The response of Cerambycid beetles (Coleoptera: Cerambycidae) to long-term fire frequency regimes in sub-tropical eucalypt forest

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Abstract

Fire has a varied influence on plant and animal species through direct (e.g. fire induced mortality) and indirect (e.g. modification of habitat) effects. Our understanding of the influence of fire regime on invertebrates and their response to fire-induced modifications to habitat is poor. We aimed to determine the response of a beetle family (Coleoptera: Cerambycidae) to varying fire treatments and hypothesised that the abundance of cerambycid beetles is influenced by fire frequency due to modifications in habitat associated with the fire treatments. Arthropods were sampled across three months in annually and triennially burnt areas (treatments starting in 1952 and 1973, respectively), an area unburnt since 1946, and a former unburnt treatment, burnt by wildfire in 2006. Eleven different cerambycid taxa were collected using flight intercept panel traps, dominated by three species (*Ipomoria tillides*, *Adrium* sp., and *Bethelium signiferum*) which made up 99 % of individuals collected. Over the sampling period the long unburnt treatment had significantly lower species richness than the triennial and wildfire treatments. Cerambycid abundance was significantly higher in the triennially burnt treatment than in all other fire treatments. *Ipomoria tillides* was more abundant in both frequently burnt treatments, *Adrium* sp. was more common in triennially burnt areas, while *B. signiferum*, was more common in the wildfire affected treatment. Some, but not all, cerambycid beetles were more common in areas with a more open understorey (i.e. resulting from frequent burning), and lower tree basal area, as this likely influences their ability to fly easily between food sources. Cerambycid abundance was positively related to the volume of coarse woody debris and healthy tree crowns. Cerambycid beetles were clearly influenced by historic fire regime, suggesting that changes in fire regime can potentially have a profound influence on arthropod assemblages, and subsequent influences on ecosystem processes, which are currently poorly understood.

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INTRODUCTION

Fire is an important worldwide disturbance factor in forest ecosystems; with the historic fire regime often influencing the diversity and abundance of plant and animal species (Gill et al., 2012, Kwon et al., 2013, McCaw, 2013, Kamczyc et al., 2017). The nature of fundamental ecological processes such as fire needs to be better understood so that land managers can have a greater understanding of which fire regimes are appropriate for specific ecosystems.

In Australia, the dry sclerophyll eucalypt forests cover large areas of the continent and are significant in terms of their biodiversity values (Hnatiuk et al., 2003, Gill & Zylstra, 2005, Bradstock, 2010, Gill et al., 2012). These eucalypt forests are frequently subject to low-intensity fire, often through prescribed burning, which can have a strong influence on plant composition and structure, therefore influencing faunal habitat. A large body of research has been undertaken on the effects of fire frequency on vegetation and habitat structure in forest environments throughout Australia (e.g. York, 1999, Pausas & Bradstock, 2007, Bowman et al., 2014, Clarke et al., 2014). Fire frequency can sometimes be important to specific plant or animal species, which may rely on or benefit from frequent fire or long periods without fire (Andrew et al., 2000, Bradstock et al., 2005, Fisher et al., 2009, Duff et al., 2013, Bowman et al., 2014). For example, caterpillars of an Australian butterfly, the Altona skipper (*Hesperilla flavescens* subsp. *flavescens*) need fresh growth stimulated by burning which encourages oviposition and feeding. However, the same species still needs some areas to remain as refuges to avoid fires and survive in the immediate post-burn environment (New et al., 2010). *Hesperilla flavescens* is just one example of many Australian invertebrate species that may be responsive to fire frequency, and in particular, time since fire. Defining an appropriate fire regime for a specific ecosystem can be particularly difficult when we lack the detailed information on responses of different biota, with sometimes conflicting preferences, within the ecosystem to the fire regime (Driscoll et al., 2010). In most cases, recommended fire regimes for biodiversity conservation are based primarily on the plant community responses and can be particularly broad (e.g. Lawes & Clarke, 2011, Duff et al., 2013, Clarke et al., 2015). As an example, in Queensland, management guidelines for dry sclerophyll forests in our study region suggest burning at intervals of between four to twenty years, with an emphasis on variability of fire intervals (QPWS, 2013, Queensland Herbarium, 2013).

