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Is bigger always better? Influence of patch attributes on breeding activity of birds in box-gum grassy woodland restoration plantings

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Abstract

Restoration plantings are an increasingly common management technique to address habitat loss in agricultural landscapes. Native fauna, including birds, may readily occupy planted areas of vegetation. However, unless restoration plantings support breeding populations, their effectiveness as a conservation strategy may be limited. We assessed breeding activity of birds in box-gum grassy woodland restoration plantings in the South-west Slopes bioregion of New South Wales, Australia. We compared breeding activity in plantings of different size (small and large) and shape (linear and block-shaped) to breeding activity in a set of remnant woodland sites. Contrary to expectations, we found that bird breeding activity was greatest per hectare in small patches. We also found a negative effect of planting age, with younger plantings supporting more breeding activity per hectare. We found no effect of patch type or shape on breeding activity, and that species' relative abundance was not predictive of their degree of breeding activity. Our results highlight the value of small habitat patches in fragmented agricultural landscapes, and indicate that restoration plantings are as valuable as remnant woodland patches for supporting bird breeding activity. We demonstrate the importance of breeding studies for assessing the conservation value of restoration plantings and other habitat patches for avifauna.

Keywords: Woodland birds, breeding success, SLOSS, restoration, fragmentation, agricultural landscapes

1. Introduction

Habitat loss due to land conversion for agriculture is a significant issue globally, with numerous effects on biodiversity and ecosystem processes (Maxwell et al., 2016; Tscharntke et al., 2012). Land clearing is increasing worldwide, particularly in productive agricultural regions (Evans, 2016; Tilman et al., 2017). The extensive removal of native vegetation creates a highly fragmented landscape in which patches of native vegetation exist primarily as small, isolated remnants. Restoration plantings in agricultural landscapes are increasingly implemented to address habitat loss and conserve threatened and declining native fauna, with hundreds of millions of hectares of vegetation being replanted around the world at a cost of billions of dollars (Crouzeilles et al., 2016). To ensure cost-effectiveness and ecological integrity, it is important to quantify the effects of revegetation on biodiversity and assess whether conservation goals are being met, particularly in the long term (Barral et al., 2015; Ruiz-Jaen and Aide, 2005).

A core assumption of restoration success is that revegetated patches provide high-quality habitat for the species they are intended to help conserve (Ikin et al., 2016; Ruiz-Jaen and Aide, 2005). In Australia, bird communities that inhabit box-gum grassy woodlands are threatened by ongoing habitat loss and degradation (Rayner et al., 2014), and are a frequent target of restoration efforts (Freudenberger, 2001; Lindenmayer et al., 2013; Smith, 2008). There is evidence suggesting that many bird species will readily occupy restoration plantings, in some cases preferentially inhabiting plantings over remnant woodland patches or other sites (Barrett et al., 2008; Cunningham et al., 2008; Lindenmayer et al., 2016), but how much do we know about the capacity of restoration plantings to support breeding populations of these species? The majority of studies examining avian responses to restoration plantings have used measures such as species richness, diversity, and relative abundance to make inferences about occupancy trends and habitat quality (Belder et al., 2018). However, focusing on occurrence patterns provides a limited picture of how birds are using a site (Chalfoun and Martin, 2007). It is therefore important to quantify whether indicators of long-term persistence, such as breeding activity, follow the same trends.

1.1. Research objectives

The underlying aim of this study was to assess whether birds are able to breed successfully in box-gum grassy woodland restoration plantings. Breeding success can be measured in several ways, with nest success and daily nest survival being commonly-used metrics (Stephens et al., 2004). However, searching for, and monitoring, nests requires considerable time and effort. An alternative, and perhaps more accessible, approach is to use indicators of breeding activity as a proxy for breeding success. For example, a scoring system developed by Mac Nally (2007) ranks observations of breeding behaviour according to how strongly they indicate breeding success (Table 1), providing a quantitative measure of the extent to which a given site supports successful breeding (Bennett et al., 2015; Mac Nally et al., 2010; Selwood et al., 2009). A method such as this provides a basis from which to commence the transition from traditional occupancy and abundance surveys to a more population-oriented approach to monitoring avian responses to restoration plantings. Importantly, it also facilitates the collection of breeding data on species of conservation concern, whose nests may be difficult to find in adequate numbers.

We sought to investigate bird breeding activity in the context of habitat restoration in a fragmented agricultural landscape. Specifically, we posed the following three questions:

Question 1. How does bird breeding activity in restoration plantings compare to breeding activity in remnant woodland patches?

We compared breeding activity in restoration plantings, similar-sized woodland remnants, and larger, more intact woodland remnants. In addition to investigating the entire bird assemblage, we assessed breeding activity for species of conservation concern, and cup-nesters vs. dome nesters (Appendix B). Remnant patches are generally considered to be high-value habitat within fragmented agricultural landscapes (Cunningham et al., 2014), and hence we predicted remnant sites would support more breeding activity than restoration plantings. We predicted that breeding activity would be highest in larger woodland remnants than in smaller, more isolated remnants and restoration plantings. We made this prediction because comparative studies have shown that species richness and abundance is typically highest in large, intact remnants (Hadley et al., 2018; Helzer and Jelinski, 1999; Martin et al., 2004; Munro et al., 2011). Many species of conservation concern are more closely associated with remnants than plantings (Kinross, 2004), so we also expected to observe more breeding activity from these species in remnants than in plantings.

Question 2. How do patch attributes affect breeding activity in plantings and remnant woodland patches?

We examined breeding activity in sites of varying size (small and large) and shape (linear and block-shaped). A key finding from pattern-based studies of bird distribution and abundance in fragmented landscapes is that larger patches support more species (Kavanagh et al., 2007; Shanahan et al., 2011; Watson et al., 2003). This is consistent with the resource concentration hypothesis, which posits that there are more resources and thus more individuals and greater species diversity in larger patches (Connor et al., 2007; Root, 1973). Previous species-specific studies have also found that avian reproductive success is positively correlated with patch size (Herkert et al., 2003; Hoover et al., 1995; Luck, 2003; Zanette et al., 2000). We therefore postulated that breeding activity would increase with patch size in parallel with bird species richness and abundance. Similarly, increasing patch linearity is typically associated with lower species richness and abundance (Kinross, 2004; Lindenmayer et al., 2018a, 2007). As such, we predicted more evidence of successful breeding in block-shaped than in linear patches.

We predicted a stronger negative response to decreasing patch size and increasing linearity for cup-nesters compared with dome-nesters. This was because edge-effects of predation are stronger in smaller and more linear sites (Fletcher et al. 2007; Helzer and Jelinski, 1999), and cup-nesters tend to be more vulnerable to predation than other nest types (Okada et al., 2017). We also predicted that species of conservation concern, many of which are area-sensitive (Ford et al., 2009; Watson et al., 2005), would show more evidence of breeding activity in larger, block-shaped sites.

We also tested for an effect of planting age. Previous studies report increases in bird species richness and abundance as plantings mature (Debus et al., 2017; Freeman et al., 2009; Lindenmayer et al., 2016). This is often attributed to the tendency of the vegetation structure and composition of restoration plantings to converge on that of remnant patches over time (Munro et al., 2011). We therefore predicted that increasing planting age would have a positive effect on bird breeding activity.

Question 3. Does breeding activity in restoration plantings and remnant woodland patches reflect species assemblage composition?

We predicted that breeding activity in our study sites would be reflective of the species assemblage present. That is, we expected the effects of patch attributes (type, size, shape) on relative abundance to be correlated with the effects of patch attributes on breeding activity scores.

2. Methods

2.1 Study area

We conducted this study in the South-west Slopes bioregion of New South Wales, Australia (Figure 1). The region is part of Australia's sheep-wheat belt and has been extensively cleared of native vegetation, with as little as 0.1% of the original vegetation remaining in intact condition (Thiele and Prober, 2000). Remnant patches consist predominantly of white box (*Eucalyptus albens*) / yellow box (*E. melliodora*) / Blakely's red gum (*E. blakelyi*) grassy woodland, which is a critically-endangered ecological community (Department of the Environment, 2018). Patches of red stringybark (*E. macrorhyncha*) woodland and mugga ironbark (*E. sideroxylon*) woodland are also present in our study region.

Figure 1 Location of study sites in the South-west Slopes Bioregion of New South Wales, Australia. Map created using ggmap for R (Kahle and Wickham, 2013).

2.2 Study sites

We used spring bird survey data collected over 12 years to select a subset of 12 restoration plantings from a set of long-term monitoring sites (Appendix A) (Cunningham et al. 2007). We selected sites on the basis that they satisfied our criteria for size and shape, and shared at least two of three key species in common – the superb fairywren (*Malurus cyaneus*), yellow-rumped thornbill (*Acanthiza chrysorrhoa*), and willie wagtail (*Rhipidura leucophrys*). We chose these species as they are relatively common, typically found in woodland communities, and encompass the two major nest types (one cup-nester and two dome-nesters). Additionally, the yellow-rumped thornbill is a species of conservation concern (Barrett et al., 2003). Nineteen of our 21 chosen sites contained all three target species, with two sites lacking the yellow-rumped thornbill. We attempted to control for the effects of competitive exclusion by selecting sites with low abundances of the noisy miner (*Manorina melanocephala*), as this hyper-aggressive species is known to have negative impacts on other species of native birds (Bennett et al., 2015; Maron et al., 2013). Our sites were separated geographically by at least 500 m to promote spatial independence.

Plantings were aged between 12 and 25 years, 1.3-7.7 ha in area, and 20-200 m in width. A typical planting contained a mature (flowering-age) *Eucalyptus* overstorey, an *Acacia* understorey, and a ground layer dominated by annual grasses (both native and exotic). The majority of planted species naturally occur in the study region. Some plantings also contained remnant trees, along with varying amounts of woody debris (fallen trees and branches).

We compared plantings with six box-gum grassy woodland remnants, also part of the long-term monitoring study. Remnant patch size ranged from 2.1 to 5.8 ha, with widths of 30-200 m. We also selected three large (47-110 ha) reference sites to represent intact remnant woodland in the study region (two travelling stock reserves, and one remnant on private property). Remnant sites were dominated by a *Eucalyptus* overstorey, with or without an *Acacia* understorey, and typically contained woody debris in the form of fallen trees and branches.

2.3 Bird surveys

To assess breeding activity, we conducted fixed time-per-unit-area surveys (one hour per hectare) in our study sites over two spring breeding seasons. The peak breeding season for the majority of bird species in our study region is September to December (Appendix B). We completed two rounds of surveys in 2015 (October and November), and three rounds in 2016 (September, October, November). We searched sites systematically, identifying and recording indicators of breeding behaviour (Table 1). We designated search areas by the size and shape of sites. For sites < 3 ha, we searched 1.3 ha within the site – this was equivalent to the area of the smallest study site. For sites > 3 ha, we searched 3 ha within the site. We surveyed block sites in a grid fashion, and linear sites along their length until we had searched the desired area (i.e. 1.3 ha or 3 ha). We surveyed sites throughout the day, with the exception of November 2016 – in this period we completed surveys in the 4 hours post-sunrise and 4 hours pre-sunset. On average, there was an interval of 4.5 weeks between surveys at each site, and we structured the order of site visits to ensure that sites were not consistently surveyed at the same time of day. We did not conduct surveys during inclement weather. All breeding activity surveys were conducted by Author 1.

