Gnangara
Sustainability
Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

# THE IMPACT OF FIRE AND *PHYTOPHTHORA* DIEBACK ON BIRDS IN THE GNANGARA SUSTAINABILITY STRATEGY

A report prepared on behalf of the
Department of Environment and Conservation for the
Gnangara Sustainability Strategy

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Edith Cowan University

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GOVERNMENT OF
WESTERN AUSTRALIA

Department of Water
Department of Agriculture and Food WA
Department for Planning and Infrastructure
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Robert A. Davis
School of Natural Sciences, Edith Cowan University

Department of Environment and Conservation

Government of Western Australia
Department of Environment and Conservation

**Gnangara Sustainability Strategy Taskforce**
Department of Water
168 St Georges Terrace
Perth Western Australia  6000
Telephone   +61 8 6364 7600
Facsimile   +61 8 6364 7601
www.gnangara.water.wa.gov.au

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This document has been commissioned/produced as part of the Gnangara Sustainability Strategy (GSS). The GSS is a State Government initiative which aims to provide a framework for a whole of government approach to address land use and water planning issues associated with the Gnangara groundwater system. For more information go to www.gnangara.water.wa.gov.au

Cover Photos (Clockwise from top right): Scarlet Robin, New Holland Honey Eater, Splendid Fairy Wren, Carnaby's Black Cockatoo. Photos by Leonie Valentine

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# INTRODUCTION

An important question for the GSS is what the impacts of key threatening processes are on the natural diversity of the GSS area. Most studies of both fire and *Phytophthora* dieback have occurred in the forest system and there have been few if any, in *Banksia* woodlands. Consequently there is an imperative to quantify the impact of these two ecological processes on the avifauna of the GSS area. To this extent, this report presents the results of a study investigating the impact of prescribed burning regimes and *Phytophthora* dieback on the diversity, composition and abundance of avifauna within the GSS.

## Fire

The impacts of fire are complex and there have been few studies that have successfully examined the impacts of fire on birds. Burbidge (2003) conducted an extensive review on the impacts of fire on birds in south-western Australia and the extensive findings of this review along with management recommendations are summarised in Table 1.

Fire clearly has a varied impact depending upon the species of interest. In general it would seem that aiming for a mosaic of burn ages, including a significant proportion of unburnt or at least 10 + years since fire would be a sensible strategy. This would also maximise diversity and provide habitats for those species that benefit from fire, as well as providing for those requiring long-term unburnt habitats and providing source sites for re-colonisation of post-fire landscapes.

Although fire is unlikely to cause the loss of any species in the GSS area, it is apparent that a high burn frequency in isolated remnants has caused the extinction of sedentary species such as the Splendid Fairy-wren (Table 1). Fire suppression in isolated remnants, possibly through mosaic burning, may be critical to ensuring the regional persistence of these species at a regional level.

**Table 1. Summary of the impacts of fire on birds including implications for the GSS**

| **Variable** | **Key Findings** | **Implications for the GSS** |
|---|---|---|
| Bird diversity and richness | • Species richness is usually highest in long-term unburnt vegetation (15 years post-burn) but is also high for several years following a fire (Burbidge 2003).<br>• Species diversity can increase and be higher in burnt areas for up to 3 years post-burn (Christensen *et al.* 1985).<br>• Species richness of honeyeaters is reduced for three years after fire, whilst other guilds are unaffected (Burbidge 2003). | • In the long-term, longer unburnt vegetation is required for maximum richness and there are short term increases after fire when open habitat species are favoured. |
| Bird abundance | • Abundance of birds drops to very low levels immediately following a fire but usually recovers within 2-3 years (Burbidge 2003).<br>• Species abundances can be higher in frequently burnt areas than unburnt areas, but diversity may be low or dominated by open-habitat favouring species (Burbidge 2003).<br>• Insectivores generally increase after fire and exceed pre-fire abundance up to 7 years after the burn whereas nectarivores decline. This is primarily due to a reduction in the number of flowering banksias in burnt areas. | • A mosaic is required as not all species are equally affected by fire. Longer unburnt habitats are generally preferred. |
| Habitat change | • Some species such as birds of prey are favoured by fire due to a preference for open habitats. Many species remain in the same area for several years after a fire (Burbidge 2003).<br>• Open-habitat preferring species may dominate for 2-6 years post fire (Burbidge 2003).<br>• No species in south-west WA occurs only in long-unburnt vegetation (Burbidge 2003), however Bamford (1985) found that Western Thornbill, Shining Bronze-cuckoo and Scarlet Robin were more common in *Banksia* woodland unburnt for 11-12 years.<br>• Changes in abundance like this are linked to changes to habitat structure and the foraging opportunities that presents. Wooller and Calver (1988) noted significant decreases in abundance of White-breasted Robin, Golden Whistler, Splendid Fairy-wren and White-browed Scrub-wren following fire and surmised that this was due to changes to vegetation structure and prey.<br>• Frequent fires lead to the decline of populations of splendid fairy-wrens by altering habitat and food availability (Brooker 1998) and increasing nest predation as a result of lack of cover. | • Open post-fire habitats favour some species but in the long-term, long-unburnt sites are required to maintain some key species which otherwise decline from food shortages and higher rates of predation. |

| **Variable** | **Key Findings** | **Implications for the GSS** |
|---|---|---|
| Population viability | • Active nests can still recruit fledglings following a low intensity fire (Kimber 1974).<br>• Fire can reduce the recruitment success of some species such as the Splendid Fairy-wren (Rowley and Russel 1990).<br>• Fire in small remnants (under 2000 ha) can cause the extinction of some species such as the Splendid Fairy-wren (Brooker and Brooker 1994) and Western Thornbill (Recher 1997).<br>• Nest predation and brood parasitism increase after fire (Russell and Rowley 1993). | • Fire suppression strategies or mosaic burning, may be essential in small remnants under 2000 ha. |
| Dispersal and recolonisation | • After an extensive fire, species recolonise from surrounding unburnt patches but this is slow for some species such as Grey Shrike-thrush, which only return after 3 years.<br>• Sedentary species in remnants are more likely to become extinct after fir than dispersive species (Burbidge 2003). | • Burn sites should be well-connected to unburnt blocks that act as source populations. Avoid burning large areas. |
| Fire Intensity | • Most studies attempting to look at the influence of fire intensity on species ecology have been confounded.<br>• Fire intensity is important with more intense fires causing a greater reduction in species richness than lower-intensity fires (Burbidge 2003).<br>• Reduced abundances are more pronounced for higher intensity fires as opposed to low intensity fires (Burbidge 2003). | • Further research on fire intensity is needed but it would seem that low intensity burns are preferable. |

## *Phytophthora cinnamomi* (Dieback)

Few studies have examined the impact of *Phytophthora cinnamomi* on avifauna but dieback is now known to be a key threatening process in south-west WA and the impacts of dieback on birds are likely to be through changes to the vegetation structure and floristics (Garkaklis *et al.* 2004; Shearer *et al.* 2007).

More than 1000 species are known to be susceptible to dieback (McDougall *et al.* 2005) and Proteaceae are highly susceptible (Cahill *et al.* 2008). In *Banksia* woodlands of the Perth region, dieback causes a decline in species richness and biomass and causes gross structural change, particularly the loss of almost all of the dominant proteaceous overstorey species (Shearer and Hill 1989).

There has been little research on the impacts of dieback on fauna and none in *Banksia* woodlands. Cahill *et al.* (2008) review the impacts of dieback on biodiversity and state that the main concern for fauna is the changes to major resources such as food items and protective cover. The loss of nectar-producing proteaceous species is likely to have a significant impact on nectarivorous birds, and changes to the invertebrate community will impact insectivorous birds. Studies that have been done on invertebrates showed a reduced abundance and richness of invertebrates related to a reduction in leaf litter in dieback-affected areas (Postle *et al.* 1986).

Another impact of dieback that relates to avifauna is the loss of tree-hollows arising from the loss of dominant overstorey species (Cahill *et al.* 2008). This will impact hollow-nesting species, although notably *Banksia* woodland is not characterized by an abundance of tree hollows and it is considered that this is less likely to be a major ecological issue than in forest habitats.

# METHODS

## Fire studies

To investigate the impacts of fire on avifaunal communities, twenty study sites of differing fire ages were established (Figure 2 and Appendix 1). A shape-file layer is held by DEC GSS and shows the location of sites which encompass the area from Warbrook Road in the South, to the very top of the GSS area at Gingin Brook Rd. Sites were chosen with the aid of a fuel age map from DEC and were all on the Bassendean dune system, so as to avoid confounding influence arising from different vegetation communities. Sites are numbered to reflect the fire age (hence names of sites are the same as the fire age). Wherever possible, to minimise logistics, sites were selected to be surveyed as pairs that is a boundary of two fire ages was chosen so that with a pair of birdwatchers, the sites could be simultaneously surveyed.

The survey methodology used was the national standard prescribed by Birds Australia and utilised in the Atlas of Australian Birds (Barrett *et al.* 2003). This involved a 20 minute, 2 ha survey at each site. Utilising the BA methodology allows compatibility with the national Atlas database which contains over 20 000 records for the GSS area. Using a GPS, the corners of all survey sites were marked out to create a site that was a 200 x 100 m rectangle. It was ensured that each survey site was fully contained within the same fuel age and of similar vegetation composition and topography throughout. Flagging tape was placed every 50 metres along the boundary to enable navigation of bird surveyors.

Sites were surveyed by recording the start time and zig-zagging throughout the entire site during the 20 minutes. Sites were surveyed during the morning from sunrise and finished before mid-day. Due to the large number of sites, surveys took two mornings, and each month, we undertook two rounds of surveys (each site was surveyed twice). The site start order was varied each day and surveys were avoided in adverse weather conditions. All surveys were undertaken by experienced bird watchers with binoculars and utilised personnel who were familiar with all local species and their calls. When a bird was sighted, a note was made of where in the vegetation structure it was found.

These were based on the following categories: N = No status recorded, G = Bird(s) observed on the ground, L = Bird(s) observed in the lower-storey (to approx. 1.5 m), M = Bird(s) observed in the mid-storey (from approx. 1.5 m to canopy), C = Bird(s) observed in the canopy, F = Bird(s) observed foraging, aerially, over the quadrat, T = Bird(s) observed in transit over or through the quadrat (not, obviously, making use of the vegetation within the quadrat - just the airspace), H = Bird(s) heard (or seen) within the same fire-age/dieback status but not within the quadrat, O = Bird(s) observed or heard from the quadrat but in a different fire-age, dieback status or habitat type (i.e. not *Banksia* woodland). The zones are depicted graphically in Figure 1. The number of birds was also noted during each sighting (thus flock size). All collected data was entered into a Microsoft Access Database.