There is a paucity of detailed scientific information on the influence of fire regime on invertebrate assemblages and their response to fire induced modifications of habitat. Studies

in Australia (southeast Queensland, Northern Territory & South Australia) and other countries (Argentina, Switzerland, southwestern United States of America and South Africa) show that response of invertebrate fauna to fire frequency varies between invertebrate groups, and some groups (e.g. ants) may show similar responses at different locations (Orgeas & Andersen, 2001, Moretti et al., 2006, Underwood & Quinn, 2010, Pryke & Samways, 2011, Beaumont et al., 2012, Calcaterra et al., 2014, Manwaring et al., 2015, York & Lewis, 2017).

Our study focused on a family of native Australian beetles, the Cerambycidae (longicorn or longhorn beetles). The diet of cerambycid beetles during their life cycle consists of a variety of living and dead plant species and this is variable among species (Linsley, 1959, Hanks, 1999, Buse et al., 2007, Nahrung et al., 2014). Cerambycid beetles are a highly abundant and diverse family (Linsley, 1959, Hanks, 1999, Slipinski & Escalona, 2013, Nahrung et al., 2014, Slipinski & Escalona, 2016), that can be surveyed relatively easy through proven trapping methodologies, using appropriate attractants (Dodds et al., 2010, Graham & Poland, 2012, Graham et al., 2012, Hanks et al., 2012, Allison et al., 2014). The abundance and diversity of cerambycid beetles, with a diverse range of feeding habits, life cycles, mating and dispersal methods, make them an important part of ecosystems (Maeto et al., 2002, Hanks et al., 2005, Paro et al., 2011, Meng et al., 2013, Nahrung et al., 2014) and a group that is likely to be responsive to different fire regimes. There has been a considerable amount of research on the effect of fire on beetle assemblages' worldwide, often reporting links between habitats affected by fire (Jenkins et al., 2008, Louzada et al., 2010, Ulyshen et al., 2010, Azeria et al., 2012). Research has shown that beetles are affected by fire through the associated changes of the habitat, but there are relatively few studies in Australia (e.g. Collett, 1999, York, 1999, Orgeas & Andersen, 2001, Radho-Toly et al., 2001, New et al., 2010, Manwaring et al., 2015), and no studies in naturally occurring sub-tropical eucalypt forests. There has been a limited amount of research undertaken on the impacts of fire regime on cerambycid beetles internationally (e.g. Moretti et al., 2010, Ulyshen et al., 2010, Azeria et al., 2012, Boulanger et al., 2013, Breton et al., 2013).

We aimed to determine the long-term response of cerambycid beetles to variation in fire frequency. We hypothesised that the abundance of cerambycid beetles is affected by fire frequency primarily through changes in habitat availability associated with the different fire histories. To test this hypothesis we made use of the longest running fire experiment in Australia, which has been managed since 1952. Our study aimed to provide a better understanding of the effects of fire frequency on this specific group of beetles to build on

existing knowledge and improve our understanding of ‘ecologically appropriate’ fire regimes for eucalypt forests in the sub-tropics. This information is needed to guide improved forest management for biodiversity conservation.

METHODS

Study area and long-term fire experiment

The study area was located within Bauple State Forest, approximately 50 km north of Gympie, Queensland (-25.88°, 152.64°). The climate is classed as sub-tropical with an average rainfall of 1250 mm, warm wet summers and cool and relatively drier winters. The study area has an elevation that varies from approximately 55–115 m above sea level, and is dominated by spotted gum (*Corymbia citriodora* subsp. *variegata*) and various co-dominant eucalypt species (Lewis & Debuse, 2012). This study site covers a large area (~900 ha) of dry sclerophyll eucalypt forest; a forest type that dominates a considerable proportion of eastern Australia including large areas of south-eastern Queensland. Due to its inherent flammability, dry sclerophyll eucalypt forest is frequently burnt through both wildfire and prescribed fire.