Table 1 Scores allocated to behavioural observations of breeding activity, modified from Mac Nally (2007).

Behaviour	**Score**
Feeding of young out of the nest	9.0
Fledglings seen	9.0
Nest with nestlings or feeding of young in the nest	8.0
Presence of juveniles or immature birds	7.5
Fledglings heard	7.5
Adult carrying food	6.0
Nest with eggs or adult on a nest	6.0
Nest empty or under construction (current breeding season)	5.0
Past breeding season’s nest	3.5
Adult gathering nest material	3.0
Courtship	2.0
Territorial behaviour	1.0
Male and female pairs	1.0

To quantify breeding activity, we used a survey method modified from Mac Nally’s (2007) scoring system. The Mac Nally (2007) method involves calculating an aggregate score of breeding activity in a study site over the course of a study. Scores are calculated based on ranking observations according to how strongly they indicate breeding success (Table 1), with a score of zero indicating no observations of breeding activity. Rather than aggregating breeding activity scores over the course of the study, we modified the method to calculate a score per survey. There were two reasons for this: first, it enabled us to test for effects of factors that may influence detectability of bird behaviour during surveys, such as weather and time of day. Second, it enabled us to account for repeat observations of the same individuals or nests across multiple surveys.

We conducted point count surveys in 2016 to quantify bird community composition and abundance in our study sites. Point count surveys in each site were typically conducted within two days of the surveys for breeding activity, and usually on the same day. Point count surveys in September were conducted by Author 1, and in October and November were completed by different observers (the entirety of each month's surveys conducted by a different observer). We divided each study site into 25 x 25 m cells, and randomly selected cells in which to conduct point counts. For sites > 3 ha, we selected six cells, and for sites <3 ha, we selected three cells. We ensured adjacent cells were not selected. At the centre of each randomly-chosen cell, we completed a five-minute count, recording counts of birds detected within 50 m of the survey point.

2.4 Statistical analyses

We used a model selection approach to investigate the effects of patch attributes on the total breeding activity score recorded in each survey (Table 2). We used linear mixed effects regression models with study site and survey year as random effects to account for repeated visits to sites over multiple years. The explanatory variables of primary interest were site type, size, and shape, and age of plantings. We included the variable "fenced", to account for potential effects of cattle grazing in our study sites (Lindenmayer et al., 2018b). Our response variable was a total breeding activity score standardised by survey area (1.3 or 3.0 ha), and was square-root-transformed to improve the distribution of the data. We also scaled and centred our continuous predictor variables. Prior to fitting models with our explanatory variables of interest, we examined variables likely to influence detectability in surveys, including time of day, temperature, and wind. In addition, we accounted for variation in activity through the breeding season by including Julian date. We found that breeding activity increased with Julian date for the woodland assemblage and all subsets of the assemblage, so included it as an explanatory variable in subsequent models. There were no other weather or temporal variables of statistical significance (Appendix D).

Prior to fitting models, we checked all explanatory variables for multi-collinearity using variance inflation factors. We corrected for multi-collinearity by removing large reference sites from models that included both size and shape. We also removed temperature due to its correlation (0.53) with time of day. We checked for a quadratic effect of time of day and found none. After fitting models, we checked for spatial autocorrelation in the data using variograms of the residuals. We detected no evidence of a nugget or sill in the variograms, and therefore assumed no spatial autocorrelation.

For our analyses, we included data for all terrestrial species recorded during breeding activity surveys, with the exception of introduced species (

Appendix B). We hereafter refer to this assemblage as the "woodland assemblage". For babblers and finches, we included data on nests only when they could be positively identified as true nests – these species build roost nests, which can be difficult to distinguish from true nests. We subset the woodland assemblage to investigate species of conservation concern, and compared cup-nesters with dome-nesters. We defined species of conservation concern as those listed as threatened in New South Wales (NSW Environment and Heritage 2018), along with those whose reporting rates declined by >20% in the South-west Slopes bioregion between the first and second *Atlas of Australian Birds* (Barrett et al. 2003). We classified cup-nesters and dome-nesters as per Morcombe (2003) and Pizzey and Knight (1997). The dome-nester group was highly correlated (0.79) with the woodland assemblage, as were species of least concern (0.91), so we did not analyse these groups separately. In addition to examining species of conservation concern and cup-nesters, we subset the woodland assemblage data to remove the most dominant species (superb fairywren, yellow-rumped thornbill, and willie wagtail).

For the woodland assemblage, and each subset, we followed a three-step modelling approach:

1. We first accounted for variation in our response variable associated with weather and temporal factors. We incorporated variables of significance into subsequent models.
2. We then modelled our response variable against site type, comparing plantings, remnants, and large reference sites.
3. Finally, we modelled our response variable against size and shape in plantings and remnants, excluding large reference sites.

In each step, we fitted global models with all combinations of the variables of interest, and ranked candidate models using Akaike's Information Criterion corrected for small sample sizes (AIC_c). We considered models with $\Delta AIC_c \leq 2$ as top-ranked models (Burnham and Anderson, 2004). Weather and temporal variables of significance identified in Step 1 were included in both Step 2 and Step 3.

We used the packages ‘lme4’ (Bates et al., 2015) and ‘MuMIn’ (Bartoń, 2018) in R version 3.4.4 (R Core Team 2018) to fit and select models. Variograms were constructed using the package ‘geoR’ (Ribeiro and Diggle 2016).

Table 2 Linear mixed model parameters. The response variable is SCORE, and all other variables are predictors.

Variable name	Description
SCORE	Square root of score of breeding activity recorded during surveys, calculated per Mac Nally (2007) and standardised by survey area (score/1.3 for small sites, score/3.0 for large sites)
TYPE	Site type (planting, remnant, reference)
SIZE	Site size (ha)
SHAPE	Measure of site shape, calculated as perimeter/width (m)
AGE	Age of planting at the commencement of the study (years)
FENCED	Site fenced from cattle (yes/no)
SUN	Subjective measure of sun during surveys, on a numerical scale of 1-4 where 1 = full sun and 4 = overcast
TEMP	Subjective measure of temperature during surveys, on a numerical scale of 1-8 where 1 = cold and 8 = hot
WIND	Subjective measure of wind during surveys, on a numerical scale of 1-8 where 1 = calm and 8 = strong wind
TIME	Time of day surveys commenced, given as no. hours post-sunrise (hr)
DATE	Julian date on which surveys were conducted

We used multivariate latent variable models from the package ‘boral’ (Hui, 2016) to compare how abundance and breeding activity for bird species responded to site type, size, and shape. This approach is useful because it allows for investigation of the association between multiple species and underlying environmental variables in a linear modelling framework, while also accounting for potential correlations among species. Specifically, we constructed one latent variable model for each response matrix, and then compared the coefficient estimates for each species and variable. For this modelling approach, only species detected both in point count surveys and breeding activity surveys could be included. We subset our data to an assemblage of interest that included woodland-dependent species (Silcocks et al. 2005) and several other small-bodied species that characterise the bird community of woodlands in our study region (Appendix C). Due to the disproportionate spatial influence of the frequently-detected superb fairywren in our initial ordination plots, we excluded it from our multivariate latent variable models.

3. Results

3.1 General findings

A total of 90 bird species was detected during point count surveys, of which 66, or 73%, displayed evidence of breeding activity (Appendix B). Additionally, two species – the hooded robin (*Melanodryas cucullata*) and brown goshawk (*Accipiter fasciatus*) – were recorded in breeding activity surveys but not detected in point counts. The most commonly detected species was the superb fairywren, which accounted for 26% of all breeding activity recorded in the study. Other frequently-detected species were the willie wagtail, yellow-rumped thornbill, grey shrikethrush (*Colluricincla harmonica*), and rufous whistler (*Pachycephala rufiventris*). The species of conservation concern we detected during surveys included the yellow-rumped thornbill, weebill (*Smicrornis brevirostris*), speckled warbler (*Pyrrholaemus sagittatus*), dusky woodswallow (*Artamus cyanopterus*), crested shrike-tit (*Falcunculus frontatus*), and hooded robin. For the woodland assemblage, breeding activity scores recorded during surveys ranged from 11.5 to 104.5, with a mean of 46.0 (n=105, SE=2.2). The mean score for cup-nesters was 19.1 (n=105, SE=1.4), with minimum and maximum scores of 0 and 76.0 respectively. For species of conservation concern, the mean score was 11.5 (n=105, SE=1.4), minimum score 0, and maximum score 55.0. We found no differences in breeding activity in sites that were fenced compared with sites that were exposed to grazing by stock.

3.2 How does woodland bird breeding activity in restoration plantings compare to breeding activity in remnant woodland patches?

For the woodland assemblage, the score for breeding activity did not differ between plantings, remnants, and reference sites (Appendix E). That is, site type did not appear as a variable of significance in any of our top-ranked models. The same was true when comparing only plantings and remnants (excluding reference sites) (Table 3). We found no effect of site type on species of conservation concern, and cup-nesters showed no response to site type. Removing the superb fairywren, willie wagtail and yellow-rumped thornbill from the woodland assemblage did not elicit any response to site type from the remainder of the assemblage.

3.3 How do patch attributes affect breeding activity in plantings and remnant woodland patches?

Modelling patch attributes of remnants and plantings (excluding large reference sites) against breeding activity score for the woodland assemblage revealed a strong negative effect of increasing patch size, which appeared consistently in the top two candidate models (Table 3). That is, there was more breeding activity per hectare in smaller patches (Figure 2). However, the removal of the superb fairywren from the woodland assemblage greatly reduced the negative effect of site size on breeding activity (Table 3). Size appeared as an explanatory variable in candidate models for breeding activity score of assemblages without superb fairywren, willie wagtail, and yellow-rumped thornbill, but its inclusion did not substantially improve the fit of the simplest model (containing only Julian date). Where size appeared as an explanatory variable, its effect was marginal, with a large standard error.

