

CHAPTER 5 The effect of prescribed burns on food availability

Part A: Recovery of seed availability after fire and patterns of habitat use by the south-eastern Red-tailed Black Cockatoo

5.1 Introduction

Fire has been an important source of disturbance in the eucalypt woodlands of Australia since the mid-Quaternary (Gill *et al.* 1981), and is now a commonly used tool for management. Although many Australian plants and animals are well adapted to fire, the widespread loss and fragmentation of woodland habitats has increased the vulnerability of some species to fire. Inappropriate fire regimes are listed as a factor contributing to the extinction of five bird taxa, and as a threatening process to at least 51 nationally threatened bird taxa (Woinarski and Recher 1997). The south-eastern Red-tailed Black Cockatoo has already suffered severe habitat loss (50%, Chapter 3), and there is evidence to suggest that food availability may be limiting the population (Chapter 4). The primary objective of the present study was to assess the effect of prescribed burns on food availability to the Red-tailed Black Cockatoo.

Food availability after fire is an important concern for birds requiring mature vegetation for nectar or seed (Woinarski and Recher 1997), but varies depending on the responses of the food plants in question. Hansen *et al.* (1991), Pannell and Myerscough (1993), Enright *et al.* (1996) and others have investigated patterns of flowering and seed production in understorey shrubs with time since fire. Fecundity may improve or decline after fire depending on how the plant regenerates and its particular life history characteristics (Gill and Groves 1980). However, there are few studies in the literature concerned with the flowering responses of eucalypts to fire. Particularly lacking is information about the effect of different fire regimes on the timing and quantity of subsequent flower and seed production. Eucalypts are typically well adapted to fire and will usually show a vigorous regenerative response following defoliation, sometimes recovering their former canopy structure within three years (Gill 1981, Strasser *et al.* 1996). However, resource allocation theory suggests that an increase in vegetative vigour will generally necessitate a reduction in the allocation of resources to reproductive growth (Reekie and Bazzaz 1987). Short term reductions in seed production after fire have been demonstrated for tropical savanna eucalypts *E. miniata* and *E. tetradonta* (Setterfield 1997), but the longer term effects of fire on eucalypt fecundity are virtually unknown.

In addition to effects on the quantity of fruits produced per plant, fire can influence the way in which flowers and fruits are displayed or held on the plant. Examples include effects on the number of fruits per inflorescence (*Teleopea speciosissima*, Pyke 1983), percentage of fruits bearing seeds (*Banksia* sp., Zammit and Westoby 1987) and seed production per unit area (*Anigozanthos* sp., Lamont and Runciman 1993). The south-eastern Red-tailed Black Cockatoo shows a strong tendency to select trees with larger crop size (capsules per tree), capsule density (capsules per branch) and cluster size (capsules per pedicel; Chapter 3). This chapter compares these aspects of capsule availability and levels of habitat use (percentage of trees and branches used for foraging) between sites grouped according to time since fire.

5.2 Methods

The broad-scale habitat survey provided the basis for the present study (methods are described in section 2.2). Measurements of capsule availability were: capsule density (the number of capsules per branch), crop size (the number of capsules per tree), cluster size (the number of capsules per pedicel) and bud production (the number of inflorescences per tree). Measurements of habitat use were the percentage of fruiting trees used by cockatoos per sampling point and the percentage of fruiting branches used per tree. Capsule volume was measured for a subset of burnt and unburnt sites as a measure of capsule profitability. The location of study sites is shown in Figure 2.2.

The precise distribution of the two stringybark eucalypts was unknown at the beginning of the study, and thus both species were included in the same survey. A small number of replicate sites for *E. baxteri* precluded the separate analysis of time since fire data for each species. However, these data were ordinated using Multi-Dimensional Scaling (MDS), and a one way Analysis of Similarity (ANOSIM) showed no significant differences between tree species for the measured aspects of capsule availability ($R = -0.049$, $P = 0.94$). Capsule availability characteristics were compared at various times post-fire on a site by site basis for each species, but a small number of *E. baxteri* sites prevented further statistical comparisons. *E. baxteri* sites were not included in analyses of habitat use by cockatoos because very few of these sites were used by cockatoos during the survey period.

5.2.1 Statistical Analysis

The survey design comprised three factors (all fixed): time since fire (3, 5, 6, 7, 9, 11 and >25 years since fire groups), site and position (edge and interior). There were three replicate sites per time since fire group, except the 3 years since fire group which had four replicate sites. Each burnt site was paired with a nearby unburnt (>25 years since fire site). The total number of unburnt sites was eleven, since

some unburnt sites were shared between nearby burnt sites. The appropriate analysis for such a design is a nested ANOVA, since there were different levels of site for each time since fire group. However, the power of these tests was typically very low (ranging between 0.1 and 0.42), leaving a high probability of making a Type II error (accepting the null hypothesis when it is false). Therefore, these tests were not included in the analysis.

Instead, three way (factorial) ANOVAs were used to test for the effects of burn status (burnt and unburnt groups), site, position (edge and interior groups) and each of the interaction terms. These tests were more powerful and also enabled interaction effects to be considered. However, the test requires a balanced design for maximum power (Zar 1984). Although sample sizes for the study were equal initially, the sample sizes were unequal for capsule availability characteristics when non-fruiting trees were excluded, and for the percentage of branches used by cockatoos when trees that were not used by cockatoos were excluded. Therefore, it was necessary to equalise sample sizes for these variables by randomly deleting samples where necessary, as suggested by Zar (1984).

The distribution of all variables was positively skewed, but there was a high level of normality for all variables following log transformation. The ANOVA test is considered to be sufficiently robust to tolerate small departures from normality (Underwood 1981). However, because some sites were non-normally distributed, Wilcoxon Rank Sum tests were used to compare burnt sites with their paired unburnt sites. These tests were also used to compare each time since fire group with its paired unburnt group. In this context, paired unburnt groups included only those unburnt sites paired with the sites in the time since fire group being considered.

In addition, the numerical data for crop size, capsule density, cluster size and bud production were converted to categorical data, as a means of avoiding the problems associated with skewed distributions, and to show trends not apparent from comparison of means. The percentage of trees in each category was determined using frequency distributions. Log-likelihood ratios (contingency tests) were used to test for differences between each pair of burnt and unburnt sites, because these tests are more accurate than Chi-Square tests in situations where low cell frequencies occur (Zar 1984).

5.3 Results

5.3.1 Recovery of seed availability after fire

Percentage of trees fruiting

The percentage of trees fruiting showed a positive relationship with time since fire, and increased steadily from 48% at 3 years since fire to 94% at 9 years since fire (Figure 5.1).

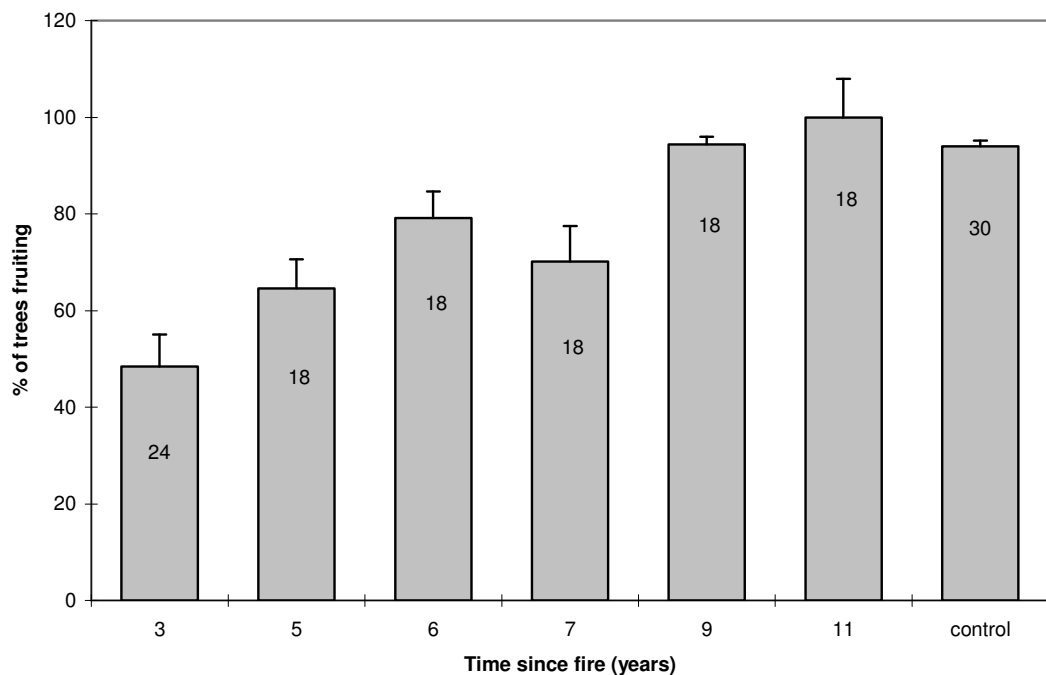


Figure 5.1. Mean (+SE) percentage of trees fruiting per sampling point in each time since fire group and the unburnt group. The total number of replicate sampling points (n) is indicated for each group.

Comparison of each pair of burnt and unburnt sites showed all sites in the 3, 5, 6 and 7 years since fire groups to be substantially lower in the percentage of trees fruiting than their paired unburnt sites (Table 5.1). Burnt and unburnt values tended to equalise as time since fire increased, with comparable values between pairs of burnt and unburnt sites in the 9 and 11 years since fire groups. Significant differences in the percentage of trees fruiting occurred for 8 out of 18 site pairs (although note the small sample sizes for these comparisons). Comparison of each pair of time since fire and unburnt groups showed significant differences for all groups except the 9 and 11 years since fire groups.

Table 5.1. Mean (\pm SE) percentage of trees fruiting per sampling point in each pair of burnt and unburnt sites. The number of trees producing at least one capsule was scored for 8 trees per sampling point and these values were converted to percentages to determine the percentage of trees fruiting per sampling point (n = no. of sampling points). The data are summarised for each time since fire (TSF) group. Significant differences based on Wilcoxon Rank Sum tests are shown with asterisks (Levels: * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$).

| TSF | Site | Species | Status | % of trees fruiting | n |
|-----------|--------------------|--------------------|---------------|---------------------|---|
| 3 | Byjuke | <i>E. baxteri</i> | burnt | 67 \pm 10 | 6 |
| | | | unburnt | 96 \pm 3* | 6 |
| | Kadnook | <i>E. arenacea</i> | burnt | 38 \pm 14 | 6 |
| | | | unburnt | 83 \pm 8* | 6 |
| | Kealys | <i>E. arenacea</i> | burnt | 65 \pm 15 | 6 |
| unburnt | | | 92 \pm 3 | 6 | |
| Tullich | <i>E. arenacea</i> | burnt | 25 \pm 7 | 6 | |
| | | unburnt | 92 \pm 3* | 6 | |
| all | | burnt | 48 \pm 7 | 24 | |
| | | unburnt | 91 \pm 2*** | 24 | |
| 5 | Roseneath | <i>E. baxteri</i> | burnt | 38 \pm 21 | 6 |
| | | | unburnt | 96 \pm 3** | 6 |
| | Tullich | <i>E. arenacea</i> | burnt | 79 \pm 7 | 6 |
| | | | unburnt | 92 \pm 3* | 6 |
| Yallakar | <i>E. arenacea</i> | burnt | 77 \pm 6 | 6 | |
| | | unburnt | 96 \pm 3 | 6 | |
| all | | burnt | 65 \pm 6 | 18 | |
| | | unburnt | 94 \pm 2*** | 18 | |
| 6 | Kalingur | <i>E. arenacea</i> | burnt | 85 \pm 6 | 6 |
| | | | unburnt | 98 \pm 2 | 6 |
| | Roseneath | <i>E. baxteri</i> | burnt | 85 \pm 6 | 6 |
| | | | unburnt | 96 \pm 3 | 6 |
| Yallakar | <i>E. arenacea</i> | burnt | 67 \pm 14 | 6 | |
| | | unburnt | 96 \pm 3* | 6 | |
| all | | burnt | 79 \pm 5 | 18 | |
| | | unburnt | 97 \pm 1** | 18 | |
| 7 | Cemetery Tk. | <i>E. arenacea</i> | burnt | 50 \pm 11 | 6 |
| | | | unburnt | 88 \pm 5* | 6 |
| | Jilpanger | <i>E. arenacea</i> | burnt | 67 \pm 15 | 6 |
| | | | unburnt | 98 \pm 2* | 6 |
| Tower Tk. | <i>E. arenacea</i> | burnt | 100 \pm 0 | 6 | |
| | | unburnt | 100 \pm 0 | 6 | |
| all | | burnt | 70 \pm 7 | 18 | |
| | | unburnt | 95 \pm 2* | 18 | |
| 9 | Jilpanger | <i>E. arenacea</i> | burnt | 92 \pm 3 | 6 |
| | | | unburnt | 98 \pm 2 | 6 |
| | Kalingur | <i>E. arenacea</i> | burnt | 98 \pm 2 | 6 |
| | | | unburnt | 98 \pm 2 | 6 |
| Nangwarry | <i>E. baxteri</i> | burnt | 94 \pm 3 | 6 | |
| | | unburnt | 100 \pm 0 | 6 | |
| all | | burnt | 94 \pm 2 | 18 | |
| | | unburnt | 99 \pm 1 | 18 | |
| 11 | Jilpanger | <i>E. arenacea</i> | burnt | 96 \pm 3 | 6 |
| | | | unburnt | 98 \pm 3 | 6 |
| | Nangwarry | <i>E. baxteri</i> | burnt | 100 \pm 0 | 6 |
| | | | unburnt | 100 \pm 0 | 6 |
| Roseneath | <i>E. baxteri</i> | burnt | 83 \pm 8 | 6 | |
| | | unburnt | 96 \pm 3 | 6 | |
| all | | burnt | 93 \pm 3 | 18 | |
| | | unburnt | 98 \pm 1 | 18 | |

The mean percentage of trees fruiting for all burnt sites combined was 74% compared with 94% for all unburnt sites combined. A three way ANOVA showed significant differences for the effects of burn status (comparing all burnt sites with all unburnt sites) and site, but not for position (edge and interior groups) or any of the interaction terms (Table 5.2). These results indicate that the effect of burn status was independent of differences caused by the effects of site and position.

Table 5.2. Results of three way ANOVA testing the effects of status (burnt and unburnt groups), site, position (edge and interior) and interaction terms on the percentage of trees fruiting (log transformed, n = 3 sampling points per position).

| Source of variation | SS | df | MS | F | P |
|----------------------|-------|-----|------|-------|---------|
| Total | 14.63 | 119 | | | |
| Status | 2.13 | 1 | 2.13 | 22.14 | <0.0001 |
| Site | 2.08 | 9 | 0.23 | 2.41 | 0.02 |
| Position | 0.06 | 1 | 0.06 | 0.61 | 0.44 |
| Status*site | 1.36 | 9 | 0.15 | 1.57 | 0.14 |
| Status*position | 0.08 | 1 | 0.08 | 0.85 | 0.36 |
| Site*position | 0.35 | 9 | 0.04 | 0.4 | 0.93 |
| Status*site*position | 0.32 | 9 | 0.04 | 0.33 | 0.96 |
| Error | 8.56 | 89 | 0.1 | | |

Capsule density, crop size and cluster size

The performance of fruiting trees in terms of capsule density (capsules per branch), crop size (capsules per tree) and cluster size (capsules per pedicel) was generally lower at burnt sites than unburnt sites in the years following fire. However, neither variable showed a strong relationship with time since fire when averaged for each time since fire group.

The mean capsule density of each time since fire group and the unburnt group (all sites >25 years since fire pooled) is given in Figure 5.2. The 3, 7 and 9 years since fire groups were all comparatively lower than the unburnt group, but the 5 and 6 years since fire groups were comparatively greater than the unburnt.

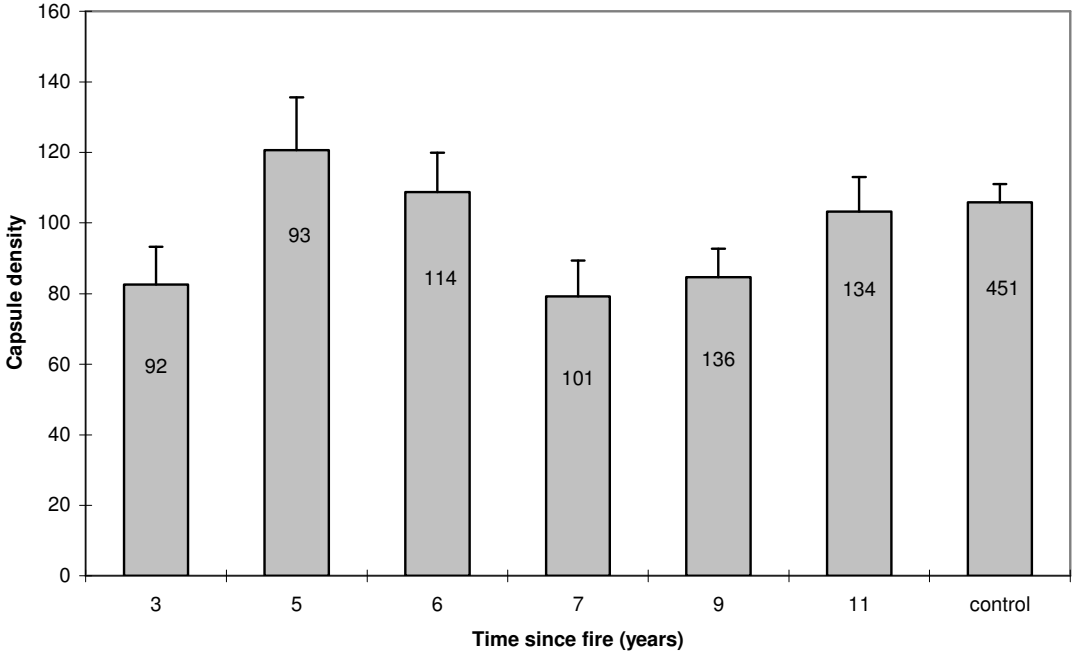


Figure 5.2. Mean (+SE) capsule density per tree for those trees fruiting in each time since fire group, and the unburnt group (>25 years since fire). The total number of fruiting trees (n) is indicated for each group.

However, the categorical data showed a stronger relationship between capsule density and time since fire (Figure 5.3). As time since fire increased, there were generally fewer trees in the low capsule density category and more trees in the medium capsule density category. The unburnt group had the lowest percentage of trees in the low capsule density category and the greatest percentage of trees in the higher capsule density categories. In contrast, the high capsule density category was relatively constant with time since fire, indicating that the high mean capsule density of the 5 and 6 years since fire groups was due to the presence of a small percentage of trees with an extremely high capsule density.

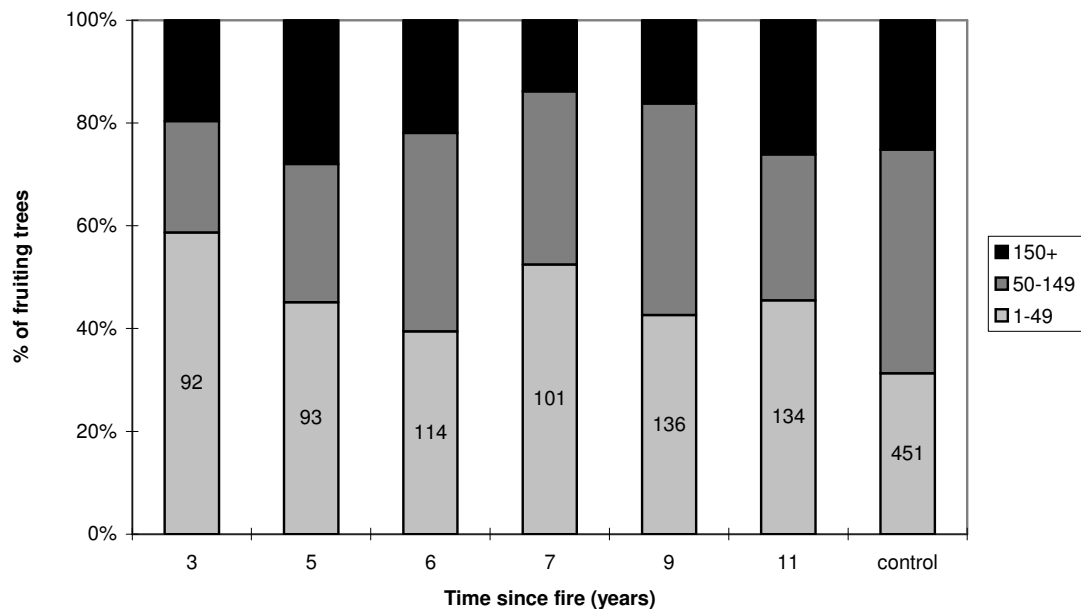


Figure 5.3. Percentage of trees with particular capsule densities (1-49 capsules per branch = low, 50-149 = medium, 150+ = high) for those trees fruiting in each time since fire group, and the unburnt group (>25 years since fire). The number of fruiting trees (n) is indicated for each group.

