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Fuels change quickly after California drought and bark beetle outbreaks with implications for potential fire behavior and emissions

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Abstract

Background An extreme drought from 2012–2016 and concurrent bark beetle outbreaks in California, USA resulted in widespread tree mortality. We followed changes in tree mortality, stand structure, and surface and canopy fuels over four years after the peak of mortality in Sierra mixed conifer and pinyon pine (*Pinus monophylla*) forests to examine patterns of mortality, needle retention after death, and snag fall across tree species. We then investigated how the tree mortality event affected surface and canopy fuel loading and potential impacts on fire hazard and emissions.

Results Drought and beetle-related tree mortality shifted mortality patterns to be more evenly distributed across size classes and concentrated in pines. Substantial changes to surface fuel loading, stand density, canopy fuel loads, and potential wildfire emissions occurred within four years following peak levels of tree mortality, with the largest changes related to increases in coarse woody debris. Nearly complete needle fall occurred within four years of mortality for all species except red fir (*Abies magnifica*). Pine species and incense cedar (*Calocedrus decurrens*) snags fell more quickly than fir species. Potential fire behavior modelling suggested that crowning and torching hazard decreased as trees dropped dead needles and fell, but as canopy fuels were transferred to surface fuels, potential for smoldering combustion increased, causing greater emissions.

Conclusions Our study increases understanding of how extreme tree mortality events caused by concurrent disturbances alter canopy and surface fuel loading and have the potential to affect fire behavior and emissions in two compositionally different seasonally dry forest types. After a major tree mortality event, high canopy fuel flammability may only last a few years, but surface fuels can increase considerably over the same time period in these forest types. The accumulation of coarse woody surface fuels resulting from multi-year drought and concurrent bark beetle outbreaks combined with the increasing frequency of drought in the western U.S. have the potential to lead to heavy and dry fuel loads that under certain weather conditions may result in more extreme fire behavior and severe effects, particularly in forest types where decades of successful fire suppression has caused forest densification.

Keywords *Pinus*, *Dendroctonus*, *Ips*, *Scolytus*, *Calocedrus*, *Abies*, Wildfire, Epidemic, Fuel hazard, Tree mortality

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Resumen

Antecedentes Una sequía extrema desde 2012 a 2016 y un estallido simultáneo del escarabajo de la corteza en California, EEUU, resultó en una extensa mortalidad de árboles. Seguimos los cambios en la mortalidad de árboles, la estructura de los rodales y los combustibles superficiales y de los doseles por cuatro años luego del pico de mortalidad en bosques mixtos de coníferas y de pino piñonero (*Pinus monophylla*) para examinar patrones de mortalidad, retención de acículas luego de la muerte de los individuos, y caída de muertos en pie en todas las especies. Luego investigamos cómo los eventos de mortalidad de árboles afectaron la carga de los combustibles superficiales y de los doseles, y su potencial impacto sobre el riesgo de incendio y las emisiones.

Resultados La sequía y la muerte relacionada con ataques del escarabajo de la corteza desvió los patrones de mortalidad para distribuirlos más uniformemente entre las distintas clases de tamaños y concentrada en los pinos. Cambios sustanciales en la carga del combustible superficial, densidad de los rodales, carga de combustible de los doseles, y en emisiones potenciales de los incendios ocurrieron cuatro años después de los niveles pico de mortalidad de árboles, con los cambios más grandes relacionados con incrementos en los residuos leñosos gruesos. Casi una completa caída de acículas ocurrió cuatro años luego de la mortalidad para todas las especies excepto para el abeto rojo (*Abies magnifica*). Los árboles muertos en pie de las especies de pinos y el cedro incienso de California (*Calocedrus decurrens*) cayeron más rápidamente que las especies de abetos. El modelado del comportamiento potencial del fuego sugirió que tanto la posibilidad de fuegos de copas como coronamiento decrecieron en la medida que los árboles perdían sus acículas y luego caían, pero dado que los combustibles del dosel eran trasferidos a combustibles superficiales, el potencial para fuegos de rescoldo aumentaba, causando mayores emisiones.

Conclusiones Nuestro estudio incrementa el conocimiento sobre cómo la mortalidad extrema de árboles causada por disturbios simultáneos altera las cargas de combustibles superficiales y del dosel arbóreo, y tienen el potencial para afectar al comportamiento del fuego y emisiones en dos tipos composicionalmente diferentes de bosques secos estacionales. Luego de un evento mayor de mortalidad de árboles, la alta inflamabilidad de los combustibles del dosel puede durar solo unos pocos años, pero los combustibles superficiales pueden incrementarse considerablemente en el mismo período en esos tipos de bosques. La acumulación de combustibles gruesos superficiales resultantes de sequías prolongadas y simultáneamente estallidos de escarabajos de la corteza, combinados con el incremento de sequías en el oeste de los EEUU, tienen el potencial de acumular combustibles gruesos y secos que bajo ciertas condiciones meteorológicas pueden resultar en comportamiento del fuego más extremo y efectos severos, particularmente en tipos forestales en los cuales décadas de supresión de incendios exitosa ha causado la densificación de esos bosques.

Background

Disturbance interactions regularly shape ecosystem dynamics in forested systems, with cascading effects that can lead to decades to centuries-long impacts on ecosystem composition and function (Buma 2015). Understanding the dynamics and consequences of interacting disturbances is particularly important as disturbance regimes are altered by anthropogenic change. Warmer temperatures and increasing aridity have been associated both with increasing drought frequency and intensity (Diffenbaugh et al. 2015) and greater frequency and size of wildfires (Westerling et al. 2006; Littell et al. 2016; Abatzoglou and Williams 2016). Furthermore, drought often triggers bark beetle (Coleoptera: Curculionidae, Scolytinae) outbreaks in conifer forests across the northern hemisphere (Kolb et al. 2016; Young et al. 2017), and when this combination causes extensive tree mortality, it may alter wildfire hazard, behavior, or resulting fire severity through changes in fuel characteristics (Kane et al. 2017; Gray et al. 2021; Wayman and Safford 2021).

Drought alone or in conjunction with bark beetle outbreaks may directly affect fire characteristics and ecological outcomes. Drought, and particularly multi-year drought, reduces fuel moisture to critical levels, increasing the risk of wildfire occurrence and widespread burning (Littell et al. 2016). Further, drought-fire interactions may result in an increase in fire severity if pre-fire drought physiologically weakens trees and predisposes them to fire-related mortality (van Mantgem et al. 2013, 2018) or drives increases in fire intensity (Prichard and Kennedy 2014; Sieg et al. 2017). When drought triggers large-scale tree-killing bark beetle outbreaks, as has been occurring in the western U.S. (Bentz et al. 2010; Fettig et al. 2019), changes to stand and fuel structure may alter subsequent fire behavior and severity (Bigler et al. 2005; Hicke et al. 2012; Wayman and Safford 2021; Stephens et al. 2018). While bark beetle outbreaks alone likely do not change the probability of fire occurrence, which is more strongly driven by climate (Mietkiewicz and Kulakowski 2016; Meigs et al. 2016; Hart et al. 2015), changes in both

canopy and surface fuels do occur as time since beetle-induced tree mortality progresses, particularly when mortality is extensive (Hicke et al. 2012). For example, when trees die from bark beetle attacks, they retain "red" dead needles with reduced water content that increases flammability of needles and increases likelihood of crown fire ignition (Page et al. 2012, 2014; Jolly et al. 2012b; Gray et al. 2021) and potential rate of spread (Hoffman et al. 2015; Perrakis et al. 2014). Yet some studies have shown decreased fire severity with time since outbreak (Meigs et al. 2016) or no influence (Harvey et al. 2014). The red-needle phase has been estimated at 1–4 years with most data coming from mountain pine beetle (*Dendroctonus ponderosae* Hopkins) and spruce beetle (*Dendroctonus rufipennis* Kirby) outbreaks (Hicke et al. 2012), but how this timeframe varies across host and insect species is not well understood. Canopy fuel loads decline following needle fall, reducing crown fire potential, while litter and surface fuels increase as needles drop and snags deteriorate (Klutsch et al. 2011; Hicke et al. 2012; Woolley et al. 2019). As time since tree mortality increases and a greater proportion of snags fall, surface fuels continue to increase, particularly coarse woody debris (CWD), likely increasing the potential for smoldering combustion and greater emissions in the event of a wildfire (Klutsch et al. 2011; Hansen et al. 2015), although the timing and degree of these impacts are not well quantified in many forested ecosystems. Over the decades following the tree mortality event, ladder fuels tend to increase as tree regeneration occurs. This timeline for change in fuel and potential fire behavior dynamics is expected to be highly variable and dependent on the severity of the tree mortality event, how long the host species affected retain needles and persist as snags on the landscape before falling, and tree and shrub regeneration dynamics (Kane et al. 2017).

