 (2.42)	0.081 (0.062)	0.87 (0.65)	2.13 (2.00)	13.63 (20.34)
			Median	0.73	4.42	0.063	0.69	1.55	4.14
	Moderate	41	Mean (s.d.)	2.62 (3.64)	4.75 (4.10)	0.071 (0.070)	0.58 (0.55)	1.39 (1.35)	8.56 (11.71)
			Median	0.59	3.41	0.061	0.44	1.15	3.27
	High	20	Mean (s.d.)	0.23 (0.64)	2.96 (2.16)	0.067 (0.075)	0.49 (0.38)	0.89 (1.25)	6.65 (14.76)
			Median	0	2.50	0.035	0.34	0.42	1.12

with very high loads. A fairly large number of plots did not have any amount of several of the fuel types, which resulted in median fuel amounts of 0 for some forest type and severity class combinations (Table 1). This was especially true for plots that burned with high severity.

Duff and litter

One year after fire, duff loads were low in plots that burned with high severity, and much higher and quite variable in low- and moderate-severity plots (Table 1, Fig. 2*a–c*). On average, 0.19 Mg ha^{-1} of duff were observed 1 year post-fire in severely burned stands. In contrast, average duff loads were 10 and 16 times greater in stands that burned in moderate- and low-severity fires respectively (Table 1).

There were no statistically significant changes in duff fuel mass over the first 9 years after fire in any of the three severity classes in either of the two forest type groups ($P > 0.22$, Table 2), indicating that the amount of residual duff present immediately post-fire does not change within the first 9 years post-fire. Statistically significant differences in duff were found among severity classes ($P < 0.006$). The highest mean duff loads were estimated for low-severity stands, followed by moderate- and high-severity stands (Table 2).

Litter loads 1 year after fire followed a similar pattern than that of duff loads: with 1.98 Mg ha^{-1} they were lowest in areas burned under high severity and much higher in areas burned under moderate and low severity (2.6 and 3.2 times respectively, Table 1). For litter in dry conifer stands, there was no evidence of an effect of fire severity in the post-fire temporal dynamics. However, both the linear and quadratic terms of TSF were significant in the model (Table 2), suggesting a non-constant rate of increase of the litter loads. Litter loads peaked at ~6 years post-fire, regardless of fire severity (Fig. 2*d–f*). In hardwood stands, the relationship between the load and time was linear (in the log scale), indicating a constant rate of increase of the litter load. We detected a significant interaction between TSF and fire severity classes for hardwoods ($P < 0.0001$ Fig. 2*d–f*). Litter loads in low-severity stands increased by 7% per year up to 9 years post-fire, while litter loads in high-severity stands increased at a higher annual rate of change by 37% per year, albeit from a very low initial load (1.16 Mg ha^{-1}) (Fig. 2*d*, *f*). For moderate-severity stands the litter loads in hardwood stands did not change significantly over time ($P = 0.69$; Fig. 2*d*).

Fine woody fuels

One year post-fire, the 1-h fuel load in stands that burned in high-severity fires was 0.062 Mg ha^{-1}, and 1.7 and 2.2 times greater in areas burned under moderate and low severity (Table 1). For 10- and 100-h fuels, the loads in stands that burned in high-severity fires were 0.35 and 0.54 Mg ha^{-1} respectively (Table 1). For the 10-h fuels, loads were 1.8 and 2.3, and for the 100-h fuels, 2.5 and 3.5, times greater for stands that burned in moderate- and low-severity fires respectively. There was not a clear pattern in the difference between hardwoods and dry conifer forest types (Table 1, Fig. 3).

In hardwood stands, the quadratic TSF term was not significant in the model. Thus, there was no evidence of a curvature in the temporal trend of 1-, 10- and 100-h fuel loads, nor was there any evidence that the rate of fine fuel accumulation differed among fire severity classes. The mean 100-h fuel load increased at a constant rate of 24% annually across all fire severity classes ($P < 0.0001$; Fig. 3*g–i*; Table 2). The 10- and 1-h fuels in

