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Effects of 38 years of wildfires on tree density in the Blue Mountains, Australia

Abstract

© 2020 Ecological Society of Australia Forests are vital for biodiversity, carbon storage and ecosystem services, but can be potentially threatened by fires. Given the significance of forests and fire in a changing climate, research into the long-term effects of fire on forests plays an important role in understanding the global carbon cycle by the forests functioning as a large terrestrial carbon sink or source. In this study, we used aerial photography from 1975 and 2013 to count the change in the number of trees in 560 dry sclerophyll plots (40 × 40m) in the Blue Mountains of Australia. We analysed the relationship between the number of fires and severe fires in that period on the change in numbers of trees. We found that the average response was an increase of 1 tree per plot over 38 years. The number of fires had a small positive effect on tree numbers; plots with 2 or 3 severe fires had 1 and 2 extra trees, respectively, than those without fire. One exception was a severe fire in 2001 that did not show this positive effect, probably because it corresponded with extensive drought. Our findings suggest that number of forest canopy trees is resilient to the number of fires and number of severe fires.

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Effects of 38 years of wildfires on tree density in the Blue Mountains, Australia

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Abstract: Forests are vital for biodiversity, carbon storage, and ecosystem services, but can be potentially threatened by fires. Given the significance of forests and fire in a changing climate, research into the long-term effects of fire on forests plays an important role in understanding the global carbon cycle by the forests functioning as a large terrestrial carbon sink or source. In this study, we used aerial photography from 1975 and 2013 to count the change in the number of trees in 560 dry sclerophyll plots (40 × 40m) in the Blue Mountains of Australia. We analyzed the relationship between the number of fires and severe fires in that period on the change in numbers of trees. We found that the average response was an increase of 1 tree per plot over 38 years. The number of fires had a small positive effect on tree numbers; plots with 2 or 3 severe fires had 1 and 2 extra trees respectively than those without fire. One exception was a severe fire in 2001 that did not show this positive effect, probably because it corresponded with extensive drought. Our

findings suggest that number of forest canopy trees is resilient to the number of fires and number of severe fires.

Keywords: Wildfire; fire frequency; fire severity; resilience; number of trees

Introduction

Around the world, forests and the trees that comprise them are vitally important for carbon storage (Quéré et al., 2015), the rainfall they generate (Andrich and Imberger, 2013) and the biodiversity they harbor (Watson et al., 2018), and these three aspects are intricately linked. Disturbance to forests can threaten those values (Brienen et al., 2015), though extended lack of disturbance can also be detrimental, and disturbance that is neither too frequent or too rare probably promotes the highest biodiversity, as predicted by the Intermediate Disturbance Hypothesis (Connell, 1978). Disturbance is defined as a disruption that can change the ecosystem structure and composition by removing biomass from the system (e.g. fires) (Seidl et al., 2017). Wildfire is one of the disturbances that consumes large quantities of carbon (French et al., 2011, Keith et al., 2014) and has the potential to impact forest structure and biodiversity (Bradstock, 2010). Understanding the effects of fire on the number of trees and other plants is vital, both intrinsically and as a habitat for animals (Gill, 2012). There is evidence of increased forest fire frequency in recent decades as a consequence of climate change (Bradstock et al., 2014, Westerling, 2016), and further increases are predicted (Bradstock et al., 2012), which increases the imperative to understand its effects.

40 Australian forests are unusual globally because the most extensive types are dominated by re-
41 sprouting eucalypts that generally survive even severe fire (Enright and Fontaine, 2014, Gill,
42 2012). Nevertheless, non-lethal stem death (called top-kill) is common in these trees. For
43 example, Bennett et al., (2016) found 45% top-kill in large trees (>70 cm dbh) after the severe
44 Black Saturday fires (2009 wildfires of southeast Australia), and Fairman et al., (2017) reported
45 >50% top-kill in sub-alpine eucalypt forests. Similarly, Parnaby et al., (2010) found 44% top-kill
46 of hollow-bearing trees following a relatively low severity prescribed fire. Very severe fires can
47 also cause substantial whole tree death (Bennett et al., 2016, Prior et al., 2016). All of these
48 studies suggest a substantial reduction in carbon storage from a single fire, which has been
49 estimated to be between 8-14% of the pre-fire levels (Prior et al., 2016).