Experimental design

The Bauple fire experiment consists of four fire treatments (> 60 ha per treatment): an area burnt annually since 1952; an area burnt every three years (triennial) since 1973; an area that has remained unburnt since 1946; and an area that was formerly part of the unburnt treatment, but was burnt by a wildfire in 2006 (Fig. 1). Within each treatment, we made use of five permanent monitoring plots (100 × 40 m) providing a total of 20 plots for sampling of cerambycid beetles. The experiment is pseudoreplicated, as the treatments are not replicated within the experimental area, and hence caution is needed before extrapolating findings to other locations. It is an operational-scale experiment that covers a large area and there was a minimum distance of 369 m, maximum of 890 m and an average distance of 545 m between individual monitoring plots (Fig. 1). Within the entire study area, selective logging and thinning has been performed (1969 – 1974 and 1984 – 1992), with periods of cattle grazing up until approximately 2000 (Lewis & Debuse, 2012). Fire treatments are low intensity burns, usually during winter or spring. However, the 2006 wildfire was a moderate intensity fire.

Cerambycid sampling

Two flight intercept panel traps (Alpha Scents Inc., West Linn, OR. USA) were located randomly within each of the 20 plots (Fig. 1). This type of trap has been identified as one of

the best designs for catching cerambycid beetles, especially when trap surfaces are painted with fluon, which enhances trapping efficiency (Graham & Poland, 2012, Graham et al., 2012, Nehme et al., 2014, Wotherspoon et al., 2014, Allison & Redak, 2017). The traps were positioned randomly in what were considered to be insect flight paths (i.e. relatively clear line of sight in at least one direction, and > 2 m from a tree). One trap of each pair at a plot was baited with a lure containing a known 'generic' cerambycid pheromone (3-hydroxy-2-hexanone) (Mitchell et al., 2012, Hanks & Millar, 2013) and the other with a kairomone lure (ethanol and α -pinene). The pheromone lure is attractive to a wide variety of cerambycid beetles across four continents (Hanks et al., 2012, Hanks & Millar, 2013, Hanks et al., 2014, Wickham et al., 2014, Hanks & Millar, 2016, Hayes et al., 2016, Silva et al., 2017). In an Australian context, of a variety of pheromone treatments tested, this treatment captured the widest diversity of cerambycid species and was the only pheromone that captured beetles at significantly higher rates than the control treatment (Hayes et al., 2016). Traps were maintained over a twelve week collection period from the beginning of March to the end of May 2015, with collection of trap catches and re-baiting approximately every two weeks (six collections in total). Traps at each plot were rotated between positions at each clearance to avoid positional bias. After each collection period, cerambycid beetles were separated from other beetles and any other arthropods (by-catch). Cerambycid beetles were identified to sub-family, genus and species, or morphospecies if it was not possible to identify to species level. This was completed using reference collection specimens, keys and text supplied by Queensland Government Department of Agriculture & Fisheries, Brisbane (Slipinski & Escalona, 2013).

Vegetation and Soil

At twenty locations, a 100 × 10 m transect was assessed using tape measures, following a random compass bearing (Fig. 1). Transects were generally nearby trapping plot, but in two cases, transects were not adjacent to the plots where the trapping had taken place. For the two plots that did not have transects immediately adjacent, treatment averages were used for these plots in the analysis. Within each transect, the diameter at breast height (DBH over bark, at 1.3 m from the ground) was recorded for all plant species with a DBH that was ≥ 10 cm. DBH of dead trees was also recorded. This allowed calculation of the basal area (m^2/ha) of live and dead trees using the equation (Shiver & Borders, 1996):

$$BA = \frac{\pi}{40000} \times DBH^2$$

Tree crown health was assessed for each living tree with a DBH that was ≥ 10 cm using the attributes of the Grimes crown scoring system (Grimes, 1978), which was developed for this type of native forest in Queensland. This involved providing scores for: Crown Position (score of 1–5); Crown Size (score of 1–5); Crown Density (score of 1–9); Dead Branches (score of 1–5); and Crown Epicormics (score of 1–3). A total Grimes crown score was calculated for each individual tree by adding the scores for each attribute to give a total out of 27 (where higher scores indicate greater canopy health).