Understanding these cascading disturbance effects is necessary from both an ecological and a fire and fuels management perspective. Changes to canopy and surface fuels in the years to decades after a major tree mortality event may lead to higher severity impacts when wildfire does occur (Wayman and Safford 2021). High-severity fire may further exacerbate changes in species composition initiated by drought and bark beetle outbreaks or lengthen the timeline for post-fire forest recovery, particularly if drought re-occurs (Meng et al. 2015; Wayman and Safford 2021; Stevens et al. 2019; Bright et al. 2019). From a management perspective, the unique timeline over which fuels change for a given system affected by large-scale tree mortality events also has implications for firefighter safety and suppression tactics (Page et al. 2013). For example, as long as dead trees retain needles, the likelihood of active crown fire may be elevated, and higher intensity crown fires and

greater rates of spread require larger safety zones and decrease escape times (Page et al. 2013; Hoffman et al. 2015; Moriarty et al. 2019). An additional concern, but difficult to accurately estimate, is the higher spotting potential from crown fires that can increase spread rates and pose safety issues (Albini et al. 2012). In the years following an outbreak, snag longevity becomes a concern as an abundance of increasingly unstable snags create extremely dangerous conditions for suppression activities despite potential declines in the probability of extreme fire behavior (Page et al. 2013; Jenkins et al. 2012). As snags fall and become surface fuels, surface fire intensity may increase even as rates of spread decline (Page et al. 2013), with firefighters reporting "surprising" increased fire behavior for fires that burned in past beetle outbreaks with heavy coarse woody surface fuels (Moriarty et al. 2019). Additionally, persistent smoldering of CWD may increase or prolong wildfire emissions, having implications for the health of both firefighters and the surrounding communities (Hyde et al. 2011).

From 2012–2016 a particularly hot, multi-year drought in California, USA resulted in the mortality of an estimated 147 million trees largely due to concurrent bark beetle outbreaks affecting multiple tree species (Goulden and Bales 2019; USDA Forest Service 2019). Mortality peaked in 2015–16 with forests in central and southern California being most highly impacted (Fettig et al. 2019). With this type of extreme tree mortality event anticipated to become more likely under future climates (Breshears et al. 2005; Allen et al. 2010), particularly as wildfire frequency and extent are expected to increase (Littell et al. 2016; Abatzoglou and Williams 2016), the 2012–16 drought and beetle outbreaks provided a valuable opportunity to assess the potential for interacting disturbances to have cascading or compounding effects, and to examine the implications of these effects for fire and fuels management.

We aimed to increase understanding of how fuels and stand structure change after two, multi-year disturbances caused widespread tree mortality within two forest types for which our understanding of fuel changes after mortality events is lacking (Stephens et al. 2018). We followed changes over four years after peak tree mortality at two sites with contrasting species compositions and examined the following questions: 1) Do patterns of mortality differ across tree species and size classes? 2) How do patterns of needle retention (i.e., dead canopy fuels) and snag fall rate differ across species? 3) How do the combination of existing stand conditions, drought, and bark beetle outbreaks affect surface and canopy fuel loading, and what impact might this have on fire hazard and potential emissions?

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We addressed these questions by following changes in tree mortality, stand structure, and surface and canopy fuels over four years following the extreme California drought of 2012–2016, and by modeling the potential for crowning and torching behavior and two types of wildfire emissions of concern.

Methods

Study sites

We established a network of plots at two sites in central and southern California, USA where tree mortality from the 2012–2016 drought and subsequent bark beetle outbreaks was extensive (Figure S1). The Sierra National Forest site (SNF) in central California spans an elevational range of 1220 to 2220 m with slopes ranging from 0 to 60%, and soils primarily well to excessively drained inceptisols with some entisols at higher and alfisols at lower elevations (NRCS 2019). Plots were co-dominated by ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), incense cedar (*Calocedrus decurrens* (Torr.) Florin), white fir (*Abies concolor* (Gord. & Glend.) Lindl. Ex Hildebr.), and black oak (*Quercus kelloggii* Newb.) at elevations below 2000 m with understories primarily dominated by *Ceanothus spp.*, *Ribes spp.*, manzanita (*Arctostaphylos spp.*), and bearclover (*Chamaebatia foliolosa* Benth.). At higher elevations, plots were co-dominated by red fir (*Abies magnifica* A. Murr.), white fir, and lodgepole pine (*Pinus contorta* var. *murrayana* (Grev. & Balf.) Critchfield), with sparse understories composed of chinquapin (*Chrysolepis sempervirens* (Kellogg) Hjelmq.), *Ceanothus spp.*, and manzanita. Ponderosa pine, incense cedar, and white fir serve as primary hosts to western pine beetle (*Dendroctonus brevicomis* LeConte), cedar bark beetles (*Phloeosinus* spp.), and fir engraver (*Scolytus ventralis* LeConte), respectively. The ponderosa pine and mixed conifer forests occurring at low and mid elevations of the western Sierra Nevada historically experienced low-intensity, frequent fire (Safford and Stevens 2017), but fire suppression and other factors in the 1900s have resulted in higher density forests with a highly departed historical fire regime (Stephens et al. 2018; Safford et al. 2021). Higher elevation mixed conifer forests have similarly experienced increasing densification, although historic fire return intervals were slightly longer than lower elevation forests (Safford and Van de Water 2014). The Los Padres National Forest site (LP) in southern California ranges from 1670 to 1890 m in elevation with slopes from 10 to 60% and was dominated by singleleaf pinyon (*Pinus monophylla* Torr. & Frém) with canyon live oak (*Quercus chrysolepis* Liebm.) and scrub oak (*Quercus berberidifolia* Liebm.) dominant in the understory. Soils are primarily well-drained

mollisols (NRCS 2019). Pinyon is a primary host to pinyon ips (*Ips confusus* LeConte). The pinyon woodlands of the transverse ranges of southern California are much less studied than the Sierra Nevada but have also experienced changes to the historical fire regime, which increased pinyon establishment in the late 1800s and early 1900s (Safford et al. 2021; Safford and Van de Water 2014).

Plot selection is detailed in Reed and Hood (2021). Briefly, we selected 49 plots at the SNF site from an existing network of plots (Pile et al. 2019), with the intention of capturing a range of tree mortality levels across stands. At the LP site, we randomly selected 27 plots within areas of both high and low mortality (Fig. 1). All plots had greater than 10.3 m^2 ha^{-1} basal area at the time of plot selection and tended to be grouped in distinct clusters across the study areas due to the initial targeting of high (> 40%) and low (< 5%) mortality areas based on 2015 Aerial Detection Survey data (https://tinyurl.com/28htd6uf). We installed fixed area 0.04-hectare circular plots (Figure S1) in 2016 and monitored plots annually through 2019 to assess changes in tree mortality status and fuels.

Field data collection

At each plot we recorded elevation, slope, and aspect. During plot establishment in 2016 we tagged and measured diameter and height for all trees > 12.7 cm in diameter at breast height (DBH; 1.37 m). We used DBH as the measure of diameter for all species except pinyon, for which diameter is more appropriately assessed at root collar (DRC) due to its short stature and tendency to have multiple stems. All references to trees henceforth refer to those that are > 12.7 cm DBH or DRC. For mortality monitoring, we recorded tree status (live or dead) and beetle attack status (unattacked, current year successful attack, prior year successful attack) annually from 2016 through 2019 for all trees. We assessed mortality agent for dead trees based on the presence and pattern of insect-specific galleries (Wood 1972). It was not possible to discern if mortality was due to drought alone or in combination with mass attack by primary bark beetles as nearly all dead trees had evidence of beetle attack. In order to assess patterns in needle retention and snag longevity, we also recorded percent needle retention (i.e., red, dead, needles still on tree), snag code (a proxy for snag age; Table S1), and measured break height for all dead trees annually.

For canopy fuels assessments, we recorded crown fuel base height, crown ratio, and crown position for each live tree. Saplings (DBH or DRC < 12.7 cm and height > 1.37 m) and seedlings (height < 1.37 m) were also assessed on a 0.004 ha subplot. We recorded diameter
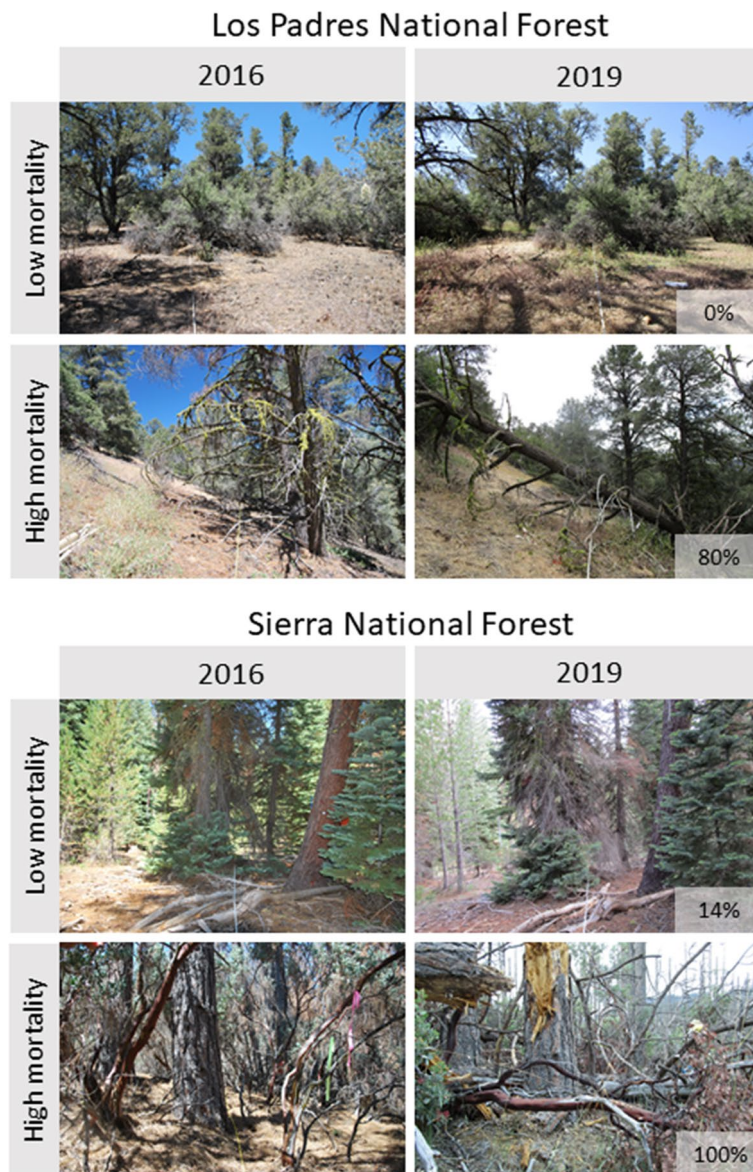
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Fig. 1 Repeat photos from 2016 and 2019 at plots with low and high tree mortality on the Los Padres and Sierra National Forests, CA, USA. Boxes in lower right corners are mortality percentages in the final sampling year (2019). Photos from U.S. Forest Service