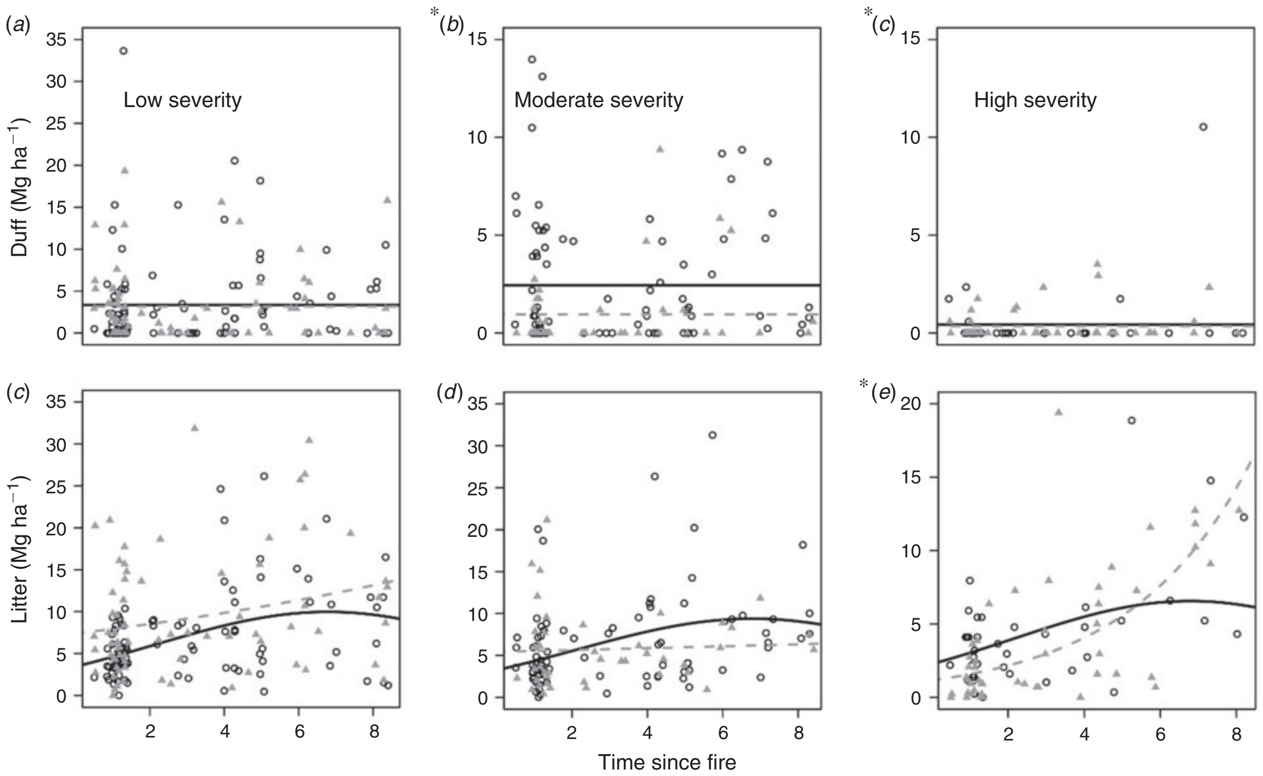

Fig. 2. Observed duff and litter loads and estimated trajectories within 9 years post-fire by fire severity and forest type. Open, black circles are observed fuel loads in dry conifer stands, while solid, grey triangles are observed fuel loads in hardwood stands. Solid, black lines describe the estimated post-fire trends in dry conifer stands, while dashed, grey lines describe the estimated post-fire trends in hardwood stands. Note that the scale of the ordinate may differ among plots. Asterisks above the *y*-axis indicate that the *y*-axis range of the panel differs from the *y*-axis range on the far left panel in the row.

hardwood stands increased at an annual rate of 15 and 18% respectively across all three fire severity classes (Table 2).

In dry-conifer stands and for 100-h fuels, the temporal rate of increase was constant, but there was a significant interaction between fire severity and TSF ($P < 0.0001$), which resulted in an estimated 6, 20 and 43% change per year for stands that burned with low, moderate and high severity respectively (Fig. 3*g–i*, Table 2). For the10-h and 1-h fuels, the post-fire trajectories had a quadratic trend ($P < 0.003$), but there was no difference across fire severity classes ($P > 0.13$). The maximum 10- and 1-h fuel loads were reached ~6 years post-fire (Fig. 3*a–f*).