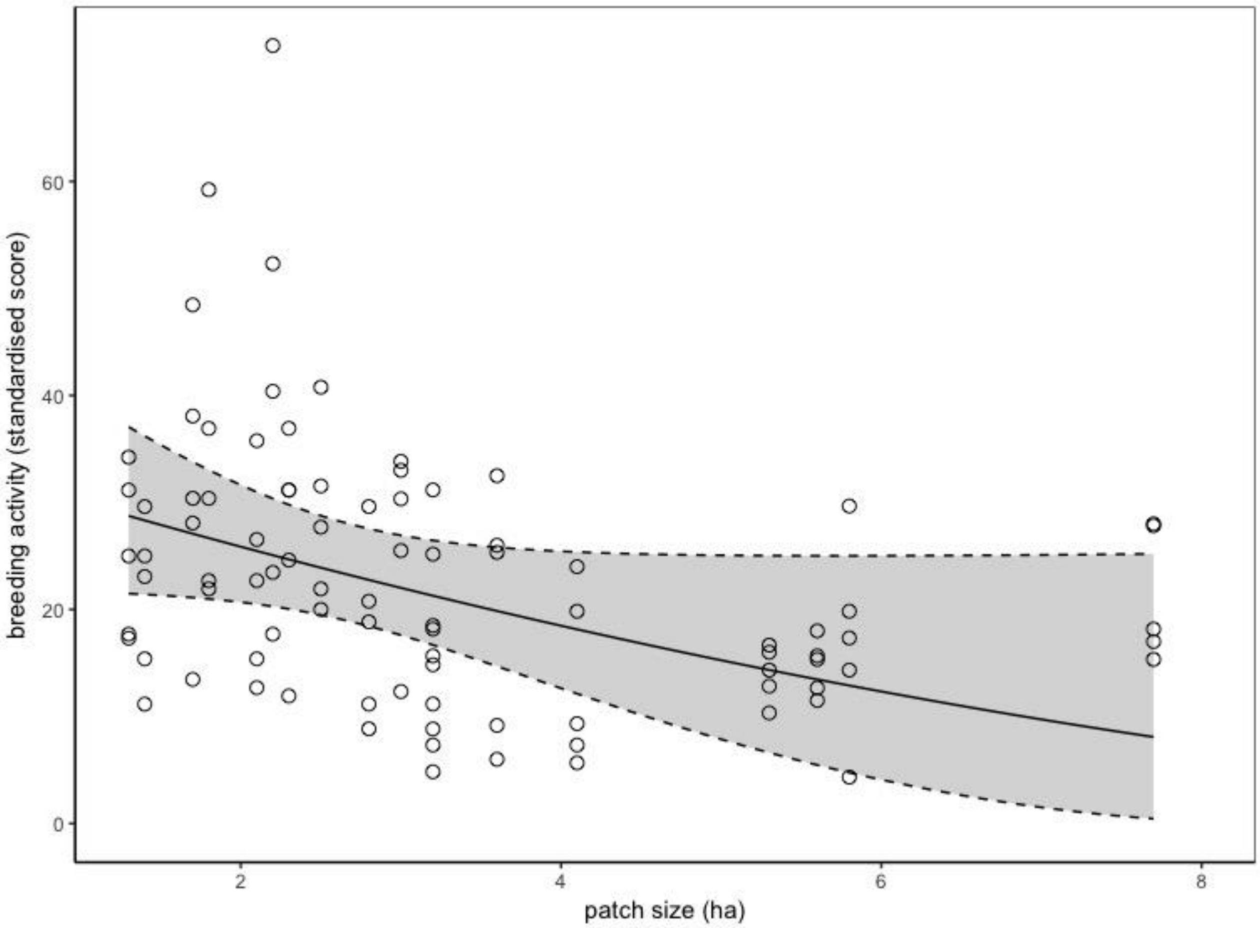

Figure 2 Effect plot illustrating the influence of patch size on breeding activity score of the woodland assemblage in restoration plantings and similarly-sized woodland remnants. Shading indicates 95% confidence intervals.

Excluding large reference sites revealed a marginal negative effect of site type, suggesting that breeding activity score was higher in plantings than in similarly-sized woodland remnants (Table 3). However, the inclusion of site type did not substantially improve the fit of the simplest model, and it failed to appear in top-ranked models after the removal of the three dominant species from the assemblage. Shape appeared in one top-ranked model when the superb fairywren was excluded from the assemblage, however, the standard error was larger than the effect size itself. Site shape therefore had no interpretable effect on breeding activity score.

For species of conservation concern, the best fitting model was the model containing only Julian date (Table 3). Consequently, there were no interpretable effects of planting size, shape, or type on breeding activity score for this subset. The same result was observed for cup-nesters, again indicating marginal or no effects of patch attributes on breeding activity score. Dome-nesters mirror the negative response to patch size demonstrated by the woodland assemblage (per 0.81 correlation).

Table 3 Parameter estimates for total breeding score recorded during breeding activity surveys, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC$_c$). Top-ranked models (ΔAIC$_c$ ≤2) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC$_c$) by ≤2 are shown.

Woodland assemblage	Rank 1 (*w* = 0.22)	Rank 2 (*w* = 0.12)				
	Estimate (SE)	Estimate (SE)				
Intercept	4.57 (0.22)	4.69 (0.24)				
DATE	0.59 (0.09)	0.58 (0.09)				
SIZE	– 0.40 (0.16)	– 0.39 (0.16)				
TYPE (remnant)		– 0.37 (0.33)				
Excluding superb fairywren	Rank 1 (*w* = 0.19)	Rank 2 (*w* = 0.11)	Rank 3 (*w* = 0.08)			
	Estimate (SE)	Estimate (SE)	Estimate (SE)			
Intercept	3.83 (0.12)	3.83 (0.12)	3.69 (0.12)			
DATE	0.38 (0.10)	0.38 (0.10)	0.37 (0.10)			
SIZE		– 0.13 (0.12)	– 0.53 (0.21)			
SHAPE			– 0.06 (0.11)			
SIZE:SHAPE			– 0.67 (0.32)			
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	Rank 1 (*w* = 0.20)	Rank 2 (*w* = 0.09)	Rank 3 (*w* = 0.09)			
	Estimate (SE)	Estimate (SE)	Estimate (SE)			
Intercept	3.19 (0.15)	3.19 (0.14)	3.38 (0.27)			
DATE	0.24 (0.10)	0.24 (0.10)	0.24 (0.10)			
SIZE		– 0.12 (0.14)				
FENCED (yes)			– 0.26 (0.32)			
Species of conservation concern	Rank 1 (*w* = 0.13)	Rank 2 (*w* = 0.11)	Rank 3 (*w* = 0.07)	Rank 4 (*w* = 0.06)	Rank 5 (*w* = 0.06)	Rank 6 (*w* = 0.05)
	Estimate (SE)	Estimate (SE)	Estimate (SE)	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	1.76 (0.23)	1.99 (0.27)	1.38 (0.43)	1.76 (0.23)	1.99 (0.26)	2.00 (0.25)
DATE	0.25 (0.10)	0.24 (0.10)	0.24 (0.10)	0.25 (0.10)	0.23 (0.10)	0.24 (0.10)
TYPE (remnant)		– 0.69 (0.47)			– 0.71 (0.46)	– 0.67 (0.43)
FENCED (yes)			0.52 (0.51)			
SIZE				0.21 (0.23)	0.23 (0.22)	0.37 (0.23)
SIZE:TYPE (remnant)						– 0.77 (0.53)
Cup-nesters	Rank 1 (*w* = 0.23)	Rank 2 (*w* = 0.10)	Rank 3 (*w* = 0.09)			
	Estimate (SE)	Estimate (SE)	Estimate (SE)			
Intercept	2.58 (0.21)	2.47 (0.25)	2.58 (0.20)			

DATE	0.27 (0.10)	0.27 (0.10)	0.27 (0.10)
TYPE (remnant)		0.32 (0.43)	
SHAPE			0.14 (0.21)

Planting age was a significant predictor of breeding activity for the woodland assemblage (Table 4). An increase in planting age was associated with a decrease in breeding activity. This result was no longer evident when the superb fairywren, yellow-rumped thornbill and willie wagtail were removed from the assemblage. However, species of conservation concern also responded negatively to an increase in planting age. No effect of planting age was observed for cup-nesters. For the latter subset, the null model was the top-ranked model.

Table 4 Parameter estimates for total breeding score recorded during breeding activity surveys, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta\text{AIC}_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown. Note that candidate models for cup-nesters are not included, as the null model was the top-ranked model for this subset.

Woodland assemblage	Rank 1 (*w* = 0.27)	Rank 2 (*w* = 0.22)	Rank 3 (*w* = 0.10)
	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	5.65 (0.51)	4.73 (0.25)	4.73 (0.25)
DATE	0.52 (0.11)	0.54 (0.11)	0.54 (0.11)
AGE	– 0.42 (0.15)	– 0.51 (0.18)	– 0.51 (0.18)
SIZE	– 0.37 (0.14)	– 0.41 (0.18)	– 0.45 (0.19)
FENCED	– 0.99 (0.53)		
SHAPE			– 0.17 (0.19)
Excluding superb fairywren	Rank 1 (*w* = 0.20)	Rank 2 (*w* = 0.13)	Rank 3 (*w* = 0.13)
	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	3.85 (0.12)	3.85 (0.14)	3.30 (0.42)
DATE	0.32 (0.12)	0.31 (0.12)	0.32 (0.12)
AGE	– 0.25 (0.13)		– 0.30 (0.12)
FENCED			0.60 (0.45)
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	Rank 1 (*w* = 0.22)	Rank 2 (*w* = 0.12)	
	Estimate (SE)	Estimate (SE)	
Intercept	3.16 (0.19)	3.16 (0.19)	
DATE	0.21 (0.11)		
Species of conservation concern	Rank 1 (*w* = 0.20)	Rank 2 (*w* = 0.12)	
	Estimate (SE)	Estimate (SE)	

Intercept	1.99 (0.21)	1.99 (0.26)
DATE	0.30 (0.13)	0.29 (0.13)
AGE	– 0.53 (0.21)	– 0.52 (0.27)
SIZE	0.43 (0.21)	

3.4 Does breeding activity in restoration plantings and remnant woodland patches reflect species assemblage composition?

Based on ordination modelling, we found that breeding activity was not strongly correlated with relative abundance for bird species in our study sites (Figure 3). Examining the effects of patch attributes on relative abundance and breeding activity revealed that many species differed in their responses according to the two metrics. For example, the abundance of the willie wagtail in remnants and plantings was similar, but breeding activity for this species was higher in remnants (Figure 3a). A similar pattern was observed for the buff-rumped thornbill (*Acanthiza reguloides*), for which more breeding activity was recorded in large reference sites than in plantings, despite the species occurring in similar abundances in the two site types (Figure 3b). The buff-rumped thornbill also displayed a positive response to increasing patch size according to relative abundance, but a negative response according to breeding activity (Figure 3c). Interestingly, there were no species whose abundance increased with patch linearity, but several species, including the black-faced cuckooshrike (*Coracina novaehollandiae*), brown treecreeper (*Climacteris picumnus*), and willie wagtail, showed more evidence of breeding activity in linear sites (Figure 3d). We note that confidence intervals around the estimates for many species were large (Appendix H).

