

## ARTICLE

# The effects of prolonged drought on vegetation dieback and megafires in southern California chaparral

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**Abstract**

Drought contributed to extensive dieback of southern California chaparral, and normalized difference vegetation index before drought and near the end of the drought was used to estimate this dieback, after accounting for other disturbances recorded in aerial photographs. Within the perimeters of two megafires that occurred after the drought, the 2017 Thomas Fire and the 2018 Woolsey Fire, there had been extensive areas of dieback. Comparing dieback with Monitoring Trends in Burn Severity measures of fire severity, there was a highly significant negative relationship between drought-caused shrub dieback and fire-caused dieback as measured by fire severity. We interpret this as further support for our remote sensing methodology for prefire dieback. Models of fire behavior suggest that one means by which dieback contributes to fire size is through increasing the density and distance of spot fires, particularly under extreme wind conditions. Lower elevation chaparral associations appear to be most vulnerable and are closer to urban environments, which should be a concern to fire managers in regions subjected to extended droughts.

**KEYWORDS**

drought impacts, remote sensing, wildfires

## INTRODUCTION

Annual summer drought is one of the identifying characteristics of a Mediterranean climate, where there is typically up to 6 months a year without measurable precipitation. To survive these conditions, plants living in these regions are characterized by various drought resistance strategies and functional mechanisms that enable them to resist water stress during the hot, dry summer months. In the extensive chaparral shrublands of California, one of the five global Mediterranean-climate regions, species resist drought through one of two survival strategies, dehydration tolerance or dehydration

avoidance (Venturas et al., 2016), and the two strategies coincide with different life history types (Jacobsen & Pratt, 2018; Keeley, 1998; Paddock III et al., 2013).

Despite this capacity to withstand the Mediterranean-climate seasonal drought, there comes a point when the duration or intensity of the drought exceeds the adaptive capacity of many shrub species. When a plant becomes water-stressed and transpiration continues, xylem cavitation of stems occurs damaging the hydraulic capabilities of the plant (Vilagrosa et al., 2012). This in turn causes an embolism where the xylem conduits are filled with air and gases leading to the death of the stem (Schenk et al., 2008). These processes start at

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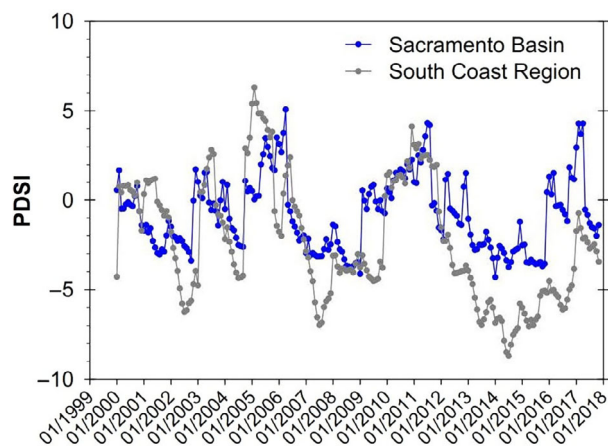
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branch tips causing individual stems to die back and under continued water stress may cause plant death (Davis et al., 2002). In mixed chaparral communities, associations comprise species with different drought adaptations (Pivovarov et al., 2016) and thus differential mortality rates depending on the species and landscape characteristics (Jacobsen & Pratt, 2018).

California has historically experienced a number of severe multiyear droughts, and these have been associated with extensive vegetation dieback (Goulden & Bales, 2019; Jacobsen & Pratt, 2018; Venturas et al., 2016; Yang et al., 2012). Not only can dieback be ecologically detrimental to the specific plants affected, but in fire-prone Mediterranean ecosystems, prolonged droughts and extensive dieback may lead to cascading ecological impacts and interactions (Field et al., 2020). It is not uncommon for fire to follow on the heels of drought in Mediterranean ecosystems. Annual summer droughts contribute to a natural fire regime of periodic large wildfires in California (Keeley et al., 2012; Keeley & Syphard, 2021). However, prolonged drought as seen this past decade in California has the potential for contributing to mass fires of extreme size and severity (Keeley & Syphard, 2021; Stephens et al., 2018). Frequent large fires in this region threaten native plant diversity due to uncharacteristically short fire return intervals (e.g., Safford & Van de Water, 2014; Syphard et al., 2019), and they also cause catastrophic human impacts (Syphard & Keeley, 2020).

California recently experienced one of the most severe droughts in its history (Griffin & Anchukaitis, 2014). It started in the latter part of 2011, and although the duration varied based on location, some of the southern portions of the state remained under year-round drought until spring 2017 (Figure 1) and were considered far worse than the northern part of the state (Dong et al., 2019). The loss of trees in California forests due to drought-related effects reached 129 million by 2017 (US Department of Agriculture–Forest Service, 2017), and there is evidence of massive dieback of shrubland vegetation in the Santa Monica Mountains, California (Figure 2), suggested by remote imagery (Gillespie et al., 2018) and field observations (Jacobsen & Pratt, 2018). Stephens et al. (2018) predicted that the huge tree dieback in the Sierra Nevada and northern California would contribute to massive forest fires—the 2020 and 2021 fire seasons in these regions would support that prediction. Likewise, following this extended drought in southern California, we saw two of the largest fires in that region: the 2017 Thomas Fire in Ventura and Santa Barbara counties and the 2018 Woolsey Fire in Los Angeles and Ventura counties. While massive tree dieback in the north has been considered a major factor behind large forest fires, it remains unknown the extent of chaparral shrubland dieback in the southern part of the state and its association with the Thomas and Woolsey Fires.

The purpose of this study was to document the connection between extreme drought and vegetation dieback in southern California shrublands within the fire perimeters of two of our largest wildfires. Since evergreen chaparral dominates these landscapes and is more sensitive to drought impacts than the often-associated summer-deciduous sage scrub (Okin et al., 2018), our focus was on chaparral. We



**FIGURE 1** Palmer drought severity index (PDSI) values from 1999 to 2018 for the Sacramento Basin and the South Coast region of California. The South Coast region includes the cis-montane areas from Pismo Beach to San Diego and is dominated by chaparral. For comparison, data are also shown from northern California, represented by the Sacramento Drainage Basin. PDSI values are based on precipitation inputs and temperature to measure drought conditions, and this index is correlated with soil moisture. Negative PDSI values indicate drought, positive values indicate wet years, and zero is the average. Data are from the National Oceanographic and Atmospheric Administration, National Centers for Environmental Information (from Jacobsen & Pratt, 2018).



**FIGURE 2** Dieback of chaparral reflected by the gray-colored shrubs in the Santa Monica Mountains, Ventura County, California, USA (photograph by Jon E. Keeley, 2016).

used remote sensing methods and aerial imagery to detect and verify chaparral dieback. To relate dieback to patterns of fire severity in the 2018 Woolsey and the 2017 Thomas Fires, we focused our study within the perimeters of these two fires. Our specific goals were to (1) quantify the spatial extent and the severity of chaparral dieback that occurred within the perimeters of the Thomas and Woolsey Fires, (2) assess the effects of potential drivers causing the dieback of chaparral vegetation, (3) examine the effects of prolonged drought on various chaparral plant communities to assess differences in vulnerability, and (4) evaluate the effect of vegetation dieback on the size of the Thomas and Woolsey Fires.