1000-h fuels

One year after fire, average 1000-h loads were 3.55 Mg ha^{-1} in severely burned stands, and 2.1 and 3.0 times greater in stands that burned in moderate- and high-severity fires respectively. The rate of increase of the 1000-h fuels was constant, but differed among severity classes for both the hardwood and dry conifer stands ($P < 0.038$ for the interaction term). In hardwood stands, the estimated annual rate of increase was 6, 31 and 48%, and 6, 18 and 32% in dry conifer stands burned at low, moderate and high severity respectively (Table 2, Fig. 4). The post-fire trajectories in hardwood and softwood stands that burned at low and moderate severity were quite similar (Fig. 4 *a, b*), whereas coarse woody fuels in high-severity softwood stands increased at a higher rate than in hardwood stands (Fig. 4*c*), because of much lower 1000-h fuel loads in hardwood stands 1 year post-fire across all severity classes (dry conifer average, 10.4 Mg ha^{-1} (s.d., 16.7); hardwood, 4.7 Mg ha^{-1} (s.d., 10.3); Table 1).

Discussion

The objectives of our study were to describe post-fire surface and ground fuel dynamics in dry conifer and hardwood forests of California in areas that burned in low, moderate, and high severity fires. Our study is a longitudinal study that follows a spatially balanced sample across the entire range of dry conifer and hardwood forests in California. From this spatially balanced sample, we selected plots that had been burned between 2002 and 2013 and had been remeasured post-fire at least twice. Therefore, the sample covers the range of variability present in such a large population, with the different conditions represented in approximately the same proportion as they occur in the population. This design allows inference to the entire sampled population (Ramsey and Schafer 2013), so that the patterns described in this study estimate the mean post-fire trends for stands in the broad forest types included in the study. In contrast, most studies focus on a limited set of conditions (e.g. Passovoy and Fulé 2006; Dunn and Bailey 2015), restricting the range of variables such as fire severity, stand age, species or cover, in an effort to limit the variability and thus increase statistical power (e.g. Rusticus and Lovato 2014). However, there is a trade-off between variability and generality – as the range of conditions

Table 2. Parameter estimates with standard errors in parentheses and significance codes of time since fire by fuel type, fire severity and forest type

Estimated duff loads (Mg ha^{-1}) based on parameter estimates with 95% confidence intervals (CI) are provided in brackets underneath the parameter estimates. The models are log-linear models, so that for each fuel type, fire severity, and forest type, the fuel load is estimated as $\exp(\beta_0 + \beta_1 TSF + \beta_2 TSF^2)$, where *TSF* is the time since fire in years. Probabilities are significant at: '***', $P < 0.001$; '**', $P < 0.01$; '*', $P < 0.05$, '^{ns}', non-significant at $P = 0.05$

Fuel type	Fire severity	Forest type	Intercept (β_0)	Linear coefficient (β_1)	Quadratic coefficient (β_2)
Duff	Low	Dry conifers	1.20*** (0.16) [3.34 (CI: 2.42, 4.59)]		
		Hardwoods	1.16*** (0.16) [3.18 (CI: 2.31, 4.38)]		
	Moderate	Dry conifers	0.89*** (0.14) [2.44 (CI: 1.84, 3.25)]		
		Hardwoods	−0.05^{ns} (0.31) [0.96 (CI: 0.51, 1.77)]		
	High	Dry conifers	−0.83^{ns} (0.63) [0.35 (CI: 0.19, 0.66)]		
		Hardwoods	−1.04** (0.32) [0.44 (CI: 0.12, 1.52)]		
Litter	Low	Dry conifers	1.24*** (0.14)	0.31** (0.09)	−0.02* (0.01)
		Hardwoods	2.00*** (0.14)	0.07* (0.03)	
	Moderate	Dry conifers	1.18*** (0.16)	0.31** (0.09)	−0.02* (0.01)
		Hardwoods	1.69*** (0.23)	0.02^{ns} (0.05)	
	High	Dry conifers	0.82*** (0.18)	0.31** (0.09)	−0.02* (0.01)
		Hardwoods	0.14^{ns} (0.25)	0.32*** (0.04)	
1-h	Low	Dry conifers	−3.06*** (0.14)	0.53*** (0.12)	−0.05** (0.02)
		Hardwoods	−1.78*** (0.17)	0.17*** (0.03)	
	Moderate	Dry conifers	−3.06*** (0.14)	0.53*** (0.12)	−0.05** (0.02)
		Hardwoods	−2.01*** (0.16)	0.17*** (0.03)	
	High	Dry conifers	−3.06*** (0.14)	0.53*** (0.12)	−0.05** (0.02)
		Hardwoods	−3.02*** (0.21)	0.17*** (0.03)	
10-h	Low	Dry conifers	−0.77*** (0.15)	0.36*** (0.09)	−0.03** (0.01)
		Hardwoods	−0.35* (0.16)	0.14** (0.05)	
	Moderate	Dry conifers	−0.77*** (0.15)	0.36*** (0.09)	−0.03** (0.01)
		Hardwoods	−0.71** (0.22)	0.14** (0.05)	
	High	Dry conifers	−0.77*** (0.15)	0.36*** (0.09)	−0.03** (0.01)
		Hardwoods	−1.54*** (0.24)	0.14** (0.05)	
100-h	Low	Dry conifers	0.73*** (0.14)	0.06* (0.03)	
		Hardwoods	−0.19^{ns} (0.20)	0.22*** (0.05)	
	Moderate	Dry conifers	0.13^{ns} (0.16)	0.18*** (0.04)	
		Hardwoods	−0.19^{ns} (0.20)	0.22*** (0.05)	
	High	Dry conifers	−0.53^{ns} (0.28)	0.36*** (0.06)	
		Hardwoods	−0.19^{ns} (0.20)	0.22*** (0.05)	
1000-h	Low	Dry conifers	2.57*** (0.22)	0.06^{ns} (0.05)	
		Hardwoods	2.15*** (0.45)	0.06^{ns} (0.07)	
	Moderate	Dry conifers	1.86*** (0.25)	0.17** (0.06)	
		Hardwoods	1.16** (0.40)	0.27*** (0.07)	
	High	Dry conifers	1.40** (0.47)	0.28*** (0.08)	
		Hardwoods	−0.49^{ns} (0.50)	0.40*** (0.10)	

