

INTRODUCTION

Tree mortality associated with drought and concurrent bark beetle outbreaks in the Western United States is expected to increase under warmer and drier climate regimes (Kolb and others 2016). The combined impacts of drought and insect outbreaks on tree mortality have important implications for predicting forest dynamics under climate change, yet forecasting tree mortality is one of the most uncertain processes in dynamic vegetation models. To improve predictions of how climate change will alter forest dynamics, models must be informed by a better understanding of how tree-level factors influence probability of mortality when multiple disturbance agents co-occur. Some tree-level early warning signals for drought-related mortality have been identified, but few patterns have emerged across tree species, and most studies do not consider drought interactions with insects (Cailleret and others 2019, Camarero and others 2015). Similarly, despite recent advances in the understanding of physiological mechanisms underlying drought-related tree mortality (Adams and others 2017, Sevanto and others 2014), much remains unclear. This is especially true when insects are involved (Anderegg and others 2015), as they can affect tree carbon and water balance, both important links to drought-related tree mortality (McDowell and others 2008). A handful of studies have examined why some trees die and others survive in the face of drought combined with insect outbreaks (Csank and others 2016, McDowell and others 2010), but it is still unclear

whether generalizable tree-level mortality risk factors or early warning signals exist in the face of these interacting disturbances.

Tree die-off events can cause rapid, drastic changes to forest structure and fuel loads, with cascading effects on potential fire behavior and severity (Hicke and others 2012). Dead trees in the red stage, when needles remain in the crowns, alter canopy fuel availability and may have higher ignition potential and burn more intensely. However, the red stage is transient, and needles fall to the ground within a few years of tree death. Surface fuel loading may then increase as foliage, branches, and trees decompose and fall to the ground. Previous research on changes to fuels in beetle-impacted forests suggests responses may vary by forest type, with studies in the Northern Rockies showing little change to surface fuel loads (Donato and others 2013, Stalling and others 2017) and others from the Southeastern and Southwestern United States reporting increased surface fuel loading and more rapid snag fall (Hoffman and others 2012, Xie and others 2020). Likewise, research into how high levels of tree mortality influence fire behavior and fire severity is also mixed and dependent on fire weather, outbreak conditions, and time since attack (Hood 2019, Sieg and others 2017).

The 2012–2016 drought and associated bark beetle outbreaks in California resulted in extensive tree mortality and provided a unique opportunity to examine questions of why some trees die while others survive these co-occurring

Chapter 11.

Tree Mortality and Fuel Changes due to Extreme Drought and Concurrent Bark Beetle Outbreaks in California

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disturbances and to increase understanding of how subsequent bark beetle-caused tree mortality alters fuel profiles and forest flammability over time. Our study objectives were to: (1) identify factors relating to conifer tree mortality and monitor changes in vegetation after mortality and (2) determine temporal changes in fuel loading and hazard due to bark beetle- and drought-associated tree mortality.

METHODS

Sampling areas were established in 2016 in two separate geographical areas affected by the 2012–2016 California drought and concurrent beetle outbreaks. We installed plots in areas of low and high recent tree mortality on the Los Padres National Forest (LPNF; 27 plots) in singleleaf pinyon (*Pinus monophylla* Torr. & Frém) woodlands and on the Sierra National Forest (SNF; 49 plots) in areas dominated by ponderosa pine (*P. ponderosa* Lawson & C. Lawson), white fir (*Abies concolor* [Gord. & Glend.] Lindl. ex Hildebr.), and incense cedar (*Calocedrus decurrens* [Torr.] Florin). Plots in the SNF sampling area utilized an existing plot network within areas of specific forest type and mortality level, excluding areas of recent forest management (Pile and others 2019). We selected a subset of 50 plots in the network in 2016 based on mortality level determined from 2015 Insect and Disease Survey (IDS) data and species composition. We selected approximately half of the plots based on low mortality (<5 percent), and the other half based on relatively high mortality (>40 percent). Plots in the LPNF sampling area were selected based on singleleaf pinyon presence and mortality