(saplings) and height classes (seedlings) following Lutes et al. (2006), and annually assessed status (live or dead), average crown ratio, and average crown fuel base height. We assessed fine surface fuels (i.e., 1-, 10-, and 100-h fuels) at eight, 1 m² subplots within each 0.04 ha plot using the photoload method (Keane and Dickinson 2007). We measured shrub and herb cover as well as litter and duff depth within each fuel subplot. For coarse surface fuels (i.e., CWD or 1000-h fuels; ≥ 7.62 cm diameter), we measured the length and end diameters of all logs within the 0.04 ha plot (Keane and Dickinson 2007; Bowen et al. 2020).

Data processing and analysis

To assess tree mortality patterns pre-drought compared to the final year of sampling (2019), we estimated tree status in 2012 using snag codes recorded in 2016. A tree was estimated to be alive prior to the drought onset if it had tight, intact bark, no sapwood decay, some needle retention, and at least 50% fine branching remaining. We further estimated year of drought-related mortality using snag codes (pre-2016 mortality) and observations of new mortality during the study period to assess snag longevity. We similarly assessed needle longevity but limited our analysis to trees that died in 2015 and later to be able

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to track dead and dying trees with a full needle component initially through time. We report trends but did not statistically compare species differences in rates of needle loss due to the short timeframe over which needles were lost. We additionally grouped dead trees into diameter classes to assess species-specific patterns of mortality pre-drought (2015) and post-drought (2019). For all analyses of tree mortality, needle retention, and snag fall, we only included species for which we had at least 20 observations.

All tree, sapling, seedling, and surface fuels data from 2016–2019 were entered into a FEAT/FIREMON Integrated (FFI v1.05.08) database (Lutes et al. 2009). We used outputs from FFI to calculate plot-level stand structure metrics for both sites (LP, SNF) from tree, seedling, and sapling data for comparison across three time steps: pre-drought (using 2012 estimated tree status as described above), 2016, and 2019 using FuelCalc v1.7 (Reinhardt et al. 2006). We excluded seedlings from the smallest height class (< 15 cm) as we wanted to capture seedlings that had a higher probability of survival. Stand structure metrics included basal area (BA), trees per hectare (TPH), quadratic mean diameter (QMD), and stand density index (SDI). For pinyon pine, DRC was converted within FuelCalc to DBH following Chojnacky et al. (2014). We compared stand metrics across timesteps using repeated measures ANOVA with pairwise t-test or Friedman test with Wilcoxon signed-rank test for pairwise comparisons, depending on whether the data conformed to normality expectations, which was visually assessed via Q-Q plots in R 4.0.3 (R Core Team, 2020).

Plot-level surface fuel loads (kg ha^{-1}) were output from FFI for fine woody debris (FWD; 1, 10, 100-h), coarse woody debris (CWD; 1000-h), live and dead surface fuels (herbs, shrubs), and duff and litter. Duff and litter loads were calculated by FFI from depth measurements using the default bulk density values of 0.882559 kg m^{-2} and 0.44126 kg m^{-2}, respectively (Lutes et al. 2009). Our analyses compared surface fuel loads at both sites between 2016, when the majority of drought and beetle-related tree mortality had occurred but most dead trees still retained needles and had not fallen, and 2019, the final year of sampling, using Wilcoxon signed-rank tests. Since changes in fuel loads between 2016 and 2019 were primarily observed for 1000-h fuels (see results below), we used generalized linear modeling (GLM) to elucidate the primary drivers of the observed changes. We modelled changes to 1000-h fuels between 2016 and 2019 for both sites (LP and SNF) with one model and included site as a fixed effect. Other fixed effects included percent tree mortality, elevation, aspect, and slope, and pre-drought stand structure variables (BA, QMD, SDI, TPH). Stand structure variables were highly collinear

so we did not include more than one stand structure variable in each model. We additionally assessed models with an interaction between stand structure variables or percent tree mortality and site because we expected that inherent site differences might influence how stand structure or degree of tree mortality affected changes in fuels. Because our dataset included many zeroes in the response variable but otherwise followed a regular exponential distribution, we used a Tweedie distribution from the exponential dispersion family for our GLM (Smyth 1996; Foster and Bravington 2013). We used Akaike Information Criterion (AIC) to assess model fit with the combination of predictor variables that yielded the lowest AIC selected as the "best-fit" model. All data analyses were conducted in R 4.0.3 (R Core Team, 2020). We used the packages *tweedie* (Dunn 2022) and *statmod* (Giner and Smyth 2016) for GLMs with Tweedie distributions. During model development we examined diagnostics for potential outliers using Q-Q plots. One plot was an outlier, but after confirming that values were not due to sampling error, we opted to include the plot in all data analyses, as excluding the outlier plot did not change model conclusions (data not shown).

We calculated four canopy fuel measures and two potential fire behavior measures using FuelCalc v1.7 (Reinhardt et al. 2006). We used surface fuel outputs from FFI combined with annual tree, sapling, and seedling data as inputs. From FuelCalc we calculated canopy fuel load, canopy bulk density, canopy base height, stand height, and crown fire hazard (i.e., Torching and Crowning Indices) for pre-drought, 2016, and 2019. Detailed descriptions of canopy fuel characteristics are included in Lutes (2020b). FuelCalc calculates Crowning and Torching Indices (Scott and Reinhardt 2001) using the embedded code of Nexus 1.0 (Scott 1999), and Nexus creates a custom fire behavior fuel model from canopy and surface fuel inputs. Torching Index is defined as the wind speed at 6 m which can initiate crown fire activity and is a function of surface fuel loads, composition, and moisture, canopy base height, and slope and wind reduction from the canopy. Crowning Index is the 6 m wind speed at which active crown fire (i.e., spreading between tree crowns) can occur and is a function of canopy bulk density, surface fuel moisture, and slope (Lutes 2020b). Higher values for both indices are indicative of more extreme wind conditions needed for torching or crowning to occur. FuelCalc assumes that dead trees have zero needle retention and thus all canopy fuel and the two fire behavior metrics do not account for retention of dead needles in tree crowns. For the modeling of both fire behavior metrics, we used the "very dry" moisture regime defaults with fuel moisture of 5% for 100-h fuels, 4% for 10-h fuels, 3% for 1-h fuels, live herb and woody fuel
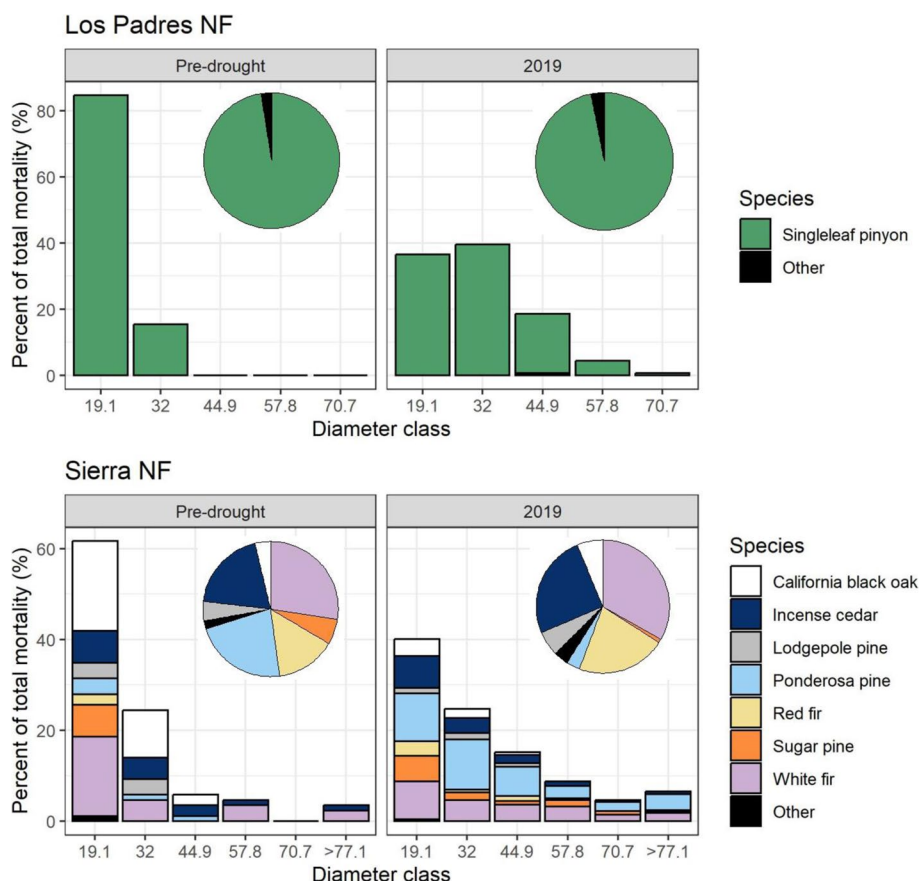
Fig. 2 Diameter distribution of tree (> 12.7 cm) mortality prior to the drought and in 2019 by species on the Los Padres and Sierra National Forests. Note variable scales for y-axes between sites. Pie charts indicate species composition based on percentage of live trees. Only species with ≥ 20 observations included