## STUDY AREA

This study was conducted in southern California, USA, on chaparral vegetation within the perimeters of the 2017 Thomas Fire (114,078 ha), which started on 4 December in Ventura County and burned for over a month in Ventura and Santa Barbara counties, and the 2018 Woolsey Fire (39,234 ha), which ignited on 8 November in Ventura County and burned for nearly 2 weeks in Los Angeles and Ventura counties (Cal Fire FRAP, 2020). These fires were located approximately 32 km from one another in the Transverse Ranges between the cities of Santa Barbara and Los Angeles. Both fires resulted from electrical power equipment failures (Cal Fire FRAP, 2020) that occurred during gale-force Santa Ana winds when live fuel moisture was extremely low due to the near-total lack of rain since June in both years. This contributed to much of the area within the fire perimeters being categorized as having moderate-to-high burn severity based on the Monitoring Trends in Burn Severity (MTBS) system (Eidenshink et al., 2007; Picotte et al., 2020).

The Transverse Ranges are a geologically complex group of east–west trending mountain ranges that separate southern California from the rest of the state. The terrain is rugged with steep slopes ranging from near sea level in both fire perimeters up to 860 m in the Woolsey Fire and 1880 m in the Thomas Fire. The soils are mostly shallow and rocky (Dibblee, 1982), and the vegetation consists primarily of coastal sage scrub, herbaceous annuals, and agricultural lands at lower elevations and various chaparral types at higher elevations. Chaparral associations include nearly homogenous stands of one species to mixtures of *Adenostoma fasciculatum*, *A. sparsifolium*, *Arctostaphylos glandulosa*, *A. glauca*, *Ceanothus crassifolius*, *C. cuneatus*, *C. megacarpus*, *C. oliganthus*, *C. spinosus*, *Cercocarpus betuloides*, *Heteromeles arbutifolia*, *Malosma laurina*, *Quercus agrifolia*, *Q. berberidifolia*, *Q. chrysolepis*, *Rhus integrifolia*, and *R. ovata*.

The climate is Mediterranean with winter rains and summer drought; average annual mean and maximum temperatures range from 13 to 17°C and from 18 to 25°C, respectively, over the area burned by the Thomas and Woolsey Fires (30-year normal from 1981 to 2010) (PRISM, 2020). The majority of precipitation occurs between the months of December and March and on average ranges from 414 to 1300 mm year<sup>-1</sup> across the elevational gradient of the area of the Thomas Fire and from 323 to 661 mm year<sup>-1</sup> in the area of the Woolsey Fire (PRISM data 30-year normal from 1981 to 2010) (PRISM, 2020). Beginning in December 2011, the state slipped into a drought that in the northern portion peaked 3 years later in 2014 but continued on in southern California until March 2017 (Figure 1).

## METHODS

### Spatial extent and severity of chaparral dieback

Remote sensing has been used to assess the impacts of drought on chaparral communities (Dong et al., 2019), estimate changes in biomass (Schrader-Patton & Underwood, 2021), and monitor changes over time (Coates et al., 2015; Gillespie et al., 2018). To assess the spatial extent of chaparral dieback that occurred following the drought of 2011–2017, and prior to the Thomas and Woolsey Fires, we used Landsat 4/5/7/8 surface reflectance NDVI data (normalized difference vegetation index) data to quantify changes in green biomass or living vegetation. NDVI is a commonly used metric of green biomass that is calculated as the difference between near-infrared (NIR) and visible or red (RED) reflectance divided by their sum (Weier & Herring, 2000):

$$\text{NDVI} = (\text{NIR} - \text{RED}) / (\text{NIR} + \text{RED}).$$

The result is a continuous metric that ranges from –1 to 1, with 1 representing the highest attainable density of green leaves. Because chaparral is evergreen, we hypothesized that the difference in NDVI (dNDVI) across this time span would be a measure of vegetation dieback when other factors affecting changes in NDVI, including both natural and anthropogenic landscape disturbances, were removed from our sample.

Many vegetation types exhibit variation in NDVI values over the course of a year due to their life history traits and the seasonality of precipitation. Chaparral species, however, are evergreen with NDVI values that are relatively constant throughout the year. Therefore, we used the annual mean NDVI data for 2010, the year prior

to the drought, and the annual mean NDVI data for 2016, near the end of the drought and preceding the two fires. We acquired these data from a hub developed to automatically generate climate metrics to user-specified preferences at a 30-m resolution (Climate Engine, 2020). Since our focus was on chaparral dieback, we clipped the NDVI data to locations identified as chaparral within the Thomas and Woolsey Fire perimeters using the Existing Vegetation (Eveg) map product by Classification and Assessment with LANDSAT of Visible Ecological Groupings (CALVEG, 2020). In order to ensure that changes in live vegetation between 2010 and 2016 were not fire-related, we also excluded all chaparral that fell within wildfire or prescription burn perimeters from 2006 forward based on the fire perimeter data in the Cal Fire database (Cal Fire FRAP, 2020). Finally, we calculated the dNDVI by subtracting the 2016 NDVI data from the 2010 data in order to create a map of potential chaparral dieback within the two fire perimeters. The outcomes were either negative, indicating an increase in live vegetation; zero, indicating no change in live vegetation; or positive, indicating a decrease in live vegetation.

This map of potential chaparral dieback, however, did not account for other possible disturbances. To address this, we selected plots where we evaluated other factors that might also contribute to changes in NDVI, using National Agriculture Imagery Program (NAIP) aerial imagery (NAIP, 2020). ArcGIS was used to randomly generate sample points within areas mapped as chaparral in the Thomas and Woolsey Fires, and 30-m buffers were created around each point to establish 0.28-ha plots. All overlapping plots were deleted, as well as plots with less than 100% chaparral based on the Eveg map. This left a total of 14,395 plots for evaluation. Each plot was assessed using the 2010 NAIP imagery to confirm that there was at least 75% chaparral cover prior to the drought and that there were no signs of human or natural disturbances, which included: clearance for development; defensible space fuel modifications; infrastructure such as roads, trails, and structures; and any recent fires that may not have been captured in the Fire and Resource Assessment Program data. The 4216 plots that did not meet these criteria were deleted. The remaining 10,179 were then reassessed using the 2016 NAIP imagery to verify that no new disturbance had occurred in the plots between our first assessment in 2010 and 6 years later in 2016.

After this assessment, there were 9322 plots remaining that were divided into two classes: those with dNDVI values  $>0$  that were verified as having vegetation dieback; and those with dNDVI values  $\leq 0$  where there was no change or an increase in vegetation. The plots having dieback were then divided by the sum of all plots

in order to get the percentage of plots that experienced chaparral dieback. Since our measure of vegetation dieback was based on the difference between the two indices, we could not express the amount of dieback that occurred within a given plot in terms of biomass lost or percentage of dead vegetation. Instead, we divided plots with chaparral dieback into four equal-sized classes or quartiles so that the severity of vegetation dieback could be quantified spatially across the area. These analyses were first performed for the two fires combined and then for the fires individually in order to assess whether there were differences in the amount and severity of chaparral dieback between the two fire areas.

## Drivers of vegetation dieback

To identify the most important drivers of vegetation dieback, we compiled a spatial database of mapped explanatory variables extending across the entire study region with both fires. We explored a number of topographic and soil variables because they are generally associated with the physiological limits of plants (Díaz de Leo'n-Guerrero et al., 2021; Dubuis et al., 2013; Thuiller, 2013) and may mediate the ability of plants to attain or retain water during long periods of drought (Okin et al., 2018). We obtained elevation from a 30-m resolution US Geological Survey digital elevation model and derived slope and aspect from the model using the spatial analyst surface tool in ArcGIS (NRCS, 2020). The aspect, which is a circular variable from  $0^\circ$  to  $360^\circ$ , was first converted into radians and then into two variables, northness (radians  $\times$  cosine) and eastness (radians  $\times$  sine), to avoid the problem of  $359^\circ$  being only one unit away from  $0^\circ$ . Available water storage capacity and minimum bedrock depth were acquired from the July 2020 Soil Survey Geographic Database (SSURGO) at a spatial resolution of 30 m, and atmospheric nitrogen deposition data were obtained from a 2002 map of total annual deposition of reduced and oxidized nitrogen (in kilograms of nitrogen per hectare per year) at a spatial resolution of 4 km (SSURGO, 2020).