from which the sample is drawn is limited, so is the inferential scope of the conclusions (Baguley 2012). Thus, there is a gradient between very detailed studies that aim to describe processes in a very controlled set of conditions, and studies such as this one, which aim to include the range of variability to ensure that the results apply to a wide population.

However, even though there is great variability among plots sampled from a large population in our study, changes in fuel loads were estimated from cohorts of burned plots that were measured at least twice – once the year after fire and then again within the 9 years following fire. Because the same plots are measured over time, this type of longitudinal study allows greater power in estimating the time effects than cross-sectional studies such as chronosequences, as the effects of plot-level variables that do not change over time are in effect removed from the analysis (Wooldridge 2002). The effect of measuring the same plot over time is similar to the effect of blocking in experimental design – precision is increased because the variability of measurements from the same plot is less than that from different plots (Kuehl 2000).

Litter and duff dynamics

The majority of litter, duff and fine fuels have been reported to combust in high-severity fires (Campbell *et al.* 2007), and rapid accumulation of litter, 1- and 10-h fuels within the first few years post-fire is expected if tree crowns have not been completely consumed by fire (Dunn and Bailey 2015). However, the combustion and accumulation of litter, duff and fine fuels is much more variable in low- and moderate-severity fires (Campbell *et al.* 2007) and not as well documented as most research focuses