Mean crop size also showed little relationship with time since fire, with only the 7 years since fire group being substantially lower than the unburnt group, and the 5 years since fire group being comparatively higher (Figure 5.4).

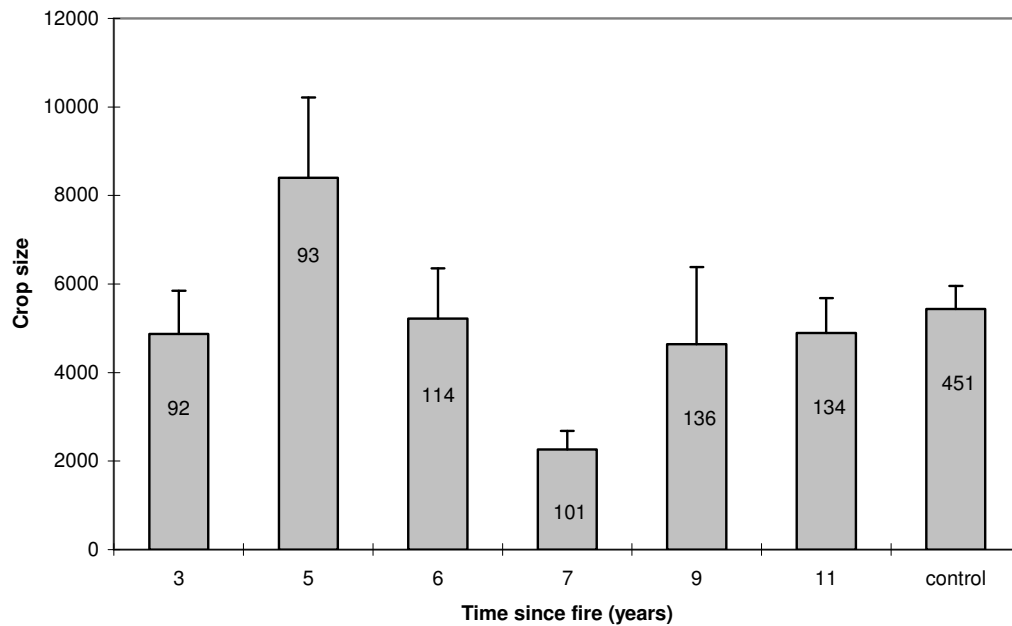


Figure 5.4. Mean (+SE) crop size per fruiting tree in each time since fire group, and the unburnt group (>25 years since fire). The number of fruiting trees (n) is indicated for each group.

The categorical data again showed the relationship between crop size and time since fire more clearly (Figure 5.5). The percentage of trees bearing low and medium crop sizes is lowest in the 3 years since fire group and highest in the unburnt group. As time since fire increased, a larger percentage of trees moved from the low crop size category into the medium crop size category, while those trees in the high crop size category were relatively constant with time since fire. Therefore, the high mean values for the 5 and 6 years since fire groups were caused by the presence of a few trees with large crop sizes.

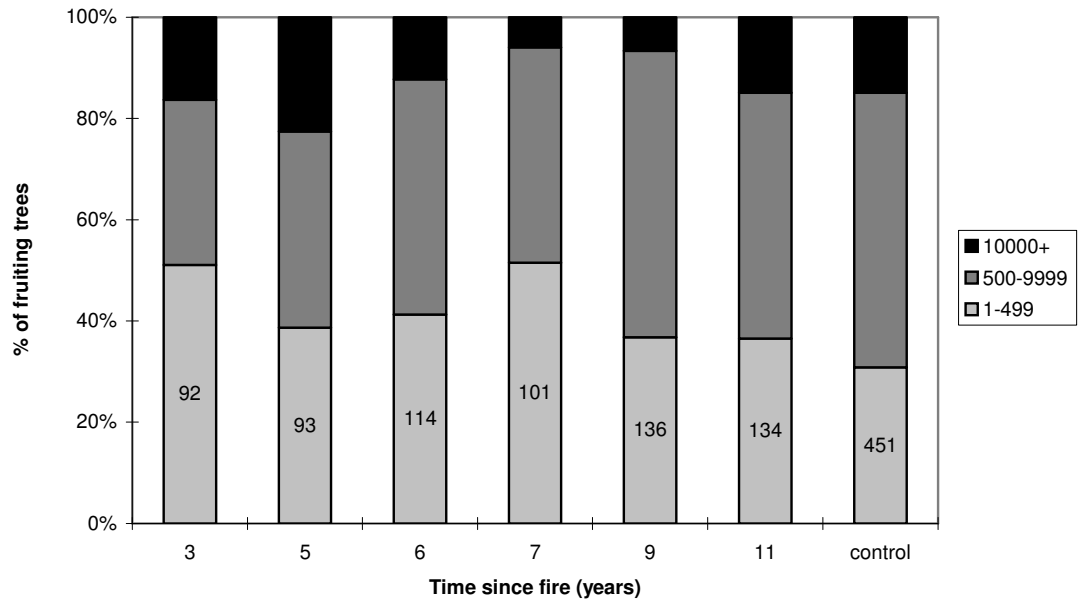


Figure 5.5. Percentage of trees with particular crop sizes (1-499 capsules per tree = low, 500-9999 = medium, 10000+ = high) for those trees fruiting in each time since fire group, and the unburnt group (>25 years since fire). The number of fruiting trees (n) is indicated for each group.

Mean cluster size showed a similar pattern to mean capsule density and mean crop size. Cluster sizes were comparatively lower in the 3, 7 and 9 years since fire groups but were comparatively higher in the 5 and 6 years since fire groups (Figure 5.6).

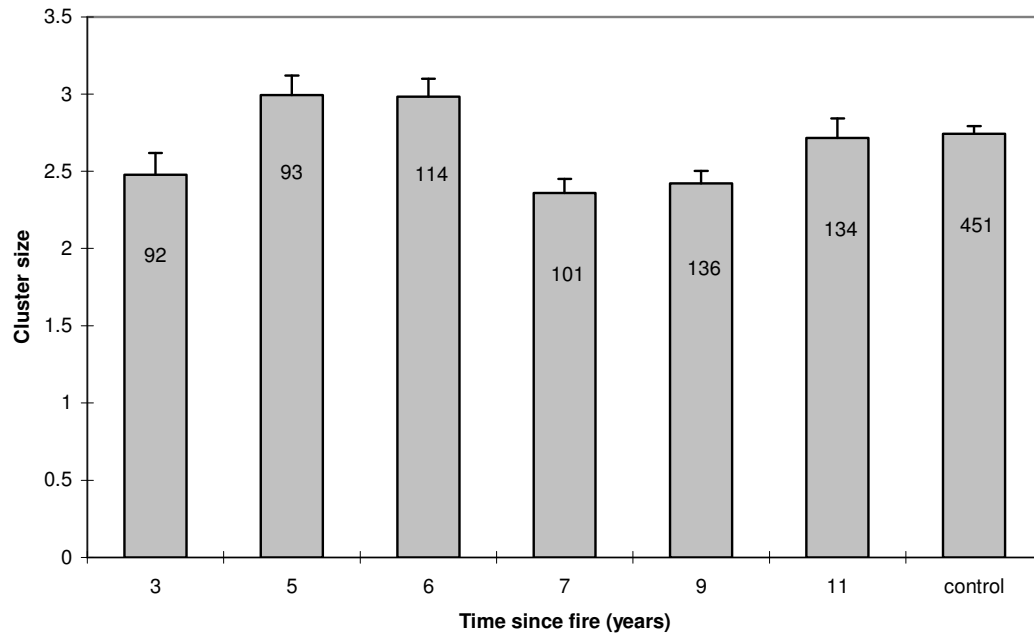


Figure 5.6. Mean (+SE) cluster size per tree for those trees fruiting in each time since fire group, and the unburnt group (>25 years since fire). The total number of fruiting trees (n) is indicated for each group.

The categorical data for cluster size showed a generally similar pattern to the means, with comparable distributions between the unburnt group and the 5 and 6 years since fire groups. However, the 3, 7, 9 and 11 years since fire groups all had a higher percentage of trees in the low category, and a lower percentage of trees in the high category, than the unburnt group (Figure 5.7).

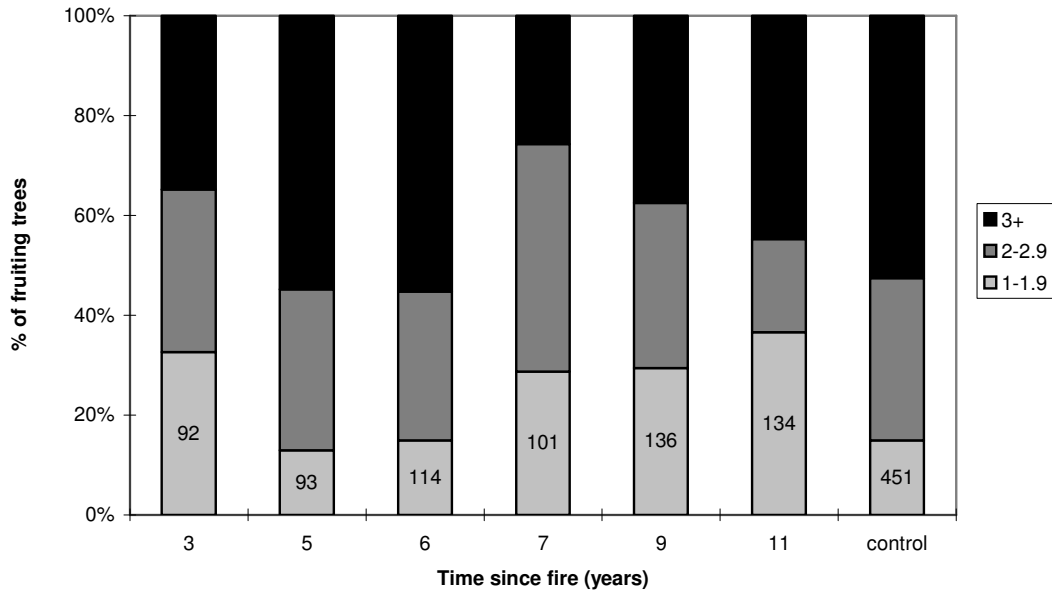


Figure 5.7. Percentage of trees with particular cluster sizes (1-1.9 capsules per pedicel = low, 2-2.9 = medium, 3+ = high) for each time since fire group, and the unburnt group (>25 years since fire). The number of fruiting trees (n) is indicated for each group.

Table 5.3 compares mean values for capsule density, crop size and cluster size between each time since fire group and its paired unburnt group. Capsule density, crop size and cluster size were all significantly lower for the 3 years since fire group than the paired unburnt group; capsule density and crop size were significantly lower for the 7 years since fire group than the paired unburnt group; and capsule density and cluster size were significantly lower for the 9 years since fire group than the paired unburnt group. The 5 and 6 years since fire groups were generally higher than their paired unburnt sites, but the differences were not consistent at the level of the individual site, and none were significantly different when pooled for each time since fire group.

Comparisons of means between each pair of burnt and unburnt sites (Table 5.3) showed large variability among sites, particularly for crop size and capsule density. The mean capsule density of fruiting trees ranged between 28 and 226, while the mean crop size of fruiting trees ranged between 743 and 11251. Burnt sites that were both higher and lower than their paired unburnt sites occurred in all time since fire groups, but the majority of site pairs were lower at burnt sites for all measures of capsule availability. Burnt sites were significantly lower than their paired unburnt sites at 8 (out of 19) site pairs for capsule density, 8 site pairs for crop size and 7 site pairs for cluster size. Burnt sites were significantly higher than their paired unburnt sites at 5 site pairs for capsule density, 4 site pairs for crop size and 2 site pairs for cluster size.

Table 5.3. Mean (\pm SE) capsule density, crop size and cluster size for those trees fruiting in each pair of burnt and unburnt sites (n = no. of fruiting trees). The data are summarised for each time since fire (TSF) group. Significant differences based on Wilcoxon Rank Sum tests are shown with asterisks (Levels: * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$).

| TSF | Site | Species | Status | Capsule density | Crop size | Cluster size | n |
|-----------|--------------------|--------------------|-----------------|--------------------|-------------------|------------------|----|
| 3 | Bjjuke | <i>E. baxteri</i> | burnt | 43 \pm 9 | 3183 \pm 1411 | 2 \pm 0.1 | 32 |
| | | | unburnt | 135 \pm 24** | 7370 \pm 1763** | 2.7 \pm 0.1** | 46 |
| | Kadnook | <i>E. arenacea</i> | burnt | 165 \pm 33 | 6050 \pm 1874 | 3.3 \pm 0.3 | 16 |
| | | | unburnt | 91 \pm 18* | 5841 \pm 2160 | 2.9 \pm 0.2 | 41 |
| | Kealys | <i>E. arenacea</i> | burnt | 104 \pm 20 | 7364 \pm 2225 | 2.9 \pm 0.3 | 31 |
| unburnt | | | 155 \pm 18* | 11251 \pm 2055* | 3.6 \pm 0.2* | 44 | |
| Tullich | <i>E. arenacea</i> | burnt | 28 \pm 11 | 1621 \pm 915 | 1.6 \pm 0.2 | 13 | |
| | | unburnt | 93 \pm 11** | 2794 \pm 717* | 3 \pm 0.1*** | 44 | |
| all | | burnt | 83 \pm 17 | 4870 \pm 977 | 2.5 \pm 0.1 | 92 | |
| | | unburnt | 120 \pm 10** | 6837 \pm 900* | 3 \pm 0.1*** | 175 | |
| 5 | Roseneath | <i>E. baxteri</i> | burnt | 50 \pm 11 | 3121 \pm 1401 | 2.4 \pm 0.2 | 18 |
| | | | unburnt | 49 \pm 9 | 1658 \pm 796 | 2.2 \pm 0.1 | 46 |
| | Tullich | <i>E. arenacea</i> | burnt | 226 \pm 28 | 1789 \pm 48 | 3.6 \pm 0.2 | 37 |
| | | | unburnt | 91 \pm 11** | 1372 \pm 33** | 3 \pm 0.1* | 42 |
| Yallakar | <i>E. arenacea</i> | burnt | 45 \pm 7 | 1799 \pm 512 | 2.6 \pm 0.1 | 37 | |
| | | unburnt | 87 \pm 8** | 3280 \pm 641* | 2.8 \pm 0.1 | 46 | |
| all | | burnt | 121 \pm 15 | 8398 \pm 1816 | 3 \pm 0.1 | 92 | |
| | | unburnt | 76 \pm 6 | 2573 \pm 417 | 2.7 \pm 0.1 | 134 | |
| 6 | Kalingur | <i>E. arenacea</i> | burnt | 133 \pm 24 | 10437 \pm 4692 | 3.4 \pm 0.3 | 41 |
| | | | unburnt | 136 \pm 16 | 10926 \pm 4130 | 2.9 \pm 0.2 | 47 |
| | Roseneath | <i>E. baxteri</i> | burnt | 112 \pm 18 | 6459 \pm 1856 | 2.4 \pm 0.2 | 41 |
| | | | unburnt | 49 \pm 9** | 1658 \pm 796** | 2.2 \pm 0.1 | 46 |
| Yallakar | <i>E. arenacea</i> | burnt | 85 \pm 10 | 2193 \pm 698 | 3.3 \pm 0.2 | 32 | |
| | | unburnt | 87 \pm 8 | 3280 \pm 641 | 2.8 \pm 0.2 | 46 | |
| all | | burnt | 109 \pm 11 | 5217 \pm 1140 | 3 \pm 0.1 | 114 | |
| | | unburnt | 91 \pm 7 | 4188 \pm 845 | 2.6 \pm 0.1 | 149 | |
| 7 | Cemetery Tk. | <i>E. arenacea</i> | burnt | 42 \pm 10 | 1758 \pm 937 | 2 \pm 0.1 | 24 |
| | | | unburnt | 49 \pm 10 | 3459 \pm 1288 | 1.9 \pm 0.1 | 42 |
| | Jilpanger | <i>E. arenacea</i> | burnt | 82 \pm 19 | 2236 \pm 799 | 2.5 \pm 0.2 | 32 |
| | | | unburnt | 136 \pm 16** | 7471 \pm 2204** | 2.9 \pm 0.1 | 37 |
| Tower Tk. | <i>E. arenacea</i> | burnt | 97 \pm 17 | 2552 \pm 552 | 2.4 \pm 0.1 | 45 | |
| | | unburnt | 185 \pm 20*** | 9342 \pm 2358** | 3 \pm 0.1*** | 47 | |
| all | | burnt | 79 \pm 10 | 2263 \pm 414 | 2.4 \pm 0.1 | 101 | |
| | | unburnt | 126 \pm 11* | 6879 \pm 1194*** | 2.6 \pm 0.1 | 136 | |
| 9 | Jilpanger | <i>E. arenacea</i> | burnt | 82 \pm 16 | 8550 \pm 5282 | 2.4 \pm 0.2 | 44 |
| | | | unburnt | 136 \pm 16* | 7471 \pm 2204* | 2.9 \pm 0.1* | 45 |
| | Kalingur | <i>E. arenacea</i> | burnt | 112 \pm 15 | 3152 \pm 805 | 2.5 \pm 0.1 | 47 |
| | | | unburnt | 137 \pm 17 | 7497 \pm 2253* | 2.9 \pm 0.1* | 46 |
| Nangwarry | <i>E. baxteri</i> | burnt | 58 \pm 9 | 2361 \pm 577 | 2.4 \pm 0.1 | 45 | |
| | | unburnt | 72 \pm 7 | 1906 \pm 543 | 2.6 \pm 0.1 | 48 | |
| all | | burnt | 85 \pm 8 | 4637 \pm 1744 | 2.4 \pm 0.1 | 136 | |
| | | unburnt | 104 \pm 9* | 4660 \pm 1155 | 2.7 \pm 0.1* | 142 | |
| 11 | Jilpanger | <i>E. arenacea</i> | burnt | 110 \pm 17 | 3798 \pm 1034 | 2.9 \pm 0.3 | 46 |
| | | | unburnt | 136 \pm 16 | 7472 \pm 2204* | 2.9 \pm 0.1 | 47 |
| | Nangwarry | <i>E. baxteri</i> | burnt | 160 \pm 17 | 9388 \pm 1773 | 3.4 \pm 0.2 | 48 |
| | | | unburnt | 72 \pm 7*** | 1906 \pm 543*** | 2.6 \pm 0.1*** | 48 |
| Roseneath | <i>E. baxteri</i> | burnt | 27 \pm 5 | 743 \pm 257 | 1.6 \pm 0.1 | 40 | |
| | | unburnt | 49 \pm 9* | 1658 \pm 796 | 2.2 \pm 0.1*** | 46 | |
| all | | burnt | 103 \pm 10 | 4888 \pm 790 | 2.7 \pm 0.1 | 134 | |
| | | unburnt | 86 \pm 7 | 3680 \pm 827 | 2.6 \pm 0.1 | 141 | |

Comparison of median values (means were highly skewed for these data) for all burnt sites combined and all unburnt sites combined showed that burnt sites were substantially lower than unburnt sites for all three variables. The median capsule density of trees at burnt sites was 56 compared to 75 at unburnt sites; the median crop size of trees at burnt sites was 812 compared to 1320 at unburnt sites; and the median cluster size of trees at burnt sites was 2.65 compared to 2.74 at unburnt sites.