50 These studies may or may not be representative of the effects of fires in general. They all report
51 the effects of a single fire event, which does not necessarily inform us about the long-term
52 health of these forests. That is because trees usually recover and many habitat elements
53 regenerate after fire. It is more important to understand the effects of the fire regime over
54 decades rather than any single event. The regime experienced at a particular location is
55 comprised of fire frequency, severity, season, and type (prescribed vs. wildfire) (Gill et al.,
56 1987). The impact of fire on tree demography is complex and involves interactions between
57 regime components and forest type and pre- and post-fire environmental conditions (Coop et
58 al., 2016, Fernandes et al., 2015, Van Mantgem et al., 2013). There have been few studies of
59 the long-term effects of fire regimes on Australian forests (as opposed to single fires). One
60 example is Gordon et al., (2018) who measured the above-ground carbon in 107 plots in dry
61 tropical forest with different combinations of fire frequency, the severity of the most recent

fire, and rainfall over a 44-year period. They found that fire frequency and severity had no detectable effect on the aboveground carbon stock over that period. In another example, Mclean et al., (2015) found that nesting hollow abundance in temperate forests was in a balance between destruction and creation, both caused by fire. Nesting hollows are destroyed when a fire completely burns and consumes a tree, but fire also causes trunk damage that leads to hollow formation. Russell-Smith et al., (2012) examined the vegetation in 48 tropical savanna woodlands monitoring plots over 15 years and found that mature eucalypt counts were negatively related to the frequency of moderately severe fires, but not significantly so. Similarly, Lehmann et al., (2008) found that high fire frequency was related to a small reduction in tree canopy cover over a 40-year period in 72 savanna plots.

Environmental conditions other than fire could also contribute to any changes in number of trees. Drought patterns, especially, over any study period can cause a general increase or decrease in tree numbers across the forest (Mitchell et al., 2014, Nolan et al., 2020, Vicente-Serrano et al., 2010).

In this study, we compared the number of trees canopies in aerial photography from 1975 and 2013 in 560 plots with a size of 1600m² (area of each plot) in the Blue Mountains National Park, NSW Australia. We examined the influence of fire regimes on the change in number of trees over that 38-year period. We examined the following questions:

- I. What is the relationship between the total number of fires in the period of 1975 to 2013 and the changes in the number of trees in the eucalypt forest?
- II. What is the relationship between severe fires and the changes in the number of trees?
- III. Does the relationship between fires and number of trees differ among fire years?

Methods

The study area was the Blue Mountains National Park, a 270,000-ha protected area dominated by re-sprouting dry sclerophyll eucalypt forest 50 km west of Sydney (Figure 1). More than 100 eucalypt species occur in the park, and the common species include *Corymbia eximia*, *Eucalyptus squamosa* and *Agophora bakeri* (Hammill and Tasker, 2010). Elevation ranges from 50 to 1200 m and rainfall from 1000 to 1200 mm (Committee, 2008). The Blue Mountains National Park was established in September 1959, 16 years before our first observations. Wildfire activity is associated with episodic drought periods (Bradstock, 2010). Wildfires can burn as much as 20% of the forested area within the park in any year (mean ~5%) (Price and Bradstock, 2011).

Data

Historical fire boundaries from 1972 onwards were obtained from the NSW (New South Wales) Department of Planning Industry and Environment (DPIE, unpublished data). Fire severity mapping derived from satellite imagery was also from DPIE (Hammill and Tasker, 2010). In our study, we used the Hammill and Tasker, (2010) severity maps, but defined their class 3 (crown scorch) and class 4 (complete crown consumption) as high severity. All of the wildfires in the major fire years (1982, 1993, 2001, 2002, 2006) for the period 1982 to 2013 were mapped. The majority of the fires were mapped using the Δ NDVI ratio from Landsat satellite imagery (30 m resolution), though the SPOT satellite was used for one fire and the Δ NBR ratio (Miller and Thode, 2007) from Landsat imagery was used for another (Storey et al., 2016). The fire boundary and severity mapping data from DPIE have both been used in several studies of fire

behavior (Barker and Price, 2018, Bradstock, 2010, Bradstock et al., 2010, Hammill and Bradstock, 2006, Price and Bradstock, 2011, Storey et al., 2016).

The study area was surveyed using an aerial photographic survey conducted by the NSW Government in 1975 and again in September 2013, a 38-year interval (Figure 1). The 1975 imagery was monochromatic and was digitized to a spatial resolution of 60 cm (Figure 2a). The 2013 imagery was taken with a Leica ADS four-band digital sensor at a spatial resolution of 50 cm (Figure 2b).