In addition, we investigated a number of climate variables that mediate the extent and duration of plant stress during droughts. Since we were primarily interested in the spatial variation of climate across our plots, we acquired climate data as mean values over the time period from 1 January 2011 to 31 December 2016. Given that plant species distributions vary across climatic gradients, mean values tend to be used in order to capture that spatial variability (Franklin, 2010). We used Climate Engine (2020) to calculate the average values of maximum, mean, and minimum temperatures from monthly

data (PRISM, 2020) and the average values of actual evapotranspiration, climate water deficit, and soil moisture from Terra Climate (2020). Precipitation data were also derived from monthly data (PRISM, 2020); however, instead of calculating the mean we summed the data in order to get the total precipitation over the time period of interest. All climate variables were acquired at a 4-km spatial resolution.

Finally, we examined vegetation type to assess whether certain chaparral communities were more vulnerable to severe drought than others. Vegetation type was extracted for each study plot from the CALVEG Eveg map (CALVEG, 2020). These included various assemblages of chaparral that were named for the dominant species or were categorized as either a lower or an upper montane chaparral association with no single dominant. The US Department of Agriculture Forest Service CALVEG (2020) manual describes the lower montane chaparral mix as an alliance of lower elevation species that extend up to 1646 m in coastal locations and 2440 m in the mountains, which include *Adenostoma fasciculatum*, *Arctostaphylos* spp., *Ceanothus* spp., *H. arbutifolia*, *Q. berberidifolia*, *Q. chrysolepis*, *Rhamnus crocea*, *Rhus integrifolia*, and *R. ovata*. Upper montane mixed chaparral is an alliance of species that generally exists above 1280 m in areas among coniferous forests and woodlands. The most common species in this alliance include *Arctostaphylos* spp., *Ceanothus cordulatus*, *C. integerrimus*, *Chrysolepis sempervirens*, and others.

After compiling our database of explanatory variables, we extracted the mean values for each plot and calculated descriptive statistics for the two fires combined, as well as individually. We used ordinary least squares regression to evaluate the effect of explanatory variables on the dNDVI as a continuous linear response variable and multiple ordinary least squares regression to assess the combined effect ( $p < 0.05$ ). We used the Durbin–Watson statistic to evaluate the influence of spatial autocorrelation; values between 1.5 and 2.5 are considered to have low autocorrelation. To ensure that multicollinearity would not be an issue, we calculated correlation coefficients among all potential explanatory variables and considered those with coefficients  $\geq 0.7$  as highly correlated (Ratner, 2009). Actual evapotranspiration, climate water deficit, and soil moisture were highly correlated with one another; therefore, only soil moisture was used in the multiple regression model because it was least correlated with other variables and had the strongest bivariate relationship with the dNDVI. The mean temperature was highly correlated with both minimum and maximum temperatures, as well as elevation, nitrogen deposition, precipitation, and soil moisture, and thus was eliminated. After constructing the multiple regression model, we calculated

the variance inflation factor of each variable, to test for remaining issues of multicollinearity. Values less than 2.5 were considered free of multicollinearity (Johnston et al., 2018). Finally, we conducted an ANOVA with a Bonferroni post hoc test to identify the vegetation types most vulnerable to the drought. Results with  $p < 0.05$  were considered significant.

## Chaparral dieback and burn severity

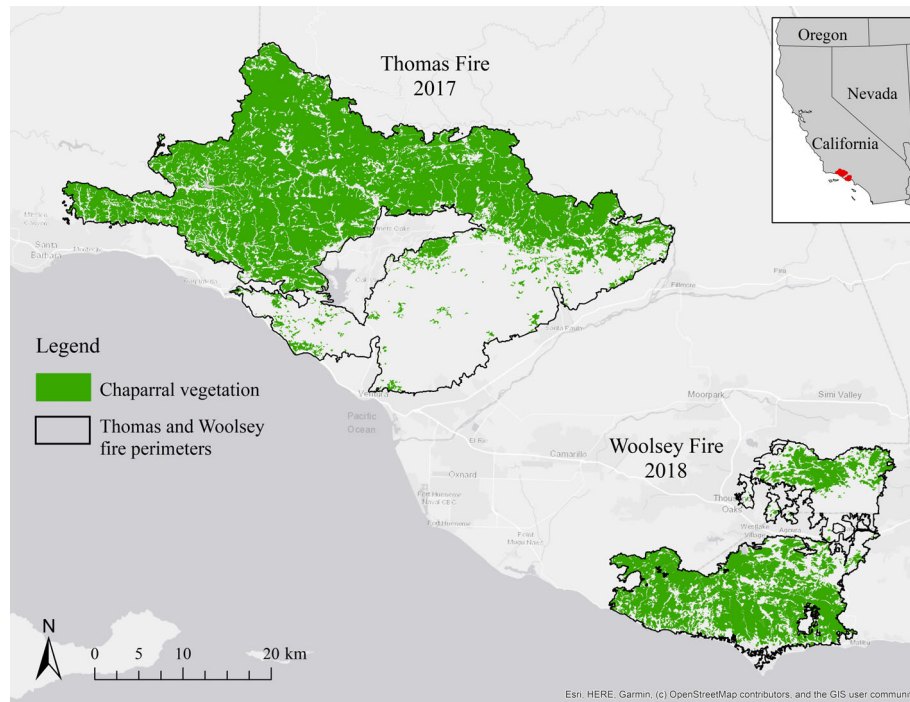
In order to evaluate the relationship between dieback and burn severity, we utilized the MTBS differenced normalized burn ratio (dNBR), at a resolution of 30 m, as our measure of burn severity (MTBS, 2020). The dNBR is the difference between normalized burn ratios prior to and immediately after a fire. We calculated the average burn severity for each plot and used ordinary least squares regression to evaluate the relationship between vegetation dieback (dNDVI) and burn severity (dNBR) as a linear response variable ( $p < 0.05$ ). The burn severity data were then divided into plots that experienced chaparral dieback (dNDVI  $> 0.0$ ) and those that did not (dNDVI  $\leq 0$ ) to determine whether they were significantly different using a  $t$  test for unequal variances ( $p < 0.05$ ).

## RESULTS

### Dieback patterns

Both the Thomas and Woolsey Fire areas were dominated by chaparral vegetation (Figure 3), with remaining wildland areas comprising summer-deciduous sage scrub, annual grassland, and oak woodland. Our study focused on just those plots dominated by chaparral. Comparisons between the two fires (Table 1) showed that chaparral in the Thomas Fire was concentrated at substantially higher elevations than in the Woolsey Fire and that climatic patterns were consistent with these differences in elevation. The average slope incline was steep, and the extreme slopes over  $40^\circ$  were present in both fires and likely would have contributed to high-severity fires.

Dieback was estimated by first measuring NDVI in 2011 and again in 2016 within both fire perimeters. We considered annual minimum, maximum, and mean NDVI and found little difference, so we utilized mean NDVI (Figure 4a–d). Potential dieback was estimated by subtracting NDVI in 2011 (Figure 4a,c) from NDVI in 2016 (Figure 4b,d). After using aerial photographs to subtract other disturbances, dieback was considered to be this modified dNDVI (Figure 5). Some level of dieback occurred in 93% of plots; however, there were marked



**FIGURE 3** Extent of chaparral vegetation within the 4 December 2017 Thomas Fire perimeter (114,078 ha) in Santa Barbara and Ventura counties and the 8 November 2018 Woolsey Fire perimeter (39,234 ha) in Los Angeles and Ventura counties, California, USA.