Results of three way ANOVAs testing the effects of status (comparing all pooled burnt sites with all pooled unburnt sites), site, position and interaction terms on each (log-transformed) variable are given in Tables 5.4, 5.5 and 5.6. The effect of burn status was significant for capsule density, but not for crop size or cluster size. However, the test for the effect of burn status on crop size and cluster size was inconclusive, since the power of the test was 0.49 for crop size and 0.09 for cluster size (leaving a 51% and 91% chance of making a Type II error, respectively).

The effect of site was strongly significant for all 3 variables, as was the interaction effect of site and status. This indicates that the effect of burning varied depending on the site being considered. The effect of position was significant for all three variables, and the interaction between status, site and position was significant for capsule density and cluster size. However, the interaction effects for position and status and position and site were not significant for any variable, indicating that the effect of position was independent of burn status and site.

Table 5.4. Results of three way ANOVA testing the effects of status (burnt and unburnt groups), site, position (edge and interior) and interaction terms on capsule density per fruiting tree (log transformed, n = 8 trees per position).

| Source of variation | SS | df | MS | F | P |
|----------------------|-------|-----|------|------|---------|
| Total | 95.08 | 319 | | | |
| Status | 1.08 | 1 | 1.08 | 4.44 | 0.04 |
| Site | 9.05 | 9 | 1.01 | 4.12 | <0.0001 |
| Position | 1.06 | 1 | 1.06 | 4.33 | 0.04 |
| Status*site | 6.98 | 9 | 0.78 | 3.17 | 0.001 |
| Status*position | 0.1 | 1 | 0.1 | 0.42 | 0.52 |
| Site*position | 2.67 | 9 | 0.3 | 1.21 | 0.29 |
| Status*site*position | 5.75 | 9 | 0.64 | 2.62 | 0.006 |
| Error | 68.39 | 280 | 0.24 | | |

Table 5.5. Results of three way ANOVA testing the effects of status (burnt and unburnt groups), site, position (edge and interior) and interaction terms on crop size per fruiting tree (log transformed, n = 8 trees per position).

| Source of variation | SS | df | MS | F | P |
|----------------------|--------|-----|------|------|-------|
| Total | 328.76 | 319 | | | |
| Status | 3.51 | 1 | 3.51 | 3.83 | 0.051 |
| Site | 2.55 | 9 | 0.28 | 2.55 | 0.008 |
| Position | 4.16 | 1 | 4.16 | 4.16 | 0.04 |
| Status*site | 3.19 | 9 | 0.35 | 3.19 | 0.001 |
| Status*position | 1.18 | 1 | 1.18 | 1.18 | 0.28 |
| Site*position | 0.92 | 9 | 0.1 | 0.92 | 0.5 |
| Status*site*position | 1.16 | 9 | 0.13 | 1.16 | 0.32 |
| Error | 256.01 | 280 | 0.91 | | |

Table 5.6. Results of three way ANOVA testing the effects of status (burnt and unburnt groups), site, position (edge and interior) and interaction terms on cluster size per tree (log transformed, n = 8 trees per position).

| Source of variation | SS | df | MS | F | P |
|----------------------|-------|-----|-------|------|--------|
| Total | 4.84 | 319 | | | |
| Status | 0.005 | 1 | 0.005 | 0.36 | 0.55 |
| Site | 0.45 | 9 | 0.05 | 3.89 | 0.0001 |
| Position | 0.05 | 1 | 0.05 | 3.65 | 0.01 |
| Status*site | 0.28 | 9 | 0.03 | 2.43 | 0.01 |
| Status*position | 0.003 | 1 | 0.003 | 0.25 | 0.61 |
| Site*position | 0.09 | 9 | 0.01 | 0.8 | 0.61 |
| Status*site*position | 0.39 | 9 | 0.04 | 3.35 | 0.0006 |
| Error | 3.58 | 280 | 0.01 | | |

Pairs of burnt and unburnt sites categorised by capsule density (fruiting trees only)

The categorical data showed a stronger effect of burn status than the categorical data, for each pair of burnt and unburnt sites. Pairs of sites categorised by capsule density (excluding non-fruiting trees) are compared in Figures 5.8 and 5.9 (sample sizes for each site are given in Table 5.10). In general, the more recently burnt sites tended to have a greater percentage of trees with lower capsule densities, although a few sites showed the opposite trend. There were significant differences between paired burnt and unburnt sites for 13 site pairs, 10 of which showed a generally greater percentage of trees with lower capsule densities at burnt sites. Only 3 sites showed a significantly greater percentage of trees with higher capsule densities at burnt sites than unburnt sites (in the 3, 5 and 11 years since fire groups). Six site pairs were not significantly different between burnt and unburnt areas.

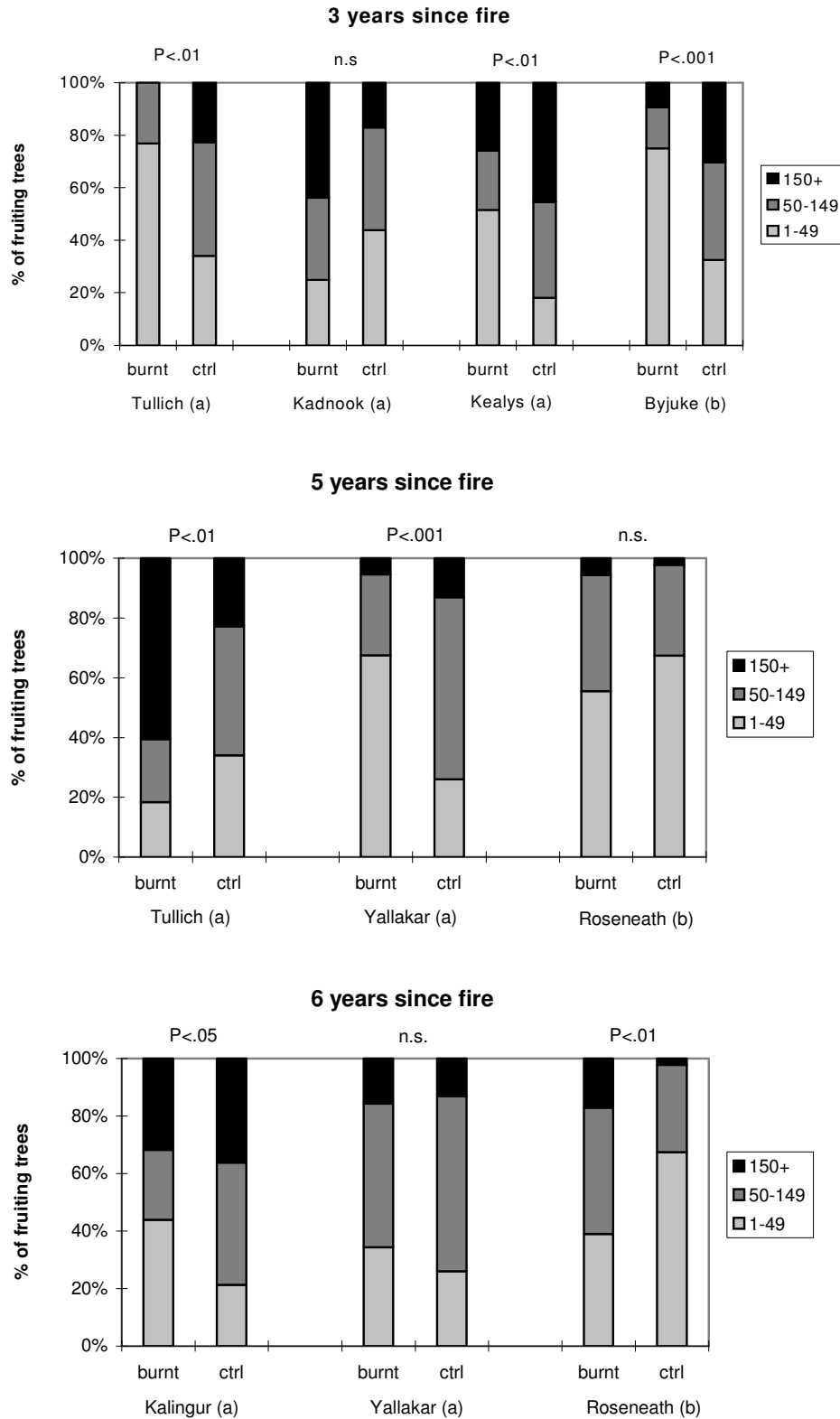


Figure 5.8. Percentage of fruiting trees with particular capsule densities (1-49 capsules per branch = low; 50-149 = medium; 150+ = high) for each pair of burnt sites and unburnt sites (a = *E. arenacea*, b = *E. baxteri*), within each time since fire group (3, 5 and 6 years since fire). The results of log-linear tests are given for each burnt-unburnt site pair comparison (n.s. = not significant).

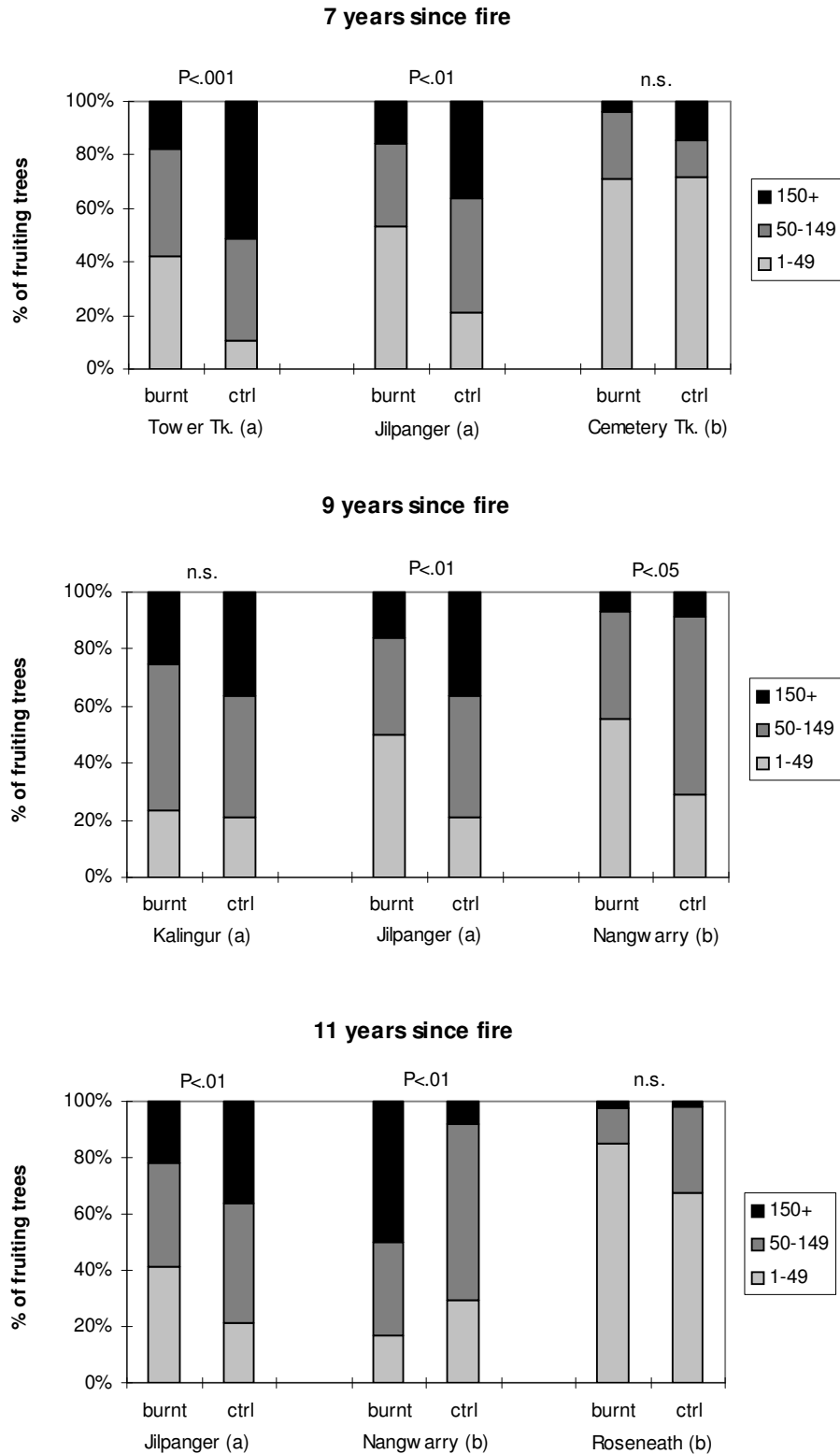


Figure 5.9. Percentage of fruiting trees with particular capsule densities (1-49 capsules per branch = low; 50-149 = medium; 150+ = high) for each pair of burnt and unburnt sites (a = *E. arenacea*, b = *E. baxteri*), within each time since fire group (7, 9 and 11 years since fire). The results of log-linear tests are given for each burnt-unburnt site pair comparison (n.s. = not significant).

Pairs of burnt and unburnt sites categorised by crop size (all trees)

Comparison of each pair of burnt and unburnt sites categorised by crop size (including non-fruiting trees) showed significant differences for all site pairs in the 3, 5, 6 and 7 time since fire groups, 2 out of 3 site pairs in the 9 years since fire group and 1 out of 3 site pairs in the 11 years since fire group (Figures 5.10 and 5.11). Sites in the 3 years since fire group had the highest percentage of non-fruiting trees and a generally lower percentage of trees in the upper crop size categories. As time since fire increases, there is a generally greater percentage of trees fruiting and a greater percentage of trees in the higher crop size categories.

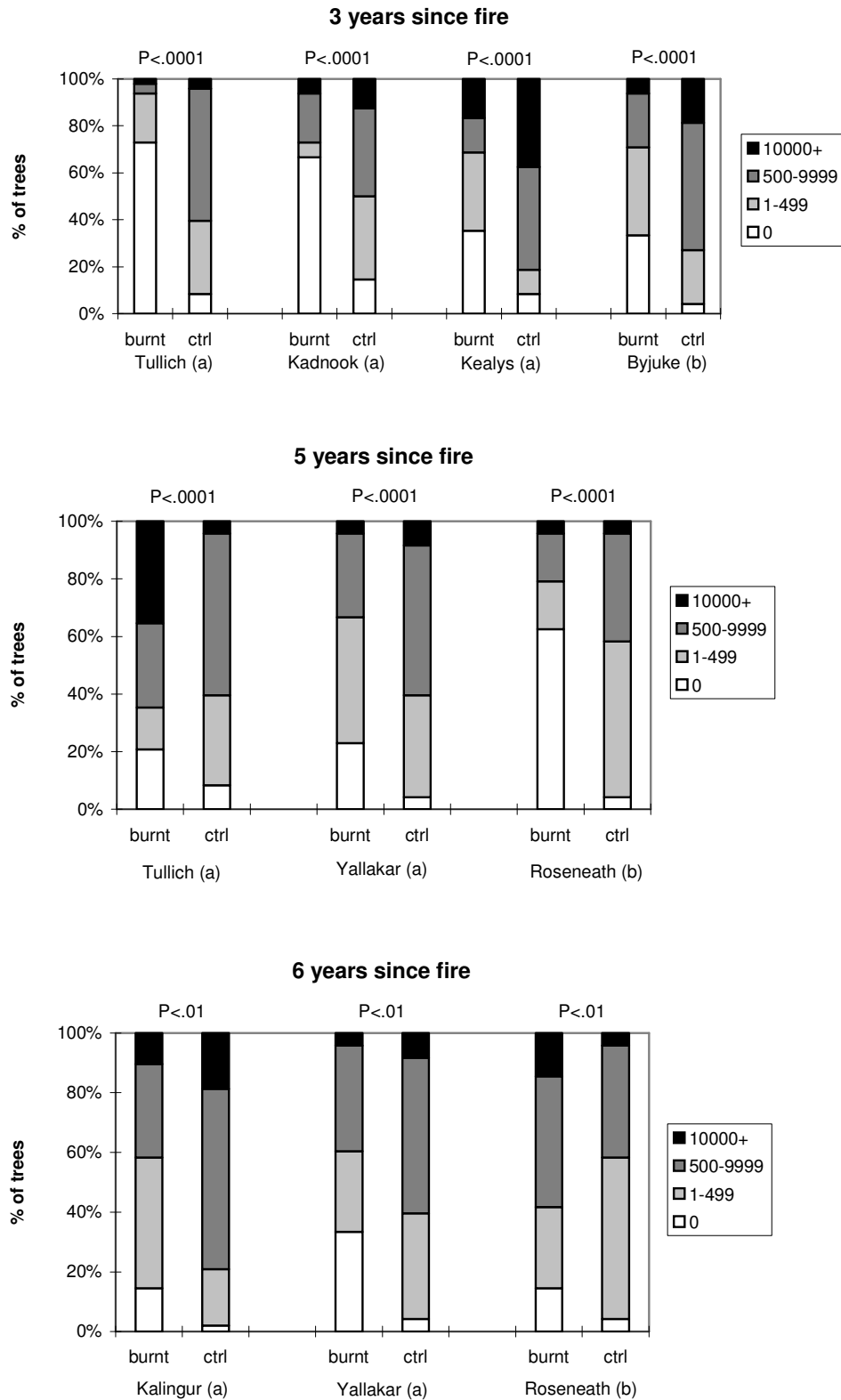


Figure 5.10. Percentage of trees with particular crop sizes (0 capsules per tree; 1-499 = low; 500-9999 = medium; 10000+ = high) for each pair of burnt and unburnt sites (a = *E. arenacea*, b = *E. baxteri*), within each time since fire group (3, 5 and 6 years since fire). The results of log-linear tests are given for each burnt-unburnt site pair comparison (n.s. = not significant, n = 48 trees per site).

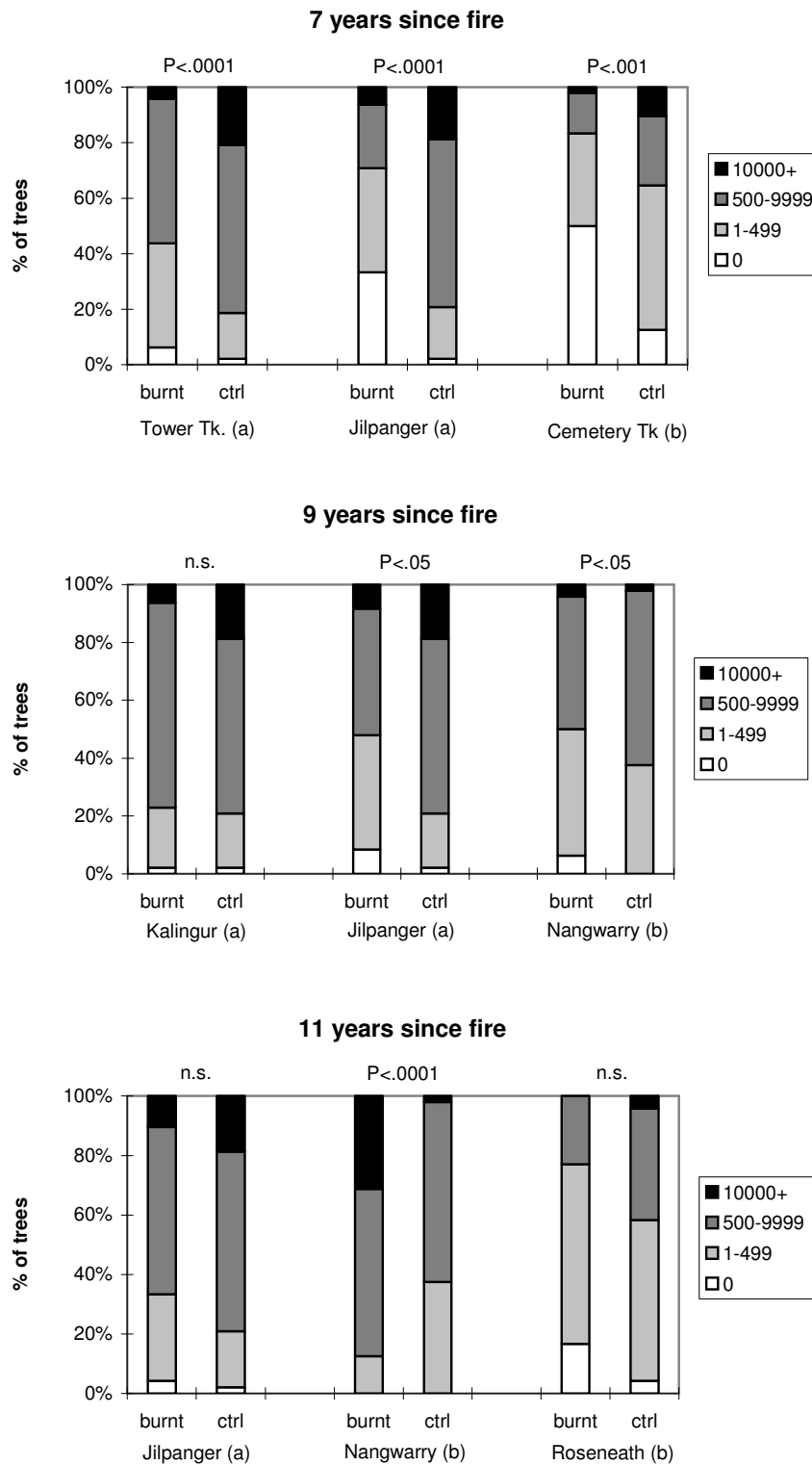


Figure 5.11. Percentage of trees with particular crop sizes (0 capsules per tree; 1-499 = low; 500-9999 = medium; 10000+ = high) for each pair of burnt and unburnt sites (a = *E. arenacea*, b = *E. baxteri*), within each time since fire group (7, 9 and 11 years since fire). The results of log-linear tests are given for each burnt-unburnt site pair comparison (n.s. = not significant, n = 48 trees per site).

