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# How drought-induced forest die-off alters microclimate and increases fuel loadings and fire potentials

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**Abstract.** Forest die-offs associated with drought and heat have recently occurred across the globe, raising concern that associated changes in fuels and microclimate could link initial die-off disturbance to subsequent fire disturbance. Despite widespread concern, little empirical data exist. Following forest die-off in the Northern Jarrah Forest, south-western Australia, we quantified fuel dynamics and associated microclimate for die-off and control plots. Sixteen months post die-off, die-off plots had significantly increased 1-h fuels (11.8 vs 9.8 tonnes $ha^{-1}$) but not larger fuel classes (10-h and 100-h fuels). Owing to stem mortality, die-off plots had significantly greater standing dead wood mass (100 vs 10 tonnes $ha^{-1}$), visible sky (hemispherical images analysis: 31 vs 23%) and potential near-ground solar radiation input (measured as Direct Site Factor: 0.52 vs 0.34). Supplemental mid-summer microclimate measurements (temperature, relative humidity and wind speed) were combined with long-term climatic data and fuel load estimates to parameterise fire behaviour models. Fire spread rates were predicted to be 30% greater in die-off plots with relatively equal contributions from fuels and microclimate, highlighting need for operational consideration by fire managers. Our results underscore potential for drought-induced tree die-off to interact with subsequent fire under climate change.

**Additional keywords:** climate change, dieback, litter, radiation, relative humidity.

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## Introduction

Die-off associated with drought and heat events has now been reported from a wide range of forest types across all forested continents (Allen *et al.* 2010). Die-off events are expected to continue as changes in the frequency, intensity and duration of extreme episodes related to global climate change continue to modify growing conditions for forests globally (Intergovernmental Panel on Climate Change 2014). Consequently, concern regarding tree mortality under climate change is growing (Allen *et al.* 2010; Martinez-Vilalta *et al.* 2012; Allen *et al.* 2015). Whereas some regions may become hotter and wetter, Mediterranean-climate-type regions, among others, are predicted to become hotter and drier (e.g. 4–6°C increase in temperature, 20–40% reduction in rainfall; Diffenbaugh and Field 2013). Such changes, superimposed onto existing challenges related to biodiversity conservation and land-use change, make Mediterranean-climate-type regions important systems for broadening our understanding of forest responses to climate change and how to manage them.

The consequences of drought-triggered forest die-off can be diverse, multiscale and complex, including responses such as species compositional changes (Allen and Breshears 1998), reduction in wildlife habitat (Lindenmayer *et al.* 2012), formation of novel communities (Anderegg *et al.* 2012*a*), changes to the carbon cycle (Hicke *et al.* 2012), alterations in nutrient and water balances (Adams *et al.* 2012; Cregger *et al.* 2014), changes to near-ground solar radiation (Royer *et al.* 2010), changing fuel characteristics and fire behaviour (Guarín and Taylor 2005) and changes in microclimatic conditions (Royer *et al.* 2011; Adams *et al.* 2012). Near-ground microclimate influences not only facilitative effects of trees on understorey plants but directly impacts processes above and below the soil surface, such as evaporation rates and associated partitioning of evapotranspiration (Huxman *et al.* 2005), nutrient cycling

(Padien and Lajtha 1992) and soil respiration rates (Raich and Tufekcioglu 2000). If tree die-off occurs at sufficiently large scales, associated shifts in microclimate (Royer *et al.* 2011) can impact important land surface-to-atmosphere feedbacks including fluxes of energy, carbon and water (Adams *et al.* 2010; Royer *et al.* 2010).

Beyond the direct impacts of drought-driven forest die-offs, interactions between climatic conditions and forest health and abiotic and biotic disturbances, such as fire and pests and pathogens, may be critical in determining how climate change expresses its long-term effects on forests. Linked disturbances, where the likelihood, intensity, or extent of a second disturbance is impacted by the first, are currently receiving a great deal of attention (Buma 2015; Gower *et al.* 2015). For instance, the link between widespread conifer mortality from bark beetles and subsequent fire potential (Donato *et al.* 2013) and fire behaviour (Harvey *et al.* 2013) continues to be a topic of concern for forest managers and policy-makers. The degree to which drought- and heat-induced forest die-off leads to linked disturbances has been widely mentioned but few examples quantifying the effect have been documented, particularly for non-conifer forest types (but see Gower *et al.* 2015). The amount, severity and duration of elevated surface fire potentials, for example, from forest die-off represent important knowledge gaps in understanding consequences of linked disturbance interactions and subsequent ecosystem processes (e.g. plant survival and recruitment) (Buma 2015).

In fire-prone Mediterranean climate-type ecosystems, altered fuel characteristics are particularly significant if hotter and drier conditions are anticipated (Mouillot *et al.* 2002; Resco de Dios *et al.* 2007). Forest die-off should alter the spatial distribution of fuels (vertical and horizontal) as tree foliage falls to the forest floor, as well as the moisture content of the fuels themselves via increased incident radiation (higher temperature, lower relative humidity) and elevated exposure to wind. Although drought-induced forest die-off is likely to affect the fuels complex (composed of different fuel strata as defined by the fuels characteristic classification system (FCSS); Sandberg *et al.* 2001) in some way, there is a paucity of studies on this relationship, particularly in highly fire-prone forests where fire spread is driven by surface litter and vegetation (Luke and McArthur 1978). Consequently, it is critical to quantify the relationships between drought-induced forest die-off and alterations to fire potentials for long-term, sustainable forest management.

During a 30-year precipitation downturn, and the driest (2010) and one of the hottest summers (2010–11) on record (Bureau of Meteorology 2011*a*, 2011*b*), multiple prominent vegetation types in the Mediterranean climate-type region of south-western Australia experienced mass forest and woodland die-off events (Department of Environment and Conservation 2011; Matusick *et al.* 2012). Within one of the most extensive forest types, the Northern Jarrah Forest, an estimated 16 500 ha were impacted, with >70% tree crown dieback (Matusick *et al.* 2013, 2015). We sought to address whether one disturbance (drought) might set the stage for a linked disturbance via fire, the most commonly occurring disturbance in the region. To that end, we quantified changes in fuel loads and obtained supplemental measures of microclimatic conditions following drought-induced forest die-off within the Northern Jarrah Forest. Based on measured surface fuels and microclimate (temperature, relative humidity and wind speed) and coupled with long-term climate records, we modelled potential fire behaviour to assess changes in rates of spread and the relative contribution of fuels vs microclimate. Such information contributes to informed fire management and aids in predicting drought–fire interactions in this fire-prone region with an extensive human–wildland interface. Specifically, we aimed to characterise die-off and control plots with respect to: (i) surface and standing fuels, along with supplemental characterisation of associated canopy structure and microclimate (potential near-ground incoming solar radiation, temperature, relative humidity and wind speed) following a forest die-off, and (ii) predicted fire behaviour associated with the die-off impacts.