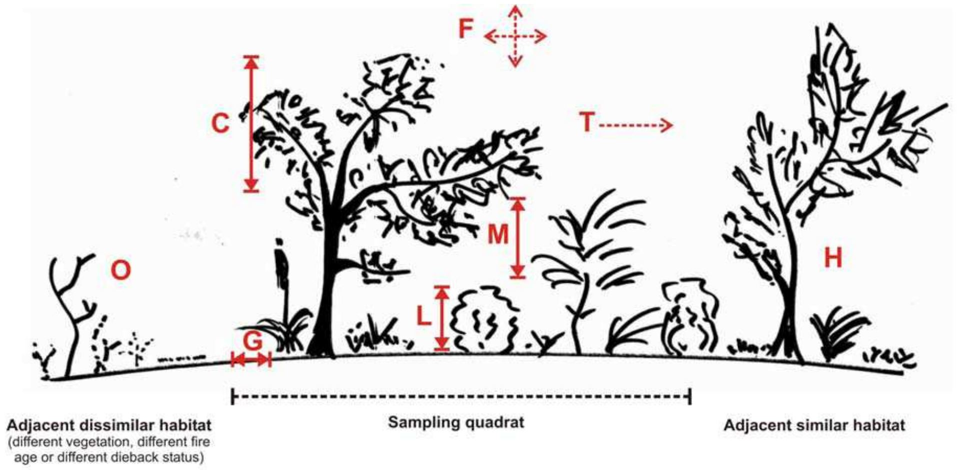

Figure 1. Graphical depiction of locations within vegetation structure recorded for bird observations: G = ground, L = lower-storey (to approx. 1.5 m), M = mid-storey (from approx. 1.5 m to canopy), C = canopy, F = foraging, aerially, over the quadrat, T = in transit over or through the quadrat, H = Bird(s) heard (or seen) nearby quadrat in similar parameters, O = Bird(s) observed (or heard)from quadrat but with different fire and dieback parameters.

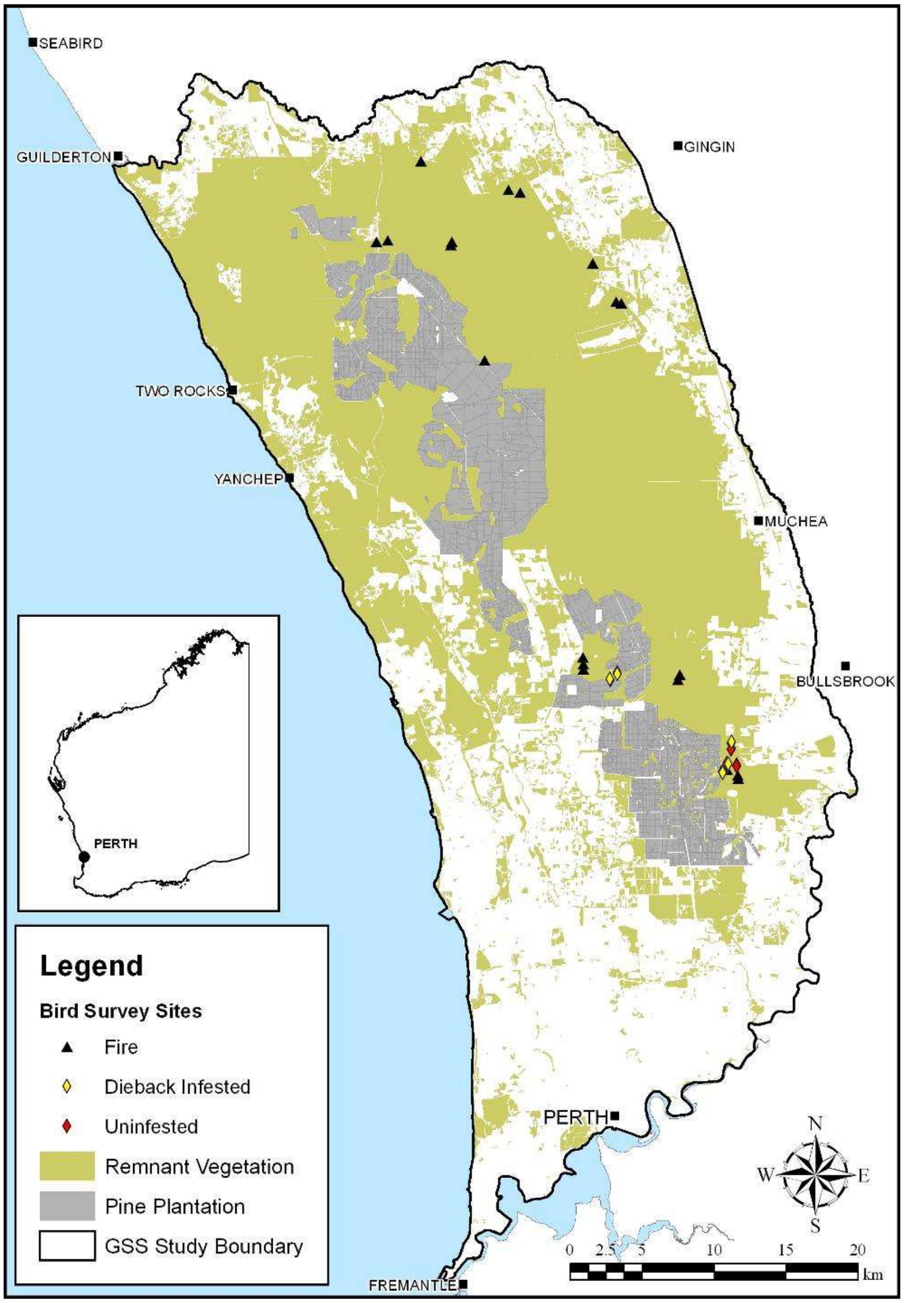

**Figure 2. Location of dieback and fire survey sites.**

## *Phytophthora* Dieback Surveys

To assess the impacts of *Phytophthora* dieback, 7 sites within the GSS were chosen (Figure 2 and Appendix 2). All sites were chosen with the aid of aerial photography and existing dieback mapping which showed areas of potential dieback. Ground-truthing was used to further identify areas that contained large dieback fronts and the *Phytophthora* dieback boundaries were interpreted and mapped made by an experienced DEC interpreter. All sites were on the Bassendean soil system.

Exactly the same survey methodology was used as the fire sites, with the exception that a paired site design was established with a 1 ha site (100 x 100 m) extending from the dieback front edge into dieback-affected vegetation and a further 1 ha extending from this front edge in the opposite direction to unaffected vegetation. Each respective half was designated as ‘unaffected’ or ‘affected’ and was surveyed for ten minutes in each. It is intended to use a paired t-test to examine the results of these surveys. As for the fire sites, a shape-file is held by DEC GSS which contains the locations of all site locations and corner points. Sites were surveyed once per month wherever possible and were all surveyed on the same day during the morning.

# RESULTS

## Fire Studies

Bird density and richness were highly variable across the 20 differing fire ages (Table 2). There was no obviously discernable trend between fire age and bird density, with the highest bird densities occurring 9, 23 and 25 years post-burn and the lowest bird densities 2, 10 and 18 years post-burn. Species richness was less variable and was also highest at 9, 23 and 25 years post-burn and lowest at 2, 6 and 10 years post-burn.

Table 2. Summary of bird richness and density at each site (mean ± standard error, brackets indicate range of data)

| Site (YSLB) | Number of Surveys | Bird Density | Species Richness |
|---|---|---|---|
| 1 | 16 | 6.03 ± 0.76 (1–13) | 4.94 ± 0.64 (1–11) |
| 2 | 16 | 3.31 ± 0.79 (0–8.5) | 3.13 ± 0.66 (0–8) |
| 3 | 16 | 4.66 ± 0.73 (0–9.5) | 4.31 ± 0.55 (0–8) |
| 4 | 16 | 5.38 ± 0.91 (1.5–12) | 4.63 ± 0.64 (1–10) |
| 6 | 16 | 4.72 ± 0.76 (1–11) | 3.19 ± 0.42 (1–8) |
| 7 | 16 | 4.09 ± 0.72 (0–10.5) | 3.88 ± 0.6 (0–8) |
| 8 | 16 | 4.84 ± 0.69 (1.5–11) | 4 ± 0.29 (2–6) |
| 8b | 12 | 4.04 ± 0.66 (0.5–7.5) | 3.25 ± 0.55 (1–7) |
| 9 | 16 | 13.72 ± 1.33 (4–21) | 6.13 ± 0.61 (3–12) |
| 10 | 16 | 3.31 ± 0.57 (0–7.5) | 2.88 ± 0.39 (0–6) |
| 11 | 16 | 5.66 ± 0.81 (1.5–14.5) | 4.31 ± 0.73 (1–12) |
| 12 | 16 | 4.41 ± 0.94 (0.5–13.5) | 3.75 ± 0.55 (1–9) |
| 13 | 16 | 5.88 ± 0.55 (2–9) | 4.31 ± 0.33 (1–7) |
| 14 | 16 | 4.78 ± 0.4 (2.5–8.5) | 4 ± 0.33 (2–7) |
| 15 | 16 | 4.59 ± 0.68 (0.5–10) | 3.56 ± 0.34 (1–5) |
| 18 | 16 | 3.97 ± 0.44 (0.5–6.5) | 3.81 ± 0.39 (1–6) |
| 22 | 16 | 4.81 ± 0.89 (1–14) | 4.38 ± 0.52 (2–9) |
| 23 | 16 | 9.53 ± 1.33 (4–20) | 5.69 ± 0.58 (2–11) |
| 25 | 16 | 10.5 ± 1.15 (4–22) | 5.31 ± 0.47 (2–8) |
| 26 | 16 | 4.28 ± 0.72 (0–10.5) | 4.19 ± 0.55 (0–9) |

Seasonal influences on birds were further explored by dividing the data into the recognised seasons of Winter (June-August), Spring (September-November), Summer (December-February) and Autumn (March-May). Surveys were undertaken in each season, however survey effort was not equal, with five autumn surveys, six in winter, four in spring and only one in summer, due to personnel and budgetary constraints.