Pairs of burnt and unburnt sites categorised by cluster size (fruiting trees only)

Pairs of burnt and unburnt sites categorised by cluster size (excluding non-fruiting trees) are compared in Figures 5.12 and 5.13. Cluster size did not show a strong relationship with time since fire. However, there was a higher percentage of trees in the lower cluster size categories at burnt sites than unburnt sites for 9 out of the 10 site pairs showing significant differences. One site pair showed a significantly greater percentage of trees with large cluster sizes at the burnt site, while the remaining 9 site pairs showed no significant differences.

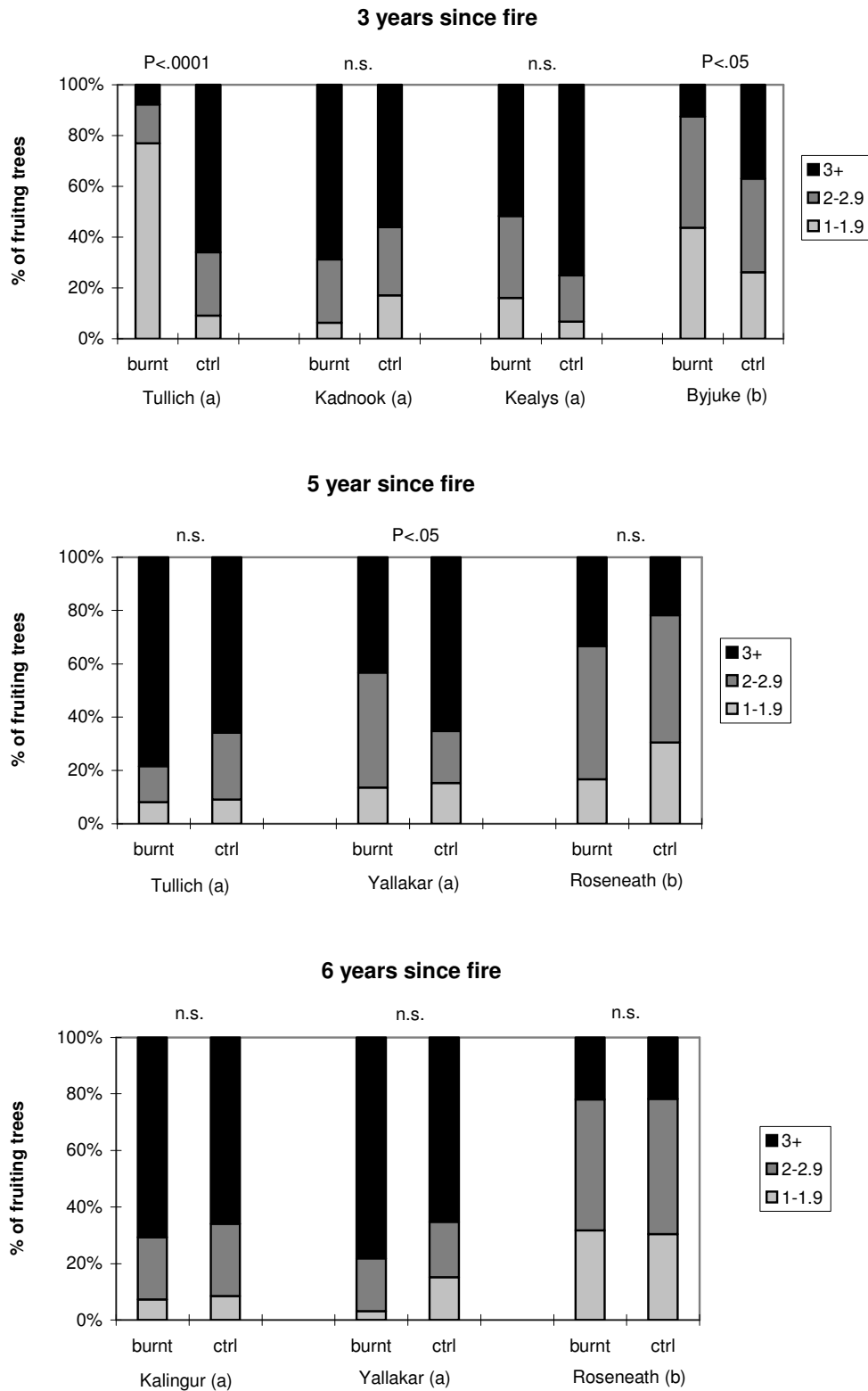


Figure 5.12. Percentage of fruiting trees with particular cluster sizes (1-1.9 capsules per pedicel = low; 2-2.9 = medium; 3+ = high) for each pair of burnt and unburnt sites (a = *E. arenacea*, b = *E. baxteri*), within each time since fire group (3, 5 and 6 years since fire). The results of log-linear tests are given for each burnt-unburnt site pair comparison (n.s. = not significant).

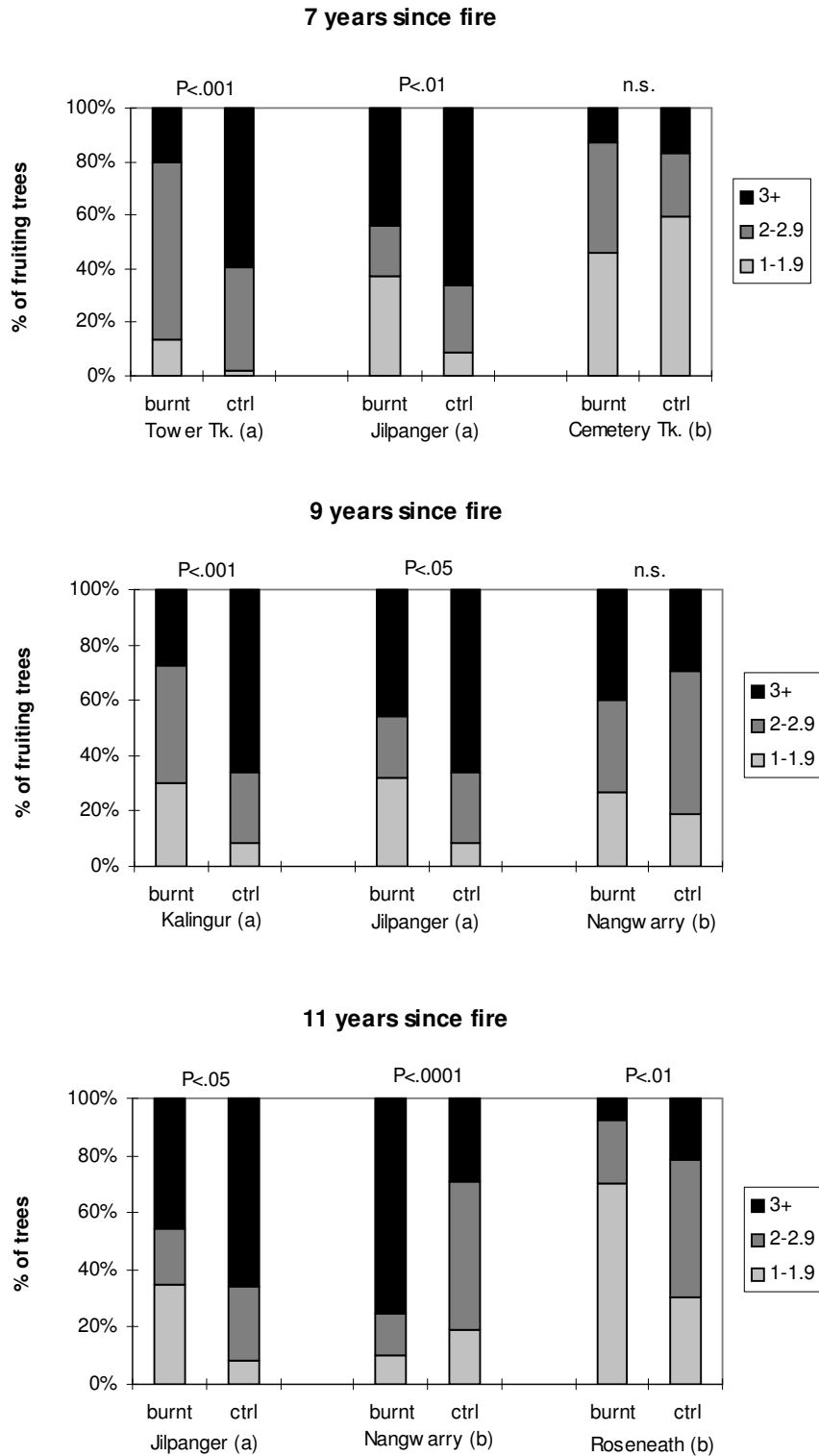


Figure 5.13. Percentage of fruiting trees with particular cluster sizes (1-1.9 capsules per pedicel = low; 2-2.9 = medium; 3+ = high) for each pair of burnt and unburnt sites (a = *E. arenacea*, b = *E. baxteri*), within each time since fire group (7, 9 and 11 years since fire). The results of log-linear tests are given for each burnt-unburnt site pair comparison (n.s. = not significant).

5.3.2 Capsule volume

There was no consistent pattern in capsule volume for the seven pairs of burnt and unburnt sites for which capsule volume was measured (Figure 5.14). Four pairs of sites were higher in capsule volume for unburnt areas than burnt areas and three pairs of sites were higher in capsule volume at the burnt site. Mean capsule volume ranged between 2362 mm³ and 4082 mm³. However, within site variability in capsule volume was relatively low. The mean (\pm SE) capsule volume of all burnt sites combined (3072 \pm 29, n = 105 trees) was comparable to that of all unburnt sites combined (3046 \pm 38).

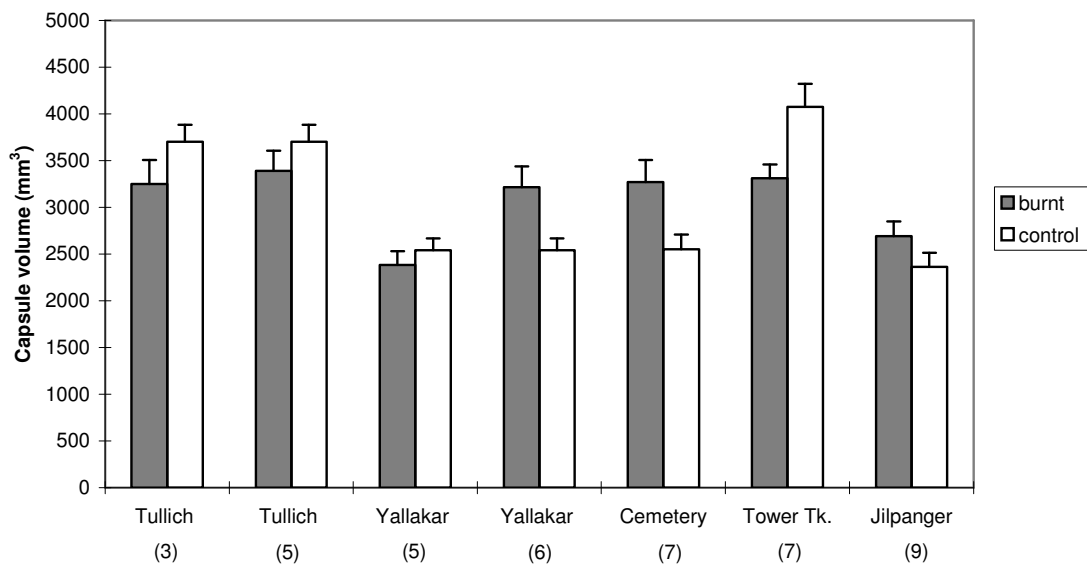


Figure 5.14. Mean (\pm SE) capsule volume (mm³) for each pair of burnt and unburnt sites (n = 15 trees per site). Capsule volume was averaged for 10 capsules per tree. The number of years since fire is given in brackets for each site.

A two way ANOVA testing for the effects of burn status, site and the interaction term showed no significant differences between burnt and unburnt groups, but the effects of site and the interaction term were significant (Table 5.7). The effect of burn status therefore depended on which site was being considered.

Table 5.7. Results of two way ANOVA testing the effects of status (burnt and unburnt groups), site and the interaction term on capsule volume (log transformed, n = 15 trees per site).

| Source of variation | SS | df | MS | F | P |
|---------------------|-------|-----|-------|-------|---------|
| Total | 2.56 | 174 | | | |
| Status | 0.005 | 1 | 0.005 | 0.45 | 0.5 |
| Site | 0.69 | 4 | 0.18 | 16.65 | <0.0001 |
| Status*site | 0.2 | 4 | 0.05 | 4.91 | 0.0009 |
| Error | 1.71 | 165 | 0.01 | | |

5.3.3 Patterns of habitat use by cockatoos after fire

In general, the cockatoos used a lower percentage of fruiting trees per sampling point and a lower percentage of fruiting branches per tree at burnt sites than unburnt sites. The percentage of trees used by cockatoos also showed a strong relationship with time since fire (Figure 5.16). The percentage of fruiting trees used in the 3 years since fire group (21%) was less than one third that of the unburnt group (66%). Values gradually increased towards unburnt levels as time since fire increased.

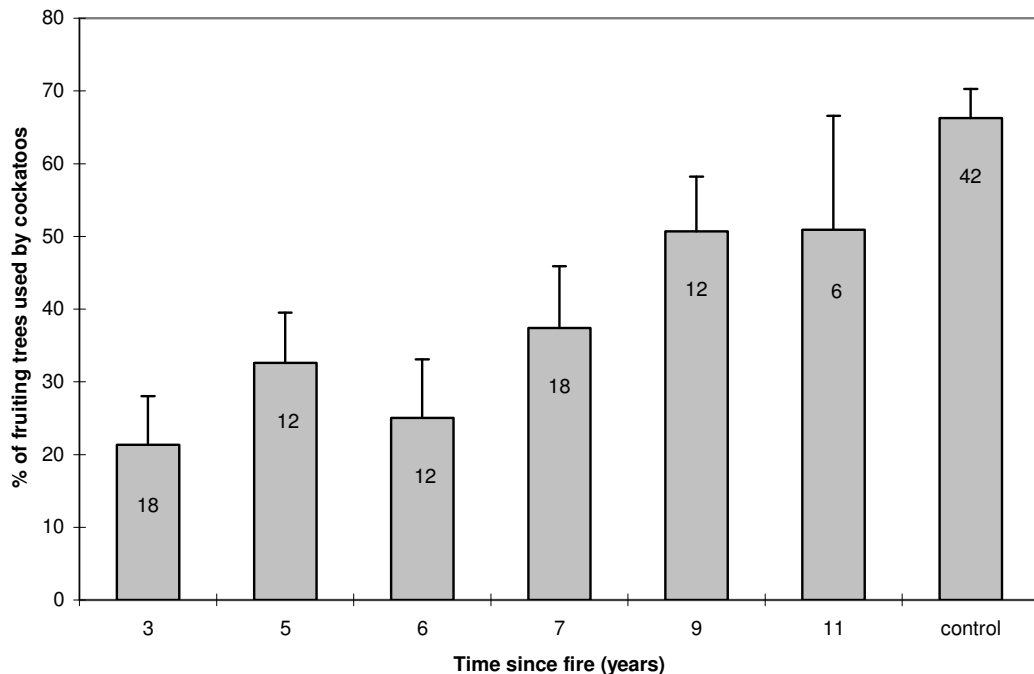


Figure 5.15. Mean (+SE) percentage of fruiting trees used by cockatoos in each time since fire group, and the unburnt group (>25 years since fire). *E. baxteri* sites have been omitted. The total number of replicate sampling points (n) is indicated for each group.

The percentage of fruiting branches used by cockatoos per tree showed no clear relationship with time since fire, but the unburnt group was comparatively higher than all time since fire groups (Figure 5.16).

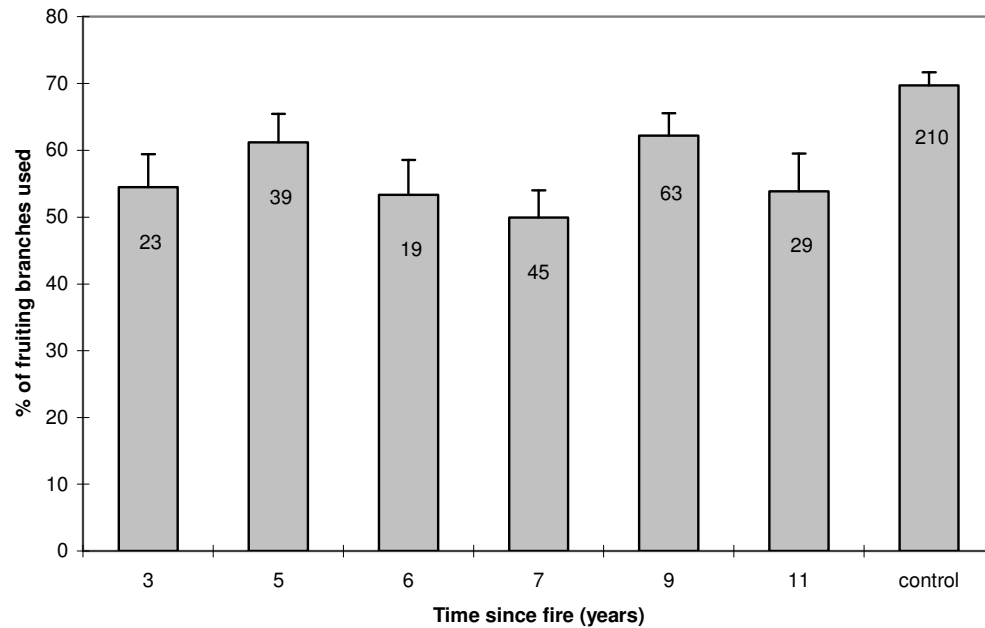


Figure 5.16. Mean (+SE) percentage of fruiting branches used by cockatoos in each time since fire group, and the unburnt group (>25 years since fire). *E. baxteri* sites have been omitted. The total number of trees used for foraging (n) is indicated for each group.