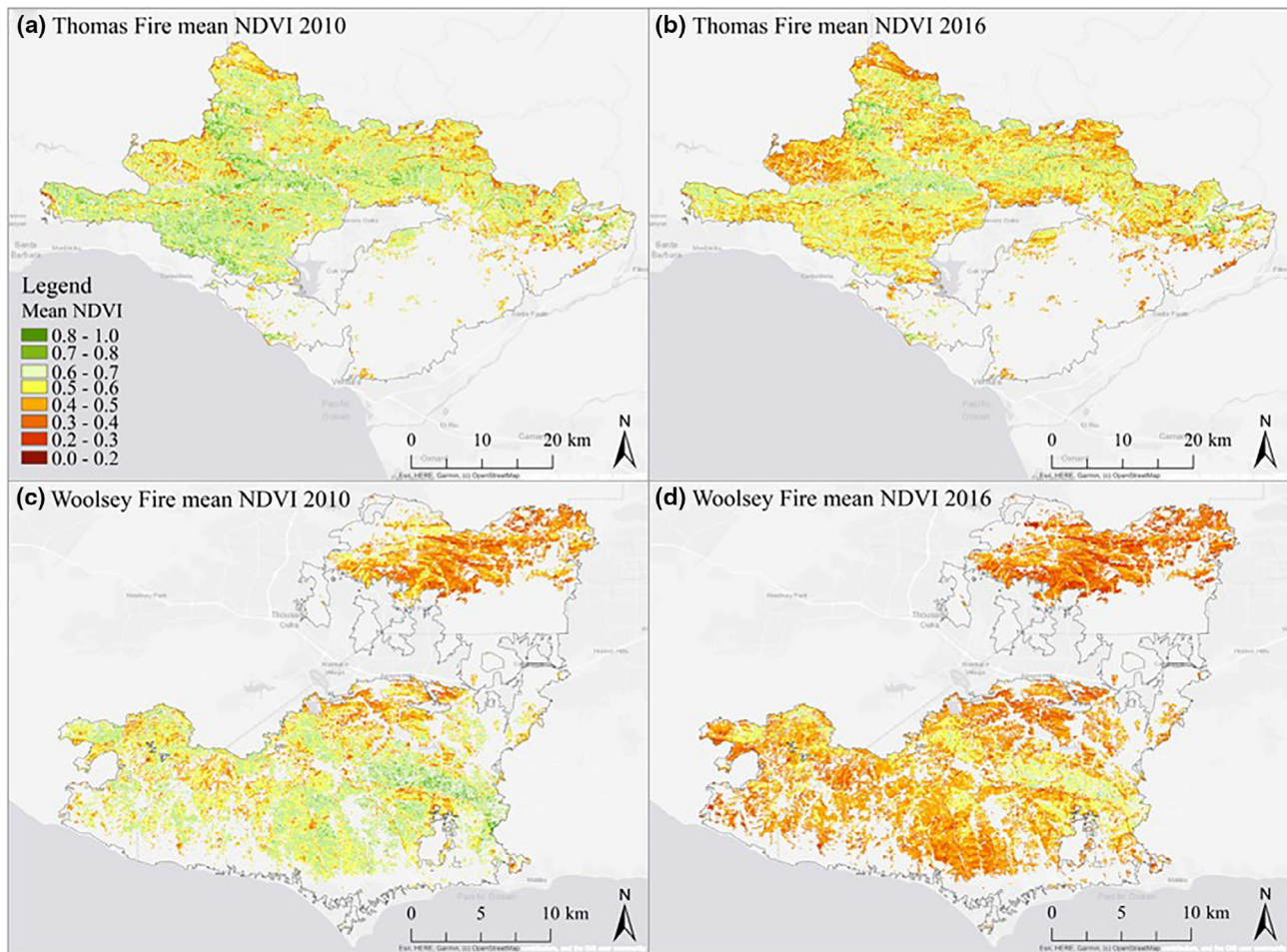
**TABLE 1** Descriptive statistics of potential drivers of chaparral dieback.

Explanatory variable	Thomas fire ( $n = 7395$ )			Woolsey fire ( $n = 1927$ )		
	$\bar{X}$	Min, max	SD	$\bar{X}$	Min, max	SD
Aspect						
Eastness	0.0	-1.0, 1.0	0.7	0.0	-1.0, 1.0	0.7
Northness	-0.2	-1.0, 1.0	0.7	-0.2	-1.0-1.0	0.7
Slope (degrees)	24.5	1.7, 47.4	7.7	22.0	3.0, 41.0	6.8
Elevation (m)	933.9	67.0, 1880.0	359.4	450.1	93.0, 860.0	151.9
Stand age (years)	39.0	14.0, 104.0	16.9	37.0	13.0, 88.0	9.1
Bedrock minimum depth (cm)	29.9	0.0, 152.0	23.0	23.1	0.0, 152.0	23.4
Available water storage (cm)	5.5	0.0, 28.5	3.4	5.5	0.0, 25.3	3.3
Climate water deficit (cm)	697.2	614.0, 781.0	30.6	767.2	733.0, 875.0	17.3
Precipitation total (cm)	358.5	208.0, 476.0	44.7	234.3	166.0, 276.0	24.9
Soil moisture (cm)	3.6	2.9, 4.1	0.2	2.5	1.5, 2.9	0.3
Actual evapotranspiration (cm)	241.9	205, 272	10.1	206.2	156, 226	13.0
Temperature ( $^{\circ}\text{C}$ )						
Maximum	22.6	19.0, 24.9	1.3	23.0	20.5, 25.5	0.8
Mean	15.5	13.3, 17.4	1.0	17.6	16.4, 18.7	0.5
Minimum	8.4	5.3, 12.3	1.8	12.2	109.0, 13.2	0.5
Nitrogen deposition ( $\text{kg N ha}^{-1} \text{ year}^{-1}$ )	7.1	2.2, 20.5	1.9	12.2	1.3, 13.8	1.4

Abbreviations: max, maximum; min, minimum.

differences between these two fires with the Woolsey Fire area experiencing a much greater percentage of plots with high dieback (Figure 6).

Vegetation comprising a mixture of many lower montane species was the main association in both fire areas (Figure 7). Plots dominated by a single taxon showed *Ceanothus* was



**FIGURE 4** Annual mean normalized difference vegetation index (NDVI) in chaparral vegetation within the perimeters of the 2017 Thomas Fire and the 2018 Woolsey Fire in 2010, prior to the start of the drought (a and c, respectively), and in 2016, after 6 years of drought and preceding the two fires (b and d, respectively). Decreased NDVI values indicate a lower density of green biomass.

markedly greater in the Woolsey Fire as was the case with *Adenostoma fasciculatum*, but in the Thomas Fire, there were greater *Q. berberidifolia*-dominated plots (Table 2). Dieback stratified by vegetation type revealed it was greatest in associations dominated by *Adenostoma* spp. and by *Ceanothus* and *Malosma/Rhus*, and these were significantly different from the lower montane mix, as well as *Q. berberidifolia*-dominated plots, and the upper montane mix.