levels similarly determined from IDS data. Individual plot selection was random within areas of high and low mortality with singleleaf pinyon dominance. In California, singleleaf pinyon is a primary host to pinyon ips (*Ips confusus* LeConte); ponderosa pine is the primary host to western pine beetle (*Dendroctonus brevicomis* LeConte); and white fir is the primary host to fir engraver (*Scolytus ventralis* LeConte) (Fettig 2016). Cedar bark beetles (*Phloeosinus* spp.) are secondary mortality agents on incense cedar (Fettig 2016).

We installed 0.04-ha macroplots in 2016 and monitored changes in vegetation and fuels annually through 2019. At each macroplot, we recorded tree species, diameter at breast height (1.37 m aboveground, d.b.h.; groundline diameter [GLD] for singleleaf pinyon), tree height, crown base height, and status of all trees ≥ 12.7 -cm d.b.h. Mortality agent was assessed for dead trees (i.e., no green needles remaining) based on presence and pattern of insect-specific galleries. We estimated the percentage of faded, dead needles remaining in the crown each year for all dead trees. We tallied saplings (≥ 1.37 m and <10-cm d.b.h.) by species, diameter class, and status and seedlings (<1.37 m) by species, height class, and status on a nested 0.01-ha microplot inside the macroplot. We collected two 4.5-mm cores near groundline on two living and two dead trees for each available species at each plot to assess tree growth patterns and responses to drought. We estimated fine fuel (1–100 hour), shrub, and herbaceous loading using the Photoload method (Keane and Dickinson 2007)

at eight 1-m² quadrats within the macroplot. We measured litter and duff depth at one point in each quadrat. Coarse fuel loading (1,000 hour) and decay classes were estimated by measuring the small and large end diameter and length of each log ≥ 7.6 cm on the macroplot.

All data were entered into FFI (FEAT/FIREMON Integrated) (Lutes and others 2009) for surface fuel and stand density calculations. We assessed all tree species for needle retention and snag fall but only singleleaf pinyon, ponderosa pine, white fir, and incense cedar for the tree ring-based analyses of tree mortality due to limited sample sizes (<14 cores per species) in the other tree species. For growth analyses and chronology building, cores were sanded and crossdated, and rings were measured to develop species-specific chronologies using standard dendrochronological techniques (Stokes and Smiley 1996). We selected a subset of six live and six dead trees per species for stable carbon isotope analysis based on tree dominance or codominance within the canopy. A subset of 10 live and 10 dead trees each for singleleaf pinyon and ponderosa pine was additionally randomly selected from the group of successfully dated trees to assess resin ducts (see Reed and Hood [2021] for full tree ring methods).

RESULTS

Tree Mortality and Risk Factors

Over the course of sampling (2016–2019), tree mortality remained fairly steady within LPNF plots (from 27 percent in 2016 to 30 percent in 2019) but increased within SNF plots

(from 43 percent to 54 percent), particularly for ponderosa pine (from 78 percent to 92 percent). At the end of the study, mean plot-level tree mortality for individual species ranged from 30 percent for singleleaf pinyon to 91 percent for ponderosa pine but varied considerably by low versus high mortality level. Mortality attributed to host-specific beetles was high for singleleaf pinyon (97 percent; pinyon ips), ponderosa pine (93 percent; western pine beetle), and white fir (91 percent; fir engraver), and all cored trees had evidence of successful attack by primary bark beetles. Mortality agents for incense cedar were unknown or attributed to suppression and drought.