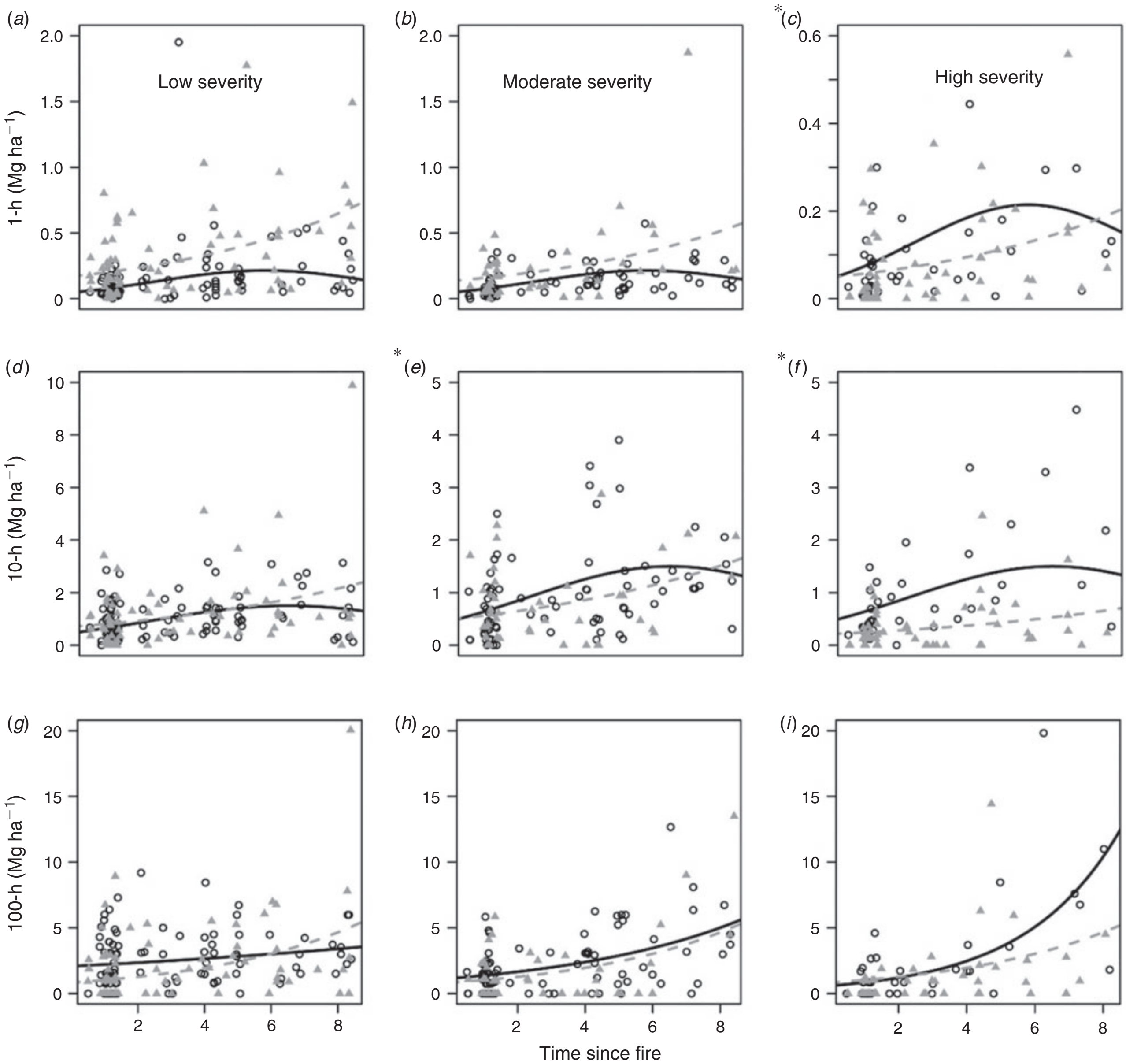

Fig. 3. Observed 1-, 10- and 100-h fuel loads and estimated trajectories within 9 years post-fire by fire severity and forest type. Observed values in dry conifer (open, black circles) and hardwood (solid, grey triangles) stands with estimated trajectories for dry conifers (solid, black lines) and hardwoods (dashed, grey lines). Note that the scale of the ordinate may differ among plots. Asterisks above the *y*-axis indicate that the *y*-axis range of the panel differs from the *y*-axis range on the far left panel in the row.

on high-severity fires (e.g. Kashian *et al.* 2013; Dunn and Bailey 2015) or prescribed burns (e.g. Keifer *et al.* 2006), which may have different behaviour and fire effects than wildfires.

The lack of change in duff loads in the first 9 years post-fire is most likely due to the lack of litter available for decomposition and the long time it takes for litter to decompose and replace fire losses in the dry, Mediterranean climate of California. In ponderosa pine forests of central Oregon, for example, litter decomposition rates were low, and were further reduced following prescribed burns, an effect that was still apparent in sites that had burned 12 years previously (Monleon and Cromack 1996). This lag in duff accumulation following fire has also been observed in a chronosequence of dry mixed conifer forests within the Eastern Cascades of Oregon, USA (Dunn and Bailey 2015). The long time it takes for duff to recover following a wildfire may influence future fire behaviour, but can also affect stand productivity and nutrient availability (Monleon *et al.* 1997).

Duff consumption during fires can be high and often increases with fire severity with consumption rates of 11–78 and 76–90% in low- and moderate-severity prescribed burns respectively (Vaillant *et al.* 2009). In a southern Oregon mixed-severity wildfire, duff consumption varied from 44% in unburned and low-severity stands to 99% in stands that burned with high severity, and was 51 and 54% in stands that burned in