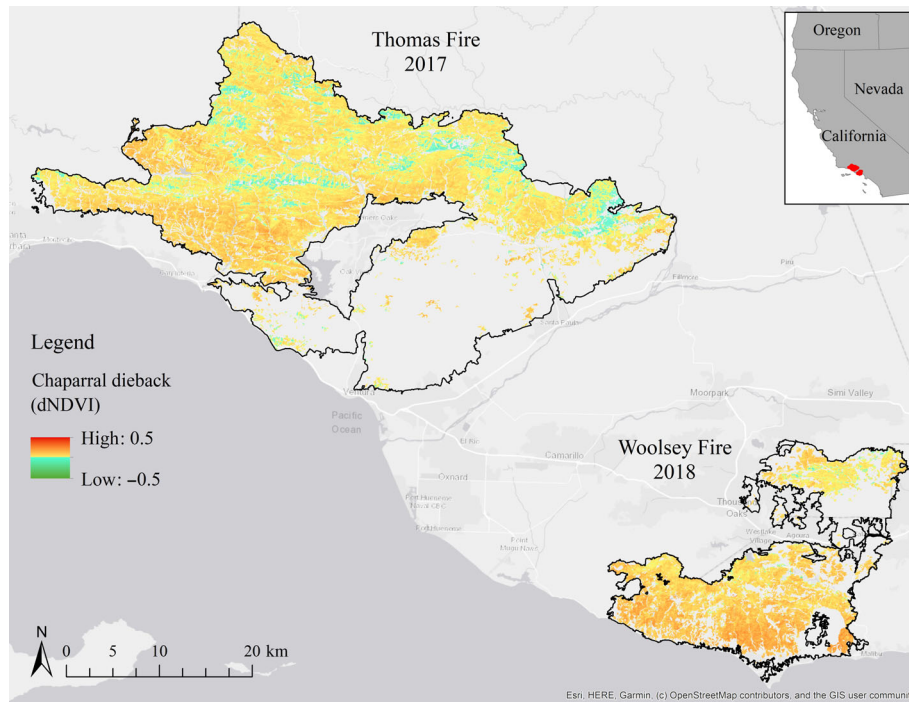
### Dieback drivers

Linear regression models evaluating the potential drivers of chaparral dieback revealed a number of significant relationships between explanatory variables and vegetation dieback (Figure 8). Elevation was the most influential driver explaining 37% of the observed variance in vegetation dieback, with dieback greater at lower elevations, and this was the only topographic variable that had a significant relationship. In particular, the lower

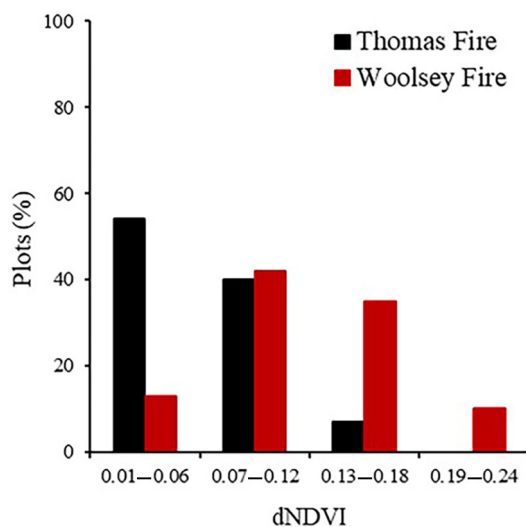
elevations with less soil moisture and higher maximum temperatures were correlated with dieback. When combined in a multiple regression model, these variables explained 45% of the variance in vegetation dieback (Table 3). In this model, elevation was the most influential predictor and was followed in importance by soil moisture and then maximum temperature.

With respect to elevation, there were marked differences between the type of chaparral and the dNDVI (Table 4). Plots that were dominated by *Ceanothus* spp., a *Malosma*- and *Rhus* spp.-dominated plot, for instance, incurred the most dieback, while plots with a *Quercus* spp. or upper montane chaparral mix had the lowest.

The most common vegetation type across nearly 70% of the plots, however, was a mix of lower montane chaparral species. The dieback in these plots, on average, was only about two thirds of the amount that occurred in vegetation types with the highest dieback. The most common vegetation type for both fire locations was a lower montane chaparral species mix; however, the



**FIGURE 5** Chaparral dieback within the 2017 Thomas Fire and 2018 Woolsey Fire perimeters, calculated as the difference in mean annual normalized difference vegetation index (NDVI) between 2010 and 2016 (difference in NDVI [dNDVI]), California, USA. Positive values indicate a decrease in live vegetation plus other human disturbances that reduced the amount of live biomass.



**FIGURE 6** Chaparral dieback (difference in normalized difference vegetation index [dNDVI]) in plots within the 2017 Thomas Fire and 2018 Woolsey Fire perimeters, stratified into four equal quartiles of increasing severity, California, USA.

percentage of plots having this vegetation type in the Thomas Fire was 72% in comparison with 51% in the Woolsey Fire (Figure 7). Most of the remainder of plots in the Woolsey Fire consisted of vegetation types that incurred higher amounts of dieback, whereas the

remainder of plots on the Thomas Fire were distributed across vegetation types with a range of dieback. We also noticed that several vegetation types only occurred in plots in one fire or the other. *Adenostoma sparsifolium*-dominated plots, for example, only occurred on the Woolsey Fire, whereas *Q. chrysolepis*-dominated plots and upper montane chaparral mixes, the vegetation types with the lowest dieback, only occurred in plots on the Thomas Fire.

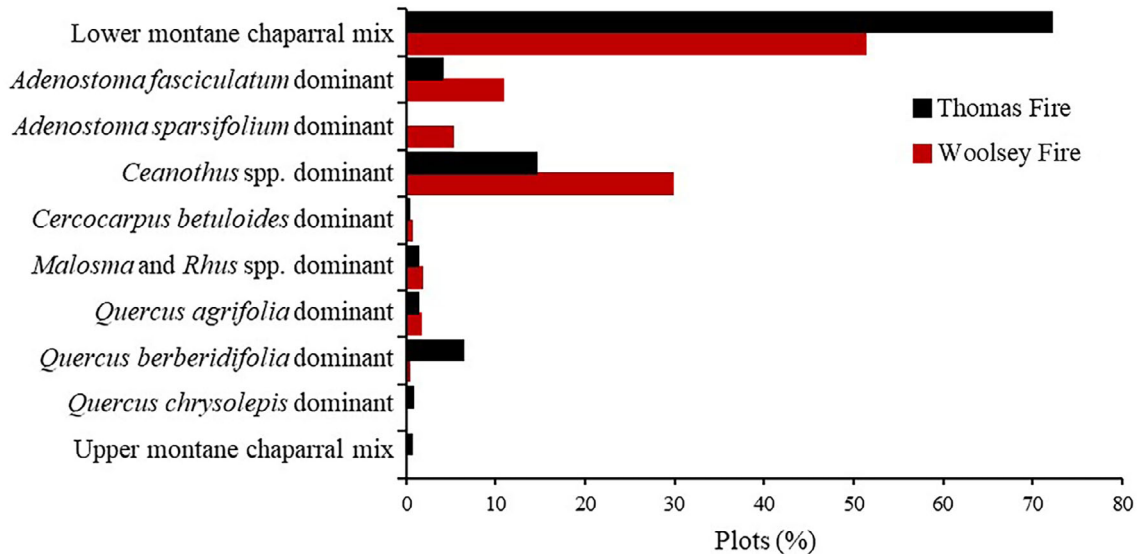
### Dieback and burn severity

For both the Thomas and Woolsey chaparral areas, burn severity ranged largely in the moderate–high range (Figure 9). The Woolsey Fire had very little low burn severity, and three fourth of the plots were in the low–moderate to moderate–high categories (Figure 10). The Thomas Fire had a much greater proportion of plots in low severity than observed in the Woolsey Fire.

Comparing patterns of burn severity with dieback showed that plots with dieback had significantly lower burn severity than plots without dieback (Figure 11).