## Materials and methods

### Study area

The Northern Jarrah Forest, in the South-west Botanical Province of Western Australia (Fig. 1), covers an area of 1 127 600 ha (Havel 1975), and ranges in form from a tall open and dry sclerophyll forest in the north to a tall closed forest in the south (Dell and Havel 1989). The present study focussed on upland areas, dominated by the trees *Eucalyptus marginata* Donn ex Sm. (jarrah) and *Corymbia calophylla* (R. Br.) K. D. Hill & L. A. S. Johnson (marri). The Northern Jarrah Forest overlies Archaean granite and metamorphic rocks capped by an extensive lateritic duricrust forming a plateau with an average elevation of 300 m, interrupted by occasional granite outcrops in the form of isolated hills (Churchward and Dimmock 1989). The climate is Mediterranean-type, with cool wet winters and most (~80%) rainfall falling between April and October (Bates *et al.* 2008) and a seasonal drought that may last from 4 to 7 months (Gentilli 1989). There is a strong west-east rainfall gradient across the forest, ranging from >1100 mm $year^{-1}$ on its western edge (Darling Scarp) to ~700 mm $year^{-1}$ in the east and north (Gentilli 1989). Further details of the study region, climate events, sites and their selection are described in Matusick *et al.* (2013) and Brouwers *et al.* (2013).

Beginning in February 2011, following 9 days over 35°C (Bureau of Meteorology 2011*a*), and after one of the driest years since comparable records (Bureau of Meteorology 2011*b*), tree crowns in the Northern Jarrah Forest began to rapidly discolour and die (see Matusick *et al.* (2013) for further details). This process continued through May 2011, resulting in discrete patches of nearly complete canopy loss. The winter rainfall period following the forest die-off was marked by above-average precipitation.

### Site selection

As part of a long-term study of drought impacts on forest dynamics, 20 patches affected by the forest die-off were selected randomly from a population of affected patches ($n = 236$) identified during an aerial survey conducted at the cessation of the die-off period (May 2011). The die-off patches were located in an area of Northern Jarrah Forest 80 × 40 km wide (Fig. 1; see map in Matusick *et al.* 2013). The population of 236 patches ranged from 0.3 to 85.7 ha in size (Matusick

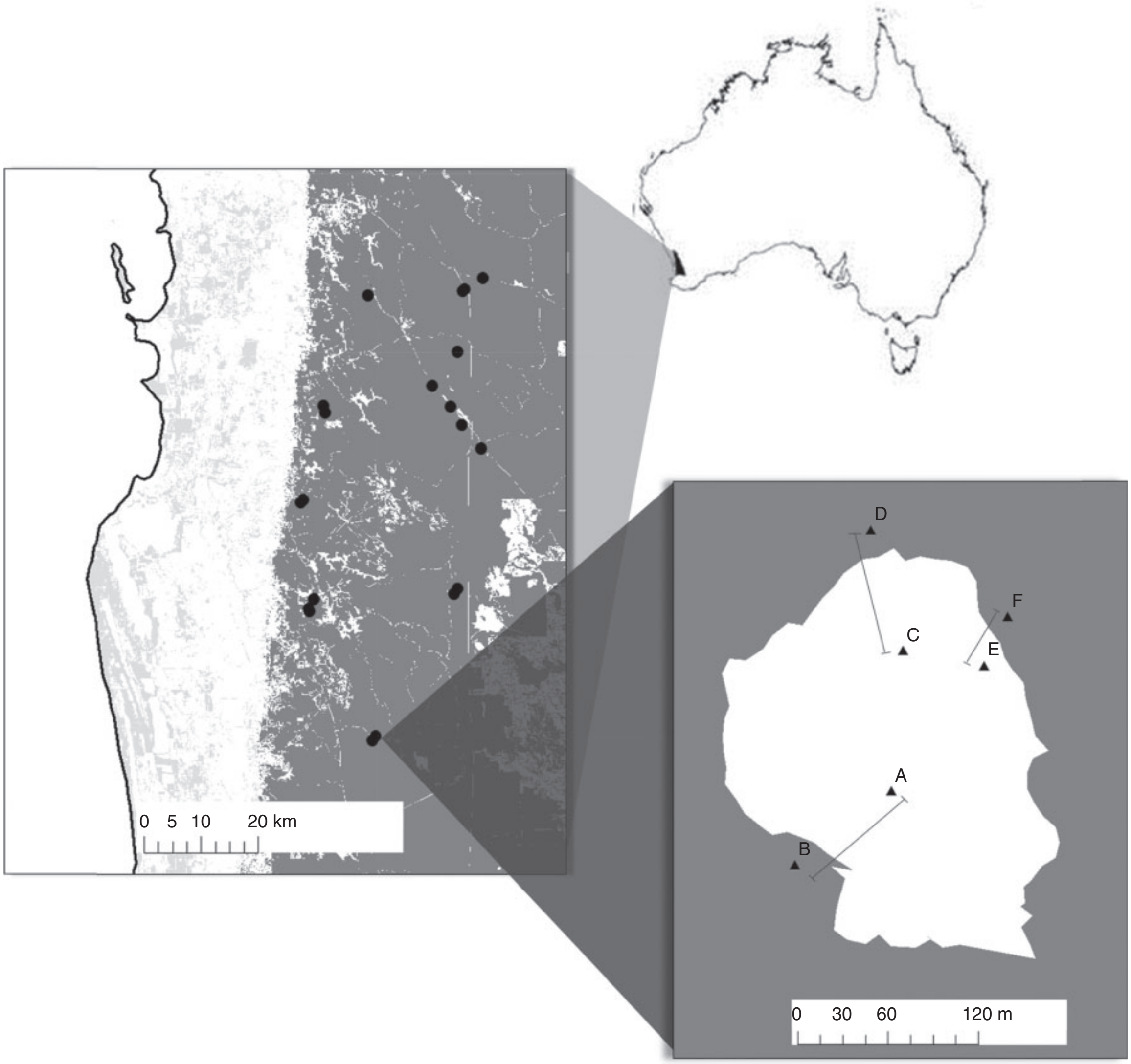


**Fig. 1.** Locations of the 20 study sites (black dots) within the Northern Jarrah Forest (dark grey shading), each containing three sets of paired die-off and control plots (black triangles) (map detail).