Coarse woody debris (CWD) volume (m^3) was estimated using the line-intersect method (Van Wagner, 1968) for each transect. The length and end diameters of each piece of CWD was ≥ 10 cm diameter that crossed the central transect (100 m) were recorded. Volume was calculated using Smalian's formula (Woldendorp & Keenan, 2005):

$$V = L (A_b + A_s)/2$$

Where V = volume, L = length of piece, A_b = cross-sectional area at large end of piece, A_s = cross-sectional area at small end of piece.

Previous research (Waldron, 2014) undertaken twelve months prior to our sampling analysed a number of top-soil (0–10 cm) variables based on soil collected at the same plots used for this research. There was twenty-four soil cores collected per plot by dividing the plot into 8 equal sub-plots with cores taken from three random locations within each subplot. Collected soil was oven dried at 40°C and sieved to < 2 mm using methods outlined by Rayment & Lyons (2012). The Environmental Analysis Laboratory of Southern Cross University, Lismore, NSW carried out soil analysis. Top-soil available phosphorus (Bray, mg/kg) and total nitrogen (%) were used in our analysis. Treatment means (and standard errors) for the above vegetation and soil attributes are provided in the supplementary material (Appendix 1).

Data Analysis

Preliminary analyses indicated a clear distinction in beetle catches in relation to the lure type; traps baited with pheromone lures recorded 99 % of individuals and the α -pinene/ethanol-baited traps recorded 1 % of individuals. Given the low number of individuals in the α -pinene/ethanol-baited traps subsequent statistical analyses focussed on captures within pheromone traps only.

We determined the influence of fire treatment and a number of co-variables on cerambycid abundance, cerambycid richness, and the abundance of the two most frequent species (*Ipomoria tillides* and *Adrium* sp.), which accounted for 94.5 % of total individuals caught, using repeated measures ANOVA. We initially investigated relationships between predictor variables using Spearman's Rank correlations; selecting the most biologically meaningful variables for inclusion as co-variables in the analysis after eliminating any predictor variables that were strongly correlated with each other (≥ 0.7). Variables included in the subsequent ANOVA were: fire treatment (annual, triennial, wildfire, unburnt), live tree BA, dead tree BA, CWD volume, Grimes crown score (total), top-soil Phosphorus (Bray) and top-soil total Nitrogen. These variables varied spatially among plots but not temporally and allocated to the plot stratum in the ANOVA, for which fire frequency assigned as the treatment factor and all other spatially varying variables assigned as co-variables. Co-variate regressions were calculated across all times and separately between treatments. Significant co-variables provide an indication of confounding with fire treatment. Where co-variables were significant, predicted means from the ANOVA adjusted accordingly. Repeated measures ANOVA investigated the effect of fire treatment, collection period (time), and their interaction.

Climatic data (interpolated data from the silo data drill website: <https://www.longpaddock.qld.gov.au/silo/datadrill/index.php>) was obtained for the study site for each day of the survey period (Jeffrey et al., 2001). These data included minimum and maximum temperatures ($^{\circ}\text{C}$), rainfall (mm), radiation (MJ/m^2) and minimum and maximum relative humidity (%). We initially investigated relationships using linear regression between minimum and maximum temperature, rainfall, radiation and minimum and maximum relative humidity and time of cerambycid sample collection. Minimum and maximum temperatures and radiation all declined significantly over the period of trapping (minimum temperature: $F_{1,4} = 23.54$, $P \leq 0.01$, $R^2_{\text{adj}} = 0.82$; maximum temperature: $F_{1,4} = 68.44$, $P \leq 0.001$, $R^2_{\text{adj}} = 0.93$; radiation: $F_{1,4} = 17.31$, $P \leq 0.05$, $R^2_{\text{adj}} = 0.77$) and were significantly correlated. Rainfall and relative humidity did not vary significantly over time ($P > 0.1$ in all cases). Given that sampling time was correlated with climatic data (temperature and radiation), we could not include all terms in the model as variation in the climatic data was explained by variation in time (Fig. 2). This needs to be considered when interpreting the influence of the temporal variation.