Correlations between burn severity and dieback were evaluated with regression analysis for each vegetation type (Table 5). In all cases, the Durbin–Watson statistic was close to 2 (1.6–1.8), indicating a lack of spatial





**FIGURE 7** Chaparral vegetation type in plots within the perimeters of the 2017 Thomas Fire and 2018 Woolsey Fire, based on the USDA Forest Service Classification and Assessment with LANDSAT of Visible Ecological Groupings Existing Vegetation map, California, USA.

**TABLE 2** Chaparral dieback (difference in normalized difference vegetation index [dNDVI]) by vegetation type in plots within the perimeters of the 2017 Thomas Fire and the 2018 Woolsey Fire, California, USA.

Chaparral vegetation type	Chaparral dieback (dNDVI from 2010 to 2016)				
	<i>n</i>	$\bar{X}$	Min, max	SD	Post hoc
Lower montane chaparral mix	6302	0.067	−0.10, 0.25	0.05	c
<i>Adenostoma fasciculatum</i> dominant	500	0.086	−0.04, 0.24	0.06	ab
<i>A. sparsifolium</i> dominant	97	0.073	−0.01, 0.17	0.04	b
<i>Ceanothus</i> spp. dominant	1634	0.093	−0.03, 0.24	0.05	a
<i>Cercocarpus betuloides</i> dominant	23	0.069	−0.03, 0.21	0.06	bc
<i>Malosma</i> and <i>Rhus</i> spp. dominant	114	0.093	0.01, 0.21	0.05	a
<i>Quercus agrifolia</i> dominant	111	0.072	−0.06, 0.20	0.05	b
<i>Q. berberidifolia</i> dominant	464	0.041	−0.08, 0.18	0.05	cd
<i>Q. chrysolepis</i> dominant	47	0.013	−0.04, 0.07	0.03	e
Upper montane chaparral mix	30	0.019	−0.03, 0.08	0.02	de

ANOVA,  $p < 0.001$

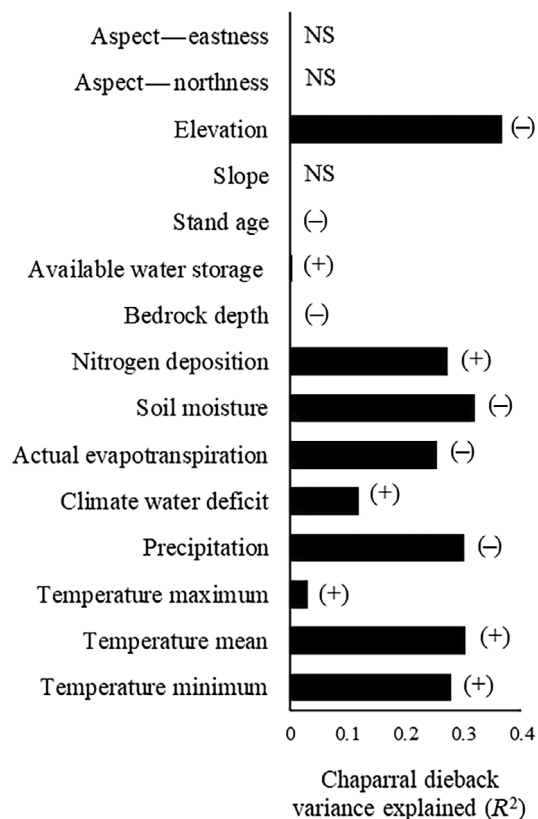
Note: Vegetation types were compared with ANOVA, and post hoc results are indicated by letters. Values sharing the same letters are not significantly different ( $p < 0.05$ ). *n* = number of plots.

Abbreviations: max, maximum; min, minimum.

autocorrelation. For both fires combined, all vegetation types had a significant regression coefficient between burn severity and dieback. For the Thomas Fire, this was consistently negative, indicating that with increased dieback, burn severity was lower. For the Woolsey Fire, the significant regression coefficients with the greatest variance explained were also negative.

## DISCUSSION

Within the perimeters of both the 2017 Thomas and the 2018 Woolsey Fires, there was substantial dieback tied to the 2011–2016 drought in southern California, revealed by the change in NDVI before and after the drought. The massive chaparral dieback that we report within the



**FIGURE 8** Chaparral dieback (difference in normalized difference vegetation index) explained by various topographic, edaphic, and climatic drivers with ordinary least squares regression. The direction of the regression relationship is indicated by (+) or (-) unless nonsignificant (NS).

perimeters of the Thomas and Woolsey Fires is likely a major contributor to the extraordinary size of these fires. There are several ways dieback could contribute to fire size. One possible factor could be the effect of dead biomass on increasing fire intensity, which would facilitate a rapid rate of fire spread, thereby complicating fire suppression activities. Fire intensity metrics require real-time measurements that were not available at the scale necessary for evaluating the impact of dieback. Although we had burn severity measurements, fire intensity and burn severity are often conflated, but they measure very different things (Keeley, 2009). The burn severity metric used in this study uses the MTBS system (Eidenshink et al., 2007; Picotte et al., 2020), which measures the difference in the live biomass before fire versus after fire; in short, it is a measure of vegetation dieback caused by fire. Thus, it is no surprise that drought-caused dieback as measured here should be negatively related to burn severity; less live biomass at the time of fire will produce less change in live biomass due to fire. Indeed, the highly significant negative relationship between drought-caused dieback and burn severity could be

**TABLE 3** Combined effect of explanatory variables on chaparral dieback (difference in normalized difference vegetation index) using ordinary least squares multiple regression.

Explanatory variable	Standard coefficient	<i>p</i>
Available water storage capacity	0.036	0.002
Bedrock depth	-0.041	<0.001
Elevation	-0.508	<0.001
Nitrogen deposition	-0.037	0.014
Precipitation	-0.048	0.003
Slope	0.087	<0.001
Soil moisture	-0.252	<0.001
Temperature—maximum	-0.169	<0.001
Temperature—minimum	0.041	0.008

Note:  $R^2 = 0.453$ ;  $n = 9322$ .

viewed as further verification of the validity of our remote sensing method of estimating vegetation dieback.

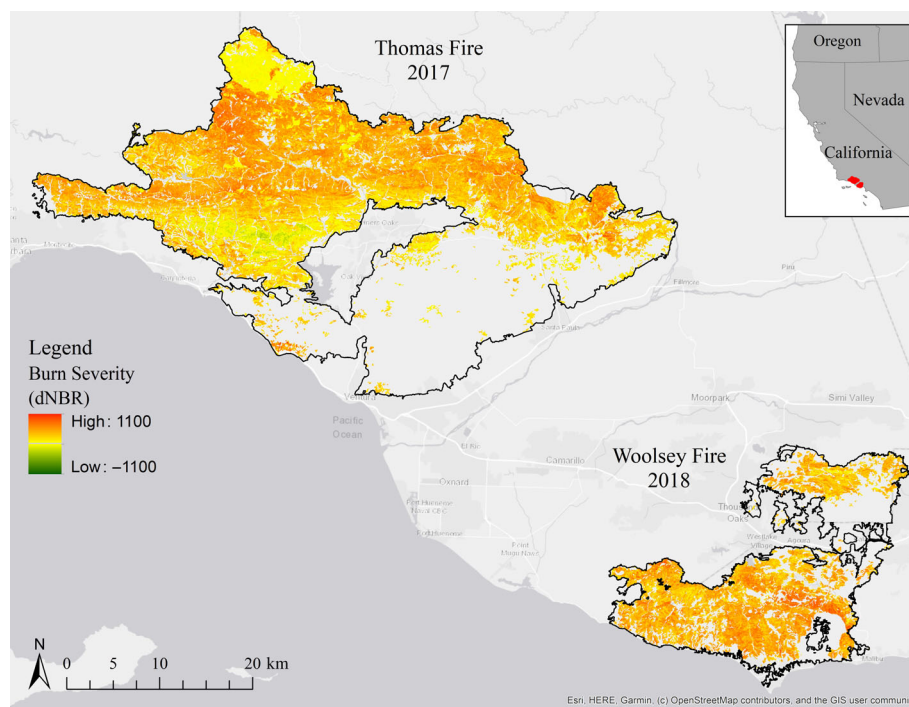
In chaparral shrublands, it has been noted that severe drought is correlated with large fires in the past (Keeley & Zedler, 2009), and this continues to the present (Keeley & Syphard, 2021). In the former paper, it was hypothesized that long droughts increased shrub dieback, and one of the factors contributing to larger fires was the dieback's impact on spotting behavior. In large fires, spotting ignition by firebrands is a significant mechanism of fire spread (Koo et al., 2010). Spotting comprises three sequential mechanisms: generation, transport, and ignition of recipient fuel. Dieback potentially contributes to all three. Dead fuels readily become embers, and much of this comprises large-diameter shrub stems that produce burning embers with a significant lifetime, which ultimately is a major factor in spotting distance (Albini, 1983). When dieback is coupled with high winds, as in the case of the Thomas and Woolsey Fires, long-distance transport is therefore likely. When embers land, ignition of recipient fuels is heavily dependent on landing on dead fuels (Pyne et al., 1996).