Mean bird density differed between seasons, however there was no consistent pattern across sites, with some sites exhibiting peak bird density in winter while others were during spring and summer (Figure 3). The standard errors of each data set also overlap, further indicating that there is no consistent and significant difference between sites. By contrast, bird species richness showed a strong and consistent seasonal trend (Figure 4). The peak species richness at each site occurred in winter, followed by autumn and was lowest in spring. It is evident that there is a significant difference in species richness across all sites and that this is related to season (Figure 4).

A correlation was performed comparing both cumulative species richness and mean bird density with fire age, and a weak positive relationship was found (Figure 5 and Figure 6). Mean density and total bird species richness were more strongly correlated with floral species richness (Figure 7 and Figure 8) with $R^2$ values of 0.5 (density) and 0.27 (richness).

The frequency of occurrence of each species is shown in Table 2. It is apparent that most species were relatively scarce. Only 4 species were recorded in more than 20% of quadrats and these were the Brown Honeyeater (74.87% of quadrats), Western Spinebill (58.89%), Splendid Fairy-wren (47.94%) and Western Thornbill (37.78%).

Most species did not show any clear trend in terms of occurrence in relation to fire age, although Splendid Fairy-wrens were more frequently recorded in longer unburnt areas (Figure 9) and Yellow-rumped Thornbills were more commonly observed in more recently burnt areas (Figure 10).

Habitat use is depicted in Figure 11 and it is difficult to generalise about any trends from this, as there seems a highly variable use of structure that does not seem to be related to fire age. The occurrence of honeyeaters has a significant positive relationship with the presence of actively flowering plants (Figure 12).

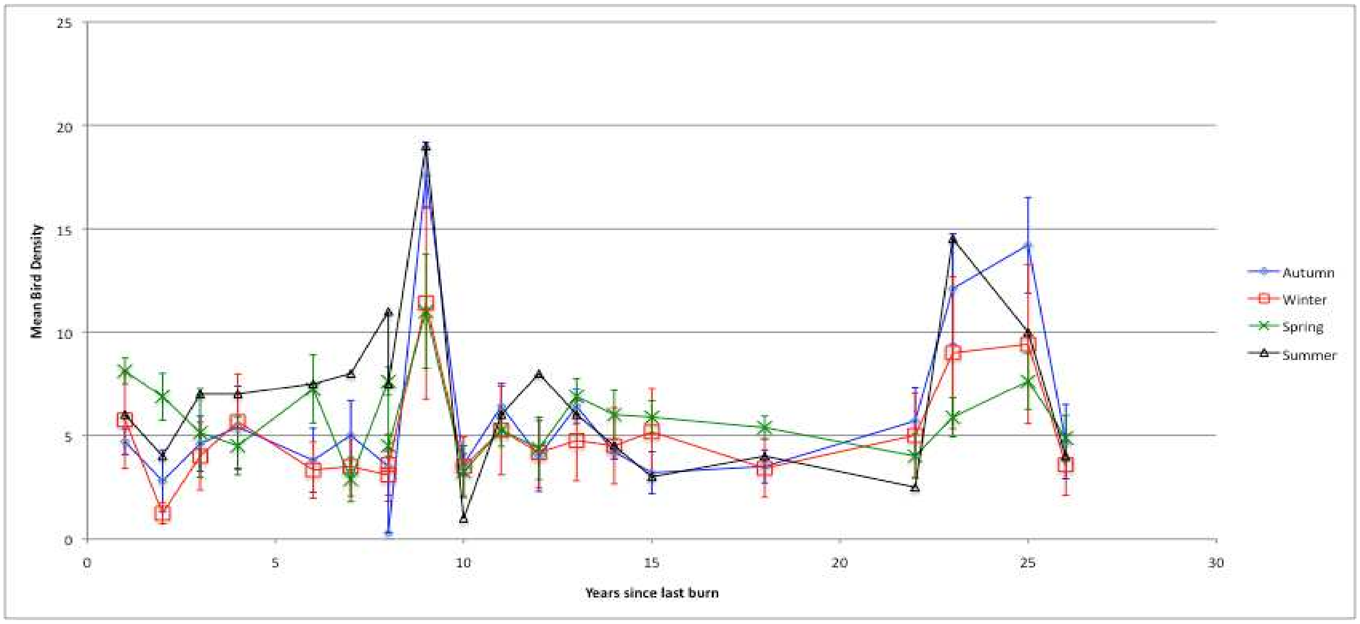

Figure 3. Seasonal variation in mean bird density of each fire age site.

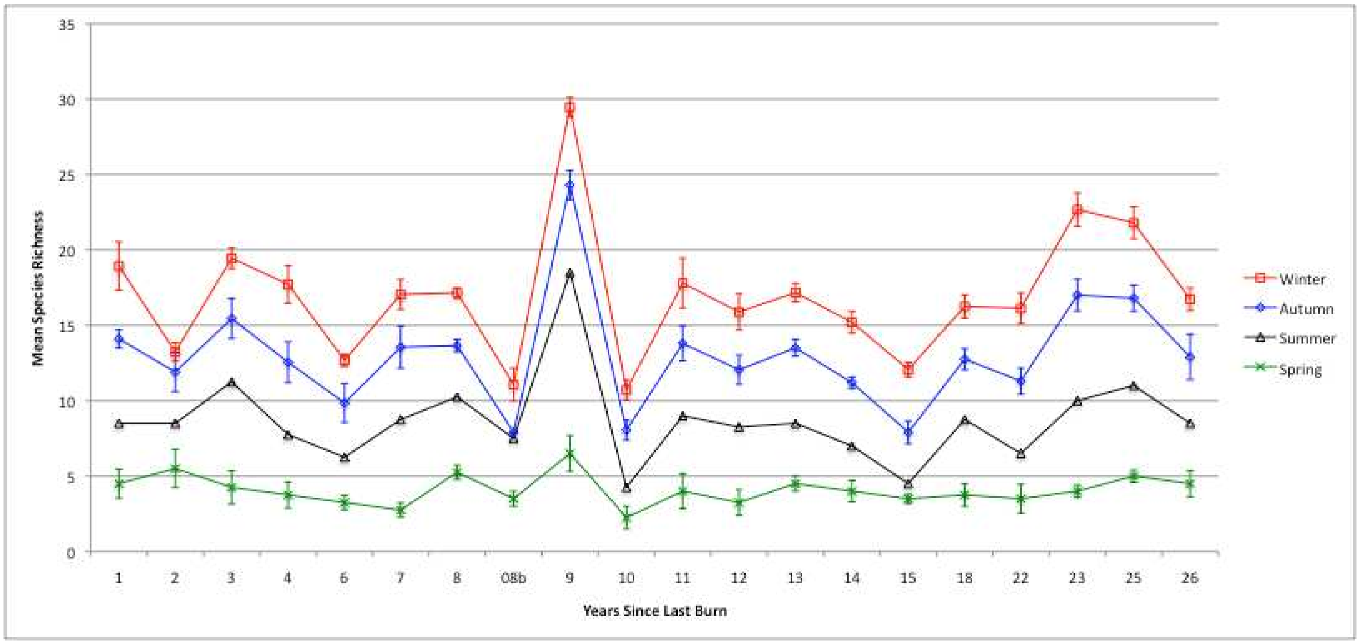

Figure 4 Seasonal variation in mean species richness, in relation to years since last burn.

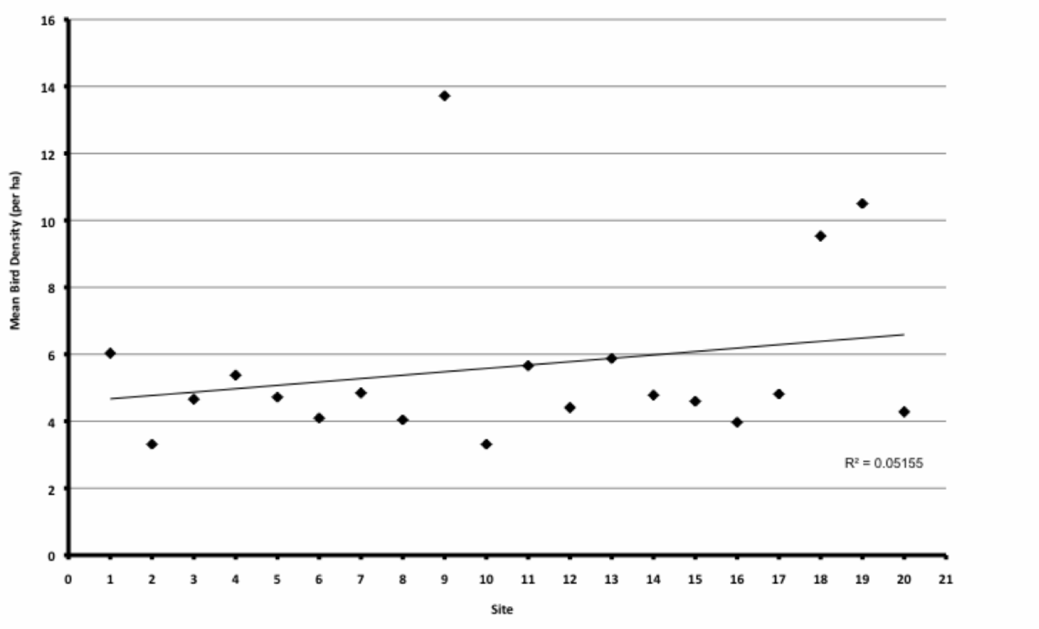

Figure 5. Mean bird density in relation to fire age. A line of best fit has been added to examine the correlation strength of the relationship.