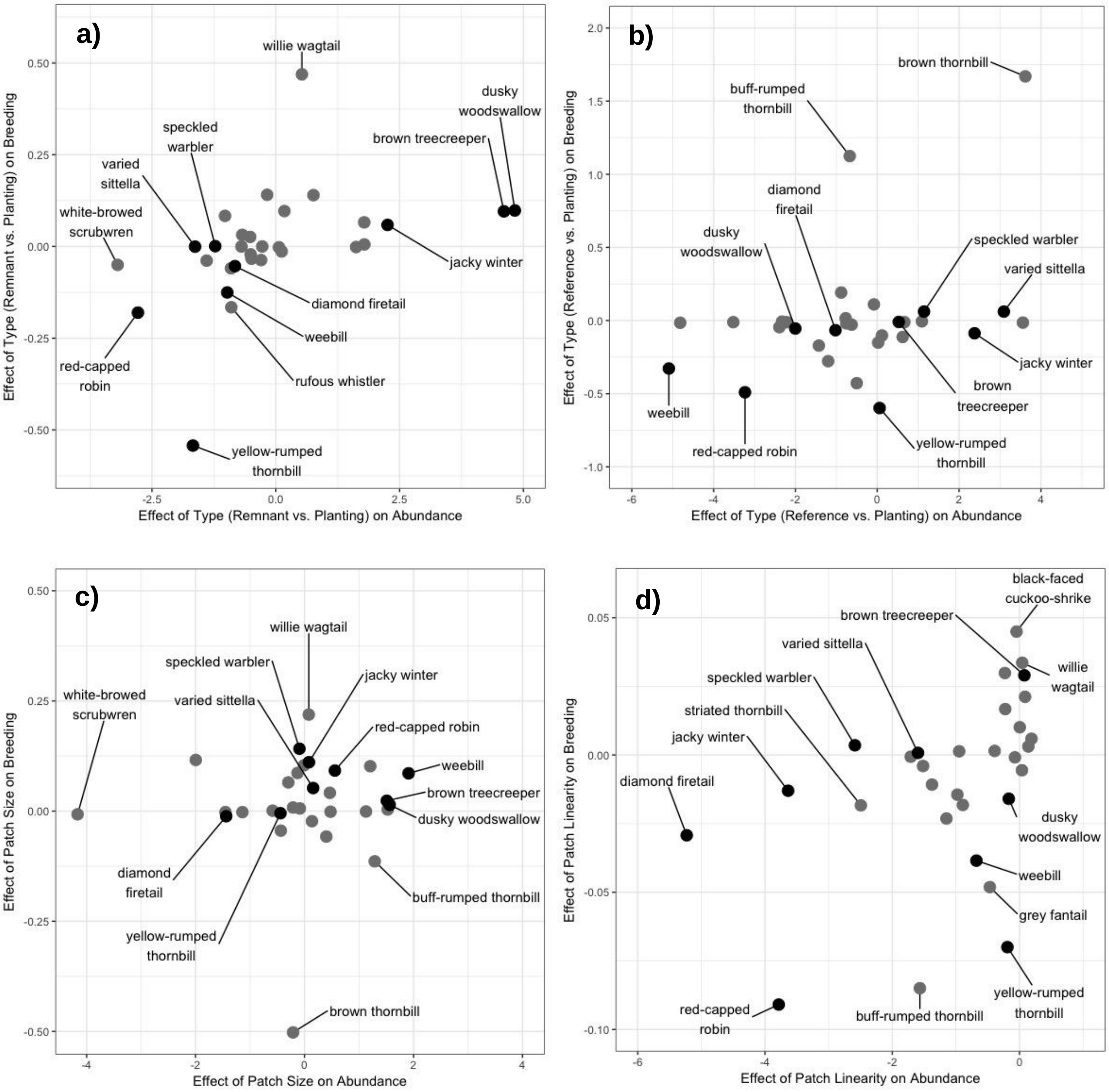

Figure 3 Bird species' relative abundance and breeding activity plotted according to the effects of a) patch type: remnants vs. plantings, in which a positive effect is associated with remnants, b) patch type: reference sites vs. plantings, in which a positive effect is associated with reference sites, c) patch size, and d) patch shape, in which the effect becomes more negative with increasing patch linearity. Effect sizes are taken from multivariate latent variable models. ● = species of conservation concern, ● = species of least concern.

4. Discussion

We recorded breeding activity of a variety of bird species in both restoration plantings and remnant woodland patches. Our analyses of the effects of patch attributes revealed several unexpected findings – most notably, a negative effect of patch size driven by one dominant species (the superb fairywren), in which there was more breeding activity per hectare in smaller patches. Inferences from our study contrast with those of numerous studies on bird species richness and abundance in fragmented agricultural landscapes, which report a positive effect of patch size. We further discuss our key findings in the remainder of this paper and conclude with some insights for bird conservation.

4.1 Patch size

Contrary to our predictions at the outset of this study based on patch size theory (Rosenzweig, 1995), we found that breeding activity score per hectare decreased as patch size increased in plantings and remnant woodland patches. This result was driven by the most commonly-detected species in the study region, and when this species was removed, there was no effect of patch size on breeding activity. Both of these findings contrast with the majority of previous studies, which have documented higher breeding success and reproductive output in larger habitat patches than in smaller patches (e.g. Burke and Nol, 2000; Zanette, 2001; Zanette et al., 2000; Zanette and Jenkins, 2000). The value of small habitat patches for biodiversity in fragmented landscapes has been highlighted via studies of bird species distribution and abundance (Fischer and Lindenmayer, 2002; Gibbons and Boak, 2002; Le Roux et al., 2015; Manning et al., 2006), and was underscored by Wintle et al. (2019) in their global synthesis of conservation studies. Our results indicate that small patches may play a substantial role in supporting bird populations, which we discuss further in the concluding sections of this paper.

According to island biogeography theory, which has been applied to fragmented agricultural landscapes, smaller patches may stimulate a concentration effect of animal populations in fragmented landscapes (MacArthur and Wilson, 1967). For example, waterbirds have been recorded breeding in greater abundances on small versus large islands (Erwin et al., 1995). This may be attributed to the relationship between resource distribution in the patch and surrounding matrix (Estades, 2001). Animals may retreat from the poor quality matrix into habitat patches (concentration effect), and then be reluctant to travel into the surrounding matrix (a so-called "fence effect”). However, Connor et al. (2007) found that animal population densities tend to be positively correlated with area, suggesting that density compensation may not be a common phenomenon.

Smaller patches, including plantings, have been found to contain bird communities with lower overall species richness and a greater proportion of generalist or edge-specialist species (Flaspohler et al., 2010; Mac Nally et al., 2010). Species that are tolerant to fragmentation may take advantage of nesting habitat provided by small patches while utilising resources in the surrounding matrix (Andrén, 1994; Driscoll et al., 2013; Estades, 2001). The superb fairywren accounted for over one quarter of all observations of breeding activity in our study, and is often described as a habitat generalist (Loyn et al., 2007; Mac Nally et al., 2010). Other commonly detected species, including the willie wagtail, demonstrated a positive relationship between breeding activity and patch size, indicating that not all species in the woodland assemblage respond similarly to patch size. Furthermore, we found no effect of patch size on the collective group of species of conservation concern.

Nest predation may have a significant influence on breeding success in birds, and can vary with predator type, patch size, and isolation in fragmented landscapes (Okada et al., 2017; Stephens et al., 2004). There is conflicting evidence pertaining to the influence of patch size on nest predation in fragmented agricultural landscapes. For example, Hoover et al. (1995) attributed lower nesting success of wood thrushes (*Hylocichla mustelina*) in smaller fragments to a greater abundance of avian predators, and Major et al. (2001) found that the grey butcherbird (*Cracticus torquatus*), a predatory species in Australian woodlands, was more abundant in smaller than in larger habitat patches. In contrast, Zanette et al. (2000) found no evidence that area-sensitivity in the eastern yellow robin (*Eopsaltria australis*) could be explained by nest predation. Lehnen and Rodewald (2009) also found no evidence of area-sensitivity in survival and recruitment of shrubland bird species of conservation concern in the eastern United States. Nest type is also confounded with predation risk. Cup-nests are inherently more vulnerable to predation than dome-nests (Okada et al., 2017), and thus species that build cup-nests may be more sensitive to edge-effects in smaller patches. However, in our study, we found no evidence of a patch-size effect on cup-nesters. Conversely, smaller patches may contain lower abundances of brood parasites such as Horsfield's bronzecuckoo (*Chrysococcyx basalis*) (Brooker and Brooker, 2003), reducing the risk of brood parasitism. Indeed, cuckoos were detected infrequently in our study sites (Appendix B). Further research is warranted to directly investigate nesting success of woodland birds in fragmented agricultural landscapes.

A potential explanation for recording greater incidences of breeding activity in smaller patches than in larger patches is that an observer may search smaller sites more thoroughly than larger ones (Woolhouse, 1983). However, we used a search method standardised by area and time in an attempt to control for potential effects of survey effort on activity detection rates. With an equivalent time spent per unit area in each survey regardless of patch size, bias towards detecting more breeding activity in smaller sites should not have influenced our results. However, we note that breeding activity surveys are inherently biased towards species that nest in lower strata (such as the superb fairywren).

4.2 Patch type

We predicted that remnant woodland patches would be characterised by more cases of successful breeding than restoration plantings. However, our results showed that there was as much breeding activity in restoration plantings as in remnant woodland patches. This result is somewhat unexpected, as previous studies have found significant differences in bird species diversity and abundance in plantings and remnants; remnants, and large remnants in particular, tend to support a more diverse species assemblage than plantings (Arnold, 2003; Cunningham et al., 2008; Lindenmayer et al., 2012; Loyn et al., 2007; Martin et al., 2011; Munro et al., 2011). Previous studies of bird assemblages in fragmented agricultural landscapes have identified bird species that are "planting specialists", which preferentially occupy restoration plantings over remnant woodland patches or other sites (reviewed by Belder et al., 2018). It was possible from the outset that breeding activity in restoration plantings would be primarily accounted for by a select few of these species, such as the generalist and edge-tolerant superb fairywren and willie wagtail. However, our modelling indicated that the same trend may hold for species of conservation concern as well as the woodland assemblage as a whole. This suggests that restoration plantings are providing habitat that is as valuable for bird populations as remnant woodland patches. We note, however, that various woodland-dependent species, including species of conservation concern such as the dusky woodswallow and brown treecreeper, show a strong affinity for remnant woodland. We therefore posit that restoration plantings play a complementary role in providing habitat for woodland birds, and caution against restoration plantings being considered a direct replacement for remnant woodland (see also Cunningham et al., 2007).

4.3 Patch shape

At this outset of this study, we predicted linear-shaped sites would support less breeding activity than block-shaped sites. We found a weak negative association between patch linearity and bird breeding activity in our study sites in only one candidate model, and therefore no strong evidence that site shape influenced bird breeding activity in our study region. Previous studies have suggested that increasing linearity negatively influences breeding birds (Helzer and Jelinski, 1999; King et al., 2009). However, Selwood et al. (2009) found more evidence of successful breeding by woodland birds in linear patches. Our ordination modelling revealed that some bird species (e.g. brown treecreeper, black-faced cuckooshrike) showed more breeding activity in sites of increasing linearity, even though this was not reflected in relative abundance. We suggest that further studies are needed to confirm whether patch linearity influences breeding success of birds in fragmented agricultural landscapes.