Models in Keeley and Zedler (2009) showed that firebrand ignitions and dispersal distance were strongly impacted by the volume of dead fuels. Not only do dead fuels increase the probability of a firebrand staying ignited during transport, but firebrands also fail to ignite unless they land on dead fuels. Increased dieback increases the probability of spot fires, which contribute to rapid-fire spread and ultimate fire size. In addition, models suggest that, as the volume of dead fuel increases, the distance of spotting increases (Figure 12). Reports from the Fire Management Officer on the massive 2003

**TABLE 4** Effect of elevation on dieback (difference in normalized difference vegetation index [dNDVI]) of different chaparral vegetation types within the 2017 Thomas Fire and 2018 Woolsey Fire perimeters, California, USA, using ordinary least squares regression.

Chaparral vegetation type	Chaparral dieback (dNDVI) variance explained by elevation							
	Thomas fire				Woolsey fire			
	<i>n</i>	<i>R</i> <sup>2</sup>	Direction	<i>p</i>	<i>n</i>	<i>R</i> <sup>2</sup>	Direction	<i>p</i>
Lower montane chaparral mix	5317	0.228	(–)	<0.001	985	0.170	(–)	<0.001
<i>Adenostoma fasciculatum</i> dominant	294	0.200	(–)	<0.001	206	0.279	(–)	<0.001
<i>A. sparsifolium</i> dominant	...	...	...	...	97	0.226	(–)	<0.001
<i>Ceanothus</i> spp. dominant	1064	0.209	(–)	<0.001	570	0.101	(–)	<0.001
<i>Cercocarpus betuloides</i> dominant	15	0.086	(–)	0.079	8	0.637	(–)	<0.001
<i>Malosma</i> and <i>Rhus</i> spp. dominant	83	0.004	(+)	<0.001	31	0.028	(–)	<0.001
<i>Quercus agrifolia</i> dominant	84	0.112	(–)	<0.001	27	0.220	(–)	<0.001
<i>Q. berberidifolia</i> dominant	461	0.254	(–)	<0.001	3	0.142	(+)	0.805
<i>Q. chrysolepis</i> dominant	47	0.183	(–)	<0.001	...	...	...	...
Upper montane chaparral mix	30	0.003	(+)	0.924	...	...	...	...

Note: The direction of the regression relationship is indicated by (+) or (–).

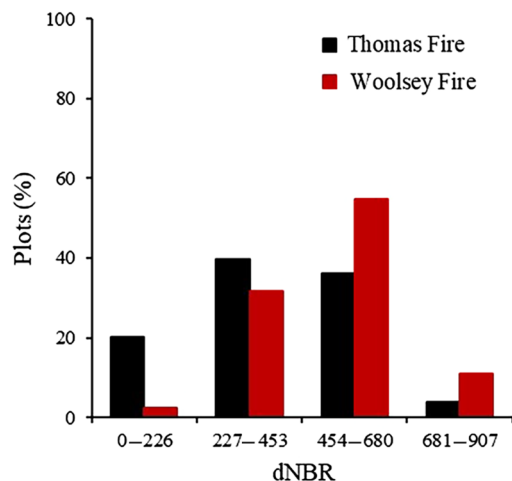


**FIGURE 9** Burn severity (differenced normalized burn ratio [dNBR]) in chaparral vegetation within the perimeters of the 2017 Thomas Fire and 2018 Woolsey Fire, California, USA.

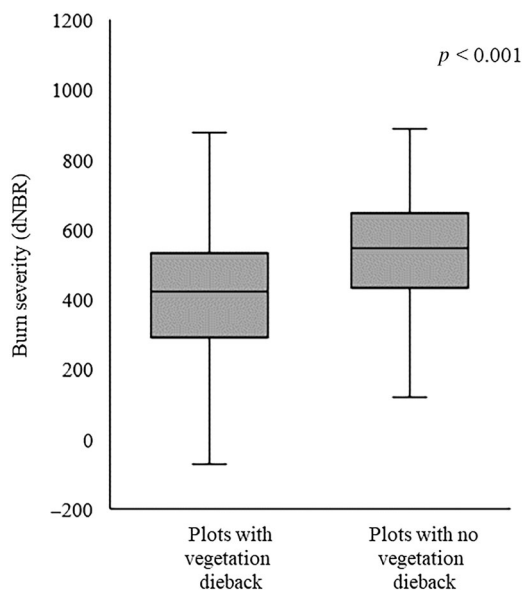
Cedar Fire attest to the critical role that spot fires played in the spread of this fire (Keeley & Zedler, 2009).

Recognizing the potential role of anomalous dieback episodes could be an important step in improving fire hazard/risk assessments (*sensu* Hardy, 2005). Fire danger indices are guided by current and recent weather indices

but use static fuel models where dead fuels are a function of stand age and do not incorporate drought-caused dieback (Woodall et al., 2005). This is problematical in chaparral since stand age is not a good indicator of dead fuels (Uyeda et al., 2015). Dead fuel moisture, unlike live fuel moisture, is closely controlled by ambient relative



**FIGURE 10** Burn severity (differenced normalized burn ratio [dNBR]) in plots within the perimeters of the 2017 Thomas Fire and 2018 Woolsey Fire, stratified into four equal quartiles of increasing severity, California, USA.



**FIGURE 11** Comparison of burn severity between plots that experienced vegetation dieback from 2010 to 2016 (difference in normalized difference vegetation index [dNDVI] > 0.0) and those that did not (dNDVI ≤ 0.0) for the 2017 Thomas Fire and 2018 Woolsey Fire combined using a Student *t* test for unequal variances.

humidity (Cohen & Deeming, 1985), and in the autumn, much lower dead fuel moisture levels can be produced than live fuel moisture levels. This is important for understanding drivers of large fires since live fuel moisture levels as low as 79% are associated with large fire events (Dennison & Moritz, 2009). In chaparral, the existence of high spatial and temporal variability results in a lack of timely fuel estimates for danger indices (Roberts et al., 2003). In southern California, typical burning indices often

perform poorly (Schoenberg et al., 2007) and there is evidence from other ecosystems that recent NDVI can improve fire risk predictions (Li et al., 2014).

Given the likely association between dieback and fire behavior, understanding factors driving chaparral dieback would be an important step in understanding fire risk. In most southern California vegetation types, dieback is greatest at lower elevations. This is of particular concern since at lower elevations, these vegetation types are closely juxtaposed with urban environments, where both ignition sources and assets at risk are most prominent (Radeloff et al., 2018; Syphard et al., 2007).