*et al.* 2013) with the 20 selected patches spanning 0.37 to 16.81 ha. The severely affected forest canopy (hereafter referred to as die-off), defined as >70% tree crown die-off, was delineated using a differential Global Positioning System (GPS) (Pathfinder Pro XRS receiver, Trimble Navigation Ltd, Sunnyvale, CA). To sample the die-off patches, a series of three 6-m-radius circular plots were randomly selected on a $20 \times 20$-m grid using *fGIS* forestry cruise software (Wisconsin DNR–Division of Forestry, Madison, WI, USA). For each plot within the affected patch, a paired plot representing minimally impacted forest (hereafter referred to as control) was located 20 m outside the area of severe die-off at the shortest straight-line distance from its affected pair (range = 24–204 m, mean = $58 \pm 5$ m) (Fig. 1).

Close proximity of control and die-off plots meant that they were comparable in relation to their soils, topographic position and fire history. Plots were visited during June and July 2011, 3 months following the first observations of forest die-off, to measure the forest structure, severity and patterns of the die-off (Matusick *et al.* 2013; Ruthrof *et al.* 2015), and plots were revisited in July 2012, 16 months post-drought, to measure surface fuels.

### Stand measurements

At each of the 120 plots (20 patches $\times$ 6 plots per patch), the stand was broken into surveys of the overstorey (all vegetation >1 cm diameter at breast height (DBH)), understorey live vegetation and surface fuels. For the overstorey, all plants (dead and live) greater than or equal to 1 cm DBH (1.37 m above the ground) were measured to determine stand attributes including basal area (Table 1). Die-off plots were densely stocked with the predominant species, *Eucalyptus marginata*. Understorey and forest floor were sampled using 1-m$^2$ quadrats ($n = 6$ per plot) at azimuths of 45°, 105°, 165°, 225°, 285° and 345°, 2 m from plot

**Table 1. Summary of stand attributes (percentage of plots with presence of dominant canopy species, relative density (%), basal area (BA)(s.e.)), and percentage cover of understorey vegetation for die-off and control plots following drought-induced forest die-off in the Northern Jarrah Forest, south-western Australia**

$n = 120$ plots (60 die-off and 60 control plots). Data reproduced (with permission) from Matusick *et al.* 2013

| Stand attribute | Species | Die-off | Control |
|---|---|---|---|
| % of plots with presence | *Eucalyptus marginata, Corymbia calophylla* | 100, 62 | 100, 68 |
| Relative density (%) | *Eucalyptus marginata, Corymbia calophylla* | 74.2 (3.2), 18.4 (2.8) | 66.8 (3.5), 20.4 (2.8) |
| BA dominant canopy species ($m^2\ ha^{-1}$) | *Eucalyptus marginata, Corymbia calophylla* | 31.6 (3.5), 6.8 (1.9) | 37.3 (3.8), 8.9 (2.1) |
| Understorey (% cover of perennial plants) | | 46.7 (4.9) | 35.4 (4.0) |

centre. Within each quadrat, two observers made ocular estimates of percentage cover (in intervals of 5%) of perennial vegetation (woody subshrubs and perennial monocots such as *Lomandra* and *Lepidosperma* spp.). In addition, within each 1-$m^2$ quadrat, litter depth (cm) was measured as the height from the bottom of the litter layer to the top of the highest litter particle.

### Fuel measurements

To establish the amount of fine fuel accumulation within each plot following drought, fuel ground sampling was undertaken in each plot (~July 2012, 16 months post-drought). Within each plot, three 25 × 25-cm quadrats were sampled by collecting all dead, non-attached material down to mineral soil. Quadrats were placed at azimuths of 45°, 165° and 285°, 3 m from the plot centre point. Thus, in total, three fuel samples were collected from 120 plots, totalling 360 samples. Fuel samples were placed into labelled paper bags, dried at 80°C for 3 days and then inventoried by time-lag size classes: 0.0–0.6, 0.61–2.5 and >2.5–7.6 cm, corresponding to 1-, 10- and 100-h time-lag classes for woody materials respectively (Fosberg 1970). There were no pieces >7.6 cm within these samples.

Owing to logistical limitations, we quantified larger coarse woody debris (CWD) at 6 of the 20 patches ($n = 36$ plots, 18 die-off and 18 control plots) using a modified planar intersect method (Brown 1971, 1974). A 20-m transect was placed in each cardinal direction from the centre of each plot and each piece of fallen dead wood (>7.6 cm) intersected by the plane of the transect tape was measured: species, diameter, decay class (1–5 where 1–3 = sound and 4–5 = rotten) (Cline *et al.* 1980). To quantify standing dead biomass, 1-m belt transects were used on either side of the planar transect with species, diameter and decay class of all standing dead trees recorded.

To estimate biomass from our planar transect and tree survey data, we followed standard approaches of first estimating volume per hectare (Brown 1974; Harmon and Sexton 1996) using published allometric equations (Hingston *et al.* 1980), followed by application of published wood densities (http://www.fpc.wa.gov.au/, accessed 7 December 2015; *Eucalyptus marginata*: 0.67; *Corymbia calophylla*: 0.65; *Allocasuarina fraseriana*: 0.62; *Banksia grandis*: 0.56 tonnes/$m^3$) to obtain mass. The aboveground biomass for each living *E. marginata* and *C. calophylla* stem was estimated using DBH measurements and previously published allometric equations (Hingston *et al.* 1980); *E. marginata*: $\ln(DW) = -3.680 + 2.84 \times \ln(DBH)$, *C. calophylla*: $\ln(DW) = -3.370 + 2.74 \times \ln(DBH)$, where DW is dry weight (kg) and DBH is diameter at breast height (cm). For records with unknown species (i.e. highly decayed logs), we applied values for the dominant species (*E. marginata*; 80% of all individuals in stand surveys). To account for mass loss of standing and downed wood progressing through decay classes, we followed the suggestions of Harmon *et al.* (2011) whereby we multiplied live wood density by 0.95, 0.74, 0.51, 0.29, 0.22 for decay classes 1–5 respectively. These calculations yielded biomass estimates of standing and down wood across live and dead pools further stratified by size and sound or rotten decay condition.

### Supplemental stand microclimate and environment characterisation

To characterise shifts in the stand physical environment (solar radiation, canopy openness and microclimate), a supplemental and intensive microclimatic investigation targeted for summer conditions was undertaken. The intensive nature of the measurements (e.g. simultaneous data logging, measurement of light) precluded spatially extensive measurements. The patch (designated no. 214, 32°13′32.60″S, 116°7′59.40″E (6.7 ha)), with its associated control and die-off plots ($n = 3$ each), was selected from the larger population of 236 patches, and represented the typical conditions of the broader sample population (tree species, dominance (100% *E. marginata* dominance at all patches), basal area ($53.5 \pm 9.0\ m^2\ ha^{-1}$ vs $44.4 \pm 2.9\ m^2\ ha^{-1}$ at other patches), % canopy cover ($25.3 \pm 4.9\%$ vs $20.0 \pm 2.0\%$ at other patches)).