moisture of 30% and 60%, respectively, and foliar moisture of 100%. Although FuelCalc does not use 1000-h fuel moisture for Crowning and Torching Index calculations, these values are likely representative of much of the average fire season at our SNF site where 1000-h fuel moistures typically range from about 5–10% (Goodwin et al. 2021). We compared values across years using Friedman tests with Wilcoxon signed-rank test for pairwise comparisons. To relate the estimated Torching and Crowning Indices to local wind patterns, we obtained average daily wind speeds and gusts during the primary fire season of July through October from 2009 to 2019 using FireFamilyPlus (Bradshaw and McCormick 2000). We used wind data from the Remote Automated Weather Stations nearest to each site (SNF: Mt Tom MMTC1, LP: Chuchupate CUUC1), as well as during the maximum fire growth period of the 2020 Creek Fire that burned near the SNF site.

We additionally investigated potential impacts of combustion by modeling emissions of particulate matter ($PM_{2.5}$) and volatilized CO_2 using the First Order

Fire Effects Model (FOFEM; Reinhardt et al. 1997; Lutes 2020a). We modeled emissions under default "very dry" conditions for 2016 fuels conditions versus those observed in 2019 using FOFEM expanded emissions factors (Urbanski 2014). Default "very dry" conditions are defined as fuel moisture contents of 20% for duff, 6% for 10-h fuels, 10% for 1000-h fuels, and soil moisture of 5%. We calculated total volatilized CO_2 emissions (kg m^{-2}) from flaming and smoldering combustion by converting CO and CH_4 into CO_2-equivalents and adding these to CO_2 emissions (S. Urbanski, pers. comm). We compared modeled values from 2016 and 2019 using paired t-tests.

Results
Overall tree mortality patterns
We captured a range of tree mortality levels at both sites with fuel changes visually observable at plots that experienced high mortality (Fig. 1). Tree mortality remained fairly constant over the sampling period at the LP site, with an average cumulative plot-level mortality of 28.1% (± 5.6, SEM) in 2016 and 31.2%

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(±6.1) in 2019, but increased considerably at the SNF site from an average of 44.1% (±3.9) in 2016 to 52.8% (±4.0) in 2019. When using estimated tree status from 2012 to compare tree mortality patterns pre-drought to 2019 observations, variability of drought-related mortality across species and size classes was clear (Fig. 2). For example, ponderosa pine mortality at the SNF site increased from 3% pre-drought to 93% in 2019, whereas black oak mortality only increased by 6% over the same time period (Table S2). At the LP site, pinyon mortality changed moderately (3–30%) from pre-drought to 2019. Pre-drought tree mortality primarily occurred in smaller size classes, but bark beetles targeted drought stressed trees in larger classes, such that tree mortality was more evenly distributed across size classes in 2019 (Fig. 2). At the SNF site, ponderosa pine mortality was low before the drought, but dominant across size classes by the end of the study period (Fig. 2). While the tree mortality event at the LP site altered forest structure, species composition did not change (Table 1, Fig. 2 insets). At the SNF site, large changes in both structure and species composition occurred (Table 1, Table S2), with plots shifting from shade intolerant pine species to shade tolerant species of incense cedar, red fir, and white fir (Fig. 2 insets).

Needle retention and snag fall

All species quickly lost their red needles, with less than 50% remaining two years after death. In general, red fir tended to lose their needles at a slower rate than any of the pines or incense cedar. Pinyon pine shed needles quickly, with an average of 56% of needles remaining the first year after death. Within two years of death, needle retention dropped greatly for all species, with a high of 48% for red fir and a low of 28% for incense cedar. Retention rates continued to decline steeply such that by year 4, all species except red fir had < 5% of needles remaining (Fig. 3a).

Most snags did not begin falling until 3–4 years after death. Approximately 5 years after dying, ponderosa pine snags fell most quickly, with red fir and white fir falling the slowest (Fig. 3b; Table S3). No red fir snags had fallen by the end of the study period. Although snag fall rates were low for red fir and white fir, 25% and 16% had broken tops, respectively, by the time snags were at least 5 years old (Table S3). It is possible that our estimates of snag-fall are conservative, as we did not begin monitoring until 2016, four years after drought-related tree mortality began, and thus, we may have missed drought and beetle-killed trees that fell between 2012 and 2015. We additionally explored relationships of diameter class and snag fall, but no patterns were apparent (data not shown).
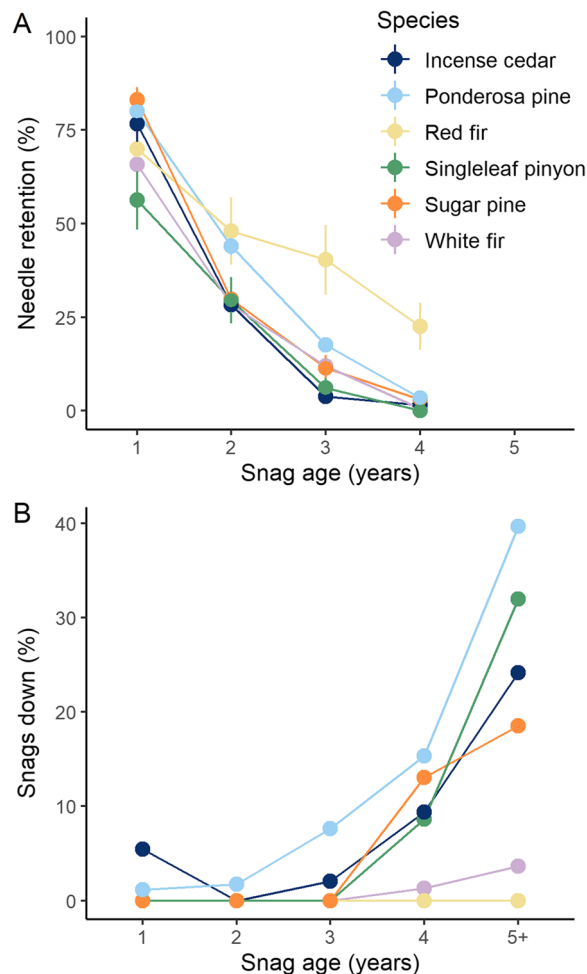

Fig. 3 Needle retention (%, ± SEM) of recently killed trees over time by species (**A**) and total snags broken below 1.37 m (%) by snag age and species (**B**). Singleleaf pinyon data are from the Los Padres National Forest site. Data from all other species are from Sierra National Forest site. Snag age beyond 5 years was estimated based on snag decay classes. No snag older than 6 years had any needles remaining. No snags older than 7 years (year of mortality = 2012) were included. No red fir snags had fallen by end of the study period in 2019

Surface fuels

Changes in fuel loads, particularly in the 1000-h class (i.e., CWD), over four years of sampling were especially pronounced at the SNF site where tree mortality was greater (Fig. 4; Table S4) and basal area was higher (Table 1), although the mean percent increase in 1000-h fuels from 2016–2019 at each site were similar (2656% ± 1723 for LP, 2830% ± 1828 for SNF). There were minor but significant changes in fine fuels in the 1-h and 100-h size classes at the LP site and for 1-h and herb cover at the SNF site (Fig. 4). The best-fit GLM model for changes in 1000-h fuels included site and the percent of drought-related tree mortality as predictors ($p < 0.01$;