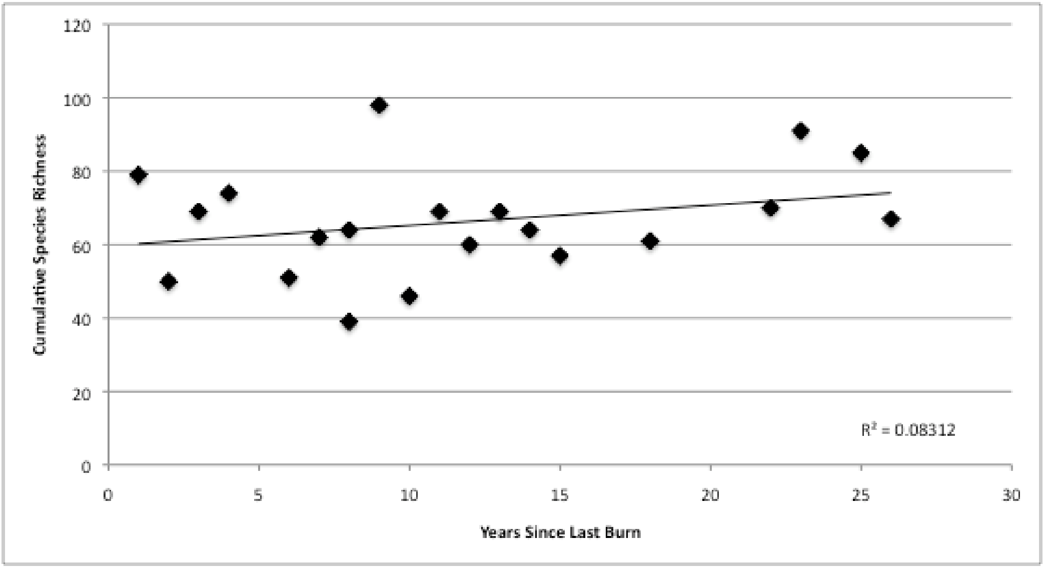

Figure 6. Cumulative species richness in relation to fire age. A line of best fit has been added to examine the correlation strength of the relationship.

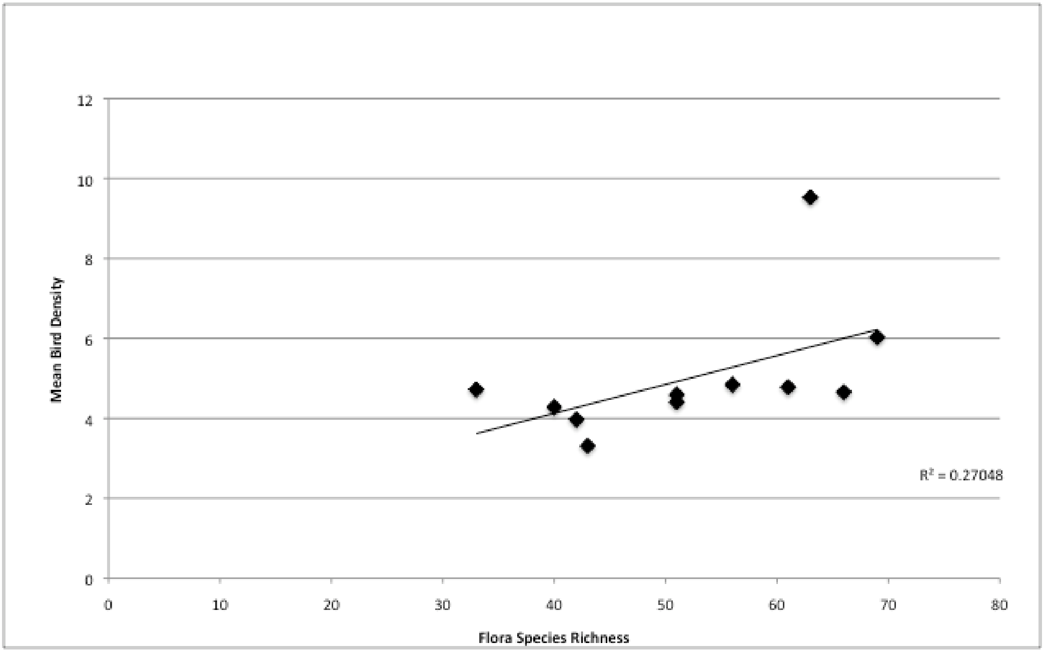

Figure 7. Correlation between mean bird species richness and floral species richness

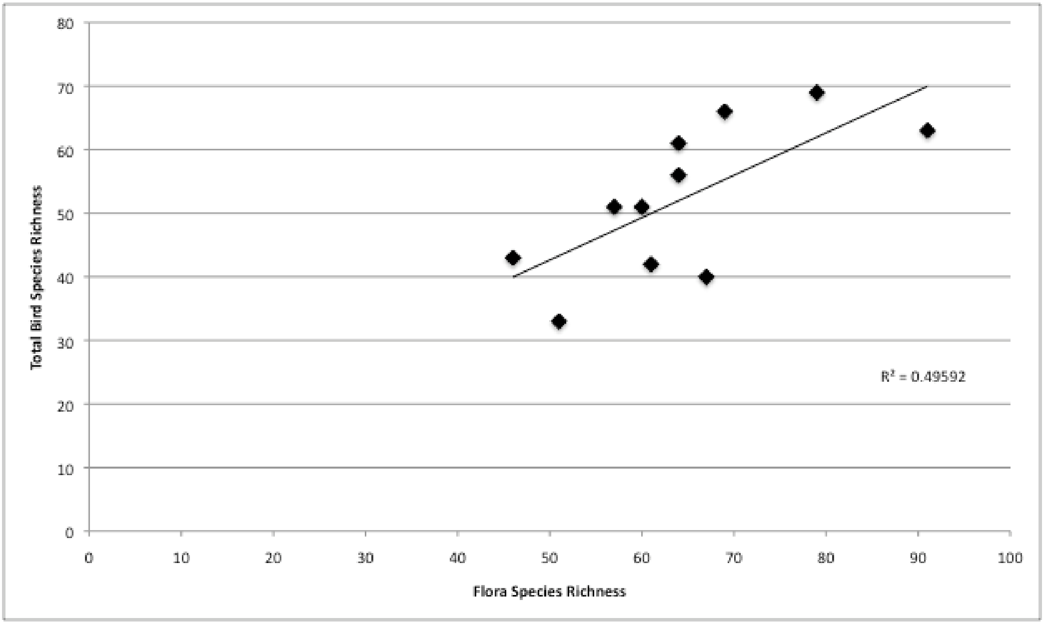

Figure 8. Correlation between cumulative bird species richness and floral species richness.

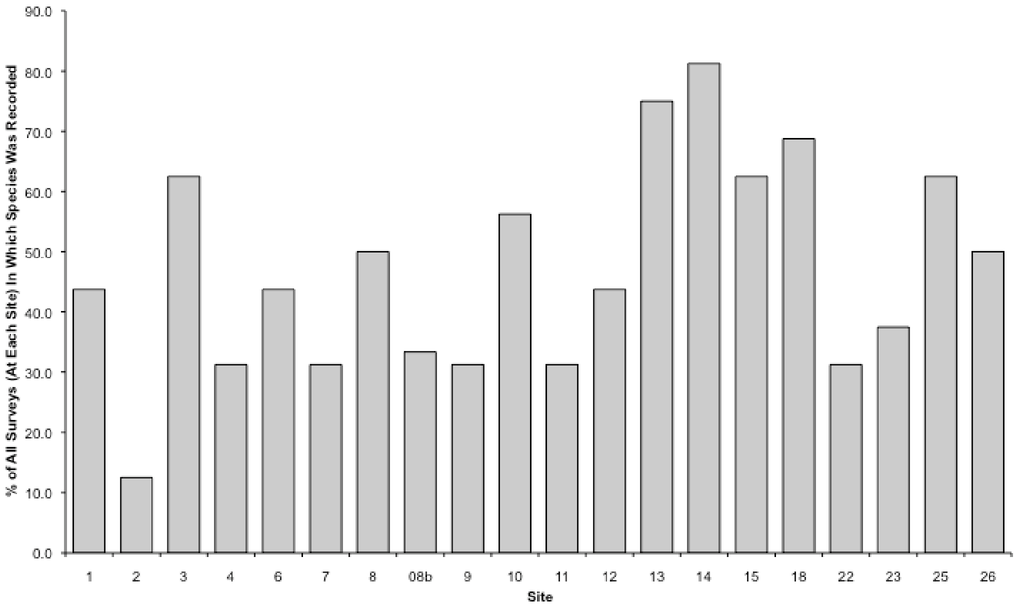

Figure 9. Relative occurrence (number of surveys in which the species was recorded) for Splendid Fairy-wrens.

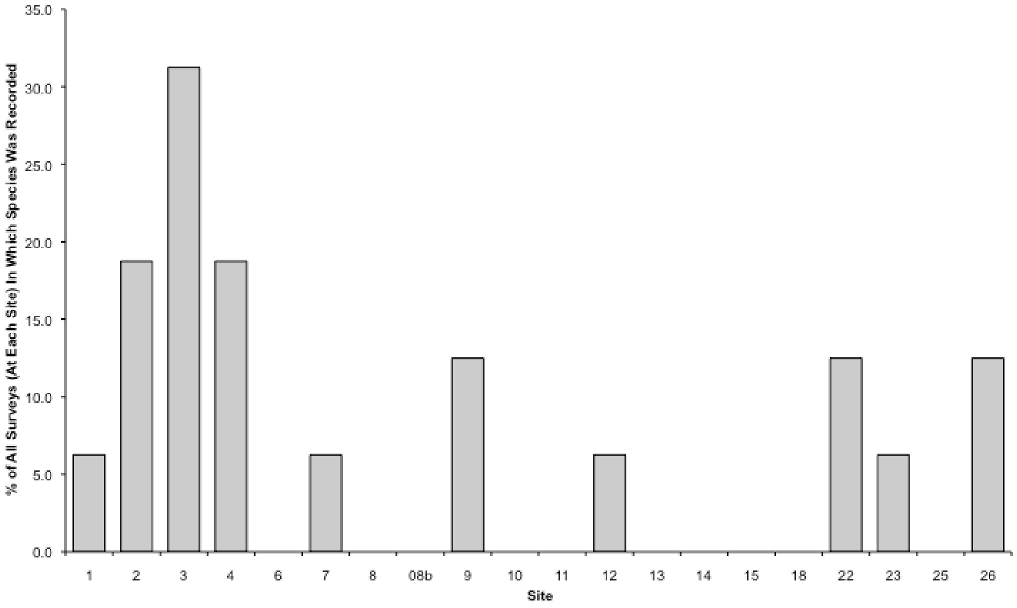

Figure 10. Relative occurrence (number of surveys in which the species was recorded) for Yellow-rumped Thornbills.