The percentage of fruiting trees used per sampling point and the percentage of fruiting branches used per tree are compared between each pair of burnt and unburnt sites in Table 5.8. The percentages of fruiting trees used by the cockatoos were substantially lower at burnt sites than unburnt sites for all site pairs and all time since fire groups. The percentage of fruiting trees used ranged between 0% and 51% at burnt sites and ranged between 45% and 88% at unburnt sites. Small sample sizes (n = 6 sampling points per site) precluded the statistical comparison of burnt and unburnt sites for individual site pairs, but differences were significant for the 3, 5, 6 and 7 years since fire groups when burnt and unburnt values were pooled for each time since fire group. The percentage of trees used in the 9 and 11 years since fire groups was also substantially lower than the paired unburnt sites for these groups, but the differences were not significant (but note also the smaller sample size for these comparisons).

The percentage of fruiting branches used per tree was generally lower at burnt sites than at unburnt sites, with some sites showing comparable values. The percentage of fruiting branches used ranged between 0% and 100%, but was usually around 50-60%. Burnt sites were significantly lower than unburnt sites for 3 site pairs. Comparison of each pair of time since fire and unburnt groups showed significantly lower values for the 5 and 7 years since fire groups, and no significant differences for the other groups.

Table 5.8. Mean (\pm SE) percentage of fruiting trees used per sampling point and percentage of fruiting branches used per (used) tree by cockatoos for each pair of burnt and unburnt sites (n = no. of sampling points for the percentage of fruiting trees used and n = no. of trees used by cockatoos for the percentage of fruiting branches used per tree). The data are summarised for each time since fire (TSF) group. Significant differences based on Wilcoxon Rank Sum tests are shown with asterisks (Levels: * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$). *E. baxteri* sites have been omitted.

| TSF | Site | Status | % of fruiting trees used by cockatoos | n | % of fruiting branches used by cockatoos | n |
|-----|--------------|---------|---------------------------------------|----|--|----|
| 3 | Kadnook | burnt | 28 \pm 15 | 6 | 49 \pm 10 | 9 |
| | | unburnt | 53 \pm 14 | 6 | 58 \pm 6 | 23 |
| | Kealys | burnt | 36 \pm 9 | 6 | 58 \pm 5 | 12 |
| | | unburnt | 59 \pm 11 | 6 | 53 \pm 5 | 26 |
| | Tullich | burnt | 0 \pm 0 | 6 | 0 \pm 0 | 0 |
| | | unburnt | 88 \pm 6 | 6 | 90 \pm 3 | 39 |
| | Combined | burnt | 21 \pm 7 | 18 | 54 \pm 5 | 23 |
| | | unburnt | 67 \pm 7*** | 18 | 55 \pm 4 | 49 |
| 5 | Tullich | burnt | 47 \pm 10 | 6 | 51 \pm 7 | 18 |
| | | unburnt | 88 \pm 6 | 6 | 90 \pm 3*** | 39 |
| | Yallakar | burnt | 19 \pm 5 | 6 | 68 \pm 6 | 7 |
| | | unburnt | 56 \pm 6 | 6 | 67 \pm 6 | 26 |
| | Combined | burnt | 33 \pm 7 | 12 | 56 \pm 5 | 25 |
| | | unburnt | 72 \pm 6* | 12 | 82 \pm 3*** | 65 |
| 6 | Kalingur | burnt | 6 \pm 4 | 6 | 41 \pm 37 | 2 |
| | | unburnt | 74 \pm 9 | 6 | 65 \pm 5 | 24 |
| | Yallakar | burnt | 44 \pm 11 | 6 | 55 \pm 5 | 17 |
| | | unburnt | 56 \pm 6 | 6 | 68 \pm 6 | 26 |
| | Combined | burnt | 25 \pm 8 | 12 | 53 \pm 5 | 29 |
| | | unburnt | 65 \pm 6* | 12 | 67 \pm 4 | 60 |
| 7 | Cemetery Tk. | burnt | 23 \pm 15 | 6 | 62 \pm 14 | 32 |
| | | unburnt | 45 \pm 9 | 6 | 65 \pm 4* | 42 |
| | Jilpanger | burnt | 51 \pm 16 | 6 | 51 \pm 16 | 24 |
| | | unburnt | 74 \pm 9 | 6 | 65 \pm 5 | 45 |
| | Kalingur | burnt | 47 \pm 14 | 6 | 52 \pm 9 | 8 |
| | | unburnt | 74 \pm 9 | 6 | 100 \pm 0* | 19 |
| | Combined | burnt | 37 \pm 9 | 18 | 59 \pm 10 | 64 |
| | | unburnt | 69 \pm 6* | 18 | 69 \pm 3** | 96 |
| 9 | Jilpanger | burnt | 54 \pm 7 | 6 | 60 \pm 5 | 24 |
| | | unburnt | 74 \pm 9 | 6 | 65 \pm 5 | 35 |
| | Kalingur | burnt | 47 \pm 13 | 6 | 67 \pm 6 | 23 |
| | | unburnt | 74 \pm 9 | 6 | 65 \pm 5 | 35 |
| | Combined | burnt | 51 \pm 8 | 12 | 63 \pm 4 | 55 |
| | | unburnt | 74 \pm 9 | 12 | 65 \pm 5 | 70 |
| 11 | Jilpanger | burnt | 51 \pm 16 | 6 | 54 \pm 6 | 24 |
| | | unburnt | 74 \pm 9 | 6 | 65 \pm 5 | 35 |

The mean percentage of fruiting trees used by cockatoos at all burnt sites combined (34%) was around half that used at all unburnt sites combined (66%). A three way ANOVA testing the effects of status, site, position and the interaction terms on the percentage of trees used per sampling point showed significant differences for burn status and site, but not for position or any of the interaction terms (Table 5.9). The effect of burn status was therefore independent of the effects of position and site.

Table 5.9. Results of three way ANOVA testing the effects of status (pooled burnt and unburnt groups), site, position (edge and interior) and interaction terms on percentage of trees used per sampling point (log transformed, $n = 3$ sampling points per position). *E. baxteri* sites have been omitted.

| Source of variation | SS | df | MS | F | P |
|----------------------|-------|----|------|-------|---------|
| Total | 32.16 | 83 | | | |
| Status | 8.29 | 1 | 8.29 | 31.65 | <0.0001 |
| Site | 4.85 | 6 | 0.81 | 3.09 | 0.01 |
| Position | 0.03 | 1 | 0.03 | 0.12 | 0.73 |
| Status*site | 2.24 | 6 | 0.37 | 1.42 | 0.22 |
| Status*position | 0.14 | 1 | 0.14 | 0.52 | 0.47 |
| Site*position | 0.36 | 6 | 0.06 | 0.23 | 0.97 |
| Status*site*position | 0.79 | 6 | 0.13 | 0.48 | 0.82 |
| Error | 15.45 | 56 | 0.28 | | |

Averaged over all burnt sites and all unburnt sites, the cockatoos used 58% of branches for foraging at burnt sites compared with 70% at unburnt sites. A three way ANOVA showed no significant differences for burn status, site, position or any of the interaction terms (Table 5.10). However, the test for the effect of burn status was inconclusive because its power was only 0.3, leaving a 70% chance of making a Type II error.

Table 5.10. Results of three way ANOVA testing the effects of status (pooled burnt and unburnt groups), site, position (edge and interior) and interaction terms on percentage of branches used by cockatoos per tree (log transformed, n = 4 trees per position). *E. baxteri* sites have been omitted.

| Source of variation | SS | df | MS | F | P |
|----------------------|---------|-----|------|--------|-------|
| Total | 8.38 | 111 | | | |
| Status | 0.15 | 1 | 0.15 | 2.12 | 0.15 |
| Site | 0.92 | 6 | 0.15 | 2.2 | 0.051 |
| Position | 0.00004 | 1 | 0.21 | 0.0006 | 0.98 |
| Status*site | 0.91 | 6 | 0.15 | 2.19 | 0.051 |
| Status*position | 0.01 | 1 | 0.01 | 0.15 | 0.7 |
| Site*position | 0.14 | 6 | 0.02 | 0.34 | 0.91 |
| Status*site*position | 0.41 | 6 | 0.07 | 0.99 | 0.44 |
| Error | 5.84 | 84 | | | |

5.4 Discussion

5.4.1 Influence of fire on seed availability to the south-eastern Red-tailed Black Cockatoo

Fire had a substantial and prolonged effect on the percentage of trees fruiting and on all aspects of capsule availability (capsule density, crop size and cluster size) likely to influence the selection of foraging sites by the cockatoos. The percentage of trees fruiting in burnt areas was significantly lower than in unburnt areas for all time since fire groups up to the 9 years since fire group, and trees that did produce a seed crop generally performed less well in terms of crop size, capsule density and cluster size over all time since fire groups. Although some sites recovered or exceeded unburnt levels in some aspects of capsule availability in the 5 and 6 years since fire groups, all burnt sites were less productive than their paired unburnt sites in at least one other aspect of capsule availability.

Variability in fire intensity probably accounts for much of the variability between sites, because trees burnt at higher intensity would be likely to have less resources available for seed production (Reekie and Bazzaz 1987). However, other climatic factors such as rainfall distribution and soil quality can also cause variation in reproductive performance between sites (Law *et al.* 2000), and may have been important sources of variability in the present study.

The overall effect of prescribed burns was to substantially reduce capsule availability over a 9 year post-fire period, with some effects persisting more than 11 years after fire. The mean percentage of trees fruiting at burnt sites (74%) was 0.79 of the percentage at unburnt sites (94%). Median crop size at burnt sites (812) was 62% of median crop size at unburnt sites (1320). When the effects of fire on both crop size and the percentage of fruiting trees were multiplied, trees in burnt areas produced

approximately half as many capsules as trees in long unburnt areas over a period of approximately 10 years (62×0.79). This estimate is simplistic, and assumes that long-unburnt trees represent “normal” levels of seed production, but it nevertheless provides a crude indication of the net reduction in seed availability for the cockatoos caused by prescribed burns. The estimate is also conservative, because the cockatoos used unburnt areas more intensively than burnt areas, and thus depleted the standing crop more in unburnt areas, despite efforts to reduce this bias.

The reduction in capsule availability corresponded to a similarly prolonged effect on levels of habitat use by the cockatoos. The cockatoos used a significantly lower percentage of trees in burnt areas than in unburnt areas for all time since fire groups up to and including the 7 years since fire group, with the percentage of trees used increasing with time since fire. Trees that were used for foraging by cockatoos in burnt areas were used less intensively than trees at unburnt sites for all time since fire groups. The cockatoos used less than half as many trees overall and an average of 82% as many fruiting branches in burnt areas. The implications of these findings for the Red-tailed Black Cockatoo population are discussed in the next section (below).

Most studies of seed production after fire have focussed on identifying the primary non-reproductive period (or time to first fruit set, Gill 1975). Although this is a biologically important measure, the present study demonstrated that the effects of fire on fecundity can linger long after this period, and that these effects can have an important impact on rates of habitat use by foraging animals. In this context, the time required by eucalypts to reach “typical” levels of seed production is probably more important than the time to first fruit set. Plants recovering from fire presumably need to invest more resources in the production of vegetative components, leaving less resources available for reproductive components (Reekie and Bazzaz 1987). The persistent effects of fire on fecundity are therefore likely to result from the continued partitioning of resources between canopy restoration and seed production. Similar effects have been noted for *Callitris collumllaris* by Hawkins (1966) and for *Banksia* sp. by Zammit and Westoby (1987).

There have been no comparable studies of seed production after fire in eucalypts to the authors knowledge, but anecdotal observations suggest that other eucalypt species may also require lengthy periods to fully regain their reproductive performance after fire. Among mallee eucalypts, first flowering on resprouting trees occurred at four years and four months after fire in *Eucalyptus gracilis* and at seven years after fire in *E. foecunda* (Noble 1985). Anecdotal observations of other South Australian eucalypt species indicate that first appearance of buds ranges from 1.99 in *E. leucoxyton* to 4.41 years for *E. obliqua* (Choate 1997). First seed set would occur between one and three years after bud initiation,

depending on the species. These observations indicate that the time to initial seed set is around six years in some eucalypt species. The time required by these eucalypts to reach normal levels of productivity may be much longer, although response times would vary according to fire intensity and other fire regime components. Although the present study targets the Red-tailed Black Cockatoo, it has important implications for beekeepers and a wide range of species using eucalypt floral resources. It also highlights an important gap in our knowledge of fire ecology. More research is needed to determine the effects of fire on eucalypt fecundity for other species, and the implications of these effects for birds and mammals that depend on eucalypt resources for their food supply.

5.4.2 Implications for the south-eastern Red-tailed Black Cockatoo

In the previous chapter, I reported that the cockatoos are likely to be limited by food supply, based on the high percentage of the day spent foraging and the increasing percentage of time spent foraging as the seed crop aged and resources were depleted. I also demonstrated that low capsule availability at the level of the branch significantly reduces foraging efficiency. The intensive use of resources documented in the present study provides further evidence to suggest that food supplies may be limiting. Of the *E. arenacea* sites surveyed, all unburnt sites and all but one burnt site had been used for feeding, and up to 88% of trees were used at some unburnt sites. Furthermore, many sites in the present study were used extensively and subsequently revisited by different flocks of cockatoos over the course of the study (Koch pers. obs.).

The high levels of habitat use at burnt sites suggested that food supplies were limiting, because the cockatoos tend to select areas for foraging with high capsule densities, crop sizes and capsule densities (Chapter 4). Assuming that they did this to maximise their foraging efficiency, they may be expected to avoid burnt areas altogether. However, the value of burnt areas as feeding habitat is likely to depend on temporal aspects of food supply. At the time of the survey, the cockatoos had been feeding on the same seed crop (*E. arenacea*) for more than two years, and the seed crop would have been relatively depleted at this time because a new seed crop typically replaces the old seed crop every one to two years (Chapter 3). Individual trees can support several hours of foraging activity and large, productive trees may support a small flock (four or five birds) for an entire day (Koch pers. obs.). Therefore, the presence of a few trees with particularly large crop sizes may help counter increased search times in burnt areas to some extent, particularly in situations when food supplies are generally low. However, with fewer trees producing large seed crops and many trees producing no seed crop at all, cockatoos feeding in burnt areas would generally have to forage over a much greater area than those feeding in long unburnt areas, and this would no doubt reduce their foraging efficiency. Given the high rates of

habitat use in long unburnt areas, it is likely that the cockatoos foraged in burnt areas as a result of resource limitation in these areas.

The combined evidence from the present study and the previous two chapters strongly suggests that food supplies are limiting, and that this limitation may be severe in years when the interval between seed crops is prolonged. In such years, fewer cockatoos are likely to increase their net energy gain beyond that required for survival, and breeding attempts may be less frequent and have higher failure rates.

The consequences of prescribed burns for the cockatoo population as a whole depend on the percentage of habitat that is less than 10 years post-fire, since this is the approximate duration of fire effects on both capsule availability and habitat use by the cockatoos (although some effects persist for more than 11 years after fire). Approximately 13% of stringybark woodlands are less than 10 years post-fire in Victoria, within the range of the Red-tailed Black Cockatoo (Cooke pers. comm.). The percentage may be substantially higher in South Australia, but Victoria makes up 88% of the current stringybark habitat within the range of the cockatoo. If the percentage of habitat burnt in the last 10 years is estimated to be 50% for South Australia, then the total percentage of stringybark habitat less than 10 years post fire is 17% ($(0.13 \times 88) + (0.5 \times 12)$); and the overall reduction in food availability attributable to fire is around 9% (a reduction in crop size of 0.51×17), across the range of the cockatoo.

If the total percentage of habitat less than 10 years post-fire was reduced to 10%, then this would increase total food availability by approximately 5%. If the total percentage of habitat less than 10 years post-fire was reduced to 0%, then this would increase total food availability by 9%. If the intensity of burns can be reduced to the point where very low levels of canopy scorch are produced, then this could also increase total food availability by up to 9%.

These calculations are simplistic, because the extent to which burnt areas are used by cockatoos is likely to depend on temporal aspects of food availability (discussed in Chapter 3), and consequently, food availability is a relative term. Nevertheless, they provide a crude estimate of what can realistically be expected from changes in fire management. The expected increases in food supply are much lower than target increases outlined in the recovery plan (50%, Burnard and Hill 2001). Therefore, this objective is probably unrealistic to some extent because target increases depended largely on improved fire management. Improved fire management will need to be supplemented with revegetation on a broad scale to achieve higher percentage increases. However, improved fire management is the only way to increase food supply to the cockatoos in the short term, because the planted trees would

probably take at least 15 years to reach levels of capsule availability comparable to those of established trees (Koch pers. obs.).

Prescribed burns reduced resource availability for the Red-tailed Black Cockatoo at the level of the site and the individual tree. At each recently burnt site there were fewer fruiting trees available for use by cockatoos and a greater percentage of trees were lower in crop size, capsule density and cluster size. Apart from the reduction in foraging efficiency associated with low capsule densities and cluster sizes per branch (Chapter 4), the lower density of suitable trees would be likely to increase search times, while the lower density of capsules on trees and branches would be likely to increase the frequency of movements within and between trees, because each branch and each tree presumably supports less foraging activity. Increased search times associated with fragmentation of seed supply can have a large impact on nesting success for Carnaby's Cockatoo (Saunders 1980). Similar effects associated with low capsule availability are likely to limit the nest success of the south-eastern Red-tailed Black Cockatoo, and the proportion of the birds that attempt to breed (Chapter 4).

It is likely that the effects of prescribed burns on capsule availability can be reduced substantially by reducing the intensity of burns, since this will reduce levels of canopy scorch and hence the resources that trees need to invest in canopy replacement after fire. The effects of fire intensity are considered in Part B of this chapter, and the implications of these two studies for fire management in the range of the cockatoo are considered in Part C.

Part B: Recovery of seed availability for the south-eastern Red-tailed Black Cockatoo following burns at different intensities

5.5 Introduction

Fire intensity is an important component of the fire regime influencing the responses of plants recovering from fire (Cremer 1962, Hawkins 1966, Gill and Groves 1980, Moreno and Oechel 1991, Wardell-Johnson 2000). Fire intensity is particularly important where eucalypts are concerned because low intensity fire generally produces defoliation of only the lower branches, whereas high intensity fire results in complete canopy defoliation and slower canopy replacement (Gill 1981, McCaw *et al.* 1995). If increased resource expenditure on canopy replacement results in fewer resources being available for reproductive functions (Reekie and Bazzaz 1987), higher levels of canopy scorch should result in greater and more prolonged effects on seed production. Setterfield (1997) demonstrated that increased fire intensity caused a decline in fecundity for two tropical savanna eucalypts, *E. tetradonta* and *E. miniata*, but there have been few other studies addressing the effects of fire on eucalypt fecundity.

The first part of this chapter demonstrated that prescribed burns have a substantial and prolonged effect on seed availability for the south-eastern Red-tailed Black Cockatoo. The present study aimed to compare the effects of partial and complete canopy scorch on capsule availability (capsule density, crop size and cluster size) to the Red-tailed Black Cockatoo, and to compare the persistence of these effects between canopy scorch groups.

5.5.1 Study sites

The study sites included a number of remnants in the Casterton, Edenhope and Harrow regions of southwest Victoria. The location of study sites is shown in Figure 2.2. Study sites composed of *E. arenacea* were: Harrow, Yallakar, Kadnook, Kealys Waits and Tullich State Forests. Sites composed of *E. baxteri* were Roseneath and Byjuke State Forests.

Strip burns, the burning of 100-500m wide strips along the perimeters of remnants or fire breaks, are performed in parts of the Red-tailed Black Cockatoo range (Edenhope and Casterton regions). The burns are typically divided by fire breaks into suitably sized segments (usually 100-1000m in length) so that small burns can be performed during favourable weather conditions. Although the burns typically result in complete canopy scorch, the different segments are occasionally burnt at different intensities. This provides an excellent opportunity to study the effects of varying canopy scorch over a small area,

which reduces the influence of other confounding effects associated with such variables as soil type and rainfall distribution.

5.6 Methods

5.6.1 Selection of trees and sites with different levels of canopy scorch

The effects of canopy scorch were studied in three time since fire classes (2, 4 and 6 years since fire) to determine whether the effects of canopy scorch on capsule availability recorded in part A of this chapter are as persistent in partially scorched trees as those in completely scorched trees. A site was considered to be suitable for the study if it included segments that ranged in intensity according to the groups specified below. There were initially three replicate sites for each time since fire class, but one site (Waits) was subsequently burnt by wildfire and was not included in the survey.