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Table 1 Average stand structure metrics ± SEM (standard error of the mean) by site for live trees > 12.7 cm DBH at three timesteps (pre-drought, 2016, and 2019). Los Padres National Forest site includes 27 plots and the Sierra National Forest site includes 49 plots of 0.04 ha each. Letters indicate significant differences between timesteps ($p < 0.05$; repeated measures ANOVA with pairwise t-test or Friedman test with Wilcoxon signed-rank test for pairwise comparisons)

|	Pre-drought	2016	2019
Los Padres National Forest			
Mortality (%)	2.3 ± 0.9 a	28.1 ± 5.6 b	33.3 ± 4.4 b
Basal area (m² ha⁻¹)	19.1 ± 1.0 a	12.9 ± 1.4 b	13.0 ± 1.5 b
Trees per ha	400.9 ± 25.8 a	303.8 ± 33.9 b	292.0 ± 35.4 b
Quadratic mean diameter (cm)	25.2 ± 0.8 a	22.9 ± 0.9 b	23.5 ± 1.0 c
Stand density index	416.8 ± 22.6 a	296.0 ± 32.4 b	303.6 ± 35.5 b
Sierra National Forest			
Mortality (%)	6.0 ± 1.1 a	44.1 ± 3.9 b	52.8 ± 4.0 b
Basal area (m² ha⁻¹)	69.1 ± 4.7 a	43.8 ± 5.0 b	37.7 ± 4.8 b
Trees per ha	433.7 ± 27.4 a	266.8 ± 28.6 b	226.4 ± 27.2 c
Quadratic mean diameter (cm)	46.5 ± 1.9 a	44.63 ± 2.3 a	43.1 ± 2.6 a
Stand density index	1439.2 ± 91.5 a	975.1 ± 101.6 b	961.4 ± 106.6 b
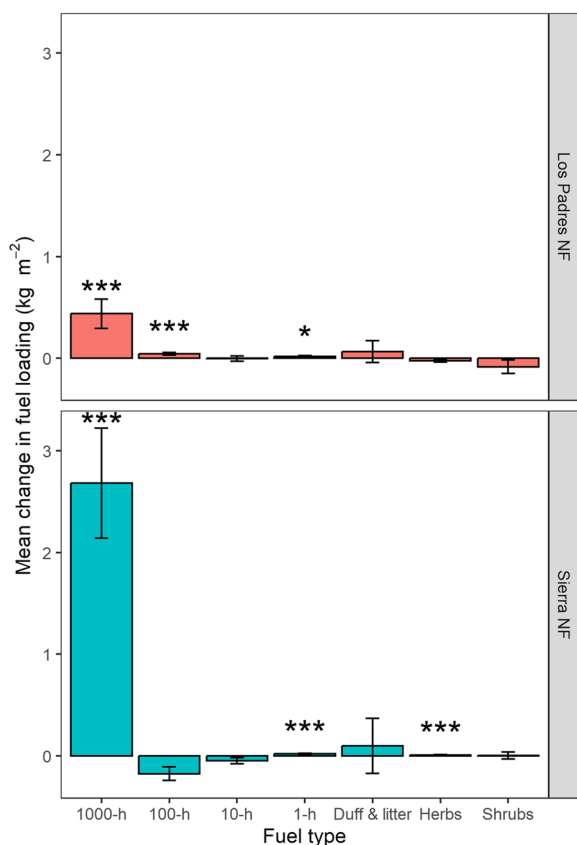
Fig. 4 Mean change in fuel loading from 2016–2019 across fuel types by particle size and component for the Los Padres and Sierra National Forests. Error bars are ± standard error of the mean (SEM). Asterisks indicate significant changes between 2016–2019 (Wilcoxon signed-rank test; * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$)

Fig. 5; Table S5). Once tree mortality reached about 60% at a plot, coarse woody fuel changes accelerated, particularly at the SNF site, although mortality did not exceed about 80% at any of the LP plots (Fig. 5). The next best model included QMD as an additional fixed effect (Table S4). No other site characteristics were significant in the model for predicting fuel changes.

Stand structure

Overstory stand structure (trees > 12.7 cm diameter) changed at both sites from pre-drought to 2019 (Table 1). BA, TPH, and SDI all declined when comparing pre-drought to 2019 values but did not necessarily change between 2016–2019. The tree mortality event caused BA to decrease an average of 32% at the LP site and 45% at the SNF site. QMD declined over the study period, but the change was only significant at the LP site. Species-specific changes in stand density metrics were highly variable, with ponderosa pine and sugar pine declining most substantially for all metrics (Table S2).

Changes for seedlings and saplings were not as apparent, and we did not attempt to reconstruct seedling and sapling status pre-drought since individuals were not tagged and were monitored by size classes (Table 2). Despite differences across years being insignificant for seedlings and saplings, we did observe trends towards decreasing QMD over the study period. At the SNF site, white fir was the most common seedling and sapling species, occurring in 59% of plots, followed by incense cedar on 27% of plots. Red fir and sugar and ponderosa pines occurred on 14% and 16% of plots, respectively. Shade tolerant species comprised the majority of regeneration in both 2016 and 2019 and changed little over
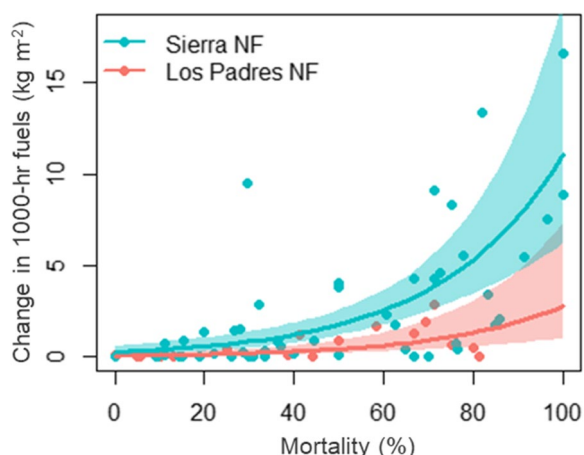
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Fig. 5 Change in 1000-h fuel loads from 2016 to 2019 by overstory (≥ 12.7 cm DBH) tree mortality (%) associated with the California drought and bark beetle outbreaks. Modeled relationships based on best-fit GLM model (lines) shows how site affects 1000 h fuel change. Shading corresponds to 95% confidence intervals. Sites are located on the Los Padres and Sierra National Forests (NF)

time, with incense cedar accounting for 59% of stems, white fir 55%, and red fir 50% in 2019. Ponderosa pine comprised 42% of regeneration in 2016, but declined to 25% in 2019. Black oak was stable with 43% of stems in 2019, as was sugar pine with 11% of stems. At the LP site, pinyon regeneration occurred on 37% of plots and did not change between 2016 and 2019.

Canopy fuels and fire hazard

Changes in canopy fuels characteristics and modeled fire hazard were variable across sites and across timesteps (Fig. 6). Both canopy fuel load and canopy bulk density decreased when comparing pre-drought to

Table 2 Average stand structure metrics ± SEM (standard error of the mean) by site for live seedlings and saplings < 12.7 cm DBH at two timesteps (2016 and 2019). Los Padres National Forest site includes 27 plots and the Sierra National Forest site includes 49 plots of 0.04 ha each. No significant differences between timesteps ($p < 0.5$)

	2016	2019
Los Padres National Forest		
Basal area (m² ha⁻¹)	0.6 ± 0.3	0.8 ± 0.3
Density (stems per ha)	283.7 ± 93.0	384.4 ± 108.2
Quadratic mean diameter (cm)	1.8 ± 0.6	1.7 ± 0.6
Sierra National Forest		
Basal area (m² ha⁻¹)	1.5 ± 0.3	1.2 ± 0.2
Density (stems per ha)	3610.8 ± 1059.7	6924.0 ± 2140.8
Quadratic mean diameter (cm)	2.7 ± 0.4	1.9 ± 0.3

2019 values as tree mortality increased (Fig. 6a,b), with changes between 2016 and 2019 being less pronounced. We observed no changes in canopy base height across timesteps (Fig. 6c). Stand height decreased at the SNF site but not at the LP site (Fig. 6d). Changes in fire hazard, as assessed by Torching and Crowning Indices, were also variable, but tended to decline over time, with higher windspeeds needed to initiate torching and crowning fires (Fig. 6e,f). Torching Index increased by 2019 at the SNF site but decreased somewhat at the LP site. By 2019 at the LP site, the average wind conditions and fuel hazard suggest that torching fires are likely in the event of a wildfire. At the SNF site, while average fuel hazard was not high enough to support torching fires by 2019, the conditions on many plots are well below that needed to support torching given average wind conditions (Fig. 6e). The tree mortality event increased Crowning Index at both sites (i.e., changes in fuels made it more difficult for crowning fires to occur), although there was little change between 2016 and 2019, and means were higher than average windspeeds observed at nearby climate stations (Fig. 6f). We caution that the crown fire hazard results do not account for red-phase dead trees and that many trees in 2016 still had a large percentage of dead needles remaining in their crowns.

Emissions potential

Changes in potential emissions if a wildfire were to burn under very dry conditions were apparent at the SNF site where tree mortality due to the drought and beetles was highest throughout the study period (Fig. 7; Table S5). Generally, modeled emissions were higher at the SNF site where fuels loading was higher than the LP (Table S6). Both modeled particulate matter (PM₂.₅) and total volatilized CO_2 emissions were higher under 2019 fuel conditions than 2016 at the SNF site where 1000-h fuels increased most considerably (Fig. 7).