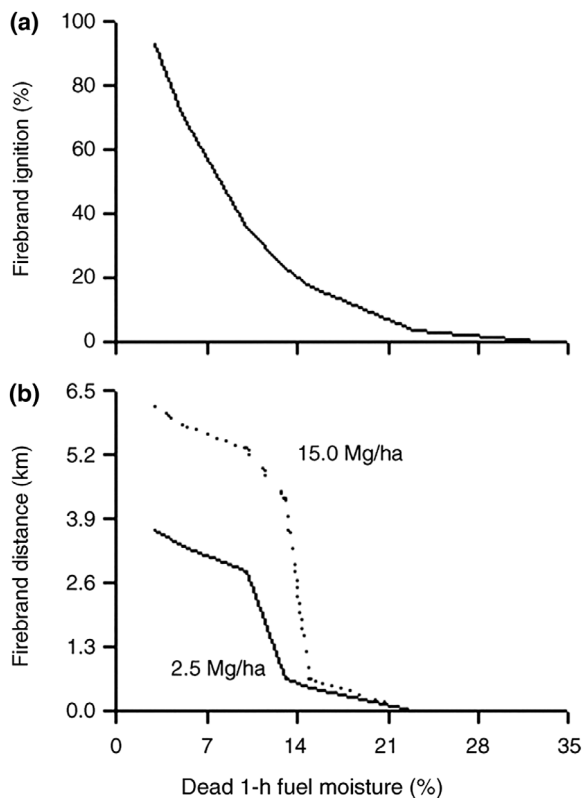
In comparing the two fire areas, plots dominated by *Adenostoma fasciculatum* and *Ceanothus* species were most abundant in the Woolsey Fire, which is where the most severe dieback occurred. These two associations tend to be most abundant at lower elevations, and this is consistent with the elevational range for that fire (Table 1) and partly explains the significant association between dieback and elevation. Rooting depth is strongly related to drought-associated mortality in chaparral (Jacobsen & Pratt, 2018), and plants with restricted root systems such as obligate seeding *Ceanothus* and *Arctostaphylos* species, as well as facultative seeding *Adenostoma* species, tend to experience greater dieback during droughts (Horton & Kraebel, 1955; Paddock III et al., 2013; Parsons et al., 1981; Venturas et al., 2016). What is interesting is that, although many of these same shallow-rooted, obligate seeding species have high stem cavitation resistance, in comparison with obligate resprouters, they still tend to be the ones most vulnerable to drought (Paddock III et al., 2013). Our findings, which show that the patterns of chaparral dieback on the landscape are heavily influenced by species composition, are consistent with results from other studies where the severity of vegetation dieback during extended drought periods was also found to be species-dependent (Coates et al., 2015; Venturas et al., 2016).

Deeper rooted resprouting species are generally less affected by drought. However, our results found that the obligate resprouting species *M. laurina* had dieback levels that were among the highest in the study and on par with the shallow-rooted *Ceanothus* species (Table 2). A study in our general study area done in 2014 near the beginning of the 7-year drought found that *M. laurina* was largely unaffected by the drought (Venturas et al., 2016). However, a later study found it suffered extensive dieback, and mortality was linked to the pathogen *Botryosphaeria dothidea* (Aguirre et al., 2018). It was hypothesized that unlike shallow-rooted *Ceanothus* where dieback is tied to short-term extreme drought, *M. laurina* dieback was tied to an extended duration of the drought, which enabled the pathogen to cause stem death. In short, chaparral species are differentially impacted by short-term extreme drought stress versus long-duration drought-caused stress.

**TABLE 5** Effect of dieback (difference in normalized difference vegetation index [dNDVI]) on burn severity (differenced normalized burn ratio [dNBR]) of different chaparral vegetation types within the perimeters of the 2017 Thomas Fire and the 2018 Woolsey Fire, California, USA, using ordinary least squares regression.

Chaparral vegetation type	Burn severity (dNBR) variance explained by dieback (dNDVI)					
	Thomas fire ( $n = 7395$ )			Woolsey fire ( $n = 1923$ )		
	$R^2$	Direction	$p$	$R^2$	Direction	$p$
Lower montane chaparral mix	0.264	(-)	<0.001	0.002	(+)	<0.001
<i>Adenostoma fasciculatum</i> dominant	0.086	(-)	<0.001	0.009	(+)	<0.001
<i>A. sparsifolium</i> dominant	...		...	0.039	(-)	<0.001
<i>Ceanothus</i> spp. dominant	0.409	(-)	<0.001	0.016	(-)	<0.001
<i>Cercocarpus betuloides</i> dominant	0.039	(-)	<0.001	0.021	(-)	0.020
<i>Malosma</i> and <i>Rhus</i> spp. dominant	0.399	(-)	<0.001	0.009	(+)	0.001
<i>Quercus agrifolia</i> dominant	0.188	(-)	<0.001	0.041	(-)	<0.001
<i>Q. berberidifolia</i> dominant	0.161	(-)	<0.001	0.602	(+)	0.534
<i>Q. chrysolepis</i> dominant	0.021	(-)	<0.001	...		...
Upper montane chaparral mix	0.000	(-)	<0.001	...		...

Note: The direction of the regression relationship is indicated by (+) or (-).



**FIGURE 12** BehavePlus 4 model results on probability of firebrands igniting (a) and spotting distance during wind-driven surface fires for two levels of dead fuel volume, using a high-load dry-climate shrub S5 fuel model and a wind speed of  $80 \text{ km h}^{-1}$  (from Keeley & Zedler, 2009).

The deep-rooted *Q. berberidifolia* was much more common within the Thomas Fire (Figure 7) and exhibited significantly less dieback than other chaparral associations (Table 2). This is likely a factor in the greater proportion of plots with more limited dieback (Figure 6).

## CONCLUSIONS

A 7-year drought in southern California was a likely factor in the extensive dieback of chaparral shrublands. Examining NDVI before the drought and near the end was used to estimate the extent of chaparral dieback within the perimeters of two megafires that occurred shortly after the drought. Both the 2017 Thomas Fire and the 2018 Woolsey Fire had extensive areas of dieback, and it is hypothesized that this played a role in the size of these fires. There was a highly significant negative relationship between drought-caused shrub dieback and fire-caused dieback as measured by fire severity, further supporting this remote sensing technique for assessing drought-caused dieback. Since dead fuels are likely to increase fire intensity, it raises concerns about the widely held assumption that fire severity is a surrogate measure for fire intensity. In addition, the impact of this short-term dieback event, which potentially contributed to fire spread, raises concerns that fire hazard models, which depend on static fuel models, need revision.

Following this drought, there were two megafires in the region and current climate projections predict that there will be longer, more intense, and more frequent droughts in the 21st century (Cook et al., 2014; Griffin & Anchukaitis, 2014; Seager et al., 2007), including in southern California (Cayan et al., 2010). Given the potential for prolonged drought to result in vegetation dieback that, in turn, can promote more extensive wildfire and related cascading ecological and human impacts, it is important to understand the conditions under which these effects are occurring. Lower elevation chaparral associations appear to be most vulnerable, and this should be a concern to fire managers in regions subjected to extended droughts.

## AUTHOR CONTRIBUTIONS

Jon E. Keeley, Alexandra D. Syphard, and Teresa J. Brennan conceived the idea, designed the study, and participated in the analysis. Teresa J. Brennan prepared the database and the final images.

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## CONFLICT OF INTEREST


The authors declare no conflict of interest.

## DATA AVAILABILITY STATEMENT

Data (Brennan-Kane & Keeley, 2022) are available from the US Geological Survey ScienceBase: <https://doi.org/10.5066/P91LIW2P>.

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