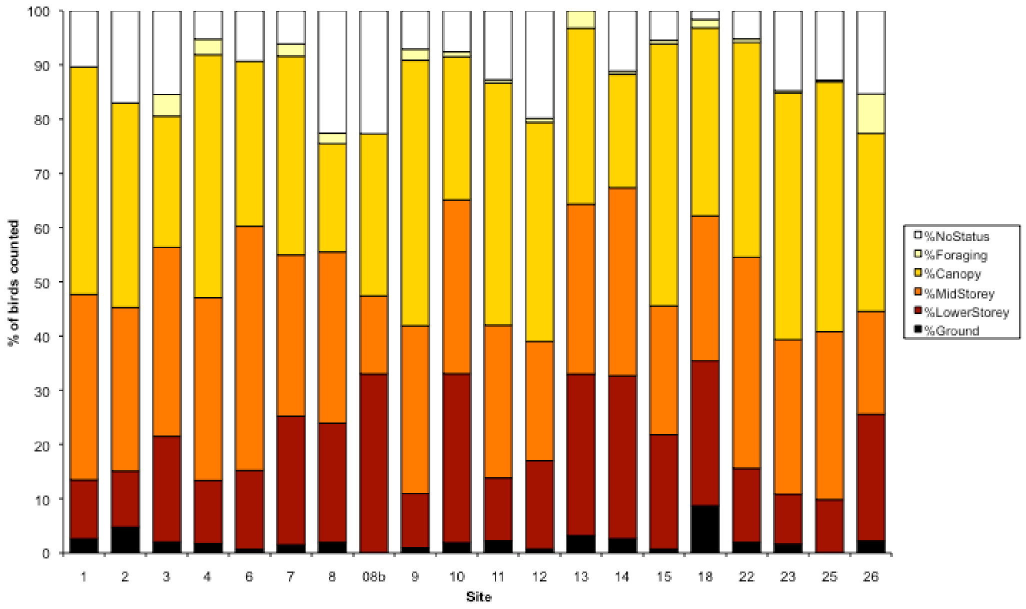

Figure 11. Habitat use by all birds in each fire age.

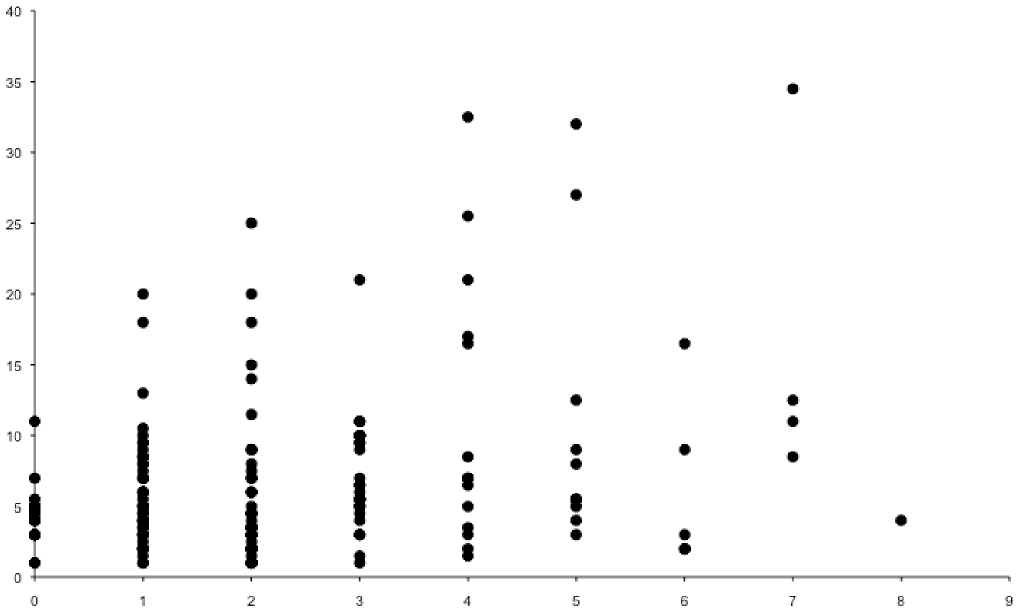

Figure 12. Plot of honeyeater abundance in relation to flowering score (R2 = 0.105).

## Dieback Studies

It is apparent that both mean species richness and bird density are lower in dieback-affected sites (Table 3). An ANOVA indicated that bird density was not significantly different (but was nearly so) with p=0.053. However, an ANOVA indicated a significant difference in species richness between sites (p = 0.007).

An examination of seasonal trends for both dieback and control sites showed that bird species richness was highest in winter and autumn and also was higher at control sites than dieback-affected sites (Figure 13). Sites 1-3 had peak richness at control sites in winter, whereas sites 5-7 had peak richness at control sites in autumn. Site 4 was the only site in which species richness was higher in the dieback-affected area than in the control area (Figure 13).

Table 3. Bird density and species richness in dieback affected and unaffected sites.

| Site | Number of Surveys | Bird Density | Species Richness |
|---|---|---|---|
| DB1.d | 8 | 5.00 ± 1.39 (0–12) | 2.38 ± 0.68 (0–6) |
| DB1.u | 8 | 5.13 ± 1.23 (0–10) | 3.25 ± 0.65 (0–5) |
| DB2.d | 7 | 3.57 ± 1.62 (0–10) | 2.00 ± 0.76 (0–5) |
| DB2.u | 7 | 3.43 ± 1.13 (0–9) | 2.00 ± 0.49 (0–3) |
| DB3.d | 8 | 6.75 ± 1.72 (2–14) | 3.13 ± 0.67 (1–6) |
| DB3.u | 8 | 9.13 ± 2.31 (3–19) | 4.63 ± 0.82 (2–9) |
| DB4.d | 8 | 4.63 ± 1.31 (0–9) | 2.38 ± 0.56 (0–5) |
| DB4.u | 8 | 2.75 ± 1.80 (0–15) | 1.50 ± 0.76 (0–6) |
| DB5.d | 8 | 2.25 ± 0.73 (0–5) | 1.25 ± 0.41 (0–3) |
| DB5.u | 8 | 7.88 ± 1.76 (1–15) | 4.00 ± 0.73 (1–6) |
| DB6.d | 8 | 5.00 ± 1.31 (2–13) | 1.88 ± 0.48 (1–5) |
| DB6.u | 8 | 7.75 ± 1.52 (1–15) | 3.25 ± 0.49 (1–5) |
| DB7.d | 7 | 3.29 ± 1.71 (0–13) | 1.71 ± 0.81 (0–6) |
| DB7.u | 7 | 6.29 ± 1.86 (0–14) | 3.43 ± 1.13 (0–8) |
| All 'dieback' sites (pooled) | 54 | 4.39 ± 0.54 (0–14) | 2.11 ± 0.24 (0–6) |
| All 'unaffected' sites (pooled) | 54 | 6.09 ± 0.68 (0–19) | 3.17 ± 0.30 (0–9) |

With the exceptions of sites 1 and 2, mean bird density was higher at controls than in dieback-affected areas (Figure 14). The seasonal trend was not as clear for bird density, with peak densities in autumn for sites 4-7, winter for site 3 and spring for sites 1 and 2 (Figure 14).

Some species showed clear trends in their response to dieback. Tawny-crowned Honeyeaters were more common at dieback-affected sites (Figure 15), whereas conversely the Western Spinebill, Brown Honeyeater and Silvereye were all more prevalent at unaffected sites (Figure 16, Figure 17 and Figure 18).

All bird species appeared to use the ground and lower-storey more in dieback-affected sites and made more use of the mid-storey and canopy in control sites (Figure 19).

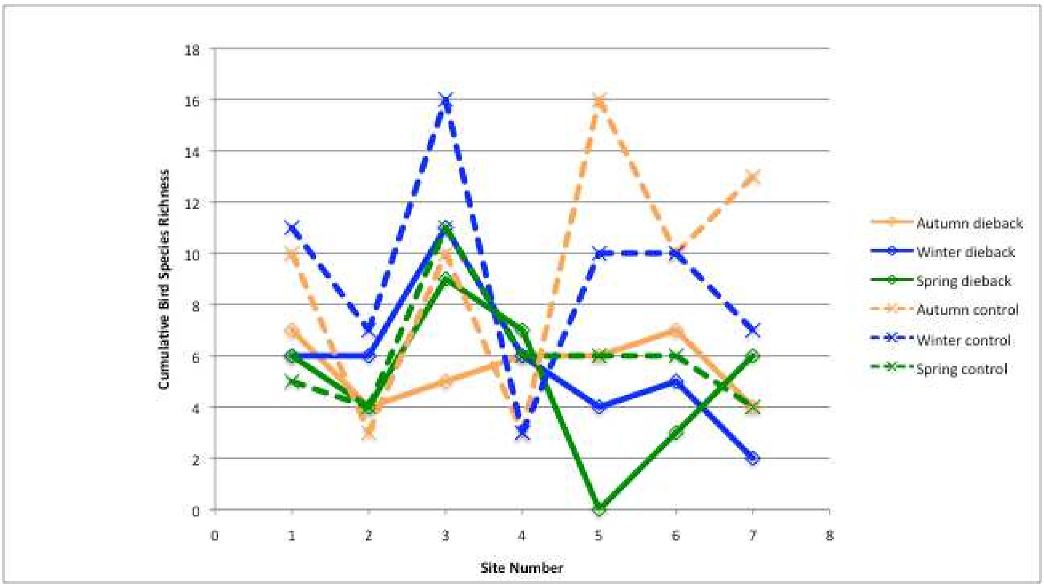

Figure 13. Cumulative species richness of dieback and control sites in relation to season.

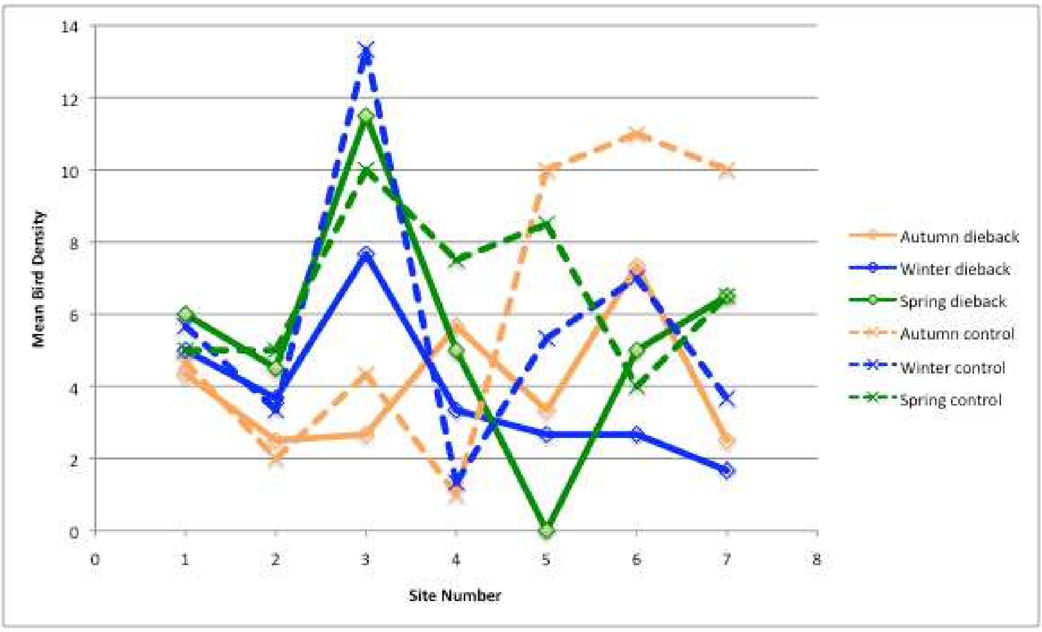

Figure 14  Mean bird density of dieback and control sites in relation to season.