Differences in radial growth between surviving and dying trees were apparent only for ponderosa pine and incense cedar. Ponderosa pine that survived tended to have lower basal area increment (BAI) than trees that died until around the late 1980s–early 1990s, a time period corresponding to a multiyear regional drought and bark beetle outbreaks, at which point growth differences between dying and surviving trees decreased. In contrast, BAI of incense cedar that survived tended to be higher than for trees that died throughout the study period, and particularly in recent decades. Across species, differences in carbon isotope discrimination ($\Delta^{13}\text{C}$) between live and dead trees were inconsistent. Ponderosa pine that survived had higher $\Delta^{13}\text{C}$ throughout most of the study period than trees that died. The other three species showed weak or inconsistent differences in $\Delta^{13}\text{C}$ between live and dead trees. Resin duct

analysis of pines showed that in the 5 years leading up to the 2012–2016 drought, some differences emerged. Surviving ponderosa pines had higher relative duct area and weakly greater duct density. Singleleaf pinyon that survived had a greater duct production (i.e., number of ducts) and weakly higher total duct area. Full results of tree mortality and risk factors are reported in Reed and Hood (2021).

Needle Retention

During the first year of sampling in 2016, there were 26 white fir, 21 sugar pine, 12 singleleaf pinyon, and 102 ponderosa pine that had recently died with high needle retention for which we could track needle loss annually. Needle loss was rapid and followed a similar pattern for all species (fig. 11.1). In the second year, 28 to 43 percent of needles remained, and by year 4, all trees had virtually no needles left.

Snag Longevity

Of the 1,402 trees (including nine species) recorded on the plots, we estimated 536 (38 percent) died from the 2012–2016 drought and successful beetle attacks. By the last year of sampling in 2019, 41 percent of the recently dead trees had broken, and 20 percent of these had broken below 1.37 m. Snag fall varied by species, with pines (all species) falling most quickly (table 11.1).

Fuel Loads

Fine woody surface fuel loads (1–100-hour fuel components) were generally low for both the LPNF and SNF sites (table 11.2). On the

LPNF, areas of high tree mortality increased in fine woody fuel more than the low tree mortality plots. On the SNF, there was not as clear of a pattern, with some fuel components increasing and others decreasing. The largest change occurred in the 1,000-hour fuels in the areas of high mortality. In just 4 years, 1,000-hour fuels in the high-mortality areas quadrupled from 0.38 kg m^{-2} to 1.50 kg m^{-2} on the LPNF and more than doubled on the SNF from 3.05 kg m^{-2} to 7.55 kg m^{-2} . Fuel loads of 1,000-hour logs

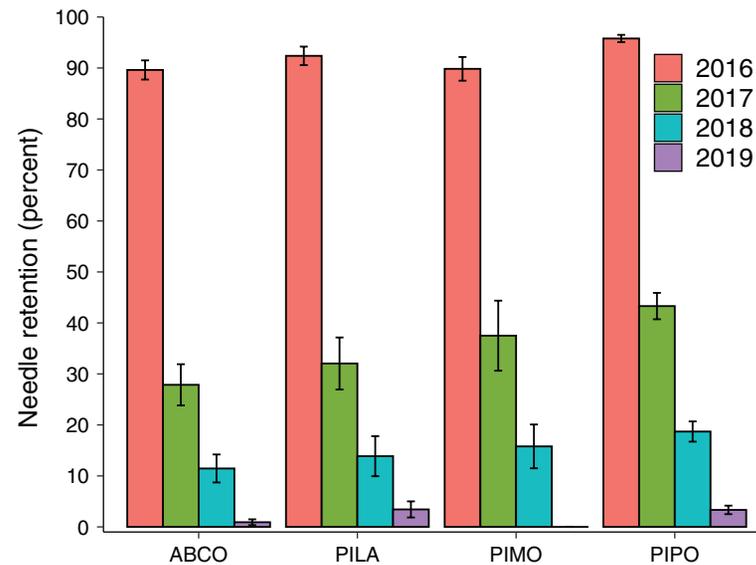


Figure 11.1—Needle retention (percent) of recently killed trees over time by species. ABCO = *Abies concolor* (white fir); PILA = *Pinus lambertiana* (sugar pine); PIMO = *Pinus monophylla* (singleleaf pinyon pine); PIPO = *Pinus ponderosa* (ponderosa pine).