Within the three die-off and three control plots, the potential annual near-ground input of solar radiation (Direct Site Factor (DSF – a dimensionless value from 0 to 1 corresponding to 1–100% direct light penetration below the canopy and the equivalent energy watts per square metre ($W\ m^{-2}$) (Royer *et al.* 2010) and references therein)) was measured at 100-cm height by estimating canopy openness (% visible sky) using hemispherical photography. Hemispherical photos can be used to obtain estimates of the proportion of annual direct solar radiation for a cloudless year reaching a location with a specific canopy structure (Royer *et al.* 2010). Photos were obtained once with a camera (Nikon Coolpix 5400, Nikon, Melville, NY, USA) fitted with a 10.5-mm Nikkor fisheye lens (Nikkor FC-E9, Minato-ku, Tokyo, Japan) in each plot at five points: the centre of each plot and midway between the centre and edge of the plot (3 m to either side of the centre) in each of the four cardinal directions. Photos were obtained under sufficient cloud cover to obscure the sun such that digital processing allowed effective differentiation of canopy structure from sky, as verified by two analysts independently, using the program *Hemi-view Canopy Analysis* (Delta-T Devices Ltd, Cambridge, UK); photographic and analysis methods are similar those provided elsewhere (Breshears and Ludwig 2010; Royer *et al.* 2010).

Near-ground microclimate measurements (100 cm above the ground surface) were taken over a 3-day period during summer (16–19 January 2012) targeted to reflect mid-summer conditions in the three die-off and three control plots (each with five replicate points in each plot, total $n = 30$). Simultaneous data for temperature, relative humidity and wind speed were recorded in each plot using Kestrel 4500 pocket weather trackers (Nielsen-Kellerman, Boothwyn, PA). Data were continuously logged at 5-min intervals over a 72-h period.

### Statistical analyses

Our overarching objectives were to (1) quantify differences in stand and fuel attributes between die-off and control plots, particularly with regards to fuel structure; (2) obtain supplemental data to characterise change in the physical environment of the stands; and (3) model fire potentials to understand the relative contribution of fuels vs microclimate. To quantify the effect of drought on fuel attributes, we analysed data at the plot scale, which required a random effect of patch (i.e. 20 patches each with $n = 3$ die-off plots and $n = 3$ control plots) and fixed effects of drought and covariates of pre-drought basal area (available from original plot measurements; Matusick *et al.* 2013) and time since fire (years: range = 3–17 years, mean = 8.25) (Department of Parks and Wildlife records, available at http://firewatch-pro.landgate.wa.gov.au/home.php, accessed 7 December 2015). Analyses were carried out for 1-h, 10-h, 100-h and total fine fuels as well as litter depth. Where necessary, data were log-transformed to achieve normality of residuals; all data analysis was carried out using R (R Core Team 2011) with the lme4 package (Bates and Maechler 2011) applying additive linear mixed-effects models using S4 classes; we had no *a priori* expectation of interactions among the covariates and die-off impact. Modelling proceeding followed the suggestions of Zuur *et al.* (2009). Because the data structure of our physical measurements (direct site factor, canopy openness) was a simple contrast within one patch, we employed a simple *t*-test on the average in each plot (rather than patch) and present means and standard errors to suit the more limited design of having sampled only one forest patch and three plots of each condition. For these supplemental data, differences and detected effects are more limited in their inference yet offer indicative quantification of the change induced by the drought event.

The supplemental microclimatic data were averaged across plots within treatment ($n = 3$ plots with five loggers per plot) and visualised to present differences. Given the limited time period (as well as spatial sample), we chose not to evaluate the data statistically. Rather, these data provided an opportunity to explore differences among control and drought stands and provide empirical observations to parameterise fire behaviour models.

### Fire behaviour modelling

To assess changes in potential fire behaviour resulting from drought-induced die-off, we incorporated stand structural and microclimatic measurements into the most widely used fire behaviour prediction algorithms in southern Australia: the Forest Fire Danger Index (Noble *et al.* 1980). The Forest Fire Danger Index (FFDI) is the principal means by which fire danger is calculated in Australia, and the rate of spread equation (FFDI $\times$ fuel load, see Noble *et al.* (1980) for equations) is still widely in use. The FFDI ranges from 0 to 200 (0–11: Low–Moderate; 12–24: High; 25–49: Very High; 50–74: Severe; 75–99 Extreme; 100+: Catastrophic) and is calculated from a combination of current weather conditions (temperature, wind, relative humidity) and antecedent rainfall. Antecedent rainfall is converted to a drought factor derived from the Keetch–Byram drought index (Keetch and Byram 1968), obtained from the nearest long-term climate station (in the present study Perth Airport; see below). The rate of spread equation (Luke and McArthur 1978) and several subsequent refinements (e.g. Burrows 1994) are widely used for operational fire-management decisions though it is widely recognised that model estimates underpredict rates of spread and behaviour under high fire danger weather conditions (McCaw *et al.* 2008; Fontaine *et al.* 2012). The forests considered here possess open understoreys with little grass, with fire behaviour driven by 'activity fuels', which include 1-, 10- and 100-h fuels (dead material) and living branches, twigs and foliage in the understorey (Luke and McArthur 1978), which were included as fuel load in the equation. Coarse fuels and standing fuels are not considered because there is no evidence to support them contributing to fire spread in a meaningful manner.

By measuring both fuels and microclimate, we were able to estimate the relative contribution of each component alone and collectively to fire potentials. Further, we chose to evaluate this contribution across a range of fire danger conditions (Low–Moderate, High, Very High fire dangers). To accomplish this, we first parameterised models using observed fuel loads and the 95th percentile weather conditions from our own microclimate data in die-off and control plots. These conditions, when compared against the closest long-term climate station (Perth Airport, 37 km to the NNW) were Low–Moderate (FFDI ~5–6). Next, we obtained High and Very High FFDI values from Perth Airport representing the 50th and 90th percentile observations over the summer months for the period 1973–2010 (Clarke *et al.* 2013). Based on our empirical data, we then estimated the climate effect of the drought die-off as a percentage of the control and applied this to the High and Very High FFDI values. This then permitted us to estimate rates of spread due to climate alone (constant fuels), fuels alone (constant climate), or in combination across Low–Moderate, High and Very High fire danger conditions. Predicted rates of spread were then summarised and presented as an indicative reflection of how drought-triggered forest die-off may impact fire behaviour and the relative contribution of fuels vs altered microclimatic conditions. Actual fire behaviour, particularly under conditions with wind speeds >12.5 km $h^{-1}$ likely exceeds predictions (McCaw *et al.* 2008).