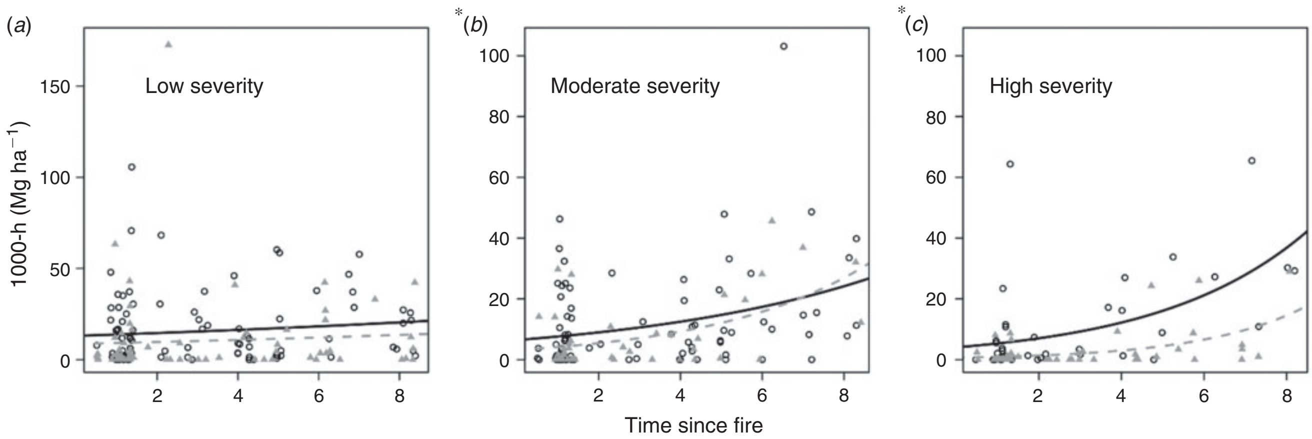

Fig. 4. Observed 1000-h fuel loads and estimated trajectories within 9 years post-fire by fire severity and forest type. Observed values in dry conifer (open, black circles) and hardwood (solid, grey triangles) stands with estimated trajectories for dry conifers (solid, black lines) and hardwoods (dashed, grey lines). Note that the scale of the ordinate may differ among plots. Asterisks above the *y*-axis indicate that the *y*-axis range of the panel differs from the *y*-axis range on the far left panel in the row.

moderate- and low-severity fires respectively (Campbell *et al.* 2007). Although the large differences in duff loads among fire severity classes in our study may be due in part to lower pre-fire loads in those stands, the nearly complete combustion of the duff layer in high-severity fires suggest that the changes are mainly due to differences in consumption. In fact, 35, 47 and 85% of the plots in low, moderate and high severities respectively did not have any duff one year after fire, suggesting that all the duff – and therefore the litter – burned in the fire. Most of these plots remained without any duff for the duration of the study.

Litter dries quickly and is easily ignited (Keane 2015). Litter consumption in fires in temperate forests was estimated to be 81% (van Leeuwen *et al.* 2014). Litter was completely combusted in a high-severity wildfire, and between 76 and 75% combusted in stands that burned in moderate- and low-severity respectively (Campbell *et al.* 2007). Litter consumption rates in low-severity, prescribed burns across California were between 32 to 84% (Vaillant *et al.* 2009). In our study, litter loads 1 year after fire were much lower in stands that burned under high severity than in those that burned under moderate and low severity, but the differences were not as pronounced as those of duff, and the litter loads were relatively high. This suggests that a large amount of the litter observed 1 year after fire may result from litterfall and accumulation amounts in the first year post-fire. As most of the litter is burned in high-severity stands (Campbell *et al.* 2007), our results suggest that, in areas that burned under high severity, dry conifer stands have a higher litter input in the first year after fire than hardwood stands, as expected given that the first pulse of litter input mainly comes from conifer needles from scorched crowns (Dunn and Bailey 2015).

Beyond that first year, litter is mainly created by re-establishing vegetation that annually contributes leaf litter and foliage (e.g. shrubs, re-sprouting hardwoods) or by scorched leaves and sloughing bark from fire-killed trees or trees for which crown foliage did not completely combust (Dunn and Bailey 2015; Stalling *et al.* 2017). We observed a similar, non-linear post-fire trend in litter for dry conifer stands across all fire severities with a peak at ~6 years post-fire, indicating that the first pulse of litter input from scorched needles and re-establishing vegetation may end around that time The continued increase in litter loads over the first 9 years post-fire observed in high-severity hardwood stands may be due to the establishment of shrubs and re-sprouting of hardwoods, and the high rate could be explained by the relatively low initial levels of litter compared with that of conifer stands. The accumulation rates for hardwood stands that burned in moderate- and low-severity fires were much lower, probably reflecting a less vigorous re-sprout response when many of the existing stems were not killed by fire. This would be consistent with previous findings that have shown California black oak (*Q. kelloggii*) sprout clump density (count and size) to be positively associated with increasing fire severity (Crotteau *et al.* 2013, 2015). Elliott *et al.* (1999) also reported prolific sprouting of evergreen shrubs in high intensity areas of the prescribed burns that they studied.