4.4 Planting age

Contrary to expectations, we found that planting age was a negative predictor of bird breeding activity for the woodland assemblage and for species of conservation concern. That is, there was less evidence of breeding activity in older plantings. This finding contrasts with that of Selwood et al. (2009), who found that the age of plantings did not influence breeding activity. Barrett et al. (2003) found evidence of bird breeding activity in plantings as young as three years, noting that the species that exhibited the most breeding activity were small, shrub-swelling species such as the superb fairywren, red-browed finch, and yellow-rumped thornbill.

A typical planting in our study region consists of a *Eucalyptus* overstorey and *Acacia* understorey. In the absence of fire, an *Acacia* understorey is likely to senesce after 20-50 years (Broadhurst et al., 2008; Parsons and Gosper, 2011), and natural regeneration of the shrub layer in planted sites may be poor (Vesk et al., 2008). The deterioration of understorey density and diversity with planting age is likely to contribute to a reduction in suitable nesting sites for common shrub-nesting species like the superb fairywren, as well as species of conservation concern such as the yellow-rumped thornbill and diamond firetail (*Stagonopleura guttata*). This may explain why the older plantings in our study, which were around 25 years of age, did not support as much breeding activity as younger plantings. The lack of an effect of age on the assemblage when our three most dominant species were removed, as well as the absence of effects for cup-nesters, may be related to the small sample size of these subsets.

4.5 Other findings

We found that examining breeding activity in our study sites provided a markedly different picture of bird species' responses to patch attributes than examining relative abundances obtained via point counts. There were several species whose responses to patch size, shape, and type based on relative abundance were opposite to their responses to these variables based on breeding activity. This indicates that 1) some bird species choose to breed disproportionately more in particular kinds of patches, or 2) the resources birds need to breed are not necessarily provided in patches that they choose to forage in (Loyn et al., 2007). The latter is of particular interest, and important for assessing the value of restoration plantings for woodland bird conservation; if birds preferentially occupy habitat patches but are unable to breed successfully in them, then those patches may become ecological traps, exacerbating population declines (Battin, 2004). This highlights the importance of conducting research that moves beyond pattern-based data collection to include more detailed, population-oriented studies (Belder et al., 2018; Ruiz-Jaen and Aide, 2005).

We found that for species of conservation concern, there were no interpretable effects of site type, size, shape, or other variables on breeding activity score. The lack of an effect of site size is surprising, as previous studies have found that site occupancy by species of conservation concern is positively associated with patch size (Ford et al., 2009; Lindenmayer et al., 2010; Montague-Drake et al., 2009).

The absence of any effect of site type was also unexpected, as we had predicted more breeding activity by species of conservation concern in remnants due to the considerable body of evidence indicating that many threatened and declining species are dependent on or closely associated with remnant woodland (Cunningham et al., 2008; Kinross, 2004; Martin et al., 2011). We note that some species of conservation concern, such as the yellow-rumped thornbill, are among “planting specialists” identified in previous studies (Belder et al., 2018) (Appendix B). It is possible that the small number of observations of species of conservation concern in our study reduced our power to detect effects of patch attributes on these species, if they do indeed exist.

4.6 Inferential limitations

Variables at the landscape level, such as the amount and proximity of native vegetation, may have a stronger influence on species richness and abundance (Cunningham et al., 2008; Fahrig, 2013; Lindenmayer et al., 2010; Radford and Bennett, 2007) and breeding activity (Hinsley et al., 2008, 1995) than the patch-level characteristics of area and shape. Investigating these variables was outside the scope of this study, but we recommend further research be undertaken to address their effects. We note the prevalence of a select few species in our data, which may be symptomatic of an environment that favours generalist and edge-tolerant species, to the detriment of richness and productivity of woodland bird assemblages in our study region. Additionally, the absence of the noisy miner in our study sites enabled us to examine the effects of patch attributes without the confounding effects of competitive exclusion, but noisy miners are regular occupants of small patches in fragmented agricultural landscapes (Major et al. 2001). We also note the small size (<10 ha) of plantings and remnants in our study. These reflect the typical size of native vegetation patches in our study region, but we caution against applying our findings to much larger-scale restoration projects, as breeding birds may respond differently to them than they do to small, isolated patches. Lastly, we note the relatively short duration of our study – two breeding seasons in years of above-average rainfall. We suggest that a better understanding of woodland bird population processes could be obtained by incorporating breeding studies into long-term monitoring projects.

4.7 Management implications and concluding remarks

Studies of bird distribution and abundance in fragmented landscapes have previously highlighted the conservation value of small habitat patches (Fischer and Lindenmayer, 2002; Flaspohler et al., 2010; Gibbons and Boak, 2002; Wintle et al., 2019). Our results add credence to these findings by providing evidence that birds not only occupy small patches, but display evidence of successful breeding within them. Previous studies of bird species richness and abundance in restoration plantings have recommended that plantings be as large as possible to maximise their conservation value (Freudenberger, 2001; Watson et al., 2001; Westphal et al., 2007). We do not seek to undermine the conservation value of very large-scale restoration projects, which were outside the scope of this study, and we fully support the planting of large areas of native vegetation as a strategy to increase vegetation cover in fragmented agricultural landscapes. However, our results suggest that the establishment and conservation of small plantings (and the conservation of small remnants) can also be of considerable value for the management of woodland bird populations (see also Schippers et al., 2009). It is often easier and more cost-effective to implement and maintain small patches (Kendal et al., 2017), so we are hopeful that our findings will encourage land managers to consider implementing small plantings wherever it is not possible to establish large plantings.

The observation of similar levels of breeding activity among the different site types in our study can be cautiously interpreted as encouraging for the conservation value of restoration plantings, as it indicates that birds in fragmented agricultural landscapes may view restoration plantings and remnant woodland patches as equally suitable breeding habitat. However, we acknowledge that breeding activity is only a proxy for breeding success, and cannot provide a true indication of whether breeding attempts are succeeding or failing. We therefore recommend further exploration using an approach such as monitoring nest success or daily nest survival.

Our finding that breeding activity decreased with planting age is of considerable interest for the management of restoration plantings. If a reduction in the condition and density of the shrub layer decreases the ability of a planting to support breeding birds, including species of conservation concern, then there is a case for active management of the shrub layer (including replanting if necessary) as a planting matures. Maintaining a complex habitat structure in restoration plantings also decreases the likelihood of colonisation by the noisy miner (Kinross and Nicol, 2004; Maron et al., 2013). Although we did not find evidence that fenced sites supported more breeding activity, previous studies have shown that the ecological benefits of restoration plantings are diminished when they are exposed to grazing by stock (Lindenmayer et al., 2018b; Selwood et al., 2009). We suggest that maintaining fences around restoration plantings may assist with preserving the shrub layer and ensuring that plantings continue to support breeding birds as they mature.

Finally, the unexpected nature of several of our key findings exemplifies the value of moving beyond pattern data (such as site occupancy information) towards a more behaviour- and population-oriented approach in monitoring and assessing the conservation value of restoration plantings and other habitat patches. As we have shown, relying solely on measures like species richness and abundance risks perpetuating critical knowledge gaps regarding habitat-use and the value of habitat patches for birds in fragmented agricultural landscapes.

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Appendices

Appendix A Attributes of study sites in the South-west Slopes bioregion.

site	site type	shape	planting year	elevation (m)	size (ha)	perimeter (m)	width (m)	fenced
WB-6	planting	block	2001	330	1.5	678	70	
SU-5	planting	block	2003	330	1.8	713	65	✓
BL-4	planting	block	1990	270	2.5	1583	70	✓
HI-3	planting	block	1990	285	5.3	983	150	✓
MT-1	planting	block	1991	280	5.6	988	200	✓
RI-5	planting	block	2002	249	7.7	1176	200	✓
PM-4	planting	linear	1997	269	1.3	1299	40	✓
PS-3	planting	linear	1989	429	1.4	1054	40	✓
MH-6	planting	linear	2000	344	1.7	1138	30	✓
FR-A	planting	linear	1997	259	3.0	2610	15	✓
FR-3	planting	linear	1997	300	3.2	1655	40	✓
MT-3	planting	linear	1993	276	3.2	2213	30	✓
SO-1	remnant	block		303	2.1	1156	80	
WS-3	remnant	block		325	2.8	1006	130	
PK-2	remnant	block		265	5.8	1755	200	✓
WB-2	remnant	linear		262	2.3	1448	30	
SU-1	remnant	linear		297	4.1	736	60	
PK-1	remnant	linear		248	3.6	2379	25	✓
GD-4	reference	block		397	47.1	3956	555	✓
KY	reference	block		347	110	5070	400	✓
MG	reference	block		298	86	5090	400	✓

Appendix B Assemblages and attributes of bird species recorded during the study. Breeding activity scores are provided for each species and site type. The number of patches in which the species was detected and in which breeding occurred are provided in brackets: (no. patches breeding/no. patches present). • denotes species recorded in point count surveys but not breeding activity surveys. Species are listed in taxonomic order (Gill and Donsker, 2018). Conservation status according to NSW threatened species listing (NSW Environment & Heritage, 2018) and bird atlas trends (Barrett et al. 2003). Categories are least concern (LC), conservation concern (CC), vulnerable (V). Information on breeding season and nest type taken from Morcombe (2003) and Pizzey and Knight (1997).