## Results

### Fuel measurements and additional site characteristics

Die-off plots had a significantly higher surface fuel loads than control plots in terms of 1-h fuels (Fig. 2*a*, Table 2, 11.8 vs 9.8 tonnes $ha^{-1}$). However, we did not detect any differences in

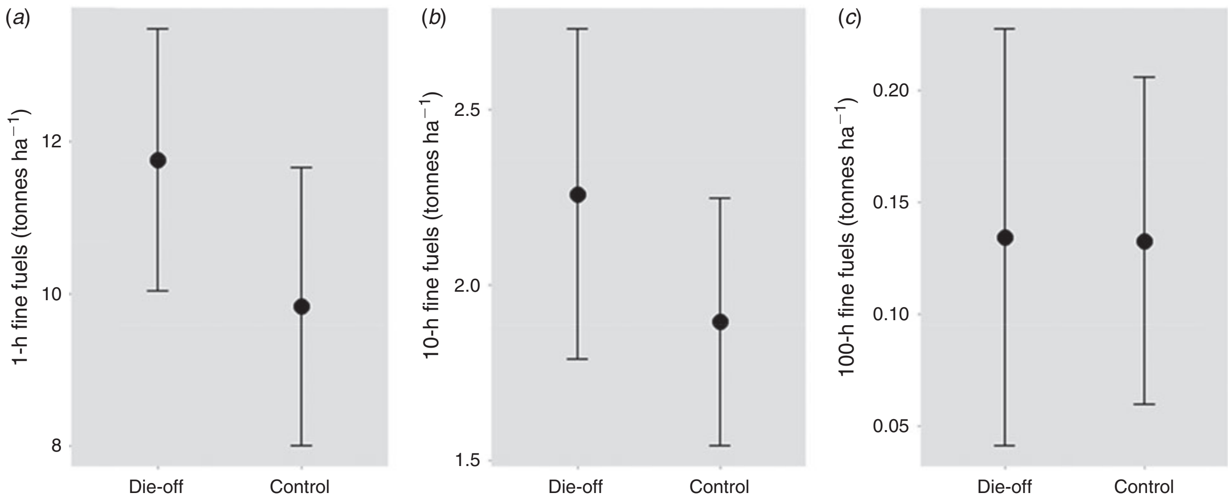


**Fig. 2.** Mean levels of fuels (*a*) 1-h; (*b*) 10-h; and (*c*) 100-h (tonnes ha$^{-1}$) (95% confidence intervals) in die-off and control plots following drought-induced forest die-off in the Northern Jarrah Forest, south-western Australia ($n$ = 360 samples, 180 die-off and 180 control).

**Table 2. Effect of drought-induced die-off on surface fuel dynamics in the Northern Jarrah Forest, south-western Australia**
$n$ = 120 plots (60 die-off and 60 control plots). Bold values indicate statistical significance at the 0.05 level

| Response | Unit of measurement | Die-off effect | s.e. | *t*-value | *P* value | Covariates in model (estimate; standard error) |
|---|---|---|---|---|---|---|
| 1-h fuels | tonnes ha$^{-1}$ | **2.2** | 0.6 | **3.6** | **0.001** | Time since fire (**0.6**; **0.2**), Basal area (**1.6**; **0.5**) |
| 10-h fuels | tonnes ha$^{-1}$ | 0.4 | 0.3 | 1.8 | 0.08 | Time since fire (0.01; 0.04), Basal area (**0.4**; **0.2**) |
| 100-h fuels | tonnes ha$^{-1}$ | 0.001 | 0.1 | 0.02 | 0.98 | Time since fire (0.01; 0.01), Basal area (0.01, 0.04) |
| All surface fuels | tonnes ha$^{-1}$ | **2.7** | 0.7 | **3.7** | **<0.001** | Time since fire (**0.6**, **0.2**), Basal area (**2.0**; **0.6**) |
| Litter depth | cm | **0.3** | 0.1 | **2.6** | **0.01** | Time since fire (**0.2**; **0.1**), Basal area (**0.3**; **0.1**) |

10- and 100-h surface fuels (Fig. 2*b* and 2*c*). The differences in 1-h fuels carried through to total fuel loads also being greater in die-off plots, as was litter depth (Table 2).

Coarse fuel (1000-h fuels and greater) mass did not differ between plot types by either rotten (die-off: 7.1 tonnes ha$^{-1}$; control: 10.7 tonnes ha$^{-1}$) or sound (die-off: 18.0 tonnes ha$^{-1}$; control: 31.7 tonnes ha$^{-1}$) with overlapping 95% confidence intervals (Fig. 3*a*). Standing dead fuels, however, were substantially greater in die-off plots compared with control plots (101 vs 11 tonnes ha$^{-1}$; Fig. 3*b*).

Analysis of hemispherical photos indicated that die-off plots had a percentage of visible sky that was 1.5 times greater than control plots (31% for die-off plots and 23% for control plots; $t = 6.8$, $df_{1,5}$ $P = 0.01$) and this was associated with a DSF of 0.52 in die-off plots versus 0.34 in control plots ($t = 4.0$, $df_{1,5}$ $P = 0.04$). The supplemental microclimatic data reflected effects of a removed canopy with die-off plots possessing greater temperature maxima (Fig. 4*a*) and lower temperature minima (Fig. 4*a*) along with commensurate shifts in relative humidity (Fig. 4*b*). Wind gusts tended to be stronger in die-off plots vs control plots (Fig. 4*c*).

### *Fire behaviour modelling*

Differences in microclimate (temperature, relative humidity and wind speed) between die-off and control stands led to empirical FFDI estimates of 5.8 and 5.1 respectively (Table 3). These values correspond to a Low–Moderate fire danger rating. Holding fuel mass constant (control stand values), these FFDI differences increased rates of spread by 12% (Fig. 5). In contrast, holding microclimate constant (control stand values) while varying fuels increased rate of spread by 16%. In concert, increased fuel quantities (i.e. increased fuel mass) and FFDI (climate) increased rates of spread ~30% in die-off plots (Fig. 5). Under Low–Moderate conditions, rate of spread was 0.11 vs 0.085 km h$^{-1}$ (die-off vs control), increasing to 0.35 vs 0.27 km h$^{-1}$ under High danger (50th percentile FFDI) and further to 0.75 vs 0.58 km h$^{-1}$ under Very High fire danger (90th percentile FFDI; Fig. 5). Values exceeding 0.14 km h$^{-1}$ are considered hazardous and require large-scale response to achieve suppression, and spread rates exceeding 0.24 km h$^{-1}$ preclude direct attack (Sneeuwjagt and Peet 1985).