Discussion

The interacting disturbances of the 2012–2016 California drought and bark beetle outbreaks resulted in widespread tree mortality, with the peak of mortality occurring in 2015–2016 across central and southern California (USDA Forest Service 2019; Goulden and Bales 2019). We found that while most mortality prior to the drought was concentrated in smaller size classes of overstory trees, drought and beetle-related mortality shifted patterns of tree mortality to be more evenly distributed across size classes and concentrated in species of pine (Fig. 2); these findings align with several other studies reporting patterns of the recent California tree mortality event (Fettig et al. 2019; Koontz et al. 2021; Robbins et al. 2022). We observed substantial changes to
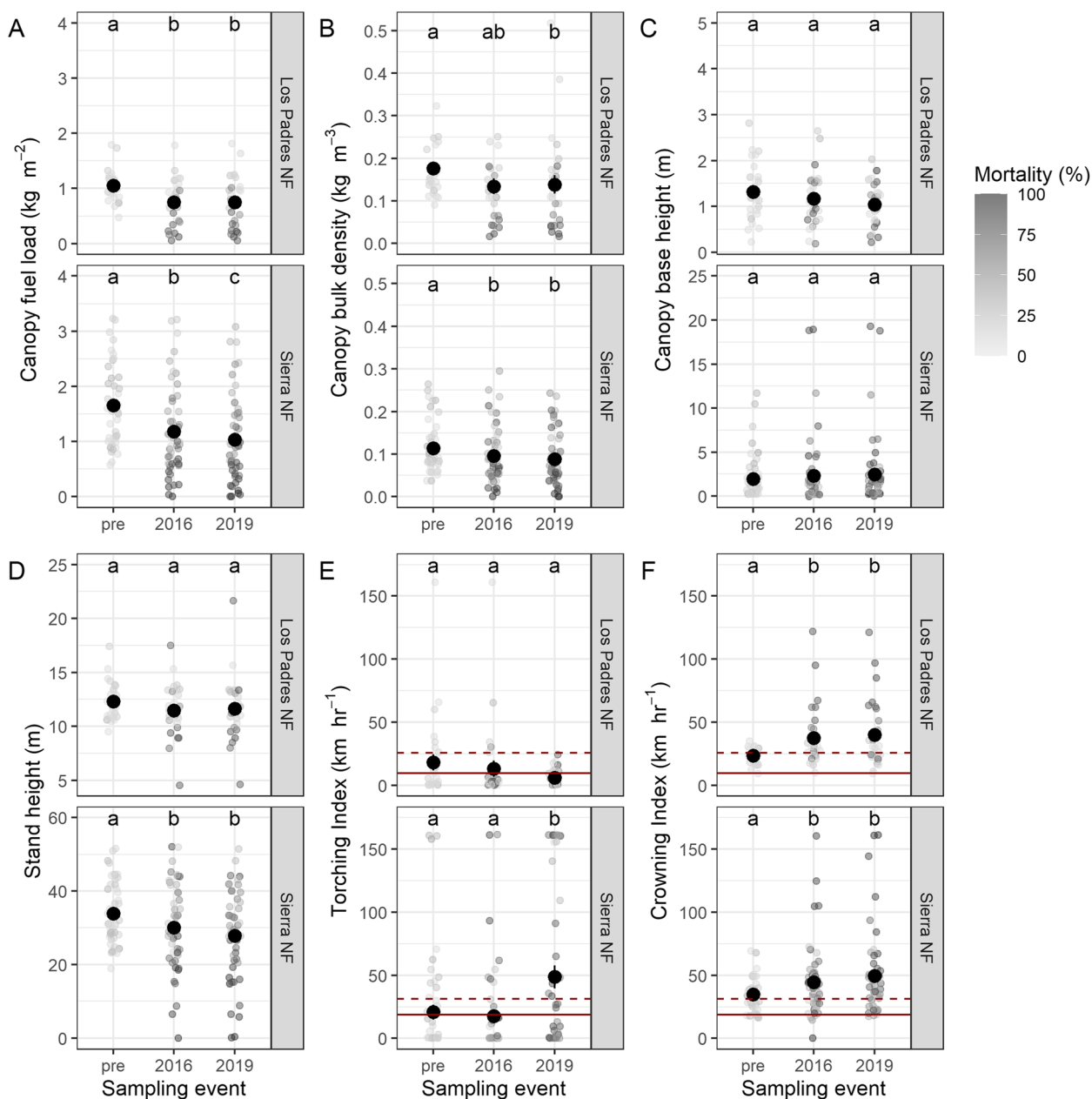
Fig. 6 Live canopy fuels (**A**-**D**) and Torching and Crowning Indices (**E**, **F**) at three timesteps (pre-drought, 2016 and 2019) on the Los Padres and Sierra National Forests (NF). Grey circles are plot-level indices with gradient indicating mortality (total) percent. Site means and standard errors are in black. Solid and dashed red lines indicate average wind speeds and average wind gusts (km hr.$^{-1}$), respectively, during fire season. Note variable scales between sites. Letters indicate significant differences between timesteps ($p < 0.05$; Friedman test with Wilcoxon signed-rank test for pairwise comparisons)

surface fuel loading, stand density metrics, canopy fuel loads, and potential wildfire emissions over a sampling period of just four years beyond when most tree mortality occurred. The most considerable changes to fuel loading occurred for coarse woody debris and appeared to be driven by the degree of drought-related mortality, with fuel loading increasing more steeply at a threshold

of around 60% overstory tree mortality (Fig. 5). In general, changes to surface fuel loading, along with stand density metrics, canopy fuel loads, fire hazard, and emissions were more pronounced at the Sierra National Forest site where fuel loads were typically higher and tree mortality more severe. Further, we observed that nearly complete needle fall occurred within about four years
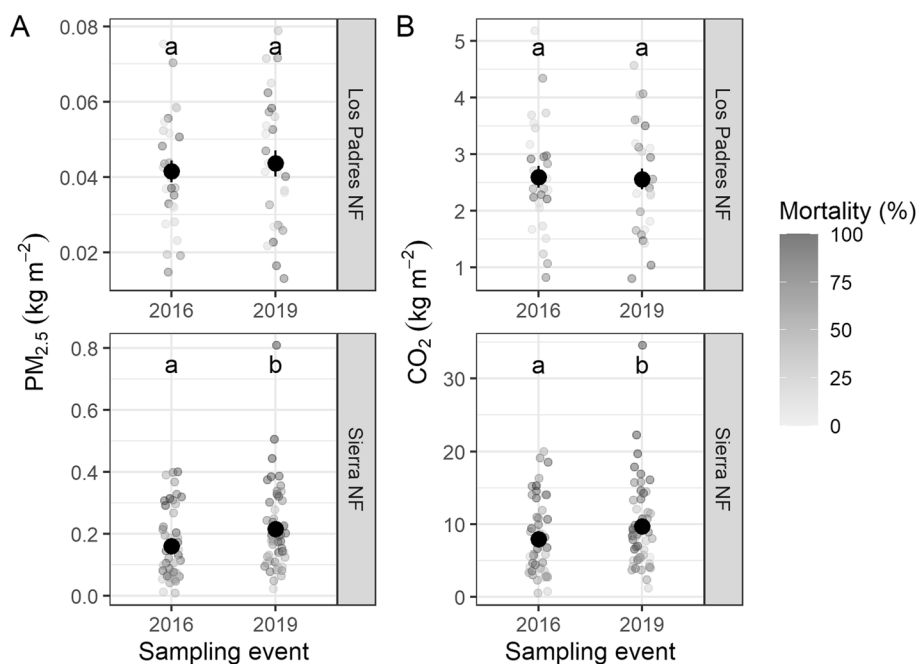
Fig. 7 Particulate matter (PM$_{2.5}$; **A**) and total volatilized CO$_2$ emissions (**B**). Grey circles are plot-level values with gradient indicating mortality (total) percent. Site means and standard errors are in black. Note variable scales between sites. Letters indicate significant differences between timesteps ($p < 0.05$; Wilcoxon signed-rank test for comparisons). Sites are located on the Los Padres and Sierra National Forests (NF)

of tree mortality for all species except for red fir. Snags appeared to fall at rates that were more speciesspecific, with pine species and incense cedar falling considerably sooner than fir (Fig. 3). No red fir had fallen by the end of the study period, which may not only be due to species specific differences in snag longevity, but the colder climate, and thus slower decay rate of these higher elevation mixed conifer forests (Rhoades et al. 2020). Less tree mortality in this forest type may have also contributed to this pattern, as the likelihood of snags falling and breaking other snags was lower. The variable snag-fall rates we observed have implications both for firefighter and recreational safety as well wildlife habitat, and our results aid in our understanding of how quickly coarse surface fuels may accumulate across these ecosystems.

Fuels change quickly after major tree mortality event

Changes to forest fuels after bark beetle-related tree mortality events tend to follow distinct phases. Initially, canopy fuels shift from live to dead and persist for a short period of time before becoming litter, a period estimated at 1–4 years in other forest types (Hicke et al. 2012). The changes we observed in canopy fuels followed similar patterns with 50–75% of dead needles lost within two years of death and near complete loss within four years, although this stage may last a bit longer in

higher elevation Sierra Nevada forests dominated by red fir (Fig. 3), perhaps due to slower decay rates in these forests. This suggests that the forest types in our study follow the predicted increase in canopy flammability immediately after tree mortality when there are an abundance of dead needles in the canopy, particularly since extensive mortality occurred within a relatively short time period (Jenkins et al. 2014). This red phase is very ephemeral however, as dead needles drop within four years and canopy bulk density is reduced.