Fine woody fuel dynamics

Fine woody fuels are important in fire behaviour. Not only do they facilitate fire spread, but they also contribute to fire intensity (Keane 2015). Vaillant *et al.* (2009) reported respective ranges of 44–79, 20–77 and 11–78% for 1-, 10- and 100-h fuel consumption rates across prescribed, low-severity burns in California, whereas van Leeuwen *et al.* (2014) reported consumption rates of 87, 79 and 73% for those three fine fuel classes.

In dry conifer stands, the observed peak in 1- and 10-h fuels 6 years post-fire is due to 1- and 10-h fuel loads mainly accumulating from the deposition of twigs and branches from scorched tree crowns (Keyser *et al.* 2009) and re-establishing vegetation (e.g. shrubs). As we do not have estimates of fine fuel loads immediately after fire, but 1 year post-fire, the observed fine fuel loads at this time may represent depositions from fine twigs from scorched tree crowns that accumulated within the first year after fire rather than those resulting from the direct effect of fire. It is difficult to tell if the lower loads in 1-h fuels in low-severity stands are due to pre-fire 1-h fuel loads, the amount burned during fire or the amount deposited since fire.

Snags are a dominant source of fine and coarse fuels, and input into fine fuel classes can be expected for 10 to 20 years post-fire as snags fragment and break down (Dunn and Bailey 2012; Roccaforte *et al.* 2012). Previous studies show that 10- and 100-h fuels reach their maxima between 14 and 18 years after high-severity fire (Dunn and Bailey 2015) and between ~19 and 20 years (Hall *et al.* 2006), whereas 1-h fuels reached their peaks ~10 years after fire (Hall *et al.* 2006). Our results indicate that the maximum is reached much earlier, at ~6 years, similar to findings by Roccaforte *et al.* (2012) who suggest peaks in 10- and 100-h fuels as early as 6 years as well. These previous studies were based on chronosequences spanning 24 years (Dunn and Bailey 2015), 160 years (Hall *et al.* 2006) and 18 years (Roccaforte *et al.* 2012) after fire. The problem with estimating a maximum with a long chronosequence may be compounded when fitting a global model, such as a quadratic, over the entire span of the study, which may be influenced by the values at a time far from the immediate aftermath of the fire.

In dry conifer stands, 100-h fuels behave similarly to 1000-h fuels. A 100-h fuel accumulation depends on breakage of large branches from scorched tree crowns and snags. The reported lower 100-h fuel accumulation rate in hardwood stands compared with dry conifer stands in high-severity fires can be attributed to the difference in snag size between those two forest type groups.

As fine fuel accumulation is mainly dependent on re-establishing vegetation and deposition of twigs and branches from tree crowns (Dunn and Bailey 2015), the continued increases in all fine fuel classes over the observed 9 years post-fire suggest that the observed fine fuel loads in hardwood stands may be likely due to re-establishing vegetation that is still increasing in cover over this fairly short time period. This is in contrast to dry conifer stands where the main source of fine fuel input seems to come from scorched crowns (Dunn and Bailey 2015).

Contrary to our findings, Ritchie *et al.* (2013) found that the quadratic TSF term was significant in their model and that 100-h fuels peaked 6 years after fire. Ritchie *et al.* (2013) further found that 1–10-h fuels were decreasing over the 8-year post-fire period of their study. These contradictory results may be explained by the fact that their study examined post-fire logging effects of five basal area retention levels whereas our study sites were not exposed to any post-fire treatments. The difference in results between Ritchie *et al.*'s (2013) study and our study suggests that the surface fuel load accumulation in salvage logged stands may differ from those observed in un-salvaged stands as presented by our study. Differences in surface fuel load accumulation may result in different potential effects in future fires (e.g. different reburn severity).

1000-h fuels dynamics

Coarse woody fuels contribute little to fire spread, but may contribute to fire intensity (Keane 2015). The consumption rates of 1000-h fuels across fire severity classes are highly variable; ranging from very low reported consumption to fairly high consumption rates (Raymond and Peterson 2005; Campbell *et al.* 2007; van Leeuwen *et al.* 2014). The post-fire 1000-h fuel accumulation as well as fine fuel accumulation has an inverse, non-linear relationship with snag fall and fragmentation (Dunn and Bailey 2015). The combustion of 1000-h fuels is highly dependent on their decay state and moisture content (Hyde *et al.* 2011). A 1000-h fuel consumption of sound *v.* rotten logs for temperate forests has been reported as 38 (s.d., 42) and 96% (s.d., 5.4) (van Leeuwen *et al.* 2014) indicating that rotten 1000-h fuels tend to be consumed almost completely, whereas sound 1000-h fuel consumption is much lower and highly variable.