## Discussion

This study is the first to quantify increased fuel loadings following drought-induced forest die-off in Australia, and builds on work of others from the northern hemisphere (Lloret *et al.* 2004; Bigler and Veblen 2011; Anderegg *et al.* 2012*b*; Hoffman *et al.* 2012). Fuel loading was found to be significantly higher in die-off plots compared with control plots, driven by greater 1-h fuels as well as large shifts from live to dead pools of standing biomass

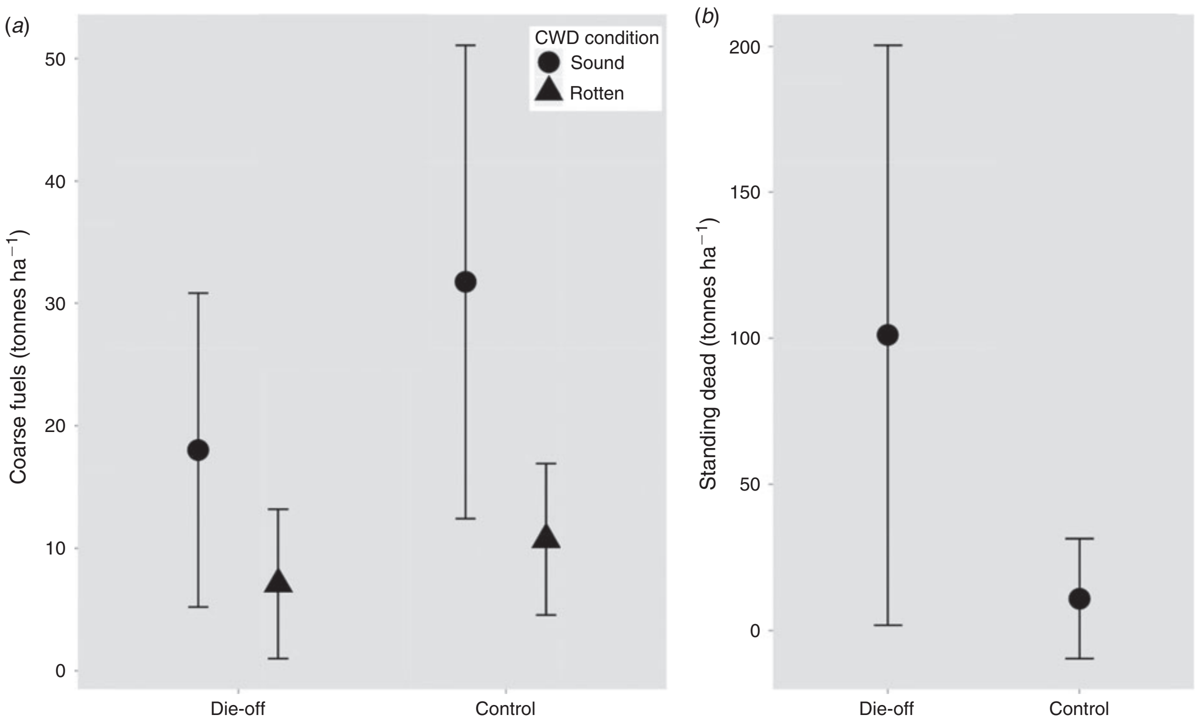


**Fig. 3.** Mean levels of fuels in terms of: (*a*) coarse woody debris (CWD), sound and rotten (tonnes ha$^{-1}$); and (*b*) standing dead (tonnes ha$^{-1}$) (95% confidence intervals) in die-off and control plots following drought-induced forest die-off in the Northern Jarrah Forest, south-western Australia ($n = 36$ plots, 18 die-off, 18 control).

(~90 tonnes ha$^{-1}$), which will slowly recruit to the surface, further increasing fuel loads at longer time scales. Thus, there has been a change in the quantity and spatial distribution (both vertical and horizontal) of biomass following the forest die-off in the Northern Jarrah Forest. The pattern of increased fuels following die-off in this study is consistent with, though lower than, observations made by Hoffman *et al.* (2012) in *Pinus ponderosa* forests in the south-western United States, where surface fuels were 2.5 times greater post mountain pine beetle forest die-off than unaffected areas. Bigler and Veblen (2011) found that immediately following tree death associated with drought and increased bark beetle activity in northern Colorado, United States, high loads of litter accumulated up to 10 times more for spruce, 5 times for fir and 2 times for pine compared with live trees. Gower *et al.* (2015) reported an increase of nearly 18 tonnes ha$^{-1}$ under trees and 2 tonnes ha$^{-1}$ under shrubs following canopy defoliation by an intense hailstorm in Western Australia. The differences in fuel load between our current study and other disturbance investigations are likely to be due to a combination of time since impact, forest types and causal factors. Irrespective, surface fuels drive fire potentials in nearly all fire-prone forests and increased fuel loading is one of the ubiquitous consequences of drought-related forest die-off (Lloret *et al.* 2004; Anderegg *et al.* 2012*b*) and has been highlighted as particularly concerning for Mediterranean climate-type ecosystems with continued climate change (Lloret *et al.* 2004).

Surface fine fuels generally increase rapidly following tree mortality, as aerial fine fuels (leaves and twigs) drop from the canopy onto the forest floor (Allen 2007). There was a clear pattern in our study, with finer (1-h) fuel loading being significantly higher in die-off plots. The pattern of higher surface fine fuels was seen in both the ground sampling and in the litter depth measurements and is also a reflection of the difference in percentage openness (and associated DSF) with severity of drought impact. Increased fine fuels were also observed in a study in the Amazon, where fuel became more abundant due to drought-related leaf and branch fall (Nepstad *et al.* 2001). In addition, a study in a mixed pine–oak forest in the south-eastern United States suggests that higher mortality rates in pine stands during more frequent and intense drought episodes may increase fuel accumulation (Klos *et al.* 2009). In both of these studies, the accumulation of fuel has been linked with more intense forest fires (Nepstad *et al.* 2001; Klos *et al.* 2009). McArthur (1962) and Peet (1965) suggest that this surface fine fuel is the most significant fuel variable affecting the behaviour of fires in eucalypt forests, though others differ in opinion (Gould *et al.* 2011).