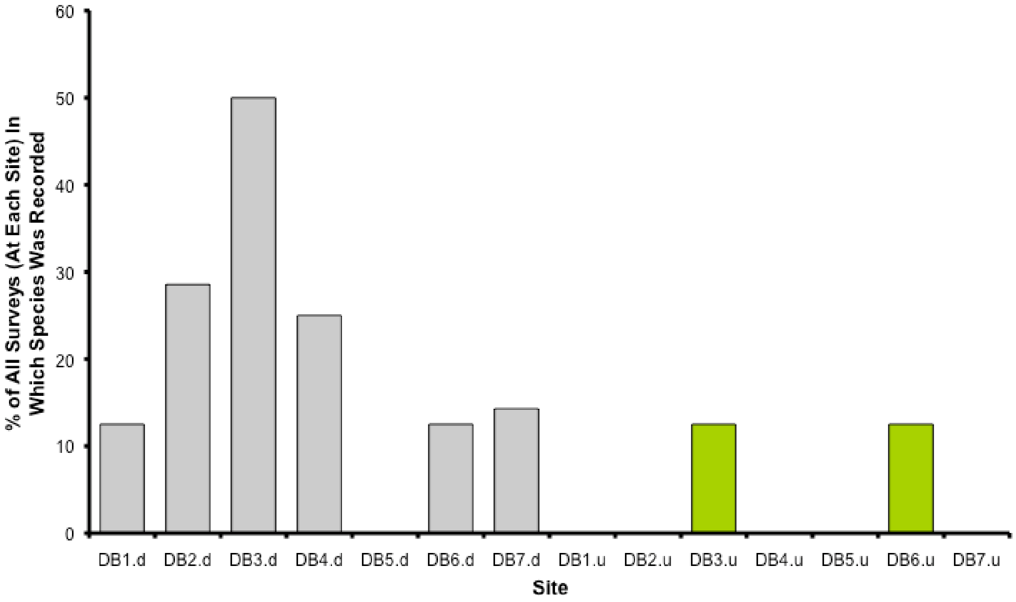

Figure 15. Occurrence of Tawny-crowned Honeyeaters in dieback (d) and unaffected (u) sites.

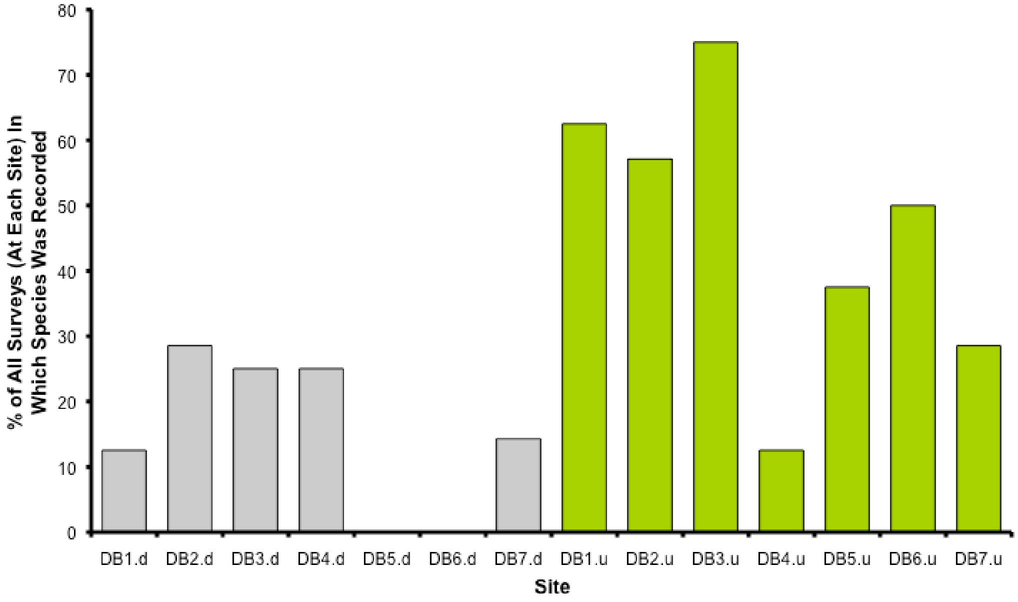

Figure 16  Occurrence of the Western Spinebill in dieback (d) and unaffected (u) sites.

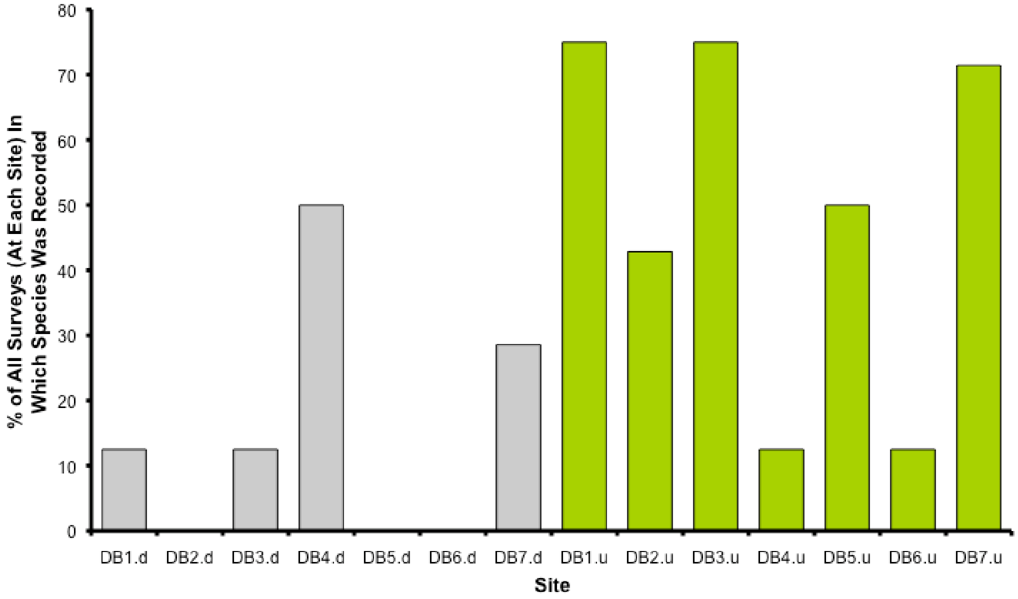

Figure 17 Occurrence of the Brown Honeyeater in dieback (d) and unaffected (u) sites.

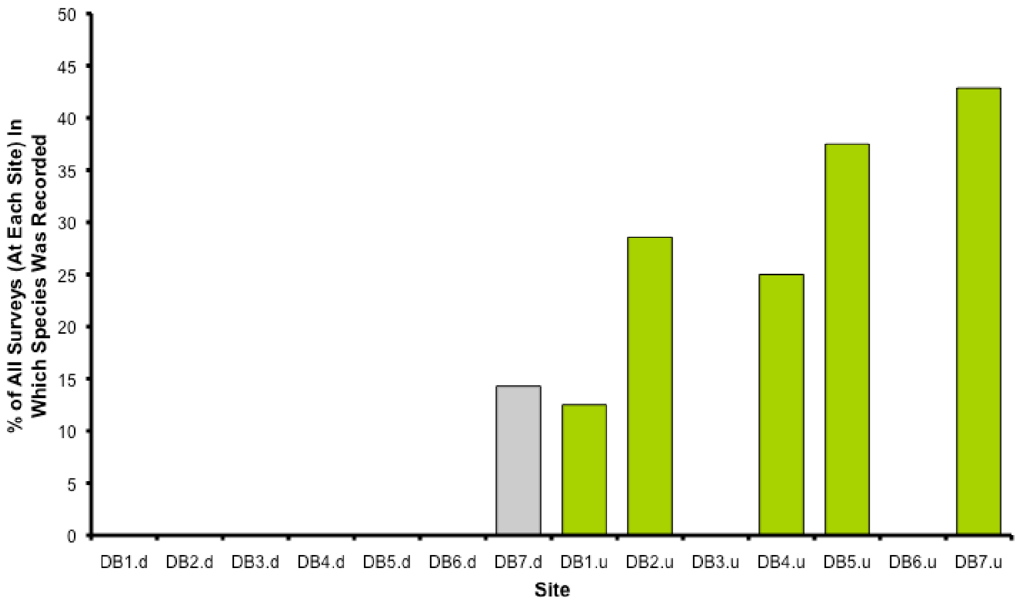

Figure 18. Occurrence of the Silvereye in dieback (d) and unaffected (u) sites.