As canopy fuels decline, surface fuels increase, first due to litter accumulation, and over longer periods, snags breaking or falling. Prior studies that followed changes in fuel loading after bark beetle outbreaks were conducted in more mesic forest types such as Rocky Mountain lodgepole pine (*Pinus contorta* var. *latifolia*) and spruce-fir (*Picea-Abies*) forests, where changes in coarse woody surface fuel loads may take decades to become apparent (Hicke et al. 2012). We were therefore surprised to find considerable changes in 1000-h fuel loads across both study sites within four years of peak drought and beetle-related tree mortality (Fig. 4). We attribute this to the speed with which snags fell over four years of monitoring, particularly for pine species (Fig. 3), which fell faster than, for example, mountain pine beetle-killed Rocky Mountain lodgepole pine (Audley et al. 2021; Rhoades et al. 2020) but at similar

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rates to beetle-killed ponderosa pine in the southwest U.S. (Chambers and Mast 2014; Hoffman et al. 2011). Landram et al. (2000) reported snag fall rates in northern California mixed-conifer forests and found that an average of 45% of ponderosa pine snags had fallen by 5 years since death, which is similar to our observation (40%). However, they found faster snag fall for white fir with an average of 25% of snags having fallen by 5 years since death, whereas our observations were lower at 4%. Our results for snag fall dynamics at our Sierra Nevada site also align with previous work following fire-killed snag fall in similar mixed conifer type forests in California, with pines falling most quickly (within ~ 10 years) and fir species more slowly (Ritchie et al. 2013; Grayson et al. 2019). However, our observations for incense cedar snag fall appear to be considerably faster than have been previously observed, with about 10% having fallen by four years after death, whereas fire-killed snag fall estimates describe a similar percentage after 10 years (Grayson et al. 2019). Taken together, these studies show widespread agreement that ponderosa pine snags quickly deteriorate regardless of mortality agent, but literature inconsistences in snag longevity for other species suggest other factors such as site and snag-cause are important to consider. The extreme drought that contributed to snag creation in our study is likely very different than other studies of snag dynamics, and additional research with larger sample sizes over longer time frames is needed to better determine how drought and bark beetles affect snag longevity.

Less is known about snag dynamics in pinyon pine-dominated woodlands of southern California like our Los Padres National Forest site, although observations from Colorado pinyon pine (*Pinus edulis* Engelm.) woodlands of southwestern Colorado indicate persistence of intact snags for an average of 8.4, but up to 25 years following tree mortality via bark beetles or root disease (Jacobi et al. 2005). Our snag fall observations may be following similar patterns, with over 30% of snags having fallen within about 5 years, although longer-term monitoring is needed to confirm these trends. With snag fall being incomplete, we expect both sites to continue to accumulate coarse woody fuels, and that these dynamics will be driven by site-specific species dominance.

Over longer timescales, regenerating trees and shrubs released from competition may increase ladder fuels (Hicke et al. 2012), although the consequences for fire behavior, particularly in stands with little remaining overstory, have yet to be quantified. Additionally, greater light availability in stands with high mortality may lead to increased shrub or herb abundance (Stone and Wolfe 1996; Pappas et al. 2020). While we saw some evidence of increasing regeneration at both of our sites (Table 2), we did not observe any increases in fuel loading from shrubs over the study period, although fuel loading from herbs did appear to increase at the SNF site (Fig. 4). A longer study is likely needed to capture these effects and determine how they may influence potential fire behavior.

Potential fire behavior and emissions

The drought and bark beetle outbreaks reduced canopy fuel load and canopy bulk density at both the SNF and LP sites, causing concomitant predicted increases in Crowning Index (Fig. 6). In other words, the tree mortality event likely reduced the chance that these forests could sustain an active crown fire post-needle drop except when windspeeds reach 50 km hr^{-1}, which is unlikely given historical wind speeds at nearby climate stations. Torching Index also increased at the SNF site but tended to decline at the LP site. Pre-drought and 2016 Torching Index at the SNF site were similar to average fire season wind speeds, but increased such that higher windspeeds than typically occur in the area would be needed to initiate torching by 2019 (Fig. 6). In contrast, by 2019 fuel conditions at the LP site were such that local windspeeds observed during the fire season would support torching fires. An important caveat is that fuel conditions on many individual plots on both the LP and SNF sites would support torching and crowning given the average windspeeds, suggesting that intense fire behavior should be expected in portions of these areas in the event of a wildfire. The challenges with fire behavior modeling and the limited ability to evaluate model predictions with empirical data (Alexander and Cruz 2013a) have led to inconsistencies and debate in the literature about whether bark beetle-impacted forests increase fire behavior relative to unattacked, green forests (Jolly et al. 2012a; Simard et al. 2011, 2012). We chose to limit potential fire behavior modeling to only crown fire hazard because of large uncertainties involved and violations of model assumptions when modeling fire behavior in beetle impacted forests (Jenkins et al. 2012; Page et al. 2014; Stephens et al. 2022) and urge caution when drawing inferences from the predicted fuel hazards, especially the 2016 estimates, as that is the peak of the red needle phase. FuelCalc, and all other fire behavior models based on Rothermel (1972) and Van Wagner (1977) fire behavior equations, assumes that dead trees have zero needle retention and thus all canopy fuel and the two fire behavior metrics we reported do not account for retention of dead needles in tree crowns. Yet in 2016, the recently killed trees had 56–83% of red needles remaining in the crown (Fig. 3a). Red and green needles from attacked trees ignite faster than green needles on unattacked trees, which could lead to increased crown fire potential (Jolly et al. 2012b).

Schoennagel et al. (2012) also show the importance of accounting for reduced foliar moisture content during the red needle phase when predicting crown fire behavior in beetle-impacted forests. Research on fire behavior in Rocky Mountain lodgepole pine forests impacted by mountain pine beetle shows that once a crown fire is initiated, fires burning under dry, windy weather conditions are likely to have similar behavior regardless of the level of bark beetle-related tree mortality (Schoennagel et al. 2012). It is unknown if these findings extend to mixed-conifer and pinyon forests of California. An additional complication for fire behavior modeling in mixed-conifer forests where bark beetle-related tree mortality is highly variable and related in part to host abundance, is that there is often a transition period where surface fuels increase, yet existing snags and live, drought-stressed trees not killed by beetles remain standing, creating highly heterogeneous fuel conditions and microenvironments. Indeed, recent work from an area of the Sierra Nevada that burned in the 2020 Creek Fire (see details below) adjacent to our SNF site found both dead biomass and live tree densities were the most important contributors to high fire severity, and that current operational fire behavior models did a poor job of replicating observed fire behavior at this site, likely because these models do not incorporate post-frontal combustion of coarse fuels (Stephens et al. 2022). Modeling efforts using physics-based models that can better account for the low foliar moisture content of fading and red needles suggest bark beetle-caused tree mortality increases fire rate of spread above non-beetle impacted forests, with spread rates peaking during the red phase, but remaining higher than in green forests even as canopy fuels decrease (Hoffman et al. 2015). Observations from the 2020 Creek wildfire and Wayman and Safford (2021) also suggest our hazard estimates may not reflect real-world conditions and strongly suggest that drought and beetle-impacted mixed conifer forests burn under higher intensity and with resulting higher severity (i.e., increased fire-caused tree mortality) even after red needles drop and bark beetle-killed forests enter the gray phase (Stephens et al. 2022). Much less is known about potential fire behavior in pinyon pine forests.

How fire severity changes due to drought and beetle-killed forests relative to green, unattacked forests depends on the phase of the outbreak, weather at the time of fire, and forest type (Sieg et al. 2017; Wayman and Safford 2021). Using a physics-based model, Sieg et al. (2017) found fire severity (i.e., tree mortality) increased in ponderosa pine-dominated beetle-impacted forests during the red phase, but was unchanged or decreased in the gray phase compared to green forests. The interactions between fire and bark beetles on resulting severity were weakened under high wind conditions (Sieg et al. 2017). The empirical findings of Wayman and Safford (2021) are consistent with physics-based model predictions, showing that under milder weather conditions (i.e., higher relative humidity), fires burning in California mixed-conifer forests in the red phase resulted in higher fire-caused tree mortality, but bark beetle killed-trees did not influence fire severity under more extreme weather conditions. Studies have shown fire severity, measured as change in vegetation, is highest in the red phase (Prichard and Kennedy 2014), decreases with time since outbreak (Meigs et al. 2016), or may be largely unrelated to bark beetle attack severity in both red and gray phase (Harvey et al. 2014).

Despite the uncertainty and challenges of modeling fire behavior with current software systems (Alexander and Cruz 2013b), heavy accumulation of coarse woody surface fuels like we observed have likely consequences for smoldering combustion and wildfire carbon and particulate matter emissions (Hyde et al. 2011), particularly when fuel aridity is high (Goodwin et al. 2021). Indeed, we observed a potential for higher CO_2 and $PM_{2.5}$ emissions over the study period at our SNF site where tree mortality and coarse woody fuel accumulation were greatest. Even if potential for crowning and torching decrease as trees drop dead needles and break or fall, potential for prolonged smoldering combustion of dry coarse woody surface fuels remains, which may also increase the possibility of fires re-igniting (Hyde et al. 2011) and cause large areas of intense heating in the interior of the fire perimeter (Stephens et al. 2022). Prolonged smoldering combustion, then, has consequences for suppression activities, higher fuel consumption rates that lead to longer duration, high-severity heating, as well as extended smoke exposure for surrounding communities and greater carbon and particulate matter emissions beyond what is released during primary fire growth.