Larger-sized fuel is available in two forms: first, if fallen, on the forest floor as surface CWD, and second, as standing dead material. In the current study, 1000+-h surface fuels were not significantly different between die-off and control plots. Work by Hoffman *et al.* (2012) demonstrated that nearly half of drought-killed trees had fallen after 5 years, resulting in fuels above recommended ranges for that forest type in 20% of plots. However, these authors undertook a slightly longer study than ours, in a different forest type, and thus further work will determine whether stand and fuel structure will succeed in a similar manner. In our study, die-off generated a substantial

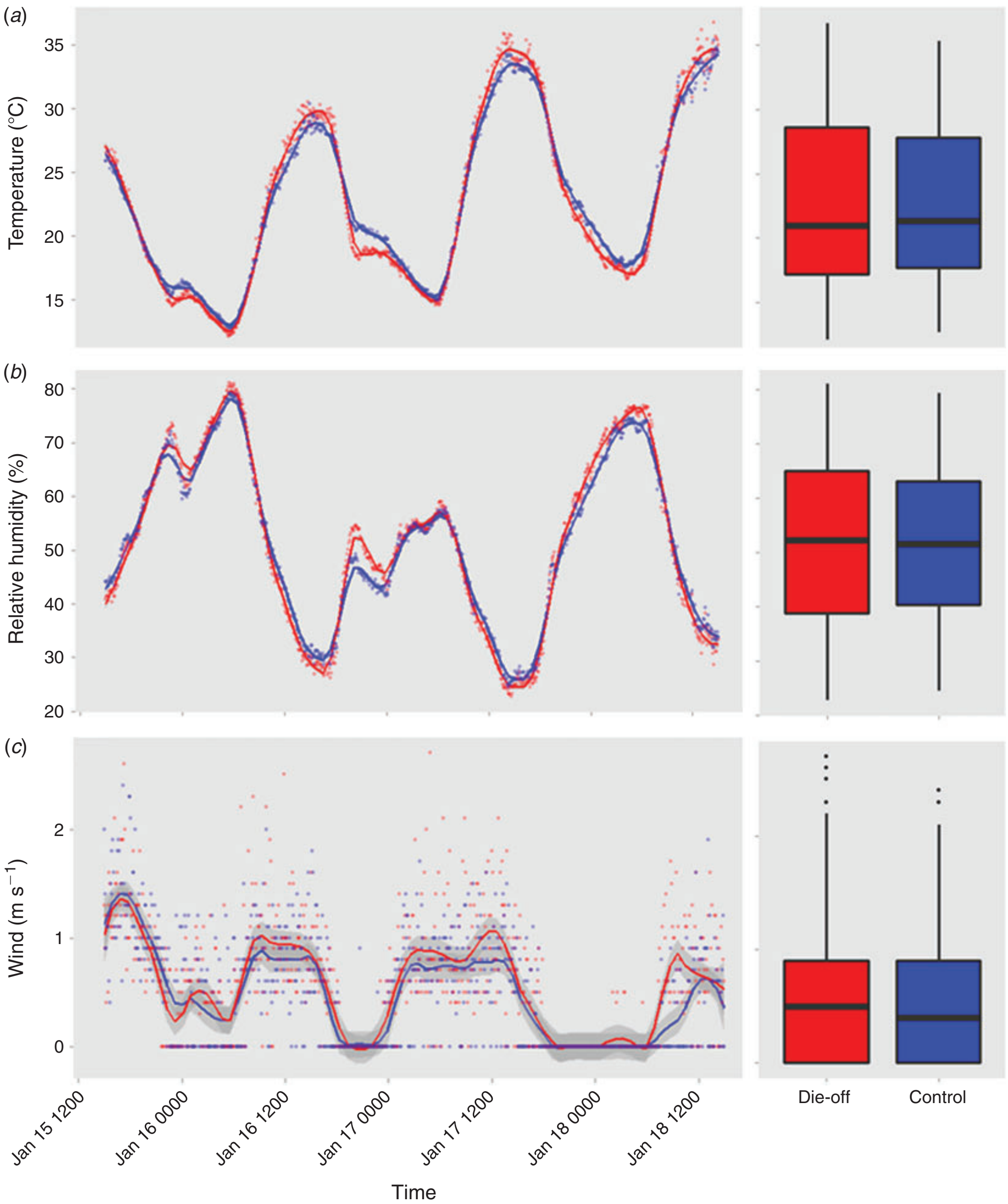


**Fig. 4.** (*a*) Temperature (°C); (*b*) relative humidity (%); and (*c*) wind speed (m $s^{-1}$) in die-off and control plots following drought-induced forest die-off in the Northern Jarrah Forest, south-western Australia ($n = 18$, 9 die-off and 9 control). Variables were measured over a 3-day period in mid-summer.

amount of dead standing material (100 vs 10 tonnes $ha^{-1}$ in control plots). Snag fragmentation and fall rates together with decay will determine contribution to future fire potentials and the temporal window of elevated hazard (if any). Eucalypts may not fall as quickly as other genera, owing to differential decomposition rates (Mackensen and Bauhus 2003) and thus the dead trees in the current study may contribute gradually to surface fuel loading over the coming decades.

Our supplemental microclimate data indicated an increase in openness and associated near-ground solar radiation in die-off patches. This has implications for properties such as soil evaporation rates, which determine soil moisture and associated plant available water, and hence seedling establishment (Royer *et al.* 2010). Although the supplemental microclimate measurements for temperature, relative humidity and wind speed were very limited in duration (3 days), they were targeted for mid-summer conditions associated with the fire season and were nonetheless suggestive of changes consistent with expectations that die-off plots should be hotter during the day, have lower relative humidity levels, and be more wind-prone. Such increased temperature ranges have also been seen in other studies when canopy is partially removed (Fontaine *et al.*

**Table 3. Climate inputs and modelled fire behaviour in die-off and control plots following drought-induced forest die-off in the Northern Jarrah Forest, south-western Australia under measured climate conditions in summer (January) 2012**

| Measure | | Die-off | Control | Difference (%) |
|---|---|---|---|---|
| 95th percentile micro-climate[A] | Temperature (°C) | 34.2 | 33.2 | 3 |
| | Relative humidity (%) | 26.4 | 27.9 | 6 |
| | Wind speed (m $s^{-1}$) | 1.2 | 1.1 | 9 |
| Fire behaviour | Fire Danger Index | 5.8 | 5.1 | 14 |
| | Rate of spread (km $h^{-1}$)[B] | 0.11 | 0.09 | 30 |

[A]Microclimate data collected 16–19 January 2012 represent 31st and 37th percentile (die-off and control respectively) of conditions based on daily maximum temperature for the month of January relative to nearest long-term hourly weather station (Perth Airport, Bureau of Meteorology, 1985–2012).
[B]Rate of spread calculated using climate values in table combined with field-measured fuel masses in control and die-off conditions.