Table 11.1—Number and percentage of new snags created during the drought and bark beetle outbreaks by species that were broken by end of the sampling in 2019

Snag condition		White fir	Red fir	Incense cedar	Lodgepole pine	Jeffrey pine	Sugar pine	Singleleaf pinyon	Ponderosa pine	Black oak	Total
		n = 91	n = 26	n = 55	n = 13	n = 2	n = 48	n = 120	n = 177	n = 4	536
Broken: any height	n	15	5	20	5	0	16	45	114	0	220
	%	16	19	36	38	0	33	38	64	0	41
Broken: below 1.37 m	n	2	0	12	4	0	9	31	47	0	105
	%	2	0	22	31	0	19	26	27	0	20

Table 11.2—Mortality and fuel loads on the Los Padres and Sierra National Forests by low- (<5 percent) and high- (>40 percent) mortality plots

	Low-mortality plots						High-mortality plots					
	2016		2019		Difference		2016		2019		Difference	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Los Padres National Forest												
Mortality (%)	8	3	13	5	5	4	61	5	62	4	1	1
1-hour (kg m ⁻²)	0.13	0.02	0.14	0.02	0.00	0.02	0.09	0.02	0.13	0.02	0.04	0.01
10-hour (kg m ⁻²)	0.18	0.05	0.15	0.03	-0.03	0.04	0.12	0.04	0.17	0.04	0.05	0.02
100-hour (kg m ⁻²)	0.06	0.03	0.08	0.03	0.02	0.01	0.02	0.00	0.10	0.02	0.08	0.02
1,000-hour (kg m ⁻²)	0.06	0.01	0.10	0.03	0.04	0.03	0.38	0.27	1.50	0.44	1.12	0.28
Sierra National Forest												
Mortality (%)	21	3	29	3	7	2	66	4	76	3	10	2
1-hour (kg m ⁻²)	0.09	0.02	0.10	0.02	0.01	0.01	0.06	0.01	0.09	0.01	0.02	0.01
10-hour (kg m ⁻²)	0.26	0.06	0.15	0.02	-0.11	0.05	0.16	0.03	0.17	0.02	0.01	0.03
100-hour (kg m ⁻²)	0.34	0.09	0.13	0.03	-0.21	0.09	0.34	0.10	0.20	0.04	-0.14	0.10
1,000-hour (kg m ⁻²)	2.65	0.62	3.43	0.64	0.78	0.24	3.05	0.62	7.55	0.95	4.51	0.89

SE = standard error

increased positively with tree mortality level (fig. 11.2). More detailed surface and canopy fuels and fire hazard analyses are ongoing.

DISCUSSION AND CONCLUSIONS

The predicted increase in the intensity and/or length of droughts, combined with changes in bark beetle dynamics associated with climate change (Kolb and others 2016), requires a better understanding of how severe disturbance events influence mortality at the tree level if we are to effectively predict future forest mortality. Risk factors associated with tree-level mortality differ