We Used a GIS (geographic information system) layer of these 560 square plots (each with area of 1600 m² (40 × 40 m)) that were randomly placed across the study area. We applied two conditions: a) that they must be in areas mapped as dry sclerophyll forest in the vegetation map of New South Wales (NSW) (Forest type: Blue Mountain's Dry Sclerophyll forest, dominated by Sydney montane dry sclerophyll class of vegetation) (Keith, 2017), and b) they must be at least 200 m from human structures. From photographs for each year (1975 and 2013), the number of tree crowns was visually assessed (S.Rahmani) and the change in tree count was calculated as the difference of the two counts (2013-1975). To ensure consistency, one operator (S. Rahmani) assessed all photos by drawing a polygon around each canopy and counting them. The fire data was used to calculate the number of fires and the number of severe fires (involving crown scorch or crown consumption) fires for each plot.

Analyses

To examine the reliability of the visual assessment of the number of adult trees, one author (O. Price) counted the number of trees in 6 additional sites by counting the number of trees in a TLS (Terrestrial Laser Scanning) scan, and the other author (S. Rahmani) counted the exact same

127 6 sites using the air photos. The TLS data was collected as a single scan using a Leica LC30
128 mounted at 1.5 m height. The data was then clipped to a 40x40 m cube and the point cloud was
129 examined visually to identify individual trees. Then, we ran an interclass correlation reliability
130 test (using IRR package in R-Studio, <https://cran.r-project.org/package=irr>) to evaluate the
131 extent of agreement between the unit of observations for two different methods of tree
132 counting (TLS and visual counting). The results show that there is an 83% interclass correlation
133 agreement between two methods of observation. Shrout and Fleiss, (1979) have mentioned
134 that value above 75% is considered as a significant reliability score. Thus, it is reasonable to
135 assume that visual tree counting using air photos is consistent and represents a true estimation
136 of tree numbers.

137 We applied linear modeling to associate the change in tree count with the number of fires and
138 the number of severe fires. We tested whether the effects of the number of fires and severe
139 fires were linear or non-linear by fitting them as either continuous or categorical variables. We
140 also examined whether the effects of severe fires differed according to which year they
141 occurred in by fitting individual predictors of whether each plot was severely burnt in each of
142 the years 1982, 1993, 2001, 2002 and 2006, each as a binary variable and fitted collectively to
143 tree number change. Model selection was undertaken using Akaike's Information Criteria (AIC).
144 Models with $\Delta AIC < 4$ points of the top model (i.e. with the smallest AIC) were considered as
145 being strongly supported. Statistics were conducted using R 3.5.1 and the GLM package (Jenkins
146 et al., 2019). To examine effects of other environmental conditions such as drought pattern, we
147 obtained and plotted the monthly trace of the Standardised Precipitation and
148 Evapotranspiration Index (SPEI) for a location in the middle of the study area (longitude 150.25,

latitude -33.75) from <https://spei.csic.es/database.html> (Mitchell et al., 2014, Nolan et al., 2020, Vicente-Serrano et al., 2010). SPEI is a simple climatic water balance model applied extensively as a proxy of drought severity (Mitchell et al., 2014, Nolan et al., 2020). SPEI data from precipitation data was inspected visually to determine whether any of the fire years corresponded to major drought.

Results

The average number of trees in each of the 560 plots was 4.73 in 1975 and 5.75 in 2013 (Figure 3). Number of trees increased in 408 of these plots, remained the same in 91 plots, and decreased in 61 plots. An increase by 1 tree was the most common result but increases up to 4 trees and decrease up to 2 did occur (Figure 3). Figure 3 displays the detail underlying the numerical analyses shown in Table 1 and 2. The individual plots experienced between zero and six fires and up to three severe fires during the study period (Figure 4). The total number of trees tended to increase with the total number of fires, but with considerable overlap of distributions (Figure 5a). The linear modeling for fire frequency showed that the nonlinear categorical variable was superior to the continuous variable ($\Delta AIC = 32.6$), with fire frequencies of 0, 3, 4 or 6 fires having a positive effect on tree number change but 1 or 2 number of fires having a negative effect (though only 4 fires was statistically significant, Table 1).

Plots that had experienced 0, 2 or 3 severe fires were more likely to have a larger increases in number of trees than those with 1 severe fire (Figure 5b). Plots that had 2 or 3 severe fires had larger increases in number of trees compared to plots with no severe fires, and very few plots had decreases in number of trees (approximately 1 and 2 trees respectively, Table 2).

The year of the severe fire had a strong effect on the tree number change. When severe fires were modeled individually, the severe fire had a positive effect on the tree number change in most years (1982, 1993, 2002, 2006), but in 2001 effect was less significant in increasing number of trees (Figure 6, Table 3).