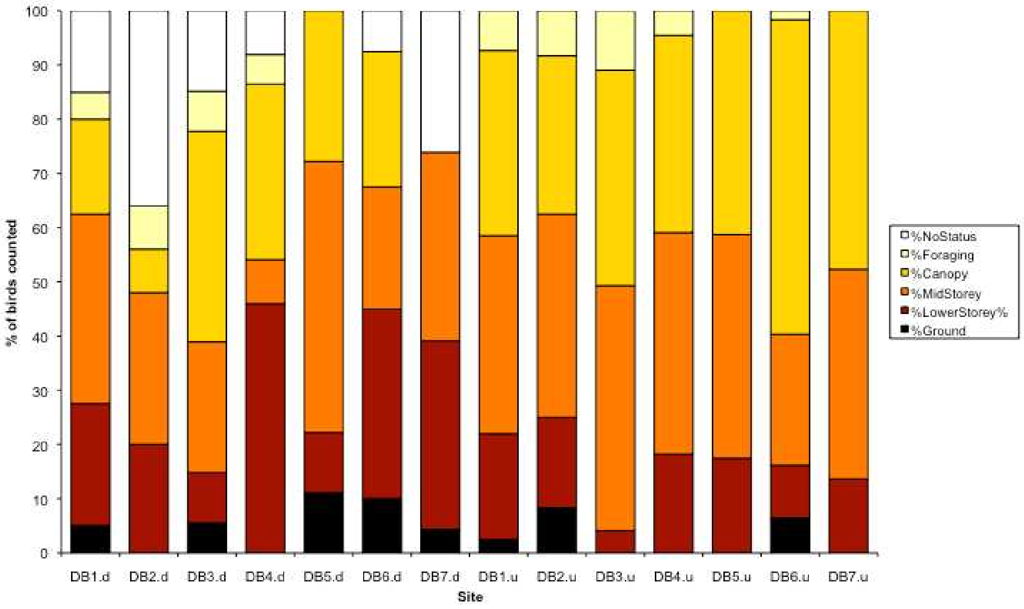

Figure 19. Habitat strata use by birds in dieback (d) and unaffected (u) sites.

# DISCUSSION

## Fire studies

Studies have found that bird species richness is usually highest in long-term unburnt vegetation (15 years post-burn) but is also high for several years following a fire (Burbidge 2003). Species diversity can also increase and be higher in burnt areas for up to 3 years post-burn (Christensen *et al.* 1985). Although these results were not clearly reflected in this study, the highest mean density and total species richness occurred at the longest unburnt sites (23 and 25 years post-fire). However, overall there was only a weak positive relationship between density and richness and time since fire.

Burbidge (2003) also reported that the abundance of birds drops to very low levels immediately following a fire but usually recovers within 2-3 years. Interestingly, the bird density data does show low densities for the 2 year old burn site followed by an increase, however there are equally low densities at the ten year-old site, so this is not likely to be meaningful.

The reasons for the weak trends observed may be an issue of power e.g. the need for more samples, an issue of correlation of measures (e.g. bird density was highest and lowest at most of the same sites as bird species richness - indicating correlation) or the importance of other factors besides fire age, contributing to the observed variation.

The observed results may also be correct. In contrast to Burbidge’s (2003) observations, a number of studies have also found no effect of fire on species richness, diversity or density. Research in Arizona, USA, found no change in species richness or evenness due to fire (Hurteau 2008). Hurteau (2008) found that some key species were affected, but over the 4 years of the study there was no detectable overall impact on community structure or abundance. Similarly, another study in the USA found that species richness and diversity were unaffected by burning (Van Dyke *et al.* 2007). Studies in Australia have also found no relationship between time since fire and species richness (Lindenmayer *et al.* 2008).

Burbidge (2003) stated the importance of the impact of fire frequency and intensity rather than just time since fire, and found that intense fires caused a greater reduction in species richness. Similarly, Lindenmayer *et al.* (2008) found that fire frequency was significantly correlated with bird species richness in Australian woodlands. It may be that these factors are more important in determining bird responses to fire and future research would benefit from examining these aspects.

Interestingly, there was a strong correlation between floral species richness and both bird species richness and density. Sites with a greater diversity and number of plants therefore had a higher richness and density of birds. Lindenmayer *et al.* (2008) reported a similar finding, with more vertically complex and species rich habitats (such as forest) supporting a higher richness of birds than more structurally simple habitats. In the current study habitat variation was minimised by ensuring that all sites were situated in the same vegetation community (*Banksia menziesii* dominated woodlands) on the same soil system (Bassendean complex), however there is still inherent variation between sites, especially related to topography and this may influence the findings.

Although we found no correlation between bird richness and density with time since fire, other studies have found that time since fire is clearly associated with difference in vegetation structure, which could in turn affect the avian community. Woinarski *et al.* (2004) found that there were marked differences in vegetation structure and composition between burnt and long-unburnt areas. In their study, more frequently burnt areas were moving towards a herbaceous annual-dominated vegetation community and longer unburnt areas were characterised by a greater richness of woody perennial species (Woinarski *et al.* 2004). In addition, species composition, basal stem area and groundcover all increased in longer unburnt habitats (Woinarski *et al.* 2004). Such changes are likely to occur in the GSS area and would suggest that longer unburnt areas would be dominated by a greater diversity of habitats, particularly denser understorey. This was supported by this study. Splendid Fairy-wrens show a preference for denser understorey habitats with most records in our study being from the understorey layer. Consequently Splendid Fairy-wrens were recorded more often in longer unburnt sites, presumably because understorey was more extensive.

Seasonal changes in the avifaunal community were observed in this study, however these varied between sites. Much of this variation is likely due to the floristic composition of sites, and particularly the influence of flowering banksias on the richness and density of nectarivores. Nectarivore densities in Australia are closely correlated with the flowering of *Eucalyptus* and *Banksia* species (McGoldrick and Mac Nally 1998; Ramsey 1989). Nectarivores are generally adapted to utilise flowering resources at a landscape level and there is a strong, positive correlation between nectarivore density and flowering intensity/nectar availability (McGoldrick and Mac Nally 1998). This trend was also evident in the current study and it is apparent that the *Banksia* woodlands of this region are an important resource for nectarivores. The primary impact of fire on nectarivore density and richness may be through changes to nectar production.

The two most abundant bird species in all surveys were nectarivores – the generalist Brown Honeyeater and the common *Banksia* woodland species, the Western Spinebill. The other two most common species were both sedentary insectivores, the Splendid Fairy-wren and Western Thornbill. The primary impact of fire on insectivores is likely to be in changing the structure of the vegetation and also on its impacts on the diversity and abundance of insect prey. As previously noted, Splendid Fairy-wrens were mostly recorded in understorey and were thus less often recorded in more open sites (e.g. after a recent fire). The Western Thornbill made equal use of several strata and was less likely to be impacted to changes in vegetation structure, thus occurring equally across a range of different fire ages. Conversely the Yellow-rumped Thornbill was more commonly recorded in burnt areas. This species favours edge habitats and open ground for foraging and thus benefits from the open habitats created by fire.

## Impacts of Dieback

The role of dieback is similar to fire in that it causes wholesale structural and compositional change to vegetation communities. In this study, it appeared that dieback had a more profound impact on birds than fire, with mean species richness and bird density lower in dieback affected sites. A study of bird utilisation of the Jarrah forest also found that in severely-dieback affected sites, the density, richness and diversity of birds was less than that of healthy forest (Nichols and Watkins 1984). Similarly, Armstrong and Nichols (2000)reported that overall bird densities were lower in dieback-affected sites as opposed to natural forest.

In the study region, the primary impact of dieback is the loss of *Banksia menziesii*, *B. ilicifolia* and *B. attenuata*, all of which are the dominant overstorey species (Shearer and Hill 1989). The dieback resistant *Eucalyptus todtiana* and *Nuytsia floribunda* are usually the only major trees left standing in dieback-affected areas (Shearer and Hill 1989). Such large changes to the structural and species composition of the vegetation naturally influence the composition of the avifaunal community.

In this study, some species showed clear trends in their response to dieback. Tawny-crowned Honeyeaters for example were more common at dieback-affected sites whereas the Western Spinebill, Brown Honeyeater and Silvereye were all more prevalent at unaffected sites. Overall, bird species made more use of the ground and lower-storey in dieback-affected sites. All of these changes are a result of the differences in vegetation structure and species composition. A study in the Jarrah forest of Western Australia found that dieback areas were characterised by bird species that favour more open habitats, edges and farmlands, such as Willie Wagtails and Australian Magpies (Armstrong and Nichols 2000). It was also noted that a number of species found in healthy Jarrah forest were in reduced abundances in dieback –affected forest, including the insectivorous Western Yellow Robin, Grey Shrike-thrush and Rufous Tree-creeper (Armstrong and Nichols 2000).

While the observed differences in Jarrah forest were due primarily to changes in tree density and canopy cover, the impacts of dieback in *Banksia* woodlands are more profound and particularly targeted as proteaceous species that provide important nectar sources for honeyeaters. Consequently (and converse to the Jarrah forest), nectar-dependent species are considered to be the most at risk from the impacts of dieback (Garkaklis *et al.* 2004). This is exemplified by the fact that the Western Spinebill and Brown Honeyeater were more frequently recorded in unaffected areas than in dieback-affected woodland.

Insectivores may also be adversely affected by dieback. Studies in the Jarrah forest concluded that litterfall and litter mass were severely reduced in dieback-affected areas and that the densities of litter invertebrates were also lower (Postle *et al.* 1986). A number of species that are rare or declining in the GSS are ground-pouncing insectivores such as the White-breasted, Western Yellow, Red-capped, Scarlet and Hooded Robins and all species are closely associated with leaf litter cover and thus likely to be detrimentally impacted by dieback primarily through changes to prey availability (Recher *et al.* 2002).

Conversely some species are favoured by dieback. Both White-winged Fairy-wrens, Yellow-rumped Thornbills and Tawny-crowned Honeyeaters were recorded more frequently in dieback areas. White-winged Fairy-wrens and Tawny-crowned Honeyeaters favour heathland habitats over woodland and Yellow-rumped Thornbills favour ecotonal edges and open habitats for their ground-based foraging. In a study in the Jarrah Forest, the Red-eared Firetail was also unaffected by dieback as it did not rely on dieback susceptible species for any of its habitat or resource requirements (Nichols *et al.* 1982). It is evident that the impacts of dieback are likely to depend upon the ecology of individual species, particularly their habitat preference and dietary requirements.

The seasonal trends observed between dieback sites, probably reflect the flowering of key proteaceous species and the influx of birds associated with this event. Although there were seasonal trends, species richness and density were higher in all seasons in control sites and this is doubtless a result of the greater diversity of habitats and resources available in control sites.

# CONCLUSIONS

In summary, it is apparent that both fire and *Phytophthora* dieback do have a significant impact on the avifauna of the GSS area. Although time since fire was not significantly related to changes in species richness or density these measures were strongly correlated with the floral richness of sites and it is likely that the primary impacts of fire are likely to be on the structure and composition of vegetation at sites and this will in turn affect avifauna. Fire intensity and frequency would be valuable information to include in further studies of this nature, as would information on habitat variables such as structure.