The fire intensity for a given site segment (referred to hereafter as a canopy scorch group) was visually assessed according to an overall estimation of canopy scorch. At each site, canopy scorch areas were selected that comprised the following canopy scorch groups: (1) high canopy scorch (100% scorch), (2) moderate canopy scorch (approximately 50% scorch), (3) low canopy scorch (10-20% scorch) and (4) unburnt (>25 years since fire). These estimations were made less than 1 year post-fire for the 2 years since fire class, at 3 years post-fire for the 4 years since fire class and at 5 years post-fire for the 6 years since fire class. It was not possible to accurately measure canopy scorch for the 4 and 6 years since fire classes, since at these sites the dead leaves had fallen from the trees. However, it was possible to determine approximate levels of canopy scorch by estimating the number of defoliated branches. Due to the imprecise nature of this measurement, canopy scorch categories were simplified in these age classes as: (1) high canopy scorch (100% scorch), (2) partial canopy scorch (less than 50% scorch) and (3) unburnt.

Although it is more common to directly measure fire intensity using a measure such as Byram's fire line intensity (heat content of fuel x fuel load x rate of forward spread, Byram 1959), it was not deemed necessary for the purposes of the present study. Simple biological measures such as leaf scorch height (the height of leaves killed by radiant heat) are often used to indicate fire intensity (eg. Koch and Bell 1980, Strasser *et al.* 1996), and have the advantage of predicting intensity at the level of the tree as well as at the level of the site. Biological measures thus eliminate the problems caused by single measures of intensity which do not account for the high variability in fire behaviour within sites (Hobbs and Atkins 1988). The ideal biological measure for fire intensity is probably the extent of crown dieback (the extent to which branches are killed by fire), which is measured as the diameter of branches killed by fire, or

terminal twig diameter (Cheney, pers. comm.). However, the height of trees precluded the measurement of terminal twig diameter in the present study.

In each canopy scorch area, ten trees were selected that had a level of canopy scorch matching the overall level of canopy scorch for the area. Hence there were 50 trees sampled per site for the 2 years since fire class (which included four canopy scorch groups) and 40 trees sampled per site for the four and six years since fire classes (which included three canopy scorch groups). Unburnt areas were typically adjacent areas that were separated from burnt areas by a minimum of 100m. The time since fire of each site was confirmed using fire history maps of each site, and by consultation with fire management authorities (Department of Environment and Natural Resources and Forestry SA staff). Each tree selected was allocated a numbered, metal tag that was nailed to the tree for identification. Flagging tape was also tied around the trunk of trees to increase their visibility upon revisitation. The position of each tree or group of trees was recorded using a Geographic Positioning System (GPS). Each canopy scorch area covered an area of approximately 1-5 ha. The trees were revisited after initial measurements were taken because not all sites fruited in the same year (details of these measurements are given below).

Trees within 10m of an induced edge, defined as any perimeter of a remnant bordering cleared land, were avoided, since edge effects contribute to variability in seed production among trees (the subject of Chapter 6). Percentage canopy scorch was determined for individual trees from counts of scorched branches (those with dead leaves) and unscorched branches. Tree size is also an important factor influencing rates of recovery from crown scorch in eucalypts (Gill 1978). For this reason, efforts were made to select trees in a range of sizes within each canopy scorch area, based on a visual assessment of the girth and height of trees.

5.6.2 Reproductive performance of trees in different canopy scorch groups

Measurements of capsule availability and bud production were made in summer 2001, and were repeated in summer 2002. *E. baxteri* sites produced a seed crop in 2000 and *E. arenacea* sites produced a seed crop in 2001. Furthermore, each species of stringybark produced a single cohort of buds over the study period. For this reason, the data collected in 2001 were combined with those collected in 2002. However, the same time since fire classes were used for both species, for the purposes of the present study.

Measurements of capsule availability were: capsule density (number of capsules per branch), crop size (number of capsules per tree), and cluster size (number of capsules per pedicel). The variables used are the same as those used in part A of this chapter, with the exception that the percentage of fruiting trees was not distinguished from crop size (non-fruiting trees were included in crop size data), in order to preserve sample sizes. Details of these measurements are given in Section 2.2.

5.6.3 The vegetative responses of trees to different levels of canopy scorch

The number and position (base, trunk or canopy) of epicormic branches per tree were used as measures of vegetative recovery after fire. Epicormic branches were defined as those branches that emerged from the fibrous-barked branches, as distinct from the smooth-barked branches that form the outermost foliage in the stringybarks studied (Koch pers. obs.). Epicormic branches at the base position were those emerging from the lignotuber. Epicormic branches at the trunk position were those emerging between the base of the tree and the first main (non-epicormic) branch that emerged from the trunk. Epicormic branches at the canopy position included all branches occurring above the first main branch of the tree.

Although ideally crown recovery should be considered in relation to other variables such as leaf weight and branching patterns (Gill 1978), the number and position of epicormic branches provides a useful comparative indication of vegetative regeneration between canopy scorch groups.

5.6.4 Statistical analysis

Two way ANOVAs were used to test for the effects of canopy scorch, site and the interaction between canopy scorch and site on crop size and number of epicormic branches. The data were log-transformed to improve normality, but were still skewed for some "cells" (canopy scorch groups at each site) in the case of crop size. However, the normality of the data was judged to be sufficient for the analysis, given

the robust nature of ANOVA (Underwood 1981). Dunnett's tests were used to compare each canopy scorch group with its unburnt group. Although sample sizes were initially equal, some trees could not be relocated after the metal tags and flagging tape fell off. Sample sizes were therefore balanced by randomly deleting the appropriate number of trees from the analysis where necessary, as suggested by Zar (1984).

In addition, the crop size data were converted into categorical data as the percentage of trees with crop sizes of 0, 1-499, 500-9999 and 10000+ capsules per tree (the same categories used in the part A of this chapter), to show trends in the data not apparent from the means.

The sample sizes were grossly unequal for capsule density and cluster size due to the exclusion of non-fruiting trees. Two way ANOVAs were not used to analyse these data, since the power of this test is substantially reduced by unequal sample sizes (Zar 1984). Although samples can be deleted to equalise sample sizes (Zar 1984), this procedure was not used here in order to preserve sample sizes. Kruskal-Wallis tests were thus used to test the effect of canopy scorch on these variables. These tests were also used to test for differences between canopy scorch groups for each individual site. Wilcoxon Rank Sum tests were used to compare each intensity group with the unburnt group.

5.7 Results

5.7.1 Reproductive performance of trees in different canopy scorch groups

Crop size (capsules per tree)

In general, patterns in crop size showed the expected trend with increasing canopy scorch at 2 and 4 years since fire, but showed no consistent relationship with canopy scorch at 6 years since fire (Figure 5.17). In the 2 years since fire class, crop size declined as the level of canopy scorch increased, although means were comparable for the low and moderate canopy scorch groups. Mean crop size ranged between 10191 capsules per tree for the unburnt group and 742 capsules per tree for the high canopy scorch group in this time since fire class. In the 4 years since fire class, crop size was substantially lower in the high canopy scorch group (4383 capsules per tree) than the unburnt group (8864 capsules per tree), and was comparable between the low canopy scorch and unburnt groups. In the 6 years since fire class, however, crop size was comparable among the unburnt and high canopy scorch groups, and was highest in the low canopy scorch group.

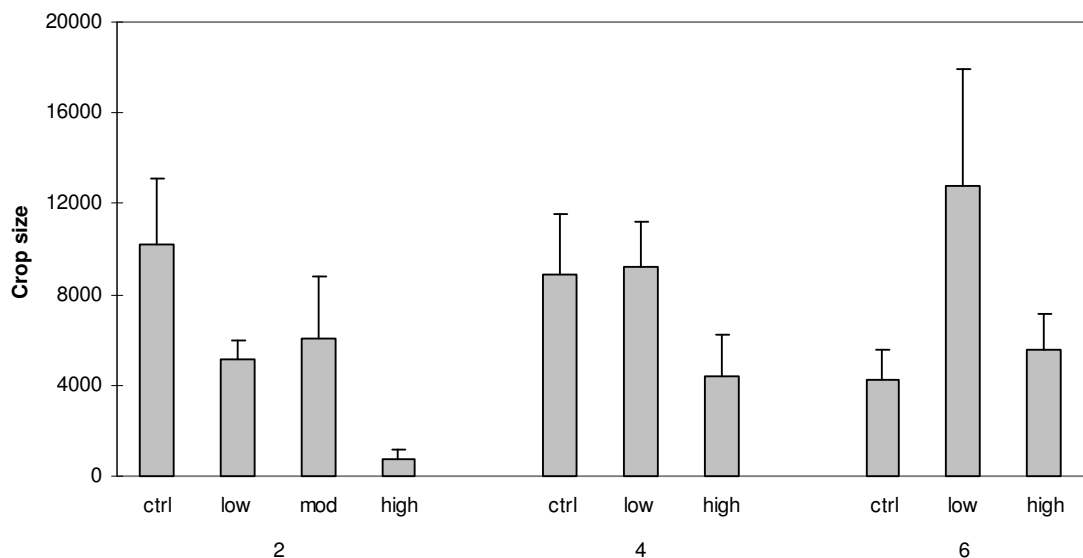


Figure 5.17. Mean (+SE) crop size (number of capsules per tree) in each canopy scorch group at 2, 4 and 6 years since fire ($n = 30$ trees per canopy scorch group for the 2 years since fire class, 24 trees for the 4 years since fire class and 16 trees for the 6 years since fire class).

The percentage of trees with particular crop sizes (0, 1-499 = low, 500-9999 = medium and 10000+ = high) is given for each canopy scorch group at 2, 4 and 6 years since fire in Figure 5.18. These data indicated that the effects of canopy scorch influenced the crop size of trees as well as the percentage of fruiting trees. At 2 and 4 years since fire, the percentage of trees with large crop sizes (greater than 500 capsules) declined with increased canopy scorch. Trees with high levels of canopy scorch had the lowest percentage of fruiting trees and the lowest percentage of trees with large crop sizes (more than 500 capsules). Trends in the categorical data for the 6 years since fire class reflected trends in mean crop size, with comparable levels between the high canopy scorch and unburnt groups, and the greatest percentage of trees with large crop sizes occurring in the low canopy scorch group.

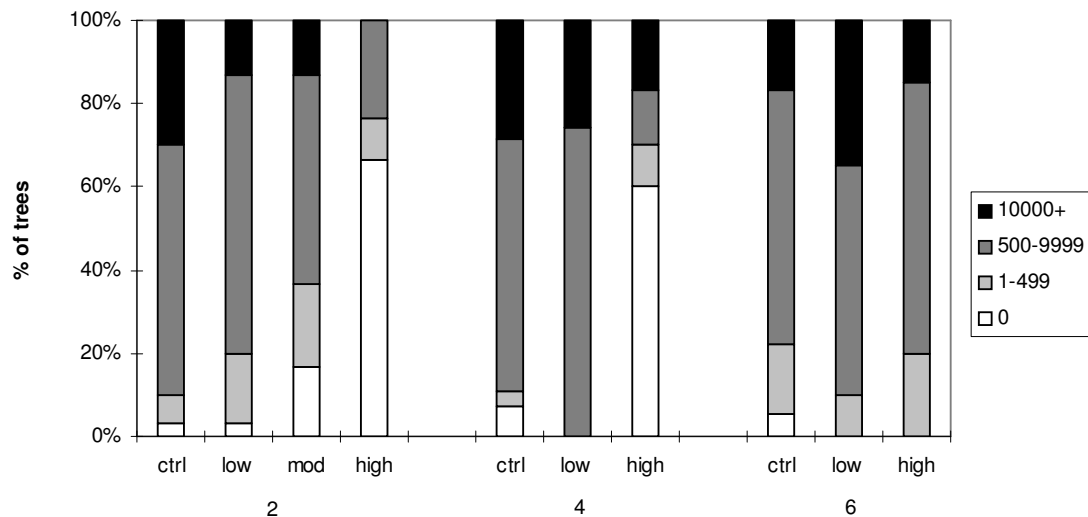


Figure 5.18. Percentage of trees with particular crop sizes (0; 1-499 = low; 500-9999 = medium, 10000+ = high) for each canopy scorch group at 2, 4 and 6 years since fire ($n = 30$ trees per canopy scorch group for the 2 years since fire class, 24 trees for the 4 years since fire class and 16 trees for the 6 years since fire class).

In the two years since fire class, crop size (log transformed) was significantly different between canopy scorch groups (Two way ANOVAs: $F_{3,48} = 21.39$, $P < 0.0001$, $n = 5$ trees per canopy scorch group) sites ($F_{2,48} = 6.83$, $P = 0.002$). The interaction between canopy scorch and site was also significant ($F_{6,48} = 3.9$, $P = 0.003$). Similarly for the 4 years since fire class, there were significant differences between canopy scorch groups ($F_{2,77} = 53.83$, $P < 0.0001$, $n = 8$ trees per canopy scorch group) and between sites ($F_{2,77} = 17.48$, $P < 0.0001$), and the interaction between canopy scorch and site was significant ($F_{4,77} = 18.09$, $P < 0.0001$). These results indicate that the effect of canopy scorch depended on which site was being considered in these time since fire classes. A significant interaction effect in the two way ANOVAs for both the 2 and 4 years since fire classes precluded the use of post-hoc comparisons between canopy scorch groups.

For the 6 years since fire class, there were no significant differences in crop size between canopy scorch groups ($F_{2, 42} = 1.07$, $P = 0.35$, $n = 8$ trees per canopy scorch group) or sites ($F_{1, 42} = 0.43$, $P = 0.47$), and the interaction between canopy scorch and site was not significant ($F_{2, 42} = 0.25$, $P = 0.78$).

Crop size is compared between canopy scorch groups for each site in Table 5.11. Within the 2 and 4 years since fire classes, all three sites were highest in crop size for the unburnt group and lowest in crop size for the high canopy scorch group, with the exception of one site (Byjuke), which was higher in crop size for the low and high canopy scorch groups than the unburnt group. High canopy scorch areas were negligible in crop size at three sites, and a complete absence of capsules occurred at two sites for high canopy scorch areas. In general, crop size was inversely proportional to the percentage of canopy scorched in these time since fire classes. Both sites in the 6 years since fire class were much higher in crop size for low canopy scorch areas than either unburnt or high canopy scorch areas. *E. baxteri* sites showed a similar general pattern to *E. arenacea* for crop size. Crop size was significantly different between canopy scorch groups for all sites in the 2 years since fire class and two out of three sites in the 4 years since fire class.

Table 5.11. Mean (\pm SE) crop size (capsules per tree) in each canopy scorch group at each site, for each time since fire class (TSF). Results of Wilcoxon Rank Sum tests comparing each canopy scorch group with the unburnt group are given for each site, for those time since fire classes showing significant differences between canopy scorch groups (Levels: * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$).

| TSF | Site | Species | Canopy scorch | Crop size | n |
|----------|-----------|--------------------|------------------|------------------|----|
| 2 | Harrow | <i>E. arenacea</i> | unburnt | 12904 \pm 5629 | 10 |
| | | | low | 6231 \pm 1638 | 10 |
| | | | moderate | 4695 \pm 1851 | 10 |
| | | | high | 0*** | 10 |
| | Roseneath | <i>E. baxteri</i> | unburnt | 4627 \pm 1496 | 10 |
| | | | low | 2782 \pm 1085 | 10 |
| | | | moderate | 1131 \pm 502 | 10 |
| | | | high | 0*** | 10 |
| | Yallakar | <i>E. arenacea</i> | unburnt | 16700 \pm 6273 | 10 |
| low | | | 5784 \pm 1893 | 10 | |
| moderate | | | 9270 \pm 7951 | 5 | |
| high | | | 2111 \pm 571** | 5 | |
| 4 | Byjuke | <i>E. baxteri</i> | unburnt | 7064 \pm 2328 | 8 |
| | | | low | 12276 \pm 5184 | 10 |
| | | | high | 13130 \pm 4020 | 8 |
| | Kadnook | <i>E. arenacea</i> | unburnt | 10220 \pm 7317 | 10 |
| | | | low | 8030 \pm 3080 | 10 |
| | | | high | 19 \pm 13** | 10 |
| | Kealys | <i>E. arenacea</i> | unburnt | 9309 \pm 3378 | 10 |
| | | | low | 7689 \pm 1927 | 10 |
| | | | high | 0** | 10 |
| 6 | Roseneath | <i>E. baxteri</i> | unburnt | 5848 \pm 2026 | 10 |
| | | | low | 12947 \pm 8082 | 10 |
| | | | high | 3773 \pm 1412 | 10 |
| | Tullich | <i>E. arenacea</i> | unburnt | 2010 \pm 719 | 8 |
| | | | low | 17365 \pm 6349 | 10 |
| | | | high | 7017 \pm 2226 | 10 |

Capsule density (capsules per branch)

Capsule density showed no obvious pattern with canopy scorch for any time since fire class (Figure 5.19). Kruskal Wallis tests showed no significant differences in capsule density for either the 2 years since fire class ($\chi^2 = 5.57$, d.f. = 3, $P = 0.13$), the 4 years since fire class ($\chi^2 = 3.48$, d.f. = 2, $P = 0.18$), or the 6 years since fire class ($\chi^2 = 1.58$, d.f. = 2, $P = 0.95$).

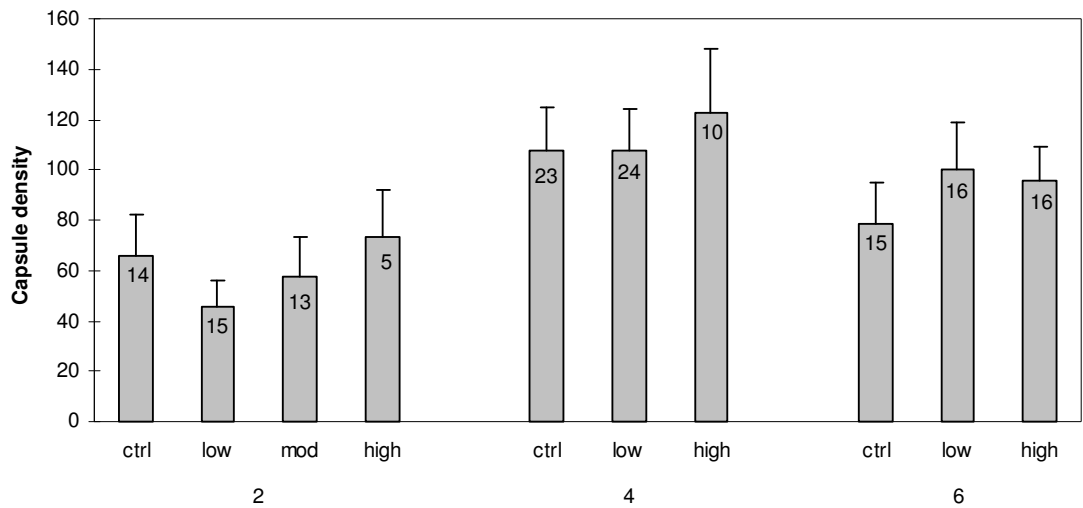


Figure 5.19. Mean (+SE) capsule density (capsules per branch) in each canopy scorch group at 2, 4 and 6 years since fire. The number of fruiting trees (n) is given for each canopy scorch group.

Cluster size (capsules per pedicel)

Cluster size was similar in pattern to capsule density and showed no consistent effect of canopy scorch (Figure 5.20). There were no significant differences between canopy scorch groups for the 2 years since fire class ($\chi^2 = 6.62$, d.f. = 3, $P = 0.08$), the 4 years since fire class ($\chi^2 = 3.7$, d.f. = 2, $P = 0.15$) or the 6 years since fire class ($\chi^2 = 2.31$, d.f. = 2, $P = 0.31$).