For analysis of cerambycid abundance and abundance of the two most common species, data were square root transformed to ensure homogeneity of variance. Cerambycid richness data did not require a transformation. To aid interpretation, we report back-transformed means in the text, but figures show the predicted means and errors. Post-hoc comparisons among treatment and time combinations conducted using LSD tests at the 5 % level. The relative importance of each co-variate used in the analysis assessed by comparing standardized beta coefficients (β_s). All statistical analyses performed using Genstat™.

RESULTS

Across the twelve-week trapping period a total of 2702 individual cerambycid beetles were collected. Eleven different cerambycid taxa were collected in traps, dominated by three species which made up 99 % of all individuals (Supplementary Material, Appendix 2). These dominant species were *Ipomoria tillides* (53 %), *Adrium* sp. (41.5 %), and *Bethelium signiferum* (4.5 %). Five of the eleven taxa were only present in pheromone baited traps and three taxa were unique to α -pinene/ethanol-baited traps. Three taxa were common to both pheromone and α -pinene/ethanol-baited traps.. Beetles collected belonged to two sub-families and four tribes within the Cerambycidae. Of the two sub-families, the majority of individuals (99.5 %) belonged to the Cerambycinae and the rest (0.5 %) were Lamiinae.

Species richness (the number of taxa recorded per plot at each trap clearance) varied significantly between fire treatments ($F_{3, 16} = 5.11$, $P \leq 0.05$) and over time ($F_{5, 80} = 4.37$, $P \leq 0.001$) while the interaction between treatment and time was not significant ($F_{15, 80} = 1.17$, $P > 0.05$). The unburnt treatment had significantly lower species richness than the triennial and wildfire, while species richness in the annual did not differ significantly to any of the other treatments (Fig. 3a). Species richness declined consistently and significantly over time (Fig. 3b; $F_{1, 4} = 283.7$, $P < 0.001$, $R^2_{adj} = 0.98$), presumably in response to the seasonal decline in temperatures and radiation. None of the co-variables investigated had a significant influence on cerambycid species richness ($F_{6, 10} = 1.54$, $P > 0.05$).

Cerambycid abundance varied significantly between fire treatments ($F_{3, 12} = 16.4$, $P < 0.001$) and over time ($F_{5, 80} = 17.9$, $P < 0.001$) while the interaction between treatment and time was not significant ($F_{15, 80} = 0.97$, $P > 0.05$). Cerambycid abundance was significantly higher in the triennial than in all other fire treatments (Fig. 4a). Species abundance declined linearly over time (Fig. 4b; $F_{1, 4} = 87.8$, $P < 0.001$, $R^2_{adj} = 0.95$), presumably in response to the decline in temperatures and radiation.-. Abundance was negatively related to live tree

basal area ($F_{1, 12} = 4.92$, $P \leq 0.05$, standardised $\beta = -4.4$) and top-soil total N ($F_{1, 12} = 15.7$, $P \leq 0.01$, standardised $\beta = -4.0$), but was positively related to coarse woody debris volume ($F_{1, 12} = 6.93$, $P \leq 0.05$, standardised $\beta = 3.2$) and Grimes crown score ($F_{1, 12} = 6.19$, $P \leq 0.05$, standardised $\beta = 2.5$).

Abundance data was driven by the two most abundant species (*I. tillides* and *Adrium* sp.) and hence these species were analysed separately. There was a significant interaction between treatment and time for *I. tillides* ($F_{15, 80} = 2.39$, $P \leq 0.01$, Fig. 5). For the first three trapping periods (March and April) *I. tillides* abundance was higher in annual and triennial than in both wildfire and unburnt (Fig. 5). For the remaining three trapping periods there was no difference between treatments, except in the fourth trapping period (late April to the 1st May) where triennial had greater abundance than unburnt. Both treatment ($F_{3, 15} = 6.21$, $P \leq 0.01$) and time effects ($F_{15, 80} = 23.8$, $P < 0.001$) were significant. Across all times, the annual and triennial had a greater abundance of *I. tillides* than wildfire and unburnt. Across all treatments, *I. tillides* abundance declined linearly over time, particularly after the first two trapping periods ($F_{1, 4} = 52.5$, $P \leq 0.002$, $R^2_{\text{adj}} = 0.91$). None of the co-variables investigated had a significant influence on *I. tillides* abundance ($F_{6, 10} = 3.00$, $P > 0.05$).