Species		**Abbreviation**	**Nest type**	**Breeding season**	**Conservation status**	**Plantings**	**Remnants**	**Reference sites**
stubble quail	*Coturnix pectoralis*	SQ	cup	Aug-Mar	LC	• (0/2)		• (0/1)
wedge-tailed eagle	*Aquila audax*	WTE	cup	Jun-Nov	LC			• (0/1)
brown goshawk	*Accipiter fasciatus*	BGOS	cup	Sep-Dec	LC	3.0 (1/0)	1.0 (1/0)	
nankeen kestrel	*Falco cenchroides*	NK	hollow	Aug-Dec	LC	1.0 (1/0)	• (0/1)	
brown falcon	*Falco berigora*	BRFA	cup	Aug-Nov	LC	• (0/2)	7.0 (1/1)	
painted buttonquail	*Turnix varius*	PBQ	cup	Aug-Feb	LC			• (0/1)
common bronzewing	*Phaps chalcoptera*	CBZ	cup	Aug-Dec	LC	6.0 (1/3)	6.0 (1/2)	
crested pigeon	*Ocyphaps lophotes*	CP	cup	Jul-Dec	LC	48.5 (3/3)	6.0 (2/5)	
peaceful dove	*Geopelia placida*	PD	cup	Oct-Jan	LC	2.0 (1/4)	12.0 (2/4)	
gang-gang cockatoo	*Callocephalon fimbriatum*	GGC	hollow	Oct-Jan	V			7.5 (1/1)
galah	*Eolophus roseicapilla*	GAL	hollow	Jul-Dec	LC	• (0/11)	• (0/6)	• (0/3)
little corella	*Cacatua sanguinea*	LCOR	hollow	Aug-Nov	LC	• (0/4)	• (0/3)	• (0/1)
sulphur-crested cockatoo	*Cacatua galerita*	SCC	hollow	Aug-Jan	LC	• (0/6)	• (0/4)	• (0/3)
crimson rosella	*Platycercus elegans*	CRO	hollow	Sep-Jan	LC	8.5 (2/7)		• (0/2)
eastern rosella	*Platycercus eximius*	ERO	hollow	Aug-Dec	LC	1.0 (1/12)	12.5 (2/6)	• (0/2)
red-rumped parrot	*Pseophotus haematonotus*	RRP	hollow	Aug-Jan	LC	4.0 (2/8)	• (0/5)	• (0/1)
Australian king-parrot	*Alisterus scapularis*	AKP	hollow	Sep-Jan	LC	• (0/1)	1.0 (1/1)	• (0/1)
superb parrot	*Polytelis swainsonii*	SUPA	hollow	Sep-Dec	V	• (0/2)	• (0/2)	
Horsfield's bronzecuckoo	*Chrysococcyx basalis*	HBC	parasitic	Aug-Jan	LC		• (0/1)	• (0/2)
shining bronzecuckoo	*Chrysococcyx lucidus*	SBC	parasitic	Aug-Jan	LC	• (0/1)		• (0/1)
pallid cuckoo	*Cacomantis pallidus*	PAC	parasitic	Aug-Dec	LC		• (0/1)	
fan-tailed cuckoo	*Cacomantis flabelliformis*	FTC	parasitic	Jul-Jan	LC	• (0/1)	• (0/1)	• (0/1)
laughing kookaburra	*Dacelo novaeguineae*	LK	hollow	Sep-Dec	LC	• (0/11)	11.0 (2/4)	5.0 (1/1)
sacred kingfisher	*Todiramphus sanctus*	SK	hollow	Sep-Jan	LC	• (0/2)	2.0 (2/3)	8.0 (1/2)
rainbow bee-eater	*Merops ornatus*	RBE	hollow	Oct-Jan	CC	• (0/3)	• (0/2)	• (0/1)
white-throated treecreeper	*Cormobates leucophaea*	WTTC	hollow	Aug-Jan	LC	• (0/1)	• (0/1)	1.0 (1/3)
brown treecreeper	*Climacteris picumnus*	BTC	hollow	May-Dec	V		29.0 (2/4)	26.0 (1/3)
superb fairywren[P]	*Malurus cyaneus*	SFW	dome	Sep-Dec	LC	831.5 (12/12)	262.5 (6/12)	146.5 (3/12)
little friarbird	*Philemon citreogularis*	LFB	cup	Jul-Nov	LC	10.0 (1/2)		

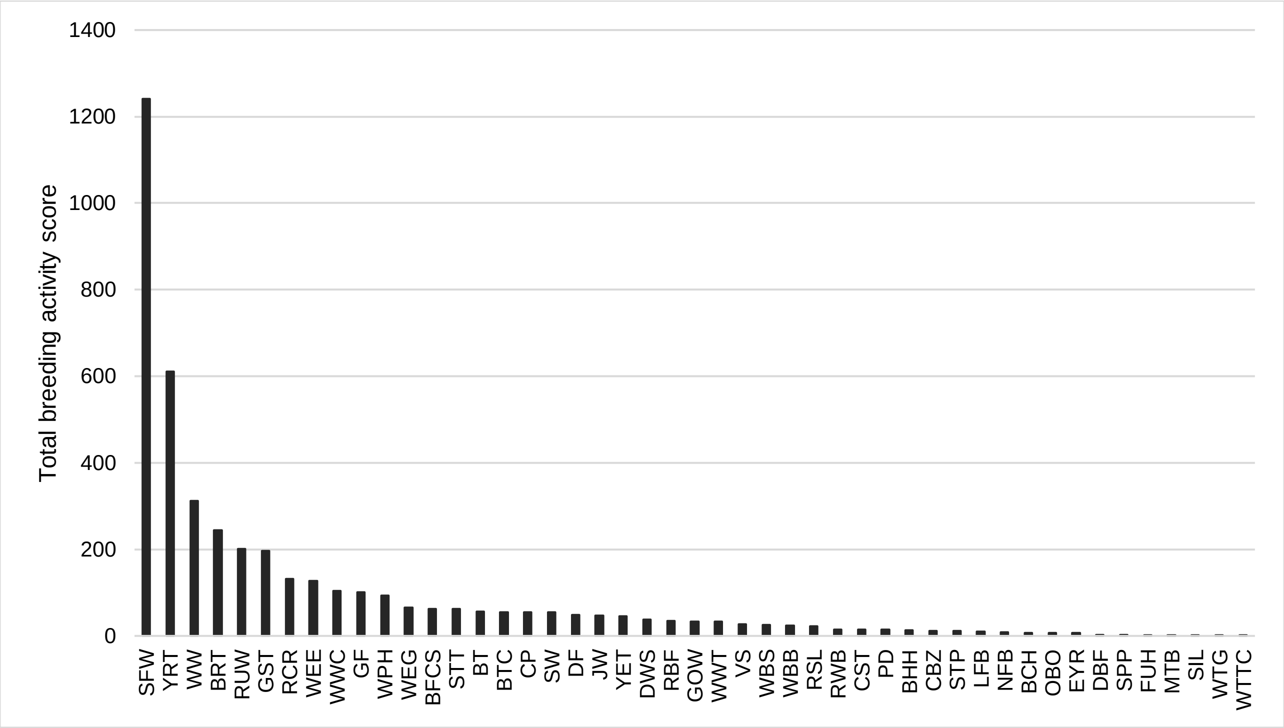
noisy friarbird	*Philemon corniculatus*	NFB	cup	Jul-Jan	LC	• (0/6)	• (0/1)	9.0 (1/3)
blue-faced honeyeater	*Entomyzon cyanotis*	BFH	cup	Jul-Jan	LC	• (0/2)		
black-chinned honeyeater	*Melithreptus gularis*	BCH	cup	Jul-Dec	V		7.5 (1/1)	• (0/2)
brown-headed honeyeater	*Melithreptus brevirostris*	BHH	cup	Aug-Jan	LC	• (0/5)	7.5 (1/2)	5.0 (1/3)
red wattlebird$_{P}$	*Anthochaera carunculata*	RWB	cup	Jul-Dec	LC	15.0 (2/9)	• (0/3)	• (0/3)
yellow-faced honeyeater	*Caligavis chrysops*	YFH	cup	Jul-Jan	LC	• (0/3)		• (0/1)
noisy miner	*Manorina melanocephala*	NM	cup	Jul-Dec	LC	4.0 (2/6)	5.0 (2/3)	
fuscous honeyeater	*Ptilotula fusca*	FUH	cup	Aug-Dec	LC			1.0 (0/1)
white-plumed honeyeater$_{P}$	*Ptilotula penicillata*	WPH	cup	Aug-Dec	LC	21.0 (3/11)	76.0 (3/6)	• (0/3)
spotted pardalote	*Pardalotus punctatus*	SPP	hollow	Sep-Dec	LC	2.0 (1/3)		• (0/1)
striated pardalote	*Pardalotus striatus*	STP	hollow	Jun-Jan	LC	11.0 (4/12)	1.0 (1/6)	• (0/3)
speckled warbler$_{P}$	*Pyrrholaemus sagittatus*	SW	dome	Aug-Jan	CC	1.0 (1/1)	2.0 (1/2)	51.5 (3/2)
white-browed scrubwren$_{P}$	*Sericornis frontalis*	WBS	dome	Jul-Dec	LC	25.0 (1/1)		
weebill$_{P}$	*Smicrornis brevirostris*	WEE	dome	Aug-Feb	CC	97.0 (7/9)	7.0 (2/2)	23.0 (2/2)
western gerygone$_{P}$	*Gerygone fusca*	WEG	dome	Aug-Nov	LC	26.5 (3/10)	2.0 (1/4)	36.5 (3/3)
white-throated gerygone	*Gerygone olivacea*	WTG	dome	Sep-Nov	LC	• (0/3)		1.0 (1/2)
brown thornbill	*Acanthiza pusilla*	BT	dome	Aug-Dec	LC	2.0 (2/1)	• (0/1)	54.0 (1/3)
buff-rumped thornbill	*Acanthiza reguloides*	BRT	dome	Aug-Dec	LC	32.5 (2/4)	55.5 (2/2)	155.5 (3/3)
yellow-rumped thornbill$_{P}$	*Acanthiza chrysorrhoa*	YRT	dome	Jul-Dec	CC	513.5 (10/10)	53.0 (2/3)	48.5 (2/3)
yellow thornbill$_{P}$	*Acanthiza nana*	YET	dome	Aug-Dec	LC	31.0 (8/9)	8.0 (2/2)	6.0 (2/3)
striated thornbill$_{P}$	*Acanthiza lineata*	STT	dome	Jul-Dec	LC	17.5 (1/2)	13.0 (1/1)	31.5 (2/3)
white-browed babbler	*Pomatostomus superciliosus*	WBB	dome	Jun-Dec	LC	7.5 (1/2)	16.0 (1/1)	
grey butcherbird	*Cracticus torquatus*	GBB	cup	Aug-Dec	LC	• (0/2)	• (0/2)	
pied butcherbird	*Cracticus nigrogularis*	PBB	cup	Aug-Nov	LC	3.5 (1/10)	7.5 (1/5)	• (0/1)
Australian magpie	*Cracticus tibicen*	AM	cup	Aug-Oct	LC	143.5 (8/12)	74.5 (4/6)	19.5 (2/3)
pied currawong	*Strepera graculina*	PCW	cup	Aug-Dec	LC	• (0/2)	• (0/2)	• (0/1)
dusky woodswallow	*Artamus cyanopterus*	DWS	cup	Aug-Dec	V	1.0 (1/0)	36.0 (2/3)	6.0 (1/1)
black-faced cuckooshrike	*Coracina novaehollandiae*	BFCS	cup	Aug-Jan	LC	28.5 (4/12)	32.0 (4/5)	2.0 (2/3)
white-bellied cuckooshrike	*Coracina papuensis*	WBCS	cup	Aug-Mar	LC			• (0/1)
white-winged triller	*Lalage tricolor*	WWT	cup	Sep-Dec	CC	25.0 (2/1)	7.0 (1/2)	1.0 (1/2)
varied sittella	*Daphoenositta chrysoptera*	VS	cup	Sep-Dec	V			27.0 (3/3)
crested shriketit	*Falcunculus frontatus*	CST	cup	Sep-Jan	CC	5.0 (3/5)	9.0 (1/4)	
golden whistler	*Pachycephala pectoralis*	GOW	cup	Aug-Jan	LC	23.5 (3/2)	2.0 (2/2)	7.5 (1/0)
rufous whistler$_{P}$	*Pachycephala rufiventris*	RUW	cup	Sep-Feb	LC	164.5 (7/12)	16.5 (3/5)	20.0 (3/3)
grey shrikethrush$_{P}$	*Colluricincla harmonica*	GST	cup	Jul-Feb	LC	135.0 (9/12)	52.5 (4/6)	8.5 (2/3)
olive-backed oriole	*Oriolus sagittatus*	OBO	cup	Sep-Jan	LC	7.5 (1/0)	• (0/1)	
willie wagtail$_{P}$	*Rhipidura leucophrys*	WW	cup	Aug-Dec	LC	92.5 (9/12)	176.5 (5/6)	42.5 (1/3)
grey fantail$_{P}$	*Rhipidura albiscapa*	GF	cup	Aug-Dec	LC	60.5 (9/12)	24.0 (3/5)	17.0 (3/3)