The SPEI in the study period 1975 to 2013 was not different from other periods in the record (Figure 7). However, 2001 occurred at the beginning of the most severe drought for the entire period.

Discussion

This study examined whether the fire regime has influenced the number of canopy trees over a 38-year period, which is only one aspect of tree response to fire. For example, the number, sizes and growth of juvenile trees were not examined. From 1975 to 2013, 72% of plots had a positive change in the number of trees and this trend was broadly consistent irrespective of the number of fires experienced in the plots, with a slight tendency for higher increases with more fires. From this, we can conclude that wildfire *per se* does not negatively affect the number of canopy trees in these forests. The occurrence of several severe fires (crown scorch or consumption) tended to increase the number of trees. In fact, the only negative fire effect detected was that plots that experienced high severity fire in 2001 experienced less increase in number of trees than others. Under severe fires, in fact, the number of trees was rarely reduced.

These results are perhaps surprising given that there is much evidence that there can be a substantial reduction in live tree stems and carbon storage from a fire event, particularly if it is

191 severe (Bennett et al., 2016; Fairman et al., 2017). Our results undoubtedly reflect the recovery
192 of trees between fires and given that we observed the number of large trees, not sizes of trees,
193 it is likely that smaller trees have grown large enough to be seen in the subsequent
194 photographs. Our results suggest that there is a balance between loss and replacement of
195 stems and that severe fires presumably cause more loss but also more replacement.

196 There are two possible mechanisms of replacement: the growth of smaller trees or the
197 recruitment of new trees. Recruitment of new trees is unlikely since the 38-year period is not
198 enough for seedlings or re-sprouts from root stock to grow large enough to be observed in our
199 method. It is much more likely that smaller trees in place before the fire have grown to replace
200 lost stems. Eucalypt trees in native forests (with similar rainfall) have typical diameter growth
201 rates of about 2 mm per year (Ngugi et al., 2015), which suggests that the trees in our study
202 would have grown by about 8 cm in diameter over the 38 years. However, juvenile eucalypts
203 have been shown to increase their growth rates in response to fire (Lewis, 2020, Werner, 2012),
204 presumably due to a flush of phosphorous released (Butler et al., 2018) and/or the increased
205 light and water available due to the removal of most of the ground and shrub layer, and some
206 of the canopy (Murphy et al., 2015).

207 Our results suggest that although the impact of fire increases with severity, the recovery also
208 increases. Whatever the reason, our results are in agreement with Gordon et al., (2018) who
209 found no signal of fire history in field estimates of carbon storage in forests in the same study
210 area and with Collins, (2020) who found repeated prescribed burning did not reduce carbon
211 stocks over 27 years in a similar forest type. Although those two studies measured different
212 aspects of trees, all three studies are consistent with the idea that canopy tree biomass

213 recovers after fire. On the other hand, Murphy et al., (2014) found that tree biomass was
214 increasing over time in 136 savanna monitoring plots, but this change was negatively correlated
215 with fire frequency.

216 Our results are not necessarily applicable to all forest types, as Denham et al., (2016) found that
217 severe fire top-killed 74% of the eucalypts in more arid dry sclerophyll forest in NSW, and
218 Fairman et al., (2017) found high top-kill in Victorian snow gum with frequent fire. Eucalypt
219 trees are generally a carbon sink (Hinko-Najera et al., 2017) even with frequent fires (Beringer
220 et al., 2007), and this may be because these forests are often in successional, growing stages,
221 sequestering more carbon than are climax communities. Carbon dioxide enrichment is not a
222 likely cause of the observed growth, since the limited evidence suggests that it does not
223 increase growth rates in natural Australian eucalypt forests (Collins et al., 2019), It may be that
224 the forest is still recovering from a more intense disturbance that occurred before our study.

225 The fires that caused the greatest loss of property in the Blue Mountains occurred in 1957 and
226 especially 1968 (Cunningham, 1984), shortly before our study. Although the exact extent and
227 severity of these fires are not known, their impact on the number of trees might be greater
228 than any of the fires in our study (Murphy et al., 2014, Werner and Peacock, 2019).

229 The fact that 2001 was the start of a severe drought period and coincided with the largest of
230 the fire seasons (Hammill and Tasker, 2010) is most likely important in explaining why the 2001
231 fire had a negative effect on tree densities relative to other years. It seems likely that the trees
232 would recover more slowly from that fire than in other years (Purdie and Slatyer, 1976). Both
233 fire and drought are known to affect growth in eucalypt trees, (Nolan et al., 2020) and drought
234 can cause extensive mortality in Australian forests (Mitchell et al., 2014). However, the

interactive effects of these disturbances have not been studied. Drought is unlikely to explain the general increase in tree numbers in our study, since the period was neither particularly wet nor dry (Figure 7).