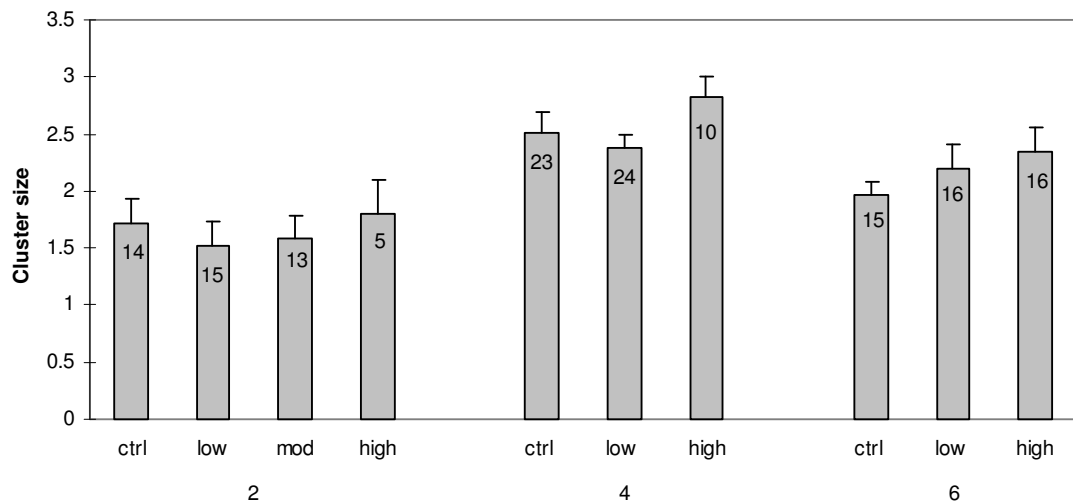


Figure 5.20. Mean (+SE) cluster size (capsules per pedicel) in each canopy scorch group at 2, 4 and 6 years since fire. The number of fruiting trees (n) is given for each canopy scorch group.

5.7.2 The vegetative responses of trees to different levels of canopy scorch

Number of epicormic branches per tree

The number of epicormic branches per tree showed a strong relationship with canopy scorch for all time since fire classes (Figure 5.21). Trees in the high canopy scorch group had the highest number of epicormic branches, while numbers were comparable between unburnt and low canopy scorch groups. The moderate canopy scorch group showed a proportionately (approximately 50%) lower number of epicormic branches than the high canopy scorch group and proportionately more epicormic branches than the unburnt group. The mean number of epicormic branches for the high canopy scorch group declined with time since fire, while numbers were comparable between time since fire classes for the low canopy scorch and unburnt groups.

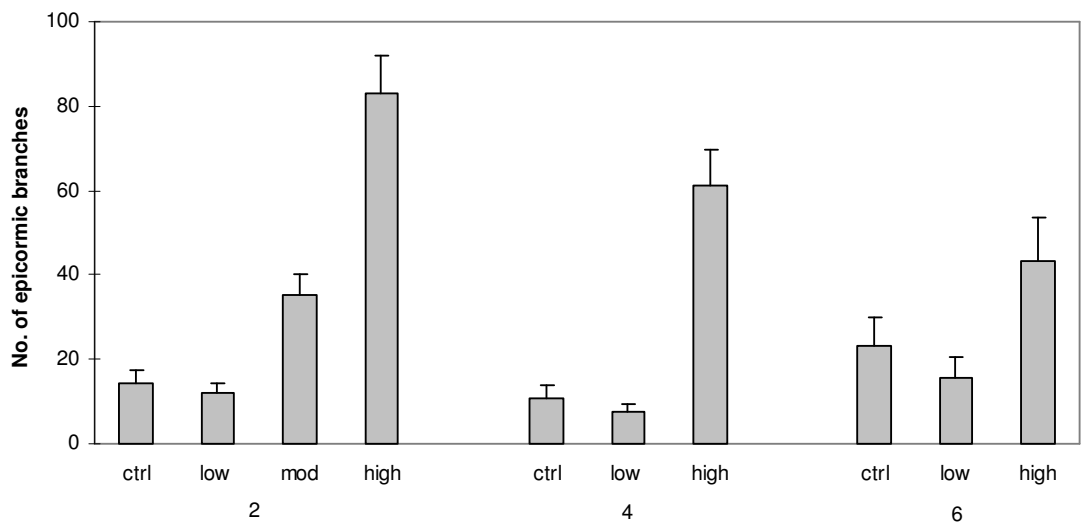


Figure 5.21. Mean (+SE) number of epicormic branches produced per tree in each canopy scorch group at 2 years, 4 years and 6 years since fire ($n = 30$ trees per intensity group for the 2 years since fire class, 24 trees for the 4 years since fire class and 16 trees for the 6 years since fire class).

The number of epicormic branches per tree was significantly different between canopy scorch groups for the 2 years since fire class ($F_{2, 89} = 30.36$, $P < 0.0001$; $n = 8$ trees per site for all time since fire classes), the 4 years since fire class ($F_{2, 63} = 32.09$, $P < 0.0001$) and the 6 years since fire class ($F_{2, 42} = 3.96$, $P = 0.03$). There were no significant differences between sites in the number of epicormic branches per tree for either the 2 years since fire class ($F_{2, 89} = 1.29$, $P = 0.28$), the 4 years since fire class ($F_{2, 63} = 1.53$, $P = 0.23$), or the 6 years since fire class ($F_{2, 42} = 0.41$, $P = 0.53$); nor was the interaction effect between canopy scorch and site significant for either the 2 years since fire class ($F_{2, 89}$

= 0.49, $P = 0.75$), the 4 years since fire class ($F_{2, 63} = 1.0$, $P = 0.41$) or the 6 years since fire class ($F_{2, 42} = 2.34$, $P = 0.11$). The results indicate that the effect of canopy scorch on the number of epicormic branches per tree was not dependent on which site was being considered.

Comparison of each canopy scorch group to the unburnt in the number of epicormic branches per tree for the 2 years since fire class showed significant differences for the high canopy scorch and moderate canopy scorch groups, but not the low intensity group (Dunnnett's tests, $\alpha = 0.05$). For the 4 and 6 years since fire class, differences were only significant for the high canopy scorch and unburnt group comparison.

Percentage of trees producing epicormic branches at base, trunk and canopy positions

A greater percentage of trees had epicormic branches on the trunk and base (lignotuber) in the high canopy scorch group than the unburnt and low canopy scorch groups, for all time since fire classes (Figure 5.22). Nearly all trees produced epicormic branches in the canopy, regardless of canopy scorch levels.

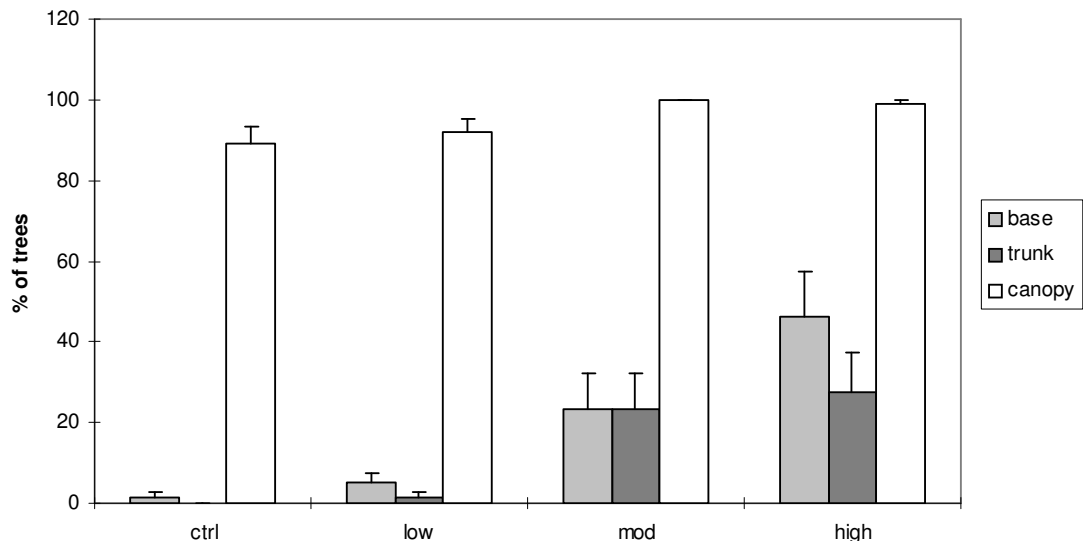


Figure 5.22. Mean (+SE) percentage of trees producing epicormic branches at base (lignotuber), trunk and canopy positions in each canopy scorch group. Data from each time since fire class were pooled ($n = 3$ sites for the moderate canopy scorch group, $n = 8$ sites for the other groups).

Time since fire classes were combined for each canopy scorch group to increase sample sizes for the analysis, and the low and moderate canopy scorch groups were pooled in the 2 years since fire class ($n = 8$ sites per canopy scorch group). Differences between canopy scorch groups in the percentage of trees with epicormic branches at each position were significant for the base positions ($\chi^2 = 12.97$, $P =$

0.0015) and trunk (Kruskal Wallis tests: $\chi^2 = 14.72$, d.f. = 2, $P = 0.0006$), but not for the canopy position ($\chi^2 = 3.36$, $P = 0.19$).

5.8 Discussion

The results of the present study confirm that fire intensity is an important component of the fire regime influencing the reproductive performance of trees recovering from fire. Resource allocation theory predicts that an increase in vegetative activity necessitates a reduction in reproductive effort (Reekie and Bazzaz 1987). In accordance with this theory, crop size was approximately inversely proportional to the percentage of canopy scorched for sites in the 2 and 4 years since fire classes. In contrast, the low canopy scorch group was usually comparable with unburnt levels, indicating that low levels of canopy scorch have a relatively low impact on subsequent capsule availability, although the effects may not be eliminated entirely.

These findings indicate that reducing the intensity of prescribed burns has the potential to substantially reduce impacts on capsule availability to the Red-tailed Black Cockatoo. Although no significant differences were detected for capsule density and cluster size in the present study, prescribed burns typically have a substantial impact on these aspects as well (part A of this chapter). Given that crop size, capsule density and cluster size are highly inter-correlated (section 4.3.1), reducing the intensity of burns would be likely to reduce impacts on all of these variables. The implications of these findings for fire management and the Red-tailed Black Cockatoo are discussed further in Part C of this chapter.

Site was an important factor contributing to variability in the responses of trees recovering from fire in the present study. This is not surprising, given the large number of factors contributing to inter-site variability, such as soil type, rainfall and grazing history. Other components of the fire regime such as fire frequency and the inter-fire interval may also have contributed to variation between sites (Cary and Morrison 1995). However, differences between trees and sites in the extent of crown dieback (measured as the diameter of branches killed by fire, or terminal twig diameter; Cheney pers. comm.) were probably the cause of the high variability between sites. The height of trees precluded measurement of the terminal twig diameter in the present study, but crown dieback should directly influence seed production because trees with more crown dieback would need to invest more resources in rebuilding the canopy, leaving fewer resources available for reproductive functions (Reekie and Bazzaz 1987). However, differences in the terminal twig diameter would only be expected among high canopy scorch areas, because the leaves of branches would be killed by fire faster than the branches

themselves (Luke and McArthur 1978), and consequently partially scorched trees would not vary in dieback percentage.

Although crop size was similar between canopy scorch groups in the 6 years since fire group, Part A of this chapter demonstrated that typical prescribed burns (which result in complete canopy scorch) can reduce capsule availability for approximately 11 years. Therefore, the sites selected for the 6 years since fire group are considered to be “above average” in their reproductive performance after fire. Nevertheless, trees in the high canopy scorch group had not yet recovered their former canopy structure in this group. Trees in the high canopy scorch group had significantly more epicormic branches than those in the unburnt group and a significantly higher percentage of trees had epicormic branches on the stem and lignotuber. Restoration of canopy structure thus took an average of more than six years for completely scorched trees.

By comparison, branching patterns were restored after three years in *Eucalyptus dives* following complete canopy scorch (Gill 1978). *Eucalyptus dalrympleana*, on the other hand, took at least six years to recover at the same study site. *Eucalyptus diversicolor* takes around five years to form an apically-dominant crown following complete canopy scorch (McCaw *et al.* 1995). The evidence from this chapter suggests that other species of eucalypts such as these may take a comparable amount of time to fully recover their floral resources after burns that completely scorch the canopy, because of the partitioning of resources between reproductive functions and canopy replacement after fire. A large number of bird and mammal species use the floral resources of eucalypts. Therefore, the level of canopy scorch caused by prescribed burns should be considered by fire management authorities as a factor influencing food availability to species depending on the floral resources of eucalypts for their food supply. Particular consideration should be given to threatened species in regions that are frequently burnt.

Part C: Implications for fire management

For conservation of the south-eastern Red-tailed Black Cockatoo, the stringybark feeding habitat should be managed to maximise food availability. However, the food requirements of the cockatoo population need to be balanced with fuel reduction objectives in order to reduce the risk of extensive wildfires.

Prescribed burns reduced capsule availability to the cockatoos by approximately 50% over a 10 year period. Although less than 20% of Red-tailed Black Cockatoo habitat is less than 10 years post fire, or more precisely, post canopy scorch, it is likely that this value could be reduced considerably if greater emphasis were placed on: (1) reducing the level of canopy scorch, (2) increasing the fire interval and (3) reducing the total area burnt, wherever possible.

Lowering the intensity of burns is probably the best way to balance the food requirements of the cockatoos with fuel reduction objectives, if such burns are a viable alternative in terms of cost and feasibility. Low intensity burns reduce effects on capsule availability in both the short term because of reduced seed shed from the canopy (Koch pers. obs.), and in the longer term through reduced effects on subsequent seed production. Moreover, low intensity burns are likely to benefit many other species because they tend to produce a mosaic of burnt and unburnt areas that increases the survival of small mammals and other animals by providing refugia (Wilson 1996). Reducing the intensity of burns may require experimentation with different burn methods, such as burning at night and/or alteration of lighting patterns. Gould *et al.* (1997) suggested that prescribed burns performed under calm conditions will produce less canopy scorch in eucalypt woodland than those performed at the same intensity under stronger wind conditions. Some Victorian management authorities are currently experimenting with methods to reduce the intensity of burns (McGuire pers. comm.).

In South Australia, intervals are generally prescribed at 3-8 years, with the exception of some areas set aside for less frequent burning (Millard 1998). Burning at such frequencies will dramatically reduce habitat quality for Red-tailed Black Cockatoos, and is further likely to reduce structural diversity (Nieuwenhuis 1987) and soil productivity (Hall 1996). Therefore, fire management authorities in South Australia should consider increasing the interval between burns, and the total area burnt. Fire management in the Horsham district of Victoria was changed in 1989 from a policy of broad-acre or "block" burning to strip burning, the burning of strips along the perimeters of remnants or along fire tracks to form strategic fire breaks. Strip burns have been successfully used for wildfire suppression in Victoria and should be considered in South Australia wherever possible and appropriate. Strip burns

reduce the area that is burnt and hence the area of stringybark that is of lower resource value to the cockatoos.

Where strategic protection from wildfire requires a high frequency of burns, managers should seek to reduce the level of canopy scorch and the total area burnt. Where the level of canopy scorch cannot be reduced, managers should seek to burn when capsule availability in the area is generally low. Forestry officers use a simple assessment procedure to evaluate fuel loads in a particular area and a similar procedure could be routinely employed for capsule availability. In particular, managers should avoid burning areas that have recently produced a new seed crop, as these areas are used preferentially by the cockatoos, and will be of high resource value to the cockatoos for the next one to two years (Chapter 3). This demands a more proactive approach to fire management, but will enable a balance between fuel reduction objectives and management for the southeastern Red-tailed Black Cockatoo.

CHAPTER 6 The value of induced edges as feeding habitat for the south-eastern Red-tailed Black Cockatoo

6.1 Introduction

Edge effects can influence a wide range of ecosystem and ecological processes that can directly or indirectly influence habitat quality for birds. The effects of induced edges on bird community composition are well studied (eg. Luck *et al.* 1999, Brand and George 2001, Berry 2001). Less well studied but also important are the effects of edges on the foraging behaviour of birds. Altered weather conditions at edges can reduce foraging activity where increased temperature variation occurs (Brotons *et al.* 2001). Other factors that have been suggested to influence foraging birds in fragmented landscapes include those processes associated with the alteration of habitats near edges, such as weed invasion, microclimatic changes, competition with edge specialists and increased tree dieback (Ford 2001).

The south eastern Red-tailed Black Cockatoo occurs in a landscape that has been extensively cleared (approximately 50%, Chapter 3), leaving “islands” of native vegetation surrounded by a “sea” of land used primarily for agricultural development. The aims of the present study were to: (1) compare capsule availability and tree characteristics between edge and interior trees; (2) determine whether the cockatoos used edge trees to the same extent as interior trees; and (3) determine if and how far edge effects extended into the habitat interior. The cockatoos tend to select trees with large crop sizes (capsules per tree), capsule densities (capsules per branch) and cluster sizes (capsules per pedicel; Chapter 4). These aspects of capsule availability were compared between edge and interior trees to indicate resource availability, and to relate capsule availability to patterns of habitat use by the cockatoos.

6.2 Methods

This chapter presents results from two separate surveys. The first survey compared capsule availability (crop size, capsule density and cluster size) and levels of habitat use by the cockatoos (percentage of trees used for foraging per sampling point) between edge and interior positions. The second survey assessed the distance that edge effects extended into the habitat interior, based on recent feeding signs left by the cockatoos. For the purposes of the present study, an edge was defined as any induced habitat perimeter that was bordered by an area cleared of vegetation for at least 20 m.

6.2.1 Comparison of edge trees and interior trees in capsule availability, tree characteristics and resource use by cockatoos

The data for this Part of the study were collected as part of the broad-scale habitat survey during the year 2000 (see section 2.2 for survey methods and details of measurements). The location of study sites is given in Figure 2.2. Only unburnt sites and sites that had a definite edge (defined above) were included in this chapter. Sites that were bordered by fire access tracks were less than 20m from adjacent vegetation were excluded. The sites included in the study were Jilpanger and Kealys. These sites were used to compare edge trees with interior trees in each of the measured resource characteristics (crop size, capsule density, cluster size, girth, height, canopy volume, percentage dieback and distance between trees). Measurements of habitat use were the percentage of trees used by cockatoos per sampling point and the percentage of branches used per tree.

The data were non-normally distributed for some sites and positions (edge and interior), even after log transformation. Therefore, nonparametric Wilcoxon Rank Sum tests were used to test for differences between positions. These tests were also used to test for differences between sites, in order to account for variability caused by site differences.

6.2.2 Patterns of resource use and capsule availability with increasing distance from edges

In September 2000, six sites in the Casterton region that had recently been used by cockatoos were surveyed to determine patterns of habitat use at varying distances from induced edges. The sites were Cooksons Track, Tullich a, Tullich b, Gills track, No. 1 Break and East Boundary Track, which all occur in the Casterton region. A map of these study sites is given in Figure 2.2. These sites were selected on the basis that they had definite edges (defined above), and that they had been recently used for feeding by cockatoos. The use of sites by the cockatoos depended on the observations of a volunteer who was monitoring the movements of the cockatoos in the Casterton district of south-west Victoria at the time. The cockatoos tend to spend several weeks or longer in a given remnant of vegetation, although duration of stay may depend on the size of the flock and the size of the remnant (Koch pers. obsv., Cooper pers. comm.).

The sites in this survey were bordered by gravel roads, Tasmanian blue gum plantations, or pine plantations. At each site, 3 replicate transects (40m wide x 150m long) running from an induced edge towards the interior of the patch were measured with a tape measure and marked with flagging tape. The total number of "feed" trees and "other" trees occurring within each segment of the transect (transect positions: 0-10m, 10-30m, 30-50m, 50-100m, 100-150m) were recorded. A tree was classified

as a feed tree if recent feeding signs were present under the canopy. Feeding signs left by the cockatoos are described in Figures 4.1 and 4.2.

A crude measure of capsule density (referred to hereafter as the capsule density index) was also recorded for each transect position, to determine whether the effects of edges on capsule availability extended into the habitat interior. The index was based on an overall visual assessment of capsule density (the number of capsules per branch) for all the trees at each transect position. A four point scale was used to estimate capsule density that included the following categories: low (<100 capsules per branch), medium 100-199 capsules per branch), high (200-299 capsules per branch) and very high (>300 capsules per branch). Each transect position (for each replicate transect) was assigned a number (1-4, from low to very high, respectively) for analysis.