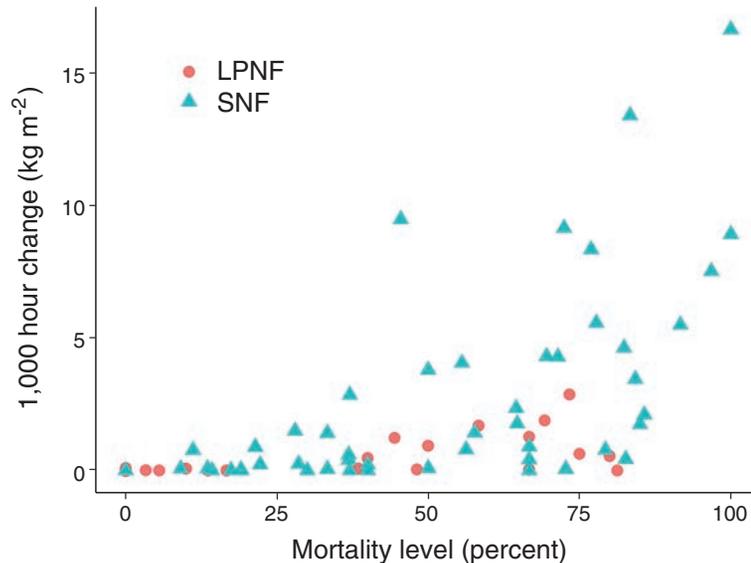


Figure 11.2—Change in 1,000-hr fuel loads from 2016 to 2019 by overstory (≥ 12.7 -cm d.b.h.) tree mortality level on the Los Padres National Forest (LPNF) and Sierra National Forest (SNF).

among tree species, and generalizable patterns become less clear when bark beetle outbreaks occur in concert with a particularly long, hot drought, likely a reflection of unique strategies for dealing with these disturbances. For ponderosa pine, both long-term differences in tree physiology and shorter term beetle-related selection and variability in defenses influence susceptibility to drought and beetle attacks. For singleleaf pinyon and white fir, however, long-term variability between trees that survived and those that died is less clear, and beetle dynamics may play a more prominent role in mortality patterns. Incense cedar, in contrast, for which we see limited evidence of beetle-related mortality, appears to be primarily influenced by long-term differences in growth rate and stand dominance, which likely impact resource availability and susceptibility to drought. It is also possible that cedar bark beetles may have contributed to incense cedar death (Stephenson and others 2019). Models that aim to predict tree-level mortality under future climate scenarios will need to account for extreme drought events and may need to consider multiple species-specific risk factors to more accurately predict tree death.

Failure of newly dead trees was high, especially for pine species and incense cedar, considering we only monitored trees for 4 years from 2016 to 2019. Our results are conservative, as our sampling does not include trees that died in the 2012–2016 drought but that fell before 2016. Landram and others (2002) conducted a large study of snag demography in California over 9 years. While our monitoring period

was much shorter and sample sizes lower than theirs, our dead ponderosa pine annual fall rates are consistent with that study at an annual 6.5 percent versus 6.9 percent. However, our fall rates are higher for lodgepole pine (7.8 versus 3.8) and incense cedar (4.3 versus 2.5) and lower for white fir (0.5 versus 3.7) than what Landram and others (2002) reported. The high snag fall raises concerns for people working, including fire fighters, and recreating in areas of high tree mortality.

The timing and degree of change to fuel profiles from bark beetle outbreaks varies greatly by forest type, with implications for subsequent wildfire behavior and severity (Hicke and others 2012, Jenkins and others 2012). Falling snags increased 1,000-hour fuel loading in the areas of high mortality, whereas we found fewer changes in fine woody surface fuels despite heavy needle loss. Given that trees dropped their needles within about 3 years of death and the high snag fall rates, we anticipate that any increase to potential crown fire behavior would be short-lived as canopy fuel loads will continue to decrease over time, but additional analysis is needed. High levels of tree mortality influence other aspects of subsequent wildland fire besides potential fire behavior, such as spotting potential, fire severity, fire suppression tactics, and firefighter safety (Fettig and others 2021, Hood 2019). The vast area impacted by the drought and outbreaks will likely influence wildfire hazard, with increased connectivity of fuels over thousands of acres. Stephens and others (2018) raised the possibility of mass fires

occurring in these forests over the next several decades that are much higher in intensity and have unpredictable fire behavior compared to individual wildfires. Ecological changes related to altered forest structure and composition from the drought and bark beetle outbreaks are also anticipated (Fettig and others 2019, Stephenson and others 2019, Young and others 2020).

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