magpie-lark	*Grallina cyanoleuca*	AML	cup	Aug-Feb	LC	10.5 (4/12)	43.0 (2/6)	5.0 (1/3)
leaden flycatcher	*Myiagra rubecula*	LFC	cup	Sep-Nov	LC	• (0/1)		• (0/1)
restless flycatcher	*Myiagra inquieta*	RFC	cup	Aug-Jan	CC	• (0/2)	• (0/5)	• (0/1)
little raven	*Corvus mellori*	LR	cup	Aug-Dec	LC	16.5 (2/8)	7.5 (1/4)	9.0 (1/2)
Australian raven	*Corvus coronoides*	AR	cup	Jul-Oct	LC	11.5 (1/12)	3.5 (1/6)	11.5 (2/3)
white-winged chough	*Corcorax melanoramphos*	WWC	cup	Aug-Dec	LC	55.5 (6/6)	40.0 (2/3)	15.5 (2/3)
eastern yellow robin[P]	*Eopsaltria australis*	EYR	cup	Jul-Dec	LC		• (0/1)	7.0 (1/1)
hooded robin[P]	*Melanodryas cucullata*	HR	cup	Jul-Dec	V			4.5 (1)
jacky winter	*Microeca fascinans*	JW	cup	Jul-Dec	CC	• (0/1)	10.0 (2/2)	37.0 (2/2)
flame robin[P]	*Petroica phoenicea*	FR	cup	Aug-Jan	V	• (0/1)	• (0/1)	
red-capped robin[P]	*Petroica goodenovii*	RCR	cup	Jul-Jan	CC	95.5 (2/2)	• (0/1)	36.0 (3/1)
welcome swallow	*Hirundo neoxena*	WS	cup	Aug-Dec	LC	• (0/4)	• (0/6)	• (0/1)
fairy martin	*Petrochelidon ariel*	FM	other	Aug-Jan	CC	• (0/1)		
tree martin	*Petrochelidon nigricans*	TM	hollow	Aug-Dec	LC		8.5 (1/2)	
rufous songlark	*Cincloramphus mathewsi*	RSL	cup	Sep-Dec	LC	6.0 (1/8)	16.0 (3/3)	• (0/1)
brown songlark	*Cincloramphus cruralis*	BSL	cup	Sep-Feb	CC	• (0/3)	• (0/1)	
silvereye	*Zosterops lateralis*	SIL	cup	Sep-Jan	LC	1.0 (1/5)	• (0/2)	• (0/1)
common starling[I]	*Sturnus vulgaris*	STA	hollow	Aug-Jan		24.0 (2/8)	10.0 (1/5)	6.0 (1/1)
common blackbird[I]	*Turdus merula*	BKB	cup	Sep-Dec		42.0 (2/3)	• (0/1)	
mistletoebird	*Dicaeum hirundinaceum*	MTB	dome	Oct-Mar	LC	• (0/1)	• (0/2)	1.0 (1/2)
diamond firetail[P]	*Stagonopleura guttata*	DF	dome	Aug-Jan	V	38.0 (2/2)	10.0 (3/2)	
red-browed finch[P]	*Neochmia temporalis*	RBF	dome	Sep-Dec	LC	29.0 (2/3)	6.0 (1/1)	
double-barred finch	*Taeniopygia bichenovii*	DBF	dome	Jul-Dec	LC	2.0 (2/2)	1.0 (1/1)	
Australian pipit	*Anthus australis*	PIP	cup	Aug-Dec	LC	• (0/3)	• (0/2)	

[P] Planting specialists (Belder et al. 2018)

[I] Introduced species

Appendix C Total breeding activity recorded for the subset of bird species included in multivariate latent model ordinations. Acronyms corresponding to particular bird species are given in Appendix B.



Appendix D Mixed effects models for breeding score modelled against weather and temporal variables, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC$_c$). Top-ranked models (ΔAIC$_c$ ≤2) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the assemblage that exclude dominant species. All models that differed from the top model (ΔAIC$_c$) by ≤2 are shown, as well as the intercept-only model.

Woodland assemblage	df	log(L)	AIC$_c$	ΔAIC$_c$	AIC$_w$
DATE	5	-144.46	299.52	0.00	0.29
DATE + TIME	6	-143.57	300.00	0.49	0.23
DATE + SUN	6	-144.29	301.44	1.92	0.11
Intercept only	4	-160.21	328.82	29.30	0.00

Excluding superb fairywren	df	log(L)	AIC$_c$	ΔAIC$_c$	AIC$_w$
DATE	5	-144.32	299.25	0.00	0.43
Intercept only	4	-150.68	309.75	10.50	0.00

Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	df	log(L)	AIC$_c$	ΔAIC$_c$	AIC$_w$
DATE	5	-145.52	301.64	0.00	0.29
DATE + TIME	6	-145.15	303.17	1.53	0.14
DATE + WIND	6	-145.30	303.46	1.82	0.12
Intercept only	4	-148.41	305.23	3.59	0.05

Species of conservation concern	df	log(L)	AIC$_c$	ΔAIC$_c$	AIC$_w$
DATE + TIME	6	-157.23	327.32	0.00	0.31
DATE	5	-158.89	328.39	1.06	0.18
Intercept only	4	-161.53	331.45	4.13	0.04

Cup-nesters	df	log(L)	AIC$_c$	ΔAIC$_c$	AIC$_w$
DATE + TIME	6	-150.37	313.60	0.00	0.23
DATE + TIME + WIND	7	-149.54	314.23	0.63	0.17
DATE + TIME + SUN	7	-149.66	314.48	0.88	0.15
DATE	5	-151.94	314.49	0.89	0.15
DATE + TIME + SUN + WIND	8	-148.96	315.42	1.82	0.09
DATE + SUN	6	-151.34	315.53	1.93	0.09
Intercept only	4	-155.94	320.29	6.69	0.01

Appendix E Mixed effects models for total breeding score modelled against site type for all sites (planting, remnant, and reference), ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta\mathrm{AIC}_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model ($\Delta\mathrm{AIC}_c$) by ≤2 are shown, as well as the intercept-only model.

Woodland assemblage	df	log(*L*)	AIC_c	$\Delta\mathrm{AIC}_c$	AIC_w
DATE	5	-144.46	299.52	0.00	0.54
DATE + FENCED	6	-144.27	301.40	1.89	0.21
Intercept only	4	-160.21	328.82	29.30	0.00
Excluding superb fairywren	df	log(*L*)	AIC_c	$\Delta\mathrm{AIC}_c$	AIC_w
DATE	5	-144.32	299.25	0.00	0.67
Intercept only	4	-150.68	309.75	10.50	0.00
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	df	log(*L*)	AIC_c	$\Delta\mathrm{AIC}_c$	AIC_w
DATE	5	-145.52	301.64	0.00	0.47
Intercept only	4	-148.41	305.23	3.59	0.08
Species of conservation concern	df	log(*L*)	AIC_c	$\Delta\mathrm{AIC}_c$	AIC_w
DATE	5	-158.89	328.39	0.00	0.34
DATE + FENCED	6	-158.10	329.05	0.67	0.24
DATE + TYPE	7	-157.23	329.62	1.24	0.18
Intercept only	4	-161.53	331.45	3.07	0.07
Cup-nesters	df	log(*L*)	AIC_c	$\Delta\mathrm{AIC}_c$	AIC_w
DATE	5	-151.94	314.49	0.00	0.60
Intercept only	4	-155.94	320.29	5.79	0.03

Appendix F Mixed effects models for total breeding score recorded during breeding activity surveys in plantings and remnants (excluding reference sites), ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta\text{AIC}_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown, as well as the intercept-only model.

Woodland assemblage	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + SIZE	6	-122.29	257.58	0.00	0.22
DATE + SIZE + TYPE	7	-121.68	258.73	1.15	0.12
Intercept only	4	-140.70	289.88	32.29	0.00
Excluding superb fairywren	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-123.81	258.33	0.00	0.19
DATE + SIZE	6	-123.18	259.37	1.04	0.11
DATE + SIZE + SHAPE + SIZE:SHAPE	8	-121.17	260.12	1.79	0.08
Intercept only	4	-130.79	270.05	11.72	0.00
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-126.14	263.00	0.00	0.20
DATE + SIZE	6	-125.82	264.65	1.65	0.09
DATE + FENCED	6	-125.82	264.65	1.65	0.09
Intercept only	4	-129.16	266.79	3.79	0.03
Species of conservation concern	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-140.41	291.53	0.00	0.13
DATE + TYPE	6	-139.40	291.81	0.28	0.11
DATE + FENCED	6	-139.90	292.81	1.28	0.07
DATE + SIZE	6	-139.99	292.99	1.46	0.06
DATE + SIZE + TYPE	7	-138.84	293.05	1.52	0.06
DATE + SIZE + TYPE + SIZE:TYPE	8	-137.85	293.49	1.96	0.05
Intercept only	4	-143.08	294.63	3.10	0.03
Cup-nesters	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-133.19	277.10	0.00	0.23
DATE + TYPE	6	-132.92	278.85	1.75	0.10
DATE + SHAPE	6	-132.96	278.93	1.83	0.09
Intercept only	4	-136.61	281.69	4.59	0.02

Appendix G Mixed effects models for total breeding score recorded during breeding activity surveys in plantings, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta\text{AIC}_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown, as well as the intercept-only model.

Woodland assemblage	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + AGE + SIZE + FENCED	8	-76.62	172.06	0.00	0.27
DATE + AGE + SIZE	7	-78.16	172.47	0.40	0.22
DATE + AGE + SIZE + SHAPE	8	-77.60	174.01	1.95	0.10
Intercept only	4	-92.38	193.49	21.43	0.00

Excluding superb fairywren	Df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + AGE	6	-79.79	173.16	0.00	0.20
DATE	5	-81.45	174.00	0.84	0.13
DATE + AGE + FENCED	7	-78.95	174.06	0.90	0.13
Intercept only	4	-84.72	178.18	5.01	0.02

Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	Df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-82.35	175.81	0.00	0.22
Intercept only	4	-84.12	176.96	1.15	0.12

Species of conservation concern	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + AGE + SIZE	7	-90.30	196.75	0.00	0.20
DATE + AGE	6	-92.08	197.75	1.00	0.12
Intercept only	4	-96.31	201.35	4.59	0.02

Cup-nesters	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
Intercept only	4	-91.13	190.98	0.00	0.17
AGE	5	-90.42	191.94	0.96	0.10
DATE	5	-90.48	192.06	1.08	0.10

Appendix H Coefficients from multivariate latent variable models used to plot the effects of site attributes on relative abundance and breeding activity of bird species. The 95% upper and lower confidence limits around the estimate are provided in brackets. Estimates for which the confidence interval does not overlap zero are shown in bold.