Two way ANOVAs were used to compare the percentage of trees used for feeding, tree density and capsule density between sites and between transect positions, since these variables were both normally distributed. The capsule availability index was log transformed to improve normality. Tukeys Honestly Significant Difference (HSD) tests were used to compare each pair of segments.

6.3 Results

6.3.1 Responses of trees to edge effects

Comparison of resource characteristics between edge and interior positions

Edge trees generally had higher scores than interior trees for capsule density, crop size, cluster size, girth, canopy volume and were lower than interior trees in branch density (Table 6.1). Values were comparable between sites for these variables, with the exception of canopy volume and branch density, which differed markedly between sites for the edge position. The distance between trees was comparatively lower for edge trees than interior trees at Kealys, but was comparatively higher for edge trees than interior trees at Jilpanger.

Table 6.1. Comparison of capsule availability characteristics and tree characteristics between edge trees and interior trees for each site (n = 24 trees) and both sites combined (n = 48 trees). Significant differences based on Wilcoxon Rank Sum tests are shown with asterisks (Levels: * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$).

| Variable | Position | Kealys | Jilpanger | Combined |
|--|----------|--------------|--------------|---------------|
| Capsule density (capsules per branch) | Edge | 159 ± 27 | 144 ± 26 | 152 ± 19 |
| | Interior | 125 ± 23 | 123 ± 20 | 124 ± 15 |
| Crop size (capsules per tree) | Edge | 12362 ± 2782 | 10926 ± 4130 | 11644 ± 2465 |
| | Interior | 8264 ± 2684 | 3707 ± 972 | 5985 ± 1451** |
| Cluster size (capsules per pedicel) | Edge | 3.57 ± 0.31 | 2.93 ± 0.15 | 3.25 ± 0.18 |
| | Interior | 3.09 ± 0.37 | 2.67 ± 0.2 | 2.88 ± 0.21 |
| Girth (cm) | Edge | 128 ± 15 | 137 ± 21 | 133 ± 12 |
| | Interior | 83 ± 9 | 95 ± 13** | 89 ± 8** |
| Height (m) | Edge | 11.8 ± 0.7 | 7.6 ± 0.5 | 9.7 ± 0.5 |
| | Interior | 10.1 ± 0.7 | 7.8 ± 0.6 | 8.9 ± 0.5 |
| Canopy volume (m ³) | Edge | 108 ± 28 | 54 ± 16 | 81 ± 17 |
| | Interior | 45 ± 13 | 32 ± 6* | 39 ± 7 |
| Percentage dieback | Edge | 31 ± 2 | 32 ± 3 | 31 ± 2 |
| | Interior | 32 ± 3 | 38 ± 3 | 35 ± 2 |
| Branch density (m ⁻³) | Edge | 1.4 ± 0.2 | 4.1 ± 1.1 | 2.8 ± 0.6 |
| | Interior | 2.2 ± 0.3** | 2.7 ± 1.3 | 2.5 ± 0.7 |
| Distance between trees (m) | Edge | 6.2 ± 0.4 | 9.1 ± 0.8 | 7.8 ± 0.4 |
| | Interior | 10.2 ± 0.9 | 8.6 ± 0.5 | 9.4 ± 0.5 |

Girth and canopy volume were significantly higher for edge trees than interior trees at Jilpanger, and branch density was significantly higher for edge trees than interior trees at Kealys. There were no significant differences between edge trees and interior trees for the other variables at these sites. Averaged over both sites, edge trees were significantly higher than interior trees in crop size (Wilcoxon Rank Sum test: $Z = 2.5$, d.f. = 1, $P = 0.01$, $n = 48$) and girth ($Z = 2.77$, $P = 0.006$), but there were no significant differences between edge and interior trees for the other measured habitat components. The mean crop size for edge trees (all site combined, 11644 capsules per tree) was almost double that of interior trees (5984 capsules per tree).

When data for edge trees and interior trees were pooled for each site, there were no significant differences between the two sites for any resource characteristic except height ($Z = 4.47$, $P < 0.0001$, $n = 48$ trees).

Capsule availability with increased distance from edges

The capsule density index was comparatively higher at the edge position than at all other positions, and declined slightly but consistently with increased distance from edges (Figure 6.1).

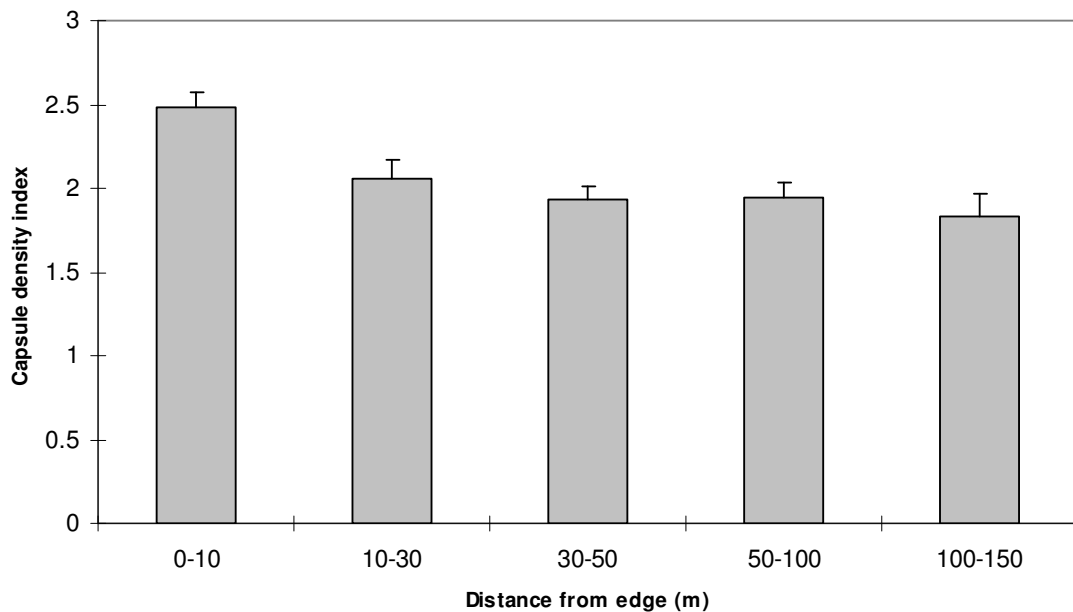


Figure 6.1. Mean (+SE) capsule density index in each transect segment at increasing distance from induced edges. The data were pooled across all sites and replicate transects ($n = 18$).

A 2 way ANOVA comparing the capsule density index between transect positions showed significant differences for the effect of position ($F_{4, 60} = 6.44$, $P = 0.0002$), but not for the effect of site ($F_{5, 60} = 0.49$, $P = 0.79$), or for the interaction effect ($F_{20, 60} = 0.35$, $P = 0.99$). The effect of position was therefore independent of site. Comparison of each pair of transect positions using Tukeys HSD tests ($\alpha = 0.05$) showed that the 0-10m transect position was significantly higher than all other positions, but there were no significant differences among the other pairs of transect positions.

6.3.2 Edge effects on patterns of resource consumption

Comparison of resource use between edge and interior trees

The percentage of trees used for foraging was comparable between edge and interior trees at both sites, although a slightly higher percentage of trees was used at Jilpanger than Kealys for both positions. Similarly, the percentage of branches used for foraging was comparable between positions for both sites but was generally higher at Jilpanger than Kealys. There were no significant differences between edge trees and interior trees for the percentage of trees used per sampling point or for the percentage of branches used per tree for foraging by cockatoos, either for individual sites or both sites combined.

Table 6.1. Comparison of the percentage of trees used per sampling point ($n = 3$ sampling points per position) and the percentage of branches used per tree ($n = 24$ trees per position) between edge trees and interior trees, for Jilpanger, Kealys and both sites combined.

| Variable | Position | Kealys | Jilpanger | Combined |
|------------------------------------|----------|------------|------------|------------|
| % of trees used per sampling point | Edge | 58 ± 3 | 71 ± 5 | 65 ± 3 |
| | Interior | 50 ± 6 | 75 ± 4 | 63 ± 4 |
| % of branches used per tree | Edge | 28 ± 6 | 44 ± 8 | 36 ± 5 |
| | Interior | 30 ± 8 | 54 ± 8 | 42 ± 6 |

A significantly higher percentage of trees was used for foraging at Jilpanger than Kealys (Wilcoxon Rank Sum tests: $Z = 2.58$, d.f. = 1, $P = 0.01$, $n = 6$ sampling points). The percentage of branches used per tree was also significantly higher for Jilpanger than Kealys ($Z = 3.61$, $P = 0.0003$, $n = 48$ trees).

Patterns of resource use with increasing distance from edges

The percentage of trees used for foraging at the edge position (58%) was higher than at all other positions, and the percentage of trees used showed a slight decline with increased distance from edges (Figure 6.2).

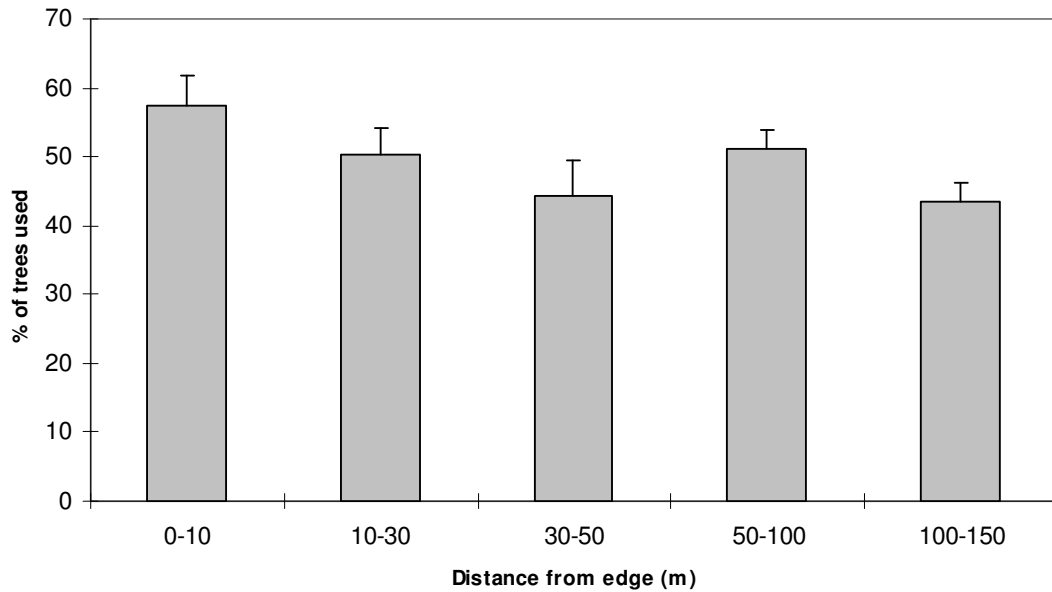


Figure 6.2. Mean (+SE) percentage of trees used for foraging in each transect segment at increasing distance from induced edges. The data were pooled across all sites and replicate transects ($n = 18$).

A 2 way ANOVA indicated significant differences in the percentage of trees used by cockatoos between transect positions ($F_{4,60} = 2.89, P = 0.03, n = 18$). The effects of site ($F_{5,60} = 1.66, P = 0.16$) and the interaction between position and site ($F_{20,60} = 0.78, P = 0.72$) were not significant (Table 2). Therefore, the effect of transect position was independent of site. Comparison of each pair of transect positions using Tukeys HSD tests ($\alpha = 0.05$) showed that the 0-10m was significantly higher in the percentage of trees used per sampling point than the 100-150m. There were no significant differences for the other comparisons of each pair of transect positions.

6.4 Discussion

There are a number of possible advantages afforded edge trees that may increase their productivity in terms of seed production. Examples include reduced competition with other trees and shrubs for resources, increased water supply through run-off from roads, and increased nutrients due to the fertilising of adjacent pastures and plantations. Increased light availability is likely to be an important advantage for edge trees, because trees at edges were seen to lean towards the cleared area (Koch pers. obs.). Differences in pollinator visitation rates might also benefit edge trees if floral displays are more attractive along edges, since rates of pollinator visitation can greatly influence fruit set (Day *et al.* 1997). Trees at induced edges were significantly larger than interior trees in crop size and girth, and were significantly greater in capsule density. However, the pattern of capsule density with increasing distance from edges indicates that the effect was mainly restricted to the outermost fringes (0-10m) of remnants.

Moreover, there were conflicting results between the two surveys with regard to the effects of edges on patterns of habitat use by Red-tailed Black Cockatoos. The survey of recent feeding signs indicated that a significantly greater percentage of trees were used near edges. Comparison of edge and interior trees using the broad-scale habitat survey data, however, showed no significant differences in either the percentage of trees used per sampling point or the percentage of branches used per tree. This result is surprising because the cockatoos show a strong tendency to select trees with large crop sizes (Chapter 4). A possible explanation is that the cockatoos select more trees at edges during the early stages of patch (or remnant) exploitation, and use interior trees once capsule availability declines on edge trees. If this is the case, edge trees may be used significantly more often only when food supplies are abundant. In accordance with this explanation, a higher percentage of trees were used overall for the broad-scale habitat survey (using evidence of feeding based on pruning effects which last for 1-2 years), than the recent feeding signs survey. This suggests that the cockatoos revisited sites following initial exploitation, because sites that had recently been exploited were used less intensively than sites that had been used over a 1-2 year period (since the production of a new seed crop on *E. arenacea*, Chapter 3). Therefore, it is likely that edge trees are only used by cockatoos to a greater extent when capsule availability is relatively abundant, such as in the early stages of patch exploitation.

Although edge trees were significantly greater than interior trees in crop size, the present study suggests that they provide little compensation for the habitat that has been lost due to clearance and logging. The edge effect was mainly restricted to a thin line of trees occurring on the outermost fringes

of remnants. The crop size of edge trees was less than double that of interior trees, which does not compensate for the large percentage of feeding habitat that has been cleared (approximately 50%, Chapter 3). Furthermore, the larger crop sizes of trees at edges were not accompanied by increases in capsule density or cluster size, characteristics which are known to significantly increase foraging efficiency for this cockatoo (Chapter 4). Nevertheless, the present study suggests that there are some small benefits provided by induced edges that have not previously been considered in the literature. Apart from the benefits provided by the generally larger crop sizes of edge trees, increased visibility at edges may help reduce search times in areas that have not yet been exploited by the cockatoos. These benefits may also extend to other species using eucalypt floral resources such as honeyeaters, but may depend on the influence of other degrading processes in some areas, such as weed invasion.

In addition, the findings suggest that the benefits of revegetating stringybark could be maximised by planting trees at lower densities, to achieve densities of mature trees that are lower than current average densities of mature trees in forest stands. If the increased productivity of trees at edges is due to reduced competition for resources (and particularly for light) as the evidence suggests, then planting trees at low densities should result in improved reproductive performance. This may benefit foraging cockatoos because of the reduced foraging costs that are likely to be associated with large crop sizes. Large crop sizes are likely to reduce foraging costs by reducing the number of flights that need to be made between trees, and the amount of search time needed to find suitable trees for foraging. Revegetation at low densities may therefore have a combined effect on foraging profitability because of increases in both the quantity of seed (after the trees reach reproductive maturity) and the feeding profitability of each tree planted. However, further research is required to determine the tree densities for planted stringybarks that maximise capsule production.

CHAPTER 7 General Discussion and Conclusions

The present study highlighted the importance of food availability as a factor influencing the movements and foraging efficiency of the south-eastern Red-tailed Black Cockatoo, and suggests that the cockatoos require a high density of capsules in order to meet their food requirements. The capsule availability characteristics of capsule density, crop size and cluster size appeared to be the most important factors influencing the selection of foraging locations. Correspondingly, the cockatoos processed significantly fewer capsules on branches with lower capsule densities, suggesting that low capsule density directly influences foraging efficiency.

These results have important implications in terms of how food availability should be defined for the south-eastern Red-tailed Black Cockatoo. Although food supply is unlikely to ever be exhausted, the density of capsules could become so low that the energetic return would not exceed the energy requirements for survival and breeding.

Periodicities between seed crops were likely to be the most important factor influencing food supply to the cockatoo population as a whole, since the evidence suggested that the cockatoos fed almost entirely on the species of stringybark that had fruited most recently. Given that capsule density had a strong influence on foraging performance, the cockatoos would be expected to spend a higher percentage of their day feeding as the seed crop aged and was depleted. In accordance with this expectation, the percentage of the day spent foraging increased by 21% in just one year following the production of a new seed crop in *E. baxteri*, with up to 88% of the day spent foraging. Although this finding should be interpreted with caution due to low sample sizes, it suggests that the cockatoos had to work much harder to meet their food requirements when the interval between seed crops was prolonged. Given the large demands on breeding males, it is unlikely that many birds could successfully breed in such years. Food supply is thus likely to be an important factor limiting the growth of the cockatoo population. If the current habitat is sufficient to support the population, it is likely to be preventing the population from increasing in numbers.

There was also substantial evidence to suggest that food supply was limiting when the birds were feeding in *E. arenacea* over the year 2000, two years after the production of a new seed crop for this species. This evidence included: the high rates of habitat use (100% of unburnt sites were used; 64% of trees were used per site, Chapter 5); the finding that nearly all burnt sites were used, despite their generally lower capsule availability (Chapter 5); and the finding that many sites were re-visited following initial exploitation (Chapters 4 and 5). Therefore, food availability appeared to be limiting the cockatoo population, regardless of which stringybark species was being considered.

If breeding males struggle to meet the food demands of supplying food for their partner and chick during periods of food shortage, we would expect a higher proportion of cockatoos to breed and a higher percentage of eggs producing fledged young in years when a new seed crop was produced.

Correspondingly, estimated nest success in 1998 (30%) was more than double that of estimated nest success in 1999 (13%), which coincided with the production of a new seed crop on *E. arenacea* in 1998. At times when the interval between consecutive seed crops was more than one year, such as the year 2000, nest success may have been even lower.

Evidence from the present study and consultation with beekeepers suggested that the two species of stringybark produce a new seed crop approximately once every three years. However, the fruiting of *E. baxteri* appeared to be irregular and varied on a 2-4 year cycle. Therefore, severe food shortages may occur at times when the period between seed crops is prolonged, and in these years breeding success may decline dramatically. Further monitoring of nest success is required to confirm this finding.

Fortunately, opportunities exist to increase food supply to the south-eastern Red-tailed Black Cockatoo through improved fire management. In particular, reducing the level of canopy scorch should be an effective means of minimising reductions in capsule availability to the cockatoos, while maintaining protection against wildfires. Fire management should concentrate on minimising the quantity of habitat that is less than 10 years post fire, or more precisely, post canopy scorch. These strategies were discussed further in Chapter 5. However, improved fire management will need to be supplemented with revegetation of stringybark, because the total increase in food availability to the cockatoo population expected from improved fire management (5-10%), is substantially lower than target increases outlined in the Recovery Plan (50%, Burnard and Hill 2001).

Revegetation should particularly target the South Australian part of the range, since the stringybark of this region has been cleared to a far greater extent. In particular, the areas west and north-west of Padthaway appear not to be used by the Red-tailed Black Cockatoo, despite the apparent suitability of these areas as feeding habitat. Cockatoos feeding in these highly fragmented areas would need to forage over a much greater area than those feeding in relatively contiguous areas, such as there are in parts of Victoria, and this would no doubt reduce foraging efficiency dramatically. If increased connectivity increases the likelihood of these highly fragmented areas being visited by cockatoos, then the total area of habitat effectively available to the birds could be substantially increased through strategic revegetation. Revegetation strategies should also focus on increasing the percentage of *E.*

arenacea, since this species makes up a lower percentage of the total stringybark and is therefore more likely to be a limiting resource than *E. baxteri*.

In conclusion, the findings of the present study suggest that food availability to the cockatoos can be substantially increased by combining improved fire management with strategic revegetation programmes. These measures have the potential to secure and improve the conservation status of the endangered south-eastern Red-tailed Black Cockatoo population.