Studies of climate change predict an increase in temperature in this region, and an associated increase in fire frequency and fire intensity (Bradstock et al., 2012). Our study suggests that an increase in the number of fires or severe fire is unlikely to cause a significant decrease in tree numbers, at least over time periods of decades. However, increased temperature is likely to reduce biomass (Gordon et al., 2018) and fire fuels (Thomas et al 2014), presumably because increased temperature reduces water availability.

Our method lacks the precision possible with field assessments. In particular, we cannot be sure individual tree detectability has remained consistent over time. For example, fire may have affected the gaps between trees and hence crown detectability. On the other hand, there were no other field studies or methods available that detected changes in number of trees over the time period, beginning with the 1970s. We are confident that any substantial decrease in the number of trees would have easily been detected by our method, and thus our conclusion that fires have not caused a substantial reduction in tree numbers is correct.

Conclusion

Our analysis has shown that trees have generally increased in number since the 1970s in the Blue Mountains and that fire is positively correlated with that increase. Further, where several severe fires have occurred, there have been no reduction in numbers of trees. The 2001 fires were less positive than fires in other seasons, probably due to its co-occurrence with an extended drought. The reason for the general increase and the positive role of severe fires

remains unclear, but likely reflects recovery of the canopy layer from smaller trees in these tree populations. Our results are congruent with an emergent consensus that the trees in dry sclerophyll forests of Eastern Australia are resilient to the fire regimes of the past four decades.

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Table 1. Across all fires, the categorical variable number of fires during the period 1975-2013 (R-square is 0.06657), the P-values, and estimates of the linear model obtained from the association of difference in number of trees in 1975 and 2013. * = $p \leq 0.05$, ** = $p \leq 0.01$

Number of fires	P-value	Estimate
0/Intercept	0.00113 **	0.89474
1	0.97020	-0.01068
2	0.43233	-0.22807
3	0.07574 .	0.51785
4	0.01336 *	0.83499
6	0.58817	0.35526

Table 2. For severe fires, the categorical variable number of fires during the period 1975-2013 (R-square is 0.101), the P-values, and estimates of the linear model obtained from the association of difference in number of trees in 1975 and 2013. * = $p \leq 0.05$, ** = $p \leq 0.01$

Number of Severe fires	P-value	Estimate
0/Intercept	< 2e-16 ***	0.80000
1	0.0884 .	0.18239
2	7.48e-08 ***	1.00392
3	2.40e-07 ***	2.00000

418 **Table 3.** For severe fire events in five individual years (cf. Table 2), the P-values and
419 estimates of the linear model. *** = $p \leq 0.001$

Year	P-value	Estimate value
1982	5.3e-11 ***	0.28594
1993	<2e-16 ***	0.30429
2001	1.98e-08 ***	-0.15185
2002	3.45e-07 ***	0.22856
2006	1.24e-12 ***	0.30369

Figure 1. Map showing plot locations (black dots) and fires frequency (shading) during 1975-2013 in the Blue Mountains National Park study area of New South Wales.

Figure 2. The full map of Blue Mountain region in a) 2013 and b) 1975 of MODIS, c) zoomed version of the blue circle in (a) in 2013, d) zoomed version of the blue circle in (b) in 1975, the yellow points are 560 plots of counted trees. The star indicates the location of the Katoomba weather station.

Figure 3. Distribution of tree numbers per plot across 560 plots in 1975 and in 2013 (top), and the distribution of the differences in tree numbers per plot between the two years (bottom left) in Blue Mountains, Sydney, Australia. The relationship between changes in number of trees in 2013 and the initial value in 1975 (bottom right)

Figure 4. The number of fires and number of high severity fires from 1975 to 2013 in Blue Mountain, Sydney, Australia.

Figure 5. The association between tree change with both the number of fires (a) and the number of severe fires (b) from 1975 to 2013 in Blue Mountain, Sydney, Australia.

Figure 6. Change in number of trees according to fire severity class in each fire year (1982, 1993, 2001, 2002, and 2006)

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Figure 7. Monthly Standardised Precipitation-Evapotranspiration Index (SPEI) at Bullaburra

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(longitude 150.25, latitude -33.75) from 1950 to 2013 (black line), and a LOESS (locally

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estimated scatterplot smoothing function) with a local span parameter of 0.1 (red line).