The recent Creek Fire of 2020 provides insight into expected fire behavior and smoke effects should our SNF plots burn. In the fall of 2020, the Creek Fire burned 153,738 ha adjacent to our SNF site in the western Sierra Nevada, with about half of its growth coming within its first week in early September (NICC 2020). A large portion of the burned area had been affected by extensive drought and beetle-related tree mortality, much like many of our plots, with high coarse woody fuel loads that had particularly low fuel moisture after years of drought and an unusually hot and dry summer in 2020 (Goodwin et al. 2021). Observed fire behavior was extreme with long-range spotting, with continued smoldering in heavy surface fuels for many weeks after initial fire growth (WFDSS 2020). Once conditions permitted, suppression activities prioritized constructing containment lines

in the area directly adjacent to our SNF plots where tree mortality was nearly 100% and coarse woody fuel accumulation was high, due to the proximity of multiple communities and concern for further extreme fire behavior within this area of heavy dead standing and surface fuels (WFDSS 2020). Smoke impacts were extensive across California with impacts on nearby communities being particularly extreme. One nearby community where a fire camp was located recorded $PM_{2.5}$ values greater than 700 µm m^3 (i.e., hazardous) during the most active period of fire growth. However, levels remained unhealthy even as fire growth slowed, possibly due in part to the pervasiveness of smoldering combustion in areas with extensive coarse woody surface fuels (Navarro and Vaidyanathan 2020). This fire spread quickly with extreme fire behavior through areas of high tree mortality similar to our SNF study site, despite most of the standing dead trees having dropped their needles by the time of the fire. While some fuel reduction projects had been completed around the communities within the fire footprint, past fuel treatments were not widespread, and recent work found no relationship between past forest treatments and fire severity on the Creek Fire (Stephens et al. 2022). Both unique weather, including unusually high vapor pressure deficit and strong winds, and low fuel moisture of coarse woody surface fuels at the time of the Creek Fire start likely contributed to the extreme fire behavior observed (Zhuang et al. 2021; Goodwin et al. 2021; WFDSS 2020). These conditions additionally constrained suppression activities and likely contributed additional particulate matter emissions due to prolonged smoldering combustion (Hyde et al. 2011). Fuel moisture content of 1000-h fuels during the Creek Fire footprint averaged about 6% (Goodwin et al. 2021), which is slightly lower than the "very dry" conditions we used in FOFEM to model emissions variables, and thus our estimates are likely conservative. Maximum wind speeds during the period of rapid fire spread reached 53 km hr^{-1}, but averaged 22 km hr^{-1}. Average predicted Crowning and Torching Indices at the SNF site in 2019 were still higher than observed average windspeeds, further highlighting the limitations of using fire behavior modeling systems in beetle-impacted forests. However, there was substantial variability in Crowning and Torching Indices across the SNF plots, with many falling under the 53 km hr^{-1} threshold.

The changes in fuels and potential fire behavior due to the tree mortality events observed on the LP and SNF have implications for public and firefighter safety. Snags created from beetle-killed trees are a safety concern along roads and for people recreating and working in these types of forests. High coarse fuel loadings increase fireline construction times and constrain fire suppression activities (NWCG 2013). Page et al. (2013) describe these impacts in detail. Firefighters have reported unexpectedly active fire behavior in mountain pine beetle-killed lodgepole pine forests (Moriarty et al. 2019) and our summary of the WFDSS Creek Fire Decision (above) supports many of these observations in mixed conifer forests impacted by western pine beetle, mountain pine beetle, and fir engraver. These considerations highlight the important distinction between understanding the effects of fire in beetle-impacted forests for planning and management purposes and the caution needed to safely manage wildland fire for firefighters and the public. Additional safety concerns relate to the inability to predict long-range embers from torching and crowning that can start new spot fires downwind of the main fire front, whether in beetle-impacted areas or not. Such fire behavior can dramatically increase fire spread rate and area burned (Stephens et al. 2018).

Site differences and outlook

We saw clear differences across our two study sites in terms of both initial fuel loads and changes post-drought, as well as with stand structure, and potential emissions. This is not surprising as there are inherent differences in species composition, stand structure, and historical fire regimes at the two sites. Mixed conifer forests in the low-mid elevational range of the western Sierra Nevada historically experienced frequent low-moderate severity fire that limited regeneration of more shade-tolerant species and biomass accumulation (Miller and Safford 2017; Safford and Stevens 2017; Stephens et al. 2018). Decades of fire suppression and resulting forest densification have led to higher accumulations of both surface and canopy fuels, as well as greater susceptibility to drought and bark beetle-related tree mortality (Young et al. 2017; Kolb et al. 2016). The tree mortality event especially impacted forest structure and species composition at the SNF site, with many plots shifting from shade intolerant pine species to shade tolerant species dominated by incense cedar and white fir in both the overstory and understory. As these species are less resistant to fire when young, but become fire tolerant with age, these shifts likely further erode resilience of this forest type to low-intensity fire and the ability for shade intolerant pine species to establish and persist. The drought and bark beetle outbreaks, combined with past forest management and fire suppression, have quickened the pace of departure from historical range of variation in Sierra Nevada mixed conifer forests (Steel et al. 2015, 2018). In contrast to our mixed-conifer site in the Sierra Nevada, the southern California pinyon pine woodland (LP) site at which we sampled had lower tree density due to being more climatically dry year-round, although it is likely that this forest type has increased in density from historic conditions as well.

These factors likely contributed to both the initial differences we observed in surface fuels and in the greater accumulation of coarse surface fuels at the SNF site as snags fell, as well as the observed differences in emissions potential. Species composition of the overstory did not change at the LP site, although this is mainly due to tree dominance by a single species (pinyon pine) compared to the mixed-conifer SNF site. However, reduced overstory density at many of the LP plots may lead to increases in shrub and herb abundance or diversity over longer time periods (Stone and Wolfe 1996; Pappas et al. 2020), which may have both ecological impacts related to wildlife habitat and carbon uptake potential (Gilliam 2007; Reed et al. 2014), as well as potential fire behavior impacts related to increasing ladder fuels or differences in flammability (Engber and Varner 2012). While the observed site differences may not be surprising from an ecological standpoint, they are important for highlighting the problems with making generalizations across seasonally dry forest types even within the same geographical region.

Conclusions

Our study begins to fill gaps in our understanding of how extreme tree mortality events caused by concurrent disturbances alter stand structure and canopy and surface fuel loading, and have the potential to affect ecological function, fire behavior, and emissions in two compositionally different seasonally dry forest types. Considerable mortality among pines at both of our study sites combined with fast accumulation of coarse woody fuels, particularly at the SNF site, are likely to further contribute to reduced resilience to wildfire in these forest types already impacted by a century of fire suppression (Safford and Stevens 2017; Bernal et al. 2022). While the length of time where canopy fuel flammability is high after a major tree mortality event may only last a few years, the timeframe over which surface fuels increase in these forest types may be shorter than more mesic or colder forest types (Hicke et al. 2012). The accumulation of coarse woody surface fuels resulting from multi-year drought and concurrent bark beetle outbreaks combined with the increasing frequency of drought in the western U.S. (Diffenbaugh et al. 2015; Seager et al. 2007), have the potential to lead to extremely heavy and dry fuel loads that under certain weather conditions may result in more extreme fire behavior or more severe fire effects as was observed during the 2020 Creek Fire (Goodwin et al. 2021; Stephens et al. 2022). This is particularly concerning as forest mortality events of this magnitude are likely to increase (Madakumbura et al. 2020; Allen et al. 2010), coupled with known limitations to accurately predict subsequent fire behavior and effects in beetle-impacted forests (Stephens et al. 2022; Jenkins et al. 2012).

Abbreviations

AIC	Akaike Information Criterion
BA	Basal area
CWD	Coarse woody debris
DBH	Diameter at breast height
DRC	Diameter at root collar
FFI	FEAT/FIREMON Integrated
FOFEM	First Order Fire Effects Model
FWD	Fine woody debris
GLM	Generalized linear modeling
LP	Los Padres National Forest
$PM_{2.5}$	Particulate matter
QMD	Quadratic mean diameter
SDI	Stand density index
SNF	Sierra National Forest
TPH	Trees per hectare

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s42408-023-00175-6.

Additional file 1: Figure S1. Left: Circles indicate general locations of study plots on the Los Padres National Forest (LP) and Sierra National Forest (SNF) overlaid on density of dead trees (trees per acre) due to the 2012 – 2016 drought and bark beetle outbreaks. Tree mortality determined from 2012-2016 Arial Detection Surveys (ADS) with data available: https://egis.fire.ca.gov/TreeMortalityViewer/. Right: Plot layout for monitoring of overstory trees (A), seedlings and saplings (B), and fuels (C). **Table S1.** Snag codes used to estimate time since tree death in the field. **Table S2.** Changes in stand density metrics at three time steps for live trees (> 12.7 cm) by species. **Table S3.** Percent of snags down (<1.37 m) and broken (>1.37 m) by age class. Total number of snags in that age class in parentheses. **Table S4.** Fuel loading (kg m-2) at initial (2016) and final (2019) sampling by particle size and component for the Los Padres and Sierra National Forests (NF). **Table S5.** Coefficients and AIC values of best-fit models of change in 1000-hr fuels between 2016 and 2019. **Table S6.** Changes in modeled emissions between 2016 and 2019.

Authors' contributions

Charlotte Reed: Investigation, Formal Analysis, Data Curation, Writing – Original Draft Preparation, Writing – Review & Editing; Sharon Hood: Conceptualization, Funding Acquisition, Methodology, Project Administration, Supervision, Investigation, Formal Analysis, Data Curation, Writing – Original Draft Preparation, Writing – Review & Editing; Danny Cluck: Investigation, Methodology, Resources, Writing – Review & Editing; Sheri Smith: Conceptualization, Funding Acquisition, Methodology, Resources, Writing – Review & Editing. The authors read and approved the final manuscript.

Availability of data and materials

Available at request of corresponding author.

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Declarations

Ethics approval and consent to participate
Not applicable.

Consent for publication
All authors consent to publication.

Competing interests
The authors declare that they have no competing interests.

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