Adrium sp. abundance varied significantly between fire treatments ($F_{3, 12} = 8.47$, $P = 0.003$) and over time ($F_{5, 80} = 4.64$, $P < 0.001$) while the interaction between treatment and time was not significant ($F_{15, 80} = 1.25$, $P > 0.05$). Abundance of *Adrium* sp. was significantly higher in triennial than all other fire treatments (Fig. 6a). Abundance of *Adrium* sp. declined significantly in the third trapping period, but increased somewhat in the fourth and fifth periods (abundance in these periods was not significantly different to that in the first two periods) then declined again in the last trapping period (Fig. 6b). *Adrium* species abundance was negatively related to live tree basal area ($F_{1, 12} = 7.52$, $P \leq 0.05$, standardised $\beta = -3.6$) and was positively related to Grimes crown score ($F_{1, 12} = 8.50$, $P \leq 0.05$, standardised $\beta = 2.9$).

Due to low abundance in some treatments, the third most common species, *B. signiferum*, was not included in our statistical analyses, but was more frequently recorded in wildfire. Most individuals of *B. signiferum* occurred in wildfire (107 individuals), with triennial, annual and unburnt recording thirteen, three and two individuals, respectively.

DISCUSSION

We investigated the response of a common family of wood-boring beetles to long-term fire frequency regimes. This study was unique in that it explored the effects of long-term fire regime rather than once-off fire or time since fire. Our hypothesis that cerambycid beetle assemblages would be influenced by fire frequency appeared to be supported. The results indicate that three common cerambycid beetle species did show a response to differences in habitat structure and soil fertility that are influenced by fire regime. For example, the abundance of *I. tillides* was higher in the two frequent fire treatments in this study. There were clear differences in understorey density and habitat features (e.g. tree basal area) between the different fire treatments (Appendix 1; Lewis & Debuse, 2012). Hence it is likely that the change in habitat structure associated with frequent repeated fire has an influence on cerambycid beetles. Cerambycid beetles are known to choose flight paths through habitat, as most species are active fliers as adults (Smith et al., 2004, Dodds et al., 2010, Allison et al., 2014). Areas with a more open understory provide more open flight paths, allowing them to move around without difficulty between food sources. The negative influence of live tree basal area on cerambycid abundance and *Adrium* sp. abundance supports this hypothesis. Other than habitat structure, fire regime in south east Queensland eucalypt forests also influences soil properties (Muqaddas et al., 2015, Muqaddas et al., 2016, Butler et al., 2018), resulting in potential flow-on effects that influence biologically available nutrients for invertebrate consumers.

As many cerambycid species are known to be host species specific (e.g. Hawkeswood, 1993, Turner & Hawkeswood, 2001, Hawkeswood, 2003), the alterations in tree species composition through fire regime at the site (e.g. see Lewis & Debuse, 2012) could influence the abundance of different cerambycid beetle taxa. For two commonly encountered species within the Tribe Callidiopini, *Bethelium* and *Ipomoria*, contrasting preferences were observed. The most abundant species caught in traps, *I. tillides*, showed significantly higher abundance in annual and triennial compared to unburnt and wildfire treatments. *Bethelium signiferum* showed a significantly higher abundance in wildfire treatment compared to the other three treatments. It is likely that *B. signiferum* prefers habitat with regenerating *Acacia leiocalyx*, which is particularly prevalent in the understorey after wildfire and is a known host for this species (Hawkeswood, 1993, Hawkeswood, 2003).