	ABUNDANCE				**BREEDING ACTIVITY**			
common name	remnant vs. planting	reference vs. planting	patch size	patch linearity	remnant vs. planting	reference vs. planting	patch size	patch linearity
black-faced cuckooshrike	0.17 (-0.36, 0.80)	1.11 (-0.30, 2.61)	-0.21 (-0.76, 0.34)	-0.04 (-0.34, 0.21)	0.09 (-0.02, 0.21)	0.00 (-0.39, 0.32)	0.01 (-0.13, 0.14)	**0.05 (0.00, 0.09)**
brown-headed honeyeater	-0.67 (-1.86, 0.46)	-0.87 (-3.62, 1.59)	0.41 (-0.46, 1.42)	**-1.61 (-3.13, -0.59)**	0.00 (-0.04, 0.04)	**0.19 (0.06, 0.29)**	**-0.06 (-0.10, -0.01)**	0.00 (-0.01, 0.01)
buff-rumped thornbill	-0.19 (-1.25, 0.98)	-0.66 (-2.64, 1.36)	**1.28 (0.58, 2.03)**	**-1.49 (-2.86, -0.22)**	0.14 (-0.08, 0.36)	**1.11 (0.36, 1.76)**	-0.11 (-0.37, 0.18)	**-0.08 (-0.18, 0.00)**
brown thornbill	0.08 (-2.13, 1.96)	**3.63 (0.82, 6.17)**	-0.23 (-1.14, 0.71)	0.10 (-1.04, 1.15)	0.00 (-0.13, 0.11)	**1.67 (1.26, 2.13)**	**-0.50 (-0.65, -0.34)**	-0.01 (-0.05, 0.05)
brown treecreeper	**4.57 (2.56, 6.97)**	0.60 (-2.67, 4.10)	**1.50 (0.65, 2.42)**	0.09 (-0.32, 0.51)	**0.10 (0.04, 0.15)**	-0.01 (-0.19, 0.18)	0.02 (-0.04, 0.09)	**0.03 (0.01, 0.05)**
crested pigeon	**1.58 (0.53, 2.72)**	-0.56 (-6.02, 5.19)	-0.94 (-4.46, 1.42)	0.20 (-0.26, 0.68)	0.00 (-0.07, 0.07)	-0.03 (-0.23, 0.20)	0.00 (-0.08, 0.07)	0.01 (-0.02, 0.03)
double-barred finch	-0.29 (-1.94, 1.45)	-2.15 (-7.56, 2.84)	-0.50 (-3.18, 1.57)	**-1.48 (-3.00, -0.35)**	0.00 (-0.01, 0.01)	-0.01 (-0.05, 0.04)	0.00 (-0.01, 0.02)	0.00 (-0.01, 0.00)
diamond firetail	-0.71 (-3.20, 0.78)	-0.89 (-6.45, 3.72)	-1.35 (-4.36, 1.22)	**-5.67 (-7.29, -2.35)**	-0.05 (-0.15, 0.04)	-0.07 (-0.32, 0.23)	-0.01 (-0.11, 0.09)	**-0.03 (-0.06, 0.00)**
dusky woodswallow	**4.79 (2.91, 6.69)**	-2.01 (-6.44, 2.55)	1.59 (-0.11, 3.24)	-0.18 (-0.93, 0.69)	**0.10 (0.04, 0.16)**	-0.05 (-0.23, 0.17)	0.02 (-0.07, 0.08)	-0.02 (-0.04, 0.01)
grey fantail	**-1.01 (-1.56, -0.48)**	-0.05 (-1.29, 1.03)	0.14 (-0.27, 0.58)	**-0.46 (-0.76, -0.18)**	0.08 (-0.01, 0.18)	0.11 (-0.19, 0.41)	-0.02 (-0.13, 0.09)	**-0.05 (-0.09, -0.01)**
golden whistler	-0.47 (-2.93, 1.57)	-1.31 (-6.81, 3.88)	-1.98 (-4.89, 0.68)	-1.19 (-2.73, 0.41)	-0.03 (-0.15, 0.06)	-0.27 (-0.59, 0.04)	**0.12 (0.00, 0.13)**	-0.02 (-0.06, 0.02)
grey shrikethrush	-0.29 (-0.66, 0.13)	0.67 (-0.59, 1.96)	-0.44 (-0.94, 0.06)	-0.07 (-0.26, 0.13)	-0.04 (-0.21, 0.11)	-0.01 (-0.15, 0.19)	-0.04 (-0.23, 0.13)	0.00 (-0.06, 0.06)
jacky winter	**2.22 (0.93, 3.76)**	2.38 (-0.16, 4.64)	0.10 (-0.71, 0.89)	**-3.64 (-5.06, -2.38)**	0.06 (-0.04, 0.14)	-0.08 (-0.38, 0.21)	**0.11 (0.01, 0.23)**	-0.01 (-0.05, 0.02)
peaceful dove	**1.75 (0.92, 2.68)**	-0.68 (-5.18, 3.70)	-1.39 (-4.14, 0.59)	-0.21 (-0.71, 0.33)	**0.07 (0.01, 0.13)**	0.01 (-0.15, 0.19)	0.00 (-0.07, 0.07)	0.02 (-0.01, 0.04)
red-capped robin	**-2.71 (-4.69, -0.39)**	-3.03 (-7.72, 0.50)	0.59 (-1.33, 2.34)	**-3.71 (-5.39, -2.48)**	**-0.18 (-0.34, -0.02)**	-0.49 (-1.05, -0.04)	0.09 (-0.09, 0.30)	**-0.09 (-0.16, -0.03)**
rufous whistler	**-0.89 (-1.30, -0.48)**	0.63 (-0.38, 1.64)	-0.09 (-0.45, 0.29)	**-0.23 (-0.43, -0.02)**	**-0.16 (-0.33, -0.03)**	-0.13 (-0.56, 0.35)	0.01 (-0.17, 0.17)	0.03 (-0.02, 0.10)
red wattlebird	-0.49 (-1.06, 0.13)	**-3.48 (-6.60, -1.03)**	**1.12 (0.31, 2.01)**	0.01 (-0.24, 0.30)	-0.02 (-0.06, 0.02)	-0.01 (-0.15, 0.11)	0.00 (-0.05, 0.05)	0.01 (-0.01, 0.03)
sacred kingfisher	**1.78 (0.84, 2.85)**	-1.34 (-4.55, 1.52)	**1.17 (0.25, 2.17)**	**-0.90 (-1.82, -0.03)**	0.01 (-0.05, 0.06)	**-0.17 (-0.37, 0.00)**	**0.10 (0.03, 0.17)**	0.00 (-0.02, 0.03)
striated pardalote	0.11 (-0.33, 0.62)	-2.27 (-4.85, 0.14)	0.46 (-0.28, 1.48)	0.13 (-0.06, 0.35)	-0.01 (-0.04, 0.01)	-0.01 (-0.09, 0.07)	0.00 (-0.03, 0.03)	0.00 (-0.01, 0.01)
striated thornbill	-0.61 (-2.61, 1.19)	**3.52 (1.05, 6.37)**	-0.12 (-1.24, 0.81)	**-2.06 (-4.95, -0.29)**	0.03 (-0.12, 0.17)	-0.02 (-0.46, 0.37)	0.09 (-0.06, 0.24)	-0.02 (-0.07, 0.03)
speckled warbler	-1.18 (-3.06, 0.41)	1.19 (-1.81, 4.39)	-0.11 (-1.32, 1.13)	**-2.69 (-4.29, -1.07)**	0.00 (-0.13, 0.13)	0.06 (-0.30, 0.48)	0.14 (-0.01, 0.28)	0.00 (-0.05, 0.05)
varied sittella	-1.65 (-4.50, 1.29)	3.10 (-0.10, 6.34)	0.14 (-0.84, 1.23)	-1.54 (-4.17, 0.57)	0.00 (-0.08, 0.08)	0.06 (-0.16, 0.30)	0.05 (-0.03, 0.13)	0.00 (-0.03, 0.03)
white-browed scrubwren	-3.00 (-7.08, 0.28)	-2.34 (-7.36, 3.75)	**-4.09 (-8.19, -0.80)**	-0.77 (-3.26, 1.14)	-0.05 (-0.14, 0.04)	-0.05 (-0.30, 0.27)	-0.01 (-0.11, 0.13)	-0.01 (-0.05, 0.02)

weebill	**-0.97 (-1.57, -0.33)**	**-5.07 (-6.81, -3.25)**	**1.90 (1.24, 2.47)**	**-0.67 (-1.01, -0.38)**	-0.13 (-0.27, 0.01)	-0.32 (-0.80, 0.12)	0.08 (-0.08, 0.26)	-0.04 (-0.09, 0.02)
western gerygone	**-0.91 (-1.62, -0.18)**	0.16 (-1.39, 1.56)	-0.01 (-0.60, 0.52)	**-0.88 (-1.30, -0.50)**	-0.06 (-0.17, 0.05)	-0.11 (-0.45, 0.25)	0.11 (-0.02, 0.24)	-0.02 (-0.06, 0.02)
white-plumed honeyeater	**0.77 (0.34, 1.12)**	**-4.75 (-7.35, -2.30)**	**1.51 (0.77, 2.36)**	0.09 (-0.10, 0.31)	**0.14 (0.03, 0.24)**	-0.01 (-0.34, 0.28)	0.00 (-0.11, 0.13)	0.02 (-0.02, 0.06)
willie wagtail	**0.53 (0.15, 0.91)**	-0.49 (-1.95, 1.13)	0.07 (-0.43, 0.63)	0.04 (-0.15, 0.24)	**0.47 (0.29, 0.66)**	-0.42 (-1.03, 0.17)	**0.22 (0.02, 0.45)**	0.03 (-0.05, 0.02)
white-winged chough	-0.52 (-1.40, 0.55)	-0.73 (-3.26, 1.75)	0.46 (-0.36, 1.36)	**-1.34 (-2.19, -0.62)**	0.03 (-0.07, 0.13)	-0.01 (-0.31, 0.28)	0.04 (-0.06, 0.16)	-0.01 (-0.05, 0.02)
yellow thornbill	**-1.39 (-2.05, -0.74)**	0.03 (-1.55, 1.66)	-0.30 (-0.98, 0.34)	**-0.39 (-0.68, -0.13)**	-0.04 (-0.11, 0.05)	-0.15 (-0.38, 0.09)	0.07 (-0.02, 0.15)	0.00 (-0.03, 0.03)
yellow-rumped thornbill	**-1.66 (-2.29, -1.08)**	0.07 (-1.53, 1.49)	-0.44 (-1.07, 0.19)	-0.19 (-0.42, 0.04)	**-0.54 (-0.84, -0.24)**	-0.59 (-1.59, 0.38)	0.00 (-0.39, 0.34)	-0.07 (-0.19, 0.04)

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