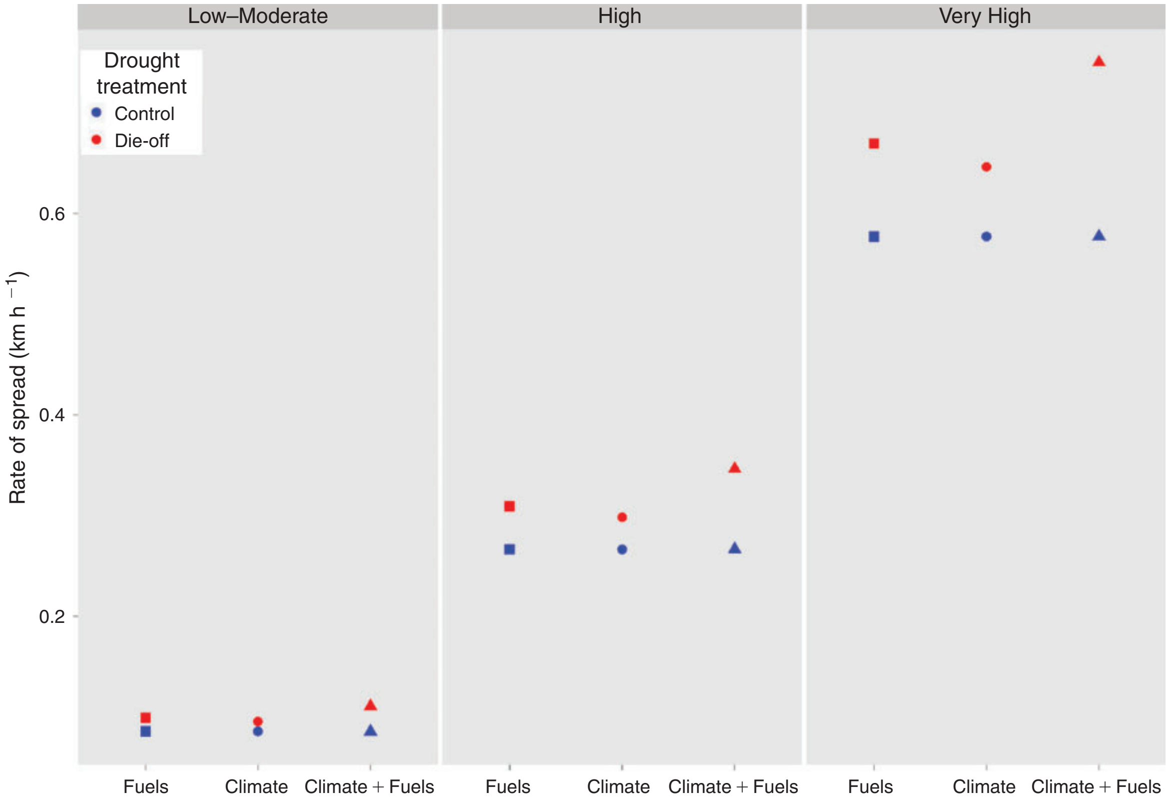


**Fig. 5.** Effect of drought-induced forest die-off on fire potentials (fire rate of spread. km $h^{-1}$) via increased fuel quantity, fire weather and combined across three levels of fire danger (Low–Moderate, High and Very High) in the Northern Jarrah Forest of south-western Australia. Fuels and climate each contributed approximately equal amounts to increased fire spread (~30% higher in die-off plots vs control plots).

2010). If the suggested changes in microclimate became substantial enough, like those documented for die-off in woodlands in the south-western United States, we would expect them to interact with numerous key abiotic and biotic processes at the local stand level, including evaporation partitioning, soil drying rates, decomposition, overstorey facilitation of understorey species, and effects of neighbouring trees on other trees (Breshears *et al.* 1998; Royer *et al.* 2010; Royer *et al.* 2012). In addition, there are implications for fire behaviour: the opening up of the canopy leads to the forest floor drying earlier in the season and could create novel habitats that are hotter, drier and windier and could lead to an extension of the fire season. Wind velocity, for example, is a key environmental factor that influences area burned by wildfires (Dimitrakopoulos *et al.* 2011). Such microclimatic changes, interacting with the predicted increases in mean daily temperatures in many regions, such as fire-prone Mediterranean type ecosystems, could have significant ramifications for fire frequency, extent and severity (Westerling *et al.* 2006; Chmura *et al.* 2011; Dimitrakopoulos *et al.* 2011; Enright and Fontaine 2014).

Modelled fire behaviour in the present study, combining empirical data on fuel loads, fuel characteristics and microclimate characteristics, has indicated that there could be non-trivial implications for fire behaviour in the short term. As we increased background fire danger from Low–Moderate to High and Very High, modelled rates of spread increased substantially, suggesting that fire encountering die-off patches would accelerate and increase in intensity and difficulty of control. The influence of particular fuel conditions on fire is still being debated (Bigler and Veblen 2011) and it will, no doubt, vary among ecosystems (Anderegg *et al.* 2012*b*). Guarín and Taylor (2005) propose that the increase in dead fuels caused by widespread tree mortality increases the risk and hazard of stand-replacing fires in forests that historically experience low- and moderate-severity surface fires. More open canopies and high loads of large surface fuels due to tree fall have been shown to significantly increase surface fireline intensities (Schoennagel *et al.* 2012). Our study shows that drought die-off and subsequent fire are potentially linked disturbances (in that one impacts the probability or severity of the next). Indeed, these could become compound interactions, conditioning the ecological response after a second event such as fire, as a crown fire in any remaining trees may depress recovery, consume canopy-stored seeds and potentially kill smaller individuals (e.g. case studies in Buma *et al.* 2013, Buma 2015).

## Conclusion

This paper presents data on the changes in fuel complexes following drought-induced forest die-off in a Mediterranean-type-climate eucalypt forest. We found increased surface fine fuel loadings and standing dead fuel, but not coarse surface fuel loading are altered with forest die-off, likely owing to low decomposition rates found in Mediterranean ecosystems. Incorporating these differences with supplemental short-term microclimate measurements into fire models resulted in predicted fire spread rates that were ~30% greater in die-off plots relative to control plots with equal contribution from elevated fuels and altered microclimate (temperature, wind). Our results indicate that the linked disturbances of drought-induced forest die-off and fire will likely have implications for ongoing forest health, which needs consideration from both forest managers and fire managers.

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