*Phytophthora* dieback appeared to have a more significant role than fire in terms of observed changes to the avifaunal community. Dieback causes major changes to the floristics, structure and resources available for birds at sites and consequently we observed major differences in species richness and density between dieback and control areas. Key species are likely to be advantaged by dieback and these are species that favour open habitats and have more generalist life strategies. Those species that depend upon proteaceous plants for nectar are most likely to be impacted, as well as ground-pouncing insectivorous birds.

# APPENDIX

**Appendix 1. Locality and fire age of fire study sites.**

| Site ID | Years since Last Burn | Location | Easting | Northing |
|---|---|---|---|---|
| 1 | 1 | Miltary Road (opp. Redwood Road) | 375694 | 6525013 |
| 2 | 2 | Redwood Road | 380764 | 6524901 |
| 3 | 3 | Warbrook Road | 399832 | 6488446 |
| 4 | 4 | Tick Flat (Redwood Road) | 376230 | 6525062 |
| 6 | 6 | Nature reserve limestone track off Sandringham Road | 385531 | 6528306 |
| 7 | 7 | Near Gingin RAAF airfield | 392522 | 6520706 |
| 8 | 8 | Off Neaves Road | 396579 | 6495093 |
| 8a | 8 | South of Duffy Road | 390609 | 6523592 |
| 9 | 9 | Limestone powerline track off Perry Road | 390047 | 6495562 |
| 10 | 10 | Clover Road | 382993 | 6516712 |
| 11 | 11 | Warbrook Road | 399832 | 6488446 |
| 12 | 12 | Gas pipeline track south of Warbrook Road | 400564 | 6487882 |
| 13 | 13 | Quinn Road | 378768 | 6530533 |
| 14 | 14 | Off Neaves Road | 396495 | 6494736 |
| 15 | 15 | Nature reserve limestone track off Sandringham Road | 384882 | 6528624 |
| 18 | 18 | Gas pipeline track south of Warbrook Road | 400564 | 6487882 |
| 22 | 22 | Redwood Road | 380764 | 6524901 |
| 23 | 23 | Limestone powerline track off Perry Road | 389975 | 6496363 |
| 25 | 25 | Limestone powerline track off Perry Road | 390047 | 6495562 |
| 26 | 26 | Near Gingin RAAF airfield | 392366 | 6520861 |

**Appendix 2 Location of *Phytophthora* dieback study sites**

| Site ID | Dieback Status | Easting | Northing |
|---|---|---|---|
| DB1_u | Uninfested | 399948 | 6488792 |
| DB1_d | Dieback Infested | 400051 | 6488789 |
| DB2_u | Uninfested | 400237 | 6489767 |
| DB2_d | Dieback Infested | 400277 | 6489856 |
| DB3_u | Uninfested | 400622 | 6488603 |
| DB3_d | Dieback Infested | 400616 | 6488703 |
| DB4_u | Uninfested | 399619 | 6488207 |
| DB4_d | Dieback Infested | 399658 | 6488140 |
| DB5_u | Uninfested | 392315 | 6495105 |
| DB5_d | Dieback Infested | 392316 | 6495017 |
| DB6_u | Uninfested | 391842 | 6494585 |
| DB6_d | Dieback Infested | 391803 | 6494669 |
| DB7_u | Uninfested | 400217 | 6490219 |
| DB7_d | Dieback Infested | 400256 | 6490314 |

**Appendix 3. The percentage of all quadrats surveyed each month, in which a species was recorded**

| Common Name | October 2008 | November 2008 | December 2008 | March 2008 | April 2008 | May 2008 | June 2008 | July 2008 | August 2008 | Mean |
|---|---|---|---|---|---|---|---|---|---|---|
| Emu | | | | 5.0 | | | | | | 0.56 |
| Quail sp. | | | | | | | | | 2.5 | 0.28 |
| Square-tailed Kite | | | 5.0 | | | | | | | 0.56 |
| Brown Goshawk | 2.5 | 5.0 | 10.0 | | | | 2.5 | | | 2.22 |
| Wedge-tailed Eagle | | | 5.0 | | | | | 2.5 | | 0.83 |
| Little Eagle | 2.5 | 2.5 | | | | | | | | 0.56 |
| Brown Falcon | 2.5 | 2.5 | 10.0 | | | | | 2.5 | 2.5 | 2.22 |
| Painted Button-quail | | 5.0 | 5.0 | | 7.7 | 5.3 | 2.5 | 5.0 | 5.0 | 3.94 |
| Common Bronzewing | 7.5 | 5.0 | 5.0 | | 2.6 | 2.6 | 2.5 | | 2.5 | 3.08 |
| Western Rosella | | 2.5 | | | | | | | | 0.28 |
| Australian Ringneck | 7.5 | | 5.0 | | 7.7 | 18.4 | 5.0 | 10.0 | | 5.96 |
| Red-capped Parrot | | | | 10.0 | | 18.4 | 12.5 | 5.0 | 5.0 | 5.66 |
| Pallid Cuckoo | | 2.5 | | | | | | | 5.0 | 0.83 |
| Fan-tailed Cuckoo | | | | | | 2.6 | | | | 0.29 |
| Horsfield's Bronze-Cuckoo | | 2.5 | 5.0 | | | | | | | 0.83 |
| Shining Bronze-Cuckoo | | | 5.0 | | | | | | 2.5 | 0.83 |
| Tawny Frogmouth | | | | | | 2.6 | | | | 0.29 |
| Australian Owlet-nightjar | | | | | | 2.6 | | | | 0.29 |
| Fairy-wren sp. | 5.0 | 7.5 | 5.0 | | | | | | | 1.94 |
| Splendid Fairy-wren | 47.5 | 60.0 | 70.0 | 40.0 | 38.5 | 60.5 | 50.0 | 30.0 | 35.0 | 47.94 |
| White-winged Fairy-wren | | 2.5 | | | | | | | 5.0 | 0.83 |
| Striated Pardalote | 2.5 | 2.5 | | | | | | | 2.5 | 0.83 |
| Weebill | 5.0 | | | | | | | | | 0.56 |
| Western Gerygone | 15.0 | 5.0 | | | | | | 7.5 | 2.5 | 3.33 |
| Inland Thornbill | | | | 5.0 | 2.6 | 7.9 | | | | 1.72 |
| Western Thornbill | 40.0 | 37.5 | 35.0 | 35.0 | 33.3 | 34.2 | 47.5 | 37.5 | 40.0 | 37.78 |
| Yellow-rumped Thornbill | 7.5 | 2.5 | 5.0 | 10.0 | 7.7 | 10.5 | 5.0 | 5.0 | 7.5 | 6.75 |

| Common Name | October 2008 | November 2008 | December 2008 | March 2008 | April 2008 | May 2008 | June 2008 | July 2008 | August 2008 | Mean |
|---|---|---|---|---|---|---|---|---|---|---|
| Red Wattlebird | 5.0 | 2.5 | 10.0 | 10.0 | 23.1 | 13.2 | 25.0 | 10.0 |  | 10.97 |
| Western Wattlebird | 7.5 | 17.5 | 20.0 | 15.0 | 20.5 | 26.3 | 25.0 | 22.5 | 7.5 | 17.98 |
| Singing Honeyeater |  | 5.0 | 5.0 | 5.0 |  | 7.9 | 15.0 | 2.5 |  | 4.49 |
| Brown Honeyeater | 85.0 | 95.0 | 95.0 | 35.0 | 64.1 | 94.7 | 85.0 | 57.5 | 62.5 | 74.87 |
| New Holland Honeyeater | 15.0 | 7.5 | 10.0 | 35.0 | 18.0 | 21.1 | 22.5 | 30.0 | 17.5 | 19.61 |
| White-cheeked Honeyeater |  | 2.5 |  | 15.0 | 5.1 | 10.5 | 15.0 | 10.0 |  | 6.46 |
| Tawny-crowned Honeyeater | 5.0 | 12.5 | 10.0 | 5.0 | 5.1 | 15.8 | 12.5 | 5.0 | 5.0 | 8.44 |
| Western Spinebill | 50.0 | 62.5 | 45.0 | 70.0 | 53.9 | 73.7 | 85.0 | 50.0 | 40.0 | 58.89 |
| Scarlet Robin | 12.5 | 20.0 | 15.0 | 15.0 | 20.5 | 15.8 | 10.0 | 17.5 | 12.5 | 15.42 |
| Red-capped Robin | 2.5 |  |  |  |  | 2.6 |  |  | 2.5 | 0.85 |
| Hooded Robin | 2.5 | 5.0 |  |  |  |  | 2.5 | 2.5 | 2.5 | 1.67 |
| Varied Sittella |  |  | 5.0 |  |  | 2.6 | 2.5 | 7.5 |  | 1.96 |
| Golden Whistler |  |  |  | 5.0 |  |  |  |  |  | 0.56 |
| Rufous Whistler | 22.5 | 27.5 | 20.0 | 20.0 | 23.1 | 15.8 | 15.0 | 17.5 | 12.5 | 19.32 |
| Grey Shrike-thrush |  | 7.5 | 15.0 |  |  | 5.3 | 5.0 | 5.0 | 2.5 | 4.47 |
| Grey Fantail | 2.5 | 5.0 | 10.0 | 20.0 | 15.4 | 21.1 | 10.0 | 7.5 | 10.0 | 11.27 |
| Black-faced Cuckoo-shrike |  | 7.5 |  |  |  |  | 2.5 |  |  | 1.11 |
| Black-faced Woodswallow | 2.5 |  |  |  |  |  |  |  | 2.5 | 0.56 |
| Dusky Woodswallow |  |  |  | 5.0 |  | 2.6 | 2.5 | 2.5 | 5.0 | 1.96 |
| Grey Butcherbird |  |  |  |  | 2.6 | 2.6 |  |  |  | 0.58 |
| Australian Magpie |  |  |  |  | 2.6 |  | 5.0 | 2.5 | 2.5 | 1.40 |
| Australian Raven |  |  | 5.0 | 10.0 | 2.6 | 13.2 | 7.5 |  |  | 4.25 |
| Welcome Swallow |  |  |  |  |  | 5.3 | 2.5 |  |  | 0.86 |
| Tree Martin |  | 2.5 | 5.0 | 15.0 | 5.1 | 2.6 | 5.0 |  |  | 3.92 |
| Silvereye | 12.5 | 15.0 | 20.0 | 10.0 | 18.0 | 23.7 | 27.5 | 5.0 | 2.5 | 14.90 |

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