Cerambycid beetles have a range of feeding preferences (e.g. saproxyly, xylophagy, phloemophagy), with adults often having differing preferences (e.g. flowers and other

external plant parts for food) to larvae, which often feed on phloem or decaying wood (Linsley, 1959, Allison et al., 2004). Hence it is likely that fire regime also influenced the availability of different food resources through the associated changes in habitat (e.g. plant species composition, availability of decaying wood). This study detected positive relationships between cerambycid abundance and both the volume of coarse woody debris at a site and average Grimes tree crown score for a site. This suggests that sites with higher volumes of coarse woody debris and healthier tree crowns might provide better food resources for some cerambycid taxa (e.g. *Adrium* sp.). The life cycle of cerambycid beetles include adults which lay eggs in cracks or under barks of trees and dead wood, then the larvae hatch and feed on the host to then pupate a few months to a year later (Slipinski & Escalona, 2013, Slipinski & Escalona, 2016). Such taxa might prefer sites with a combination of large, healthy tree canopies as a food source for adults and dead wood (e.g. coarse woody debris) as a larval food source (Linsley, 1959, Hanks, 1999). This relationship could be driven by availability of oviposition host material as some species of cerambycid beetles use dead material as hosts.

There was a significant decline in cerambycid abundance, driven primarily by the abundance of two common species (*I. tillides* and *Adrium* sp.) over the period of this study; a trend most likely related to the seasonal decline in temperature and solar radiation in this period (Fig. 2). Additionally, there was significant rainfall during the 4th and 5th collection times, from the 30th of April to the 2nd of May, which may have contributed to the drop in cerambycid abundance, due to the high rainfall and increased cloud cover. This temporal variation in abundance is not surprising as many cerambycid beetles are more active in the warmer months (Keena & Moore, 2010, Hanks et al., 2014). The interaction between fire treatment and time of sampling for *I. tillides* suggested greater variation between fire treatments earlier in the study when weather conditions were warmer. It is possible that treatment effects are more obvious during warmer conditions for some species, due to higher levels of activity during warmer months. For example, mating and feeding might be easier in the more open habitat that was frequently burnt, in the warmer conditions, prior to a decline in abundance in cooler months when they have finished mating and feeding. Variation in cerambycid abundance between treatments may also have been influenced by other factors, including predation, resource competition, or attraction of beetles from outside the plots, although the latter was unlikely as the pheromone used here is likely to act over a relatively short range (Samways et al., 2010, Millar & Hanks, 2017). Ideally, further surveys should be

carried out throughout all seasons of the year and over multiple years to expand on the findings of this study, and include the investigation of other beetle families.

The trapping methodology utilised here may have introduced some biases in trap catches. Traps were all set at the same height, relatively close to the ground, limiting the survey to beetles that regularly fly at that height. Other studies have recognized that cerambycid species also occur in the canopy of forests, and cerambycid beetles can also have high abundances on the edges of large patches of forest (e.g. Vance et al., 2003, Webb et al., 2008, Dodds et al., 2010, Graham et al., 2012). For placement of the traps, the unburnt and wildfire treatments had a higher density of woody plants in the understorey, so it was more difficult to identify clear flight paths than in the frequently burnt treatment areas. Thus it is difficult to ascertain whether the differences in vegetation associated with the different fire treatments have influenced cerambycid abundance due to associated habitat preferences or through increased probability of capture due to availability of clear flight paths.

This study clearly demonstrates that historic fire regime has an influence on cerambycid beetles, most likely through changes in habitat suitability. It is highly likely that a range of other invertebrate taxa are also responsive to fire regime, through the modifications in habitat availability (e.g. Vanderwoude et al., 1997, York, 1999, York & Lewis, 2017). Such changes in invertebrate composition may have flow-on effects to other components of the ecosystem (e.g. through pollination services and providing a food source to other organisms, Brown & York (2017)). Further studies are needed to understand the consequences associated with altered invertebrate assemblages in terms of ecological processes (e.g. nutrient cycling, tree health etc.). This study demonstrates the limitations of setting prescribed burn regimes that are based solely on one facet of biodiversity (i.e. plant composition).

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