The observed increase in annual rate of 1000-h fuel accumulation from low- to high-severity fires can be explained by the inverse relationship between 1000-h fuel load and snag fuel load, as snags are the only source of 1000-h fuels (Passovoy and Fulé 2006; Ritchie *et al.* 2013). The abundance of snag fuels 1 year post-fire increases for low- to high-severity fires (Eskelson *et al.* 2016). Therefore, there will be increasing rates of 1000-h fuel inputs along fire severity gradients from low- to high-severity fires, as snags break and fall down. In high-severity stands 1000-h fuels increase more rapidly in dry conifer stands than in hardwood stands, likely because the greater size of snags in conifer forests. Thus, 1000-h post-fire fuel dynamics seem to be greatly influenced by pre-fire stand conditions, as reported by Dunn and Bailey (2015).

Snag fall rates vary by tree species and the environmental factors that caused tree mortality (Raphael and Morrison 1987; Morrison and Raphael 1993). For example in ponderosa pine dominated forests, the majority of snags have been found to convert to surface fuels very rapidly, mostly in less than 10 years (Passovoy and Fulé 2006; Ritchie *et al.* 2013). Therefore, the time frame of our study may have captured most of the surface fuel input in the dry conifer stands, which would align with findings by Passovoy and Fulé (2006) who did not detect any major changes in 1000-h fuels after 8 to 9 years post-fire.

Management implications

As fires in western North America increase in severity, duration, frequency, intensity or size (Stephens 2005; Westerling *et al.* 2006; Miller *et al.* 2009; Dennison *et al.* 2014), land managers will have to include previously burned areas into their landscape management plans, making decisions about post-fire fuel reduction treatments, reforestation and wildfire management to increase the benefit of existing and planned fuel treatments (Thompson *et al.* 2007; North *et al.* 2012, Stevens-Rumann *et al.* 2012). In order to make these decisions and develop restoration strategies, land managers need to be able to consider the possibility of reburns, which is affected by post-fire fuel variability, post-fire stand composition and structure and post-fire fuel dynamics (Hudec and Peterson 2012; Dunn and Bailey 2015; Coppoletta *et al.* 2016). Surface fuels receive the most attention in fire management because the surface fuel layer is most commonly used to predict fire behaviour (Keane 2015). Therefore, our results on post-fire ground and surface fuel dynamics in dry conifer and hardwood stands of California will help fire managers assess the probability of reburns and potential surface fire behaviour across fire severity and their evolution over time (Hudec and Peterson 2012).

Our results are representative of the fuel dynamics for the forest types and fire severity classes included in this study and can be generalised across dry conifer and hardwood forests across California. Fire and fuels management need to consider the natural heterogeneity that occurs across forest types and severity classes (Hudec and Peterson 2012), an objective that is met if the results are based on a random sample of sites across the population of interest, in contrast with studies based on selected sites along a chronosequence of a single forest type and fire severity class (Passovoy and Fulé 2006; Kashian *et al.* 2013; Dunn and Bailey 2015). The major limitation of a longitudinal study, such as this one, is the limited timeframe of available post-fire remeasurements, restricting the conclusions to the first 9 years post-fire. Similar to Dunn and Bailey (2015), the length of our post-fire temporal period limits our ability to make any conclusions about ground- and surface-fuel loads after snags have fallen and stopped dominating fuel inputs. We were also unable to identify the time at which duff loads start to increase post-fire. The temporal scope of the study will increase as new measurements are taken in the course of the regular inventory cycle.

Another limitation of our study is the lack of pre-fire data, which limited the possible interpretation of differences in fuel loads observed 1 year post-fire. The initial (i.e. 1 year post-fire) amounts of surface- and ground-fuel loads may in some cases differ by fire severity class (Hudec and Peterson 2012). These amounts may be affected by fuel consumption, but may also be influenced by differences in pre-fire loads (Eskelson *et al.* 2016). Therefore, comparing differences in fuel loads 1 year post-fire is difficult without having information on pre-fire loads. However, this is a common limitation when studying wildfires, as it is very rare that a pre-fire sample is available.

Conflicts of interest

The authors declare that they have no conflict of interest.

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